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Including man-nature relationships in environmental sustainability assessment of forest-based production systems

Thesis submitted in fulfillment of the requirements for the degree of Doctor (PhD) in Applied Biological Sciences

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Nederlandse vertaling:
Het includeren van verhoudingen tussen mens en natuur in duurzaamheidsanalyse van op bos gebaseerde productiesystemen

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Cover illustration:
Picture taken of the studied Scots pine stand in Brasschaat by the 'Instituut voor Natuur- en BosOnderzoek (INBO)'.

There is a Dutch saying which can be linked to this picture: 'door de bomen het bos niet meer zin' (literally translated: 'because of the trees not seeing the forest anymore'). It means, by focussing too much on detail one may forget the bigger picture. On the other hand, the reverse Dutch saying is also used: 'Door het bos de bomen niet meer zien' (literally translated, 'because of the forest not seeing the trees anymore'). Consequently implying the opposite, by looking too much at the bigger picture, one may lose sight of details.

For this PhD, a balance needed to be struck between details and the complete picture, thus a balance between both latter sayings. Research had to be performed on the overall best management and services of a forest but this without losing attention for details.

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List of Abbreviations

A	Ascendency
AMI	Average Mutual Information index
ANAFORE	ANAlysis in FORest Ecosystems model
BM	Biomass
CEENE	Cumulative Exergy Extracted from the Natural Environment
CIPAM	Canopy Intereception and PArticulate matter removal Model
CUR	Current Climate scenario
DALY	Disability-Adjusted Life Years
DD	Dry Deposition
ECEC	Ecological Cumulative Exergy Consumption
ENA	Ecological Network Analysis
EUR	Europe scenario
FCI	Finn's Cycling Index
FEA	Flemish Environmental Agency
FU	Functional Unit
FS_x	Fijn Stof met een aerodynamische diameter kleiner dan x µm
GPP	Gross Primary Production
IO(A)	Input-Output (Analysis)
IPCC	Intergovernmental Panel on Climate Change
IRCEL	the Belgian Interregional Environment Agency
LAI	Leaf Area Index
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LIM	Linear Inverse Modelling/Model
MOD	Moderate climate scenario
MSWI	Municipal Solid Waste Incineration
NMVOC	Non-methane Volatile Organic Compounds
NPP	Net Primary Production
PES	Payment for Ecosystem Services
PM_x	Particulate Matter with an aerodynamic diameter smaller than x µm
REF	Reference scenario
RM	Removal
SEV	Severe climate scenario
SL	Specific water storage capacity per leaf area index

SW	Specific water storage capacity per wood area index
TES	Techno-Ecological System
TSTF	Total System Throughflow
TSTP	Total System Throughput
WAI	Wood Area Index

Abstract/summary

After realizing the impact of its human/industrial system on nature and indirectly on itself, mankind became aware of its need for a sustainable relationship with nature. To obtain this sustainable relationship, assessments are required to unravel which managements of human/industrial and natural systems are best suited for that purpose. In our study, we have attempted to assess the environmental aspect of this sustainable relationship in a better manner, this exemplified for our relation with forest ecosystems. Latter ecosystem is of major importance as it covers 30% of the land surface and provides essential services to mankind (FAO, 2010). A challenge we wanted to overcome in order of revealing best practices, is to include the dynamic responses of natural systems, e.g. effect of thinning on forest growth and thus carbon dioxide uptake. Practically, methodological improvements were performed and the improved methods were applied to one specific forest, an intensively managed Scots pine stand in Belgium (Europe).

Firstly, a framework was developed with which the environmental impact and benefit of an integrated human/industrial-natural system can be assessed (**chapter 2**). We focus here on the life cycles of products, such systems are the collections of the various processes needed to produce, use and dispose a product. A case study was performed on the impact/benefit caused by the life cycle of 1 m^3 sawn timber, encompassing wood growth in the Scots pine stand and industrial processing into sawn timber, usage of latter and burning of the wood. The results indicate that the (wood growth in the) forest was responsible for the larger share of the environmental impact/benefit. As the forest was intensively managed, this implied a biodiversity loss compared to a natural system. This loss, representing damage to ecosystem quality, was responsible for almost all of the diversity loss over the complete life cycle: 1.60E-04 species*yr m^{-3} sawn timber. Next to that, since the Scots pine stand is a plantation and managed intensively, the growth of natural vegetation was prevented, leading to the main loss of natural resources per amount of sawn timber, expressed in exergy (the amount of useful energy obtainable out of a resource, e.g. exergy content of biomass): 3.99E+02 GJ_{ex} m^{-3}. Regarding impact on human health over the life cycle, a total prevention of loss of 0.014 healthy life years m^{-3}

sawn timber is obtained. This health remediating effect could be mainly attributed for 77% to the deposition of particulate matter < 2.5 µm ($PM_{2.5}$) on the vegetative canopy of the Scots pine stand, and to CO_2 uptake for the other share. This case study revealed the potential importance of considering impact of ecosystems in environmental sustainability assessment.

As PM removal appeared to be such a relevant provided forest service, we developed a model to calculate PM removal by a forest ecosystem (**chapter 3**). More specifically, we quantified the amount washed off via rainfall from the plant surface after net-deposition on it. For the Scots pine stand, this resulted in a removal of 7.38 kg $PM_{2.5}$ ha^{-1} yr^{-1} in the year 2010. Integrating this model into a larger forest ecosystem growth model ANAFORE (Deckmyn et al., 2011, 2008), allowed us to calculate PM removal while the forest grows under different conditions. This model was run for different airborne $PM_{2.5}$ concentration scenarios for the Scots pine stand during the period 2010-2030. Estimated avoided health costs due to $PM_{2.5}$ removal of 915-1075 euro ha^{-1} yr^{-1} were obtained for these scenarios. Comparing these values with a rental price of 143.6 euro ha^{-1} yr^{-1} (based on the selling price for the Scots pine stand of 16000 euro ha^{-1}, obtained from the current owner Agency of Nature and Forest, and on a local land buy to rent price ratio) possibly illustrates the for now underrating by society of this (ecosystem service delivered by the) forest.

Additionally, Ecological Network Analysis (ENA) was improved for application in environmental sustainability assessment (**chapter 5**). ENA is a methodology to study and characterize flux networks among defined ecosystem compartments over a certain period of time via indicators, e.g. cycling of nitrogen between different trophic levels of a forest ecosystem over a year. Main reasons for improvement and application of ENA are that a change in ENA-indicator can represent an impact on ecosystems, as an alternative for diversity loss, and ecosystem networks, studied via ENA, may be easily included in environmental sustainability assessment because of the same mathematical backbone. However, prior to application in environmental sustainability assessment the following matter should be addressed. There are no standards yet for the different choices in the ENA methodology, which can have an influence on the indicator values. Hence, defining such standards is a next important research step.

Finally, in light of the overall aim of the PhD, we performed an environmental impact assessment and monetary ecosystem service assessment of the Scots pine stand under different management and environmental change scenarios from the year 2010 up until 2090 (**chapter 4**). For the monetary valuation of ecosystem services, specific monetary values valid for Flanders were used, e.g. 150 euro kg^{-1} $PM_{2.5}$ removed (Broekx et al., 2013; Liekens et al., 2013b). Disservices (e.g. NO_x emission by the forest) are also considered and attributed negative economic values to them. An environmental impact assessment methodology was applied using our previous framework. In that particular framework the uptake of harmful compounds such as CO_2 is considered (Schaubroeck et al., 2013), **chapter 2**, thus the benefit and the damage done by the Scots pine stand to mankind and nature was assessed. The addressed flows/ecosystem services in this analysis are: PM removal ($PM_{2.5}$ and $PM_{2.5\text{-}10}$), freshwater loss, CO_2 sequestration, wood production, NO_x emission, NH_3 uptake and nitrogen pollution/removal. Note that is just a limited number of services/flow, e.g. freshwater loss due to evapotransipartion is considered a disservice while we did not consider the benifical effect of evapotranspiration: the counteracting of global warming by surface cooling (Bonan, 2008).

The management and environmental change scenarios represent the possible (indirect) influence we have on the forest. The ANAFORE model results of these scenarios therefore stand for the potential (indirect) effects which might occur through our actions on the forest, e.g. less wood growth by the forest induced by too much harvest. In latter model, the new PM removal submodel was integrated (**chapter 3**). In practice, three management and three environmental change scenarios were applied, resulting in nine overall scenarios.

Following main results were obtained. The monetary valuation results highlight the importance of services provided by the forest, with a total yearly average of 361-1242 euro ha^{-1} yr^{-1}. $PM_{2.5}$ removal is the key service with a value of 622-1172 euro ha^{-1} yr^{-1}. This is a factor 2.5-8.6 higher than the earlier mentioned rental price. Concerning environmental impact assessment, with CO_2 sequestration and thus the prevention of its damage as the most relevant contributor, a yearly average prevention in loss of healthy life years of 0.014 to 0.029 ha^{-1} yr^{-1} is calculated. There is however a yearly average biodiversity loss of -1.09E-06 to 7.3E-05 species*yr ha^{-1} yr^{-1}, mostly through the intensive land use but counteracted by CO_2 sequestration with 46-101%. The differences between climate scenario results are inferior to the discrepancies induced by the management scenarios. Regarding environmental change we can conclude that the less pollution of mainly $PM_{2.5}$ through more stringent legislation, the less there can be pollution removal, an ecosystem service, and thus overall value provided. Concerning management scenarios, both approaches favor the use of the least intensive management scenario mainly since CO_2 sequestration and PM removal are higher for these, latter induced by a higher needle surface area per ground area. Our framework has thus resulted in the clear selection of the best management scenario of the considered ones and this for the accounted ecosystem services/flows.

Overall, different methodological aspects were improved. Though, there are still a lot of methodological improvements needed. However during this study, it became clear that there was a more urgent issue, the lack in a clear consensus on which matters to prioritize in sustainability assessment. The most important question concering this topic is: ‘Which is more important to maintain: man or nature?’. A simple conceptual framework was proposed for sustainability assessment in which the total impact/benefit on human well-being was put central again, in correspondence with the original definition of sustainable development: ‘the development that meets the needs of the present without compromising the ability of future generations to meet their own’(WCED, 1987).

Samenvatting

Na het realiseren van de impact van zijn humaan/industrieel systeem op de natuur en indirect op zichzelf, is de mens zich bewust geworden van de noodzaak aan een duurzame relatie met de natuur. Om deze duurzame verstandhouding te bekomen, moet onder andere door onderzoek ontrafeld worden welke beheervormen van humaan/industriële en natuurlijke systemen hiertoe het meest geschikt zijn. In onze studie hebben we geprobeerd om het milieuaspect van deze duurzame relatie in een betere manier te kwantificeren, dit geïllustreerd met bosecosystemen. Laatstgenoemd type ecosysteem is van groot belang aangezien het 30% van het landoppervlak beslaat en essentiële diensten aan de mens biedt (FAO, 2010). Een uitdaging die we wilden overwinnen bij de selectie van het beste beheersscenario, is het includeren van de dynamische respons van (bos)ecosystemen, zoals ondermeer het effect van houtdunningen op bosgroei en dus op opname van koolstofdioxide. Specifieke methodes werden hiertoe ontwikkeld of verbeterd. Ter illustratie werden ze toegepast op één bepaald bos: een intensief beheerd grove dennenbestand te Brasschaat, gelegen in Vlaanderen.

Ten eerste werd een raamwerk ontwikkeld waarmee de negatieve/beschadigende en positieve/mitigerende, door bijvoorbeeld CO_2 vastlegging, milieu-impact van een geïntegreerde humaan/industrieel-natuurlijk systeem beoordeeld kan worden (**hoofdstuk 2**). Wij focussen hier op de levenscycli van producten, dergelijke systemen zijn de verzamelingen van verschillende processen die aangewend worden tijdens de productie, het gebruik en finale verwerking van het product. Een casestudie werd uitgevoerd op de levenscyclus van 1 m^3 gezaagd hout, dit omvat de groei van stamhout in het grove dennenbestand, industriële verwerking tot zaaghout, gebruik van deze en finale verbranding ervan. De resultaten van deze studie tonen aan dat (de houtgroei in) het bos verantwoordelijk is voor het grootste aandeel van de milieu-impact. Aangezien het bos intensief beheerd werd, leidde dit tot een verlies aan biodiversiteit ten opzichte van een natuurlijk bos. Dit verlies, dat schade aan ecosystemen representeert, was verantwoordelijk voor bijna alle diversiteitsverlies over de volledig beschouwde levenscyclus: 1.60E-04 soorten*jr m^{-3} gezaagd hout. Daarnaast werd het grove

dennenbestand ook aangeplant en intensief beheerd wat inhoudt dat de groei van natuurlijke vegetatie voorkomen werd op deze locatie, wat leidde to het grootste verlies aan de natuurlijke grondstoffen over de keten, uitgedrukt in exergie (de hoeveelheid energie die men uit een grondstof kan halen ten opzichte van de standaardomgeving): 3.99E+02 GJex m^{-3} gezaagd hout. Inzake impact op menselijke gezondheid, voorkwam dit systeem een verlies van 5.11 gezonde menselijke levensdagen m^{-3} gezaagd hout. Laatste is voor 77% toe te kennen aan de filtering, door depositie op het plantoppervlak, van fijn stof met een diameter < 2.5 µm ($FS_{2.5}$) uit de lucht. Deze studie toont het potentieel belang aan van het in beschouwing nemen van ecosystemen in duurzaamheidsanalyse.

Sinds verwijdering van fijn stof (FS) een relevante geleverde dienst door het bos blijkt te zijn, hebben we een model ontwikkeld om de verwijdering van FS door een bosecosysteem te kwantificeren (**hoofdstuk 3**). Meer specifiek berekent dit model de hoeveelheid fijn stof afgewassen van het plantoppervlak via regenval na netto-depositie op het oppervlak. Voor het grove dennenbestand calculeerden we een verwijdering van 6,58 kg $FS_{2.5}$ ha^{-1} $jaar^{-1}$ voor het jaar 2010. Het integreren van dit model in een groter bosgroeimodel ANAFORE (Deckmyn et al., 2011, 2008), laat toe om FS verwijdering te bepalen over de loop van de tijd heen terwijl het bos groeit onder verschillende omstandigheden. We hebben dan ook FS verwijdering voor het grove dennenbestand tijdens de periode 2010-2030 kunnen bepalen voor verschillende toekomstscenario's qua $FS_{2.5}$ luchtconcentratie. De uitgerekende geschatte vermeden gezondheidskosten als gevolg van $FS_{2.5}$ verwijdering voor deze scenario's bedroeg 915-1075 euro per hectare per jaar. Het vergelijken van deze waarden met een huurprijs van 143.6 € ha^{-1} $jaar^{-1}$ (op basis van de verkoopprijs voor het bestand van 16 000 € ha^{-1}, bekomen van de huidige eigenaar Agentschap van Natuur en Bos, en op een lokale verkoop- tot huurprijs ratio) illustreert mogelijks het onderschatten door de samenleving van (deze ecosysteemdiensten geleverd door) dit bos.

Daarnaast werd de methodologie Ecologische netwerkanalyse, *Ecological Network Analysis* (ENA), verbeterd voor toepassing in duurzaamheidsanalyse (Schaubroeck et al., 2012) (**hoofdstuk 5**). ENA is een methode om de fluxnetwerken tussen bepaalde ecosysteemcompartimenten over een periode heen aan de hand van indicatoren te bestuderen en karakteriseren, bijvoorbeeld het hergebruik (*cycling*) van stikstof tussen verschillende trofische niveaus van een bosecosysteem over het verloop van een jaar. Belangrijkste mogelijke applicaties en reden tot verbetering van ENA in duurzaamheidsanalyse, is dat een verandering in een ENA-indicator een impact op de ecosystemen kan representeren, als alternatief voor verlies aan diversiteit, en ecosysteemfluxnetwerken, bestudeerd via ENA, gemakkelijk kunnen worden geïntegreerd in levenscyclusanalyse, een methode voor duurzaamheidsanalyse, aangezien beiden dezelfde wiskundige methodologie gebruiken. Echter, voorafgaand aan toepassing van ENA in duurzaamheidsanalyse dient het volgend euvel eerst opgelost te worden. Er zijn nog geen standaarden voor de verschillende keuzes die gemaakt kunnen worden bij het uitvoeren van de ENA methode. Deze hebben weliswaar een invloed op de bekomen indicatorwaarden. Vandaar dat het definiëren van dergelijke standaarden een belangrijke volgende stap hoort te zijn in dat onderzoeksgebied.

In het kader van de algemene doelstelling van dit doctoraat werden een milieu-impactanalyse en een monetaire beoordeling van ecosysteemdiensten van het grove dennenbestand onder verschillende klimaat- en beheerscenarios voor de periode 2010-2090 uitgevoerd (**hoofdstuk 4**). Om de ecosysteemdiensten in geldwaarden uit te drukken, werden specifieke monetaire waarden geldig voor Vlaanderen aangewend, bijvoorbeeld 150 € kg^{-1} fijn stof < 2.5 µm ($FS_{2.5}$) (Broekx et al., 2013; Liekens et al., 2013b). Ondiensten (bijvoorbeeld emissie van NO_x door het bos) zijn ook in beschouwing genomen. Aan deze werden negatieve economische waarden toegekend. Een methodologie voor milieu-impactanalyse werd toegepast met behulp van ons eerder vermeld raamwerk waarin de opname van schadelijke stoffen zoals CO_2 wordt beschouwd (Schaubroeck et al., 2013), dus zowel het mitigerend als het schadelijk effect werd beoordeeld. De beschouwde fluxen/ecosysteemdiensten in deze analyse zijn: verwijdering van fijn stof ($FS_{2.5}$ en $FS_{2.5\text{-}10}$), verlies aan zoetwater, CO_2-opslag, houtproductie, uitstoot van NO_x, NH_3 opname en verontreiniging of verwijdering van stikstof. Merk op dat dit slechts een beperkt aantal fluxen/diensten zijn, bijvoorbeeld waterverlies door evapotranspiratie is in rekening gebracht als ondienst maar het voordelig effect van evapotranspiratie door koeling van het aardoppervlak, wat de globale opwarming tegenwerkt, is niet in beschouwing genomen (Bonan, 2008).

De beheer- en klimaatscenarios vertegenwoordigen de mogelijke (indirecte) invloed die de mens op het dennenbestand heeft. De resultaten van het ANAFORE model onder deze verschillende scenarios, stellen dan de (indirecte) effecten van onze acties op het bos voor, bijvoorbeeld minder houtgroei bij te veel oogst. Het toegepaste ANAFORE model bevatte ons ontwikkeld model voor fijn stof verwijdering (**hoofdstuk 3**). Specifiek werden drie beheer- en drie klimaatscenario's toegepast, wat resulteerde in negen algemene scenario's.

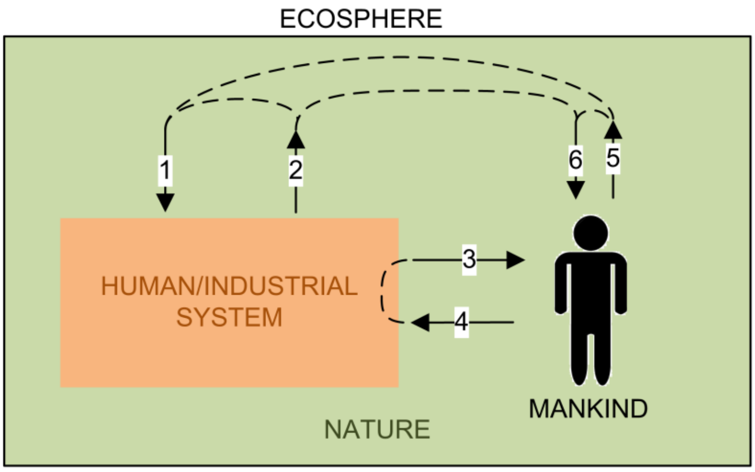
De volgende resultaten en conclusies werden bekomen uit deze analyse. De in geld uitgedrukte ecosysteemdiensten van het bos benadrukken ten eerste het belang van deze, dit met een totaal jaarlijkse gemiddelde van 361-1242 euro per hectare bos per jaar. Dit is een factor 2.5-8.6 hoger dan de eerder vermelde huurprijs ha^{-1} yr^{-1}. $FS_{2.5}$ verwijdering is de belangrijkste ecosysteemdienst, aan een waarde van 622-1172 € ha^{-1} $jaar^{-1}$. Betreffende analyse van de milieu-impact, waarbij CO_2-opslag de meest relevante dienst is, werd een jaarlijks gemiddelde preventie van verlies van 0,014-0,029 gezonde menselijke levensjaren ha^{-1} $jaar^{-1}$ bekomen. Er is echter een jaarlijks gemiddeld verlies aan biodiversiteit van 7.3E-05 tot -1.09E-06 soorten*yr ha^{-1} $jaar^{-1}$. Dit is grotendeels veroorzaakt door het intensieve landgebruik maar ook teniet gedaan door CO_2-opslag aan 46-101%. Het verschil tussen uitkomsten van de drie klimaatscenario's is inferieur aan de discrepantie veroorzaakt door de drie verschillende beheerscenario's. Betreffende klimaatverandering, kunnen we concluderen dat minder vervuiling van voornamelijk $FS_{2.5}$ door strengere wetgeving tot minder verwijdering van desbetreffende polluenten leidt, en daarmee de totale hoeveelheid geleverde diensten door het bos doet verkleinen. Qua beheerscenario's, favoriseren beide benaderingen het gebruik van het minst intensief scenario omdat CO_2-opslag en FS verwijdering groter waren voor deze, veroorzaakt door een hogere naaldoppervlak per grondeenheid. Onze methodiek heeft dus geresulteerd in een duidelijke selectie van het beste beheerscenario onder diegene beschouwd, wat initieel beoogd werd, dit natuurlijk wel enkel voor de beschouwde ecosysteemdiensten en -fluxen.

Verschillende methodologische aspecten van duurzaamheidsanalyse werden verbeterd en geïntroduceerd. Echter, er zijn nog veel verbeteringen nodig. Tijdens het uitvoeren van deze studie werd het duidelijk dat er een meer prangend kwestie is, namelijk het ontbreken van een duidelijke consensus over welke zaken prioriteit hebben bij de beoordeling van duurzaamheid. De belangrijkste vraag hierbij is: 'Wat is belangrijker om te behouden/beschermen: de mens of de natuur?'. Een eenvoudige conceptuele kader wordt voorgesteld voor de beoordeling van duurzaamheid waarbij de totale impact op het menselijke welzijn opnieuw centraal wordt gesteld, dit in overeenstemming met de oorspronkelijke definitie van duurzame ontwikkeling: 'de ontwikkeling die de behoeften van het heden beantwoordt zonder het beperken van het vermogen van toekomstige generaties om in hun eigen noden te voorzien' (WCED, 1987).

Chapter 1 Introduction

1.1 A need for a sustainable relationship between nature and mankind

Similar to every species, since the beginning of its existence mankind exploits its environment (the 'ecosphere' (Huggett, 1999)), from oxygen consumption over application of medicinal herbs to a wound, to maintain itself in harsh/damaging environmental conditions. The human species has been extremely adaptive and, functioning as an ecosystem engineer, created and performed numerous processes in its environment to aid in its own survival, and to increase its life quality (Smith and Zeder, in press). As the role of humans in the ecosphere became ever more prominent, an abstract boundary was drawn between this collection of processes, the so called 'human/industrial system' or 'technosphere', and the rest of the environment, mostly referred to as 'nature', besides mankind itself that we consider as a separate entity (Figure 1.1).



1: ecosystem resources for the human/industrial system (e.g. wood)
2: emissions (e.g. CO_2) to the natural system by the human/industrial system
3: economic/industrial products and services (e.g. car) and toxic compounds (e.g. airborne particulate matter)
4: management of the human/industrial system, human emissions (e.g. garbage) to be processed
5: emissions by humans (e.g. CO_2 exhalation)
6: ecosystem goods and services directly provided, but also damaging ones (e.g. diseases)

Figure 1.1. Flows, material and non-material, between mankind, its human/industrial system (orange) and nature (green), all three considered as separate entities of the ecosphere. For clarification, the most common interpretation of parts of these flows are given. Some indirect effects are illustrated with dotted lines.

Man invested a lot of time and energy in improving its human/industrial system in exploitation of nature to satisfy its needs. Later on, we however discovered the adverse effects, besides the positive ones, of our human/industrial processes and activities on our own health, on nature and the rest of the human/industrial system (Carson, 2002; Rockström et al., 2009). In this context, the need for a sustainable development, being "the development that meets the needs of the present without compromising the ability of future generations to meet their own", was called forth in the famous Brundtland commission (WCED, 1987). In practice, this is often implemented as the 'Triple Bottom Line": social, environmental and economic sustainability (Elkington, 1999). However, environmental sustainability should be prioritized because a society and its economy are bounded by planetary/environmental limits, e.g. a limited amount of fresh water (Griggs et al., 2013; Muys, 2013). Here we will therefore focus only on environmental sustainability.

To meet this demand for an environmentally sustainable relation/development between mankind and nature, effort has been put in management of the human/industrial system and nature to obtain minimization of the adverse impact of our activities and processes, but also maximization of products and services of nature. The interactions within nature may after all be altered through human management/intervention, e.g. reduction in global warming gas emission by converting a swamp to a rice field (Jiang et al., 2009).

The dynamic responses of all the main actors makes the achievement of an environmental sustainable relationship though a hard nut to crack. An alteration of nature (by flows 2 and 5 in Figure 1.1) might change or even endanger the provision of products and services of it to humans (flow 6 and indirect flow 1 in Figure 1.1), e.g. drought induced by climate change may lead to loss of harvestable biomass and recreational services of an ecosystem (Banerjee et al., 2013). A change in industrial processes by mankind will also have an impact on humans, a feedback. These dynamic responses and the indirect effects they induce should be accounted for as well (dotted lines Figure 1.1).

Overall, tools are needed/developed that assess the environmental sustainability of the relationship between mankind (possibly via its industrial system) and nature, and that cover the indirect effects evoked through dynamic responses of these systems. The most important tools and their shortcomings are addressed shortly in the next sections.

1.2 Tools to assess the environmental sustainability of the mankind-nature relation

In general, development and application of tools to assess the environmental sustainability of interactions between mankind and nature has skyrocketed since the Brundtland report written in 1987 (WCED, 1987).

On the one hand, tools were developed to assess the impact of the human/industrial system on humans and on nature (focusing on flows 2 and 3 of Figure 1.1). Different methodologies have been developed, with the main difference between them the entity to which they attribute and normalize this impact: to a product or service (life cycle assessment), to a region or project (environmental impact assessment), to a substance flow over time (substance flow analysis) (Heijungs, 2001). For convenience, we will focus on the most popular of these methods, namely Life Cycle Assessment (LCA), though findings/improvements are also applicable to the other methodologies. In particular LCA quantifies the environmental impact, only the damage aspect of it, of resource extraction and emissions of a human/industrial product's life cycle (ISO, 2006a, 2006b) (Figure 1.2).

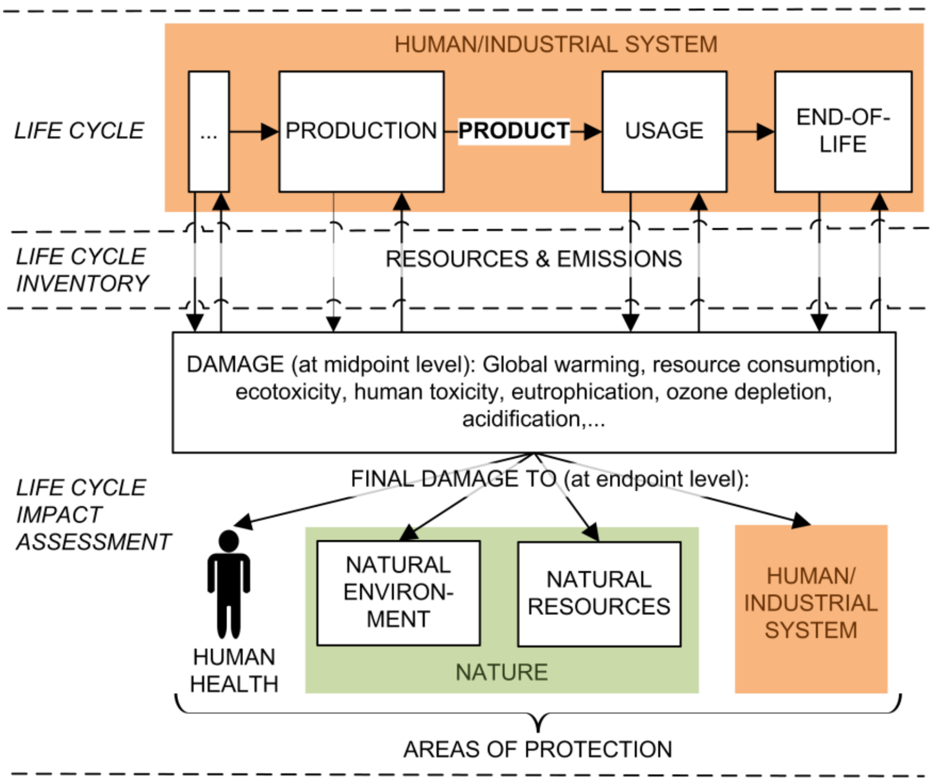


Figure 1.2. Concept of life cycle assessment of a product with the different stages. The impact at endpoint level addresses four areas of protection, brought forward by de Haes et al. (1999), among which the human/industrial system is often not considered.

On the other hand, another important tool in environmental sustainability assessment of the man-nature relationship exists: the 'ecosystem services' assessment. Ecosystem services are described as the direct (flow 6 of Figure 1.1, pg. 2) and indirect (flow 1 of Figure 1.1, pg. 2) contributions of ecosystems to human well being (de Groot et al., 2012; Maes et al., 2013), well described in the work of Hassan et al. (2005) (Figure 1.3). The ecosystem services span a wide range of commodities, e.g. for forests from wood to recreation. This concept and thus also corresponding tools which only assess these services, emphasize on benefits of an ecosystem towards mankind.

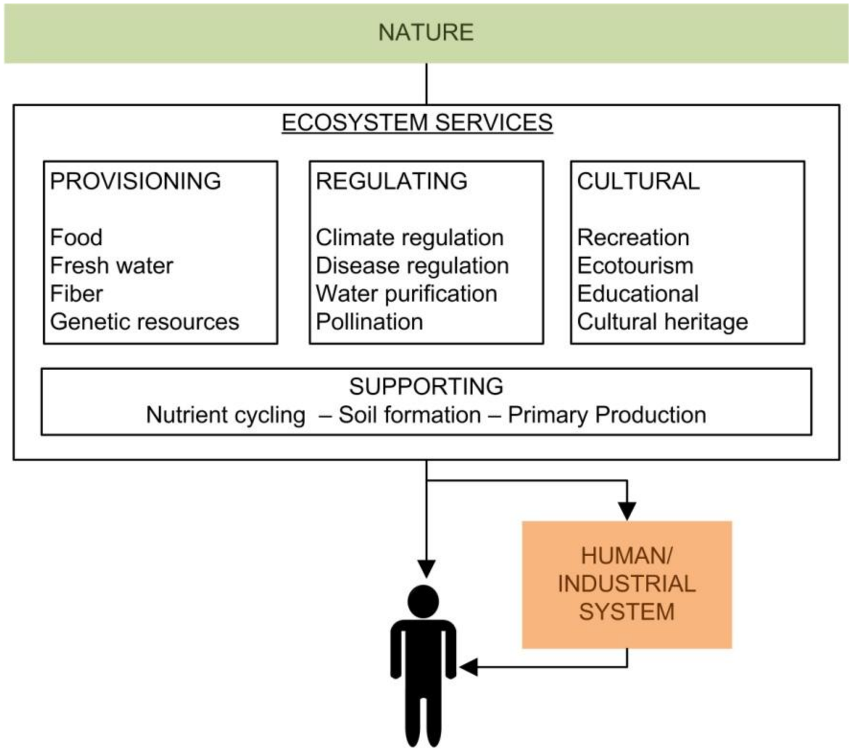


Figure 1.3. Concept of ecosystem services, the services provided by ecosystems towards mankind (Hassan et al., 2005). The different types of services are presented with some examples.

One could consider that these two types of tools are presumably derived from two different (popular) environmental paradigms: 'the human/industrial system harms nature and mankind' for LCA and 'nature is good for mankind' for ecosystem services assessment. Both are one-sided perspectives. A more comprehensive approach is therefore needed, which includes all relationships between humans and nature, the harmful and beneficial ones, from nature to mankind and vice versa. An overall methodology to cover all these fluxes (Figure 1.1, pg. 2) and their damaging and beneficial aspects, is though still lacking. This is a first important need.

The mentioned methodologies might include some modelling of the human/industrial system and nature to assess indirect effects, e.g. airborne industrial emitted NO_x can form particulate matter which harms mankind, though this is often in a preliminary manner assessed using predefined characterization factors (Goedkoop et al., 2009). Better models, discussed in the next section, should be used in combination or integrated into previous assessment tools, representing a second relevant need.

1.3 Tools to quantify dynamic responses of and their induced indirect effects between mankind and nature

To characterize indirect effects, including feedback loops, in the flows/relationships between mankind, its human/industrial system and nature, depicted in Figure 1.1, pg. 2, measurements/observations or models can be used. In light of obtaining an environmentally sustainable relationships under changing conditions (such as climate change), predictive models to characterize responses to a flow/relationship changes are a must. The goal is mainly to select the best, in this case most environmentally sustainable, management practices using models.

Human interactions with the human/industrial system are straightforward: through work and management humans indirectly obtain products from this system. Economic models are used to predict the system's performance. A lot of research is done on this matter, see the work of Basu and Kronsjo (2009) for an overview, but it is not the focus in this manuscript.

We focus on environmental sustainability and will only consider modelling of ecosystems. A review on ecological modelling is given by Fath et al. (2011). One of the most applied models are the empirical models which predict wood growth and stand characteristics of forest under certain management practices (Pretzsch, 1999).

Note that integrated models exist, which model both nature and the human/industrial system, though these are not used in our study, since as mentioned we will only include ecosystem modelling. A fine example of an integrated model is that of Arbault et al. (2014).

1.4 Forests and their ecosystem services, with focus on particulate matter removal

In this work we will study one specific ecosystem type and its interactions with mankind, namely the forest. According to the Food and Agriculture Organization (FAO, 2010), a forest is defined as: "Land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use." It is one of the main ecosystem types of nature as forests covered approximately 31% of the world land area, just over 4 billion hectares, in 2010 (FAO, 2010) (Figure 1.4).

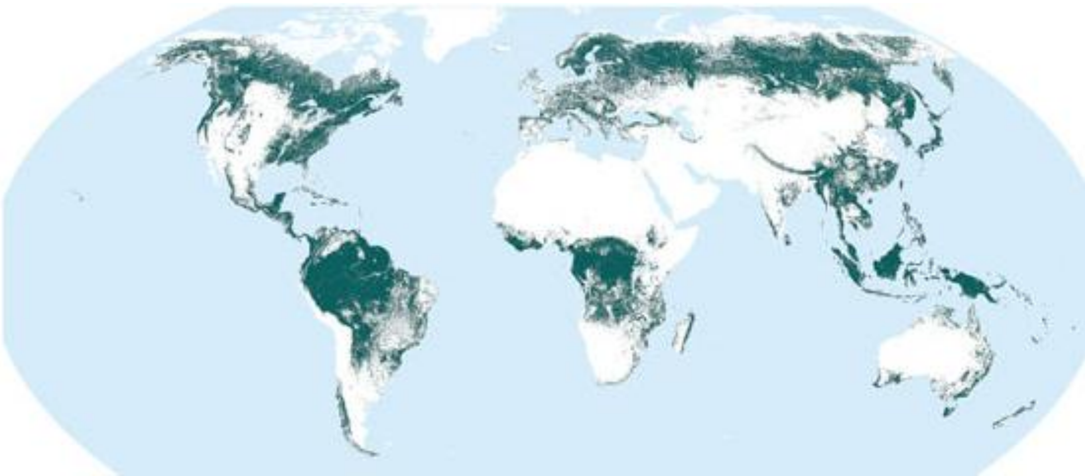


Figure 1.4. Map of global forest (>10% tree cover) area (shown in green) for the year 2005 (FAO, 2010).

Due to deforestation, this total area is slowly reducing but the rate of removal is however lowering from 0.20% per year between 1990-2000 to 0.13% between 2000-2010 (FAO, 2010). This deforestation is mainly due to conversion of forests to agricultural land. As other land uses often offer more direct benefit to mankind, the various benefits of forest are frequently overlooked (Figure 1.5). Renewable production of wood is a unique irreplaceable asset of forests. In 2005 3.4 billion m^3 wood was reported to be harvested worldwide (FAO, 2010). Let us not neglect to mention one of the more recently highlighted important services of forests: the sequestration of carbon dioxide (Pan et al., 2011). Next to that, removal of particulate matter by forests, through deposition on plant surface, is regarded as an important provided service (Fowler et al., 2009; Nowak et al., 2013). Though research is still needed to model the quantity of particulate matter removed (Petroff et al., 2008; Pryor et al., 2008). In this work we will attempt to quantify this service in a better manner.

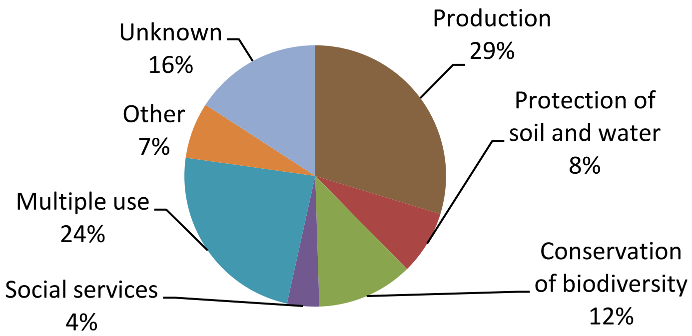


Figure 1.5. Designated/primary functions of forests in 2010 (FAO, 2010). Percentages represent share of forests which has this specific designated/primary function.

Regarding all these provided services, deforestation is by consequence also counteracted by afforestation to maintain this crucial ecosystem and its goods and services (FAO, 2010). Research is however still ongoing and needed to unravel all relevant environmental sustainability features of forest and how to best manage them. This aspect we also want to address in our study.

1.5 Objectives and outline of the work

The overall aim is the development of an improved framework, in which the environmental sustainability of the man-nature relationship is better assessed, while including the dynamic response of ecosystems, e.g. to climate change, to highlight best ecosystem management practices, this illustrated with an application on a man-forest relationship. In particular, (the perspectives of) LCA and ecosystem service assessment need to be both used. The two main shortcomings of these assessment tools are: a lack in considering all bidirectional relationships (damaging and beneficial) between man-nature and a need for integration with ecosystem models to better address the dynamic response of ecosystems, evoking indirect effects, both shortcomings are mentioned in section 1.2, pg. 4.

In the given context, four main objectives are defined:

1. Development/improvement of an LCA-based tool to assess the environmental sustainability assessment of the bidirectional man-nature relationship, in particular an integrated human/industrial-natural system, in a better manner.
2. The development of a model to quantify one of the most relevant ecosystem services, more precisely particulate matter removal by a forest.
3. Providing a framework that addresses the above mentioned overall aim.

   Research question: How to pinpoint the most environmentally sustainable way of managing an ecosystem, in this case a Scots pine stand?
4. Apply the developed tools to forest ecosystems and derive best management practices.

The fourth objective narrows down this relationship to only with one ecosystem type, to make the other objectives realizable and tangible. More precisely, the methodological improvements were applied to one specific forest ecosystem, a Scots pine stand, discussed in the next section, 1.6.

To better grasp, the objectives, specific research questions can be formulated:

1. What is the environmental impact/benefit change of a (wood) production system if a(n) (Scots pine forest) ecosystem is included? Does nature, e.g. the Scots pine forest, or the human/industrial system has the highest environmental impact/benefit; which is more relevant to better manage?
2. How much particulate matter will a certain forest remove in the future?
3. How to pinpoint and what is the most environmentally sustainable way of managing an ecosystem, in this case a Scots pine stand?

To realize these objectives, different studies have been performed. These are presented in the different chapters of this dissertation besides 'Introduction' (**chapter 1**), and 'Conclusions and perspectives' (**chapter 6**).

First, to achieve the first objective, tools to assess the environmental sustainability of the mankind-nature relationship/flows were improved in **chapter 2**. In this particular chapter, we erased the abstract boundary between the human/industrial system and nature and combined the ecosystem services and Life Cycle Assessment (LCA) approach, to assess the complete life cycle of an economic product in a more objective and complete manner. A case study was performed on sawn timber in which also the impact of the forest, where the wood was originally grown, and the uptake of pollutants by that forest, an ecosystem service, are assessed. An important conclusion of this chapter was that deposition of airborne particulate matter smaller than 2.5 µm ($PM_{2.5}$) on plant surfaces of a forest ecosystem can be the most important benefit over the complete life cycle for human health, even more than CO_2 sequestration, provided by a forest. Prior to addressing forest ecosystems in a dynamic manner, a model needed to be developed to quantify this PM removal better (second objective), which is done in **chapter 3**.

In chapter 2 only a static approach was considered, without a dynamic response of the systems. Therefore, the developed new methodological framework was integrated with a forest growth model, ANAFORE (Deckmyn et al., 2011, 2008), for different management and climate scenarios in **chapter 4**. Latter scenarios induce indirect effects brought forward (partially) by mankind and its human/industrial system. Chapter 4 thus addresses the third objective.

Damage to ecosystems in general, as in chapter 2, is often addressed through loss in species diversity in LCA. As Goedkoop et al. (2009) point out that this approach only covers the loss/change in information and not that of material and energy. In search for alternatives, Ecological Network Analysis (ENA) was improved to potentially meet this demand, elaborated in **chapter 5**. Next to that, because of the same mathematical backbone of ENA and LCA, ecosystem flow networks of ENA-studies can be easily integrated in the framework of chapter 2 to include ecosystems in product life cycles. Figure 1.6 gives an outline of the dissertation.

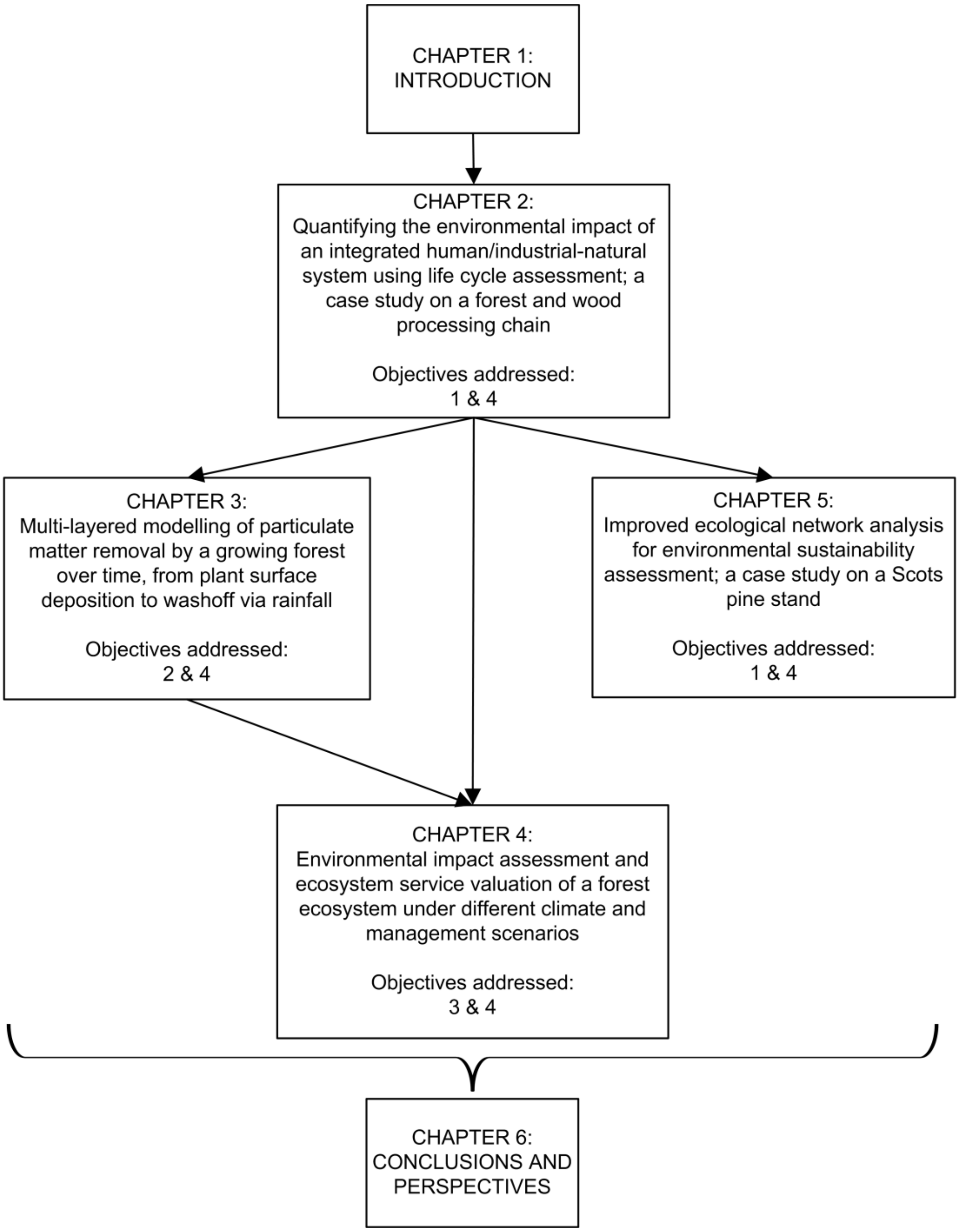


Figure 1.6. Outline of the PhD dissertation.

1.6 The Scots pine stand

The studied forest ecosystem is a managed 2-hectare Scots pine (*Pinus Sylvestris* L.) stand located in the forest 'De Inslag' (150 ha mixed coniferous/deciduous forest) 20 km NE of Antwerp, situated in the Campine region of Flanders (Belgium), country of Europe (51°18'33''N, 4°31'14''E) (Figure 7).

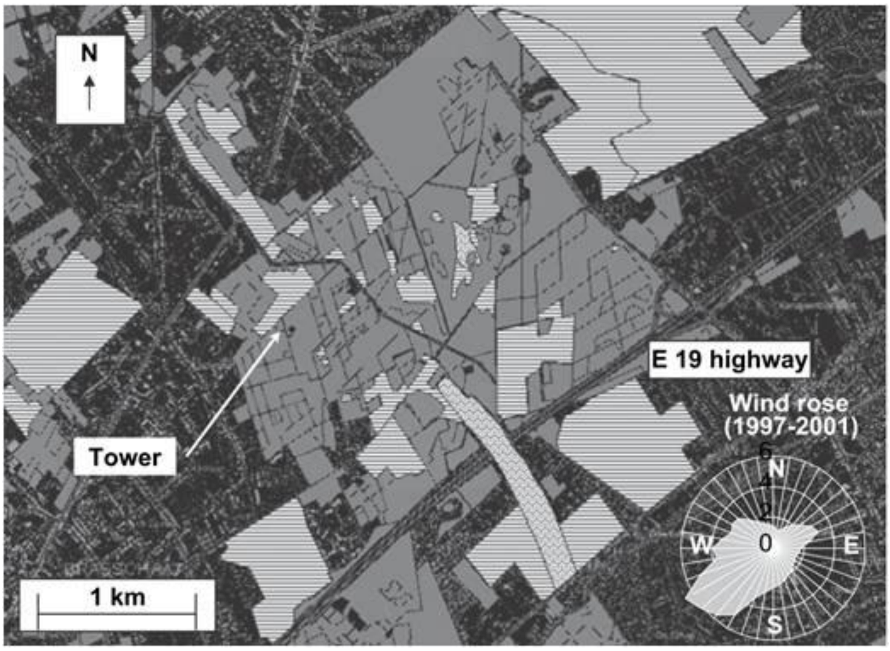


Figure 1.7. Location of the Scots pine stand with measurement tower in the experimental forest site (grey: forest, black: residential areas, waves: water pools, horizontal bands: low vegetation types such as meadows, clearcuts or moorlands). The presence of the E19 highway has an important influence on the particulate matter concentration, this also since the wind is mostly coming from the southwest. This map is retrieved from Neirynck et al. (2007).

This site is a level II observation plot of a European program for intensive monitoring of forest ecosystems, their vitality, effect on air pollution and carbon flux measurements. It has been thoroughly researched in numerous studies, mainly by the University of Antwerp and the Flemish research institute for Nature and Forest. For chapter 2 and 5, data representative for the period 2001-2002 of the forest was used and for chapter 3 that for 2010. In case of chapters 4 and 5, the ANAFORE forest growth model was used, parameterized to this Scots pine stand (Deckmyn et al., 2011, 2008).

The area of the Scots pine stand has a maritime **climate** with a mean annual temperature of 11 °C and a precipitation of 830 mm (Campioli et al., 2011). The site has a flat topography with a slope of less than 0.3% and an elevation of 16 m above sea level. It is also characterized by a high nitrogen deposition of 48 kg N ha^{-1} yr^{-1} (Neirynck et al., 2008).

The **soil** is classified as a podzol and consists of an ectorganic layer with a moder type of humus, an aeolian sand layer with an hemi-organic surface layer (6-8cm), an eluviation horizon and a distinct humus and iron B horizon, on a substratum of Campine Clay (40% clay) at a depth varying between 1.2 and 2.5 m. The groundwater table is usually at a depth of 1.2-1.5 m (Baeyens et al., 1993). The soil is moist, but rarely saturated, because of the high hydraulic conductivity of the upper layers. A more detailed soil description is given by Neirynck et al. (2002) and Janssens et al. (1999).

Regarding **vegetation**, Scots pine is a coniferous, evergreen species. It is an abundant species in Flanders, the most important considering standing volume (INBO, 2007), and a major tree species in Europe (Skjøth et al., 2008; Tröltzsch et al., 2009). The Scots pine trees were planted in 1929 and the plot consist only of this tree, considering 100% canopy cover (Figure 1.8). Table 1 gives an overview of stand characteristics. In the years 2001-2002, the surface area was considered to be covered for 54% by black cherry (*Prunus serotina* Ehrh.), 8% by rhododendron (*Rhododendron ponticum* L.) and 20% by purple moor grass (*Molinia caerulea* L. Moench), with a non-vegetated area of 18% (Nagy et al., 2006). These areal percentages were also used as estimated contributions of the different understory species to the understory biomass.

Figure 1.8. The Scots pine stand and its measurement tower, shown in the upper right. The not completely closed canopy cover is pictured in the upper left one. The tree stand consists of an even-aged single-species as can be seen from the similar tree stems on the bottom left and right pictures.

Table 1.1. Stand characteristics of the Scots pine stand. Tree height for the year 2010 was not measured but that of 2008 is given. DBH: Diameter at Breast height.

Parameter	Unit	2001-2002	2010
Source		(Yuste et al., 2005)	(Gielen et al., 2013)
Used in chapters		2 and 5	3
Stand density	Trees ha^{-1}	377 (winter 2001) – 361 (winter 2002)	361
Average DBH	cm	29	33
Basal area	m^2 ha^{-1}	24	31
Tree height	m	21.4	21.2 (2008)
Tree age	years	72-73	80

Management of the forest consisted out of several thinnings but also removal of understory vegetation. Latter was repeatedly done until 2001 (Gielen et al., 2013). The known harvesting of trees and tree densities over time are mentioned in Table 1.2. Frequent thinning occurred between the period 1980-1997, according to Neirynck et al. (2008). The thinning of 1999 was however done because of poor site management in the past, and it was mainly surpressed trees that were removed, as stated by Xiao et al. (2003). We believe that this poor site management could be a reason for the incomplete canopy closure later on in the forest. Op de Beeck et al. (2010a) after all mention a gap fraction of 42% in the period 2007-2008 (Figure 1.8).

Table 1.2. Management/ history of the Scots pine stand, the tree density and harvest quantities for which numbers are given in literature are mentioned. N/A: data or info not available.

Year	Age	density	Harvest (winter)	source
	years	Trees ha^{-1}	Trees ha^{-1}	/
1980	50	1390	N/A	(Neirynck et al., 2008)
1987	57	899	N/A	(Neirynck et al., 2008)
1995	65	538	N/A	(Xiao et al., 2003)
1999	69	377	163	(Xiao et al., 2003)
2001	72	376	1	(Xiao et al., 2003)
2002	73	361	15 (fell in storm)	(Xiao et al., 2003)
2010	80	361	N/A	(Gielen et al., 2013)

During 2001-2002 on average 8 trees ha^{-1} yr^{-1} were harvested (Yuste et al., 2005), with properties assumed to be equal to the average tree of the stand. Only stem wood was harvested, and the remaining aboveground parts were left as slash.

Chapter 2 Quantifying the environmental impact of an integrated human/industrial-natural system using life cycle assessment; a case study on a forest and wood processing chain

Redrafted from:

Schaubroeck, T., Alvarenga, R. A. F., Verheyen, K., Muys, B., Dewulf, J., 2013. Quantifying the Environmental Impact of an Integrated Human/Industrial-Natural System Using Life Cycle Assessment; A Case study on a Forest and Wood Processing Chain. Environ. Sci. Technol. 47, 13578-13586.

Abstract

Life Cycle Assessment (LCA) is a tool to assess the environmental sustainability of a product; it quantifies the environmental impact of a product's life cycle. In conventional LCAs, the boundaries of a product's life cycle are limited to the human/industrial system, the technosphere. Ecosystems, which provide resources to and take up emissions from the technosphere, are not included in those boundaries. However, likewise to the technosphere, ecosystems also have an impact on their (surrounding) environment through their resource usage (e.g. nutrients) and emissions (e.g. CH_4). We therefore propose a LCA framework to assess the impact of integrated Techno-Ecological Systems (TES), comprising relevant ecosystems and the technosphere. In our framework, ecosystems are accounted for in the same manner as technosphere compartments. Also, the remediating effect of uptake of pollutants, an ecosystem service, is considered.

A case study was performed on a TES of sawn timber production encompassing wood growth in an intensively managed forest ecosystem and further industrial processing. Results show that the managed forest accounted for almost all resource usage and biodiversity loss through land occupation but also for a remediating effect on human health, mostly via capture of airborne fine particles. These findings illustrate the potential relevance of including ecosystems in the product's life cycle of a LCA, though further research is needed to better quantify the environmental impact of TES.

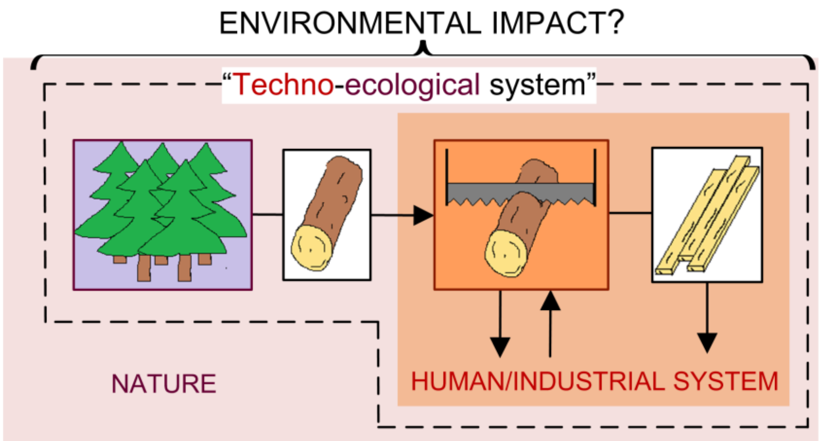


Figure 2.1. Graphical abstract

2.1 Introduction

It is a challenge to provide metrics that quantify the environmental sustainability of a product. In this context, tools such as Life Cycle Assessment (LCA) play an important role as they quantify the impact on the environment of a product's life cycle, comprising its production and optionally its use and its end of life phase (ISO, 2006a, 2006b). In conventional LCAs, a product's life cycle is limited to the boundaries of the human/industrial system, the technosphere (Figure 2.2, case A). The environmental impact of the product's life cycle in the technosphere is assessed as the total impact of resource extraction from and emissions into the environment, i.e. the rest of the ecosphere (Huggett, 1999) (Figure 2.2). The technosphere is therefore studied to obtain these system specific resources and emission flows. Commonly, the impacts of these emissions and resource extraction on the environment, among which impact on ecosystems and human health, are then quantified using Life Cycle Impact Assessment (LCIA) methods. In these methods, often typical generic cause-and-effect relationships are considered, which are series of general ecosystem processes, instead of local or regional ones. For example, the damaging effect of certain metals on the species living in ecosystems is quantified through a generic approach, though the damaged amount of species depends on the amount present in the affected ecosystems (Goedkoop et al., 2009). This generalization in the cause-and-effect relationships can thus be debated. Next to that, the particular interacting ecosystems also use specific resources, take up explicit human/industrial emissions and release specific harmful substances, this all in particular amounts, interacting both with the natural and human/industrial system. For example a forest provides wood but can also emit quantities of NO, CO_2 and other compounds, requires solar energy and occupies a piece of land. There is thus a need for LCAs on specific combinations of particular ecosystems and human/industrial systems, to be considered as integrated systems (Berkes et al., 2000; Liu et al., 2007; Young et al., 2006). On one hand, one can do so by improving the LCIA methods through making them specific for considered ecosystems (and their processes) which provide or receive emissions. In light of this, new regionalized impact assessment methodologies are being developed which assess the impact for a specific affected region, and thus its particular ecosystem (R. A. F. Alvarenga et al., 2013; Baan et al., 2012; Mutel et al., 2012; Saad et al., 2013). On the other hand, one may consider the environmental impact of the integrated human/industrial-natural system and thus expand the system boundaries of the life cycle in an LCA study beyond the technosphere to include specific relevant ecosystems, accounting for their ecosystem processes in the product's life cycle at the inventory stage (Figure 2.2). We call such an integrated system of the human/industrial system and particular ecosystems a Techno-Ecological System (TES).

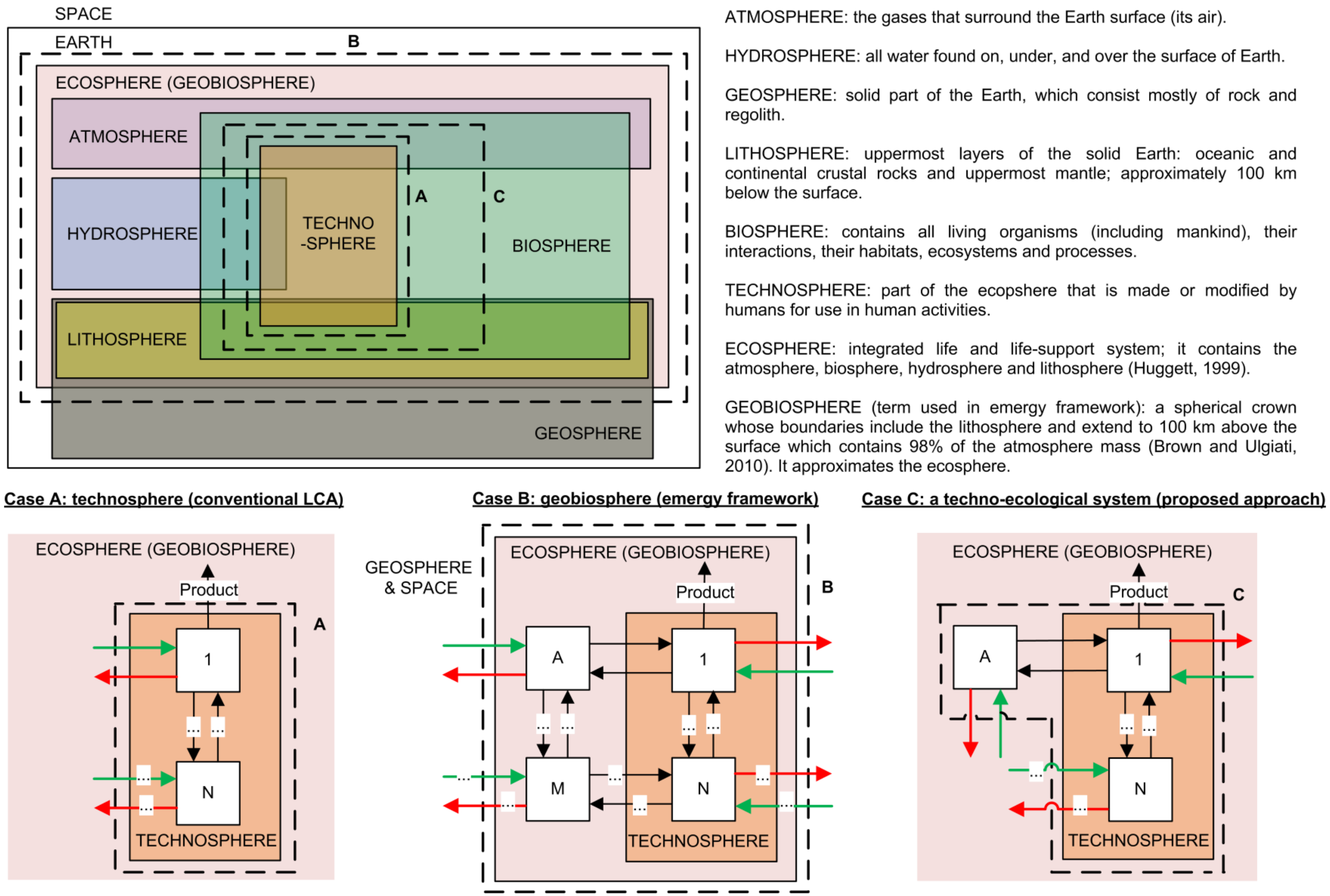


Figure 2.2. System boundaries for different approaches in environmental sustainability assessment. At the bottom are the different cases: A, B and C. In the graphs illustrating the cases, the technosphere consists out of 1 to N compartments and the rest of the ecosphere out of A to M compartments. In case C, as an example of a Techno-Ecological System (TES), one ecosystem compartment A is included in the system boundaries, but more might be included. System boundaries are in dotted lines. Resources from and emissions to the surrounding environment are represented by green and red arrows, respectively.

In this work, focus is on the second option and we therefore propose a framework to perform an LCA on an integrated TES, illustrated in case C of Figure 2.2. in which an ecosystem compartment A is included in the studied system besides the technosphere. An ecosystem compartment is an ecosystem (process) or a part of it, conventionally not considered as part of the human/industrial processes, e.g. a forest stand.

After Cumberland (1966) drew attention to it, Isard (1968) and Daly (1968) were the first to work out frameworks in which ecosystem compartments and human/industrial compartments are integrated, being TES, with interactions between and within them. In their frameworks ecosystems are considered in a similar manner as economic systems, making no distinction between them. In the work of Isard (1969, 1968), this was applied to real case studies. The goal of their research was only to study a certain region and the effect of changes on it, e.g. construction of a town in a bay area (for more information regard ISA (2013)). Similar, our framework will consider ecosystem compartments as human/industrial but ours is specifically applicable to LCA. Heijungs (2001) discusses the framework of Isard (1968). He states that in this context, ecosystem compartments, more specifically processes, should not be considered in a similar manner as economic ones, because only of the latter, the operating time can be regulated. To the contrary, from a thermodynamic point of view, there is no essential difference between (the regulation of) ecosystem and economic processes. Thus, the approach of Isard (1968) and ours, which make no difference between human/industrial and ecosystem compartments, are still valid options.

Besides mentioned works, the inclusion of ecosystem accounting in environmental sustainability assessment has been done in (related methodologies of) the ‘emergy’ accounting framework, well described in the works of Odum (1996) and Brown and Ulgiati (2010). In that framework, the resource amount needed for the production of a good is quantified as the cumulative amount of exergy, called emergy, needed from outside of the geobiosphere (Brown and Ulgiati, 2010). System boundaries surround the complete geobiosphere, which approximates the ecosphere (Figure 2.2, case B). The TES in the emergy framework thus equals the complete ecosphere and cannot be freely chosen. By consequence all ecosystems and physicochemical processes, often as generic processses, such as rainfall are included besides the human/industrial processes. The focus of emergy accounting is to obtain an environmental cost for different commodities, while that of LCA is environmental impact assessment of these. Hence, the harmful effect of emitted compounds is not accounted for in the emergy framework but is in ours, e.g. effects of CO_2 on climate change. But for resource accounting in LCA, some methodologies have been developed by attributing emergy values to resources and thus including ecosystem production processes in the product’s life cycle chain (Liao et al., 2011; Rugani et al., 2011; Rugani and Benetto, 2012; Zhang et al., 2010a). Though, these frameworks, likewise emergy, do not address the impact of emissions.

An important step in LCA is the construction of the Life Cycle Inventory (LCI), this is an inventory of all the resource and emission quantities of the considered life cycle/production system for a given amount of product. In practice, Linear Inverse Models (LIM) are often used to calculate these quantities. Three approaches have been developed in the field of LCA to apply LIM to a system: process based, input-output based and a combination of these two, called hybrid (Suh and Huppes, 2005). If ecosystem compartments are to be included, considering them in a similar manner as human/industrial ones, an adequate mathematical model needs to be applied which also quantifies the flows of all these compartments. Isard already had the idea too and took the first steps to apply LIM to a TES (Isard and Office, 1972). In the world of LCA, this has been achieved by use of LIM in a process based approach in the outline of Rugani and Benetto (2012) and in the Ecological Cumulative Exergy Consumption (ECEC) framework of Hau and Bakshi (2004) (the latter was adapted to the LCA framework by Zhang et al. (2010a)). These models, however, only calculate the amount of resources of a TES, not the amount of its emissions. Moreover, in the framework of Rugani and Benetto (2012) only flows to the technosphere can be accounted for and not flows from the technosphere to the ecosystem compartments. This makes the framework incapable of a full integration in a TES, accounting for flows from and to ecosystems. However, a mathematical model is needed which also accounts for emissions besides resources and which is capable of full integration of a TES, a first objective.

Some additional assets are included in the framework. Ecosystems may take up harmful compounds and/or process them, e.g. CO_2. In this framework, the uptake of such compounds by compartments, also human/industrial, is considered. This is a second objective. This is done by accounting for the avoided environmental damage which these compounds would otherwise exert. Next to that, in LCA methodology compartments are assumed to be in steady state, no change in storage/stock (Suh, 2004), which is in reality hardly true. This problem is circumvented by considering a long enough process operation times so that steady state is approximated (Suh, 2004). A better solution needs to be presented and used in our framework, which is the third objective.

Finally, the new framework will be applied to a case study, notably a full life cycle of sawn timber which includes the growing of stem wood in an intensively managed forest, further processing to sawn timber in a forestry industry, usage and final disposal.

2.2 Material and methods

2.2.1 Framework

In our framework ecosystem compartments and their environmental impact will be included in a LCA following the ISO 14040/14044 guidelines (ISO, 2006a, 2006b). In the goal and scope definition, particular ecosystem compartments are included in the system boundaries of the considered life cycle. Each envisaged ecosystem compartment can be considered as a process or a sector, in the same manner as human/industrial processes or sectors are approached in the process or input-output based approach, respectively (Suh and Huppes, 2005).

Subsequently, the Life Cycle Inventory (LCI) is created through quantification. Some important rules are made in light of the second and third objective for the creation of a LCI. These apply to any type of compartment: human/industrial or ecosystem ones:

1) In this framework, the uptake of harmful compounds, causing damage to the areas of protection (de Haes et al., 1999), by compartments is considered, the second objective. Practically, this is done by representing the amount of a particular harmful compound taken up by the compartment as a negatively valued amount in the inventory. By consequence, the environmental impact might have a negative value after performing a Life Cycle Impact Assessment (LCIA), meaning the system provides an environmental benefit: a remediation effect. For example, for a CO_2 uptake of x kg by a forest, – x kg CO_2 is inventoried, which leads to a Greenhouse Warming Potential (GWP) of – x kg CO_2 equivalents.

2) Stock changes occur in almost all systems. They are net changes in mass or energy over the time period in which a compartment is studied and are here considered as flows. Depletion is an ingoing flow of commodity in time but not in space, as the depleted stock originates from the period before the considered time period. Increment is in fact an output flow as it is the accumulation of a commodity. As done with output flows, increment flows are regarded as products or as wastes. This approach has been reintroduced by Schaubroeck et al (2012), Chapter 5, pg. 133.

Regarding the first objective, if a LCI needs to be modelled, we propose to use the existing approaches based on Linear Inverse Modelling (LIM) described in Suh and Huppes (2005). These modelling approaches are not changed. Ecosystem compartments are just considered as additional economic ones when using them. A general methodology is described in supporting information (SI), section 2.5.1. The steps of LCIA and life cycle interpretation follow the approaches of a conventional LCA.

2.2.2 Case study

2.2.2.1 Scope definition and system description

As a case study a process-based LCA will be applied on a life cycle in which an upstream ecosystem, providing a resource, is added to an industrial production process using the abovementioned framework, following the ISO 14040/14044 guidelines (ISO, 2006a, 2006b). The studied TES is a complete life cycle of sawn timber with as foreground system the production of wood in an intensively managed forest, the further industrial processing, usage (only considered transport) and disposal through burning in a municipal solid waste incineration plant with net production of heat and electricity (Figure 2.3). On this TES, a cradle-to-grave LCA is performed for the sawn timber product. Note that here only the wood amount which has grown during the period of study is harvested, making it a completely renewable resource. The specific forest ecosystem is the Scots pine stand described in section 1.6, pg. 12, with studied period 2001-2002. The foreground human/industrial chain in the technosphere is modelled using different processes from Ecoinvent v2.2 (Swiss Centre for Life Cycle Inventories, 2010). The net electricity generated is a product of the wood disposal through burning. To account for this, system expansion is used by displacement of the processes needed to conventionally generate this electricity amount for the Belgian grid. Net produced heat is considered to be wasted. The background processes of the technosphere, which provide goods or services for the foreground industrial processes, are represented by those in the Ecoinvent v2.2 database. One m^3 of sawn timber wood is selected as a Functional Unit (FU) for the complete production chain. For additional information concerning system description, see supporting information, section 2.5.2.

2.2.2.2 Life cycle inventory

The resource and emission flow data of the ecosystem compartment, the Scots pine stand, were obtained from the work of Schaubroeck et al. (2012), Chapter 5 (pg. 133), and the data sources used therein. Some changes have been made though and these are hereafter elaborated. Firstly, a minimal amount of harvest occurs in the Scots pine stand during the studied period. However the harvest and the effects on the Scots pine stand are not considered part of the Scots pine stand compartment, only wood production is. Harvest is considered as a human/industrial process which is done after production. Secondly, additional data were collected on the fluxes of following compounds: SO_2 (Neirynck et al., 2011), O_3 (Neirynck et al., 2012), NO_x and Non-methane Volatile Organic

Compounds (NMVOC) (Gielen et al., 2013) (Figure 2.3). The reference flow of the Scots pine stand, is the production of 1 m^3 of stem wood under bark. The other outputs of the ecosystem, e.g. nitrate leaching and root growth, are here considered as wastes.

Data for the industrial life cycle for the production and disposal of 1 m^3 of sawn timber originated from the Ecoinvent database v2.2, Werner et al. (2007) and Doka et al. (2009). Additional information on the total LCI can be found in the supporting information, section 2.5.3. The LIM incorporated in Simapro version 7.3 (Pré Consultants bv, Amersfoort) was used to calculate the LCI of the process-based life cycle.

2.2.2.3 Life cycle impact assessment

Two LCIA methodologies were applied on the LCI using Simapro version 7.3 software. To quantify the total resource consumption the Cumulative Exergy Extracted from the Natural Environment (CEENE) indicator method (Dewulf et al., 2007) is used, which is considered as one of the two best thermodynamic resource indicators (Liao et al., 2012; Rugani et al., 2011). Herein, all resource flows are expressed in terms of exergy and summed up, leading to the total CEENE amount. This idea and the exergy calculations/values are based on the work of Szargut et al. (1988) and Valero et al. (1986). In general, resource depletion impact assessment has still some scientific gaps and needs further research (Hauschild et al., 2013), therefore we preferred to keep the resource assessment at an early stage, i.e., evaluating the quantity and quality of the consumed resources expressed in exergy. This method expresses resources in one scientifically sound metric, covering all resource types, whereas others do not (Swart et al., submitted). Few other methods include land occupation, which is relevant to acccount for in this case. We used an updated version of the CEENE method, version 2.0 (R. A. F. Alvarenga et al., 2013). In this improved method land occupation on a specific location is accounted for by the amount of potential Net Primary Production (NPP), expressed in exergy, normally produced during the time of occupation by the natural environment present on that specific geographic location, which is modelled via hte Lund–Potsdam–Jena dynamic global vegetation model (Haberl et al., 2007). Rain, sun and other natural inputs of the occupied land are indirectly accounted for in the potential NPP. Since the Scots pine stand vegetation is not the natural one, we can consider this deprivation in terms of NPP for land occupation. The CEENE characterization factor for land occupation at the exact location (defined by its coordinates) of the Scots pine stand, is 278 GJ_{ex} ha^{-1} yr^{-1} (R. A. F. Alvarenga et al., 2013).

For environmental damaging effects of land occupation and emissions, the ReCiPe method (Goedkoop et al., 2009) version 1.07 was used. ReCiPe is a recent holistic LCIA methodology which includes impact assessment methods for many different categories, of which only the emission related and land occupation were used in this case study

(Figure 2.4). The extent of impact can be assessed at an early stage of the cause-and-effect chain, called midpoint level. The impact can also be assessed as the final effect on the environment, at an endpoint level. The final damage in the endpoint approach of ReCiPe is estimated in terms of damage to human health, expressed in Disability-Adjusted Life Years (DALY), and loss of ecosystem diversity/biodiversity, expressed in total loss of species. The hierarchical (H) approach was chosen because it is based on the most common policy principles with regards to time-frame and other issues (Goedkoop et al., 2009). Improvement to LCIA methods in general will occur in the future/are under development concerning impact on biodiversity (Curran et al., 2011), this most importantly for land use and land use change (Koellner et al., 2013).

Nitric, nitrous acid and ozone in air have no characterization factors in the ReCiPe methodology version 1.07. These matters are resolved, as described in this paragraph. Tropospheric ozone is a photochemical oxidant and taken up by the forest. Its precursors NO_x and Non-Methane Volatile Organic Compounds (NMVOC) are accounted for in the ReCiPe methodology. Latter have the same characterization factor, for endpoint, 3.9E-08 DALY kg^{-1}. As an estimation, this is regarded the same for ozone itself. Nitric and nitrous acid in air pollute through terrestrial acidification and marine eutrophication. For marine eutrophication, these compounds have been replaced by an amount of nitrate nitrogen present in air. For terrestrial acidification, this was not done, though an estimation points out that these compounds are relatively low compared to others in terrestrial acidification. Consider NO_x with a terrestrial acidification factor of 0.56 kg SO_2 eq. kg^{-1} NO_x or assuming an average composition of NO_2 and NO resulting in a factor of 1.52 kg SO_2 eq. kg^{-1} N. Multiplied with 4.1 kg N of NO_2 and NO results in 6.232 kg SO_2 eq ha^{-1}. This is 9.0364 kg SO_2 eq. per m^3 sawn timber wood, only about a tenth of the impact as it is now and by consequence negligible.

2.3 Results & discussion

2.3.1 Case study

In the industrial processing 2.65 m^3 of stem softwood under bark is needed to obtain 1 m^3 of sawn timber. The remaining wood amount, 1.65 m^3, ends up as co-products. Hence, 1 m^3 stem softwood under bark and its production in the forest ecosystem are allocated to 1 m^3 of sawn timber. The LCI of latter is represented in Figure 2.3. The observed Scots pine stand productivity is very low compared to what yield tables predict (Nagy et al., 2006), discussed in supporting information section 2.5.4.2. 1.45 ha*yr land occupation is needed per m^3 stem softwood (under bark) produced and thus per m^3 sawn timber. As a consequence, all resources and emissions are relatively high. A forest ecosystem filters particulate matter through dry deposition. Here only data for capturing of nitrate and ammonium in particulates with a diameter smaller than 2.5 µm ($PM_{2.5}$) were considered, the total amount equals 33.35 kg per m^3 sawn timber (Neirynck et al., 2007) (Figure 2.3).

The carbon balance is important in this production system. 0.71 tonnes (t) C FU^{-1} is sequestered by the Scots pine stand, of which 0.17 t C is leached to the underlying soil, 0.29 t C remains in the forest (e.g. as biomass) and 0.25 t C is harvested as stem wood and processed into timber, but latter amount is emitted again during burning. The amounts of carbon needed by the technosphere are insignificant since the estimated net fossil carbon consumption is smaller in absolute value than 0.01 t C FU^{-1} (Figure 2.3). Almost double the amount of carbon present in the timber is thus sequestered during the life cycle.

Results of the impact assessment of the product's life cycle are given in Figure 2.4. Impact values are positive when there is a damaging effect. But when harmful compounds are taken up, thus remediating their impact if emitted, or a process is displaced, here the case for electricity production, the value is negative.

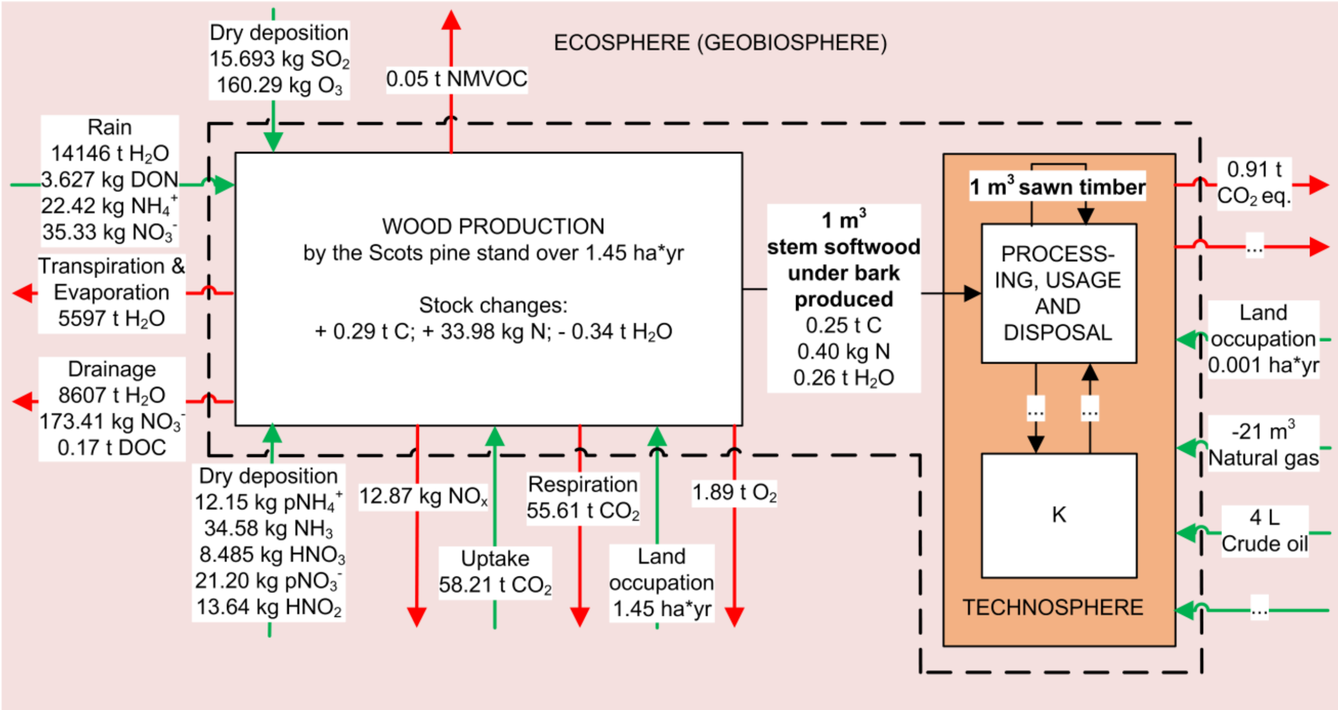


Figure 2.3. Overview of the life cycle inventory of the production chain of 1 m^3 sawn timber. In the industrial processing 2.65 m3 of stem softwood under bark is needed to obtain 1 m^3 of sawn timber. 1 m^3 stem softwood is allocated to 1 m^3 sawn timber as the residual 1.65 m^3 ends up as co-products. The industrial part comprises the complete technosphere with as foreground system the 'processing, usage and disposal', described in section 2.5.3.2 of supporting information, up to K background processes. All specified flows for the Scots pine stand and the most relevant for the technosphere are given. Negative flows to or from the technosphere are flows prevented through displacement of electricity generated during burning of wood. The net emitted amount of oxygen by the forest is estimated as the net equimolar amount of sequestered CO_2. System boundaries are in dotted lines. Resources and emissions are represented by green and red arrows, respectively. NMVOC: Non-methane volatile organic compounds. DOC: Dissolved organic carbon; DON: Dissolved organic nitrogen; p: present in particulate matter with a diameter < 2.5 µm; t: tonne.

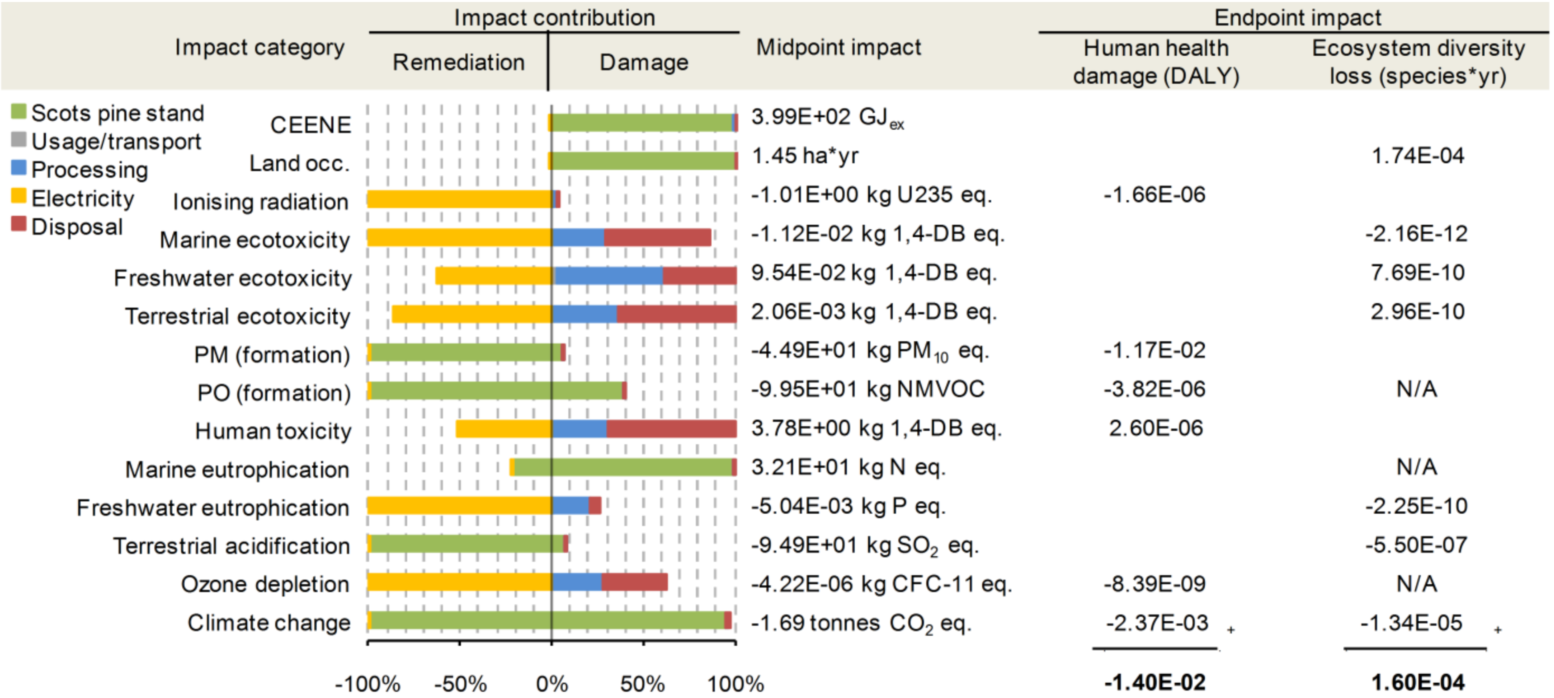


Figure 2.4. Impact assessment of the studied life cycle for 1 m^3 of sawn timber produced (Figure 2.3). The impact contributions of different processes are visualized in the graph, with damage normalized to the maximum absolute value of remediation and damage. The Cumulative Exergy Extracted from the Natural Environment (CEENE) accounts for the resource consumption. The other emission categories are from the ReCiPe methodology. The impact at midpoint and endpoint level are given. The final damage, at endpoint, is the damage done to human health, expressed in Disability-Adjusted Life Years (DALY), and/or ecosystem diversity, expressed in total loss of species. For some impact categories, no quantitative endpoint value is available although there is a link, this is marked as not available (N/A). The estimated total endpoint impacts are printed in bold. Occ.: occupation; PO: Photochemical oxidant; U236: Uranium-236 isotope; eq.: equivalents, 1,4-DB: 1,4-Dichlorobenzene, PM_{10}: Particulate Matter with a diameter smaller than 10 μm; NMVOC: Non-Methane Volatile Organic Compounds; N: nitrogen; P: phosphorus; CFC-11: Trichlorofluoromethane.

Resource usage (CEENE) is high, about 400 GJ_{ex}, equivalent to the exergy of 9.5 tonnes of fossil oil, is extracted from the natural environment per m^3 of sawn timber, and can be almost completely assigned to the land occupation by the Scots pine stand for the wood production (403 GJ_{ex}). This is slightly counteracted by the generation of electricity which prevents the extraction of 5 GJ_{ex}. There is a remediation effect on human health by the complete system since the total impact is negatively valued as -1.40E-02 DALY, approximately 5 days, FU^{-1}. The loss of ecosystem diversity is estimated with a total amount of 1.60E-04 species*yr FU^{-1}. Note that there are some impact categories for which no endpoint modeling is available for effect on ecosystem diversity (Figure 2.4).

The estimated loss in ecosystem diversity can be almost completely assigned to the occupation of land by the intensively managed forest. In the ReCiPe methodology, the species diversity of an intensively managed forest, such as the Scots pine stand, is represented by that of a broadleaf plantation. The difference between this species diversity and that of a considered natural reference system, results in the estimated loss in ecosystem diversity. The natural reference system in Europe is considered to be broadleaf forest (Goedkoop et al., 2009).

The largest share (77%) of the human health net remediation effect can be attributed to the impact category particulate matter (formation). The depostion of the particulate nitrate and ammonium share of $PM_{2.5}$ by the Scots pine stand accounts for 70%. Ammonia and sulphur dioxide, which normally coagulate with other chemicals to form such fine particles, were also deposited in the forest ecosystem and contribute, respectively, with 23% and 7%, while the emission of nitrogen oxides, also precursors, counteracts this by 6%. *Pinus sylvestris* is in fact known for its efficiency in capturing PM (Sæbø et al., 2012). This remediation effect is even underestimated because only ammonium and nitrate content of $PM_{2.5}$ is considered. A rough estimation, accounting for total PM_{10} and resuspension, leads to a removal of 110 kg PM_{10} by the forest, this suggests a 2.7 times higher impact reduction for this category and higher gain in DALY FU^{-1} (see supporting information, section 2.5.4). Besides this deposition of $PM_{2.5}$ there is a negligible small impact in this category by emissions of the technosphere, only 1% of the negative impact value, mostly from wood disposal.

The impact of the production system on climate is less important since it only accounts for 17% of the quantified endpoint impact on human health and remediates ecosystem diversity loss at 8% of the total. There is a high emission of CO_2 by the forest ecosystem, but its uptake is even higher. In total, 2.60 t CO_2 FU^{-1} is sequestered by the Scots pine stand. The greenhouse gas emissions by the technosphere amount to 0.91 t CO_2 equivalents (eq.) FU^{-1}, almost all from burning the wood. This leads to an impact value of -1.69 t CO_2 eq. per m^3 of sawn timber for the total life cycle. If the Scots pine stand would not be included in the system boundaries, only the CO_2 sequestered as the carbon present in the biomass product is considered according to the greenhouse gas protocol ("Greenhouse Gas Protocol," 2013). This carbon amount is almost all released again during burning, resulting in a total impact of -0.01 t CO_2 eq. FU^{-1}, remarkably less. However by doing so, the additional carbon sequestration which inevitably occurs in the ecosystem during production of the biomass product (e.g. root growth in case of a stem wood product) is not taken into account. On the other hand, the fates of these other forest carbon flows (litter, grown biomass and leachate) and their possible rerelease of greenhouse gases are not considered here.

A discussion on the impact of the other categories can be found in the supporting information, section 2.5.4. If a conventional LCA would have been performed (without Scots pine stand) findings would differ considerably, this is discussed in the supporting information, section 2.5.4.1.

The impact of the Scots pine stand in the production chain is the most important: it accounts for quasi all of resource usage, the final remediation effect on human health and estimated biodiversity loss through land occupation. Even for normal productivity compared to the low productivity of the studied Scots pine stand, the forest ecosystem would thus still play an essential role (supporting information, section 2.5.4.2). This showcases the potential importance of including ecosystem processes in product life cycles for sustainability assessments. As such, a better view on the overall impact of these life cycles on the environment, is obtained (Figure 2.2).

2.3.2 Framework for LCA on techno-ecological systems

In this chapter, a framework is introduced to conduct a LCA on an integrated Techno-Ecological System (TES). A mathematical model based on Linear Inverse Modelling (LIM) is proposed to calculate the Life Cycle Inventory (LCI). In this framework, no distinction is made between human/industrial and ecosystem compartments. New in our framework is that either a process based, input-output or hybrid approach can be conducted and this while accounting for both resources and emissions of the human/industrial and ecosystem compartments of the TES (Suh and Huppes, 2005), and allowing interactions from human/industrial to ecosystem compartments and vice versa. This is not so for the previous models of Rugani and Benetto (2012) and Hau and Bakshi (2004). There are some other additional assets in our framework. Stock changes of compartments are addressed as inputs or outputs if they are depletions or increments, respectively. And taken up amounts of harmful compounds, e.g. CO_2 sequestration, are accounted for by considering these as negatively valued amounts which leads to negative impact values. After all, by taking up such compounds their harmful effect is prevented, what should be accounted for. Methods for accounting for the uptake of harmful compounds have already been worked out specifically for uptake of carbon dioxide. Intergovernmental Panel on Climate Change (IPCC) 2006 guidelines elaborate on the uptake of carbon dioxide by land and on the effect of change in land use on uptake (IPCC, 2006). The greenhouse gas protocol accounts for the uptake of CO_2 in biomass products ("Greenhouse Gas Protocol," 2013). However in our framework a general approach is presented and this for all harmful compounds.

Still, methodological standards are needed that define which ecosystems or parts of them to account for and which not. For example, according to the methodology of Alvarenga et al.(2013) intensively managed ecosystems, such as the forest ecosystem of the case study, should be included in the system boundaries of a LCA study. Our framework could then be used to assess the environmental impact of a TES with selected included ecosystem compartments. In the emergy framework and related works, by definition the boundaries are fixed to that of the geobiosphere/ecosphere while our approach allows choosing where to draw them. Boundaries could optionally be expanded and fixed to that of the biosphere. Our mathematical model even allows one to include non-ecosystem compartments from the ecosphere, e.g. rain production, going beyond TESs. System boundaries are then expanded to that of the geobiosphere/ecosphere as done in the emergy framework illustrated in case B of Figure 2.1. For all these options, practically, databases will need to be developed which contain data for the different (ecosystem) compartments. As an example, for the technosphere, diverse databases already exist, of which Ecoinvent (Swiss Centre for Life Cycle Inventories, 2010) is the most popular.

As for the mathematical model proposed in this framework to calculate the LCI of a LCA, a LIM is one of the most basic models to simulate a response of a system. Better mathematical models should in the future be used since the assumption of linearity in LIM is sometimes far from correct for real systems, especially for ecosystems (Suh, 2005). However by subdividing the studied time window in smaller intervals for which the assumption of linearity are more valid, the results will be more representative, as is shown by Collinge et al. (2013). The time window in total could in fact also be chosen so that a linear approximation can be obtained. For intensively managed ecosystems, this is for example a full harvest cycle. However a harvest cycle is not present in non-managed ecosystems. In the case study, the time window of the forest ecosystem is two years which is in fact narrow to have representative results. Only empirical data were collected for the case study and there was only data for that time period. Data collection in the field is after all in general a time demanding and costly undertaking. A solution for this is to use output data from ecosystem growth models, e.g. such as ANAFORE (Deckmyn et al., 2011, 2008) for a forest ecosystem, which provide output data on larger, more representative time scales.

Using this framework, specific ecosystems may be studied in particular to quantify their emissions and resources as is done with the human/industrial compartments. All the available LCIA methods can then still be applied on the LCI. The environmental impact is still the impact on the ecosphere without the technosphere, even though the ecosystem compartments of a TES belong to the ecosphere. Additionally, as mentioned in this chapter's introduction (section 2.1), LCIA methods could still be improved to assess for a more site-specific impact, e.g. the site-specific land occupation impact assessment in the CEENE methodology applied in the case study (R. A. F. Alvarenga et al., 2013). Next to that, ecosystem compartments might demand other inputs or have other outputs which current LCIA do not account for, e.g. nitrogen input from the atmosphere, production of oxygen, emigrating organisms (which might damage other ecosystems) and so on. In this sense, new LCIA methods should be developed to account for these impacts, besides assessing impacts on specific ecosystems. Both improvements of LCIA methods would allow one to better evaluate the impact of a TES. Next to only considering the environmental damage of emissions through attributing certain environmental impacts to them, the amount of resources needed to degrade them may also be considered. This might be done via an avoidance method (Sciubba, 2004; Szargut et al., 1988) or by including the specific degrading processes, human/industrial or ecosystem ones.

An aspect which is not dealt with here, is the aspect of time in LCA. Take for example how to account for the temporary aspect of storage of harmful compounds, not taken into account in our study, which is presumably relevant for temporary CO_2 storage in the sawn timber products in the case study. Research is done and ongoing to make LCA methodology time-specific. Yet, for now this is a difficult issue in the field of LCA and mostly not considered. A lot of questions are present on this matter, with temporary carbon storage being an important matter of debate and research (Brandão and Levasseur, 2011; Cherubini et al., 2011; Guest et al., 2013), and there is a lack of consensus. The most promising framework, in our opinion, is that of Collinge et al. (2013). If the issues are resolved, we advice to implement this time aspect in our framework.

Ecosystems provide numerous services which are beneficial for mankind, see section 1.3 (pg. 6). It is by some desired to take up all these services in sustainability assessments. An overview of methods and their accounting for ecosystem services is given by Zhang et al. (2010b). Using LCA, one mainly accounts for the provisioning services by considering the amount of resources, e.g. using CEENE as impact methodology. Recently, damage to ecosystem services is proposed as another endpoint besides loss of ecosystem diversity, the latter used in ReCiPe applied in the case study in order to assess the damage to ecosystem quality in LCA (Koellner and Geyer, 2013). This has already been worked out for land use impact on some services, besides biotic production (Saad et al., 2013). Additional in our framework, by accounting for uptake of harmful compounds, we also account for a regulating service. However our framework also accounts for this service by human/industrial compartments, but in the case study this was negligible. A vital remark on this matter is that human/industrial compartments might after all also have similar other beneficial services: cycling nutrients, uptake of CO_2, et cetera. While often only the damaging effect of human/industrial compartments is emphasized, these services of human/industrial compartments are often inconsistently forgotten. In a TES no differentiation is made between the human/industrial and ecological part, and thus also no differentiation should be made between their impacts and/or services. In general, further research is needed to better account for the environmental impact, damaging as well as remediating, of systems, especially techno-ecological systems.

2.4 Acknowledgements

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2.5 Supporting information

The supporting information of this chapter gives additional information on the mathematical model section 2.5.1), case study system description (section 2.5.2), life cycle inventory (section 2.5.3) and discussion of its results (section 2.5.4).

2.5.1 Mathematical model

As a mathematical model the existing Linear Inverse Modelling (LIM) methodology is used, including its different approaches within Life Cycle Assessment (LCA): process, input-output (IO) and hybrid. A general methodology is described which is valid for the different existing LIM approaches. A life cycle is always first subdivided into different compartments. In conventional LCAs these were only human/industrial compartments (compartments 1 to N in Figure 2.2, case A). Now, ecosystem compartments are included. This is, for example, one compartment A in Figure 2.2 case C, but there can be more. No differentiation is further made between human/industrial and ecosystem compartments in the mathematical model. The compartments are interlinked through exchange of energy and mass in the form of products, expressed in certain units, e.g. money or mass content (these are the arrows between the compartments in Figure 2.2). These compartments deliver different products to the rest of the system and the environment. For each product output i, by considering it as a reference flow, a set of other product amounts directly needed for its production is quantified. When different products are generated by one compartment or by multiple compartments, allocation or system expansion is needed (for more information we refer to Suh et al.(2010)). A square matrix T is then constructed (common in process based) or calculated (the Leontief matrix (Leontief, 1936) in the IO-based approach) which contains the exchanges; each ij-th element of matrix T is the negative amount of product output i directly needed as an input for the production of product output j, per amount of product output j. The product output of the reference flow is by consequence positively valued in matrix T. This matrix represents the direct relationships inside the system boundaries between the different outputs of compartments indifferent if they are from human/industrial or natural origin. Consider vector x of which each i-th element is the total amount of product output i produced. Also consider vector y of which each i-th element is only the produced amount of product output i which leaves the system. Vector y represents the final demand of different product quantities of the system. Assuming linear relationships between the product flows of the compartments, equation 1 can be constructed:

$$Tx = y \quad (1.1)$$

This assumption of linearity is important since it means that if a product output needs to be generated with a factor s greater, all the product flows needed for its production will increase with the same factor s. Equation 2 is obtained by reformulating equation 1.

$$x = (T)^{-1}y \quad (1.2)$$

Using equation 2, a vector x can be calculated, which is the amount of product outputs of the compartments, for a given demand, vector y, and a given system, matrix T. The inverse matrix of the T matrix, $(T)^{-1}$, contains the indirect and direct negative amount of product output i (if output positive) needed for the production of product output j, per amount of product output j. Matrix T only contains the direct amounts.

Consider a matrix B of which the ij-th element is the amount of emission/resource i of product output j per product output of j. Assuming linearity, multiplying this matrix with vector x, results in a vector c of which the i-th element is the emission/resource i quantity for the complete studied system (equation 3).

$$c = Bx = B(T)^{-1}y \quad (1.3)$$

As such the LCI, containing all resource and emission quantities of the entire system, can be calculated based on assumptions of linearity. For more information, on the difference between the process, input-output or hybrid based approach, we refer to the work of Suh and Huppes (2005). In essence the mathematical model stays the same in our new approach but now there is inclusion of ecosystem compartments.

2.5.2 Additional information on system description

The industrial life cycle is the complete collection of processes in Ecoinvent (Swiss Centre for Life Cycle Inventories, St-Gallen), representing the technosphere. The foreground system is a collection of processes from Ecoinvent which comprise the processes needed for harnessing the forest, harvesting and processing of wood to sawn timber, transport during usage of the timber and the disposal of wood through burning with cogeneration of heat and electricity (Figure 2.5). The processes for harnessing the forest, harvesting and processing of wood are representative for Central Europe (Werner et al., 2007). Herein the saw mill is considered to be at 40 km driving distance from the forest.

The sawn timber wood will mainly be used in construction of buildings, furniture, et cetera. After disposal, it may be recycled several times (optionally as chip board; for paper only wood waste from processing in saw mills or veneer industries is considered) in a so called cascaded use. We assume no degradation or alteration of the composition, e.g. by impregnation, during the use phase. Eventually, one will have to dispose of the wood. In Belgium it is collected and burned, latter with a possible industrial cogeneration of heat and electricity in the best case (land filling of wood is prohibited in Belgium). Using ecoinvent processes, these use and end-of-life phases were considered. For the use phase, only transportation needs to be considered, this will be estimated as the same amount for collection of used wood (about 0.01 t*km per kg wet waste wood (Doka, 2009)). Collection and burning of the waste Scots pine wood are based on the ecoinvent process 'Disposal, building, waste wood, untreated, to final disposal', with as functional unit 1 kg of wet waste wood.

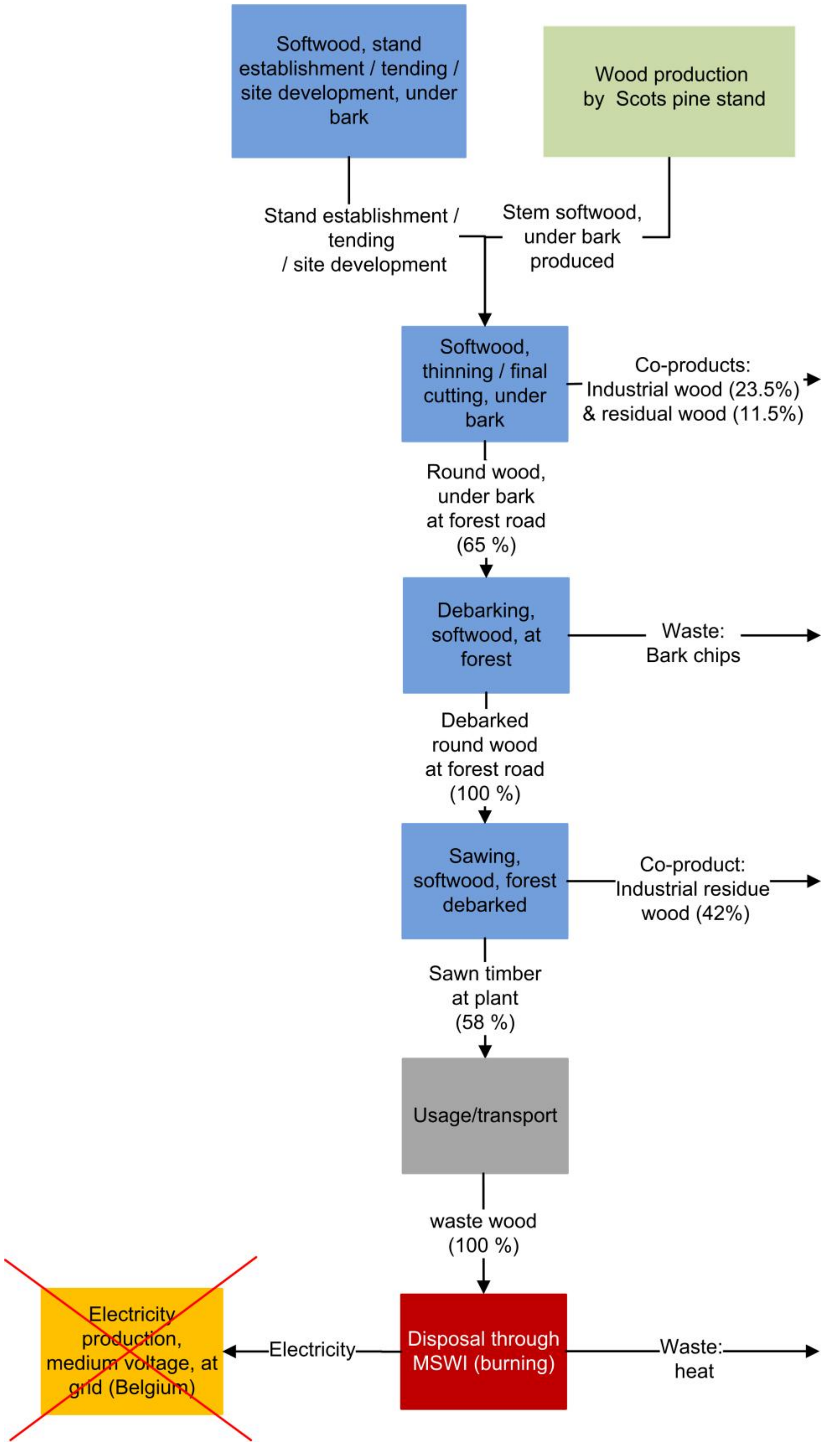

Figure 2.5. Overview of the human/industrial foreground system 'processing, usage and disposal' (Figure 1). The industrial processes originate from the Ecoinvent database (Swiss Centre for Life Cycle Inventories, 2010). The wood production by the Scots pine stand is in green and does not belong to the industrial foreground system. The colours for the human/industrial processes are the same as for Figure 3. Volumetric allocation factors (Werner et al., 2007) are given between brackets. Bark chips are considered as a waste product leading to 100% allocation to the 'debarked, round wood at forest road'. Conventional electricity produced at medium voltage at the grid in Belgium is displaced, this is visualized by a cross over this process. MSWI: Municipal Solid Waste Incineration.

The burning of the wood occurs through municipal solid waste incineration, based on the average data of Switzerland (Doka, 2009). Besides burning and cogeneration, this process also includes the subprocesses treatment of flue gasses (with additional treatment of wastewater streams) and land fill of residuals (Figure 2.6).

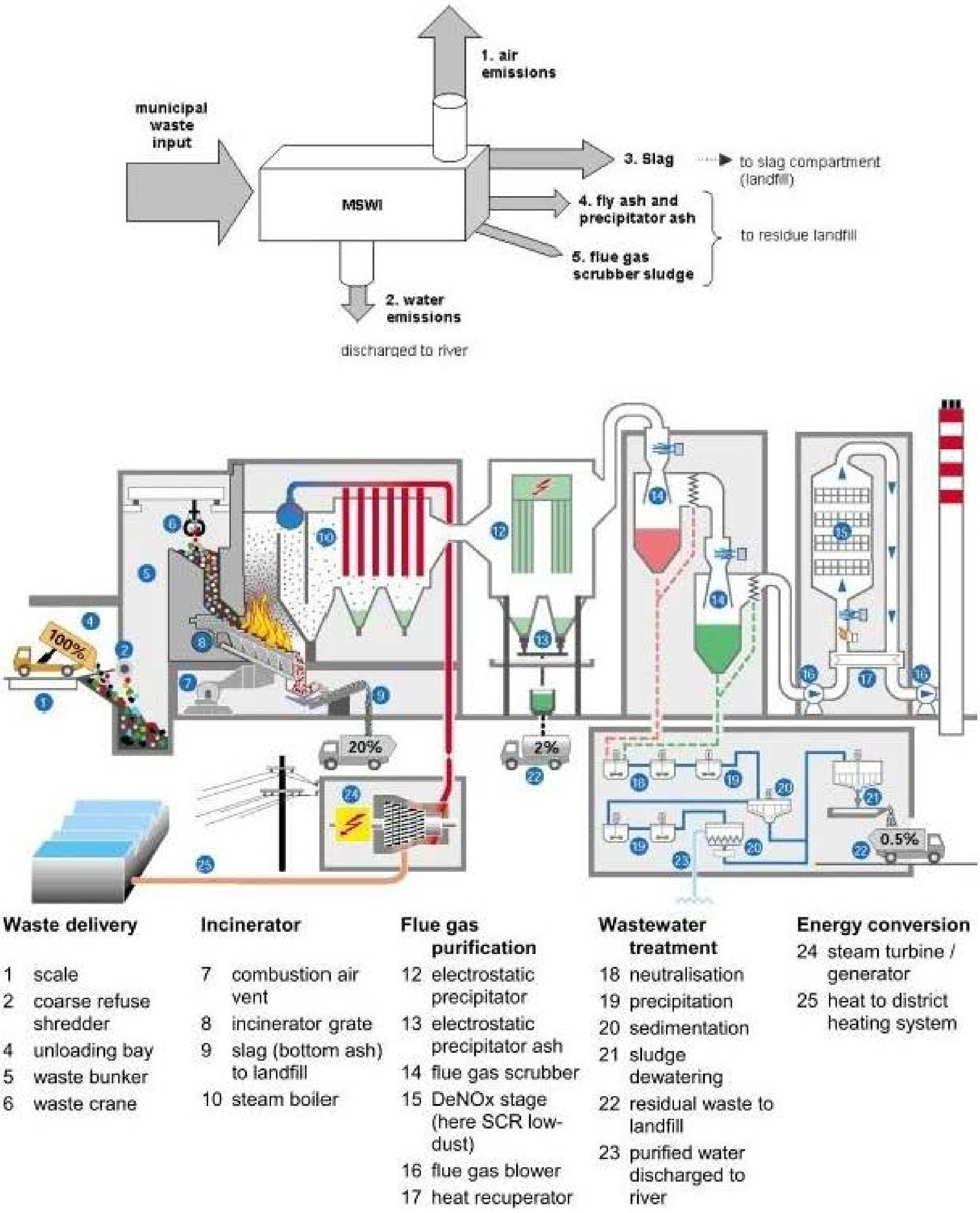

Figure 2.6. Municipal Solid Waste Incineration (MSWI) (Schemes retrieved from Doka (2009)). The overall flows considered in the ecoinvent process are depicted in the above scheme. The below scheme is of a typical Swiss MSWI, specifically that of MSWI Buchs AG with a deNOx installation (selective catalytic reduction low-dust) after the wet scrubber.

2.5.3 Additional information on life cycle inventory

2.5.3.1 Scots pine stand

Most data from this stand date from 2001 to 2002. These data were completed with data from the literature. For specific data sources we refer to Schaubroeck et al. (2012), Chapter 5 (pg. 133). Different from Schaubroeck et al. (2012) no harvest was considered in the two-year study period to only account for the production of wood in the forest ecosystem (harvest is considered to be a human/industrial process which will occur after the study period). This is acceptable since in reality only 8 trees out of 377 trees were harvested per hectare and this at the end of the considered time frame: november 2002 of the period 2001-2002 (Yuste et al., 2005). This makes the effect of harvest negligibly small on the other flow values. Wood harvest and slash, being plant residues after harvest (including bark), were by consequence also not considered. Their amounts were attributed to the plant increment. Secondly, additional data was collected on the fluxes of following compounds: SO_2 (Neirynck et al., 2011), O_3 (Neirynck et al., 2012), nitrogen oxides and Non-methane Volatile Organic Compounds (NMVOC) (Gielen et al., 2013) (Figure 2.3, pg. 28). Deposition of SO_2 and O_3 were obtained from Neirynck et al.(Neirynck et al., 2011) and through personal communication with dr. ir. Johan Neirynck related to another study (Neirynck et al., 2012), respectively. Nitrogen oxides were converted from a nitrogen amount to an estimated total mass amount, by assuming a molecular weight equal to that of NO_2. For an estimation of the emission of Non-methane Volatile Organic Compounds (NMVOC), we used the emission value of carbon present in isoprene and monoterpenes, which Gielen et al.(2013) derived using the work of Schurgers et al. (2009). This value was converted to a total mass amount using the molecular weight of isoprene.

The functional unit of the Scots pine stand ecosystem, is the production of 1 m^3 of stem wood under bark. To calculate the amount of stem wood under bark produced per hectare yearly, the original amount of stem increment 0.4 ton dry matter (DM) biomass (BM) ha^{-1} yr^{-1} (Yuste et al., 2005) was divided by its density 0.502 ton dry biomass (BM) m^{-3} (Yuste et al., 2005) and multiplied with a factor (1.135) in which 0.135 (range: 0.10-0.17) is the average fraction of bark volume per stem volume for pine (Werner et al., 2007). The other outputs of the ecosystem, e.g. nitrate leaching and root growth (increment), are considered as wastes.

2.5.3.2 Human/industrial life cycle

Data originated from Ecoinvent v2.2 (Swiss Centre for Life Cycle Inventories, 2010) and Werner et al. (2007). Different co-products (industrial and residual wood) are created during the processing of stem softwood into sawn timber. For the processing chain the allocation methodology was adjusted from an original economic one in the Ecoinvent v2.2 (Swiss Centre for Life Cycle Inventories, 2010) database to a volumetric one, thus based on a physical property which is recommended by the International Organization for Standardization (ISO) (2006b) standards rather than economic allocation. This volumetric allocation can be seen as a mass/energetic/exergetic allocation, since composition does almost not alter in the debarked wood. As an assumption, bark is considered as a waste product which remains in the forest, therefore allocation of the debarking process is 100% to the debarked round wood at forest road. It must be noted that bark can be valorized in different manners but this is not dealt with in the current study. Natural drying at the saw mill is assumed, leading up to a moisture content of about 17%.

For the respective byproducts we used allocation instead of system boundary expansion, since no alternative pathway is available for a quasi identical product. System expansion is however often interesting for waste valorization. That is why it is applied for the electricity produced during burning of wood.

As the emissions and resources of this process are dependent on the type of waste burned, a calculation tool is provided to calculate these based on the composition of the input waste stream (Swiss Centre for Life Cycle Inventories, 2010). We used this tool to generate outcomes as specifically as possible for the Scots pine wood. To do so, the composition of the wet wood is needed. Ecoinvent already provides composition values for average untreated wood with a moisture content of 17.4% (Swiss Centre for Life Cycle Inventories, 2010). Carbon and nitrogen amounts present were displaced by these of the Scots pine wood of our study. The Phyllis database ("Phyllis database," 2013) was also used to retrieve average relative amounts of Scots pine wood for oxygen, hydrogen, sulfur, chlorine, potassium and sodium. The higher heating value was estimated using the method proposed by Sheng and Azevedo (2005). The lower heating value for wet biomass was derived from this, by not accounting for the latent heat of vaporization for water (2.442 MJ kg^{-1}). Table 2.1 summarizes the input data needed for the calculation tool.

Table 2.1. Input data, mostly composition data, of the wet waste Scots pine wood needed for the calculation tool (values are per kg wet wood). Carbon and nitrogen data are retrieved from Schaubroeck et al. (2012) and Neirynck et al. (2008), respectively. Other data specific for Scots pine wood were retrieved from the Phyllis database ("Phyllis database," 2013), denoted as phyllis. The rest of the data is that for untreated wet wood already present in the ecoinvent database (Swiss Centre for Life Cycle Inventories, 2010), denoted by ecoinvent. The higher heating value was estimated using the method proposed by Sheng and Azevedo (2005). The lower heating value for wet biomass was derived from this, by not accounting for the latent heat of vaporization for water (2.442 MJ kg^{-1}).

Parameter	Unit	Value	Source
Higher Heating Value (HHV)	MJ kg^{-1}	1.67E+01	Calculated
Lower Heating Value (LHV)	MJ kg^{-1}	1.50E+01	Calculated
Water content	kg kg^{-1}	1.74E-01	ecoinvent
Oxygen (without O from H_2O)	kg kg^{-1}	2.91E-01	phyllis
Hydrogen (without H from H_2O)	kg kg^{-1}	4.38E-02	phyllis
Carbon	kg kg^{-1}	4.26E-01	Assumption; half of dry matter
Sulfur	kg kg^{-1}	7.01E-04	phyllis
Nitrogen	kg kg^{-1}	6.81E-02	Neirynck et al. (2008)
Phosphorus	kg kg^{-1}	1.09E-04	ecoinvent
Boron	kg kg^{-1}	2.11E-06	ecoinvent
Chlorine	kg kg^{-1}	4.23E-04	Phyllis
Bromium	kg kg^{-1}	0.00E+00	ecoinvent
Fluorine	kg kg^{-1}	2.11E-05	ecoinvent
Iodine	kg kg^{-1}	0.00E+00	ecoinvent
Silver	kg kg^{-1}	0.00E+00	ecoinvent
Arsenic	kg kg^{-1}	4.22E-07	ecoinvent
Barium	kg kg^{-1}	0.00E+00	ecoinvent
Cadmium	kg kg^{-1}	2.01E-07	ecoinvent
Cobalt	kg kg^{-1}	8.68E-08	ecoinvent
Chromium	kg kg^{-1}	6.58E-07	phyllis
Copper	kg kg^{-1}	2.15E-04	phyllis
Mercury	kg kg^{-1}	3.20E-07	ecoinvent
Manganese	kg kg^{-1}	5.31E-05	ecoinvent
Molybdenum	kg kg^{-1}	8.30E-07	ecoinvent
Nickel	kg kg^{-1}	5.56E-07	ecoinvent
Lead	kg kg^{-1}	2.79E-05	ecoinvent
Antimony	kg kg^{-1}	0.00E+00	ecoinvent
Selenium	kg kg^{-1}	0.00E+00	ecoinvent
Tin	kg kg^{-1}	0.00E+00	ecoinvent
Vanadium	kg kg^{-1}	0.00E+00	ecoinvent
Zinc	kg kg^{-1}	1.78E-05	ecoinvent
Beryllium	kg kg^{-1}	0.00E+00	ecoinvent
Scandium	kg kg^{-1}	0.00E+00	ecoinvent
Strontium	kg kg^{-1}	0.00E+00	ecoinvent

Titanium	kg kg^{-1}	0.00E+00	ecoinvent
Thallium	kg kg^{-1}	0.00E+00	ecoinvent
Tungsten	kg kg^{-1}	0.00E+00	ecoinvent
Silicon	kg kg^{-1}	0.00E+00	ecoinvent
Iron	kg kg^{-1}	1.55E-05	ecoinvent
Calcium	kg kg^{-1}	1.31E-04	ecoinvent
Aluminium	kg kg^{-1}	6.33E-06	ecoinvent
Potassium	kg kg^{-1}	6.94E-04	phyllis
Magnesium	kg kg^{-1}	1.98E-04	ecoinvent
Sodium	kg kg^{-1}	2.97E-05	phyllis
Share of iron in waste that is metallic/recyclable	%	0	ecoinvent
Share of carbon in waste that is biogenic	%	0	ecoinvent
Degradability of waste in a municipal landfill within 100 years	%	1.5	ecoinvent

According to the tool 0.40 kWh electricity (of which 0.144 kWh consumed by the Municipal Solid Waste Incineration (MSWI) is already substracted) and 3 MJ heat is produced per kg wet Scots pine wood. An electricity producing efficiency of 13% is hereby assumed. However, higher efficiencies have been noted in literature for cogeneration of biomass by the European joint research center of the European Commission (Vatopoulos et al., 2012). An average efficiency of 21% can be derived for the cogeneration out of biomass. Latter efficiency was used resulting in a production of 0.74 kWh and 1.75 MJ heat per kg wet Scots pine wood. A total of 0.62 tonnes of wet Scots pine wood needs to be burned. This results in a production of about 460 kWh electricity and 1081 MJ heat. The produced electricity displaces electricity from the Belgian grid (39% from fossil fuel combustion, 54% nuclear energy, 3% renewable and others) and the impact to conventionally produce this electricity amount is subtracted from the total impact. Concerning leftover heat, Niphuis (2013) estimated an efficiency of 31% using the sewer grid as transportation means. This is only 20% of the produced energy and will therefore be neglected. Above that, infrastructure for heat transport should also be accounted for. Next to that, heat networks are not common yet.

The different abatement technologies of the MSWI plant reduce the amount of air pollutants considerably. Results would be very different if no air purification technologies were present. An important example is nitrogen oxides which were reduced by 69% in weight by the average Swiss MSWI plant. Next to that an electrostatic precipitator removes almost all particulate matter, leading to an emission of only 6 mg PM_{10} per kg wet waste, thus only 3.8E-03 kg per m^3 sawn timber wood, compared to an estimated uptake of 110 kg PM_{10} FU^{-1} by the forest.

2.5.4 Additional discussion of case study results

The specific midpoint and endpoint impact values are again given in Table 2.3 and Table 2.4, this for the Scots pine stand, technosphere processes and in total. The total endpoint impact on human health and ecosystems diversity for the Scots pine stand, the technosphere and the complete life cycle are shown in Table 2.2. We must first note that the impact of land occupation on the site of the Scots pine stand is considered, however the terrestrial acidification, its damage to biodiversity and that by (precursors of) photochemical oxidants on the Scots pine stand are not directly considered. For photochemical oxidant formation, no endpoint modelling for damage to species diversity is available so this cannot be considered at that stage. Next to that, terrestrial acidification is indirectly accounted for through a lower productivity (see section 2.5.4.2). Above that, when considering the impact reduced by taking up pollutants responsible for terrestrial acidification, their impact on diversity will be negligible compared to the intensive land occupation. Alternatively, one could not consider uptake of the respective compounds in these categories.

The largest share (77%) of the human health net remediation effect can be attributed to the impact category particulate matter (formation). The depostion of the particulate nitrate and ammonium share of $PM_{2.5}$ by the Scots pine stand accounts for 70%. This particular remediation effect is even underestimated because only ammonium and nitrate content of $PM_{2.5}$ is considered. A study by the Flemish government (Vlaamse milieumaatschappij, 2011) estimated that their share of the total $PM_{2.5}$ is 33%, this about 35 km away of the forest for the years 2009-2011. The same study also quantified that $PM_{2.5}$ accounted for 73% of the total PM_{10}, at a distance of about 6 km of the forest. Next to that, resuspension of PM is not accounted for which can lead to a reduction in removal of magnitude 50%, but we estimate this to be rather low (magnitude of 20%) due to high rain fall and relatively low average wind speed (4 m s^{-1} at 10 m height) at the region of the Scots pine stand (Nowak et al., 2013). A rough estimation, accounting for total PM_{10} and resuspension, leads to an uptake of 110 kg PM_{10} by the forest, this suggests a 2.7 times higher impact reduction for this category and higher gain in DALY FU^{-1}.

Table 2.2. Endpoint Impact assessment of the emissions and land occupation of the studied life cycle for 1 m^3 of sawn timber (Figure 2.3). The final damage, at endpoint, is the damage done to human health, expressed in Disability-Adjusted Life Years (DALY), and/or ecosystem diversity, expressed in total loss of species*yr.

Area of protection	Unit	Scots pine stand	Processing	Usage /transport	Disposal	Electricity	Total
Human health	DALY	-1.54E-02	6.10E-05	2.52E-06	1.55E-03	-2.61E-04	-1.40E-02
Ecosystem diversity	Species *yr	1.53E-04	4.68E-07	1.06E-08	8.21E-06	-1.30E-06	1.60E-04

Table 2.3. Midpoint impact the studied life cycle for 1 m^3 of sawn timber and its different processes (Figure 2.3). U236: Uranium-236 isotope; eq.: equivalents, 1,4-DB: 1,4-Dichlorobenzene, PM_{10}: Particulate Matter with a diameter smaller than 10 µm; NMVOC: Non-Methane Volatile Organic Compounds; N: nitrogen; P: phosphorus; CFC-11: Trichlorofluoromethane.

Impact category	Unit	Scots pine stand	Processing	Usage/transport	Disposal	Electricity	Total
Climate change	kg CO_2 eq.	-2.60E+03	2.85E+01	1.20E+00	1.03E+03	-1.56E+02	-1.69E+03
Ozone depletion	kg CFC-11 eq	0.00E+00	2.96E-06	1.94E-07	3.80E-06	-1.12E-05	-4.22E-06
Terrestrial acidification	kg SO_2 eq	-9.56E+01	1.68E-01	6.94E-03	1.01E+00	-4.33E-01	-9.49E+01
Freshwater eutrophication	kg P eq	0.00E+00	1.42E-03	1.63E-05	3.87E-04	-6.86E-03	-5.04E-03
Marine eutrophication	kg N eq	3.19E+01	1.15E-02	4.02E-04	1.89E-01	-1.22E-02	3.21E+01
Human toxicity	kg 1,4-DB eq	0.00E+00	2.36E+00	5.40E-02	5.37E+00	-4.01E+00	3.78E+00
Photochemical oxidant formation	kg NMVOC	-9.95E+01	3.83E-01	1.21E-02	1.63E+00	-3.19E-01	-9.78E+01
Particulate matter formation	kg PM_{10} eq	-4.52E+01	7.44E-02	3.09E-03	3.76E-01	-1.45E-01	-4.49E+01
Terrestrial ecotoxicity	kg 1,4-DB eq	0.00E+00	5.30E-03	1.47E-04	9.83E-03	-1.32E-02	2.06E-03
Freshwater ecotoxicity	kg 1,4-DB eq	0.00E+00	1.54E-02	6.10E-04	1.00E-02	-1.65E-02	9.54E-03
Marine ecotoxicity	kg 1,4-DB eq	0.00E+00	2.29E-02	9.49E-04	4.49E-02	-8.00E-02	-1.12E-02
Ionising radiation	kg U235 eq	0.00E+00	3.32E+00	3.72E-02	6.45E-01	-1.05E+02	-1.01E+02
Agricultural land occupation	ha*yr	1.45E+04	1.02E+00	4.26E-03	1.12E-01	-2.00E+00	1.45E+04
Urban land occupation	ha*yr	0.00E+00	1.08E+01	1.28E-02	1.30E-01	-5.26E-01	1.04E+01
Total land occupation	ha*yr	1.45E+04	1.18E+01	1.71E-02	2.42E-01	-2.52E+00	1.45E+04
CEENE	MJ_{ex}	4.03E+05	8.32E+02	2.12E+01	5.30E+02	-5.12E+03	3.99E+05

Table 2.4. Endpoint impact of the studied life cycle for 1 m^3 of sawn timber (Figure 2.3) and its different processes, quantified using RECIPE. The final damage, at endpoint, is the damage done to human health, expressed in Disability-Adjusted Life Years (DALY), and/or ecosystem diversity, expressed in total loss of species*yr. For marine eutrophication no quantitative endpoint value is available although there is a link.

Impact category	Unit	Scots pine stand	Processing	Usage/transport	Disposal	Electricity	Total
Climate change	DALY	-3.64E-03	4.00E-05	1.68E-06	1.45E-03	-2.18E-04	-2.37E-03
	Species*yr	-2.06E-05	2.26E-07	9.49E-09	8.19E-06	-1.24E-06	-1.34E-05
Ozone depletion	DALY	0.00E+00	7.69E-09	5.12E-10	9.57E-09	-2.62E-08	-8.39E-09
Terrestrial acidification	Species*yr	-5.54E-07	9.72E-10	4.03E-11	5.80E-09	-2.51E-09	-5.50E-07
Freshwater eutrophication	Species*yr	0.00E+00	6.34E-11	7.26E-13	1.65E-11	-3.06E-10	-2.25E-10
Human toxicity	DALY	0.00E+00	1.66E-06	3.78E-08	3.71E-06	-2.81E-06	2.60E-06
Photochemical oxidant formation	DALY	-3.88E-06	1.50E-08	4.72E-10	6.32E-08	-1.24E-08	-3.82E-06
Particulate matter formation	DALY	-1.18E-02	1.94E-05	8.04E-07	9.69E-05	-3.77E-05	-1.17E-02
Terrestrial ecotoxicity	Species*yr	0.00E+00	7.99E-10	2.22E-11	1.46E-09	-1.98E-09	2.96E-10
Freshwater ecotoxicity	Species*yr	0.00E+00	1.32E-11	5.25E-13	8.13E-12	-1.42E-11	7.69E-12
Marine ecotoxicity	Species*yr	0.00E+00	4.04E-12	1.67E-13	7.74E-12	-1.41E-11	-2.16E-12
Ionising radiation	DALY	0.00E+00	5.44E-08	6.10E-10	9.96E-09	-1.73E-06	-1.66E-06
Agricultural land occupation	Species*yr	1.74E-04	1.66E-08	5.25E-11	1.31E-09	-2.40E-08	1.74E-04
Urban land occupation	Species*yr	0.00E+00	2.23E-07	2.65E-10	2.42E-09	-1.09E-08	2.15E-07
Total land occupation	Species*yr	1.74E-04	2.40E-07	3.17E-10	3.73E-09	-3.49E-08	1.74E-04

Some other impact categories are here discussed. The emission of nitrate to the ground water in the Scots pine stand is high due to a high atmospheric nitrogen load (Neirynck et al., 2008). Nitrogen in water bodies lead to (marine) eutrophication. The endpoint impact is not modeled in ReCiPe and it is thus difficult to assess its importance compared to other categories in our case study. The eutrophication is mainly induced by nitrate leaching in our specific case. The uptake by the system of ammonia, ammonium, nitrate via wet and dry deposition, remediate the damaging impact with about 21%. The contribution of the technosphere is negligible (0.47% of negative). Photochemical oxidant formation, with an endpoint impact of -3.82E-06 DALY and unknown species diversity loss, is due to the uptake of ozone by the forest. This is partially remediated by emissions of the Scots pine stand (38%), of which 21% by nitrogen oxides and 79% by NMVOC. Due to enhanced mixing of air layers, ozone is entrained from above the canopy into the trunk/canopy space where it reacts with stored NO (Neirynck et al., 2012).

The Scots pine stand remediates terrestrial acidification, making technosphere impacts negligible, this mainly because of an uptake of ammonia. The emission of nitrogen oxides leads only to a damage effect of 7% compared to the remediation. The rest is induced by emissions of technosphere processes based on the known flux data. Note that if other forest fluxes relevant to these categories would be known, results would differ. Besides that, the impact in these categories is here discussed as such. Human toxicity, marine, terrestrial and freshwater ecotoxicity are mainly due to leaching of metals and carbohydrates and of minor importance. The displaced electricity production remediates considerable shares of the damage done through processing and disposal in latter categories. Release of halogenated hydrocarbons induces ozone depletion and has no notable share in the final impact. There is even a remediation effect through the displacement of electricity, 1.6 times the damaging effect. Ionising radiation is mainly induced by carbon-14 during electricity production. Since Belgian electricity is for 54% of nuclear energy, there is a net remediation effect of 1.66E-06 DALY FU^{-1}. Also phosphate emissions, leading to freshwater eutrophication, are coming from electricity production. By consequence there is considerable net remediation effect in this category.

Note that if one would attribute/allocate the total 2.65 m^3 wood under bark needed (see section 2.3.1) to the production of 1 m^3 of sawn timber, the values for the Scots pine stand would be a factor 2.65 higher. This would also mean that the 2.65 m^3 of the wood would not be allocated to the different co-products.

The resource consumption by the technosphere is here more elaborated (Figure 2.7). The 5.12E+03 CEENE displaced for electricity consumption is namely nuclear energy (53%) and fossil fuels (40%), reflecting the profile of electricity production. During processing resources are mainly fossil fuels used in fuel-driven machinery for harvesting and sawing the wood, and their transport to the site. For the disposal, fossil fuels are needed for the transport of and the natural gas and ammonia used in the abatement of NO_x (Doka, 2009).

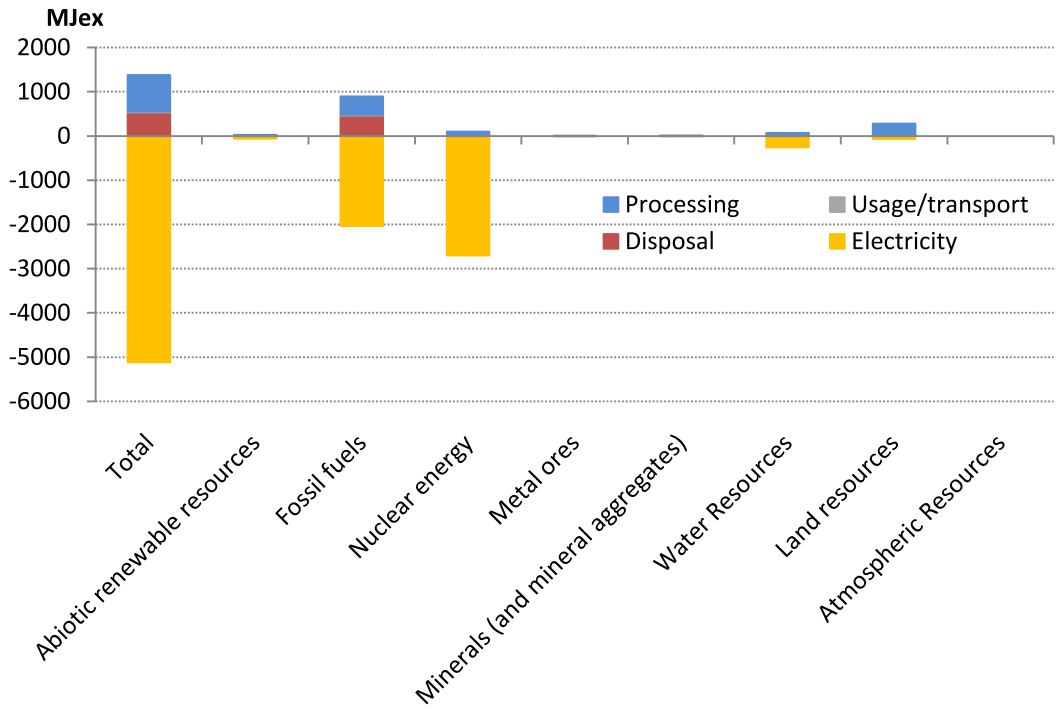

Figure 2.7. Resource consumption (and profile), expressed as the cumulative exergy extracted from the natural environment, per m^3 of sawn timber wood of the technosphere part of the product's life cycle.

2.5.4.1 Comparison with a conventional LCA; without Scots pine stand

A conventional LCA is performed, assessing the environmental impact of the technosphere in industrial processing of wood into sawn timber, i.e. without the Scots pine stand and all its emission and resource flows (see Figure 2.8, Table 2.2, Table 2.3 and Table 2.4). This is done to compare its findings with that of the original LCA performed. Impact in categories ionizing radiation, marine, freshwater, terrestrial and human (eco)toxicity, and freshwater eutrophication are unaltered since these are only brought forth by human/industrial processes, based on the known data.

Regarding emissions and land occupation, a damaging impact on the human health (1.35E-03 DALY FU^{-1}), 10 times lower, and ecosystem diversity (7.39E-09 species*yr FU^{-1}), 440 times lower, would be considered for the conventional LCA. There would thus be no positive impact considered on human health. The most important category would be climate change since it accounts for 94% and 97% of the impact on human health and ecosystem diversity, respectively. The greenhouse gases emitted during disposal of the wood sum up to 1032 kg CO_2 eq., 97% of the total. This is by 15% remediated by the displacement of greenhouse gas emissions during electricity production. The second biggest impact (6%) on human health is through particulate matter formation. This impact can be mostly contributed to the emission of nitrogen oxides during the disposal phase, burning of wood. The rest of the ecosystem diversity loss (3%) is due to urban land occupation. The sawmill plant accounts for the majority of this occupation. Findings differ considerably from the original LCA; most important there is a much lower absolute impact and no remediation effect on human health. And climate change is by far the dominant impact category.

If the production by the Scots pine stand is not considered, the stem wood extracted from nature should be accounted for as a renewable resource. Hence, the exergy value of 1 m^3 of stem wood would be added to the total CEENE value of the conventional LCA on the studied system. This exergy value, 1.06E+04 MJex m^{-3}, is calculated by multiplying the wood densitiy 502 kg Dry Matter (DM) m^{-3} stem wood (Janssens et al., 1999) with the specific exergy content, 21.1 MJex per kg DM, obtained via the group contribution method using data of the Phyllis database ("Phyllis database," 2013), similar to Alvarenga et al (2013).

Regarding resource consumption (Figure 2.8), the total CEENE value of the conventional LCA (6.9 GJex FU^{-1}) is about 60 times smaller than the total CEENE value of the original LCA (3.99E+02 MJex FU^{-1}). The Scots pine stem wood represents (88%) of the exergy input as a renewable resource. All other input flows of the technosphere of the original LCA are the same for the conventional LCA. By consequence, the absolute values of the other resource fingerprint categories (nuclear, water, et cetera) are identical. Their share does however differ. 12% of the resource consumption now originates from processing and disposal. The displacement of electricity production now has a considerable share in resource preservation by 43%. Marine eutrophication is dominated by the emission during disposal of nitrate and nitrogen oxides, latter compounds also lead to the high share in photochemical oxidant formation and terrestrial acidification.

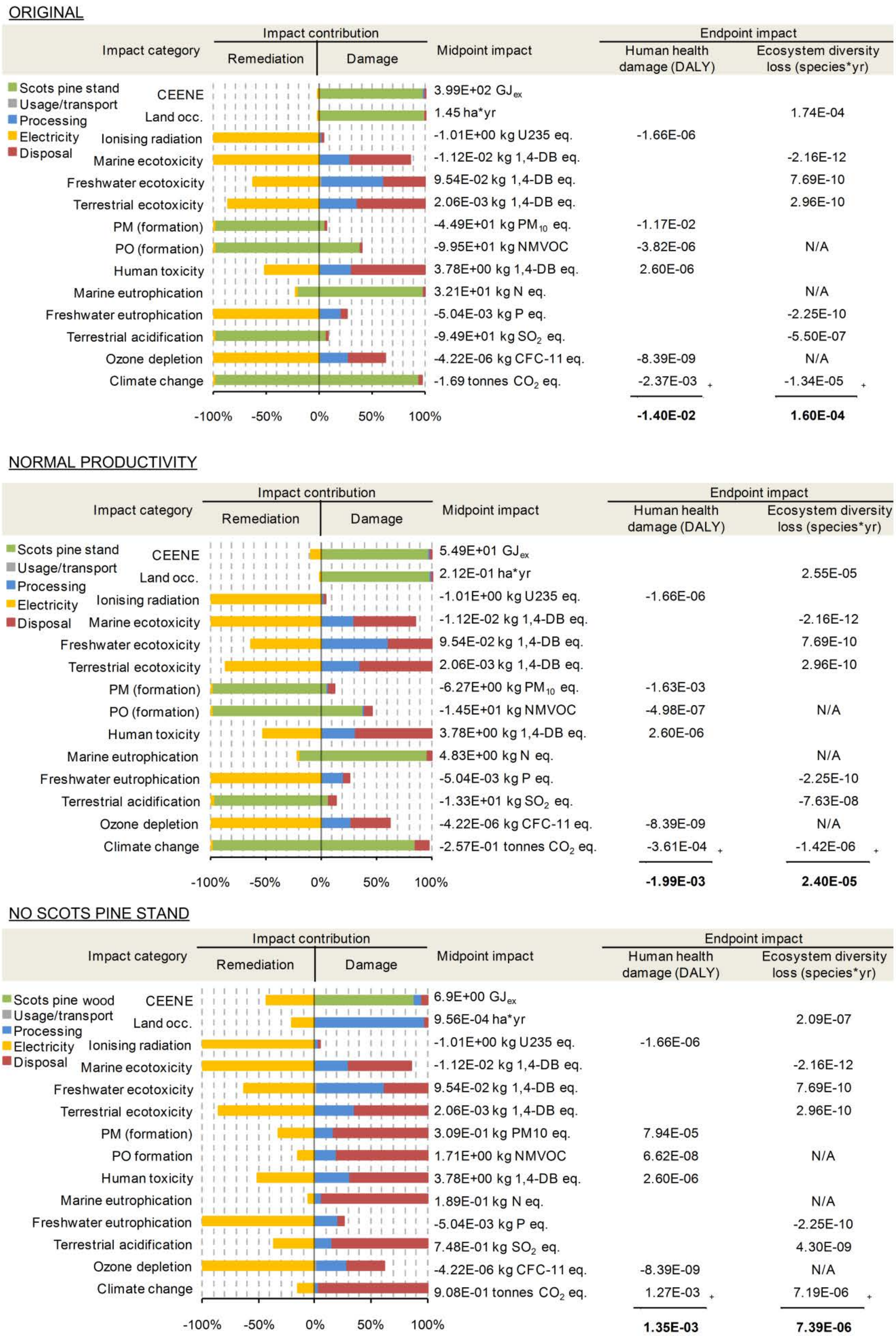

Figure 2.8. Different impact assessment results of the studied life cycle for 1 m^3 of sawn timber produced (Figure 2.3). Three different results are shown: the original results, results if normal productivity was valid and results if the scots pine stand (nor its resources and emissions; only the wood itself) were included in the LCA. cf. Figure 2.4.

2.5.4.2 Case study results with normal wood productivity

The Net Primary Production (NPP) of the Scots pine forest ecosystem for the period 2001-2003 is 9.4 tonnes biomass (BM) ha^{-1} yr^{-1}, of which 8.2 is of the trees and 1.2 by the understory vegetation. There is only a stem NPP of 0.4 tonnes BM ha^{-1} yr^{-1} (Yuste et al., 2005). The increment of the Scots pine stand in total is 1.45 tonnes BM ha^{-1} yr^{-1} (including 0.2 tonnes BM ha^{-1} yr^{-1}). We do not consider any stem litter thus the stem increment equals 0.4 tonnes BM ha^{-1} yr^{-1}. Using yield tables an estimated wood production of 5.5 m^3 ha^{-1} yr^{-1} (2.75 tonnes BM ha^{-1} yr^{-1}) is obtained (Jansen et al., 1996). The low stem growth would be because of a low canopy cover and a high soil acidification, a high deposit of such compounds is measured (Neirynck et al., 2011), which indirectly demands lots of energy consumption by the roots to overcome this (personal communication with prof. dr. Ivan Janssen of the research group studying the stand).

For the natural vegetation, based on results of Harbel et al. (2007), an NPP production of 13 tonnes BM ha^{-1} yr^{-1} is estimated. This is somewhat higher than the production of 9.4 tonnes BM ha^{-1} yr^{-1} by the Scots pine stand, retrieved from yield tables, but in the same order. For the natural vegetation about 40%, assuming a 50/50 mix of temperate humid evergreen and deciduous vegetation, can be appointed to wood (Luyssaert et al., 2007). This would mean a 3.76 tonnes BM ha^{-1} yr^{-1} productivity. This is close to the estimated 2.75 tonnes BM ha^{-1} yr^{-1} obtained using yield tables but almost 7 times bigger than the measured production of 0.4 tonnes BM ha^{-1} yr^{-1}, the reason for this difference is already explained above.

The discrepancy between natural and the actual low wood increment, 0.4 tonnes BM ha^{-1} yr^{-1} (Yuste et al., 2005), clarifies the high CEENE resource input of land occupation by the forest (as it equals the natural NPP production on the land needed to produce 1 m^3 stem wood in the Scots pine stand). A normal wood production for the Scots pine stand, 2.75 tonnes BM ha^{-1} yr^{-1} instead of 0.4 tonnes BM/ha/yr, leads to a 6.875 (2.75/0.4) times lower land occupation input by the Scots pine stand. Also, all the emissions of the forest would be that factor lower, except for the carbon dioxide input. This one is lower but also the additional carbon dioxide amount stored as carbon in the wood is accounted for. The absolute impact of the forest would thus be a factor 6.875 lower. The results would be different, see Figure 2.8.

In total impact there is a considerable drop compared to the original results: 7.3 times lower CEENE value, 7 times lower benefit for human health and 6.7 times lower loss of species diversity. Otherwise the share in impacts and the contribution of the different processes have not changed considerably. The most remarkable change is the increase in the contribution of climate change which is now 18% for human health and a remediation effect of 6% for diversity loss.

Chapter 3 Multilayered modelling of particulate matter removal by a growing forest over time, from plant surface deposition to washoff via rainfall

Redrafted from:

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Abstract

Airborne fine Particulate Matter (PM) is responsible for the most severe health effects induced by air pollution in Europe (European Environment Agency, 2013). Vegetation, and forests in particular, can play a role in mitigating this pollution since they have a large surface area to filter PM out of the air. Many studies have solely focused on surface dry deposition of PM on trees, but deposited PM can be re-suspended to the air or may be washed off by precipitation dripping from the plants to the soil. It is only the latter process that represents a removal. To quantify this removal all these processes should be accounted for, which is the case in our modeling framework. Practically, a multi-layered PM removal model for forest canopies is developed. In addition, the framework has been integrated into an existing forest growth model, ANAFORE (Deckmyn et al., 2011, 2008), in order to account for alteration in PM removal efficiency during forest growth.

A case study was performed on a Scots pine stand in Belgium (Europe), resulting for 2010 in a dry deposition of 31 kg $PM_{2.5}$ (PM < 2.5 μm) ha^{-1} yr^{-1} from which 76% was resuspended and 24% washed off. For different future emission reduction scenarios from 2010 to 2030, with altering $PM_{2.5}$ air concentration, the avoided health costs due to $PM_{2.5}$ removal was estimated to range from 915 to 1075 euro ha^{-1} yr^{-1}. The presented model could even be used to predict nutrient input via particulate matter though further research is needed to improve and better validate the model.

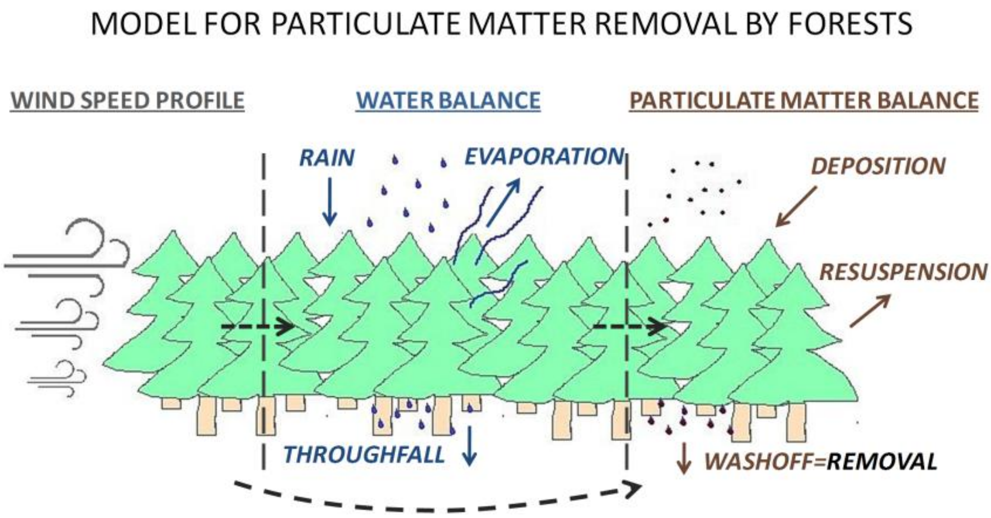

Figure 3.1. Graphical Abstract

3.1 Introduction

Airborne Particulate Matter (PM), occurring as solid or liquid matter, has a considerable damaging effect on human health by contributing to cardiovascular and cerebrovascular diseases (Anderson et al., 2012; Nelin et al., 2012; Wu et al., 2014). According to the World Health Organization (WHO) PM air pollution contributes to approximately 800 000 premature deaths each year, ranking it as the 13th leading cause of mortality worldwide (WHO, 2002). With regard to severity of human toxicity, an increase in damage has been associated with a decrease in particle size (De Nocker et al., 2010; Mirowsky et al., 2013), though Perronne et al. (2013) argue this matter. Airborne particles are commonly subdivided according to their size via their (aerodynamic) diameter, e.g. $PM_{2.5}$ denotes all particles with an (aerodynamic) diameter smaller than 2.5 µm. Important (emission) sources for PM, thoroughly discussed in the review by Belis et al. (2013), consist of traffic, crustal/mineral dust, sea/road salt, biomass and fossil fuel burning, (industrial) point sources and atmospheric formation of secondary aerosol.

Trees/forests can mitigate the damaging effect of PM through removal and subsequent lowering of its concentrations in the air (Nowak et al., 2013; Schaubroeck et al., 2013; Tiwary et al., 2009), see also Chapter 2, pg. 17. This ecosystem process is being increasingly regarded as an important ecosystem service. Various experimental and modelling studies have by consequence been made to characterize PM removal by trees and/or forests (Fowler et al., 2009; Hirabayashi et al., 2012; Katul et al., 2011; Lin and Khlystov, 2012; Petroff et al., 2009, 2008; Popek et al., 2013; Pryor et al., 2008; Sæbø et al., 2012; Terzaghi et al., 2013; Wang et al., 2006).

To quantify the total removal by vegetation, all relevant underlying dynamic processes should be addressed. These are: Dry Deposition (DD) on the vegetation surface, the subsequent (delayed) dry resuspension from the vegetation surface, wash-off due to precipitation events (Nowak et al., 2013), and dissolution in water, plant uptake and/or encapsulation into the wax layer (Popek et al., 2013; Sæbø et al., 2012; Terzaghi et al., 2013). Removal of PM is defined here as the amount that cannot be resuspended again, thus the washed-off, taken-up, dissolved and encapsulated amounts, and not just the deposited share. No values are currently known for the rates at which dissolution, encapsulation and uptake of PM occur and they are therefore not considered further on. To our knowledge Nowak et al. (2013) present the only framework which also covers wash-off besides deposition and resuspension, and thus integrates PM and canopy interception modelling. However, considerable improvements can be made to their model, as will be explained further on. Additionally, a dynamic modelling of PM removal over time as the forest grows and alters under different management and weather/climate scenarios is lacking. In this study, an improved modelling framework, called Canopy Intereception and PArticulate matter removal Model (CIPAM), is therefore presented in order to estimate PM removal by trees, and its integration into a process-based forest growth model (Figure 3.2). The selected forest growth model is the ANAlysis in FORest Ecosystems (ANAFORE) model (Deckmyn et al., 2011, 2008). Note that through integration, our framework also has improved the ANAFORE model. CIPAM may, however, be used on its own if the necessary inputs are provided. The focus of this paper is on the overall framework, and less on the separate submodels.

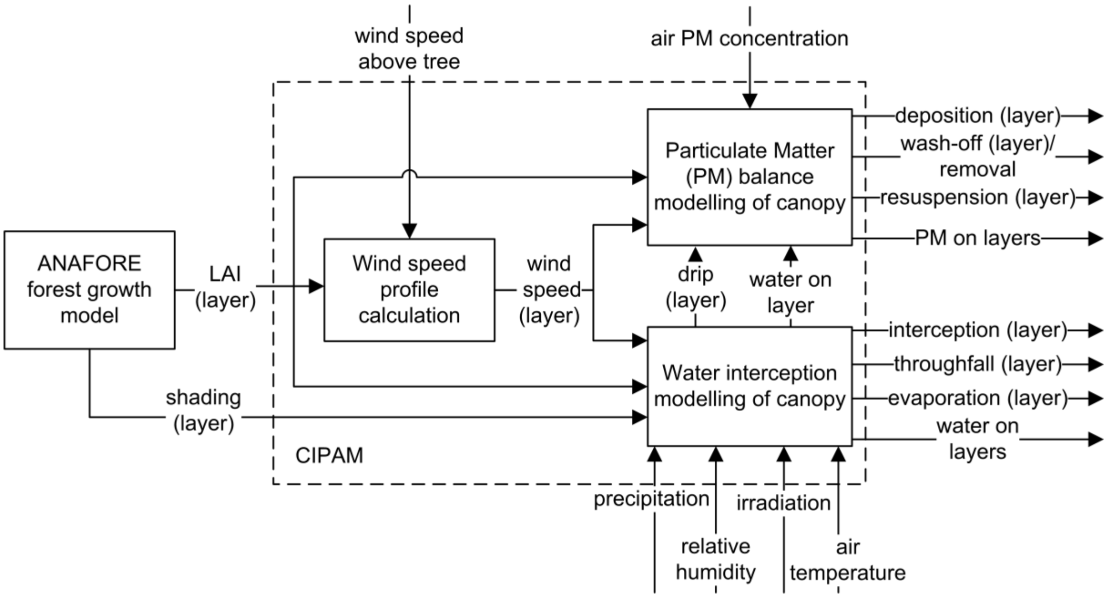

Figure 3.2. The introduced Canopy Interception and Particulate matter removal Model (CIPAM) and its integration with the ANAlysis in FORest Ecosystems (ANAFORE) model. The leaf area index and shading per canopy layer is provided by ANAFORE as input for CIPAM. Model calculations are made at a certain time interval (e.g. 30 min) and per canopy layer with living foliage. Only variables with the mentioning of 'layer' need to be known or are calculated per layer. The feedback loops within the dry matter balance and interecpetion modelling are not depicted. PM: Particulate Matter; LAI: Leaf Area Index.

The most important methodological improvement in our study is the consideration of different vegetation layers, as the Leaf Area Index (LAI) may vary considerably along a vertical gradient of a canopy (Aber, 1979; Van der Zande et al., 2009), with water and PM exchange between layers and a layer-specific characterization of wind speed, evaporation, dry deposition, etc. Nowak et al. (2013) though only perform calculations for a total tree canopy without subdivision in layers and considering a site average wind speed. Besides that difference, contrary to Nowak et al. (2013) we have also considered dry deposition, resuspension and interception evaporation during precipitation events, since PM concentrations do not drop to zero when it rains (Feng and Wang, 2012; Gonçalves et al., 2010).

Nowak et al. (2013) also assumed that all the deposited PM is washed off by precipitation and they acknowledge this as a limitation. Here, the PM quantity that is washed off depends on the amount of throughfall drip from each layer. Despite these improvements in the modelling framework, there are still considerable assumptions and shortcomings, which are listed in Table 3.3, pg. 80, in the supporting information.

The model is applied to a case study of a Scots pine stand in the Campine region of Flanders (northern Belgium) for the year 2010 using model runs with half-hourly calculations. To illustrate the potential importance of PM removal, Scots pine is a relevant example since studies have reported its good PM removal efficiency (Pullman, 2009; Sæbø et al., 2012). This is, amongst other features, caused by its evergreen and coniferous canopy (Beckett et al., 2000). Scots pine is also a major tree species in Flanders and Europe (Skjøth et al., 2008; Tröltzsch et al., 2009). Airborne PM is a major health concern in these highly populated and heavily industrialized areas. In the period 2009-2011 more than 90% of the European population was exposed to a yearly average $PM_{2.5}$ concentration that exceeds the current threshold value of the WHO: 10 µg m^{-3} (European Environment Agency, 2013). Also in Flanders this threshold value is exceeded since the average $PM_{2.5}$ concentration in 2011 was 17-24 µg m^{-3} at different sites (Vlaamse milieumaatschappij, 2011). In the future, PM concentrations are predicted to decrease in Flanders in response to the implementation of emission legislation (Van Steertegem, 2009). These concentration changes have been modelled for different scenarios until the year 2030 (Van Steertegem, 2009). The model introduced below will also run until 2030 for these emission reduction scenarios to examine the response to changes in PM concentrations and to predict the future PM amounts removed by the studied forest.

3.2 Methods

3.2.1 Modelling framework and integration into ANAFORE

A modelling framework CIPAM is introduced, which encompasses three submodels that estimate: (1) the wind speed along the tree crowns, (2) the water interception and (3) the particulate matter (PM) balance of the forest canopy (Figure 3.2, pg. 58). This framework is integrated into the ANAFORE model (4). These four different aspects are explained separately further on. The ANAFORE model provides leaf area (index) values, which is an important input variable for all three submodels, and shading values, which influence interception evaporation in the canopy layers. Wind speed is an important driver for the other submodels since it affects canopy evaporation and dry deposition and resuspension of PM. The interception model yields per canopy layer the amount of water dripping to the lower layers, used to estimate the wash-off of PM, and the interception water amount remaining per layer, which protects PM from being resuspended. Figure 3.3, pg. 60, gives an overview. These calculations are done per horizontal layer with living foliage and are thus restricted to the canopy part with living foliage. The thickness and amount of layers can be freely selected as long as LAI and shading of all the layers are given as inputs. The change in vertical distribution of foliage over time is hence accounted for through the change in LAI-values of the layers over time. If foliage or whole trees die-off or are cut (due to thinning) the water and PM on the layers are considered as throughfall and removed PM, respectively. The calculations of CIPAM can be done for a given time interval, e.g. half-hourly in the case study, and are performed per layer starting from the top layer and continue progressively towards the lower layers. Additionally, the ANAFORE model allows one to estimate the included processes while the forest is growing and is being managed. In the following text, subscript denotes a specific canopy layer with living foliage, where layer counting starts from the top of the tree, unless mentioned otherwise.

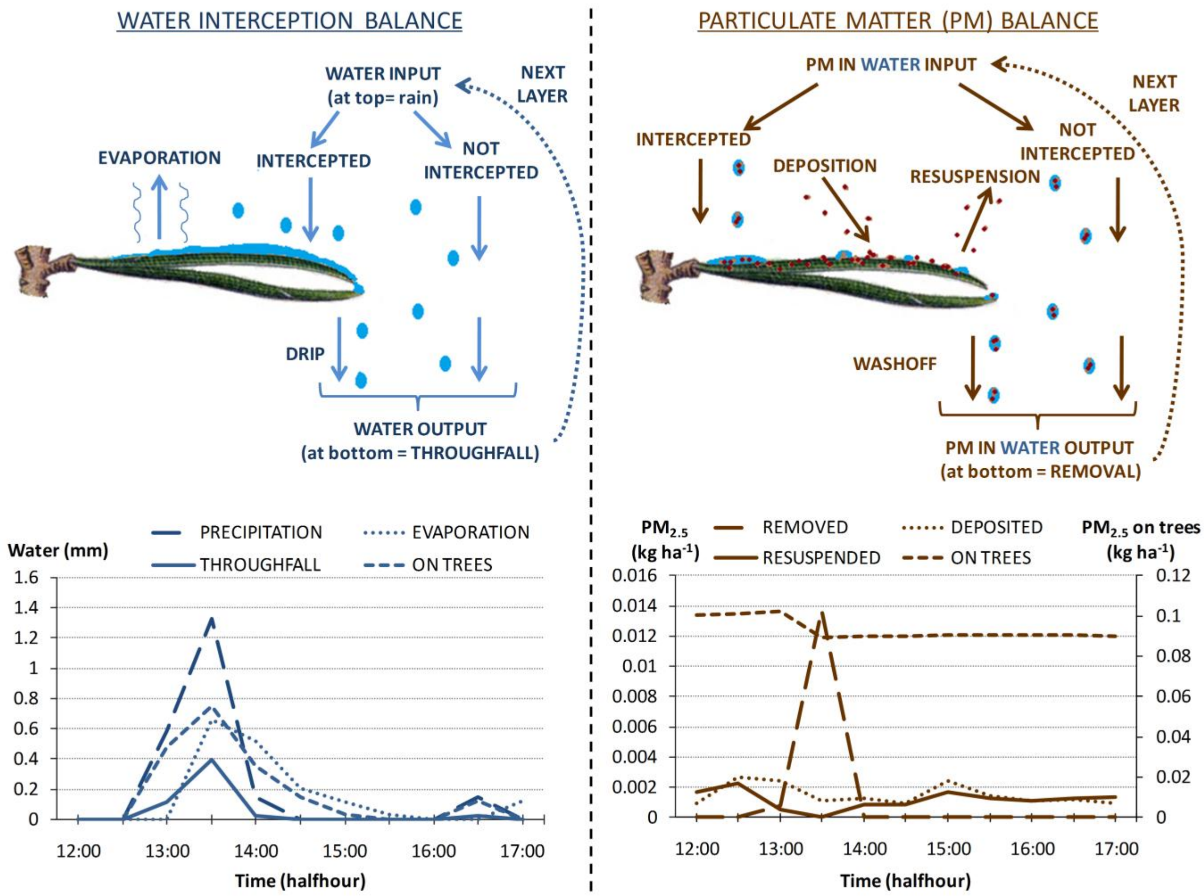

Figure 3.3. Considered fluxes per layer (drawing) and modelled results of these for the total tree over time (graphs), concerning the water and particulate matter (PM) canopy balances in the modelling framework. Wash-off and drip occur when the water on the layer exceeds the storage capacity. The loop to the next tree layer is depicted using dotted arrows in the drawings. The graphs represent the case study results of the complete canopy for the Scots pine stand on the 1st of July 2010. 'On trees' implies present on the surface area of the trees.

The horizontal layers all have a same height and extend over the complete forest, i.e. layer 1 of a tree cohort is at the same height as that of all the other tree cohorts. They are derived and thus defined in the same manner as those of ANAFORE (Deckmyn et al., 2008), see figure below. As the trees grow, more layers are created.

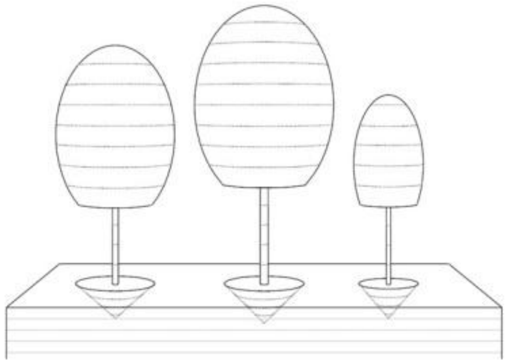

Figure 3.4. Simplified scheme of a forest stand in ANAFORE: individual trees of each cohort described as a truncated elliptic crown, cone-shaped root volume, horizontal layers in the crown and soil. This figure was redrafted from the work of Deckmyn et al. (2008).

Note that in this framework no horizontal change of deposition across the forest is considered. The forest stand is considered to be surrounded by other stands of similar height, so that forest edge effects can be neglected (Wuyts et al., 2008). The PM removal by understory vegetation is also considered when including their LAI values and introducing respective model parameter values. We assume that water and PM are inert to other processes (aggregation, plant uptake, encapsulation,...) than the ones described below. Practically, the programming code is written in FORTRAN and compiled with Intel Fortran compiler 14.02.

3.2.2 Wind speed calculations

Wind speed varies considerably along the vertical profile of the forest and is thus a function of height (Sypka and Starzak, 2013). The ANAFORE model already provides a calculation of wind speed for a specific height using an natural logarithmic function based on the work of Raupach (1994). Sypka and Starzak (2013) disapproved the use of such a function given the S-shaped wind speed profile in a forest that they and others observed. They advised to use the equation of Yi (2008) within the canopy, which we will apply here. Yi (2008) derived a formula to calculate wind speed at different heights through canopies with a uniform vertical distribution of the leaf area index (LAI; m^2 leaf area m^{-2} ground area):

$$U(h) = U_H*exp(-1/2*LAI*(1-h/Hc)) \quad (3.1)$$

In this formula, $U(h)$ is the wind speed (m s^{-1}) at the height h (m) within the canopy, U_H is the wind speed at the top of the canopy and H_c is the height of the canopy (m). This is done for each layer based on the wind speed of the above layer and assuming a tree stand with similar configuration as the particular layer. The wind speed for each layer *i*, U_i (m s^{-1}), is considered as the average of the wind speed at the top and bottom of the layer.

3.2.3 Interception modelling

Interception modelling is well reviewed by Muzylo et al.(2009). Many studies focus only on water interception and storage by the leaf surface area. Llorens and Gallart (2000) however, point out the importance of including the wood area as well, particularly for Scots pine, which was done accordingly in the present study. Note that we did not account for stemflow. Stemflow is though mostly a minor flow and not always accounted for in interception modelling (Muzylo et al., 2009). Out of the review and results of Crockford and Richardson (2000) we concluded that stemflow is rarely higher than 10% for tree species, and only 2% is reported for Scots pine by Llorens et al. (1997). The calculation of the water mass balance per canopy layer constitutes the basis of this submodel (Wang et al., 2008):

$$\Delta W_i = f_i I_i - E_i - D_i \quad (3.2)$$

where W_i (mm) is the water amount of layer *i*, I_i (mm) the water input, f_i (-) the fraction of intercepted water, E_i (mm) the evaporation rate and D_i (mm) the drip rate to the next layer, per layer *i* and time interval.

The interception fraction f_i, as calculated by Deckmyn et al. (2008), is based on Van Dijk and Bruijnzeel (2001):

$$f_i = 1 - exp(-k * LAI_i) \quad (3.3)$$

Factor *k* is here called the interception coefficient. This constant is considered equal to 0.7 for forest (Deckmyn et al., 2008; Wang et al., 2008).

The water input I_i is the sum of the drip and not-yet intercepted, free throughfall, amount of water input received from the above layer:

$$I_i = D_{i-1} + (1-f_i) * I_{i-1} \quad (3.4)$$

For the top(most) layer, this input, I_1, is the precipitation (mm) over the given time interval. A crucial parameter for calculating evaporation and drip is the storage capacity

S_i (mm), which is the amount of water which can be stored/retained/accumulated by a layer on its foliage and wood (stem and branches). Specific storage capacity amounts can be measured for both tree parts as well as for layers (Liu, 1998; Llorens and Gallart, 2000). The following formula for storage capacity was derived by Llorens and Gallart (2000):

$$S_i = LAI_i*2*SL+SW*WAI \quad (3.5)$$

where *SL* (mm) and *SW* (mm) are the specific storage capacities per LAI and wood area index (WAI; m^2 wood area m^{-2} ground area), respectively. The wood area can be related on an empirical basis to the average leaf area of a certain time interval t by *R*, the ratio of LAI per WAI. This results in (Llorens and Gallart, 2000):

$$S_i = LAI_i*2*SL+SW/R*\pi*LAI_i(t) \quad (3.6)$$

Alternatively, it is possible to model the Wood Area Index (WAI) needed in the calculation of water storage by woody tree parts since the modelling of stem and also tree branches was already included in the ANAFORE model (Deckmyn et al., 2008), though branching occurs in a simple manner compared to more complex models such as that of Lintunen et al. (2011). First the Branch Area Index (BAI) is calculated. Note that only maximum 10 branch whorls are considered and no further branching of branches. However these non-primary branches are most probably negligible in projected area. BAI is calculated as follows, considering a triangular branch shape (with *q* the number of the whorl):

$$BAI = \sum_{q=1}^{10} BRq*2*BLq*BLR/2 \quad (3.7)$$

BRq is the branch radius at the stem (m), *BLq* the branch length (m) and *BLR* the branch length ratio (-), being the ground projected branch length per branch length. As an assumption the basal area was used for the stem and the WAI of the tree was divided over the tree layers based on the leaf area index (LAI) distribution (assuming the ratio of woody to leaf area as a constant fraction over the canopy per time step), resulting in a WAI per layer of:

$$WAI_i = (BA + BAI)*LAI_i/(\sum_{i=1}^{s} LAIi) \quad (3.8)$$

Herein, BA is the basal area (m^2 stem m^{-2} surface area). BAI and WAI are calculated on a yearly basis. This WAI$_i$ calculation is inserted in equation 3.5, by substituting the term WAI.

Prior to calculation of the actual evaporation rate per layer E_i, the potential evaporation rate (Ep_i) needs to be calculated per time interval. This potential evaporation rate is the rate of actual evaporation if the considered canopy surface would be fully covered by water. It is here calculated using the widely applied Penmann(1948)–Monteith(1965) equation as done in the ANAFORE model, but not including the stomatal resistance applicable for transpiration. This rate is based on meteorological conditions: wind speed (varying for each layer), solar radiation, humidity and temperature (Deckmyn et al., 2008). As solar radiation has a considerable influence on evaporation rates, we took into account the influence of shading. Separate potential evaporation rates are calculated for the shaded (Eps_i) and sunlit (Epl_i) canopy parts, based on the associated different irradiation inputs per layer. An overall potential evaporation rate is then estimated by the weighted average of the separate ones, as represented in the next equation:

$$Ep_i(U_i) = SF_i*Eps_i(U_i)+LF_i*Epl_i(U_i) \quad (3.9)$$

SF_i and LF_i are the fractions of the layer which are shaded and lit, respectively, computed by ANAFORE. Having calculated the potential evaporation rate, the actual evaporation rate might be calculated via the following equation (Wang et al., 2008):

$$E_i = (W_i/S_i)^{2/3}*Ep_i(U_i) \quad (3.10)$$

The values W_i and S_i represent the values at the beginning of the considered time interval. No actual evaporation rate is computed separately for the shaded and sunlit part as the specific water amounts on these parts are not known and the parts of the tree which are lit or shaded, change during daytime as the sun position alters. A complex geometrical model is needed to address this matter. Indirectly we thus assume that the water per surface area is equal for the sunlit and shaded parts of each layer.

Drip D_i (mm) from a layer to the next layer below occurs if the water input W_i (mm) exceeds the storage capacity S_i (mm) of a layer at the end of a time interval:

$$D_i = W_i - S_i \quad (3.11)$$

The W_i (mm) is in that case set equal to S_i at the end of an interval.

The forest throughfall over a certain time interval, T (mm) is then the water leaving the lowest layer s with living foliage:

$$T = D_s + (1-f_s)*I_{s-1} \quad (3.12)$$

The total canopy evaporation rate, CE (mm), is the sum of the evaporation from all layers per time interval:

$$CE = \sum_{i=1}^{s} E_i \quad (3.13)$$

This multi-layered interception model, with evaporation based on Penman-Monteith for each layer and inclusion of wood area and shading, appears to be conceptually a high-end model among the ones mentioned by Muzylo et al. (2009). Our submodel is a considerable improvement compared to the original approach in the ANAFORE model, in which canopy evaporation and drip were considered very simple using constant fractions (0.5) of the intercepted rain amount, based on Sampson et al. (2001).

3.2.4 Particulate matter modelling

The particulate matter (PM) amount on a tree layer changes over time. The basic mass balance is the following:

$$\Delta P_i = DD_i - RS_i - WO_i + PI_i * f_i \quad (3.14)$$

where P_i (µg) is the PM amount on the surface of foliage and wood of the layer, DD_i (µg) is the dry deposition, RS_i (µg) the resuspension and WO_i (µg) the wash-off amounts per layer and time interval. The f_i term is the interception fraction as explained in the above section. The last term of this equation denotes the input, besides through deposition, of PM, PI_i, over layer i and the given time interval. This is the sum of wash-off from and non-intercepted input of the above layer:

$$PI_i = WO_{i-1} + (1-f_i)*PI_{i-1} \quad (3.12)$$

Dry deposition is the combined removal of particles from the atmosphere by sedimentation, Brownian motion, impaction and direct interception (Petroff et al., 2008). Sedimentation can be neglected for smaller-size particles belonging to class $PM_{2.5}$ (Neirynck et al., 2007). Different research with associated approaches exists to address dry deposition on vegetation surfaces (reviewed by Petroff et al. (2008)). The direct dry deposition rate or flux of a pollutant, here PM, per leaf area, without considering resuspension, DD_i (µg m^{-2} time interval^{-1}) can be estimated as:

$$DD_i = V_i(U_i) * C \quad (3.15)$$

where V_i is the dry deposition velocity of the pollutant, here PM, per surface area (m time interval^{-1}) and C is the concentration of the pollutant, here PM (µg m^{-3}) (Hicks et al., 1989; Nowak et al., 2013). Deposition is usually expressed per ground surface area instead of per plant surface area, but here we refer to the one per plant area, unless mentioned otherwise. To obtain the deposition rate per layer, the deposition velocity is multiplied with the surface area of the layer. This deposition velocity per plant surface area depends on the wind speed, particle size and tree configuration, defined, amongst others, by the tree species (Nowak et al., 2013; Petroff et al., 2008). For example, pine needles are highly dissected and have a high surface area compared to flat broadleaves, per length of primary branch, and have been found to have ten times higher deposition velocities than broadleaves (Beckett et al., 2000). These species-specific deposition velocities, related to wind speed (or friction velocity) and PM size, need to be derived from experiments (empirically), via wind tunnel or field measurements, or calculated (mechanistically). Although the latter approach has been widely used (Kouznetsov and Sofiev, 2012; Petroff et al., 2009, 2008; Piskunov, 2009), we here consider an empirical approach, similar to Nowak et al. (2013). This approach was selected for its simplicity, linkage with measured results and inclusion of rebound, i.e. the direct removal of particles during impaction (Paw U and Braaten, 1992).

Resuspension, more precisely delayed resuspension, is the resuspension of material, such as PM, from surfaces, strictly speaking only the quantity which was deposited via atmospheric pathways, through wind shear or mechanical actions (Nicholson, 1993; Pryor et al., 2008). Though it is shown to be an important process (Gillette et al., 2004; Nicholson, 1993; Pullman, 2009), it is rarely addressed in studies on PM removal by dry deposition onto vegetation. Sometimes a fixed constant for resuspended fraction per deposited amount is considered, such as 50% for PM_{10} (Hirabayashi et al., 2012; Zinke et al., 1967). However, resuspension depends on the accumulated PM amount on the tree (layer) and the wind speed (Nicholson, 1993; Pryor et al., 2008; Pullman, 2009). The more particles accumulated on the foliage, the more particles can be removed. In addition, we consider the prevention of resuspension due to the water present on the canopy. Though, not the complete surface of the canopy (layer) is wet, only a part. Here, we estimate this fraction by the ratio of W_i on S_i. Resuspension is then calculated using the following formula:

$$RS_i = RSf_i(U_i)*P_i*(1-W_i/S_i) \quad (3.16)$$

In this equation RSf_i (-) is the fraction of resuspensed PM per PM present on the layer per time interval of layer i. The values of W_i, S_i and P_i are those at the beginning of the time interval. Note that W_i can be maximally equal to S_i at the beginning of a time interval (see section 3.2.3, pg. 62). RSf_i is influenced by the wind speed (Nicholson, 1993; Pullman, 2009). To our knowledge, no mechanistic approach to calculate these values has been reported yet. Empirical values should therefore be used. The wash-off of PM due to drip is calculated as:

$$WO_i = P_i*D_i/(S_i + D_i) \quad (3.17)$$

In contrast to the approach of Nowak et al. (2013), not all PM is considered to wash off during canopy drip, which implies an important difference. The total PM removed by a forest, *PR*, (μg) over a certain time span is (with *s* the lowest canopy layer with living foliage):

$$PR = WO_s + (1\text{-}f_s)^*WO_{s-1} \quad (3.18)$$

The total resuspension, *TRS*, is the sum of the resuspension of all layers with living foliage:

$$TRS = \sum_{i=1}^{s} RS_i \quad (3.19)$$

3.2.5 Integration into the ANAFORE model

The process-based ANAlysis of FORest Ecosystems (ANAFORE) model was developed by Deckmyn et al. (2008) and later on improved with a better soil submodel (Deckmyn et al., 2011). It has already been applied and validated to the Scots pine stand considered here (Deckmyn et al., 2011, 2008). For more information, see section 3.5.2, pg. 81.

Because of the high variance in time of wind speed, PM concentrations, weather conditions and rainfall, it is crucial that the calculations are done using appropriate small time intervals. Our submodels were therefore integrated at the lowest, half-hourly, time step of ANAFORE. The inputs for the submodels are the leaf area (index) of the different canopy layers and the shading. The leaf area and LAI are recalculated on a daily basis, whilst the share of sunlit and shaded leaf area is determined on a half-hourly basis. The layer height is variable and is here set at 0.6 m, the smallest that can be used in the model, as LAI may vary considerably along a tree stem.

3.2.6 Case study

The model is tested for $PM_{2.5}$ exchange between the atmosphere and the Scots pine stand for the year 2010, see section 1.6 (pg. 12), and also ran for different future scenarios for the period 2010-2030 while the forest grows.

3.2.6.1 Model input data for the specific Scots pine stand

The main input variables concerning the Scots pine stand for the ANAFORE model are derived from Gielen et al. (2013) and Neirynck et al. (2008), and are mentioned in the supporting information, section 3.5.3, pg. 82. The modelled forest consists of trees which are assumed to be identical and no understory vegetation was considered to be present. A yearly value of nitrogen deposition to the soil is considered of 40 kg N ha^{-1} yr^{-1} with a share of 0.21 NO_y-N and 0.79 NH_x-N (Neirynck et al., 2008) and 390 ppmv CO_2. Half-hourly values for wind speed above the tree tops, air temperature, precipitation, relative humidity and radiation are obtained specifically for the Scots pine stand for the year 2010 from the Research Institute of Nature and Forest ("Instituut voor Natuur- en Bosonderzoek," 2013) and were measured as described in Neirynck et al. (2007). These meteorological values are considered to be the same for all other years in all scenarios, e.g. windspeed on a specific time in 2030 is the same of that on the same time in 2010. A distribution of these yearly wind speed values is depicted in Figure **3.5**. Less than 1% of the time points had no value and were given the average wind speed value of 2010, namely 1.931 m s^{-1}.

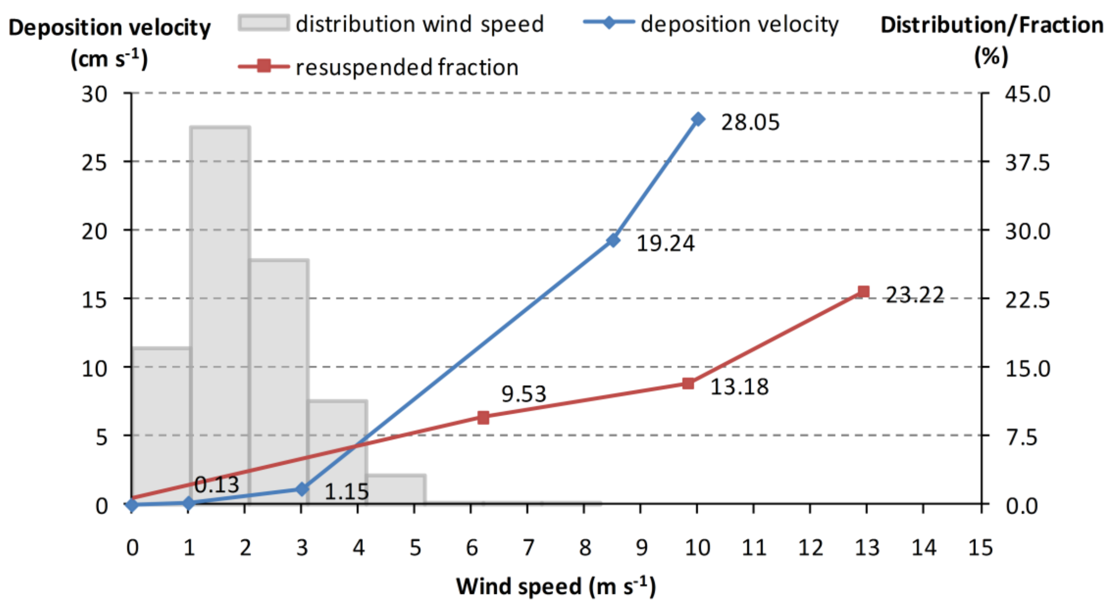

Figure 3.5. Distribution (%) of measured wind speed values (m s^{-1}) above the tree top at the Scots pine stand, considered deposition velocities (cm s^{-1}) per leaf area and resuspended fraction (%) as a function of wind speed (Beckett et al., 2000; Nowak et al., 2013; Pullman, 2009). Wind speed was measured on a half-hourly basis as described in Neirynck et al. (2007). All discrete values (labelled points on the graph) of the deposition velocity and resuspended fraction, except the (0,0) points, which are set by default, are retrieved from Beckett et al. (2000) and Pullman (2009), respectively. Linear interpolation between these discrete values, represented by the straight lines, is used to obtain values for other wind speeds.

Hourly $PM_{2.5}$ concentrations above the Scots pine stand for the year 2010 are obtained from the Belgian Interregional Environment Agency ("IRCEL - CELINE Belgium," 2014), which uses interpolation techniques to derive the concentration at other locations than those measured in discrete points by the Flemish Environment Agency ("Vlaamse Milieumaatschappij," 2013). For these data the more accurate RIO (Residual Interpolation Optimised) model was used with a resolution of 4x4 km (Janssen et al., 2008). In 2010 the modelled average $PM_{2.5}$ concentration above the Scots pine stand was 17.65 µg m^{-3} (71% of PM_{10}), highly determined by a nearby highway, see section 1.6 (pg.

12). Note that the distance to residential area is less than 0.5 km, PM removal in this area is by consequence most likely relevant. For the predictions until 2030 only the PM concentrations were considered to alter. Every five year (2015, 2020, 2025, 2030) PM concentrations were predicted for 3x3 km grids in Flanders for two alternative scenarios as was done by Van steertegem (2009). This was done based on the integrated approach of Deutsch et al. (2008) in which the outcomes of the BelEUROS model, the integrated Eulerian air quality modelling system for European Operational Smog adapted to model PM in Belgium (Deutsch et al., 2008), was interpolated with RIO, Residual Interpolation Optimised for ozone and extended to other pollutants (Deutsch et al., 2008; Hooyberghs et al., 2006), for the year 2007. The values for the years within the five-year intervals were determined using interpolation. The two alternative environmental policy scenarios are specific for Flanders and are those presented by a Flemish report of the VMM (Van Steertegem, 2009), the Reference scenario (REF), representing future conditions under an unaltered Flemish environmental policy of the year 2008, and the Europe scenario (EUR), in which the stricter environmental guidelines implemented by the European Union are followed, specific for PM this is given by the emission policy presented by Amman et al. (2008). The hourly PM concentrations are divided by the yearly 2010 average and multiplied with the predicted value. These values are shown in Figure 3.8, pg. 77.

3.2.6.2 Model parameter values for Scots pine

The parameter values are given in Figure **3.5**, pg. 68, and Table 3.1.

Table 3.1. Parameter values and their sources used in this study to model the canopy water and particulate matter balance. The parameter values used by Nowak et al.(2013) are mentioned in the last column.

	This study			Nowak et al. (2013)
Parameter	Value	Species	Source	
Water balance				
Extinction coefficient	0.7	All tree species	(Deckmyn et al., 2008; Wang et al., 2008)	0.7
Specific leaf storage capacity	0.0435 mm m^{-2} (windy) 0.1040 mm m^{-2} (still) Average considered: 0.0735 mm m^{-2}	Scots pine	(Llorens and Gallart, 2000)	0.20 mm m^{-2} for all tree species(Wang et al., 2008)
R-ratio	11.62	Red pine	(Deblonde et al., 1994)	not considered
Parameters of potential evaporation	(Deckmyn et al., 2008)			(Wang et al., 2008)
Particulate matter balance				
Deposition velocities	PM with mean diameter of 1.28 (±0.07) μm; see Figure 3.5	Black pine	(Beckett et al., 2000)	For different tree species, see Table 2 in Nowak et al. (2013)
Resuspension fractions	For $PM_{3.0}$; see Figure 3.5	Averages for 3 conifers: white pine, Japanese yew and eastern Hemlock	Data of Pullman (2009) interpreted by Nowak et al. (2013)	Identical values to this study but applied to all tree species

Regarding the leaf and woody storage capacity, specific values for Scots pine were adopted from the study of Llorens and Gallart (2000). The leaf storage capacity value is less than half of the value used by Nowak et al. (2013) for all their considered tree species.

Since no branch area is known for the Scots pine stand in 2010, we used the R-ratio (LAI/WAI) to quantify interception, see equation 3.6. This ratio is though variable among tree stands. Deblonde et al. (1994) denote an R-ratio of 3.0-11.6 for different red and jack pine stands. This R-ratio is dependent on different stand characteristics. Therefore, in our study we have selected the R-ratio value of 11.6 (i.e. the value of a stand most similar to ours). See supporting information, section 3.5.4, pg. 83. A variable R-ratio in function of (these) stand characteristics or a direct calculation of WAI is needed to better address this matter.

For the deposition velocity per foliar surface area as a function of wind speed, we used values for black pine (*Pinus nigra*) reported by Becket et al (2000), based on wind tunnel tests with pot-grown small trees using particles of 1.28 (± 0.07) µm diameter. Since *Pinus sylvestris* and *Pinus nigra* belong to the same family, their branching (amount, structure and orientation) and needle structure is rather similar, justifying the use of black pine values for Scots pine. However, the deposition velocity also depends on the particle size (Fowler et al., 2009; Petroff et al., 2008). Since the particles used by Beckett et al. (2000) were smaller than 2.5 µm, applying their values is a reasonable choice, though it might be a crude estimation. Deposition velocities and resuspension fractions are only given for three discrete values of wind speed (Figure **3.5**, pg. 68), so that functions are needed to determine these values as a function of wind speed. Similar to Nowak et al. (2013), for 0 m s^{-1} the deposition velocity was set to 0 cm s^{-1} and the resuspended fraction 0, this by default, and linear interpolation was used to derive estimated values between the discrete values.

Pullman (2009) studied the resuspension of $PM_{3.0}$ (with a mass-based average of 2.5 µm) from tree branches of three coniferous species in wind tunnel tests during 5, 10 and 20 minutes. We used the data of this study as reinterpreted by Nowak et al. (2013) to address resuspension fractions. Note that latter authors used these values for all types of different tree species over an hour. Since Scots pine is a conifer, as are the tested species of Pullman (2009), it is appropriate to apply her values. Also here the values are used for a half-hourly interval, which is closer to the original intervals reported by Pullman (2009). However, since the values are for $PM_{3.0}$, an overestimation of $PM_{2.5}$ resuspension is probable.

3.2.6.3 Experimental setup for collection of throughfall data and PM removal data

The experimental measurement data of 2010 used to validate the modelled values, are obtained via following methodologies. For half-hourly throughfall data two measurement campaigns were performed in the Scots pine stand using two gutter-like throughfall collectors. The respective material and methodology used for these measurements are described in the work of Neirynck et al. (2007). More or less 4-weekly removal rates of sodium and chloride present in particulat matter, are obtained via the canopy budget model of Staelens et al. (2008), based on that of Ulrich (1983). In practice, these values are thus obtained by substracting wet deposition values via rainfall (collected just outside the forest) from the total amount of wet and dry removal (collectors underneath forest canopy). The respective material and methodology used for these measurements are described in the work of Staelens et al. (2008).

3.3 Results and discussion

3.3.1 Case study results for 2010, validation and interpretation

3.3.1.1 Interception modelling

Figure 3.3, pg. 60, shows half-hourly example results of the water and particulate matter (PM) balances of the Scots pine stand at the smallest time interval. The total measured rainfall of 2010 was 842 mm of which, according to two measurement campaigns, 678 and 720 mm were measured. Our modelling framework, using a half-hourly time step, estimated a throughfall amount of 697 mm, which is very close to the measured amounts. The associated canopy evaporation was 145 mm. For a better data validation, throughfall measurements may be compared with modelled results on smaller time scales. The smallest possible time scale is half-hourly. The correlation coefficients between measured and modelled data are lower at the half-hourly time level than at the biweekly level (Table 3.2 and Figure 3.6). This is most probably due to the time delay during the transport of water through the canopy, which is not included in the model. Nevertheless, the biweekly correlation values indicates in our opinion that the model has a very good accuracy, although the measurements are slightly underestimated.

Table 3.2. Pearson correlation coefficients for different time intervals between the measured (2 series) and modelled throughfall data of the Scots pine stand in 2010.

Time interval	Throughfall 1	Throughfall 2
Halfhourly	0.88	0.88
Biweekly	0.99	0.99

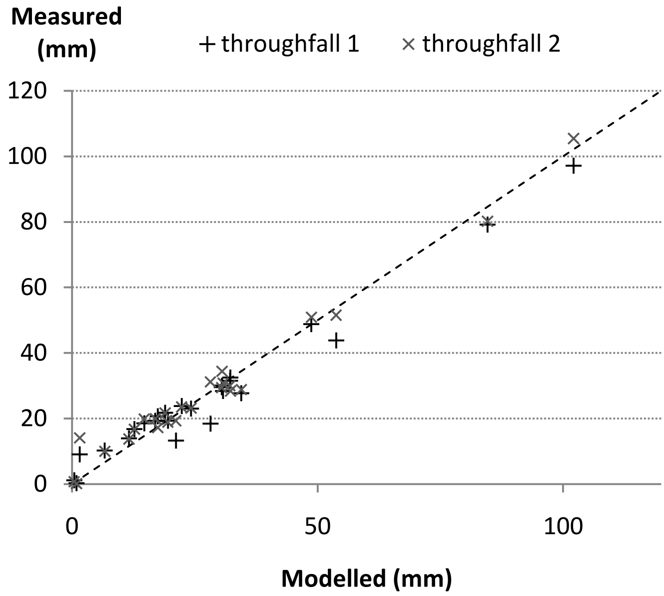

Figure 3.6. Comparison of modelled and measured (2 series) biweekly throughfall values for the Scots pine stand in 2010.

The ANAFORE model as presented in Deckmyn et al. (2008), unadjusted, would have obtained a throughfall of 518 mm. Hence, according to these first results, our modified ANAFORE model leads to more accurate results in terms of canopy interception modelling.

3.3.1.2 Particulate matter removal modelling

Concerning the PM balance, our modelling framework, CIPAM with ANAFORE, calculated for 2010 a total dry deposition of 31.43 kg ha^{-1} yr^{-1} $PM_{2.5}$, from which 23.93 kg was resuspended, 7.38 kg was considered as definitely removed (dripping of the canopy to the forest floor) and 0.11 kg was still present on the tree canopy at the end of the year. No total values of dry deposition or removal of $PM_{2.5}$ are known for the studied Scots pine stand for the year 2010. However, the estimated fluxes of some inorganic compounds via wet deposition (rainfall) and throughfall are known. Using the canopy budget model of Staelens et al. (2008), based on that of Ulrich (1983), allowed to retrieve more or less 4-weekly removal rates. As sodium and chloride are considered not to leak or to be taken up by the canopy, we will focus on these two elements. Important to note here is that the shares of these compounds in $PM_{2.5}$ are not constant over time (Bencs et al., 2008). Next to that, these are removal rates of sodium and chloride present in all PM with different sizes, not only $PM_{2.5}$. One may assume that the distribution of these elements over different PM size classes constant is. On a 4-weekly basis, results of the removal of chloride and sodium can be obtained, although there can be a considerable error since the canopy budget model is commonly applied at the (semi)annual time scale. Results are shown in Figure 3.7. The Pearson correlation coefficient between the modelled $PM_{2.5}$ removal and the removal of sodium and chloride were 0.64 and 0.62, respectively. So, there appears to be a reasonably similar trend, favoring the model. More data is however needed for a better validation.

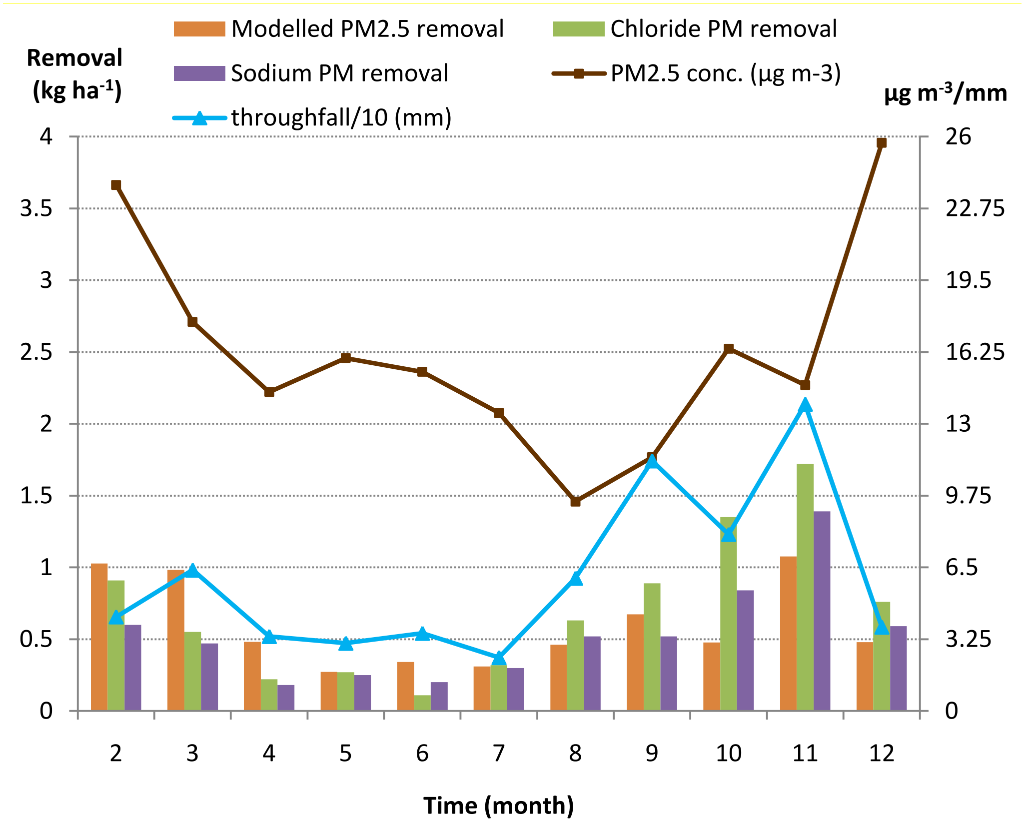

Figure 3.7. Particulate matter removal (kg ha^{-1} period^{-1}), average measured throughfall divided by 10 (mm) and time average airborne $PM_{2.5}$ concentration in 2010. The modelled $PM_{2.5}$ removal is determined using the modelling framework of the present study, the calculated chloride and sodium fluxes are determined using a canopy budget model.

From this graph, also the influence of throughfall and average airborne $PM_{2.5}$ concentrations on the PM removal rates can be derived. Both throughfall and $PM_{2.5}$ concentration influence removal. The correlation between the average throughfall and the removal rates is high (> 0.60). The influence of airborne $PM_{2.5}$ concentration is however also clear. For months 2, 4, 5, 6, 7 and 12, throughfall amount is similar, though the PM removal is more or less double as high for months 2 and 12, this due to higher airborne $PM_{2.5}$ concentrations.

The contribution of resuspension is 76%, which is rather high, but not unrealistic as Hirabayashi et al.(2012) assumed a resuspension fraction of 50%, based on the work of Zinke et al. (1967) and Nowak et al. (2013) obtained an average of 34% with a range of 27-43%. However, one needs to keep into account the differences, discussed in this manuscript, between our model and that of Nowak et al. (2013). On the other hand, the applied parameter values need to be defined more precisely. Hence, it is clear that further research is required to improve the model parameters. Regarding dry deposition, most of the concerned studies have reported deposition velocities per ground area of forest. For our study we obtained a yearly value for $PM_{2.5}$ (based on the yearly total deposition and average $PM_{2.5}$ concentration) of 0.56 cm s^{-1} and a yearly average (of half-hourly deposition velocities) of 0.71 cm s^{-1} with a standard deviation of 0.83 cm s^{-1}. This is within the normal range of 0.1-1 cm s^{-1} reported by Belot et al. (1994) and Pryor et al. (2008). Specifically for this Scots pine stand Neirynck et al. (2007) calculated a deposition velocity for particulate NH_4^+ as fraction of $PM_{2.5}$ of 1.2 cm s^{-1} from September 1999 to October 2000 and of 1.5 cm s^{-1} from January till March 2001. These values are about double as high as ours, though a different approach was used to obtain their values and they were only valid for the NH_4^+ fraction of $PM_{2.5}$. Deposition velocity values for Scots pine mentioned in the review by Petroff et al.(2008) range from 0.15 to 4 cm s^{-1}, although this is for different particle sizes. There is thus still large variation in reported deposition values and more research on this matter may still be needed. Additionally, an accurate size distribution of the PM considered needs to be known, which is not the case here, to calculate and use more precise deposition velocities as a function of particle diameter and wind speed. Regarding model uncertainty, the exact size of uncertainty is impossible to define as no uncertainty intervals are known for all input and parameter values. It will however for sure be considerable with a roughly estimated deviation of 15-50% for the final $PM_{2.5}$ removal. For a better understanding of CIPAM, its assets and limitations, the influence of parameters wind speed, precipitation, $PM_{2.5}$ concentration and LAI was assessed by altering these parameter values for the Scots pine stand in 2010. This sensitivity analysis is given in supporting information section 3.5.5, pg. 84.

A qualitative summary is given here. The influence of wind speed on canopy evaporation is minimal in this case as the high humidity after rain events counteracts its beneficial effect on evaporation and irradiation is much more important, this especially for the temperate humid climate at the Scots pine stand. PM deposition obviously increases as wind speed is heightened though the relative share in resuspension also increases over time, resulting in an overall lower increase in removal for higher wind speeds. Increasing precipitation, while maintaining the same rain pattern, increases PM removal and decreases resupension, although this effect diminishes considerably if precipitation is already high. An increase in PM concentration, results in a linear increase of deposition, resuspension and removal. The resuspension share remains however constant as the wind speed does not vary. More leaf area per surface area leads to a considerable drop in average wind speed within the canopy, this in a logarithmic trend. As canopy evaporation is not strongly influenced by wind speed changes, a higher LAI leads to less throughfall but this is less pronounced the bigger the LAI is. Regarding PM processes, dry deposition increases in a logarithmic manner while resuspension increases logarithmically though in a lower manner. As a result, the share of resuspension decreases while LAI increases. The beneficial effect of more surface area for PM deposition is counteracted by a decrease in wind speed. As a result, PM removal increases logarithmically for increasing LAI values.

3.3.2 Predictions for future scenarios until 2030

First we discuss the results of the current scenario (no change in PM concentration), in order to define the influence of the change in forest growth. The most important variable of the forest in this context is LAI, its change over time is depicted in Figure 3.8. Note that the number of trees is assumed to stay the same (no management or dieback). The average LAI of 2010 was calculated as 2.17. LAI dropped slightly at the beginning, then increased and further on remained quasi constant at 2.3. Its increase might be attributed to canopy closure as there was still a gap fraction of 43% in the period 2007-2008 (Op de Beeck et al., 2010b). Dry deposition (DD) and removal (RM) of $PM_{2.5}$ follow the same pattern as LAI, although the relative increase in DD and RM is less pronounced compared to the LAI increase. This is mainly due to the fact that increasing LAI reduces the wind speed within the canopy, which is a negative feedback on deposition and removal. For more information see supporting information, section 3.5.5.4, pg. 87.

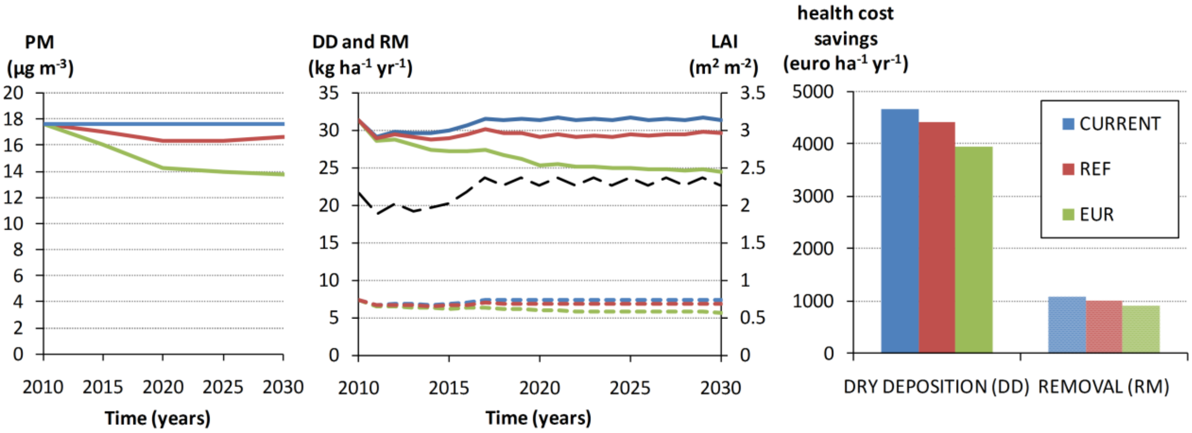

Figure 3.8. The change of airborne $PM_{2.5}$ concentrations (left graph), the associated dry deposition (DD, full lines) and removal (RM, dotted lines) of PM and the leaf are index (LAI, black dotted line) as the forest grows over time (middle graph). This all is shown for the 'CURRENT' (blue) scenario (PM concentration unchanged since year 2010) and for the two future scenarios 'REF' (red), a business-as-usual scenario, and 'EUR' (green), a scenario where environmental European guidelines are followed. In the last graph DD and RM are presented in a monetary unit (based on the value of 150 euro health costs saved kg^{-1} $PM_{2.5}$ removed), depicted for the three scenarios.

For the two future scenarios, REF and EUR, the $PM_{2.5}$ concentration declines over time (Figure 3.8), which is due to a decrease in secondary PM formation because of a reduction in emission of precursors such as NO_x (transport sector), NH_3 (cattle) and SO_2 (energy and household sectors). This decline is, as such, not caused by a decrease of (primary) PM emission, which only decreases till 2015 but then starts to increase again till 2030 because of a rise in emissions from the industry and energy (coal burning) sectors due to economic growth (Van Steertegem, 2009). This drop in $PM_{2.5}$ concentration is logically more profound for the EUR than the REF scenario.

In the first years, DD and RM decrease for both the REF and EUR scenario mainly in response to the LAI decrease, next to the $PM_{2.5}$ concentration decreases. After that, DD and RM in the EUR and REF scenarios follow the same pattern as the current scenario but reaching lower values. The decrease in PM outweighs the increase in LAI and subsequently DD and RM decrease, still RM absolute in much lower amounts. After 20 years, the relative decrease in DD and RM is quasi identical to the relative decrease in airborne $PM_{2.5}$ concentration. Overall, change in land characteristics and PM concentrations need both to be predicted in order to estimate PM removal.

3.3.3 Associated health/economic benefit

Specific for Flanders, based on hospital stay, work absence and willingness-to-pay to avoid health damage costs, the health benefit of $PM_{2.5}$ removal can be converted to an estimated average monetary values of 150 euro kg^{-1} $PM_{2.5}$ removed, while this is only 25 euro kg^{-1} in case of $PM_{2.5\text{-}10}$, often called PM-coarse (De Nocker et al., 2010; Liekens et al., 2013b). The derivation of this value in literature is summarized in supporting information section 3.5.6, pg. 89. As the site is situated close to populated areas (see section 1.6, pg. 12) and the region Flanders for which the number is valid, is a densely populated area, this validates to a certain extent the use of a single estimated value as an approximation. We applied the value to our case study results (see Figure 3.8). For the year 2010 this results in a benefit of 1107 euro ha^{-1} yr^{-1} for removed $PM_{2.5}$, compared to 4763 euro ha^{-1} yr^{-1} if only deposition without resuspension would be considered. Over the period 2010-2030, an average range of 915-1075 euro ha^{-1} yr^{-1} is obtained for $PM_{2.5}$ removal for the different future scenarios; the lowering in $PM_{2.5}$ concentration due to emission legislation, decreases its removal by the Scots pine stand. In 2030 a larger difference is obtained: 853 euro ha^{-1} yr^{-1} for the EUR scenario compared to 1093 euro ha^{-1} yr^{-1} for the current scenario. Comparing these values with a rental price of 143.6 euro ha^{-1} yr^{-1} (based on the selling price for the Scots pine stand of 16000 euro ha^{-1}, obtained from the current owner Agency of Nature and Forest, and on a local land buy to rent price ratio) illustrates for all scenarios the for now underrating by society of this ecosystem service.

3.3.4 Future perspectives

Firstly, besides the perspectives mentioned here, the limitations and assumptions (see supporting information Table 3.3) can be elucidated through additional research. Secondly, CIPAM results should be validated with more experimental results. Thirdly, the model can be adapted to other tree species for further improvement and validation. Fourthly, the model can be extended to other (gaseous) pollutants besides PM, such as ozone, sulfur dioxide, etc. In addition, dry deposition of atmospheric particles is, besides wet deposition via rainfall, an important pathway for relevant chemical compounds (e.g. nitrogen compounds), which do not only affect forest growth but also alter global biogeochemical cycling, water and soil pollution (Fowler et al., 2009). Concerning nitrogen deposition for the Scots pine stand, studied in this study, Neirynck et al. (2007) calculated that dry deposition of the particulate NH_4^+ and NO_3^- comprised in $PM_{2.5}$ was already responsible for 20% of the total, showcasing the importance of this pathway for nitrogen input. In that field of science so-called canopy budget models are mostly applied to derive removal (in that context called deposition) and canopy exchange of different compounds from measured data of throughfall and wet deposition, but they are inapt for predictive purposes (Adriaenssens et al., 2013; Hansen et al., 2013; Staelens et al., 2008). CIPAM can in fact be seen as a predictive canopy budget model which is only suited for removal of PM. It does not account for gaseous compounds and does not allow for canopy exchange, although the model might be extended for these purposes.

Of the studied Scots pine stand, at maximum only 15% of the wet and dry nitrogen deposition (also including deposition of gases) was estimated to be taken up by the canopy (Neirynck et al., 2007). Nonetheless, not considering interactions on the vegetation surface between water, PM and the vegetation itself, is an important limitation of the proposed model, which should be kept in mind. If rates of these processes are known, they should be integrated into the modelling framework. Considering these interactions, resuspension and removal could change considerably. CIPAM may, however, be further used as a tool to study these interactions as it generates half-hourly water and PM amounts on plant surfaces per canopy layer.

3.4 Acknowledgements

We want to express our special gratitude to Charlotte Vanpoucke, Line Vancraeynest, Jordy Vercauteren, Frans Fierens, the Flemish Environment Agency (VMM) and the Belgian Interregional Environment Agency (IRCEL-CELINE) for providing the airborne PM data, and the Research Institute for Nature and Forest (INBO) for supplying meteorological and flux data on the Scots pine stand. The authors also gratefully thank Bert Gielen and Marilyn Roland for data and discussion on the 2010 stand characteristics. We also like to thank David Nowak, Pilar Llorens and Satoshi Hirabayashi for scientific communication on this topic.

3.5 Supporting information

The supporting information gives additional information on limitations and assumptions of the presented modelling framework (section 3.5.1), additional information on ANAFORE (section 3.5.2), Scots pine stand input data (section 3.5.3), the R-ratio (section 3.5.4) and sensitivity analysis (section 3.5.5).

3.5.1 Limitations and assumptions

Table 3.3. Assumptions and limitations of the submodels of the presented modelling framework.

1	The modelling framework is only adapted to rainfall as a precipitation process.
2	No freezing of the water is taken into account.
3	Specific storage capacity can vary in function of wind speed (Llorens and Gallart, 2000), humidity and (foliage) age(Adriaenssens et al., 2010), but this was not accounted for.
4	Forest edge effects were not considered in the model (Wuyts et al., 2008).
5	Sweeping of trees was not considered.
6	Rainfall intensity may influence storage capacity, drip and thus washoff but this is not considered.
7	No spatial or temporal PM concentration change is considered within a time step.
8	We consider PM and water on foliage as inert: no PM aggregation, no uptake of water or PM by foliage, no encapsulation of PM,...
9	No splash evaporation is taken into account.
10	Deposition is influenced by the foliage surface area but not by that of the other plant parts (stem, branches, cones), irrespective of the amounts on the foliated branches.
11	The time needed for water and PM transport is not accounted for. For example, the delay of throughfall water compared to rainfall is not included.
12	The influence of plant surface wetness/humidity on dry deposition is not accounted for.
13	Occult deposition by water droplets in mists or clouds is not included.
14	Precipitation interception and PM deposition by foliage and branches of dead layers are not considered.
15	Wind speed per layer is the average of the top and the bottom of the layer.
16	The distribution of water over the tree is assumed to be equal between shaded and sunlit parts and the potential evaporation per layer is the surface-weighted average of both.
17	Wind turbulence and its effect on the modelled processes is not accounted for.
18	The ratio of LAI to WAI, the R-ratio, is considered constant all the time.
19	Stemflow was not considered in the interception model.
20	Deposition and thus also resuspension of PM on the trunk are not considered.
21	The amount of PM which is washed off is the amount present multiplied with the ratio of drip to drip+storage amounts, hence we assume that the PM is evenly suspended in the water layer.

3.5.2 Additional information on the ANAFORE model

The ANAFORE model simulates carbon (C), water (H_2O) and nitrogen (N) fluxes, tree growth, and wood tissue development of cohorts of trees in a stand, this all in response to management, climate and stand characteristics. The model follows a bottom-up approach: leaf level processes such as photosynthesis and transpiration are simulated at a half-hourly time step for sunlit and shaded leaves of crown leaf layers and implemented into a daily-operating tree architecture and C allocation module. This model allows one to subdivide the tree population into different tree cohorts with different characteristics. A complex soil sub-model was added later on to the model (Deckmyn et al. 2011). No regeneration is included in the used version of ANAFORE though note that the acid soil of coniferous forest may somewhat (indirectly) inhibit plant growth. The ANAFORE model is sensitive to climate change in a complex way, e.g. CO_2 has direct influence on photosynthesis and stomatal conductance while temperature affects many processes (transpiration, photosynthesis, soil processes, respiration).

ANAFORE requires 132 input parameters for a tree species, and 124 parameters concerning soil functioning. These are the parameters that can be included in a Bayesian procedure routine, introduced by Deckmyn et al. (2009) and based on the method described by Van Oijen et al. (2005) making it in fact a hybrid model: mechanistic and empirically based. Furthermore, half-hourly, daily or monthly values of temperature, precipitation, radiation, wind speed, relative humidity, ambient CO_2 concentration, stand inventory, forest management and soil characteristics are necessary input values.

We must note that ANAFORE is a highly detailed and parameter-rich model and is therefore less suited if not sufficient input data are available (van Oijen et al., 2013). However this is not an issue in this case, as the model has already been applied and validated to the here considered Scots pine stand (Deckmyn et al., 2011, 2008).

3.5.3 Scots pine stand input for ANAFORE

Table 3.4. Scots pine stand characteristics and inputs for the ANAFORE model. The organic and mineral soil C is distributed over the different soil layers as was done in Deckmyn et al. (2011)· Quality of the data is shown by colour (from ideal (white) to assumption (black)).

Parameter	Unit	Value	Year	Source
Tree number	trees ha^{-1}	361	2010	(Gielen et al., 2013)
Initial tree height	m	21.2	2008	(Gielen et al., 2013)
Crown depth	m	4.5	2008	.(Gielen et al., 2013)
Start living crown	m	16.7	2008	(Gielen et al., 2013)
Crown radius	m	2.85	2010	Was 2.43 m in 1996, as an estimation 2.85 m is here considered
Stem biomass	kg C $tree^{-1}$	213	2010	(Gielen et al., 2013)
Coarse roots	kg C $tree^{-1}$	40.6	2010	(Gielen et al., 2013)
Fine roots	kg C $tree^{-1}$	5.56	2010	(Gielen et al., 2013)
Foliage	kg C $tree^{-1}$	5.56	2010	(Gielen et al., 2013)
Stem radius	m	0.164	2010	(Gielen et al., 2013)
Initial heart wood	%	1	2010	Fixed
Initial parenchym filled	%	99	2010	Fixed
Tree age	year	80	2010	(Gielen et al., 2013)
Foliage nitrogen	kg N $tree^{-1}$	0.21	2010	Calculated using ratio of Neirynck et al. (2008) for 1999: 1.88% of dry matter
Soil, organic	ton C ha^{-1}	31.9	2010	(Gielen et al., 2013)
Soil, mineral	ton C ha^{-1}	68.2	2010	(Gielen et al., 2013)
Maximum tree height	m	22	2010	Estimation, measured tree height was 21.2 in 2010 (Gielen et al., 2013) and was more or less the same as that of 21.4 in 2001-2002 (Nagy et al., 2006), validating this maximum tree height

3.5.4 The R-ratio

The ratio R of leaf area index (LAI) per wood area index (WAI) is 9 for older Scots pine stand according to Bréda (2003), interpreting the results of Walter and Himmler (1996). However this variable is dependent on stand characteristics and not a constant according to age as from the study of Deblonde et al. (1994) different R-ratios may be derived for different red and jack pine stands with 100% stand closure, in which these differences are not only defined by age. They are presented in Table 3.5.

Table 3.5. Stand characteristics and stem to leaf area ratio for various red and jack pine stands with 100% stand closure (as mentioned in table 1 of Deblonde et al.(Deblonde et al., 1994)). The R-ratio was considered as the inverse of latter parameter. The R-ratio of the first-mentioned Red pine stand was eventually used as its characteristics were most similar to the here studied Scots pine stand.

tree\unit	age yr	density Stems ha^{-1}	Mean DBH cm	LAI $m^2\ m^{-2}$	Basal area $m^2\ ha^{-1}$	Stem:leaf area ratio $m^2\ m^{-2}$	R-ratio $m^2\ m^{-2}$
Red pine	60	430	29.5	2.9	29.3	0.086	11.62
Red pine	60	850	25.5	4.9	43.3	0.099	10.10
Red pine	60	1269	24	6.2	57.3	0.119	8.40
Jack pine	30	1299	13	1.6	17.3	0.194	5.15
Jack pine	30	1368	16.1	2.2	28	0.184	5.43
Jack pine	30	1510	13.2	1.7	20.7	0.229	4.37
Jack pine	30	2705	9.8	1.5	20.5	0.33	3.03
Jack pine	60	781	16.9	2	17.5	0.10	10

Interpreting the results of that article, striking relationship are revealed. The R-ratio increases as stem density decreases (Pearson correlation of -0.88) and stem thickness (Pearson correlation of 0.89) increases. More research is however needed on this matter. In our case, the first mentioned red pine stand resembles most the stand studied in this research, see Table 3.4, pg. 82, and therefore its R-ratio was selected in this study.

3.5.5 Sensitivity analysis

Only for the modelling framework, thus not for ANAFORE, and the year 2010 some parameter values were individually altered to analyze the sensitivity of the model to input changes.

3.5.5.1 Influence of wind speed

An important driver for water and particulate matter processes is wind speed. The wind speed pattern within the canopy is here not altered, but the value of the wind speed above the tree is multiplied with a certain factor. As can be seen from Figure 3.9, the average wind speed within the canopy is quasi constantly 64-65% of the wind speed above the trees.

Canopy evaporation increases minimally and thus throughfall decreases minimally under varying wind speed, as can be seen from Figure 3.9. Theoretically, an increase in wind speed reduces the water boundary layer height around the leaf, which leads to a faster transport of water to the atmosphere. The driver for canopy evaporation, the transport of water from the leaf surface to the atmosphere, is the extent of under saturation of the air humidity, the vapor pressure deficit. Reason why this influence is minimal in our case is due to a high humidity values and thus a low vapor pressure deficit. The climate of the Scots pine stand is a humid climate with a high yearly average relative humidity of 79% in 2010 (based on half-hourly values). Next to that, canopy evaporation only occurs if water is present on the foliage, which is just after rain events, however then humidity is high, generally > 90%. The influence of sunlight on canopy evaporation is therefore much higher than that of wind speed. These conclusions can also be found in literature for evapotranspiration for humid climates (Irmak et al., 2006; Tabari and Hosseinzadeh Talaee, 2014). Note that in these literature sources transpiration is also considered which does not only occur just after rain events.

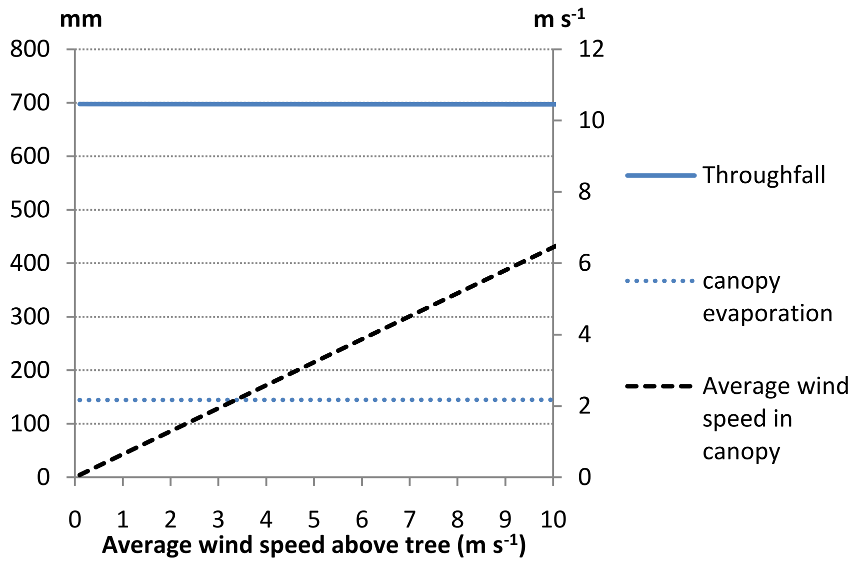

Figure 3.9. Water fluxes for the Scots pine stand in 2010 and average wind speed in the canopy as a function of wind speed above the canopy.

On the other hand, the $PM_{2.5}$ balance is strongly influenced by the wind speed (Figure 3.10). Over the complete tested wind speed range, sigmoidal curves are obtained for dry deposition and resuspension with an inflection point around 6 m s^{-1}. For wind speed values up to 3-4 m s^{-1}, the increase is exponential/quadratic. At a wind speed of 1 m s^{-1} a deposition of 10 kg ha^{-1} hr^{-1} is obtained and at 3 m s^{-1} this is around 90 kg ha^{-1} yr^{-1}. The relative share of resuspension increases over time in a logarithmic manner from 18% at 0.1 m s^{-1} to 67% at 1 m s^{-1} and up to 80% at 2.7 m s^{-1}. Removal also increases with a maximum removal of 64.74 kg $PM_{2.5}$ ha^{-1} yr^{-1} at 10 m s^{-1}. However this increase is much more significant for lower wind speeds.

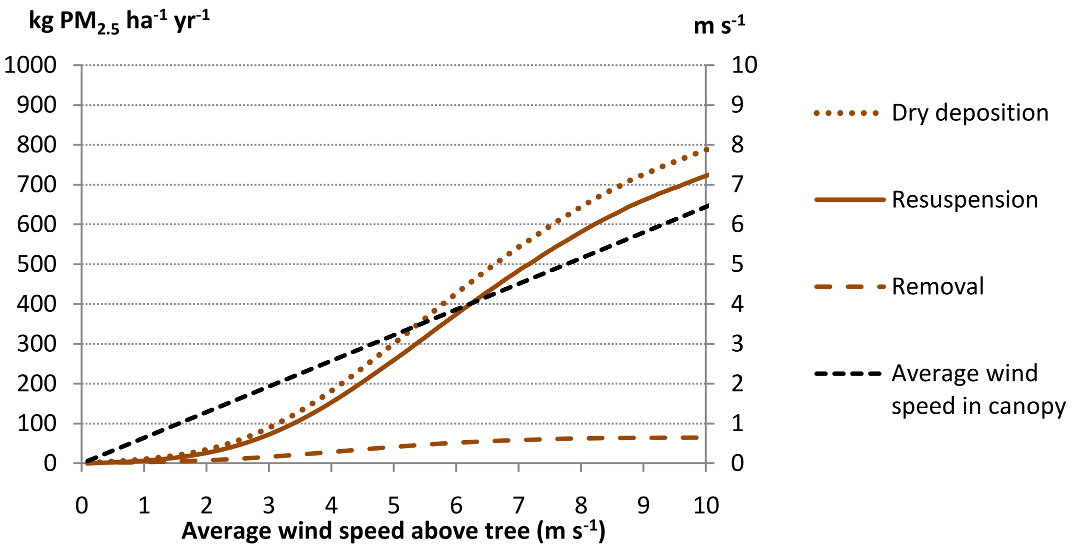

Figure 3.10. Particulate matter ($PM_{2.5}$) fluxes for the Scots pine stand in 2010 and average wind speed in the canopy as a function of wind speed above the canopy.

3.5.5.2 Influence of precipitation

Here, yearly rainfall is multiplied with a certain factor and thus also the amount of rainfall events, in other words, varying rainfall amounts but with the same pattern of 2010. As precipitation increases, throughfall increases in a near linear manner (Figure 3.11). Canopy evaporation also increases with higher rainfall but this is in a logarithmic way. The flattening of this increase is expected as intercepted rainfall will more and more just end up dripping from the foliage as the canopy is saturated since the water storage capacity is considered to be constant under different meteorological conditions. Precipitation as such does not influence the modelled dry deposition. However precipitation is important for removal (washoff) and resuspension. Since deposited $PM_{2.5}$ ends up as either one (the amount remaining on the trees is negligible), their curves are mirrored. Resuspension and removal both have a sigmoidal change as a function of precipitation with an inflection point around 150-200 mm. Overall, increasing precipitation, increases PM removal and decreases resupension, although this influence decreases if precipitation is already high. Alterations would be different if the rain pattern (also) changes.

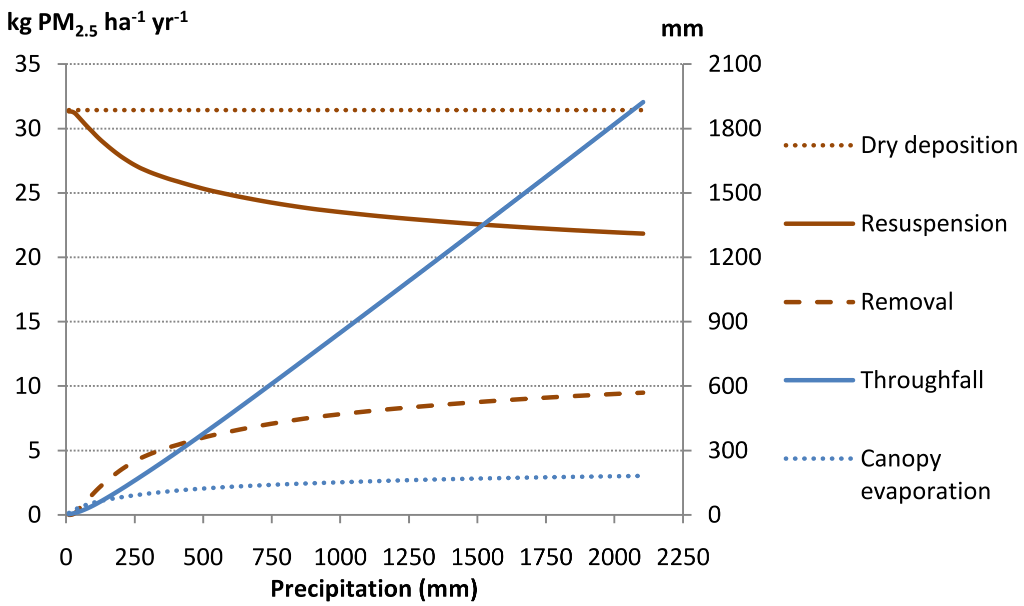

Figure 3.11. Particulate matter ($PM_{2.5}$) and water fluxes for the Scots pine stand in the year 2010 as a function of yearly rainfall with a pattern similar to that of 2010.

3.5.5.3 Influence of $PM_{2.5}$ concentration

If yearly $PM_{2.5}$ concentrations increase ($PM_{2.5}$ pattern is constant), dry deposition will increase linearly (y=1.7807*x). Hence, also resuspension (y=1.36*x) and removal (y=0.42*x) will increase (Figure 3.12). The resuspension share remains constant at 76%. We can conclude that under higher $PM_{2.5}$ concentrations, the forest will remove more $PM_{2.5}$, i.e. 0.42 kg ha^{-1} yr^{-1} per 1 µg m^{-3} $PM_{2.5}$ concentration increase in this case. So the higher the PM pollution, the higher will this PM removal ecosystem service be.

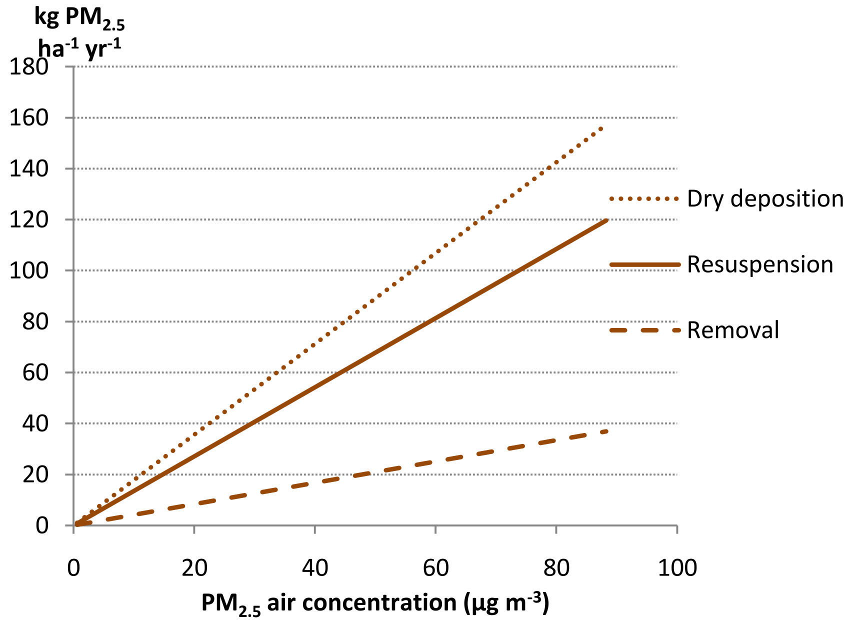

Figure 3.12. Particulate matter ($PM_{2.5}$) fluxes for the Scots pine stand in the year 2010 as a function of yearly airborne $PM_{2.5}$ concentration with a temporal pattern similar to that of 2010

3.5.5.4 Influence of leaf area (index)

Here the influence of leaf area and leaf area index on the outcome of the modelling framework is assessed. In this sensitivity analysis we varied LAI from 0.1 to 17.4 m^2 m^{-2} (such high LAI values have been reported) to assess a broad range. The first obvious conclusion is that the average wind speed within the canopy decreases over time with increasing LAI in a logarithmic way (flattening towards the end), in response to the exponential function of the wind speed calculation. Looking at the water fluxes, throughfall decreases and canopy evaporation logically increases, both in a logarithmic fashion, if LAI increases (Figure 3.13). This is not due to a drop in wind speed, as can be derived from results of section 3.5.5.1, pg. 84, within the canopy but due to an increase of storage capacity.

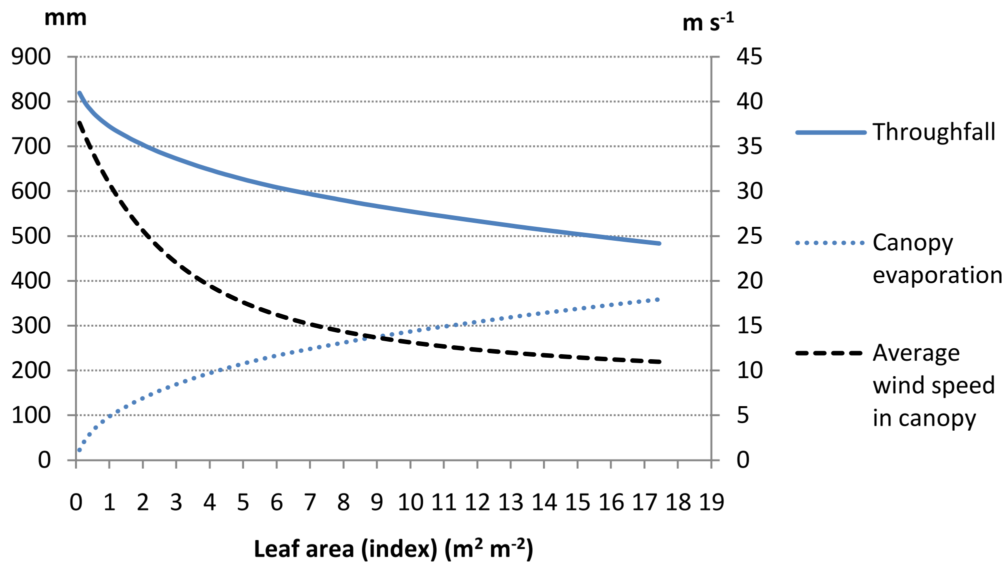

Figure 3.13. Water fluxes and average wind speed in the canopy for the Scots pine stand in the year 2010 as a function of varying leaf area (index).

Dry deposition, directly linked to wind speed, increases in a logarithmic manner (Figure 3.14). Resuspension increases logarithmically but starts to decrease with a low slope at LAI 4-5. Besides, the share of resuspension decreases over time. This is both considered to be influenced by the decrease in wind speed. As a result, PM removal increases logarithmically for increasing LAI values until a value of 2, and then increases linearly. Overall, we can conclude that increased leaf area (index) increases the modelled dry deposition and removal of PM for realistic leaf area (index) values. Please note that the removal and resuspension of PM do not decrease the overall ambient PM concentration in this approach, otherwise the results might be quite different.

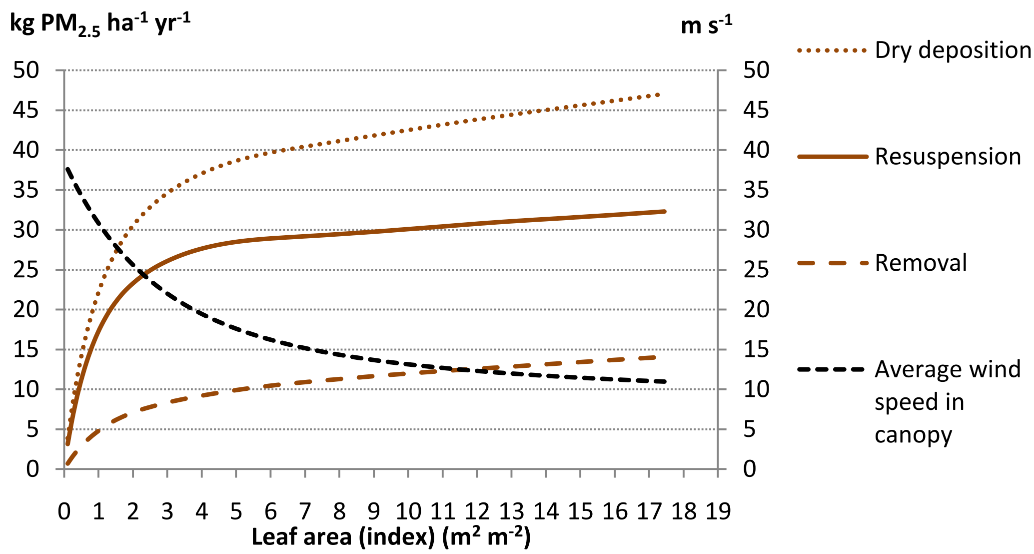

Figure 3.14. Particulate matter ($PM_{2.5}$) fluxes and average wind speed in the canopy for the Scots pine stand in the year 2010 as a function of varying leaf area (index).

3.5.6 Derivation of the health cost value per kg $PM_{2.5}$ removal

The monetary value per kg $PM_{2.5}$ (150 euro per kg $PM_{2.5}$) applied in our study was retrieved from the Flemish study of Liekens et al.(2013b) in which this value is used to represent the removal of $PM_{2.5}$ by vegetation in Flanders. Originally, the value is computed by De Nocker et al. (2010) to represent the external costs of emission by industry of transport in Flanders.

De Nocker et al. (2010) present a framework to link emission of different pollutants with monetary costs specifically valid for Flanders. **Firstly**, a change in emission of $PM_{2.5}$ is linked to a change in air concentrations (µg m^{-3}) over Flanders. This is calculated for a 20% decrease in emission by the industry and transport sectors, assuming this represents well enough a marginal change, using models, such as the BelEUROS model, mentioned in the manuscript.

Subsequently, the impact of the concentration changes on human health is assessed. The considered dose-effect detrimental effects of $PM_{2.5}$ on the human health are:

- Long-term effects during constant exposure
    - New cases of chronic bronchitis
    - Early mortality
- Short-term effects
    - Premature death
    - Hospitalization due to respiratory and cardio-issues of the local population
    - Use of bronchodilators by children and adults
    - Lower airways health issues with children and adults
    - Days with lessened activity/restricicted activity days
    - Days with minor lessened activity/minor rads
    - Days lost due to work absence/work loss day

For quantification, global dose-effect values of epidemiological studies are used (specific studies are given in De Nocker et al. (2010)) the effects are calculated based on changes in yearly average $PM_{2.5}$ concentration and a linear response is considered.

Thirdly, the different costs, valid for the year 2009, associated with the above mentioned detrimental health effects are computed. The following costs are accounted for:

- Costs for medical care and medication (specific for Flanders)
- Costs of loss in productive and free time because of work absence or bedridden (specific for Flanders)
- Willingness to pay to lower risks of disease and early mortality (specific for Europe)

Finally, dividing the monetary value associated with a $PM_{2.5}$ concentration change by the emission change results in a ratio of euro health costs per kg $PM_{2.5}$ emission. For more information, regard the study of De Nocker et al. (2010).

Chapter 4 Environmental impact assessment and ecosystem service valuation of a forest ecosystem under different future environmental change and management scenarios

Redrafted from:

Schaubroeck, T., Deckmyn, G., Giot, O., Campioli, M., Vanpoucke C., Dewulf J., Verheyen, K. Muys, B., To be submitted. Environmental impact assessment and ecosystem service valuation of a forest ecosystem under different future environmental change and management scenarios.

Abstract

In order to achieve a sustainable development we cannot only manage our human/industrial system in a sustainable manner but also ecosystems. To achieve the latter goal, we have 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 scenarios. In literature such studies have been performed though they cover a limited amount of services and the tools used to compare between them always have an incorporated subjective aspect and represent the final result in a non-tangible unit such as 'points'. In this study we want to resolve these matters, and assessed the environmental impact (on human health, diversity and natural resource) and performed an ecosystem service valuation based on monetary values (including ecosystem disservices with associated negative monetary values) on an ecosystem. We applied these approaches to a Scots pine stand from 2010 to 2089 for a combination of three environmental change and three management scenarios. The addressed flows/ecosystem services, including disservices, are: particulate matter (PM) removal, freshwater loss, CO_2 sequestration, wood production, NO_x emission, NH_3 uptake and nitrogen pollution/removal. The environmental change scenarios include alterations in temperature, precipitation, nitrogen deposition, wind speed, PM concentration and CO_2 concentration.

The monetary valuation highlights the importance of services provided by the forest, with a total yearly average of 361-1242 euro ha^{-1} yr^{-1}. $PM_{2.5}$ (< 2.5 µm) removal is the key service with a value of 622-1172 euro ha^{-1} yr^{-1}. Concerning environmental impact assessment, with net CO_2 uptake the most relevant contributing flow, a prevention in loss 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 as CO_2 sequestration and PM removal are higher for this one, latter induced by a higher leaf area index.

4.1 Introduction

To obtain an environmentally sustainable future for mankind, we can control our human/industrial system by reducing its environmental impact caused by emission of harmful compounds and resource extraction, increasing its productivity/efficiency and by remediation of environmental damage. However, next to only controlling our own direct actions on the environment, we may control/manage ecosystems in a manner so that they also aid us in achieving these sustainability efforts, e.g. provisioning of more renewable resources, in the best way possible. One of the most relevant terrestrial ecosystem types is the forest ecosystem. Forests covered 31% of total land area in 2010 (FAO, 2010), provide valuable goods and services to us, such as the provisioning of wood, and may mitigate climate change, e.g. through the well known sequestration of carbon dioxide (Pan et al., 2011).

Next to the direct influence of mankind on forests through harvest and management practices there is also the influence of changing environmental conditions, such as climate change, which is important to account for, and their interactions with mentioned management practices (FAO, 2012). Future environmental conditions are however not exactly known. Different scenarios may occur and some long-term ones were specifically predicted by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2014, 2000). These should be used (indirectly) as inputs for forest models.

As already said, providing wood is just one of the functions that forests fulfill. Since the 1950s, forest management in many regions over the world moved toward multifunctional management aimed at optimizing several services including wood production, soil and water protection, recreation and conservation (Luyssaert et al., 2010; Quine et al., 2013). A well known concept to assess the different goods and service is the one of 'ecosystem services', an anthropocentric concept. Ecosystem services are described as the direct and indirect contributions of ecosystems to human-well being (De Groot et al., 2012), well described in the Millenium Ecosystem Assessment reports (2005). By consequence one can account for these different aspects/services of forests through a set of indicators and compare their obtained values, possibly using a multi-criteria analysis (MCA) methodology (Hails and Ormerod, 2013). Overall, there is a need for solid methods that account for the combined influence of different environmental change and management scenarios on ecosystem functioning, the environment and mankind, latter via a change in provided ecosystem services (Figure 4.1).

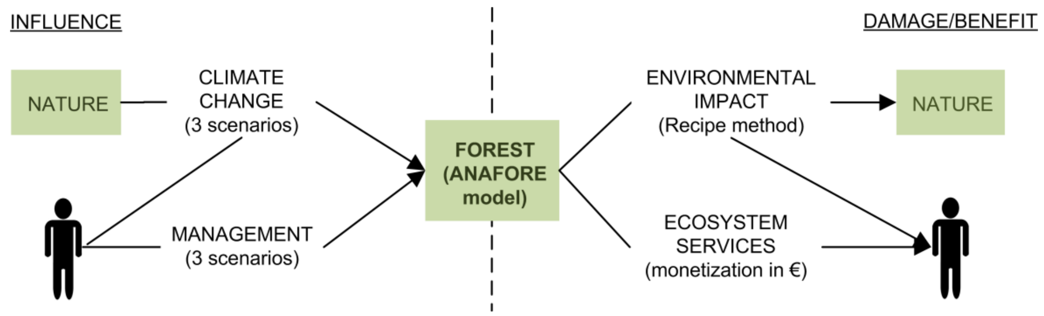

Figure 4.1. Overall scheme of the influence of mankind and nature on forests and the subsequently indirectly induced damage and benefit to themselves by that influence. Practically in this study, we set up a framework to attempt to quantify these relationships and effects using different scenarios and methods, shown between brackets. The Recipe method is an environmental impact assessment method (Goedkoop et al., 2009).

In practice such result can be obtained via a framework that includes the evaluation of a set of calculated indicator values for the different scenarios, possibly obtained from forest models (Wolfslehner and Seidl, 2010).

Because this is a relevant topic, different studies of that kind have already been performed. Table 4.1 gives an overview of 5 known studies and their properties. The goals of this study are to also perform such a study on a Scots pine stand, located in Belgium (Europe), with only a focus on environmental sustainability and to address certain aspects of it in a better manner, as explained further on in the introduction.

A first aspect we want to better address is the approach to compare the different scenarios in terms of the various services presented in different units which are induced by them. This is a difficult issue. 2 out of 5 studies of Table 4.1 still do this based solely on individual's personal insight, without the use of an additional methodology (Lasch et al., 2010; Temperli et al., 2012). Such an approach might be plausible if not too many indicators are considered but is nonetheless strongly based on subjective opinions. Duncker et al. (2012) make use of principal component analysis to aid in their judgment though it does not result in a single outcome. An additive utility model is used by Fürstenau et al. (2007) which includes the weighting of the indicators in different manners based on specific stakeholder group, e.g. environmental organization, priorities and expert knowledge, and adding them up. Seidl and Lexer (2013) use a complex framework that is partially based on selected weighting, some by stakeholder groups, of indicators. A single score is obtained for both 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. There is thus a need for methodologies which result in a single or a small set of outcomes based on more/only objective calculations. Next to that, the outcomes of the applied multi-criteria analysis methods have no units (they are represented as 'scores' or 'points'), expressing no real tangible quantity, giving no message regarding the impact of a scenario (choice).

Table 4.1. Overview of considered studies in which the combined influence of management and environmental change scenarios on the provisioning of goods and services by forest ecosystem was investigated. Duncker et al. (2012b) did not assess the influence of environmental change, though was considered because of their broad accounting of ecosystem services. T: temperature; PM: Particulate Matter. SRES IPCC: Special Report on Emission Scenarios of Intergovernmental Panel on Climate Change. DBH: Diameter at Breast Height; MCA: Multi-Criteria Analysis.

	Duncker et al. (2012)	**Fürstenau et al.** (2007)	**Pizzirani et al.** (2010)	**Temperli et al.** (2012)	**Seidl and Lexer** (2013)	**This study**
Site(s) with tree species	Virtual Central European (German) forests; Spruce and beech	Kleinsee study area in East Germany; mainly Scots pine	Inshriach forest in Scotland; mainly Scots pine	Black forest in Germany; mostly Norway spruce initially	Austrian Federal forests; mainly Norway spruce	2-hectare stand in forest 'de Inslag' in Belgium; Scots pine
Management scenario's	5: from none to intensive	6: from none to intensive;	5: from none to intensive	5: 2 classic and 3 adaptive management regimes	2: Bussiness as usual and climate change adapation strategy	3: intensive with different thinning regimes
Environmental change Scenario's	/	2: modelled with climate models; change in T, precip. and CO_2	1: increasing intensity of biotic threats	3: modelled with climate models, only monthly T and precip.	3: modelled with climate models, only T and precip.	2: partially based on SRES IPCC, change in T, precip., CO_2, N-depostion, PM concentration
Timber/ Biomass production	Timber production (monetarized); DBH considered	Income from timber production	Cost and value of wood production	Timber production	Productivity stem wood	wood production with value; DBH considered
Groundwater recharge	As such	As such	/	/	/	Loss in recharge due to evapotranspiration
Biodiversity	Abundance of dead wood, large DBH trees, tree species div. & woodland key habitats	Coarse woody debris and deciduous trees	expert opinion and analysis of various biodiversity sub-indicators	tree species diversity and stand structure complexitity via stand maturity index)	tree species diversity and standing deadwood volume	/
Carbon sequestration/ stock changes	Yes	forest and wood products (incl. end of life)	/	/	Carbon storage	Yes
Influence on water quality	Nitrate leaching	/	/	/	/	N-pollution/removal

Continuation Table 4.1

	Duncker et al. (2012)	**Fürstenau et al.** (2007)	**Pizzirani et al.** (2010)	**Temperli et al.** (2012)	**Seidl and Lexer** (2013)	**This study**
Others	- Soil fertility -acidification/nu-trient loss	/	- Recreational value - Employment value - Carbon stocks	/	Disturbances (bark beetle, storm, snow breakage)	- Removal of PM - processing of NH_3 - emission of NOx
Comparisson method	Judgement, aided by principal component analysis	MCA method with weigthing by stakeholder	Judgement	Judgement	MCA with different weighting scenarios based on stakeholders	Monetization of ecosystem services and environmental impact assessment
Model(s) used	Hybrid model W+, (Yue et al., 2008)	Process-based C4 model (Bugmann et al., 1997)	No forest model used but predictions	Process-based forest model LandClim (Schumacher et al., 2004)	Hybrid model PICUS v1.4 (Seidl et al., 2005)	Hybrid model ANAFORE (Deckmyn et al., 2011, 2009, 2008)
Conclusion on best management practice	there is a trade-off between services (biodiversity trade-off with biomass and sequestration)	Depends on stakeholder group which management scenario is the best	Best scenario is the one with partial natural regeneration and partial cultivation	Trade-offs between forest resource use and environmental objectives	Depends on stakeholder group which management scenario is the best	The least intensive management scenario is the best

In this article two such already developed methodologies are put forward and applied. In the first method, ecosystem services and goods will be valuated through monetization and adding them up to a single monetary amount (Baveye et al., 2013; Broekx et al., 2013; de Groot et al., 2012; Liekens et al., 2013b; TEEB, 2010). Note that this is not an analysis on financial or economic feasibility/profit, a cost-benefit analysis, of a selected scenario, such as has been done by Garcia-Quijano et al. (2005) for climate mitigation through CO_2 uptake. Monetary valuation is not purely objective though it delivers a tangible overall value and no normalization of services is needed. Environmental impact assessment methodologies represent another type of tool in which a small set of indicator values might be obtained, e.g. Garcia-Quijano et al. (2005) use such a methodology to assess land use impact for different forest management practices. This type of method is the second one applied in this study. Latter approach also regards the impact on nature while ecosystems service assessment focuses on the benefit for mankind (Figure 4.1). And an application of such tool will in return allow one to attribute certain environmental impacts to specific management scenarios and their products (in life cycle assessment), e.g. the environmental impact per m^3 of harvested wood if it is managed in an intensive or extensive way. This approach is even more objective and also presents the outcome in tangible units, though possibly not in a single one. Schaubroeck et al. (2013), Chapter 2 (pg. 17), pointed out that in the environmental impact assessment of an integrated system of forest and wood processing, the forest could have the most important share in impact. These authors also provide a framework which allows one to assess the environmental impact better by including uptake of harmful compounds, which will also be used here. Note that other approaches exist to aid stakeholders in selecting the best 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 still a lot of subjective choices need to be made in these and no overall tangible outcome when addressing multiple criteria is obtained.

A second aspect to improve is that a broader range of services needs to be considered, and this in a more realistic manner, to pinpoint the best management strategy under changing environmental conditions (Smith et al., 2013). Most important additional service considered here is particulate matter removal from the air, based on the work presented in Chapter 3, pg. 55 (Schaubroeck et al., 2014). The goods and services addressed in this study will be elaborated on further on.

4.2 Material and methods

4.2.1 Site description

See introduction section 1.6, pg. 12.

4.2.2 Model selection

Difficulties in this field of study are the slow growth and thus response time of forests. Models were therefore developed to help predict their growth. The first were based on empirical relationships in the forest, obtained through measurements, and are called empirical models, for a historical overview see Pretzsch (1999). Empirical models on their own are less reliable to quantify ecosystem responses under unprecedented future conditions, e.g. occurring through environmental change, since no forest responses are measured for these. Process-based models offer a better solution as they are based on more fundamental physicochemical relationships on molecular level in the forest, e.g. the influence of increased CO_2 concentration on photosynthesis and forest growth may be more realistically studied (Fontes et al., 2010). A combination of empirical and process-based modelling is called a hybrid model (Muys et al., 2010). Hybrid modelling can occur through Bayesian parameterization of a process-based model with measured data (Van Oijen et al., 2005).

Except for Pizzirina et al. (2010), which uses no model, almost all other forest sustainability evaluation studies under different management and climate scenarios, apply models that are at least partially process-based (see Table 4.1, pg. 96). However some remarks should be made. In the study of Temperli (2012) the Landclim model does not simulate emissions to the environment. An important part which is lacking in all the used models, except PICUS 1.4 applied by Seidl and Lexer (Seidl and Lexer, 2013), is the modelling of the soil, its microbiota and its processes such as respiration. It is important to assess (fluxes of) soil respiration, denitrification and nitrate leaching. For example, soil respiration of the Scots pine stand accounts for about half of its CO_2 emission (Nagy et al., 2006).

Here, we applied the process-based model ANAFORE, described in section 3.5.2 (pg. 81).

To assess particulate matter (PM) removal, we created a model operating on a halfhourly basis, and integrated it into the ANAFORE model and applied it already to the Scots pine stand studied here, see Chapter 3 (pg. 55). This particular submodel will also be used in our study for $PM_{2.5}$ (PM with a diameter < 2.5 µm) and $PM_{2.5\text{-}10}$ removal. The input needed to run this model, is airborne PM concentrations and wind speed of which the data sources and calculation are given in Chapter 3, pg. 55. The parameter values mentioned in Chapter 3 (pg. 55) for $PM_{2.5}$ removal by Scots pine are used and are also applied for $PM_{2.5-10}$ removal. For the wood area calculation, needed in the interception modelling, we applied the alternative approach, mentioned in section 3.2.3 (pg. 62), in which the branch area is calculated.

4.2.3 Management scenarios

The studies described in Table 4.1 (pg. 96) test management scenarios within a broad spectrum, from no management to intensive, this 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, this to show that different outcomes can be obtained by the framework, even on that level of detail.

For all scenarios, the considered management of the Scots pine stand is a 80 year rotation period, starting from 10 000 planted one-year old trees per hectare after a clear felling of the current pine forest in 2010 until the next clear cut in 2090. 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 3.4 of chapter 3. The distribution over the soil layers is retrieved from a previous run of ANAFORE on the same site. Two year old saplings were planted. In order to initialize the soil conditions after a clear cut in 2010, a preparatory model run was performed for a full rotation length of pine. We consider only one tree cohort in the ANAFORE model, which is reasonable since it is a planted forest.

For all scenarios, after 14 years a tending occurs in which 30% of the trees are cut. The subsequent thinning is different between the scenarios. For the LOW management no thinning occurs. For the MID and the HIGH management, thinning is performed every 5 years, as done 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 this is all the increment. Random trees are cut when thinning. This is a simplification, given that different thinning procedures exist, e.g. thinning from below or from above, but in practice a mix is applied. Only wood from stem and big branches is harvested when thinning. Root, needle and small branch residues are left behind in the forest.

4.2.4 Environmental change scenarios and their parameter values

Three of the discussed studies in Table 4.1 (pg. 96) use different environmental change models to assess different scenarios (Fürstenau et al., 2007; Seidl and Lexer, 2013; Temperli et al., 2012). Pizzirani et al. (2010) just consider increase in biotic threats. Out of the three others, two only consider increase in temperature and precipitation. Besides these two variables, Fürsternau et al. (2007) also take into account an increase in CO_2.

For this study, simulations were performed from 2010 till 2089 with three different environmental change scenarios that aim to capture the possible trends in environmental change: one assuming no change as a reference, the current (CUR) scenario, and two alternative future scenarios. The latter two will be 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 more socio-environmental sustainability. Next to that, another matter to address when specifying environmental change scenarios is if there will be a convergence of different communities, i.e. similar conditions and policies among world-wide communities (IPCC, 2000). Here we consider a heterogeneous/non-convergent world for both MOD and SEV since we deal with more local policies. The environmental change scenarios differed in 6 out of the 8 meteorological and environmental variables driving the model: air temperature, CO_2 concentration, NO_y deposition, NH_x deposition, wind speed and airborne particulate matter concentration, more specifically that of $PM_{2.5}$ and PM_{10}. These environmental changes are applied on a yearly level, except for precipitation and temperature which also vary on a seasonal level. The scenarios are aggregated out of other similar scenarios from different references, this because no single reference provided values for all considered parameters and we wanted to use more site-specific values. For an overview, see Table 4.2.

Table 4.2. The aggregated climate scenarios and the scenarios that are used to model the respective parameters.

	Moderate (MOD)	Severe (SEV)	Values	Reference
Temperature Precipitation Wind Speed	G+	W+	Table 4.3	(van den Hurk et al., 2006)
CO_2 concentration	A2	B2	Figure 4.2	(IPCC, 2001)
N deposition: NH_x and NO_y PM air concentration: $PM_{2.5}$ & $PM_{2.5\text{-}10}$	Europe	Reference	Figure 4.2	(Van Steertegem, 2009)
Overall similar IPCC scenario	A2	B2	/	(IPCC, 2000)

In this study monthly values of weather conditions (temperature, precipitation and radiation) are used as model inputs while yearly for the others. Important to include is interannual variation concerning these weather conditions, this definitely since forest growth in the beginning years is sensitive to weather conditions (Cunningham et al., 2006; Dzwonko and Gawroński, 2002; Taeger et al., 2013a, 2013b). Besides that, as uncertainty of future weather predictions is more dependent on the randomness of years in the beginning, see figure 11.8 of IPCC (2014), only this aspect of future uncertainty was included and e.g. not uncertainty in average amount of temperature increase. This is practically done via the following approach. If one assumes 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 needed as weather input for a run from 2010 to 2089. Fifty random periods are thus created and these serve as weather input for the model runs of the current scenario. This all creates a natural variation in weather conditions. On these random samples a change in temperature and precipitation is superseded to obtain the weather inputs for the two future scenarios. This is not done for irradiation, since future radiation changes are expected to be very minor in Belgium (Campioli et al., 2011), and also not on an average yearly basis for precipitation, as projected yearly changes are small compared to internal variability (i.e., smaller than one standard deviation of 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 CO_2 increase (IPCC, 2014). The considered changes, shown in Table 4.3, are obtained for the moderate and severe scenarios for the region of study till 2090 based on the scenarios G+ (equivalent to B family results) and W+ (equivalent to A family results), respectively, of the Royal Dutch Meteorological Institute (Demarée, 2008; van den Hurk et al., 2006). The latter scenarios and the values for the period 1999-2008 were constructed combining results of global circulation models, regional climate models and local measurements. On average, a warmer climate with wetter winters and drier summers compared to the current climate is predicted for the moderate and even more for the severe scenario (Table 4.3).

Wind speed only influences particulate matter removal and evapotranspiration of water, is forecasted to increase for the G+ (moderate) and W+ (severe) scenario, see Table 4.3. 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 4.3. Considered changes for 2100 compared to 1990 in precipitation and temperature 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 CO_2 concentrations, the current scenario concentration was set constant at 390.103 ppmv (IPCC, 2001). Future CO_2 projections for moderate (B2) and severe (A2) scenarios foresaw a gradual CO_2 increase up to 585 and 762.5 ppmv, respectively, in 2090, based on averages of the reference scenario of models ISAM and Bern-CC (IPCC, 2001) (Figure 4.2).

Future projections of nitrogen deposition and particulate matter (PM) concentrations (Figure 4.2) 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 up until 2030 (Van Steertegem, 2009). For the moderate scenario the 'Europa'-scenario is picked in which environmental change is based on meeting specific European environmental policy directives (Amann et al., 2008), applied to Flanders, this is in line with more socio-environmental sustainability. For the severe scenario, the 'reference' scenario is selected, a business-as-usual approach. The FEA provided us specific future prediction values valid for the location of the Scots pine stand for the years 2010, 2015, 2020, 2025 and 2030. Linear interpolation was used to address the years between latter ones and after 2030 values are considered to remain 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 with 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 that of 40 kg N ha^{-1} yr^{-1} with a share of 0.21 NOy-N and 0.79 NHx-N, valid for the period 1992-2007 for the Scots pine stand (Neirynck et al., 2008). The effect of change in vegetation on dry deposition of PM and thus on the nitrogen deposition is not considered as 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_{10} and 16.77 µg m^{-3} $PM_{2.5}$ (with a resolution of 3*3 km), of which the methodology is explained in Chapter 3, pg. 55.

For modelling of PM removal, halfhourly precipitation and PM concentration need to be known (Chapter 3, pg. 55). For 2010, these values were measured for precipitation and calculated by IRCEL and FEA for PM as addressed above. Halfhourly precipitation and hourly PM values for other years were obtained via recalibration using the yearly values, e.g. halfhourly precipitation values of 2011 = halfhourly precipitation values of 2010*yearly precipitation 2011/yearly precipitation 2010. This is not done using monthly values for precipitation instead of yearly values as unrealistic results could be obtained because of higher variation between months than years. Note that every year has the relative same rain and PM pattern as the reference year 2010. Humidity is set constant at a high 0.7 since the climate of the Scots pine stand is a humid one.

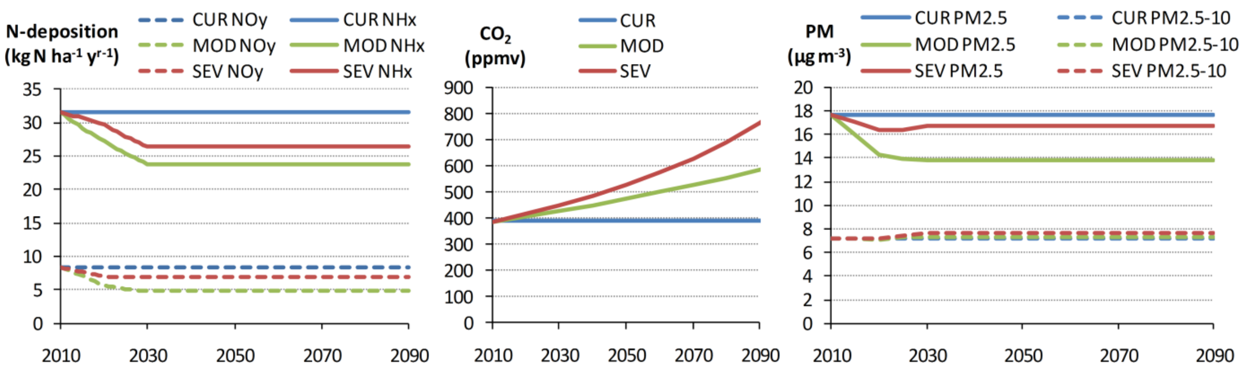

Figure 4.2. Nitrogen (N) deposition, CO_2 and airborne particulate matter (PM) values over time for the current (CUR; blue), moderate (MOD; green) and severe (SEV; red) environmental change scenarios.

4.2.5 Ecosystem services and their monetary valuation

Besides the definition given in the introduction (see section 1.2, pg. 4), we interpret an ecosystem service as a property, function, process or a collection of these of an ecosystem which provide a benefit to mankind, variable in time and space (Lyytimäki and Sipilä, 2009). For example, a forest may prevent runoff to a nearby river, lowering risk of flooding, but this could lead to a water shortage later on in another region which stores water from this river in a reservoir. Assessments of these may thus be very case specific and should therefore in practice be considered for fixed time and space boundaries if possible. For this study, only the services provided during the management period will be accounted for. The area benefiting from the services may vary between the services, from local (water recharge) to global (global warming potential), and is as much as possible Flanders in this case study. Ecosystem services are furthermore subdivided into different categories: provisioning (e.g. food, water), regulating (removal of pollutants,...), supporting (these support other services; e.g. nutrient cycling, primary production,...) and recreational/cultural services (Figure 1.3, pg. 5). Besides services, ecosystems may also provide disservices, which are negative for human well being (Lyytimäki and Sipilä, 2009), e.g. infectious disease spreading, crop damage by pests, emission of volatile organic compounds and allergenic pollen (Dunn, 2010; Escobedo et al., 2011; Lyytimäki and Sipilä, 2009). These disservices should thus also be regarded if possible (Lyytimäki and Sipilä, 2009).

We need to first select ecosystem (dis)services, and the processes or aspects responsible for them, which can be directly attributed to the forest 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 not/less harmful ones. Above that, our selection of ecosystem (dis)services is also restricted by the ones for which monetary values are present. Besides that criterium, data should be of course available or modelled, by ANAFORE in our case, to account for a (dis)service. In Table 4.4, the here considered ecosystem (dis)services are given.

In the other 5 studies (Table 4.1, pg. 96), wood production, because it was longtime considered the primary function of forest, and biodiversity are by all in a certain manner accounted for. Carbon sequestration is considered by 3 and groundwater recharge by 2 of them. In our study, we consider: wood production, carbon sequestration, water evapotranspiration, $PM_{2.5}$ and $PM_{2.5\text{-}10}$ removal, NO_x emission, NH_3 processing and nitrogen pollution/removal. The reasons for not considering biodiversity are given further on.

To quantify ecosystem services, monetary values may be attributed to them (Baveye et al., 2013; de Groot et al., 2012). Specific for the region of Flanders, such values have been developed by the Flemish institute for technological research (Broekx et al., 2013; Liekens et al., 2013b). Economic values can be attributed to ecosystem services via different approaches. For provisioning services this can be straightforward their normal market price. Willingness to pay for a service is another approach, used for recreational/cultural services (Liekens et al., 2013a). For regulating services, a first calculation option is the avoided damage cost; the second option is the avoided abatement cost. Note that the variety in methods induces different outcomes for a certain service and thus variability in its monetization (Kumar et al., 2013). Other methods for monetization of ecosystem services are not addressed here. For disservices, the negative value of the opposite service is considered. In Table 4.4 is shown which monetization approaches are used for the different considered (dis)services. Since monetary values may vary from year to year, 2010, the beginning of the management period, was selected as the reference year.

Table 4.4. Ecosystem (dis)services considered of the forest ecosystem with their monetary valuation and the characterization factors for environmental impact assessment, based on values of Recipe version 1.08 (2009), Alvarenga et al. (2013) and Pfister et al. (2009). Calculation of the service is done via modelling using ANAFORE (Deckmyn et al., 2011, 2008) or just retrieved from data. 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 services							**Impact assessment**	
Ecosystem services		Calculation		Monetary valuation			/	
Ecosystem 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	Mod-elled		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	Mod-elled		20 euro ton^{-1} CO_2	Avoided abateme nt 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 (Neirynck 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
Emission of NO_x (disservice)	Emission of NO_x to the air	Data	5.29% of N deposition (Neirynck et al., 2007)	0.6 euro kg^{-1} NO_x*	Avoided damage cost	(De Nocker et al., 2010)		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^-
							Terrestrial acidification	1.42E-8 species*yr kg^{-1} NH_3 3.25E-9 species*yr kg^{-1} NO_x
Water purification/ pollution via N-removal/emission (regulating)	The net amount of eutrophication potential (kg N eq.) of the forest (see section 4.2.5)	Data & Mod-elled	kg N eq. input – kg N eq. output (based on the values of marine eutrophication)	5 euro kg^{-1} N**	Avoided abateme nt cost	(Broekx et al., 2013)	PM formation	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 4.4

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, which is taken up by the foliage and ends up on the soil	Mod-elled	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 fresh water (disservice)		Mod-elled	rain – infiltration = transevapo-ration + runoff#	-0.075 euro m^{-3} H_2O Tax for water extraction	product price	(Broekx, 2013)	Freshwater consumptio n	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 euro, or high, 74 euro, kg^{-1} N removed from water. Expert knowledge of prof. dr. ir. Siegfried Vlaeminck points out 5 euro as the fitting value.
#: runoff is negligible for the Scots pine stand since it has almost no slope

On the considered services, we elaborate more in the following text. Next to that, we explain why some considered services or approaches presented in literature are not taken into account in our study. The **provisioning of fresh water** through seepage is by some accounted for as a service provided by the forest (Fürstenau et al., 2007; Ninan and Inoue, 2013). This could be questioned since it is the complete hydrological cycle which produces rain that falls on land and may end up as available fresh water. Attributing this service solely to a terrestrial ecosystem, such as a forest, is not fitting. The land ecosystem may however influence the fate of the fresh water through its influence on runoff, evapotranspiration and infiltration, and could thus locally/regionally influence the available stock in fresh water. Runoff and evapotranspiration may lead in fact locally to a potential loss of fresh water as there is less infiltration which refills groundwater reservoirs. This loss by evapotranspiration is already pointed out (Jobbágy and Jackson, 2004; Maes et al., 2009). Note that if runoff ends up in another natural fresh water reservoir, it may not be lost. Also, on a larger scale, evaporated water could end up again as freshwater somewhere else (Keys et al., 2012). For the studied Scots pine, 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). On top of that, if the groundwater table is high enough, tree roots may be able to directly take up ground water, and thus potential fresh water, besides rain which percolates through the soil (Dawson, 1996; Jobbágy and Jackson, 2004). This is clearly shown by a study done on another Scots pine stand on 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 therefore be regarded as a regulating ecosystem disservice, being the loss in freshwater. This will be accounted for in this study. Duncker et al. (2012) just consider runoff and neglect to regard evapotranspiration. Ecosystem services prevention of erosion and reducing impact of flooding through water retention are not relevant since the Scots pine stand is not located in an area where this is of importance (Broekx, 2013). On the other hand, 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. Next to that, 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 by consequence not done.

Water may not be provided directly by an ecosystem, though, just as its fate, its composition may be altered. (**Water) purification** is an important ecosystem service, which has been put forward many times (Duncker et al., 2012; Ninan and Inoue, 2013). Specific, 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 is improved. In this study only the water quality aspect of nitrogen content of the water is considered. To just account for the quality of the water leaving the system is not fitting, since the occurrence of pollutants which are already present in the initial input, rainfall and deposition, is not accounted for. 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 needs to cope with a total nitrogen input through rain fall and dry deposition, and the service provided is the amount which does not end up in the water/ the gain in water quality. This service is provided through the ecosystem by taking up input nitrogen into biomass and through converting it into mainly non-harmful dinitrogen gas via microbial processes. A disservice may be the extra presence of N in watery flows through depletion of the nitrogen stocks.

In addition, the damaging effect depends not only on the amount of nitrogen but also in which forms, e.g. nitrate, this amount is present. When 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 the ReCiPe methodology (given in Table 4.4, pg. 107) and sum them up. For nitrogen compounds in watery flows this factor is just the amount of nitrogen per compound, e.g. 0.78 kg N kg^{-1} NH_4^+. For nitrogen in gaseous fluxes and particulate matter deposition, these values are lower due to their lower potential in ending up in watery flows (Goedkoop et al., 2009). Also, since the uptake of these fluxes is influenced by the forest, this is more suitable. After all, through enhancing dry deposition via plant surfaces, forest may aid in bringing N-compounds from air into water, and thus actively contribute to eutrophication. After obtaining the single summed up value in kg N equivalents, we may convert it to a monetary amount by multiplying with 5 euro kg N^{-1} as mentioned in Table 4.4, pg. 107. To calculate this in a good manner, the composition of nitrogen compounds must be known. For each environmental change scenario, the total amount of NH_x-N and NO_y-N input are already given (see section 4.2.4, pg. 101). Based on the values of Neirynck et al. (2007) specific component amounts can be obtained. NH_x-N consists of dry deposited NH_3-N (51.44%) and NH_4^+-N (17.06%), and wet deposited NH_4^+-N (31.50%). NO_y-N consists out of wet deposited NO_3^--N (42.64%), and dry deposited NO_3^--N (25.58%), HNO_2-N (21.71%) and HNO_3-N (10.08%). The dry deposited amounts of HNO_2-N and HNO_3-N are considered to be NO_3^--N amounts as done in Schaubroeck et al. (2013). The nitrogen leaving the system are considered 100 % NO_3^- via drainage and emission of NO_x, 5.29% of the N deposition (Neirynck et al., 2007).

Two services of air, thus not water, purification or pollution by nitrogen compounds are also considered: the **emission of NO_x** and the uptake with subsequent **processing of NH_3**, this based on the values given in previous paragraph.

The total already **stored amount of nutrients** present in an ecosystem is also considered by some as an ecosystem service, for example 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 should be quantified, only the increment or depletion during that time period and the maintenance of the stock should be accounted for. The already stored nutrient amounts such as carbon and nitrogen are results of sequestration before the studied time period, which is thus outside the system boundaries. By previously mentioned authors, the monetary value for CO_2-C sequestration during a considered time period is assigned to the already present carbon stock. On the other hand, the maintenance of the stock can be considered. If the ecosystem would be deteroriating at the beginning of the study period, the stocks would deplete due to degradation and harmful components such as CO_2 and NO_3^- may be emitted again, but also less harmful ones, such as N_2. These fluxes are thus prevented by the ecosystem and could be measured or modelled. Moreover, in the future the stocks may be apprehended to overcome perturbations such as diseases. Note that it costs energy for an ecosystem to maintain its ordered state. A supporting service is thus provided by an ecosystem through maintaining its stocks. It is however difficult to quantify these services. In our study a full management period is studied starting from planting on a just clear-cut field up until clear cut, this to somewhat cover this service.

In general, the benefit of supporting services, such as water retention (Broekx, 2013) and the already stored amount of nutrients described above, is difficult to assess since the benefits occurs through the other services (provisioning, regulating and cultural) which they induce over time. They may by consequence have the potential of providing other services in the future. How to address this in particular needs to be researched further.

One of the most discussed issues related to ecosystem services is **biodiversity**. It is an important asset as it supports different ecosystem services (Costanza et al., 1997; TEEB, 2010). It can be considered as an ecosystem service on its own (Mace et al., 2012). Lyytimäki & Sipilä have however a skeptical view on biodiversity: "Securing especially the regulating and (other) supporting ecosystem services is used as an argument for protection of biodiversity. However, this rationale for biodiversity conservation or enhancement is lost if it turns out that the services can be replaced with similar or better man-made services or services produced by heavily manipulated ecosystems or ecosystems with very low biodiversity.". In fact the benefit of biodiversity increase should be approached with care since for example, according to latter authors based on the work of McKinney (2008) and Destefano and Deblinger (2005), emergence of invasive species into urban green areas can increase biodiversity but decrease ecosystem services. Whether or not agreeing with last statement, in general, the relationship between biodiversity and (other) ecosystem services is a complex one and needs further unraveling (Mace et al., 2012). Further research is needed to put a figure on the functionality of biodiversity and its link with other services, such as the planned experiments on functionality of tree species diversity of Verheyen et al. (2013) and Baeten et al. (2013). Consequently, the direct effect and (economic) value of biodiversity for mankind is not quantified yet (Cardinale et al., 2012).

Few forest models exist which may quantify, besides tree diversity, the total species diversity of plants, let alone animals. Though indicators which represent assets which influence diversity do exist (Mäkelä et al., 2012), such as coarse woody debris or amount of aged trees (Table 4.1, pg. 96).

According to Gao et al. (2014), four stand structure parameters influence plant species diversity: canopy coverage, age of canopy trees, tree species composition and canopy stratification. Our studied Scots pine stand is single-species even-aged densely planted forest and among the scenarios only the harvest regimes differ. For this forest, age of canopy trees does not differ and tree species composition will vary slightly, (growth of new trees on open spaces created through harvest though the acid soil in coniferous forest inhibits growth). Canopy coverage might vary more but not tremendously through harvest. Canopy stratification on the other hand will also be not complex as this is an even-aged forest. Harvesting of biomass may influence biodiversity through following aspects: destruction of habitats, providing open spaces for new species growth, leaving behind of dead wood for other organisms to use (Mäkelä et al., 2012). As for the dead wood, none is left behind in our management practices, inducing no discrepancy. For the two other aspects this is difficult to consider as habitats are provided (open space) but also destroyed. However growth is for various species difficult in this coniferous woods on sandy soils, as soil acidification occurs (de Schrijver et al., 2012). Hence biodiversity of the site will most probably be low, as will be the differences between the different management scenarios. Because of latter reason and the lack in monetization and evaluation of biodiversity as a service, biodiversity is hence not considered in the ecosystem service assessment.

Even though they may have a high monetary value (Broekx et al., 2013; Liekens et al., 2013a), cultural/recreational services will not be addressed here since we do not consider the socio-economic aspects/benefits. Moreover, Broekx et al. (2013) mentioned that there are issues related to scientific reliability of the methodology to estimate cultural services: the methods are based on only one study and tend to dominate results in most case studies. Next to that, the differences between our management scenarios are most probably negligible concerning cultural services. The method of Liekens et al. (2013) does also not provide any difference for such a marginal change, presumably for the same reason.

Through harvest, wood is provided to mankind. The price for the ecosystem service **wood provisioning** is that of the market price per cubic meter standing wood (€ m^{-3}) prior to harvesting, in function of its circumference (cm) at 1.5 m (Figure 4.3). These values are obtained from the Belgian federation of forestry experts ("Fédération Nationale des Experts Forestiers," 2013). Price data for Scots pine from the year 2010 are here used. For each size class, a minimum and maximum price and circumference are given. In this study we appointed the average of minimum and maximum price to the average of the minimum and maximum circumference of the respective class. This resulted in discrete data points. A zero value was assigned to a circumference of 20 cm or smaller. Between these coordinates linear interpolation was used to determine price values for the intermediate sizes. Beyond a circumference of 134.5 cm the price equals 35 € m^{-3}. This results in a sigmoid-like curve with an inflection point at 79.5 cm circumference and 20 € m^{-3}. Important to note is that these wood prices fluctuate significantly over time, see supporting information section 4.4.1, pg. 128.

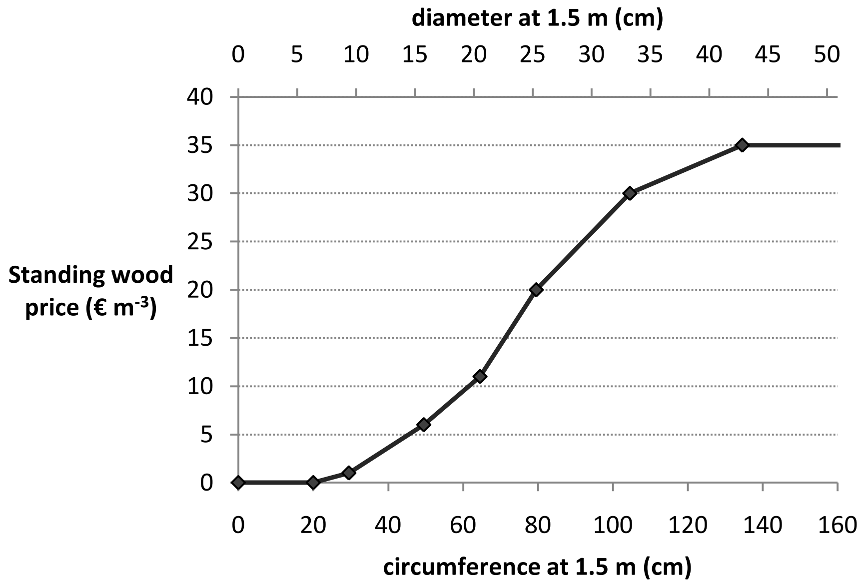

Figure 4.3. Price (€ m^{-3}) of standing scots pine wood in function of its circumference (cm) at 1.5 m. The diameter was calculated out of the circumference, assuming a perfect circle. For circumference larger than 134.5 and lower than 20 cm, the price is equal to 35 and 0 € m^{-3}, respectively. Linear interpolation, represented by the straight lines, between the other discrete values ("Fédération Nationale des Experts Forestiers," 2013) was used to obtain intermediate values.

4.2.6 Environmental impact assessment (methodologies)

To assess the environmental impact of resources and emissions of the forest ecosystem, the same framework as in Schaubroeck et al. (2013), see Chapter 2 (pg. 17), is applied. In this framework, the absorption of harmful compounds e.g. CO_2, by the forests, is considered as negatively valued impact, reflecting the remediation effect. The impact on three areas of protection is considered here: ecosystem quality, human health and natural resources (de Haes et al., 1999). Based on the same reasoning as in Schaubroeck et al. (2013), ReCiPe 1.07 (Goedkoop et al., 2009) is selected to assess the impact of emissions and land use on ecosystems, expressed as diversity loss (species*yr), and human health, expressed in Disability Adjusted Life Years (DALY). Important to note is that for the impact category marine eutrophication, no quantification in endpoint diversity loss is available yet though this effect is acknowledged (Goedkoop et al., 2009). Hence, this is just expressed in kg N equivalents. Furthermore, the impact of net loss of fresh water, mainly through evapotranspiration, on human health and ecosystem diversity is also assessed. This is done via the methodology of Pfister et al. (2011, 2009) in which the local and marginal impact of consumption of freshwater from lakes, rivers or aquifiers, the so called 'blue' water, is assessed. For this aspect, specific values for Belgium are given in Table 4.4, pg. 107. For human health impact, the value is 0 DALY m^{-3}, this is due to the fact that in the methodology of Pfister et al. (2009) for developed countries as Belgium, the loss in freshwater is assumed to be dealt with. For resource accounting also the Cumulative Exergy Extracted from the Natural Environment (CEENE) methodology is used (R. A. F. Alvarenga et al., 2013; Dewulf et al., 2007). Since this is an intensively managed forest, the resource usage according to this methodology, is only the deprived net primary production, expressed in exergy, normally occurring during the same period of land occupation at the considered site, modelled via a global vegetation model. The CEENE characterization factor for land occupation at the exact location (defined by its coordinates) of the Scots pine stand, is 278 GJ_{ex} ha^{-1} yr^{-1} (R. A. F. Alvarenga et al., 2013). The natural NPP production is induced by the combination of biotic and abiotic (such as rain, sunlight,...), thus indirectly these other input flows are also accounted for as resources. Here the solution of the environmental impact assessment methodology is thus a set of three values which represent damage to human health (DALY), ecosystem (species*year) and resources (CEENE). Since not a single value is obtained, these values need to be interpreted altogether or a multi-criteria assessment methodology needs to be applied. In this case they will just be interpreted. Using this approach already narrows down the different units to 4.

4.3 Results & discussion

4.3.1 Gross forest flows

Firstly, we will focus on the carbon fluxes, as these are characteristic for the forest growth and associated with CO_2 sequestration. In the first years, there is a negative carbon balance, this because of high heterotrophic respiration in the soil and thus the release of carbon dioxide. Only from year 5 onwards, a positive carbon balance is reported. After 9-10 years the total balance is again positive (see Figure 4.7), i.e., this is the repayment time needed to reach a net carbon uptake given the initial soil conditions. This period is shorter if less carbon is stored in the beginning, as is the case for the short rotation coppice on a low carbon soil in Flanders studied by Njakou Djomo et al. (2013) in which about a year is needed to have a net carbon uptake. Besides that, the nitrate leaching is also much higher in the beginning period, it drops about a factor 10 over 10 years. The overall reason for this is that the huge amounts of dead organic matter made available after harvest are processed by microorganisms and leave the system in reduced forms: most importantly carbon dioxide and nitrate. It is known that soil carbon decreases significantly after harvest (Nave et al., 2010; Zummo and Friedland, 2011). Also other modelling approaches underline this finding as they show significant decreases in soil carbon stores following intensive harvesting. Our result emphasizes the relevance of considering the right initial soil conditions (here just after a clear-cut) and the use of a forest growth model including a soil module (Deckmyn et al., 2011), which is not used in the other studies mentioned in Table 4.1, pg. 96, expect that by Seidl and Lexer (2013).

The average differences in carbon flows between the climate scenarios are not large. The main reason for this is that the dissimilarity between the scenarios in terms of CO_2 concentration (see Figure 4.2, pg. 104) increase over time, i.e. low in the beginning, and the most 'active' period of the forest is situated in the beginning 20 years. The Gross Primary Production (GPP), C uptake, peaks at about 10 years and decreases slowly afterwards. Though as expected, the average GPP is the highest for the severe and lowest for the current scenario (see Figure 4.4), in agreement with the difference in air CO_2 concentration and increase in temperature. However this increase in C uptake is counteracted by an increase in plant respiration, induced by the respective temperature increases, a driver for this respiration (Deckmyn et al., 2008). As a result, carbon assimilation, the net primary production, by plant is similar for all scenarios. The differentiation in soil respiration is much less, a maximum difference between corresponding management scenarios of only 0.1 t C ha^{-1} yr^{-1} was modelled. In the end, C sequestration, besides not differing considerably, is marginally highest for the CUR scenario and almost equal for the MOD and SEV scenarios, this for all management scenarios. Overall the increase in carbon uptake raise is counteracted by a plant respiration increase, in this case. Drought may have a considerable effect on forest growth and productivity (Allen et al., 2010). However as the climate is a humid one and precipitation does not alter considerably between climate scenarios, this effect is expected to be minimal for our case study. Above that, the groundwater table level is not a climate scenario variable. Hence the effect of drought on the carbon fluxes is minimal. New climate scenarios, especially if a potential drought effect on the forest could be induced, should include a change in groundwater level.

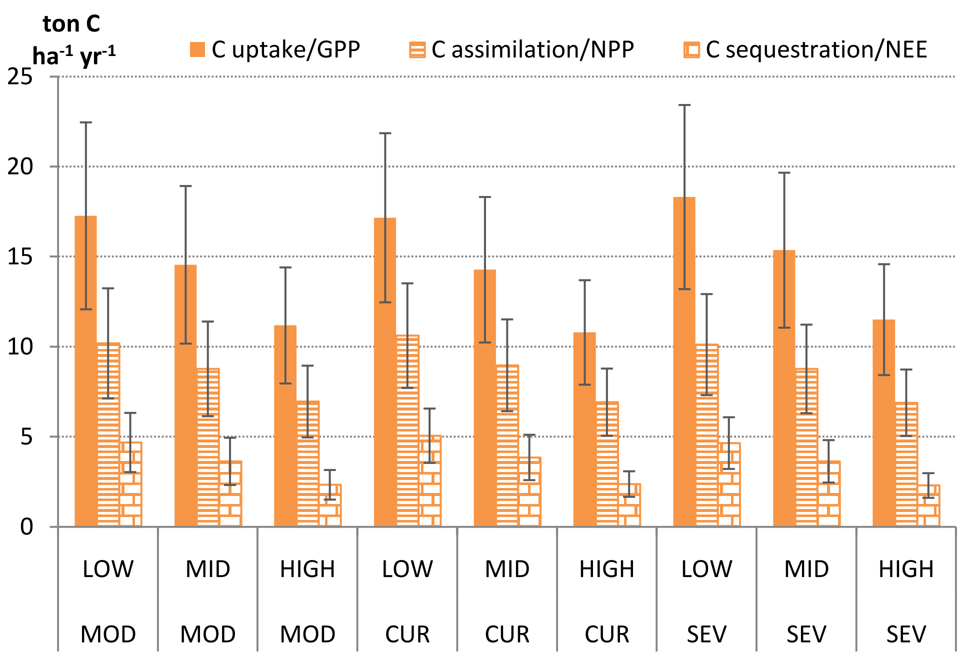

Figure 4.4. Yearly average carbon uptake, assimilation and sequestration for the combination of the three different management scenarios (low, mid & high) and the three different climate scenarios: current (CUR), moderate (MOD) and severe (SEV). Standard deviation induced by natural variability of weather (precipitation, irradiation & temperature) conditions (see section 4.2.4, pg. 101) is depicted with error bars. GPP: Gross Primary Production; NPP: Net Primary Production; NEE: Net Ecosystem Exchange.

Differences between management scenarios for this aspect are much larger, a spreading with a difference of 6.1 to 6.8 ton C ha^{-1} yr^{-1} in GPP between HIGH and LOW was modelled. This absolute discrepancy is lower for NPP and NEE respectively due to a higher plant respiration (2.7-3.6 ton C ha^{-1} yr^{-1} higher for LOW compared to HIGH) and soil respiration (0.9-1.0 ton C ha^{-1} yr^{-1} higher for LOW compared to HIGH) for the more intensive scenarios, induced most probably by a higher carbon uptake. However the relative differences are larger for NPP and NEE. Concerning the latter, i.e. carbon sequestration, the relative difference is most pronounced as about a double amount of sequestration is obtained for the LOW (4.7-5.1 t C ha^{-1} yr^{-1}) compared to the HIGH scenario (2.3-2.4 t C ha^{-1} yr^{-1}). The effect of environmental change scenarios on the differences between management is minimal; in this study no other management scenario of the ones presented should be applied under environmental change conditions. The best management scenario in this perspective is thus the 'LOW', least intensive, management scenario, this is in accordance with known findings in literature (Duncker et al., 2012; Fortin et al., 2012; Fürstenau et al., 2007; Schwenk et al., 2012; Seidl et al., 2007). Main reason for this is most probably that there is less efficient use of resources when there are less trees and after thinning events, time is needed for the forest to re-establish itself, e.g. canopy closure needs to occur. In line with this, less stem wood is available at the end of the rotation period but also overall produced, the more intensive the management scenario is, see Figure 4.5. However we have to note that this is not always the case as other management, different rotation length, thinning quantities, etc., may lead to more stem wood growth. One of the other main reason though why more intensive forest thinning is applied, is to obtain thicker tree (stems) with a higher price value ratio (Figure 4.3, pg. 113) evoked by more growth space per tree. Our results also clearly show that trees with a higher circumference are obtained at the end of the rotation period, when clear-cut occurs, for the MID (52.4-55.0 cm) and even more for the HIGH (61.8-65.0 cm) compared to the LOW (46.2-49.1 cm) management scenario (Figure 4.5). On the other hand, the weighted average circumference over the complete management period does not differ much between the scenarios, due to contribution of thinner trees originating from the periodic harvests. We have to note that the obtained circumference values are of course dependent on other management factors, which are considered here as constant, such as rotation length, harvest frequency, etc.

The values for wood production vary from circa 1000 to about 1500 m^3 ha^{-1}, 12.5-18.75 m^3 ha^{-1} yr^{-1}. These are too high values to be realistic; yield tables predict a productivity of circa 8 m^3 ha^{-1} yr^{-1} for the Scots pine stand (Jansen et al., 1996). Yield tables provide empirical maximum amount of wood productivity through a defined management procedure for various stand qualities. To have a productivity more than 150% of the ones from the corresponding stand quality of a yield table is most probably unrealistic. However, the monetary value of this ecosystem service (Figure 4.10) is minor compared to the other services (Figure 4.7). Hence the overall results will almost not vary if these productivity values were more correct. More research is though ongoing to resolve this matter. The other values calculated for the other ecosystem services are considered realistic. As the ANAFORE model is a complex entity it is difficult to trace back the influence of and reason for this outcome.

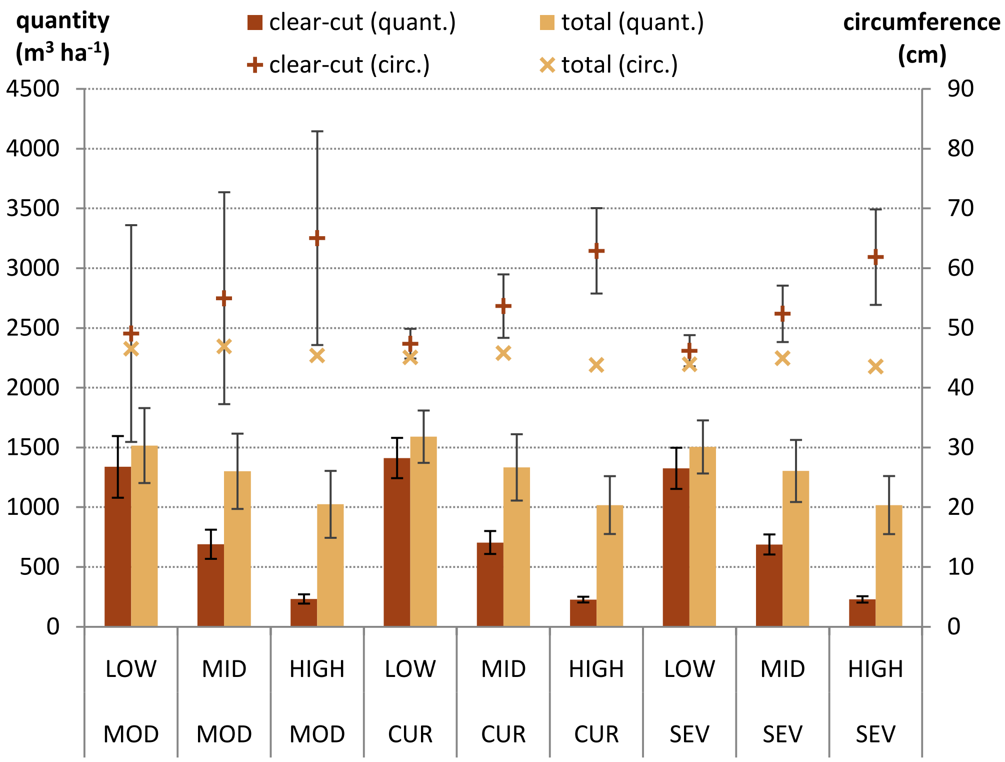

Figure 4.5. Wood quantity produced/harvested (column-left axis) and circumference (points-right axis) for the combination of the three different management scenarios (low, mid & high) and the three different climate scenarios: current (CUR), moderate (MOD) and severe (SEV). The total amounts are of the complete management period and the clear-cut ones for the clear-cut in the last year. The circumference for the total amount is a weighted average. Standard deviation induced by natural variability of weather (precipitation, irradiation & temperature) conditions (see section 4.2.4, pg. 101) is depicted with error bars.

An additional forest flux relevant to discuss is the removal of particulate matter (PM), more precisely $PM_{2.5}$ and $PM_{2.5\text{-}10}$ removal (Figure 6). For each climate scenario, the difference in leaf area index (LAI) is the main driver for the discrepancy in PM (Pearson correlation > 0.99), as expected. LAI is logically lower for more intensive forest management practices due to more tree harvest. The differentiation in removal between the environmental change scenarios is related to that of the considered airborne PM concentrations (see Figure 4.2, pg. 104). The variation in wind speed between the environmental change scenarios had a negligible influence, as could be expected from the relatively low change and findings made in Chapter 3, pg. 55.

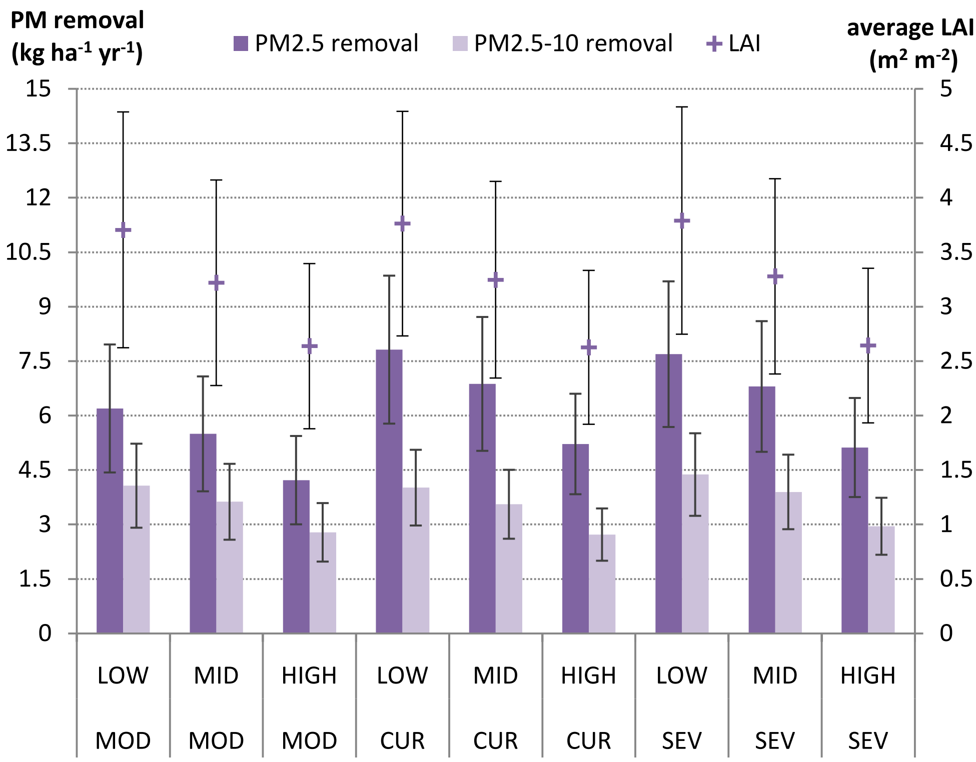

Figure 4.6. Yearly average particulate matter (PM) removal, $PM_{2.5}$ and $PM_{2.5\text{-}10}$, (column-left axis) and leaf area index (LAI) (points-right axis) for the combination of the three different management scenarios (low, mid & high) and the three different climate scenarios: current (CUR), moderate (MOD) and severe (SEV). Standard deviation induced by natural variability of weather (precipitation, irradiation & temperature) conditions (see section 4.2.4, pg. 101) is depicted with error bars.

The previous presented modelled values had considerable deviation, which shows that the influence of natural variation of weather effects is here, and may thus overall be, considerable on forest growth and its delivered services.

4.3.2 Monetary valuation of ecosystem services

The ease of monetary valuation of ecosystem services is that these are all presented in one tangible unit and can thus be easily compared and interpreted. The profile over time of the provisioning of services is highly similar for all 9 scenario combinations and presented for one of these in Figure 4.7. Only after 4-5 years a total positive monetary balance is obtained, and a cumulative positive balance over 10-11 years, i.e. only then the Scots pine stand will provide a net overall service to mankind, according to our results. This is due to high CO_2 losses and nitrate leaching explained in the previous section. After this period, almost all services maintain a linear increase, except CO_2 sequestration which decreases in slope and wood harvest which occurs in steps, associated with harvest operations, over time. The clear-cut at the end of the management period is responsible for an important share of monetary value of this service, leading to the steep increase at the end.

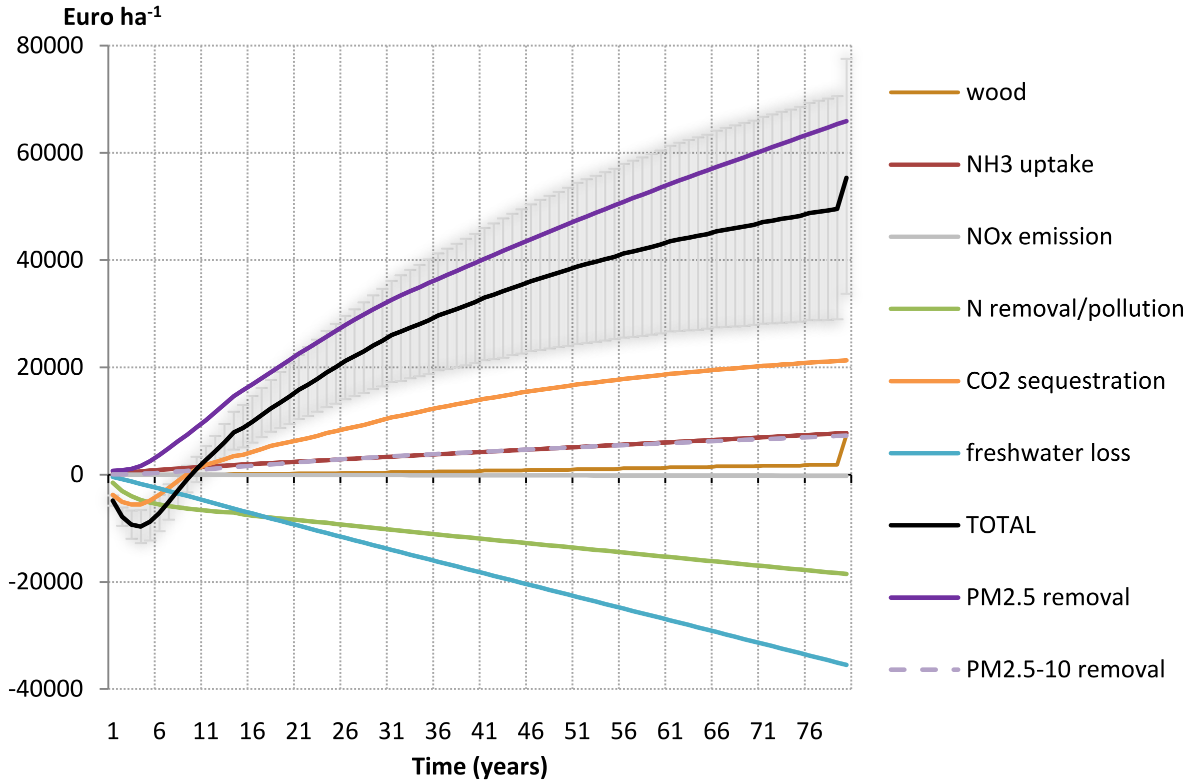

Figure 4.7. Ecosystem services provided by the Scots pine stand for the moderate environmental change scenario with a mid management type, presented in monetary values, cumulative over time. Standard deviation induced by natural variation in weather conditions is represented with shading for the total.

In Figure 4.8 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 monetary value is $PM_{2.5}$ removal with 622-1172 euro ha^{-1} yr^{-1}. In fact, when not considering this service, the balance would be negative in total for all scenarios. Next in line is CO_2 sequestration (168 – 371 euro ha^{-1} yr^{-1}). $PM_{2.5\text{-}10}$ removal, NH_3 removal, and wood production all have a yearly average around 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}. These last two services however do almost not vary between the different scenarios. The emission of NO_x is here a negligible disservice with a value lower than 5 euro ha^{-1} yr^{-1}.

The difference between environmental change scenarios (-60 euro ha^{-1} yr^{-1} for the severe and -289 euro ha^{-1} yr^{-1} for the moderate scenarios on average compared to 939 euro ha^{-1} yr^{-1} for the current) are not large and can be mainly attributed to the discrepancy in PM removal, and in a lesser degree NH_3 removal (and in a small extent to the other services), both can be allocated to the differences between the lower input/airborne concentration of these pollutants for the future scenarios compared to the current one. Simply put, the less 'pollution', the less a forest can remove these, and the lesser the provided removal services. Note however that nitrogen also serves as a nutrient and a complete depletion in nitrogen input can be detrimental.

Concerning management scenarios, CO_2 sequestration, PM removal (induced by LAI differences as previously discussed) 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 has a 1.25-1.30 higher value and the high scenario a 1.71-1.92 times lower total monetary value for ecosystem services, favoring the lowest thinning amounts. The difference in CO_2 sequestration is relatively the largest and that of PM removal and wood provisioning are similar in relative differences. Selection of the management scenario has here and can have a considerable impact on the delivered services by a forest. If one only regards the provisioning of wood, the same trend is visible only to a much lesser extent. Regarding all services, the LOW management scenario is the preferred one.

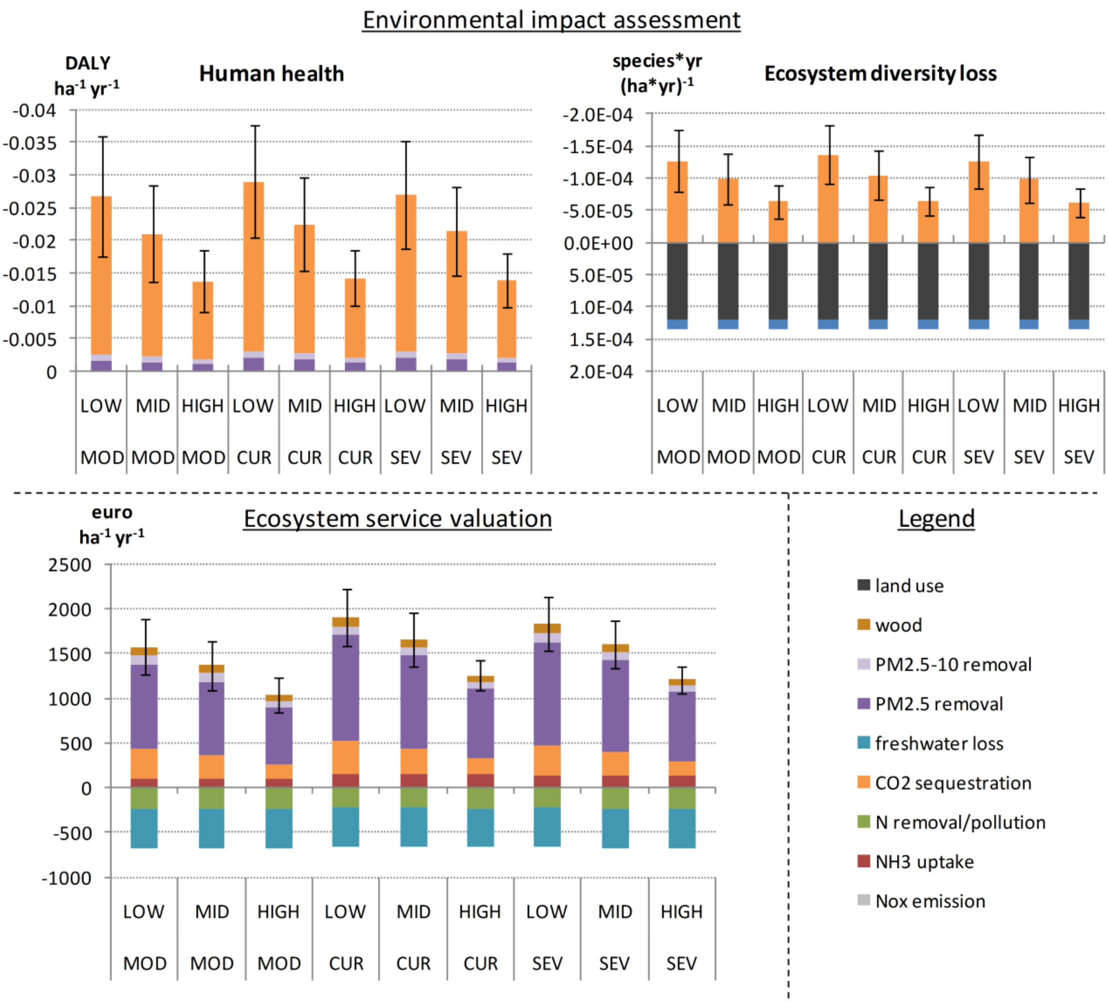

Figure 4.8. Comparison of the results of the 9 different combinations of management (low, mid & high) and the three different environmental change scenarios: current (CUR), moderate (MOD) and severe (SEV). Yearly averages of the environmental impact assessment and monetary valuation of ecosystem services are given. The environmental impact assessment expresses the impact on human health in disability adjusted life years (DALY), impact on ecosystems by ecosystem diversity loss and that of resource consumption is constant at 278 GJex ha^{-1} yr^{-1}. Standard deviation induced by natural variability of weather (precipitation, irradiation & temperature) conditions (see section 4.2.4, pg. 101) is depicted with error bars for the total values.

Certain tradeoffs and synergies might exist between the services. For NO_x emission and NH_3 uptake this is not interesting to research as their amounts solely depend on the amounts of N-input, fixed for every climate scenario. Using correlation, this can be tested from an empirical point of view (see Table 4.5).

Table 4.5. Correlation between the monetary valuation values for different ecosystem services

	Wood	CO_2 sequestration	PM removal	N removal /pollution	Freshwater loss
wood	1	0.97	0.84	0.60	-0.82
CO_2 seq.	0.97	1	0.87	0.65	-0.88
PM rem.	0.84	0.87	1	0.92	-0.79
N rem./poll.	0.60	0.65	0.92	1	-0.60
Freshwater loss	-0.82	-0.88	-0.79	-0.60	1

In our case, mainly synergies exist. High positive correlation exists between CO_2 sequestration, PM removal and provisioning of wood. Reason for this is that more PM is removed and CO_2 sequestered because of higher leaf area/LAI. And higher LAI occurs for the forest with lesser management (Figure 4.6) which though have a higher wood provisioning value. There is less nitrogen pollution, less nitrate in the leached water, in a small extent when all the other services increase, except for freshwater loss. A higher uptake of nitrogen by the plants is considered the main reason for this. Freshwater loss is negatively correlated with the other services, it decreases when another increases. This is most probably due to the fact that more water is transpirated with a higher leaf area/LAI. There thus exists a tradeoff between freshwater loss and each other here discussed ecosystem service.

4.3.3 Environmental impact assessment

The environmental impact assessment is expressed in three units: disability adjusted life years (DALY), species diversity loss and resource consumption (Figure 4.8). As the latter is only defined per hectare, 278 GJex ha^{-1} yr^{-1}, there is no difference between the scenarios. Concerning impact on human health, clearly a positive effect is obtained in all cases, 0.014-0.029 DALY ha^{-1} yr^{-1}, equal to 5.0-10.6 days ha^{-1} yr^{-1}, is prevented by the forest ecosystem. Over a complete management cycle, this is prevention in 1.1-2.3 disability adjusted life years ha^{-1}. The largest contributor is the uptake of CO_2, at least 85%. The rest is attributed to PM removal. Concerning biodiversity, there is a calculated loss in biodiversity, mainly due to the intensive management of the forest. The value for this is 1.2 E-04 species*yr ha^{-1} yr^{-1} which is the general difference between the natural environment and an intensive forest (Goedkoop et al., 2009). Above that, freshwater loss also leads to a minor diversity loss. The 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 diversity loss but only expressed in kg N, the spreading is very slim between the scenarios: 43.2-47.2 kg N ha^{-1} yr^{-1}, with the LOW scenario having the lowest values. This however shows that the forest has a negative impact on its environment concerning (marine) eutrophication, as the forest aids in bringing airborne particulate nitrogen into the leached fresh water via dry deposition. Though note that the quantified diversity loss does not completely cover damage to ecosystem (quality) as for some impact categories the impact categories is not expressed yet in diversity loss, such as marine eutrophication.

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

4.3.4 Allocation to wood produced

The forest delivers different biomass products: wood, roots, etc. When only considering wood as a product, the impact and the provided services of the forest can be fully allocated to the wood produced in m^3 by simply dividing the values per hectare by the wood production per hectare (m^3 ha^{-1}). This though implies that the higher the wood productivity the lower the environmental benefit/impact and services associated with a m^3 of wood. Since in our case, the productivity (Figure 4.5, pg. 118), besides provided services and the environmental benefit, are higher the lesser intensive the scenario, the differences per cubic meter stem wood produced are less pronounced. The standing value of this wood varies from 5.2-5.9 euro m^{-3}, but when including the already added value due to provided ecosystem services during production, a total economic amount of 28.2-62.5 euro m^{-3} is delivered to society. Next to that, the resource usage does differ, 14.0-21.8 GJ_{ex} m^{-3}, and is lower for scenarios with higher productivity, thus the LOW scenarios. In each of previous aspects, the wood of the LOW management scenario comes out on top.

We may compare latter values with the ones of Schaubroeck et al. (2013), namely 355 GJex m^{-3} wood. This shows that when considering a complete rotation period with realistic productivity, values can be considerably different. Schaubroeck et al. (2013) already pointed out that their considered time window was too narrow. Comparing the impact on human health, biodiversity and marine eutrophication is not fitting since more flows are considered in Schaubroeck et al. (2013), and especially for PM removal only deposition is considered by latter authors. Similary one may allocate everything to another service, e.g. to 1 t of CO_2 emission reduction as done by Garcia-Quijano et al. (2005). Note however that our wood productivity values are too high to be very realistic.

4.3.5 Discussion of methodological approaches

In our case the same management scenario, the LOW one, is the best according to both approaches. Keep however in mind that this is just a selected amount of ecosystem services/flows and not all relevant ones (see section 4.2.5, pg. 105). Though the differences in values between the monetary ecosystem services approach and the environmental impact assessment highlight the distinctions between these methods and also acknowledge that results may vary according to applied environmental sustainability assessment method. The most important differences are: PM removal is by far the most important for ecosystem service valuation but CO_2 sequestration for environmental impact assessment, wood production is not accounted for in the latter one and biodiversity loss not as an ecosystem service. Next to that, the ReCiPe methodology is somewhat older than and not site-specific as the used monetary valuation of ecosystem services. ReCiPe does not distinguish between the difference in health damage between $PM_{2.5}$ and $PM_{2.5\text{-}10}$, even though research pinpoints that this is the case (De Nocker et al., 2010; Mirowsky et al., 2013), though Perronne et al.(2013) argue this matter. Next to that, the endpoint for damage to ecosystems is assessed as species diversity/richness, which covers directly only one aspect: information. As discussed in section 4.2.5, pg. 105, the value of biodiversity is not quantified yet. Theoretically, an increase in richness does not ensures 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 manual of the ReCiPe methodology they already pointed out that the damage to ecosystem also addresses disruption of mass and energy fluxes, besides information (Goedkoop et al., 2009). Other approaches might by consequence be used, based on changes in mass, energy or exergy fluxes, storage and dissipation (Maes et al., 2011; Schaubroeck et al., 2012; Silow and Mokry, 2010).

On the other hand, ecosystem service and environmental impact assessment clearly overlap (see for example Table 4.4, pg. 107) and are integrated more and more. Take for example, the uptake of harmful compounds which is considered by Schaubroeck et al. (2013), Chapter 2 (pg. 17), the framework used in this study, and based on regulating ecosystem services. In fact, in that framework more than just an ecosystem service is accounted for since also the beneficial effect on the ecosystem is assessed besides that for human health. In our study this has been additionally done through using the midpoint characterization of marine eutrophication to convert all N-flows into a kg nitrogen equivalents, used in the ecosystem service valuation. Novel life cycle impact assessment methodologies are developed which assess the potential damage on (the provisioning of) ecosystem services (Arbault et al., 2014), this mostly for different land use (Koellner and Geyer, 2013; Saad et al., 2013). However an essential issue should not be forgotten: the conceptual difference between ecosystem service and environmental impact assessment, being that the first one only considers the benefits for mankind while the latter accounts for the total environmental impact. When integrating and comparing these, one should keep this in mind.

Overall, more methodological improvements, striving towards unification in methods such as ecosystem service assessment and environmental impact assessment, and more research on ecosystem flows and modelling is needed.

4.3.5.1 Monetization of ecosystem services, the holy grale?

In this study, a valuation of ecosystem services was done using monetary values assigned to them. Though it allows to put a single value on all the services provided, one should keep in mind when judging these results that an economic value does not (completely) represent/capture an intrinsic value, the benefit for mankind, and that the given value depends on the socio-economic framework used to obtain it, e.g. avoided damage. Baveye et al. (2013) reviewed different scientific opinions on this matter. There is by consequence still a need to express ecosystem services in a unit which represents better their intrinsic value (Baveye et al., 2013). However given the economics-oriented society we live in, monetization makes the concept more tangible (Quine et al., 2013). It is though better just used to compare different alternative scenarios which influence the provisioning of ecosystem services (Kumar et al., 2013), as is done in this study. Monetization is nowadays a necessity if one wants to easily account for it in our society. Using these values one may in fact readily consider ecosystem services as economic products. As a step further, one could thus financially reimburse land owners for the services provided by their land, this is called the Payment for Ecosystem Services (PES). If we additionally consider these services as tradable products, without fixed prices, different market mechanisms are set loose on the prices which alter them over time. However it is not guaranteed that these changes in economic price represent a change in intrinsic value of the service for mankind, e.g. regard the volatility on the carbon emission market (Chevallier, 2011). It is advised to control, potentially fix, these prices by governments or institutions. On the other hand, this might induce rent-seeking. Note that, in this particular accounting/policy method a difference should herein be made between services that improve the life quality of the total (global) community, such as climate change, or the local/regional ones (Kumar et al., 2013). Some of the potential downfalls induced by PES and the necessary regulation to restrict these is well discussed by Kronenberg and Hubacek (2013). Focusing on our case study, the specific Scots pine stand has about a selling price of 16000 euro ha^{-1} in 2010 (price retrieved from owner Agency for Nature and Forestry). When considering the same ratio for rental and selling price as in the nearby city Brasschaat, a rental price of 143.6 euro ha^{-1} yr^{-1} is obtained. This is about a factor 2.5-8.6 lower than the here calculated value provided to mankind by the forest through ecosystem services, showcasing an undervaluation of these services by society.

If we consider fixed prices, the services are provided over a certain time span. On the market, the profit Y however 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) and calculated as $Y/(1+R)^T$. Yet again, this can be regarded as a variation in price and thus value. Next to that, inherent to changes based on percentages, they 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 benifit for future mankind is regarded as less important than for the current one. Above that, we consider valuation of ecosystem services in this study and no cost-benefit analysis/PES. We therefore mainly consider results without discontinuation of price (R=0%), but results of NPV with a constant discount rate of 2 and 4%, as done in Fürstenau et al. (2007), have been obtained. Mainly the same conclusions are drawn as without discounting. For more info, see supporting information section 4.4.3, pg. 131. In practice, the 'ecosystem service' concept could replace the 'sustainable forest management' principles as a framework for management selection, though for now they should co-exist, mostly since not all ecosystem services are well enough (e)valuated (Quine et al., 2013).

4.3.5.2 Influence of space and time boundaries

As already noted, the temporal and spatial boundaries influence the results of these 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 only over 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 a very high human development index (> 0.88) for Belgium, the malnutrition vulnerability induced by loss in agricultural crops is set equal to zero. From a marginal and local perspective this is acceptable (ceteris paribus principle), but if huge quantities are withdrawn, this will always have a direct effect and will result in a loss in agricultural products on the global market which may thus effect human health, though possibly not locally. These boundaries should be kept in mind and possibly broadened through further research. Just as mentioned in Schaubroeck et al. (2013), the aspect time, e.g. the amount of time carbon dioxide is stored, and the regional differentiated aspect of impact/effect need to be better integrated, this also in the ecosystem service valuation.

4.3.6 Acknowledgements

We want to express our special gratitude to Sandy Adriaenssens, Jordy Vercauteren, Frans Fierens, the Flemish Environment Agency (FEA) and the Belgian Interregional Environment Agency (IRCEL-CELINE) for providing the airborne PM data, Line Vancraeynest and the FEA for the nitrogen deposition data, and the Research Institute for Nature and Forest (INBO) for supplying meteorological and flux data on the Scots pine stand. We are also appreciative towards two forest experts Pierre Fonteyn and Jean-Christophe Naets ("Fédération Nationale des Experts Forestiers," 2013), which provided advice concerning the standing wood price in function of circumference of Scots pine. Model runs were carried out using the STEVIN Supercomputer Infrastructure at Ghent University, funded by Ghent University, the Flemish Supercomputer Center (VSC), the Hercules Foundation and the Flemish Government – department EWI. We also would like to thank Leendert Vergeynst for aid in modelling on latter platform.

4.4 Supporting Information

In this section additional information is given concering price of standing wood (section 4.4.1) and monetary valuation with discontinuation of ecosystem services (section 4.4.3).

4.4.1 Price of standing wood

It is important to note that the wood price may fluctuate considerably in time. Figure 4.9 shows the evolution of the average price of Scots pine over more than a decade according to the values of the national federation of forest experts ("Fédération Nationale des Experts Forestiers," 2013). The price more than doubled over a decade from 2003 to 2013. Presumably, because of an increase in interest in biomass products (as energy source) the price is going up. There is a downfall in 2008-2009 probably inflicted because of the economic crisis but afterwards the price reestablished. From 2010 on the price remained more or less the same, making the choice for this as reference year more acceptable.

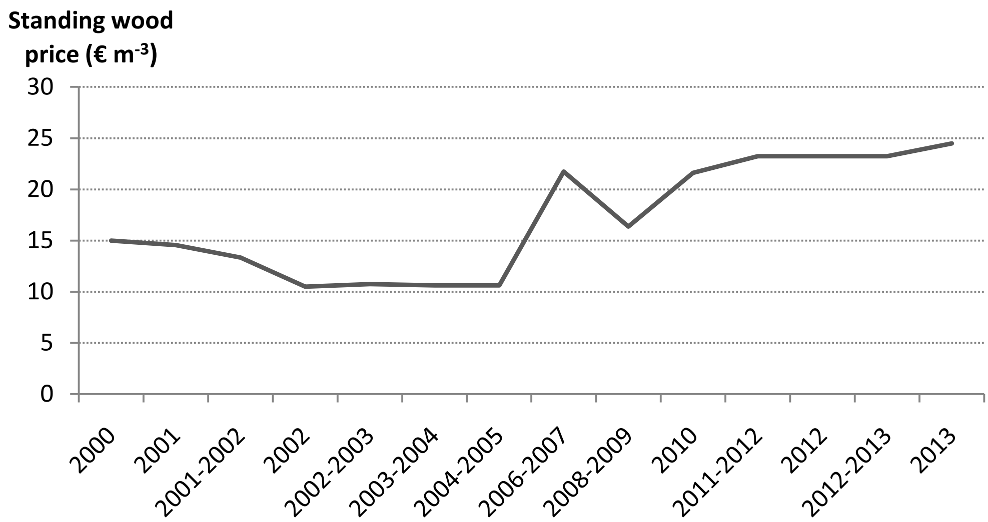

Figure 4.9. The price of standing Scots pine wood over time ("Fédération Nationale des Experts Forestiers," 2013).

4.4.2 Monetary valution of wood provisioning

Figure 4.10 represents the distribution in monetary value of the provided wood between the different scenarios.

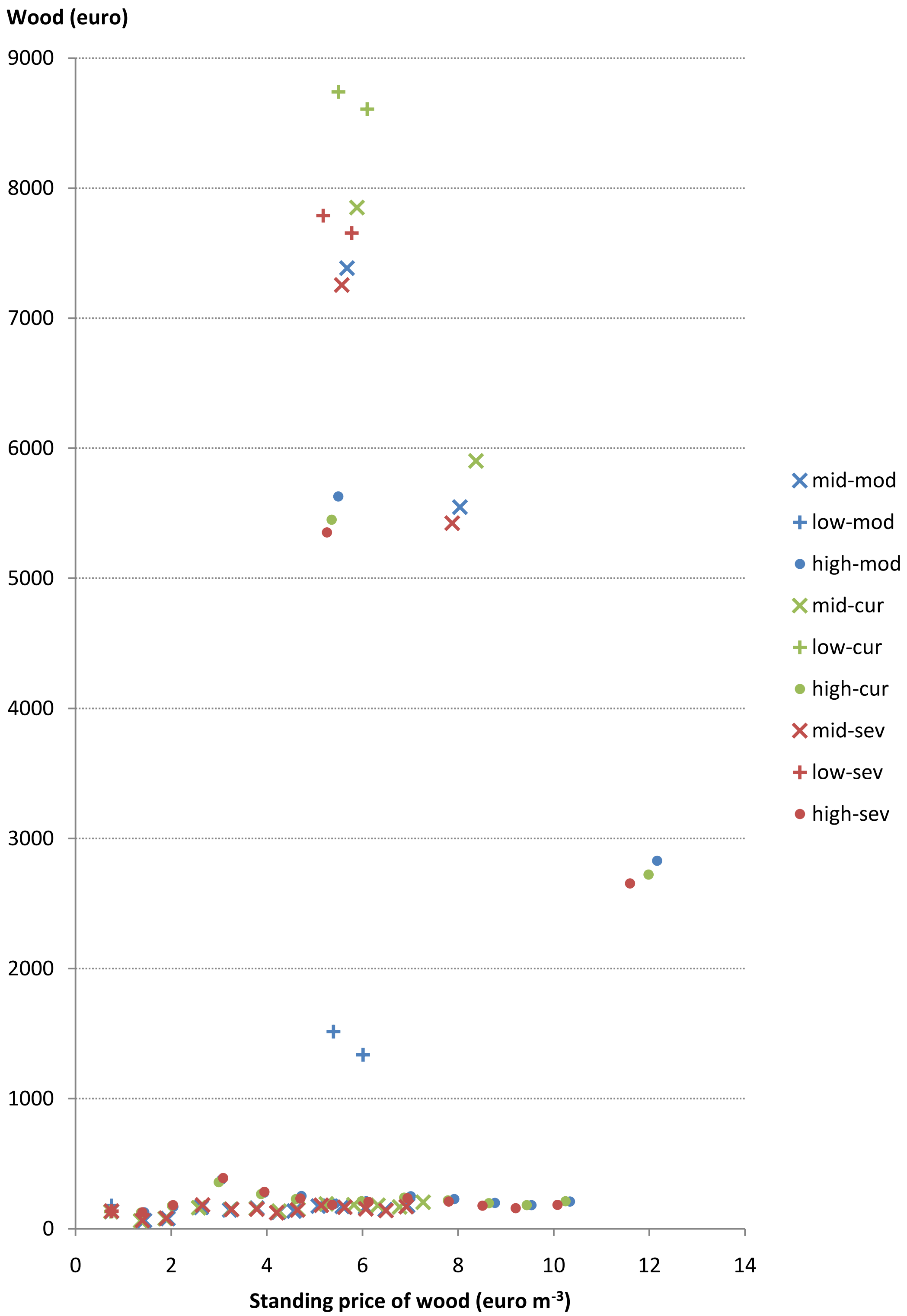

Figure 4.10. Monetary value of harvested wood in function of its standing price value for the different scenarios. The highest data point for each scenario represents the total amount over the complete period at the respective weight average standing price value.

4.4.3 Monetary valuation with discontinuation of ecosystem services

As mentioned and explained in the article, we applied a monetary discontinuation with a discount rate of 2% and 4% to the results, presented in the graphs below. In overall discontinuation increases contributions at earlier years and decreases contribution at later years over a given period.

The Net Present Value (NPV) at a discount rate of 2% results in somewhat lower yearly average values of 233-577 euro ha^{-1} yr^{-1} (Figure 4.11). The differences in total value between the management scenarios are less pronounced as the management is applied over time and only differs starting from the year 20. The differences, compared to without discontinuation, is the most pronounced regarding wood provisioning, as the clear-cut occurs at the end of the rotation period. The profit for this service is circa 20 euro ha^{-1} yr^{-1} and the highest for the MID scenario instead of the LOW.

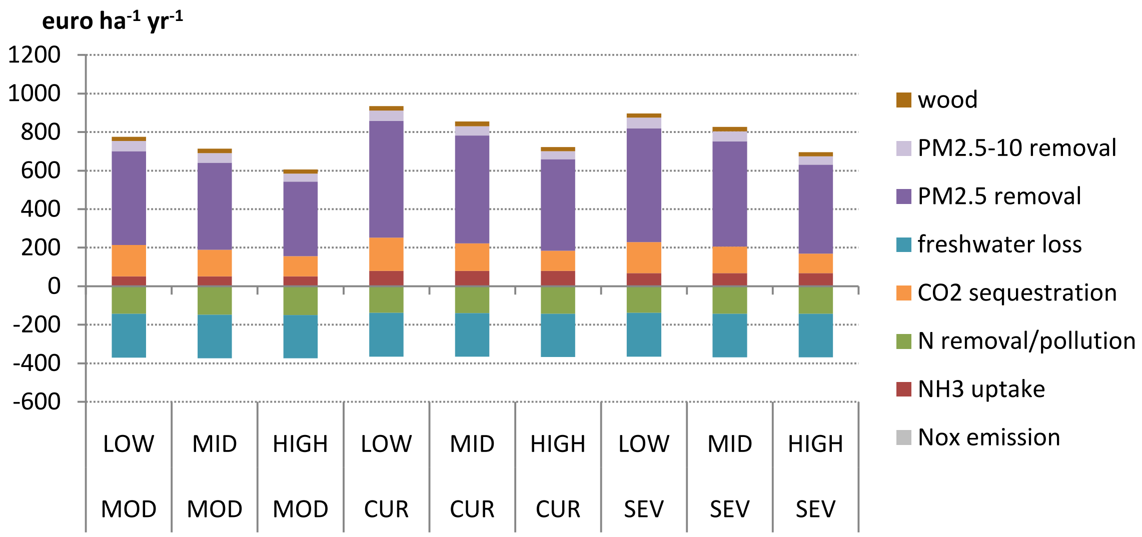

Figure 4.11. Average yearly monetary valuation of ecosystem services with discontinuation rate of 2%

For a discount rate of 4%, the latter mentioned effects are more drastic, this results in yearly averages of 86-302 euro ha^{-1} yr^{-1} (Figure 4.12). Regarding wood profit, the HIGH management scenario results in the highest added price, on average 8.9 euro ha^{-1} yr^{-1}, and the LOW to the lowest with an average of 5.6 euro ha^{-1} yr^{-1}. This is also an important reason why in real life more intensive management scenarios, with more frequent harvest, are applied, this to ensure a higher economic turnover.

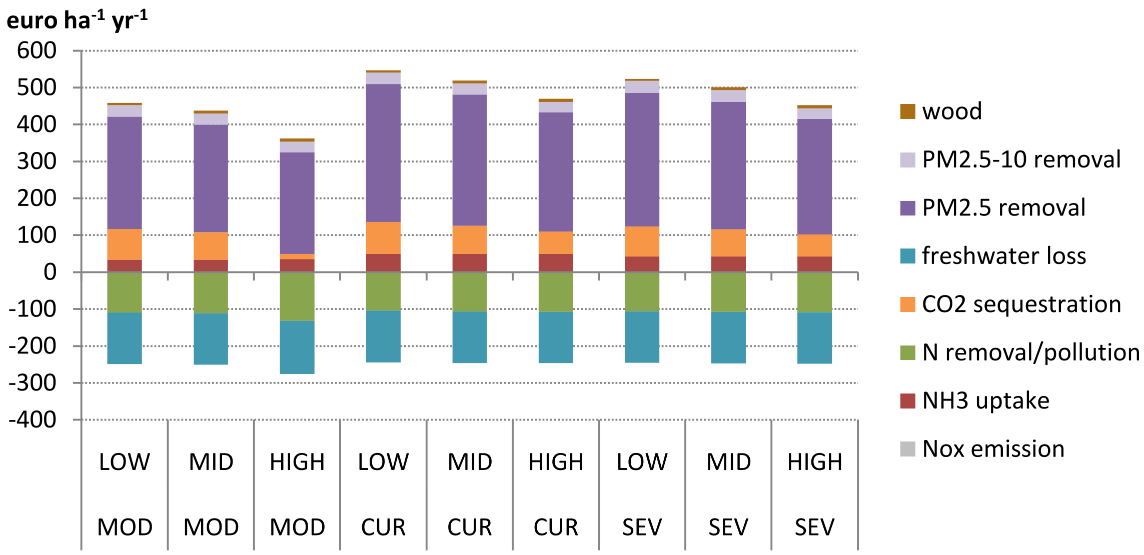

Figure 4.12. Average yearly monetary valuation of ecosystem services with discontinuation rate of 4%.

Chapter 5 Improved ecological network analysis for environmental sustainability assessment; a case study on a forest ecosystem

Redrafted from:

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Abstract

To assess the environmental sustainability of industrial products and services, tools such as Life Cycle Assessment (LCA) have been developed. In LCA, the total environmental impact of resource extraction and emissions during a product's life cycle is quantified. To better quantify this impact, first the alteration of ecosystems induced by those processes needs to be accounted for. Second, the flow networks of ecosystems, responsible for the formation of the extracted resources, should be included in the product's life cycle. To achieve these two objectives, a tool was selected which studies the flow networks of ecosystems: Ecological network analysis (ENA). In ENA, total system indicators are calculated which assess an ecosystem's functioning (e.g. cycling). Alterations of ecosystems can be represented by changes in the values of those indicators. ENA is based on the computational framework of Input-Output Analysis (IOA). This framework is also used in LCA allowing for a possible extension of a product's life cycle in an LCA with the ecosystem flow networks of ENA. The ENA/IOA framework itself was revised and improved in this study to better fit in an LCA framework, prior to integration and application in LCA. The major adaptation was to enable physical compartmentalisation of the surrounding environment of the studied (eco)system. This allows for a specification of destinations and sources of export and import flows, respectively, which is desired in LCA to assess the impact of these flows. Next to that, the adapted framework was made applicable to non-steady state systems by applying Finn's concept (1977, 1976), in which increase, increment, and decrease, depletion, in stock are considered abstract export and import flows, respectively. As an example, the adapted ENA framework was applied to a forest ecosystem. However, there are no standards yet for the different choices in the ENA methodology, which can have an influence on the indicator values. Hence, defining such standards is a next important research step.

5.1 Introduction

The growing awareness of global resource limitation for human development should drive the human/industrial system towards a more sustainable employment of natural resources and energy. To sustain the human/industrial system, ecosystems are indispensable as sources and sinks of energy and materials, apart from other provisioning, regulating and cultural services they provide to human well-being (see Hassan et al. (2005) for an overview). It is a challenge to provide metrics that quantify how sustainably the human/industrial system deals with energy and material flows from and to the ecosystem. In this context, tools such as Life Cycle Assessment (LCA) play an important role, as they quantify the effects on the environment induced by a product life cycle in terms of resource extraction and emissions (ISO, 2006a). To quantify this impact, we need to account for the formation of natural resources in ecosystems and the response of the ecosystems towards emissions and resource extractions. Therefore, the network of flows within an ecosystem and its alterations induced by those processes should also be considered in an LCA.

Tools that study the network of flows in an ecosystem have been developed in the domain of systems ecology. One of them is Ecological Network Analysis (ENA), founded by Hannon (1973) (**Figure 5.1**). This methodology is based on the computational framework of Leontief (1936), well explained by Suh (2005), called Input-Output Analysis (IOA) (Figure 5.1).

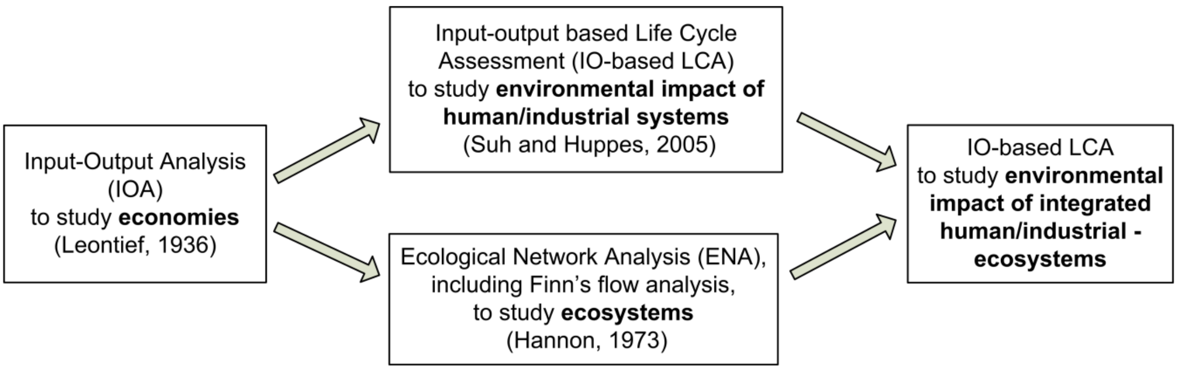

Figure 5.1. Different methodologies based on the Leontief input-output analysis framework (Leontief, 1936) and integration of ecosystems studied using ecological network analysis in an input-output based life cycle assessment.

An advantage of these ecological network indicators is that they can be much better constrained than the uncertain system flows, e.g. carbon sequestration in a forest, from which they are calculated and are thus robust estimators of the network functioning (Kones et al., 2009). ENA has been used on several ecosystem types, but particularly on aquatic systems (Baird et al., 2011; Chen et al., 2010; Christian et al., 2009; Link et al., 2009; Miehls et al., 2009). Besides application to ecosystems, ENA has also already been applied to human/industrial systems (Liu et al., 2011a, 2011b; Pizzol et al., 2013; Singh, 2012; Xu et al., 2011) and integrated human/industrial – ecosystems (Zhang et al., 2010c, 2009a, 2009b). Leontief's IOA framework is also used in some LCA studies, called Input-output based (IO-based) LCA (Figure 5.1). A product's life cycle is divided into different compartments of which the emissions and resources are assessed. The computational outline of IOA can then be used as a tool to create the Life

Life Cycle Inventory (LCI) of such a LCA, which is an inventory with all emissions and resources of a products life cycle. This LCI can be obtained by means of linear inverse modelling (Suh and Huppes, 2005). ENA and LCA are in fact both methodologies for system analysis. In ENA the network of flows in an (eco)system is studied, whereas LCA examines the environmental impact caused by the resource extraction and emissions of a product system responsible for the production of a certain good or service.

Practically, a LCA consists of a scope definition, system boundaries selection, construction of the Life Cycle Inventory (LCI), i.e. quantification of emissions and resources of a products life cycle for a certain quantity of the product, and Life Cycle Impact Assessment (LCIA), i.e. assessment of the impact of the emissions and resource extraction on the environment (incl. mankind). Integration of ENA could improve the LCA of a product or service in these different steps.

The application of the same Leontief framework in some LCA and all ENA studies opens opportunities to better incorporate ecosystems within the system boundaries, more specifically their flow networks, studied with ENA in a products life cycle (Figure 5.1). This is desired if one wants to account for the resources needed and the emissions of the particular ecosystems in the LCI, and their impact in the LCIA. Additionally, it allows for a linear approximation of these flows and those within the ecosystem for a given product quantity. One can thus easily perform an LCA with an IOA framework on an integrated human/industrial – ecosystem, socio-ecological system, studied with ENA. We quote Fath (2004): "... the most promising application of network analysis may be as a platform for integrated environmental assessment models to address sustainability issues of combined human-natural systems."

In LCIA there is both classification and characterization: the resources and emissions are classified in certain impact categories with representative indicators (classification), and their potencies of effect in the selected impact categories are determined (characterization) (ISO, 2006a). For example, carbon dioxide is classified into the impact category 'Global Warming Potential' with a characterization of 1 kg CO_2-equivalents per kg. Characterization models are needed to calculate the indicator values for each emission or resource. The impact categories can be divided into midpoint categories, which translate impacts into environmental themes such as acidification, and endpoint categories, which assess the final damage done to mankind and/or ecosystems. Often endpoint indicators are derived from the midpoint indicators. One class of endpoint indicators focuses on damage done to ecosystem quality (de Haes et al., 1999). This damage is regarded as disruption of mass, energy and information flows by anthropogenic activities by Goedkoop et al. (2009), in their manual on a recent holistic impact methodology (ReCiPe). In ReCiPe and most other methodologies only damage to information is considered, assuming it represents adequately the quality of ecosystems. This damage to information is represented by the endpoint indicator damage to ecosystem diversity (ED) expressed in Potentially Disappeared Fraction (PDF) (of species). Endpoint indicators based on ENA indicators could be ideal to fill the gap of assessment of the disruption of energy or mass flows. Characterization models for such indicators could be ecosystem models which directly or indirectly deliver the flow network of an ecosystem in response to emission or resource extraction (e.g. climate and/or management scenarios applied to a forest growth model) and calculate the change in ENA indicators over a time span. Next to that, ENA indicators could be used to study the product system itself, as has been done by Singh and Bakshi (2011).

In this chapter, an environmental sustainability assessment itself is not applied or altered but the focus of this study is to improve the ENA/IOA framework prior to application in environmental sustainability assessment, and more particularly in LCA studies. To do so, certain difficulties need to be overcome. First, in ENA studies, flows leaving the system (export flows) are commonly categorized in only two types, i.e. useful and non-useful, and flows entering the system (import flows) are only categorized as one type, following the convention of Hirata and Ulanowicz (1984). For an LCA study, the flows between the environment and the studied system (i.e. ecosystem) need to be quantified and specified in their external destination or source, as this is needed to assess the impact of these flows, and thus the system, on the rest of the environment. A second difficulty is that most ENA methodologies can only calculate all indicators if the system under study is in steady state, meaning no change in storage of the system nor of its entities (Allesina and Bondavalli, 2004; Fath and Borrett, 2006; Gattie et al., 2006; Schramski et al., 2011). However, many natural and managed systems in the world are not in steady state. Recently, complex solutions to apply ENA directly on non-steady state systems have been proposed by Matamba et al. (2009) using network particle tracking (Tollner and Kazanci, 2007) and by Shevtsov et al. (2009) using rigorous algebraic calculation methods. However, simple solutions have previously been developed (Finn, 1977, 1976; Suh, 2005). In this chapter, we apply the concept of Finn (1977, 1976), and explain why it is the best simple solution for applying ENA on non-steady state systems. A third difficulty is that there are no strict standards yet in the ENA methodology, giving room for choices in different steps of the methodology (Fath et al., 2007).

The main goal of this study is to revise the ENA methodology so that it can be better applied in LCA studies, providing an adapted general framework from data gathering up until the final calculations of ENA indicators. Three sub goals were defined: (1) to adapt the ENA methodology so that all ingoing and outgoing flows of the studied system are categorized according to their source and destination, (2) to account for non-steady state systems in a simple but adequate manner and (3) to qualitatively assess the influence of procedural choices in the methodology on the outcome.

In the 'Methodology' section (5.2, pg. 138), the conventional ENA methodology for steady-state systems is first explained step by step and then adapted to our needs. The adapted methodology is illustrated with a case study of a managed Scots pine forest ecosystem in Belgium (section 5.3, pg. 149). To the best of our knowledge, we only know of two other managed forest ecosystems studied using ENA (Finn, 1980; Heymans et al., 2002). In the subsequent section on 'Influence of methodology' (section 5.4, pg. 155), the influence of procedural choices on the outcome values of indicators is shown.

5.1.1 Notation

In this chapter, bold characters represent matrices (upper case) and vectors (lower case), while lower case italics with subscripts are used for elements of the corresponding matrix or vector, e.g. z_{ab} is the element of row a and column b of matrix $\mathbf{Z}$. Hat (^) diagonalizes vectors. Matrix $\mathbf{I}$ and $\mathbf{i}$ represent the unity matrix and a vector with all elements equal to 1, respectively. The matrix and vector dimensions depend on the calculations in which they are used. An i or a j refers to an internal compartment and a k to an external compartment (see section 5.2.1.1, pg. 139, more information). We consider n number of internal compartments and r number of external compartments. The external stock compartment of an internal compartment, is represented by s (see section 5.2.2.3, pg. 147, for more information).

5.2 Methodology

5.2.1 Conventional ENA methodology

The different steps necessary for an Ecological Network Analysis (ENA), are: (1) system identification and selection of system boundary, (2) compartmentalisation, (3) selection of energy-matter flow currency, (4) identification and quantification of flows, (5) data balancing, (6) construction of an input-output table and (7) calculations of indicators.

The first six steps are similar for any method using the Input-Output Analysis (IOA) framework. The explained procedure is partially based on Fath et al. (2007). The different steps do not necessarily have to be taken in the given order, though it is the most common one. The conventional methodology explained here will only deal with steady-state systems. Step one is case specific.

5.2.1.1 Compartmentalisation

The first task after selecting a study system and its system boundaries, is dividing the system into compartments. This internal compartmentalisation can be done in different ways and is an important choice. In the original IOA applied on industrial systems, the internal compartments (notation i or j) were usually economic sectors, such as the petrochemical industry. In reported ENA studies, the focus is generally on food web interactions. By consequence the internal compartments of ENA studies typically consist of trophic levels (Hannon, 1973), groups of species or individual species (Baird et al., 2011). For a forest ecosystem, Heymans et al. (2002) subdivided trees into foliage, wood and roots compartments, as is also done in our case study (see section 5.3, pg. 149).

A system also has interactions with its surrounding environment outside the system boundaries in terms of mass or energy flows. Rather than keeping them together as total import/export, these interactions can be subdivided in different external compartments (notation k). This external compartmentalisation can again be performed in different ways, and is another important choice. According to the Hirata and Ulanowicz (1984) convention, used in almost all ENA studies, the subdivision occurs by the type of interaction: import to the system, export of usable products (i.e. organic matter) and export of unusable products (i.e. respiration/dissipation).

An important criterion for compartmentalisation is the availability of data to quantify the flows between the compartments. A quick scan of the available data for flow quantification is therefore recommended before compartmentalisation.

5.2.1.2 Selection of energy-matter flow currency

In this step a currency in which to quantify the flows must be chosen. When studying an economy, money is the common currency. For ENA, typically biomass (which can be expressed in carbon), nutrients or energy are used as a currency for the flows through a food web (Baird et al., 2011; Fath et al., 2007; Finn, 1980; Hannon, 1973). More recently, other currencies have also been used: exergy (Liu et al., 2011) and emergy (Zhang et al., 2009a), but also information (Chen et al., 2011).

5.2.1.3 Identification and quantification of flows

After compartmentalisation and the selection of a currency, the flows between compartments can be identified and quantified. The different kind of possible flows and their nomenclature are represented in Figure 5.2 and Table 5.1.

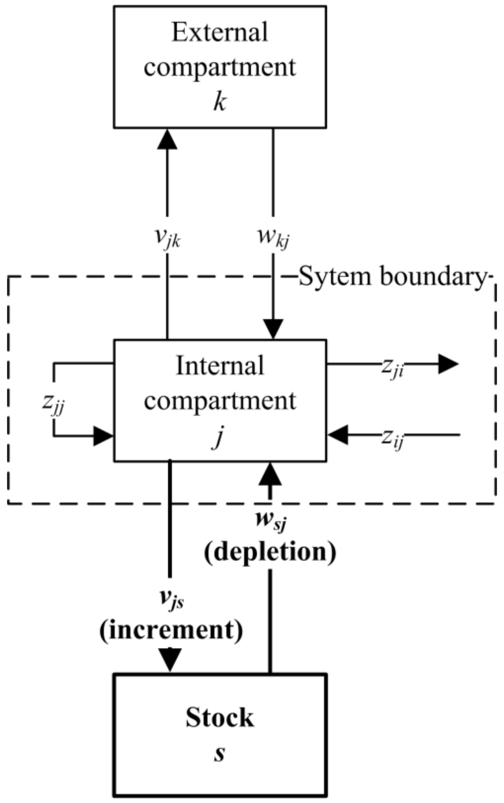

Figure 5.2. Diagram of the different type of compartments and the type of flows between them (cf. Table 1). Following the framework of Finn (1977, 1976) for non-steady state systems, the stock of a compartment (*s*) is considered as an external compartment and depletion and increment are represented by flows to and from it, respectively. The stock compartment and these flows are printed in bold.

Flow identification, i.e. defining all flows between the compartments, and quantification, i.e. assigning values to flows, can be done simultaneously. However, to be sure to take into account all flows, we recommend to first identify all possible flows in a system and to select from these the ones that will be quantified, neglecting those who are presumable negligible in quantity and/or not applicable for the case study. For ENA studies on food webs, there are some tools available to help quantify the flows (see e.g. Ulanowicz and Scharler, (2008)).

Table 5.1. Different types of flows between internal and external compartments and their nomenclature. Internal compartments (*i*, *j*) are those within the system boundary and external (*k*) those outside of it. There are *n* and *r* number of internal and external compartments, respectively. The flows for a non-steady state system are printed in bold. Following the framework of Finn (1977, 1976) for non-steady state systems, the stock of a compartment (*s*) is considered as an external compartment and depletion and increment are represented by flows to and from it, respectively.

Notation/ formula	Name	Description
f_{ij}		Flow from compartment *i* to compartment *j*
	Input of *j*	Flow entering compartment *j*
z_{ij}	Internal input	Flow from internal compartment *i* to internal compartment *j*
w_{kj}	Import; external input	Flow from external compartment *k* to internal compartment *j*
$x_i = \sum_{i=1}^{n} z_{ij}\ (1) + \sum_{k=1}^{r} w_{kj}\ (2)$	Total internal (1) and external (2) input of *j*	All internal (1) and external (2) flows entering compartment *j*
$\boldsymbol{w_{sj}}$	**Depletion**	**Abstract flow from the stock of *j* to compartment *j***
$\boldsymbol{I_i = \sum_{i=1}^{n} z_{ij}\ (1) + \sum_{k=1;\neq s}^{r} w_{kj}\ (2)}$	**Total real internal (1) and external (2) input of *j***	**All real internal (1) and external (2) flows entering compartment *j***
	Output of *j*	Flow leaving compartment *j*
z_{ji}	Internal output	Flow from internal compartment j to internal compartment i
v_{jk}	Export; external output	Flom from internal compartment *j* to external compartment *k*
$x_j = \sum_{i=1}^{n} z_{ji}\ (1) + \sum_{k=1}^{r} v_{jk}\ (2)$	Total internal (1) and external (2) output of *j*	All internal (1) and external (2) flows leaving compartment *j*
$\boldsymbol{v_{js}}$	**Increment**	**Abstract flow from compartment *j* to its stock**
$\boldsymbol{O_i = \sum_{i=1}^{n} z_{ji}\ (1) + \sum_{k=1;\neq s}^{r} v_{jk}\ (2)}$	**Total real internal (1) and external (2) output of *j***	**All real internal (1) and external (2) flows leaving compartment *j***
z_{jj}	Self-cycling of *j*	Flow from internal compartment *j* to itself
	On a system level	
$\sum_{i=1}^{n}\sum_{k=1}^{r} v_{jk}$	Total system input	All flows entering the system
$\sum_{k=1}^{r}\sum_{j=1}^{n} w_{kj}$	Total system output	All flows leaving the system

It is possible to include flows from compartments to themselves (see Figure 5.2 and Table 5.1), which is called self-cycling (Suh, 2005). This can e.g. occur for a compartment of carnivores of a food web of which some carnivores eat other carnivores (Chen et al., 2010). These self-cycling flows can be neglected even if they are large in quantity without changing the equality between total input and total output since the self-cycling flow is an input and an output flow. Inclusion or exclusion of self-cycling is another choice to be made in the methodology.

A prerequisite for matrix calculations and ENA is that all flow values should be exact values. However, a common problem with ecological systems is that the flows cannot be unequivocally determined using available data (Kones et al., 2009). There can be a high standard deviation on values or no exact value but only an interval is given. An exact value can then be determined based on common knowledge or to make the system balanced (see section 5.2.1.4). Mathematical solutions for this problem on a system level also exist (Kones et al., 2009).

5.2.1.4 Balancing

After collection of all data, mass and/or energy balances should be checked. Total input should equal total output of each compartment and of the total system (Equtation 5.1):

$$\forall\, i\colon \sum_{j=1}^{n} z_{ij} + \sum_{k=1}^{r} v_{ik} = \sum_{j=1}^{n} z_{ji} + \sum_{k=1}^{r} w_{ki} \quad (5.1)$$

If this is not the case, additional balancing should be performed. Balancing is an important step in an IOA, more specifically ENA, since it can have major consequences on the final data, and can be done in different ways. One way is to alter or choose flow values to balance data, e.g. if the value of the flow is not exactly known but an interval can be defined, a suitable value from this interval can be chosen which leads to a balanced system. Balancing is then obtained during flow quantification. Another option is to use calculation methods, as discussed by Allesina and Bondavalli (2003). Take note that these methods are purely based on mathematics and do not take into account ecological principles. Which balancing procedure(s) to use is again a methodological choice.

5.2.1.5 Construction of input-output table

Based on the principles introduced by Leontief (1936), input-output tables can be constructed from the balanced data. The basic structure of the input-output table of a system can be seen in Figure 5.3. Different parts of the tables are defined as matrices, with the nomenclature as given in Figure 5.3. Each *ij*-th element of the input-output table besides the vectors **x** and **x'** represents a flow from the compartment of row *i* to the compartment of column *j*. If balancing is properly performed, the sum of the elements of matrices **Z** and **V** of row *i* equals the sum of the elements of matrices **Z** and **W** of column *i*, as these represent the same compartment.

Row\column	Internal compartments	External export compartments (incl. stock for increment)	Total output
Internal compartments	Transaction matrix (**Z**) z_{ij}: flow from internal compartment i to internal compartment j	Export matrix (**V**) v_{ik}: flow from internal compartment i to external export compartment k	Total output matrix (**x**) x_i: total output of internal compartment i
External import compartments (incl. stock for depletion)	Import matrix (**W**) w_{kj}: flow from external import compartment k to internal compartment j		
Total input	Total input matrix (**x'**) x_j: total input of internal compartment j		

Figure 5.3. Basic input-output table with matrices. For non-steady state systems, according to the framework of Finn (1977, 1976), stock is included in the external export and import categories for increment and depletion, respectively.

During construction of the input-output table, the external compartments must be categorized as import (import flow value in matrix **W**) and/or export (export flow value in matrix **V**) external compartments for each internal compartment. If a compartment solely acts as a source or a sink for all internal compartments, receiving only import or export flows, respectively, categorization can be straightforward as an import or export external compartment, respectively. This is the case for the external compartments defined by Hirata and Ulanowicz (1984): import to the system is an import external compartment and export of usable products and export of unusable compartments are export external compartments.

5.2.1.6 Calculations

A large variety of calculations can be done using the input-output table and its matrices. Some calculation methods can be used to model a linear response of the system to a change in input to or output from the system (see section 5.7.1, pg. 160). This modelling method is used to quantify the amount of resources and emissions of a product's life cycle for a given product quantity in LCA studies based on IOA. In ENA, calculation methods are primarily used to obtain a set of values or indicators which characterize the system under study.

First, throughflow, an important term in ENA which is used in the calculations of many indicators, must be explained. Throughflow was introduced by Finn (1980, 1976) and defined as the total input and total output of a compartment (Table 5.2).

Table 5.2. Indicators of ecological network analysis used in this study, their notation and formula. Every indicator is designed to represent a certain aspect of the functioning of an ecosystem at system level, as given in the first column 'category'. Revisions needed for the non-steady approach of Finn (1977, 1976) are printed in bold. More information on the original indicators can be found in section 5.7.2, pg. 161, and specifically for Throughflow in 5.7.4, pg. 164.

Category	Indicator (notation)	Formula	Original reference
/	Throughflow of compartment *I* (T_i)	$\sum_{j=1}^{n} z_{ij} + \sum_{k=1}^{r} v_{ik} = \sum_{j=1}^{n} z_{ji} + \sum_{k=1}^{r} w_{ki}$	(Finn, 1980, 1976)
Activity	Total System Throughflow (TSTF)	$\sum_{i=1}^{n} \mathrm{T}_i$	(Finn, 1980, 1976)
Activity	**Revised** Total System Throughput (**r**TSTP)	$\sum_{i=1}^{n}\sum_{j=1}^{n} z_{ij} + \sum_{i=1}^{n}\sum_{k=1; \neq s}^{r} v_{jk} + \sum_{k=1; \neq s}^{r}\sum_{j=1}^{n} w_{kj}$	(Rutledge et al., 1976)
Cycling	Finn's Cycling Index (FCI)	$\frac{1}{\mathrm{TSTF}} \sum_{i=1}^{n} (\frac{l_{ii}^{a} - 1}{l_{ii}^{a}})^{b} \times \mathrm{T}_i$	(Finn, 1980)
Organisation	**Revised** Average Mutual Information-index (**r**AMI)	$\mathrm{k}^{c} \sum_{i=1; \neq s}^{n+r} \sum_{j=1; \neq s}^{n+r} \frac{f_{ij}}{\mathrm{TSTP}} \log_2 \frac{f_{ij} \times \mathrm{TSTP}}{I_j \times O_i}$	(Rutledge et al., 1976)
Development	**Revised** Ascendency[d] (**r**A)	$\sum_{i=1; \neq s}^{n+r} \sum_{j=1; \neq s}^{n+r} f_{ij} \log_2 \frac{f_{ij} \times \mathrm{TSTP}}{I_j \times O_i}$	(Ulanowicz, 1980)

[a] l_{ii} represents the *ii*-th element of the Leontief (and the Ghosh) invers matrix

[b] This term is called the cycling efficiency of compartment *i*

[c] By convention k is set equal to one (Latham II and Scully, 2002)

[d] Ascendency is AMI with the scalar k equal to TSTP and can thus be seen as the product of AMI (with k=1) and TSTP

It represents the quantity of matter flow through a compartment during the period of study. Vectors **x** and **x'** of the input-output table are by consequence the throughflow values of the different compartments. Notation of the throughflow of compartment *i* is 'T_i'. A vast set of indicators exist in the field of ENA (Latham II, 2006). Some of these indicators are based on the Leontief inverse matrix or the Ghosh matrix, which characterize the direct and indirect relationships between the different internal compartments of an ecosystem. In this chapter, as an example, only a set of frequently used indicators are used (Table 5.2): Total System Throughflow (TSTF), Total System Throughput (TSTP), Average Mutual Information index (AMI), Ascendency (A) and the Finn's Cycling Index (FCI). Of these indicators only the FCI is based on the Leontief (and Ghosh) inverse matrix. These indicators are more thoroughly explained in section 5.7.2, pg. 161. Calculations were performed using Microsoft Excel, see section 5.7.6, pg. 180.

5.2.2 Adaptations of the ENA methodology

5.2.2.1 Adaptations in quantification of flows

A general procedure is proposed here to collect all the data needed to quantify the different flows (in order of application):

1. Collect site and time specific data
    a. Empirical data
    b. Data obtained by modelling
2. If specific data are lacking, collect generic data
    a. Empirical data
    b. Data obtained by modelling
3. If not enough data could be collected, fill data gaps by using
    a. Input-output balance(s)
    b. Equations based on ecological principles
    c. Inclusion of net flow

If insufficient data is present for a direct calculation of a flow, three options are proposed here. Using input-output balances refers to mass or energy balances of the different compartments. Equations based on ecological principles are straightforward, e.g. 50% of the absorbed CO_2-C is stocked in biomass. The concept of 'net flow' has been used by Patten (1992) in network utility analysis which is extensively explained in the work of Fath (2007). Consider internal compartments *i* and *j* and a flow z_{ij} from *i* to *j* and a flow z_{ji} from *j* to *i*. In this case, it is possible to replace both flows by one net flow from *i* to *j* (z_{ij}-z_{ji}) or from *j* to *i* (z_{ji}-z_{ij}). This option can be chosen in case of a shortage in data or to exclude cycling, as cycling occurs between compartments *i* and *j* by means of the flows z_{ij} and z_{ji}. Also, if z_{ij} and z_{ji} are unknown, only the net flow z_{ij}-z_{ji} or z_{ji}-z_{ij} can be quantified using input-output balances.

5.2.2.2 External compartmentalisation

From an environmental point of view, the destination and sources of the flows leaving and entering the system, respectively, are crucial to know. For that purpose, we developed here a physical external compartmentalisation, dividing the environment in physical compartments (e.g. shore and ocean/sea for estuarine ecosystem). In that manner the export and import flows to and from these physical external compartments define their destination and source, respectively, and the flow quantity. A general methodology is introduced here, which allows any kind of external compartmentalisation in the ENA framework.

Choosing and defining other external compartments is not a difficult task. During construction of the input-output table, care must, however, be taken in the categorization of the external compartments (see section 5.2.1.5, pg. 142), as in a general approach external compartments may act both as a sink and a source on a system level (e.g. for an estuarine ecosystem, flow of water to and from the ocean) but also for each internal compartment specifically.

If an external compartment acts purely as a source or a sink for all internal compartments, categorization is straightforward in the **W** and **V** matrix, respectively, as is done for the external compartments of Hirata and Ulanowicz (1984).

For external compartments which act both as a sink (export external compartment) and a source (import external compartment) for an internal compartment, one can categorize it as one of the two (a value in the **W** or **V** matrix for internal compartment) or as both (a value in the **W** and one in the **V** matrix for the internal compartment). However, when categorizing the particular external compartment only as an export or import external compartment, both flows (export and import) will be reduced to a net flow (see section 5.2.2.1, pg. 145). For example if it is only categorized as an export external compartment, only a net flow value will be given in matrix **V** for the corresponding internal compartment. In that case, it can happen that the net flow value turns negative. If the external compartment is categorized as both, values will be present in the **W** and **V** matrices for the import and export flow, respectively. This categorization is an extra choice to be made as a consequence of this adaptation.

To be able to calculate the indicators AMI and A (Table 5.2, pg. 144), all values in the input-output table must be positive since a logarithmic function is used. Therefore, we propose the rule that categorization must always be such that all values in the **W** and **V** matrices of the input-output table are positive. When applying this rule, there still remains an option in the categorization procedure. More exactly, if an external compartment acts both as a sink and as a source for one specific internal compartment, either the export and import flows can be considered or only the positive-valued net flow. Categorization as such is an extra choice which needs to be made.

In the field of modelling (IOA), external compartmentalisation and categorization of external compartments provide additional options, for more information see section 5.7.3, pg. 163.

5.2.2.3 Non-steady state

Applying ENA on non-steady state systems has always been a difficult issue. Some solve this problem by modelling the system until it reaches steady state (Allesina and Bondavalli, 2003) or by assuming steady state by neglecting changes in storage and outbalancing the differences between inputs and outputs in the balancing procedure (see section 5.2.1.4, pg. 142). In those cases the term 'balancing' also refers to the process of obtaining a steady state, which is not the case in this study. A first major drawback of altering data to obtain a steady state is that not the system itself is studied but an abstract steady state of it. And the farther a system is from a steady state because of large stock changes, the bigger the differentiation between the studied system and its abstract steady state. Also, it is important to notice that if a model is used, time and effort needs to be invested in the selection of the model and fitting it to the network flow data. Next to that, the values of the steady state, including all further outcomes of calculations (indicators), depend on the used model. It must be noted that some models cannot reach a steady state for some systems. As a consequence, a second major drawback is that there is no consistency in the extent and manner of data alteration to obtain an abstract steady state.

A simple adequate manner to account for non-steady state systems was already introduced by Finn (1977, 1976). He introduced the abstract external compartment storage or stock (notation *s*) of an internal compartment (Figure 5.2; in bold). A flow between the internal compartment and its external stock compartment should be regarded as a change in stock. Regarding changes in indicators for non-steady state, throughflow, specifically, also included change in storage in the original definition of Finn (1980, 1976): if the change in storage was negative or positive, this change was considered as an output or input, respectively. Other definitions for throughflow in case of non-steady state have also been developed, but here we stick to the original definition. An overview on throughflow can be found in section 5.7.4, pg. 164. As such the throughflow (T_i) of a compartment is equal to x_i, the total input of i and the total output of i. Recent approaches by Schramski et al. (2011, 2009) do not consider stock as an abstract external compartment and by consequence do not (indirectly) include it in the calculation of indicators.

As an external compartment, the stock compartment needs to be categorized. Originally, the stock compartment was categorized both as an export and an import external compartment by Finn (1977, 1976) (see Figure 5.3, pg. 143). Consequently, an export flow from the internal compartment to the storage, representing increment (v_{js}) could be distinguished from an import flow from the stock compartment to the internal compartment, representing depletion (w_{sj}) (see Figure 5.2, pg. 140, and Table 5.1, pg. 141; in bold). A steady state can then mathematically be defined by equation (Equation 5.2).

$$\forall\, j: s_j = 0 \Leftrightarrow \forall\, j: v_{js} = 0 \; and \; w_{sj} = 0 \; (5.2)$$

However, more recently the stock compartment has been categorized as an export external compartment for all internal compartments (Latham II, 2006; Suh, 2005). This is also done in studies of economies using IOA by addressing capital changes as an export compartment. By categorizing it only as an export external compartment, the interpretation of certain indicators may not be valid anymore. This is the case for FCI (see section 5.7.2.2, pg. 162, for full explanation of FCI). If the stock compartment is categorized only as an export compartment, throughflow will not equal $\boldsymbol{x}_i$ if the stock of compartment i diminishes (negative export flow value). By consequence the elements l_{ii} of the Leontief inverse matrix will not represent the flow from compartment i to i per throughflow of i. Therefore, the cycling efficiency of compartment i (Table 5.2, pg. 144) cannot be regarded as its cycled throughflow fraction of compartment i and FCI not as the cycled part of the Total System Throughflow (TSTF) as it is defined. The genuine interpretation of FCI is thus no longer correct in that case. In the manual of Eurostat (2008) on the framework of Input-Output Analysis (IOA) of economies, changes in stock are also addressed in the same manner using abstract compartments, namely import for depletion and export for increment, for monetary flows called 'Consumption of fixed capital' and 'Formation of fixed capital', respectively. If the same framework is used in separate studies of human economies and ecological systems, these can be easier interlinked using that particular framework. For these two reasons, we consider the framework of Finn (1977, 1976), in which categorization of the stock compartment of an internal compartment as an export and import external compartment if there is an increment or depletion, respectively, as more adequate.

However because most ENA indicators are based on a steady state framework, their logicality needs to be revised. Total System Throughflow (TSTF) does not need to be further revised since it is the sum of the compartimental throughflows and throughflow is here defined with storage inclusion. Finn's cycling index (FCI) logicality is improved in this non-steady approach as mentioned above. After all, it is necessary to include change in storage in the concept of cycling. If there is a depletion of a compartment, a part of the output of a compartment originates from its stock and not from other compartments, which means it cannot be a cycled portion. If there is an increment of a compartment, a part of the input ends up as stock and is no longer available for circulation and thus also not for cycling. Mathematically, this comes to expression as follows: if there is a storage change in a compartment i, l_{ii} will be lower, resulting in a logical lower amount of cycling. TSTP represents the activity/growth of the system by being equal to the sum of all flows in the system (Rutledge et al., 1976; Ulanowicz, 1980). As the stock changes are not real physical flows, change in storage is excluded in the revised rTSTP calculation. rTSTP thus only accounts for the sum of all real flows (see formula in Table 5.2, pg. 144). AMI represents the amount of organization in a system as the assessment of the evenness in flow quantity between the different possible flows connecting the compartments of the system (Rutledge et al., 1976). As stock flows do not connect real physical compartments, it is more adequate to neglect stock changes in the calculation of AMI (see revised rAMI in Table 5.2, pg. 144). Ascendency is the product of TSTP and AMI. Since both factors exclude storage change, it is also excluded in their product: the revised Ascendency (rA) (Table 2). A simple manner to calculate these revised indicators is to set the stock change flows to zero, replace the elements of **x'** and **x** by the real inputs (*I*) and outputs (*O*) of the compartments (Table 1) and calculate the indicators in the original manner.

5.3 Case study

5.3.1 Case description

The studied ecosystem is the Scots pine stand as described in Introduction section 1.6 (pg. 12). The considered period studied is 2001-2002.

5.3.2 ENA study

The ENA framework with adaptations was applied on the casus, meaning data collection as proposed in section 2.2.1, external physical compartmentalisation and accounting for non-steady state using the methodology of Finn (1977, 1976). For additional information on the performed ENA study, we refer to section 5.7.5, pg. 165. Here the most important choices in the study and the results are elaborated.

In a managed forest ecosystem, the tree population is the most important biomass pool aimed at for human resource extraction. Internal compartmentalisation was done with that focus (Table 5.3). Trees were therefore divided into foliage, wood and roots to give a better picture of the change in the tree stand. The rest of the forest ecosystem was divided into soil, understory vegetation and the other aboveground organisms (e.g. herbivores, predators). However, the latter compartment was excluded for the Scots pine stand, as its stock and all ingoing and outgoing flows were negligible.

Table 5.3. Description of the internal compartments of a forest ecosystem in the case study

Internal compartment	Description	Content
Foliage	Aboveground overstory tree parts which perform photosynthesis	Foliage
Wood	Aboveground overstory tree parts which do not perform photosynthesis	Stem, branches and reproductive organs
Roots	Belowground overstory tree parts	Roots, incorporated symbiotic nitrogen-fixing micro-organisms and mycorrhizal funghi attached to the roots[a]
Understory vegetation	Plants not belonging to the overstory tree stand	Complete understory plants, incorporated symbiotic nitrogen-fixing micro-organisms and mycorrhizal funghi attached to the roots[a]
Soil	Soil of the ecosystem without plant roots itself and with a depth equal to that of the root zone	Layer of organic material (ectorganic layer), mineral soil containing the root zone, soil solution, all organisms living in the soil except plants and mycorrhizal funghi not directly attached to the roots[a]
Aboveground organisms	Aboveground living organisms different from vascular plants	Aboveground living heterotrophs and autotrophs different from vascular plants

[a] After removal and washing of roots

The external compartmentalisation is a physical one to assess the destiny and sources of ingoing and outgoing flows, respectively (Figure 5.4, Figure 5.5 and Figure 5.6, all given below). The environment was divided into the compartments atmosphere, human-industrial system, underlying soil and adjacent soil.

The three most important mass flows in a natural ecosystem are water (H_2O), carbon (C) and nitrogen (N). Most data on the considered Scots pine stand are also available for these three currencies. Hence, these three currencies were picked and data were collected for them (Figure 5.4, Figure 5.5 and Figure 5.6, all given below).

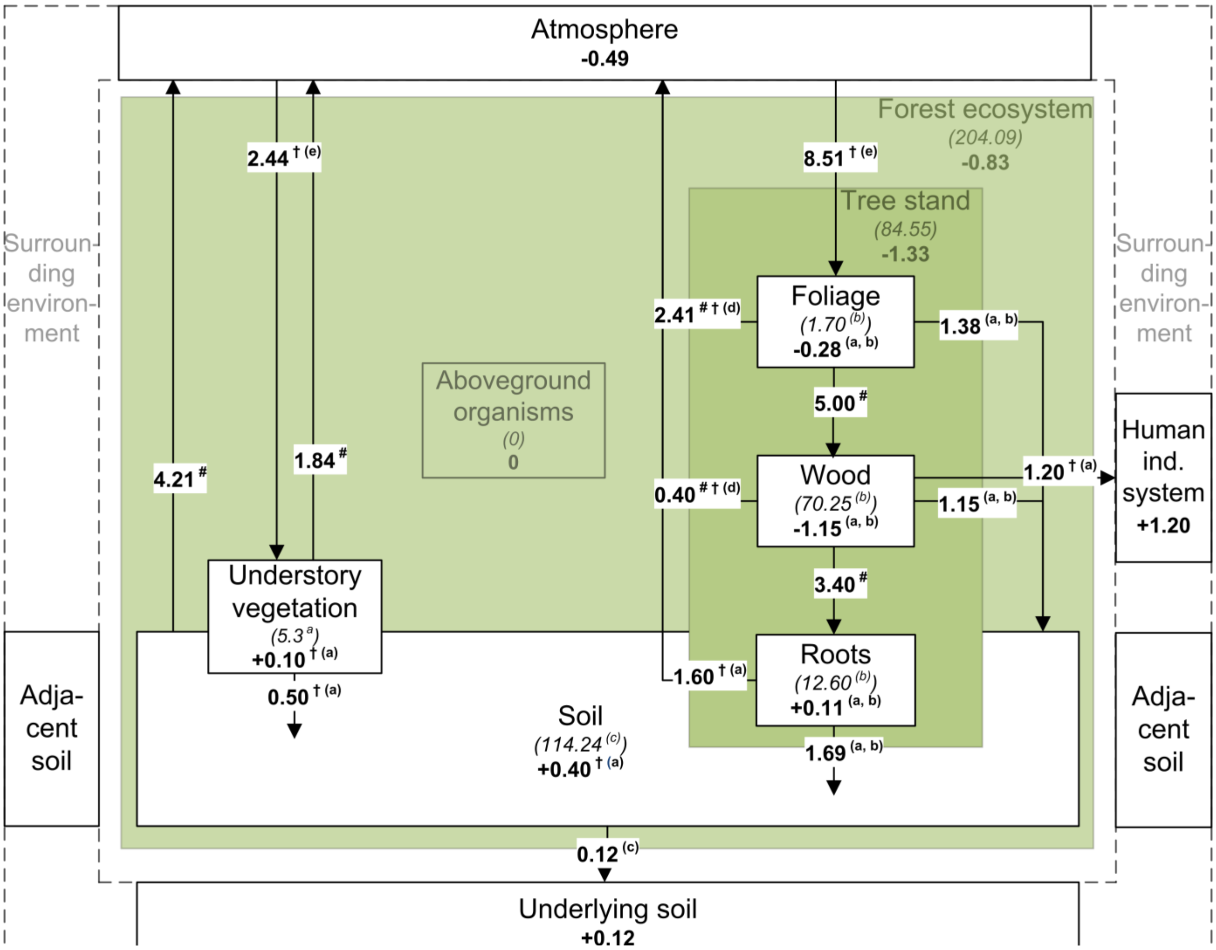

Figure 5.4. Carbon flows (ton C ha^{-1} yr^{-1}; in bold) and pools (ton C ha^{-1}; in italic between brackets) of the Scots pine stand with a wood harvest of 8 trees ha^{-1} yr^{-1}. Data are as much as possible based on the 2-ha studied Scots pine stand of the forest 'De Inslag' at Brasschaat (Belgium) during the period 2001-2002. A change in storage is depicted by a value in the compartment itself, with negative or positive values referring to depletion or increment, respectively. Superscript symbols mean that some data needed to calculate a flow value did not originate from reported site-specific measurements (†) or that input-output balances were used (#). The letter(s) between brackets in superscript refer to the reference(s) used to obtain this value: (a): Nagy et al. (2006); (b): Yuste et al. (2005); (c): Gielen et al. (2011); (d): Khomik et al. (2010); (e): Nagy et al. (2006), Sampson et al. (2006), Lamaud et al. (2001) and Misson et al. (2007).

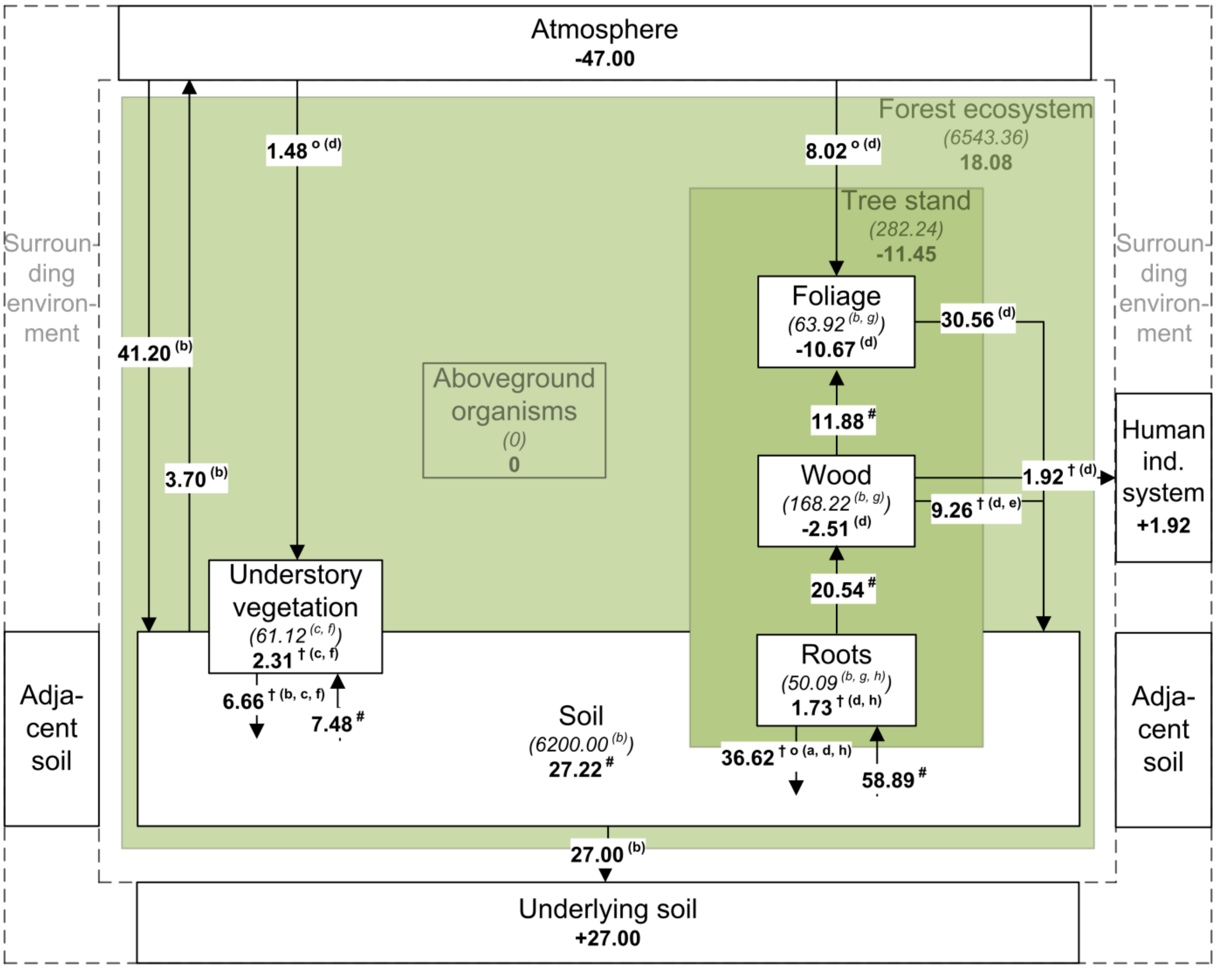

Figure 5.5. Nitrogen flows (kg N ha^{-1} yr^{-1}; in bold) and pools (kg N ha^{-1}; in italic between brackets) of the Scots pine stand with a wood harvest of 8 trees ha^{-1}yr^{-1}. See caption Fig. 3 for more information. If a rule of thumb was used, a superscript symbol is shown (°). References: (a): Verbeiren (1998); (b): Neirynck et al. (2008); (c): Nagy et al. (2006); (d): Nagy et al. (2006), Neirynck et al. (2008) & Yuste et al. (2005); (e): Mälkönen (1974); (f): Phyllis database (2011); (g): Yuste et al. (2005); (h): Gordon and Jackson (2000).

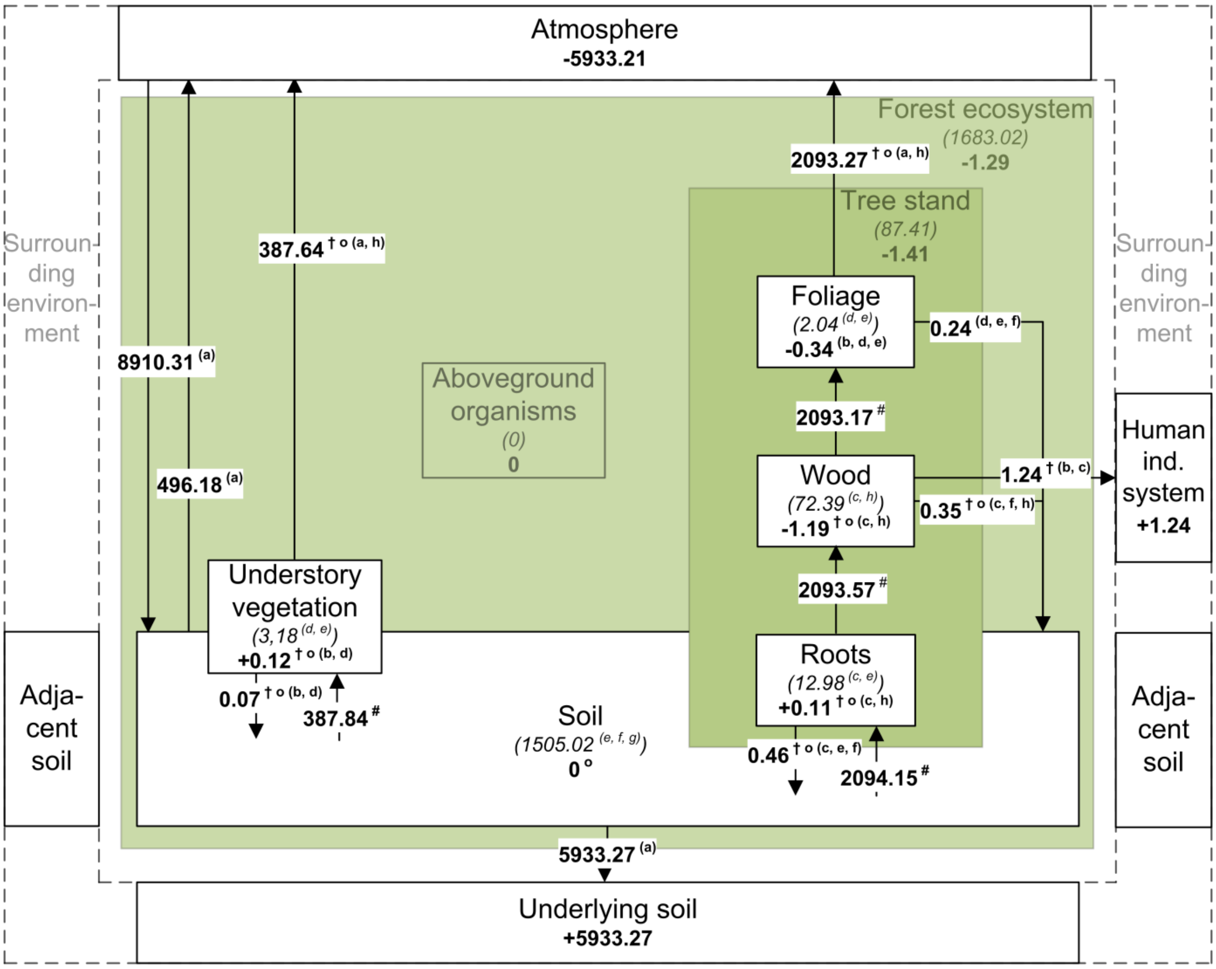

Figure 5.6. Water flows (ton H_2O ha^{-1} yr^{-1}; in bold) and pools (ton H_2O ha^{-1}; in italic between brackets) of the Scots pine stand with a wood harvest of 8 trees $ha^{-1}yr^{-1}$. See caption Fig. 4 for more information. References: (a): Gielen et al. (2010); (b): Nagy et al. (2006); (c): Kravka et al. (1999); (d): Gond et al. (1999); (e): Yuste et al. (2005); (f): Kelliher et al. (2004); (g): Verstraeten et al. (2005); (h): Nagy et al. (2006) and Yuste et al. (2005).

Using these data, five indicators were calculated (Table 5.4, shown below). When comparing Total System Throughflow (TSTF) and revised Total System Throughput (rTSTP), both representing activity, H_2O flux activity seems to be about 600 times larger than C flux activity, which in itself is about 100 times larger than N flux activity. This shows the large difference in flow quantity between these three important ecosystem currencies.

Table 5.4. Calculated indicators (cf. Table 5.2, pg. 144) for the Scots pine stand. The first column mentions what the indicators represent. TSTF: Total System Throughflow; ; rTSTP: revised Total System Throughput; FCI: Finn's Cycling Index; rAMI: revised Average Mutual Information-index; rA: revised Ascendency.

Category	Indicator	Carbon (ton C ha^{-1} yr^{-1})	Nitrogen (kg N ha^{-1} yr^{-1})	Water (ton H_2O ha^{-1} yr^{-1})
Activity	TSTF	25.51	245.77	15581.68
Activity	rTSTP	34.42	252.03	24490.23
Cycling	FCI[a]	0	0.40	0.00010
Organization	rAMI[a]	1.55	1.19	1.92
Development	rA	53.45	300.73	47096.93

[a] These indicators are dimensionless

There are a lot of differences between the Finn's Cycling Index (FCI) values of the networks. For C, FCI equals zero, as expected since no cycling occurs in the C flow network (Figure 5.4, pg. 151). The FCI for H_2O approaches zero due to the small amount of H_2O present in litter and slash which ends up on the soil and is later on taken up by the roots (Figure 5.6, pg. 153). The FCI of the N flow network of the forest stand was 0.40, which means that almost half of the total throughflow was recycled. Cycling of N occurs also due to N uptake by roots from the soil, which in turn receives N from the trees by the litter and slash.

The revised average Mutual Information index (rAMI) of H_2O is the highest, most probably due to the equality of the different transfer flows between the tree compartments and between the soil and the roots. The revised Ascendency (rA), representing development, is about 800 times larger for the H_2O network compared to that of C, which in itself is about 150 times larger compared to that of N. Since rA is the product of rAMI and rTSTP, the differences between the currencies in terms of rTSTP have been enlarged.

5.4 Influence of methodological choices

The change in indicator value output due to different methodological procedures, and thus also their interpretation, is very case specific; it depends on the choice (e.g. different internal compartmentalisation, inclusion of self-cycling) and the extent of the change (e.g. the specific alternative compartments, the quantity of the self-cycling flow). As an illustration different choice scenarios have been tested for the Scots pine stand in section 5.7.7, pg. 180. In general, only the possibility of each methodological choice to alter the indicator values can be determined. This is done here for the different selected indicators (Table 5.5, shown below).

Table 5.5. Influence of choices in methodological procedure on the given indicators (cf. Table 2). If the choice does not have an influence on the indicator, 'N' is shown. If there can be an influence, 'Y' is shown. If the choice always results in an increase or decrease, an '↑' or an '↓' is given, respectively. TSTF: Total System Throughflow; rTSTP: revised Total System Throughput; FCI: Finn's Cycling Index; rAMI: revised Average Mutual Information index; rA: revised Ascendency.

Methodological choice\Indicator	TSTF	rTSTP	FCI	rAMI	rA
Internal compartmentalisation	Y	Y	Y	Y	Y
External compartmentalisation	N	N	N	Y	Y
Quantification of flow(s)	Y	Y	Y	Y	Y
Inclusion of self-cycling flow(s)	Y (↑)	Y (↑)	Y (↑)	Y	Y
Inclusion of net flow(s) between internal compartments	Y (↓)	Y (↓)	Y (↓)	Y	Y
Balancing	Y	Y	Y	Y	Y
Categorization external compartment(s)	Y	Y	Y	Y	Y

Another type of internal compartmentalisation alters the input-output table in dimension and changes some values, possibly altering all indicator values. External compartmentalisation only changes rAMI and rA as only these terms deliberately take into account the elements of matrix **W** and **V** (see Table 5.2, pg. 144). Quantification of flows and balancing can have a direct influence as these procedures alter flow values directly. Specifically for the quantification of flows, including self-cycling flow(s) results clearly, amongst other effects, in more cycling and in a higher activity (rTSTP and TSTF). If a net flow is included, input and output of both particular compartments will drop, leading to lower TSTF and rTSTP values, and there will be no direct cycling between them, resulting in a lower FCI. Categorization of external compartment(s) changes the values of the **W** and **V** matrices (Figure 5.3, pg. 143) and indirectly those of **x** and **x'**. These changes can directly and indirectly alter the given indicator values.

5.5 Discussion

5.5.1 Successful specification of external flows; extending ENA/IOA methodology

The ENA/IOA methodology was successfully adapted to account for the specific destinations and sources of external flows by applying a physical external compartmentalisation, dividing the environment into physical compartments. However, categorization of external compartments as import and/or export external compartments during construction of an input-output table is now an additional choice to be made. In fact a framework has been set up to allow for any kind of external compartmentalisation, offering the following new possibilities.

In the field of LCA, ENA can be applied on product life cycles of an LCA if all flows are put in the same unit, without loss of information concerning destination and sources of the external flows. On the other hand, a flow network of an ENA study is more adequate to be implemented in the life cycle of an LCA study if they are altered to take into account sources and destinations of external flows, as has been done in the case study.

In IOA modelling, which is also used in LCA studies, any kind of external compartmentalisation can be chosen. This allows one to choose which kind of external response is calculated for a given external input. When using physical compartmentalisation, the quantity of export or import flows to specific destinations and from specific sources can be quantified, e.g. a linear approximation of the amount of carbon (C) leached to the underlying soil and other export flows of a forest ecosystem for a certain amount of C input into the system. It could also be applied in other IOA frameworks than that of ecosystems, e.g. of economies. As such, interactions of economies with its trading economies could be studied.

Consider an LCA performed on an integrated human/industrial – ecosystem, socio-ecological system, using the IOA framework. In such an LCA, an ENA can be performed on the ecosystem, the human/industrial part and/or the integrated system to calculate network indicators which deliver additional insight for the assessment of the sustainability of the studied life cycle or parts of it.

Note that another type of external compartmentalisation and/or categorization may have an influence on indicator values. For the tested indicators, another external compartmentalisation altered rAMI and rA, and another categorization may alter all of them.

5.5.2 Accounting for non-steady state systems in a simple adequate manner

The prerequisite of steady state for a system to be able to apply ENA on it is a major drawback in any field of application. Furthermore, the solution of assumption of steady state and the use of simulation models to obtain a steady state can alter results considerably and one studies a virtual steady state condition of the ecosystem and not the ecosystem itself. Because a model or assumption is used, there is no consistency in the extent of alteration of the results. To solve this problem, in this work we reverted to Finn's concept (1977, 1976) in which storage is an external compartment and categorized as an import or export external compartment if there is depletion or increment, respectively (see section 5.2.2.3, pg. 147). Compared to the concept applied by Suh (2005) and Latham II (2006), in which storage is only an export external compartment, the interpretation of FCI is correct, if there is a depletion. Compared to the concepts of Matamba et al. (2009) and Shevtsov et al. (2009), the main advantage is the simplicity. Next to that, the non-steady approach using abstract stock compartments is used in IOA of human economy systems (Eurostat, 2008). The construction of an ENA/IOA framework out of such an IOA of a human economy system and an ecosystem can therefore be easily set up. It is important to notice that the non-steady approach in this chapter influences the outcome values of ENA indicators. This has been studied for the set of indicators used in this study. FCI and TSTF appear more logical. The calculations for the other three indicators were revised, resulting in rTSTP, rAMI and rA.

5.5.3 Comparing (quality of) ecosystems using ENA (in LCA); a need for standardization

The choices made in the methodological procedure of ENA may alter the results considerably. All these choices except the influence of balancing, were tested in this study. But as balancing alters data, its influence is obvious. Though, balancing, in the context that it also includes obtainment of a steady state, has already been tested by Baird et al. (2009). They showed that balancing and internal compartmentalisation influenced the outcome of ENA using different scenarios on the same case study.

Consequently, to be able to compare ecosystems or one ecosystem over different periods in an appropriate manner using ENA, it is required to make the same choices regarding internal and external compartmentalisation, self-cycling, net flows, balancing (including necessity of steady state) and categorization of external compartments. However, in a strict sense, using the same choices is not sufficient, because e.g. balancing can occur in the same manner but the extent of the changes can differ meaningfully between different ENA studies, which reduces the credibility of comparing them. Nevertheless, standardization of the methodological procedure of ENA would already be a large step forward in the adequacy of ENA to compare flow networks of different studies. Specifically in the construction of endpoint indicators for ecosystem quality out of ENA indicators for LCA, standardisation should have a high priority. This lack in standard concerning compartmentalisation is already addressed by Pizzol et al. (2013).

In the case of food webs, a certain convention is already maintained in literature: self-cycling is included, net flows are not used and categorization and external compartmentalisation are done via the convention of Hirata and Ulanowicz (1984) (see section 5.2.1.1, pg. 139). Balancing and internal compartmentalisation have however no clear convention yet.

We propose two standards for ENA: the categorization of external compartments during construction of the input-output table should be as such that only positive values are obtained and, following Finn (1977, 1976), the stock compartment should be categorized as an export or import if there is increment or depletion, respectively.

Another issue in ENA is the interpretation of the different indicators, which is not straightforward and case-specific, demonstrated by the lack of consensus in the interpretation of some of these indicators and their vast number, as shown in the work of Latham II (2006). The inclusion of change in storage flows and management flows in the network provides an additional challenge in the interpretation of indicators. These flows are after all indirectly implemented in the calculation of indicators. For example, a full tree harvest of the complete Scots pine stand would lead to very high values for TSTF in the C and N cycles. We can therefore conclude that using ENA, and its indicators, to compare (eco)systems for any kind of purpose (including sustainability assessments such as LCA) should definitely happen in a standardized manner and with caution towards the interpretation and comparison of indicator values.

Regarding construction of endpoint indicators of ecosystem quality, ENA indicators prove to be a good match in representing damage to ecosystem quality as disruption in energy and matter flows. Because of their holistic nature, ENA indicators account for the complete ecosystem and also include indirect effects of e.g. wood harvest. Also, the concepts that some ENA indicators aim to represent are closely related to ecosystem quality. The indicators used in this study are all total ecosystem indicators and represent such concepts: activity (rTSTP and TSTF), organisation (rAMI), cycling (FCI) and development (rA) (elaborate interpretations of these indicators can be found in the section 5.7.2, pg. 161). Further research is therefore recommended and needed in the construction of endpoint indicators out of ENA indicators in general.

Concerning the results of the case study, because of the difference in applied methodology, it is not very useful to compare its outcome indicator values with that of other studies. For the Scots pine stand, it was possible to compare the indicators for C, H_2O and N as the same ENA procedure was used in all of them.

5.5.4 Adaptations in data collection

Regarding the data collection procedure (see section 5.2.2.1, pg. 145) to facilitate this difficult task in future ENA studies, it should be considered as a template and not as a standardized procedure.

Regarding the use of a net flow (see section 5.2.2.1, pg. 145), it offers a solution for lacking data and excludes cycling between two specific compartments. However, one should keep in mind that inclusion of net flows may alter the indicator values (see section 5.4, pg. 155).

5.5.5 Conclusions

ENA can be valuable in different manners in the research field of LCA. First, the ecosystem flow networks of ENA studies can be easily linked to the product life cycle of LCA studies based on IOA. Second, the alterations in ecosystem functioning caused by emissions and resource extraction can be represented by changes in ENA indicator values. More particularly, endpoint indicators (ISO, 2006a) in LCA studies could be based on the change in these indicator values, e.g. the change in Ascendency or Finn's Cycling Index (FCI) over a certain period of time. When studying for example a forestry life cycle production system, the effect of wood harvest and CO_2 emission on ecosystems could be addressed by such particular endpoint indicators. Third, the ENA indicators can assess the functioning of the industrial and/or ecological part of a product's life cycle. The proposed adapted ENA/IOA framework improves this functionality by being applicable to any (eco)system (steady state and non-steady state) in an adequate but simple manner and accounting for the compartmentalisation of the studied ecosystem its environment in any possible way. Specifically, using a physical external compartmentalisation, the environment is divided into physical compartments (e.g. atmosphere and industry) and thus flows leaving or entering the (eco)system can be linked to these, specifying their destinations or sources, respectively. In this manner, ENA studies can be applied in a more fitting way in sustainability assessment (such as LCA) if it is desired to identify and quantify the interacting flows between a system and its environment. Yet, for an adequate use of ENA as a comparative tool, standards should be implemented in its methodology as several of its procedural choices may considerably alter the results of the calculated indicator values. This is an important next step in the field of ENA.

5.6 Acknowledgements

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5.7 Supporting information

In this section additional information is given concering modelling framework and matrices (section 5.7.1), the used indicators (section 5.7.2), opportunities in modelling by external compartimentalisation (section 5.7.3), throughflow (section 5.7.4), ecological network analysis of the Scots pine stand (section 5.7.5), software used (section 5.7.6) and influence of methodological choices illustrated with the case study on the Scots pine stand (section 5.7.7).

5.7.1 Modelling framework and matrices

Input-output analysis originally was conceived by Leontief (1936) for quantifying the response of an economic system on a change of external output (demand of final products). The model constructed for that purpose is called the Leontief demand-driven model (Leontief, 1936).

Dividing the flow from compartment i to j (z_{ij}) of matrix $\mathbf{Z}$ by the total input of compartment j (x_j) results in the input coefficient of the ij-th element, a_{ij} ($=z_{ij}/x_j$), being the flow quantity from compartment i to j per unit input of compartment j. Doing this for all elements in the transaction matrix (**Z**) results in the input coefficient matrix (direct requirements matrix), **A** (=**Zx^**$^{-1}$), a normalized version of the transaction matrix.

Of the total balanced system, a balance can be made on an element (Equation 5.3) and matrix level (Equation 5.4). By substitution, matrix **A** can be implemented in the matrix version of the balance equation (Eq. 4) resulting in equation 5.5. The matrix (**I**-**A**) and (**I**-**A**)$^{-1}$ are called the Leontief and Leontief inverse matrix, respectively. Equation 5.5 shows that by using this model the total output of all compartments (**x**) can be calculated from the total export of all compartments (**Vi**), or in other words that the system response can be calculated from a change in system export.

$$x_i = \sum_{j=1}^{n} z_{ij} + \sum_{k=1}^{r} v_{ik} \quad (5.3)$$

$$\Leftrightarrow \; \mathbf{x} = \mathbf{Zi} + \mathbf{Vi} \quad (5.4)$$

$$\mathbf{x} = \mathbf{Ax} + \mathbf{Vi} \;\Leftrightarrow\; (\mathbf{I}\text{-}\mathbf{A})\,\mathbf{x} = \mathbf{Vi} \;\Leftrightarrow\; \mathbf{x} = (\mathbf{I}\text{-}\mathbf{A})^{-1}\mathbf{Vi} \quad (5.5)$$

As only the total export is taken into account (**Vi**), the amount of export external compartments has no influence on the resulting output values of the model for a given total export. However when using $\mathbf{v}_k$, a vector with only exports of external compartment *k*, instead of **Vi**, the part of the total output for each compartment necessary for the specific export flow to compartment *k* can be calculated, providing additional insight.

A variation on the Leontief demand-driven model has been developed by Ghosh (1958), and is called the Ghosh supply-driven model. In this model output coefficients b_{ij} (=z_{ij}/x_i) are calculated. The matrix (**I**-**B**) and (**I**-**B**)$^{-1}$ are the Ghosh and Ghosh inverse matrix, respectively. In the Ghosh model, response of the model to a certain external supply is calculated (**x'** = **i'W**(**I**-**B**)$^{-1}$). System ecologists have shown more interest in this model than in the original Leontief model since it simulates the response of a system to a change in system input (Suh, 2005). It can also be used to clarify the distribution of system inputs over the different compartments of the system. For an even more thorough explanation of these calculations, we refer to Suh (2005).

5.7.2 Explanation of used indicators

5.7.2.1 Total system throughflow and throughput

The total flow quantity in an ecosystem is seen as a good indicator of the activity in the ecosystem (Finn, 1980). There are two indicators to measure this total flow quantity:

Total System Throughflow (TSTF) and Total System Throughput (TSTP). By summing up all the throughflows of the compartments, the TSTF is obtained (Finn, 1980, 1976) (Table 5.2, pg. 144). The TSTP is the sum of all flows between all compartments (Rutledge et al., 1976; Ulanowicz, 1980) (Table 5.2, pg. 144).

5.7.2.2 Finn's cycling index

Cycling in ecosystems is an important phenomenon, particularly for nutrients. It is difficult to correlate cycling with maturity or development of an ecosystem because the relation appears to depend strongly on the considered case and nutrient (Baird et al., 1991; Kazanci et al., 2009). Odum (1969) observed that mature systems, as compared to developing ones, have a greater capacity to retain nutrients through cycling. On the contrary, enhanced cycling of carbon can be seen as a sign of a stressed community (Norton et al., 1992). Cycling is after all a buffering mechanism that allows ecosystems to face a shortage of nutrient inflows, which is site-dependent (Jørgensen, 2009). Cycling should thus be best regarded as a specific aspect of an ecosystems 'health'. Finn (1980) was the first one to use ENA to assign a value to cycling by introducing the Finn's Cycling Index (FCI) (Table 5.2, pg. 144 and Equation 5.6). This index is the fraction of throughflow flux that is cycled (TSTFc) relative to the total system throughflow flux (TSTF) (Equation 5.6). According to Finn (1980), TSTFc is the sum of the products of the cycling efficiency, RE_i, of each compartment and its throughflow, T_i. RE_i is the fraction of throughflow returning back to the compartment, the cycled fraction, and is calculated using the diagonal elements of the Leontief inverse matrix, l_{ii}, which represent the direct and indirect flow from a compartment to itself.

$$FCI = \frac{TSTFc}{TSTF} = \frac{1}{TSTF}\sum_{i=1}^{n} RE_i \text{ x } T_i = \frac{1}{TSTF}\sum_{i=1}^{n} \frac{l_{ii-1}}{l_{ii}} \text{ x } T_i \quad (5.6)$$

5.7.2.3 Average Mutual Information index

Another aspect of an ecosystem which could be quantified with ENA is the level of organization of the interrelationships between the different compartments of an ecosystem. In that context, Rutledge et al. (1976) applied an index of communication theory, the Average Mutual Information (AMI) index (Table 5.2, pg. 144), to ecological networks. For a good interpretation of this term, we refer to Latham II and Scully (2002). The AMI represents the organization inherent in a system because it captures the average amount of constraint exerted upon an arbitrary amount of mass as it flows from any one compartment to the next (Rutledge et al., 1976). In short, AMI can be seen as the assessment of the evenness in flow quantity between the different possible flows connecting the compartments of the system. A high value for AMI is correlated with a higher evenness. It is difficult to link AMI with development of an ecosystem as no consensus in literature can be found on this topic (Latham II, 2006; Latham II and Scully, 2002). Because of the log function in AMI (Table 5.2, pg. 144) it is essential that all values in the input-output table are positive.

5.7.2.4 Ascendency

Building further on the concept of AMI, Ulanowicz (1980) developed an indicator which encompasses the natural growth and organization of ecological systems and attempts to represent in a mathematical manner the development of an ecosystem. This indicator is called ascendency (A) and is simply the product of TSTP and AMI (Table 5.2, pg. 144). The term ascendency is more thoroughly discussed and explained in the works of Latham II and Scully (2002) and Latham II (2006) and there also appears to be no consensus on its interpretation.

5.7.3 Opportunities in modelling by external compartmentalisation and categorization of external compartments

For modelling purposes, the external compartmentalisation and categorization is important as it defines the input and simulated output flows. In this appendix the notation '^' signifies a diagonalisation of a matrix.

Define matrix **B**, containing the input coefficients of matrix **W**, being the w_{ij} element divided by x_j. Using the Leontief model, the import of the system (**W**) can be calculated out of a given export of the system, matrix **V**, the Leontief inverse matrix and matrix **B**:

$$\mathbf{W} = \mathbf{B}((\mathbf{I}\text{-}\mathbf{A})^{-1}\mathbf{Vi})\text{\^{}} \quad (5.7)$$

Define matrix **C**, containing the input coefficients of matrix **V**, being the v_{ij} element divided by x_i. Using the Ghosh model, the export of the system (**V**) can be calculated out of a given import of the system, matrix **W**, the Ghosh inverse matrix and matrix **C**:

$$\mathbf{V} = (\mathbf{i}'\mathbf{W}(\mathbf{I}\text{-}\ \bar{\mathbf{A}})^{-1})\text{\^{}}\mathbf{C} \quad (5.8)$$

By choosing the external compartments and how they are categorized, one can choose which external flows act as input and output of the used model. For example, if atmosphere is only considered as an external export compartment of all internal compartments of an ecosystem (i.e. only in **V**), the import of compounds such as carbon from the atmosphere to the compartments of the system cannot be calculated using the Leontief model for a given net export value.

5.7.4 Throughflow

Throughflow was introduced by Finn (1980, 1976) as the total input and total output of a compartment including the change in storage. If the change in storage was negative or positive, this change was considered as an output or input, respectively. Throughflow thus represents the quantity of matter flow through a compartment during the period of study. Notation of the throughflow of compartment *i* is T_i.

Some authors (Gattie et al., 2006; Schramski et al., 2011; Shevtsov et al., 2009) have not included storage in throughflow. As a consequence, a distinction is then made between input throughflow, representing all inputs exclusive change in storage of a compartment, and output throughflow, representing all outputs exclusive change in storage of a compartment when the system is not in a steady state[1]. This approach is advantageous to assess the system from an input or output point of view (see e.g. Schramski et al. (2011)).

However, for further calculations based on throughflow, a single throughflow value is needed. Without including storage, only if there is a steady state, input throughflow equals output throughflow for each compartment and a single value for throughflow is obtained. The necessary steady state can be obtained by assumption or by using a model. As modelling may influence the outcome of calculations and consequently the indicators based on throughflow, we stick to the original definition of Finn (1980, 1976) which does not require a steady state. We thus do not distinguish between input and output throughflow and regard throughflow equal to the total input and the total output.

[1] Kazanci et al. (2009) also excluded storage change in their definition of throughflow and defined throughflow as the sum of flows from an internal compartment to other internal compartments and to the environment (output throughflow definition of Schramski et al. (2011)). They did not define input throughflow but immediately equaled throughflow to output throughflow.

5.7.5 Additional information on Ecological network analysis of the Scots pine stand

5.7.5.1 System and system boundaries

For the soil we choose a depth up until the C horizon, which contains the parent material. In this Scots pine stand, the Cg horizon is at a depth of ±70-80 cm (Gielen et al., 2011), thus a depth of 75 cm was taken as a system boundary. Gielen et al. (2011) also considered the Cg horizon as the ecosystem boundary.

5.7.5.2 Compartmentalisation

A difficult issue for compartmentalisation in forests are mycorrhizal fungi, as these organisms are attached to the tree roots and form a web of hyphae all over the soil. Although root samples are washed after sampling, it is likely that not all mycorrhizal fungi are washed off then. Therefore, mychorrizal fungi were categorized both in the root and soil compartment (Table 5.3, pg. 150).

5.7.5.3 Selection of energy-matter flow currency

The three most important mass flows in a natural ecosystem are water (H_2O), carbon (C) and nitrogen (N). As most data on the Scots pine stand in Brasschaat were also available for these currencies, these were used in the case study.

5.7.5.4 Identification and quantification of flows

The flows between the compartments in C, H_2O and N needed to be identified. This has been done in general for a managed forest ecosystem, see Figure 5.7, pg. 166, and Table 5.6, pg. 166 (Duvigneaud, 1974; Verbeiren, 1998). For the H_2O cycle, metabolic water was not taken into account.

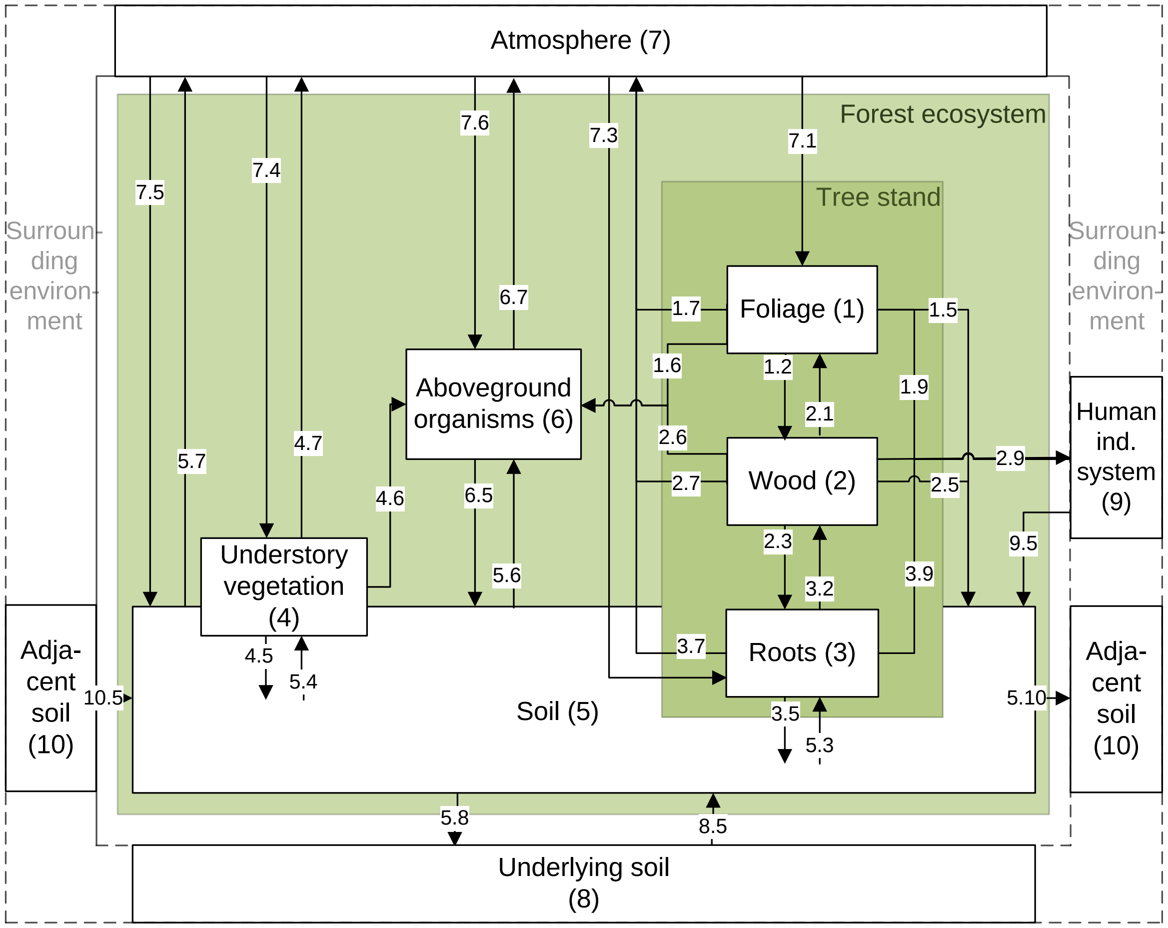

Figure 5.7. Identification of carbon (C), nitrogen (N) and water (H_2O) flows in a forest ecosystem with tree harvest included (Duvigneaud, 1974; Verbeiren, 1998). The number of a flow refers to a row in Table 5.6.

Table 5.6. Identification of carbon (C), nitrogen (N) and water (H_2O) flows in a forest ecosystem with tree harvest included (Duvigneaud, 1974; Verbeiren, 1998). The number of a flow in Figure 5.7 refers to a row in this table. If the flow is neglected or included in another flow for the specific case of the Scots pine stand, this is indicated in the last column (negligible, no data, not applicable or included in other flow). BVOC: emission of Biogenic Volatile Organic Compounds. NO_x: NO & NO_2 (gases); NO_y: $_pNO_3$, HNO_3 & HONO (p: particulate matter).

Number	From	To	Comp -ound	Description	Neglecting & explanation
1.2	Foliage	Wood	C	Transfer	
			N	Transfer	Included in net flow 2.1.N
			H_2O	Transfer	Included in net flow 2.1. H_2O
1.5	Foliage	Soil	C	Litter & slash	

			N	Litter & slash	
			H_2O	Litter & slash	
1.6	Foliage	Aboveground organisms	C	Herbivory	Negligible & no data
			N	Herbivory	Negligible & no data
			H_2O	Herbivory	Negligible & no data
1.7	Foliage	Atmosphere	C	Autotrophic respiration & BVOC	BVOC: negligible & no data
			N	Emission	Negligible & no data
			H_2O	Transpiration	
1.9	Foliage	Human industrial system	C	Harvest	Not applicable
			N	Harvest	Not applicable
			H_2O	harvest	Not applicable
2.1	Wood	Foliage	C	Transfer	Included in net flow 1.2.C
			N	Transfer	
			H_2O	Transfer	
2.3	Wood	Roots	C	Transfer	
			N	Transfer	Included in net flow 3.2.N
			H_2O	(transfer)	Included in net flow 3.2. H_2O
2.5	Wood	Soil	C,	Litter & slash	
			N	Litter & slash	
			H_2O	Litter & slash	
2.6	Wood	Aboveground organisms	C	Herbivory	Negligible & no data
			N	Herbivory	Negligible & no data

			H_2O	Herbivory	Negligible & no data
2.7	Wood	Atmosphere	C	Autotrophic respiration	
			N	/	
			H_2O	Transpiration	Negligible & no data
2.9	Wood	Human industrial system	C	Harvest	
			N	Harvest	
			H_2O	Harvest	
3.2	Roots	Wood	C	Transfer	Included in net flow 2.3.C
			N	Transfer	
			H_2O	Transfer	
3.5	Roots	Soil	C	Litter, slash, root exudates, herbivory	Herbivory & root exudates: negligible & no data
			N	Litter, slash, herbivory	Herbivory: negligible & no data
			H_2O	Litter, slash, herbivory	Herbivory: negligible & no data
3.7	Roots	Atmosphere	C	Autotrophic respiration	
			N	/	
			H_2O	Transpiration	Negligible & no data
3.9	Roots	Human industrial system	C	Harvest	Not applicable
			N	Harvest	Not applicable
			H_2O	Harvest	Not applicable

4.5	Under-story vegeta-tion	Soil	C	Litter, root exudates, herbivory	Herbivory & root exudates: negligible & no data
			N	Litter, herbivory	Herbivory: negligible & no data
			H_2O	Litter, herbivory	Herbivory: negligible & no data
4.6	Under-story vegeta-tion	Aboveground organisms	C	Herbivory	Negligible & no data
			N	Herbivory	Negligible & no data
			H_2O	Herbivory	Negligible & no data
4.7	Under-story vegeta-tion	Atmosphere	C	Autotrophic respiration, BVOC	BVOC: negligible & no data
			N	Emission	Negligible & no data
			H_2O	Transpiration	
5.3	Soil	Roots	C	/	
			N	Uptake & transfer from external mycorrhiza	
			H_2O	Uptake & transfer from external mycorrhiza	
5.4	Soil	Understory vegetation	C	/	
			N	Uptake & transfer from external mycorrhiza	
			H_2O	Uptake & transfer from external mycorrhiza	
5.6	Soil	Aboveground organisms	C	Carnivory, herbivory	Negligible & no data
			N	carnivory & herbivory	Negligible & no data

			H_2O	Carnivory, herbivory & water uptake	Negligible & no data
5.7	Soil	Atmosphere	C	Heterotrophic respiration, autotrophic respiration, Slash decomposition, methane emissions	Methane emissions: negligible & no data
			N	emissions of (NH_3, (N_2), NO_x,NO_y & N_2O)	
			H_2O	Soil evaporation	
5.8	Soil	Underlying soil	C	Seepage	
			N	Seepage	
			H_2O	Drainage	
5.10	Soil	Adjacent soil	C	Surface & subsurface runoff	Negligible & no data
			N	Surface & subsurface runoff	Negligible & no data
			H_2O	Surface & subsurface runoff	Negligible & no data
6.5	Above-ground organ-isms	Soil	C	Deposits	Negligible & no data
			N	Deposits	Negligible & no data
			H_2O	Deposits	Negligible & no data
6.7	Above-ground organ-isms	Atmosphere	C	Heterotrophic & autotrophic respiration	Negligible & no data
			N	/	
			H_2O	Transpiration	Negligible & no data
7.1	Atmo-sphere	Foliage	C	Uptake	
			N	Uptake	
			H_2O	Uptake	Negligible & no

					data
7.3	Atmo-sphere	Roots	C	/	
			N	Symbiotic nitrogen fixation	Not applicable
			H_2O	/	
7.4	Atmo-sphere	Understory vegetation	C	Uptake	
			N	Uptake (via cuticula en stomata)	
			H_2O	Uptake	Negligible & no data
7.5	Atmo-sphere	Soil	C	Gas uptake, throughfall & stemflow (dissolved organic carbon)	Negligible
			N	Nitrogen fixation, stemflow & throughfall (inorganic & organic nitrogen)	Nitrogen fixation: not applicable stemflow: negligible
			H_2O	Stemflow & throughfall (rainfall – interception evaporation)	
7.6	Atmo-sphere	Aboveground organisms	C	Uptake	Negligible & no data
			N	Uptake	Negligible & no data
			H_2O	Uptake	Negligible & no data
8.5	Under-lying soil	Soil	C	Uptake from groundwater	Negligible & no data
			N	Uptake from groundwater	Negligible & no data
			H_2O	groundwater	Negligible & no data
9.5	Human industrial	Soil	C	Fertilizer	Not applicable
			N	Fertilizer	Not applicable

	system		H_2O	irrigation	Not applicable
10.5	Adjacent soil	Soil	C	Surface & subsurface runoff	Negligible & no data
			N	Surface & subsurface runoff	Negligible & no data
			H_2O	Surface & subsurface runoff	Negligible & no data

Not all identified flows were quantified for the Scots pine stand. This could be because the flow was not applicable for the stand, it was (presumably) negligible and no data was available or it was included in a net flow. With respect to data collection for quantification, the procedure mentioned in the methodology (see section 5.2.2.1, pg. 145) was followed.

Few values in Figure 5.4, Figure 5.5 and Figure 5.6 were calculated using only site and time-specific data of the Scots pine stand during 2001-2002. We indicated if an input-output balance or a rule of thumb was used for the quantification or if no site-specific data were used to calculate the flow value. Besides that, most values were not time-specific. However, the extent of necessity of a value to be time and/or site-specific depends on its nature, e.g. the flux of N from soil to atmosphere was 3.7 kg N ha^{-1} yr^{-1} (Figure 5.5) over the period 1996-2007, deviating from the goal period 2001-2002, but it did not change much over the different years since the standard deviation was only 0.3 kg N ha^{-1} yr^{-1}. During quantification often no exact values could be obtained but no mathematical methods were applied to obtain the exact values, with a preference for an average data value, if available, or for the most opportune value in case only an interval was available.

Next to that, it was impossible to match all data found in literature, for example: the total ecosystem respiration (TER) in our database amounts to 10.46 ton C ha^{-1} yr^{-1} as a sum of all individual respiration flows while from data in literature (Gielen et al., 2011) a TER of 9.65 ton C ha^{-1} yr^{-1} could be derived. But, as shown in this example, differences were always small.

5.7.5.5 Self-cycling

Self-cycling was deliberately excluded in this case study because it was not possible to quantify these values in a realistic manner for all compartments.

5.7.5.6 Net flows

The transfer between foliage and wood and the transfer between wood and roots are considered as net flows for C, H_2O and N. In reality there are flows going both directions between these compartments, e.g. C flow of photosynthesis products from foliage to wood during day and from wood to foliage at night. However, we did not want to account for the cycling between different tree compartments as we solely wanted to assess for cycling on a system level. Next to that, no data were available for the flows between these compartments. As a consequence, they could only be quantified using input-output balances, which resulted in the net flow values.

5.7.5.7 Short conclusion on data collection

In this case study, generating a database using literature data was a cumbersome task due to different reasons. As the available data were not gathered by the authors of the consulted articles for the purpose of a holistic ecological network analysis, some data were lacking, data originated from different time periods and data did not always fit. Even though the compartmentalisation of the forest ecosystem and its surrounding appears simple, quantifying the flows between all compartments was not straightforward. Nevertheless, to the best of our knowledge, the database generated still is exceptional in the quantification of all those flows within a single forest ecosystem.

Because of the difficulties in the streamlining of data from literature to the ENA accounting framework, the quality of the data is variable, but the data are definitely realistic. It is very understandable why other studies collect data specifically aimed for an ENA study. However, this study already shows that literature contains a vast amount of data and information which support ENA studies, although it is recommended to pick a well-studied ecosystem.

5.7.5.8 Balancing

In this case study, balancing was needed for the water cycle of the Scots pine stand. The total input did not equal the total output of the quantified flows of the soil compartment and the exact change in water stock is not known. Using the knowledge that the mean annual soil water content is relatively stable, we set the change in stock of the soil compartment equal to zero as a rule of thumb. Output flows were balanced by contributing part of the difference between the flows weighted by their relative quantity. For the other data, balancing was not needed since a lot flows were determined using input-output balances which indirectly made the data balanced.

5.7.5.9 Construction of input-output tables

Input-output tables were constructed for C, H_2O and N of the Scots pine stand (Table 5.7, Table 5.8 and Table 5.9). The grey rows and columns in the tables can be left out for the Scots pine stand. The throughflow values of the different compartments, also needed for some indicators, do not need to be separately calculated any more since these are equal to the total input and output of each compartment and are visible in the input-output tables as the vectors $\mathbf{x}$ and $\mathbf{x}$'.

Table 5.7. Carbon input-output table (ton C $ha^{-1} yr^{-1}$) of a Scots pine stand. Grey categories can be left out.

Column (j) / Row (i)	Foliage	Wood	Roots	Under-story	Soil	Above-ground organisms	Adjacent soil	Atmos-phere	Human ind. system	Under-lying soil	Incre-ment	Total output
Foliage	0	5.00	0	0	1.38	0	0	2.41	0	0	0	8.80
Wood	0	0	3.40	0	1.15	0	0	0.40	1.20	0	0	6.15
Roots	0	0	0	0	1.69	0	0	1.60	0	0	0.11	3.40
Understory	0	0	0	0	0.50	0	0	1.84	0	0	0.10	2.44
Soil	0	0	0	0	0	0	0	4.21	0	0.12	0.40	4.73
Above-ground organisms	0	0	0	0	0	0	0	0	0	0	0	0
Adjacent soil	0	0	0	0	0	0						
Atmosphere	8.51	0	0	2.44	0	0						
Human ind. system	0	0	0	0	0	0						
Underlying soil	0	0	0	0	0	0						
Depletion	0.28	1.15	0	0	0	0						
Total input	8.80	6.15	3.40	2.44	4.73	0						

Table 5.8. Nitrogen input-output table (kg N ha^{-1} yr^{-1}) of a Scots pine stand. Grey categories can be left out.

Column (j) / Row (i)	Foliage	Wood	Roots	Under-story	Soil	Above-ground organisms	Adjacent soil	Atmo-sphere	Human ind. system	Under-lying soil	Incre-ment	Total output
Foliage	0	0	0	0	30.56	0	0	0	0	0	0	30.56
Wood	11.88	0	0	0	9.26	0	0	0	1.92	0	0	23.05
Roots	0	20.54	0	0	36.62	0	0	0	0	0	1.73	58.89
Understory	0	0	0	0	6.66	0	0	0	0	0	2.31	8.96
Soil	0	0	58.89	7.48	0	0	0	3.70	0	27.00	27.22	124.30
Above-ground organisms	0	0	0	0	0	0	0	0	0	0	0	0
Adjacent soil	0	0	0	0	0	0						
Atmosphere	8.02	0	0	1.48	41.20	0						
Human ind. system	0	0	0	0	0	0						
Underlying soil	0	0	0	0	0	0						
Depletion	10.67	2.51	0	0	0	0						
Total input	30.56	23.05	58.89	8.96	124.30	0						

Table 5.9. Water input-output table (ton H_2O ha^{-1} yr^{-1}) of a Scots pine stand. Categories in grey can be left out.

Column (j) / Row (i)	Foliage	Wood	Roots	Under-story	Soil	Above-ground organisms	Adjacent soil	Atmo-sphere	Human ind. system	Under-lying soil	Incre-ment	Total output
Foliage	0	0	0	0	0.24	0	0	2093.27	0	0	0	2093.51
Wood	2093.17	0	0	0	0.35	0	0	0	1.24	0	0	2094.76
Roots	0	2093.57	0	0	0.46	0	0	0	0	0	0.11	2094.15
Understory	0	0	0	0	0.07	0	0	387.64	0	0	0.12	387.84
Soil	0	0	2094.15	387.84	0	0	0	496.18	0	5933.27	0	8911.43
Above-ground organisms	0	0	0	0	0	0	0	0	0	0	0	0
Adjacent soil	0	0	0	0	0	0						
Atmosphere	0	0	0	0	8910.31	0						
Human ind. system	0	0	0	0	0	0						
Underlying soil	0	0	0	0	0	0						
Depletion	0.34	1.19	0	0	0	0						
Total input	2093.51	2094.76	2094.15	387.84	8911.43	0						

5.7.5.10 Calculations

Firstly, the Leontief inverse matrices for C, H_2O and N of the Scots pine stand were determined. In this case, this was necessary to calculate the FCI. For carbon, Equation 5.9 gives the Leontief matrix $(\mathbf{I}\text{-}\mathbf{A})$ and Equation 5.10 the Leontief inverse matrix $(\mathbf{I}\text{-}\mathbf{A})^{-1}$.

$$(\mathbf{I}-\mathbf{A}) = \begin{bmatrix} 1 & -0.81 & 0 & 0 & -0.29 \\ 0 & 1 & -1 & 0 & -0.24 \\ 0 & 0 & 1 & 0 & -0.36 \\ 0 & 0 & 0 & 1 & -0.11 \\ 0 & 0 & 0 & 0 & 1 \end{bmatrix} \quad (5.9)$$

$$(\mathbf{I}-\mathbf{A})^{-1} = \begin{bmatrix} 1 & 0.81 & 0.81 & 0 & 0.78 \\ 0 & 1 & 1 & 0 & 0.60 \\ 0 & 0 & 1 & 0 & 0.36 \\ 0 & 0 & 0 & 1 & 0.11 \\ 0 & 0 & 0 & 0 & 1 \end{bmatrix} \quad (5.10)$$

For nitrogen, the Leontief, $(\mathbf{I}\text{-}\mathbf{A})$ and Leontief inverse, $(\mathbf{I}\text{-}\mathbf{A})^{-1}$, matrices are given in Equations 5.11 and 5.12, respectively.

$$(\mathbf{I}-\mathbf{A}) = \begin{bmatrix} 1 & 0 & 0 & 0 & -0.25 \\ -0.39 & 1 & 0 & 0 & -0.07 \\ 0 & -0.89 & 1 & 0 & -0.29 \\ 0 & 0 & 0 & 1 & -0.05 \\ 0 & 0 & -1 & -0.83 & 1 \end{bmatrix} \quad (5.11)$$

$$(\mathbf{I}-\mathbf{A})^{-1} = \begin{bmatrix} 1.17 & 0.43 & 0.48 & 0.40 & 0.48 \\ 0.50 & 1.30 & 0.33 & 0.28 & 0.33 \\ 0.65 & 1.67 & 1.88 & 0.73 & 0.88 \\ 0.04 & 0.09 & 0.11 & 1.09 & 0.11 \\ 0.68 & 1.75 & 1.96 & 1.64 & 1.96 \end{bmatrix} \quad (5.12)$$

For water, the Leontief, $(\mathbf{I}\text{-}\mathbf{A})$ and Leontief inverse, $(\mathbf{I}\text{-}\mathbf{A})^{-1}$, matrices are given in Equations 5.13 and 5.14, respectively.

$$(\mathbf{I}-\mathbf{A}) = \begin{bmatrix} 1 & 0 & 0 & 0 & -0.000027 \\ -0.9998 & 1 & 0 & 0 & -0.000039 \\ 0 & -0.9994 & 1 & 0 & -0.000052 \\ 0 & 0 & 0 & 1 & -0.0000083 \\ 0 & 0 & -1 & -1 & 1 \end{bmatrix} \quad (5.13)$$

$$(\mathbf{I}-\mathbf{A})^{-1} = \begin{bmatrix} 1.000027 & 0.000027 & 0.000027 & 0.000027 & 0.000027 \\ 0.999904 & 1.000066 & 0.000066 & 0.000066 & 0.000066 \\ 0.999388 & 0.999551 & 1.000118 & 1.000118 & 0.000118 \\ 0.000008 & 0.000008 & 0.000008 & 0.000008 & 0.000008 \\ 0.999397 & 0.999560 & 1.000126 & 1.000126 & 1.000013 \end{bmatrix} \quad (5.14)$$

5.7.6 Software used

Different free programs are available which perform ecological network analysis: NetMatCalc (Latham II, 2006), Wand (Allesina and Bondavalli, 2004), Econet (Schramski et al., 2011) and Ecopath from EwE (Christensen and Walters, 2004). The adapted framework of this study cannot be applied in any of them. For this study, Microsoft Excel was used as a calculation tool.

This could also have been done using the netindices package (Kones et al., 2009) (http://cran.r-project.org/web/packages/NetIndices/index.html) in the R-software, available for free. For the revised Ascendency, Average Mutual Information index and Total System Throughput, abstract export and import flows should be set equal to zero.

5.7.7 Influence of methodological choices illustrated with the case study on the Scots pine stand

This section illustrates the influence of choices in methodology with data of the case study. Different scenarios are presented in which other choices are made in the methodology and the corresponding changes of indicator values for these scenarios are given. The change in indicator values, and thus also their interpretation, depends on the case study, the type of change and the extent of the change.

5.7.7.1 Compartmentalisation

In a scenario in which roots, wood and foliage are combined into one tree compartment, the calculated indicators of the nitrogen flow network change accordingly for the Scots pine stand: TSTF: -14%, rTSTP: -12%, FCI: +19%, rAMI: -30% and rA: -48%. The type of internal compartmentalisation thus can have a profound effect on the outcome of an ENA.

If there would be only one external compartment besides the compartments for change in storage, this would result in the following changes for the carbon cycle: 0% for TSTF, rTSTP and FCI and -8.3% for rAMI and rA. Only AMI and A alter as only these terms deliberately take into account the destination or the source of an export or import flow, respectively (Table 5.2, pg. 144). So, choices regarding external compartmentalisation also influence the indicator values.

5.7.7.2 Identification and quantification of flows

Which flows are identified and quantified has an obvious direct influence on the calculations. For example include self-cycling, suppose detrivores on the soil eat organic matter at a flow rate of 1 ton C ha^{-1} yr^{-1}, being self-cycling for the soil compartment. This has the following effects on the calculated indicators of the carbon flow network of the Scots pine stand: TSTF: +3.9%, rTSTP: +2.9%, FCI increases from 0 to 0.04, rAMI: -4.5% and rA: -1.7%. Concequently, including self-cycling results clearly, amongst other effects, in more cycling and in a higher activity (TSTP and TSTF).

With respect to including a net flow, e.g. suppose a N retranslocation flow of 5 kg N ha^{-1} yr^{-1} of the foliage is included, meaning there is an additional flow of 5 kg N ha^{-1} yr^{-1} from foliage to wood and the flow from wood to foliage is increased with 5 kg N ha^{-1} yr^{-1}. The calculated indicators for nitrogen then alter in the following manner for this case study: TSTF: +4.1%, rTSTP: +4.0%, FCI: +3.1%, rAMI: -0.4% and rA: 3.6%. The use of one net flow instead of two flows between two compartments influences the outcome of an ENA; total input and output of the both compartments will drop and there will be no direct cycling between them.

5.7.7.3 Categorization of external compartments

Suppose that in the case study atmosphere is only categorized as an external import compartment in the N flow network (by doing so the flow from soil to atmosphere in Fig. 5 is removed and the one from atmosphere to soil becomes a net flow and decreases with 3.70 kg N $ha^{-1}yr^{-1}$). This has the following consequences on the calculated indicators: TSTF: -1.5%, rTSTP: -2.9%, FCI: +3.0%, rAMI: -0.2% and rA: -2.8%. The categorization of the external compartments during construction of input-output tables has an influence on the outcome of the results.

Regarding the stock compartment, if we follow the convention and categorize this only as an external export compartment, the calculated indicators of the nitrogen flow network of the Scots pine stand change accordingly: TSTF: -5.4%, rTSTP: +5.2%, FCI: +22%, rAMI: +1.2% and rA: +6.5%.

Chapter 6 Conclusions and perspectives

6.1 Conclusions

As a whole, we improved important aspects of the environmental sustainability assessment of the mankind-nature relationship, this in particular for forests, a major terrestrial biome. Case studies to illustrate the pertinence of the proposed improvements were all performed for a specific Scots pine forest stand, described in the Introduction, section 1.6 (pg. 12).

6.1.1 A step forward in environmental sustainability assessment (chapter 2)

A Life Cycle Analysis (LCA)-framework was developed, in which environmental impacts and benefits of an integrated human/industrial-natural system can be assessed. This framework, introduced in chapter 2, is a step forward in environmental sustainability assessment for two reasons. Firstly, natural and human/industrial system were accounted for as a whole and not just as separate systems, giving a broader, more correct image of the life cycle of products in the ecosphere. Secondly, besides damaging effect of the studied system also the beneficial effects of uptake of harmful compounds (e.g. CO_2) were considered. A case study was performed on the impact caused by the production system, life cycle, of 1 m^3 sawn timber, encompassing wood growth in the Scots pine stand and further processing into sawn timber and final wood burning with electricity generation in the human/industrial system. The results indicate that the (wood growth in the) forest was responsible for the larger share of the environmental impact/benefit. Because the forest was intensively managed, this implied a biodiversity loss compared to a natural system. This loss, representing damage to ecosystem quality, was responsible for almost all biodiversity loss over the complete life cycle: 1.60E-04 species*yr m^{-3} sawn timber. Concerning quantification of biodiversity loss, more research is though needed and ongoing to address this in a better manner (de Souza et al., 2013; Koellner and Geyer, 2013; Verheyen et al., 2013). Next to that, since the Scots pine stand is a plantation and managed intensively, the growth of biomass from natural vegetation is strongly prevented, leading to the main loss of natural resources per amount of sawn timber, expressed in exergy (the amount of useful energy obtainable out of a resource, e.g. exergy content of biomass): 3.99E+02 GJ_{ex} m^{-3}. This approach for resource consumption can be questioned as this impact is so high, just by considering the managed Scots pine stand as a non-natural one (R. A. F. Alvarenga et al., 2013). Regarding impact on human health over the life cycle, a total prevention of 1.40E-02 disability adjusted life years m^{-3} sawn timber is obtained. This health remediating effect could be mainly attributed for 77% to the deposition of particulate matter < 2.5 µm

($PM_{2.5}$) on the vegetative canopy of the Scots pine stand, and to CO_2 uptake for the other share. This case study revealed the potential importance of considering impact of ecosystems in environmental sustainability assessment, more specifically LCA.

No influence of human activities on the Scots pine stand, e.g. through forest management, was yet assessed in chapter 2. A dynamic approach is needed that addresses these indirect effects. Therefore, a dynamic forest growth model was selected, namely ANAFORE (Deckmyn et al., 2011, 2008), used in chapter 3 and 4.

6.1.2 Modelling particulate matter removal by a forest canopy (chapter 3)

Particulate matter (PM) deposition is an important ecosystem service, this especially in highly industrialized and densely populated regions such as Flanders. In chapter 2 we even illustrated that it is one of the most relevant benefits provided by the studied Scots pine stand. The selected ANAFORE forest growth model, used later on, did however not account for this process. A modelling framework to assess the removal of airborne PM by forest in a dynamic and better manner was thus needed. Subsequent to dry deposition of PM on the tree, different processes may though still occur with/to this deposited PM: delayed resuspension and removal via washoff through precipitation, encapsulation into wax layer, dissolution and plant uptake. It is only these latter processes which lead to definitive removal. The developed model (CIPAM: Canopy Interception and Particulate Matter removal model) included washoff of PM via rainfall and PM resuspension, neglecting the other processes. CIPAM builds further on and improves the methodology of Nowak et al. (2013) on this matter. Ours comprises the following modules: calculation of wind speed profile over the forest canopy (1), of rainfall interception, evaporation and throughfall (2), and calculation of PM deposition, resuspension and removal through washoff via rainfall (3). The calculation of the wind speed profile is essential as wind speed is a driver for PM deposition, PM resuspension and canopy evaporation. A multi-layered approach is considered in which calculations are done per layer. Application of this model to the Scots pine stand for $PM_{2.5}$ (PM with a diameter < 2.5 µm) resulted in a throughfall calculation of 697 mm compared to 700 mm measured and $PM_{2.5}$ deposition of 31.43 kg $PM_{2.5}$ ha^{-1} yr^{-1} of which 24% was removed and 76% resuspended. These numbers are considered realistic though the share of resuspension is somewhat high compared to values mentioned in literature (Hirabayashi et al., 2012; Nowak et al., 2013; Zinke et al., 1967). The integration into ANAFORE, allowed for a calculation over time and assessment of the influence of indirect effects on PM removal, such as wood harvest and climate change. Here, also the indirect effect of change in $PM_{2.5}$ airborne concentration, induced by different emission legislations, was assessed for the period 2010-2030, while the forest grows. This resulted in an estimated avoided health costs due to $PM_{2.5}$ removal within a range of 915-1075 euro ha^{-1} yr^{-1} during the considered period.

6.1.3 A framework to unravel best management practices based on (dis)services provided and impacts/benefits of the forest (chapter 4)

Finally, we performed an environmental impact assessment, using the framework of chapter 2, and monetary ecosystem service assessment of the Scots pine stand under different management and climate scenarios from the year 2010 up until 2089, using the ANAFORE model including the developed PM removal submodel (chapter 3).

For the monetary valuation of ecosystem services, specific monetary values valid for Flanders were used, e.g. 150 euro kg^{-1} $PM_{2.5}$ removed (Broekx et al., 2013; Liekens et al., 2013b). These values are valid for the selected reference year 2010, which is an important limitation. The environmental impact assessment methodology ReCiPe (Goedkoop et al., 2009) was applied using our previous framework. In this framework the uptake of harmful compounds such as CO_2 is considered (Schaubroeck et al., 2013), chapter 2, thus the benefit and the damage done by the Scots pine stand to mankind and nature was assessed. In the ecosystem service assessment we have considered disservices (e.g. NO_x emission) and hence accounted not only for the beneficial effect of a forest ecosystem but also its detrimental effect on mankind through disservices. A negative monetary value is attributed to a disservice. We have by consequence attempted to consider in both approaches the bi-directional relationship between mankind and nature in a better manner. The addressed flows/ecosystem services in this analysis are: PM removal ($PM_{2.5}$ and $PM_{2.5\text{-}10}$), freshwater loss, CO_2 sequestration, wood production, NO_x emission, NH_3 uptake and freshwater (nitrogen) pollution/removal. Note that is just a limited number of services/flow, e.g. freshwater loss due to evapotranspiration is considered a disservice while we did not consider the benifical effect of evapotranspiration on counteracting global warming through surface cooling (Bonan, 2008).

The management and environmental change scenarios represent the possible (indirect) influence we have on the forest. The model results of these scenarios therefore stand for the potential (indirect) effects which might occur through our actions on the forest, e.g. less wood growth by the forest induced by too much harvest. In practice, three management and three environmental change scenarios were applied, resulting in nine overall scenarios.

The environmental change scenarios consists of a current one, and two future environmental change scenarios. These future scenarios are based on certain socio-economic incentives, this to reflect the effects of socio-economic choices on the results. The Moderate scenario is associated with a more sustainability-oriented policy and the Severe scenario represents a business as usual approach. The environmental change scenarios include changes in temperature, precipitation, nitrogen deposition, wind speed, $PM_{2.5}$ and $PM_{2.5\text{-}10}$ concentration, and CO_2 concentration.

Regarding management, the Scots pine stand was modelled as a forest plantation with 10 000 trees ha^{-1} at the beginning of the 80 years long rotation period. At year 14 a conventional precommercial thinning of 50% occurred. Three management scenarios were applied differing in the subsequent five-yearly thinning quantities, relative to the amount of wood grown over that period: 0% (Low), 50% (Mid), 100% (High). At the end a clear-cut was always performed.

Following main results were obtained. The monetary valuation highlights the importance of services provided by the forest, with a total yearly average of 361 (scenarios High and Moderate) -1242 (scenarios Low and Current) euro ha^{-1} yr^{-1}. $PM_{2.5}$ removal is the key service with a value of 622 (scenarios High and Moderate) -1172 (scenarios Low and Current) euro ha^{-1} yr^{-1} (Figure 4.8, pg. 122). Next to that, these advantages are less pronounced for the Severe and even more for the Moderate scenarios. Mainly since for these scenarios the airborne $PM_{2.5}$ concentrations decrease over time, and there is thus less $PM_{2.5}$ removal possible. Straightforward, the lower pollution of mainly $PM_{2.5}$ through more stringent legislation, as is the case for the Moderate scenario, the less there can be pollution removal. Care must be taken in interpreting and using these results as monetary values cannot truly represent intrinsic values of services to mankind and always have a subjective aspect. Concerning environmental impact assessment, with CO_2 sequestration and thus the prevention of its damage as the most relevant contributor, a yearly average prevention in disability adjusted life years of 0.014 (scenarios High and Moderate) to 0.029 ha^{-1} yr^{-1} (scenarios Low and Current) is calculated (Figure 4.8, pg. 122). There is however a yearly average biodiversity loss of -1.09E-06 (scenarios Low and Current) to 7.3E-05 species*yr ha^{-1} yr^{-1} (scenarios High and Severe), mostly through the intensive land use but counteracted by CO_2 sequestration with 46-101% (Figure 4.8, pg. 122). The biodiversity loss through nitrogen water eutrophication is even not accounted for as the ReCiPe method has no means yet to quantify this, though differences between scenarios are minor on a midpoint level. The differences between environmental change scenario outcomes is negligible for the environmental impact assessment. On the other hand, the discrepancy between the results of the three management scenarios are superior in both assessment methods. Both approaches favor the use of the least intensive management scenario, the ‘low’ scenario, since CO_2 sequestration and PM removal are higher for these, latter induced by a higher needle area per surface area. Our framework has thus resulted in the clear selection of the best management scenario of the considered ones, of course only in light of the considered aspects.

6.1.4 Improvements to ecological network analysis prior to inclusion in environmental sustainability assessment (chapter 5)

Ecological Network Analysis (ENA) is a methodology to study and characterize flux networks over a certain period of time among defined ecosystem compartments via indicators, e.g. cycling of nitrogen between different trophic levels of a forest ecosystem over a year assessed with the Finn's cycling index (FCI) (Finn, 1980). This tool was investigated and improved in light of its twofold potential relevance for environmental sustainability assessment. First, there is a possibility of better impact assessment on ecosystem quality, an area of protection, besides diversity loss via a change in (an) ENA-indicator(s). Secondly, as the mathematical backbone of ENA is derived from linear inverse modelling, which is also used in environmental sustainability assessments such as life cycle assessment, ecosystems studied using ENA can be easily integrated into them as needed for the framework of chapter 2. Improvements have been made to some aspects of ENA. A difficulty in ENA was its application to non-steady state systems. This was resolved by reintroducing the concept of Finn (1977, 1976) on this matter in which per compartment an abstract external stock compartment is considered with flows to and from it as increment and depletion, respectively. The major adaptation was to enable physical compartmentalisation of the surrounding environment of the studied (eco)system. This offers possibilities for specification of destinations and sources of export and import flows, respectively, which is desired in LCA to assess the impact of these flows. Application to the C, N and H_2O flux networks of the Scots pine stand, resulted into FCI values of 0, 0.40 and 0.00010, respectively. Prior to application in environmental sustainability assessment the following matter should be addressed. There are no standards yet for the different choices in the ENA methodology, which can have an influence on the indicator values. Hence, defining such standards is a needed important research step.

6.2 Perspectives

At the end of each chapter, different future perspectives are given in the context of the chapter. Besides these, here are more general ones discussed.

6.2.1 Further integrations

The presented and improved tools may be even further integrated in the future, this to increase even more their added values. We will shortly discuss some pipelines for further integrations.

As pointed out in chapter 5, a change in an indicator of ecological network analysis, an ENA-indicator, may serve as an indication for damage done to ecosystems, an area of protection. The evolution of ENA-indicators while the ecosystem grows over time under different management scenarios should be quantified to assess the discrepancy in values between management options and natural development. This could for example be done by using the ANAFORE model for forests, applied in chapter 2 and 3, to compute for distinct time periods, e.g. every year, ENA-indicator values. Practically, a flow network needs to be set up and indicator values calculated based on them for every time step. The challenges for further research in this field of study are that there are no standards yet for ENA-calculations, see section 5.5.3, p. 157, and a change in ecosystem services is more tangible and can be linked to damage done to mankind. Otherwise, one could still use these tools to study the ecosystems and maybe these indicators may serve as midpoint indicators for a change in ecosystem services.

Another important aspect of network analysis and its indicators, is the assessment of resilience of a human/industrial, natural or coupled system in response to perturbations and stress (Haberl et al., 2004; Pizzol et al., 2013; Singh, 2012; Xu et al., 2011). Resilience, the capability to retain similar structures and functioning after disturbances for continuous development, is an often overlooked sustainability aspect (of a system) (Liu et al., 2007). Network analysis seems one of the best ways to assess resilience as it may quantify (in an indirect manner) the amount and quality of various pathways a system possesses to overcome a disruption of one of them, an indication of resilience.

In environmental sustainability assessment one often assesses the impact of a static system on the environment, this is an attributional point of view. However, systems are dynamic and react to policy changes and one could hence also include the change in environmental impact of these alterations. For example, increased production of bio-ethanol out of sugar cane in Brazil, indirectly could lead to an additional loss of natural ecosystems, as more land is needed to cultivate biomass in Brazil, unless other practices, such as husbandry, are intensified (Alvarenga et al., 2013). In life cycle assessment (LCA), the respective ways of assessment are called the attributional and consequential LCA (JRC-IES, 2010). In particular a consequential LCA is the convergence between LCA and economic modelling approaches (Earles and Halog, 2011). However, not only the response of the human/industrial system can be modelled but also that of ecosystems. This is already done in a simple manner in impact assessment methodologies, in which the indirect effects of emissions are also considered, e.g. secondary formation of particulate matter out of nitrogen oxides (Goedkoop et al., 2009). Integration of complex ecosystem models with life cycle assessment should be further researched to better assess this dynamic response. The environmental impact assessment of a forest in chapter 4 serves as a good example.

In that same chapter, only an impact assessment was performed of the forest itself not of the related human/industrial processes, such as wood harvest. One could get a better image through accounting for the environmental impact of these activities. This could be done in a similar fashion as in chapter 2 though latter chapter already pointed out that the growing of the wood in the forest is probably the most important process and not human/industrial ones for forestry practices.

The integrated framework of chapter 4 can be even further improved through accounting for more biophysical output values of the model in the quantification of the impact or monetary value of an ecosystem service. A good illustration of this is the inclusion of stem thickness in the price for wood. This could even be further improved by including the quality of the wood, e.g. number of knots induced by too much branching, in the wood price calculation.

6.2.2 Application; extrapolation to Flanders

Different new or improved tools/frameworks have been presented and applied in this dissertation. However to apply these, a lot of input data are needed. In fact, we applied these to one well-studied forest for which this was possible as plenty of necessary data on its characteristics or fluxes were available. Hence thorough in-depth analyses of this single case were possible. These serve as fine examples to what results and findings the tools/frameworks may lead. Above that, the respective outcomes already show the possible significant impact that (forest) ecosystems may have on the environment, including mankind.

One can extrapolate/generalize the results to illustrate this. For example, Flanders has about 35000 ha of Scots pine forests (INBO, 2007) and an average $PM_{2.5}$ concentration of 17-24 µg m^{-3} in 2010 (Vlaamse milieumaatschappij, 2011). This concentration resembles that at our studied Scots pine stand in 2010: 18 µg m^{-3}, see chapter 3. Assuming that this concentration is also equal for all Scots pine stands and that their composition is similar to the one of our studied Scots pine stand, this would mean that in 2010 39 million euro yr^{-1} of health costs were potentially saved in 2010 through particulate matter removal of Scots pine forests in Flanders. Taking into account uncertainty in the result, it is though more fitting to express this in a magnitude of 3.9-390 million euro yr^{-1}. If we consider the Scots pine stands in Flanders to be managed in the same manners as the one in chapter 4 and this for the different climate scenarios, the value of the considered ecosystem services for the period 2010-2089 results in 13 to 43 million euro yr^{-1} for Flanders. Also here a magnitude 10 times lower and higher should be considered.

The readily application of the developed tools/frameworks to other ecosystems will be however low as for few ecosystems data quality and quantity will be adequate enough. On the other hand, this study highlights which data need to be collected and are of importance for the application of certain tools and thus to obtain respective findings. For example, to quantify the environmental benefit or ecosystem services of a forest, it seems crucial to collect data on its removal of particulate matter (chapter 2).

6.2.3 Future methodological challenges

Different methodological choices can be made in the application procedure of the presented and improved tools. There are no best options yet for a lot of these choices and their influence can be significant. For example, for ecological network analysis this is discussed in section 5.5.3, p. 157. One needs to keep this aspect in mind when using the respective tools.

Concerning maturity, the research field of environmental sustainability assessment is in fact still in its infancy, mainly since it is a challenging one as it combines various other research domains. In essence, a complete environmental sustainability assessment will only be achieved if we can accurately predict the future of the world, which is nowadays impossible, since damage/benefit is spread over time and sustainability includes the well-being of future generations. For now, practically, a lot of methodological improvements are needed to better perform environmental sustainability assessment, including more accurate future predictions. Every step in such an assessment has in fact **a lot of methodological difficulties to be resolved**. This is discussed below, and in particular for life cycle assessment (LCA), see section 1.2 (pg. 4). Overall, the audience should be aware of these shortcomings when consulting results of an environmental sustainability assessment. These limitations are nicely illustrated for bioenergy systems by Holma et al. (2013) and Cherubini and Strømann (2011).

As mentioned in chapter 2, the **boundaries** of the system (life cycle for LCA) for which the environmental impact/benefit should be assessed is not standardized yet, only the fact that this should occur in the first phase of such an assessment, though in that chapter the possibility and relevance of including ecosystems is shown. Next to the boundaries in space, the ones in time are as important and have to be chosen. For example through the selection of an impact assessment method, one sets the boundary in time for the impact assessment, e.g. global warming potential over 20 years or over 100 years. Also, in the inventory step, boundaries are drawn. In chapter 2 the further fates of the stored carbon after the considered period 2001-2002 were not assessed. It is possible that this carbon is rereleased again after some years. If this is accounted for the results of the study may alter considerably.

The construction of the **inventory** of in- and outgoing flows/relationships of a studied system, resources and emissions of the life cycle inventory (LCI) of LCA, has also some important challenges. Most importantly, models applied to obtain an LCI are only simple linear inverse models (Suh and Huppes, 2005), as brought forward in Chapter 2 (pg. 17). A solution is the application and integration of more complex non-linear models, as done in this dissertation using ANAFORE, a forest growth model.

A here not addressed major issue is that of **allocation** of environmental impact/benefit of a system between different coproducts of that system, e.g. allocation based on physical or economic value. Results might vary a lot depending on how this matter is addressed (Svanes et al., 2011). On top of that, the 'emergy' approach, explained in chapter 2, uses a completely different approach in allocation compared to LCA, resulting in major discrepancies between results. This is well explained by Rugani and Benetto (2012). A scientific consensus on this matter is yet far from realized.

The last step, the **impact assessment**, calculation of benefit/damage from the inventory flows of a system, can be done in numerous ways following various methodologies (Moldan et al., 2012). This reflects the lack of a scientific agreement on this topic. Every method has its limitations and flaws but undoubtedly also its strengths and advantages. We will illustrate this with a skeptical analysis of the CEENE methodology, in which resource consumption is expressed as cumulative exergy extracted from the natural environment (CEENE) (Dewulf et al., 2007), applied in chapter 2 and 5. The reason for its selection, is that the method expresses resources in one scientifically sound metric, covering all resource types, whereas others do not (Swart et al., submitted), and that other methods which assess the more final impact, through depletion, are not considered adequate enough due to scientific gaps (Hauschild et al., 2013). On the other hand, this CEENE-indicator does thus not assess resource depletion or scarcity. A thermodynamic concept, such as exergy, has its limits in expressing environmental sustainability since for mankind and natural entities the values of goods and resources is much more restricted than just the first and second thermodynamic laws, e.g. mankind does not need a definite amount of exergy for its well-being, but a composition of certain goods (food, oxygen,...). Related to that, aggregation of data in one metric also leads too data loss and thus loss in the possibility to assess environmental sustainability of systems. For that reason, Zhang et al. (2010a), advise to still represent the raw inventory data in an environmental sustainability assessment. On top of that, the only functionality, and thus value, covered by exergy of a good is its usage as an energy source (Romero and Linares, 2014). To the contrary, goods are used for more than just energy purposes, e.g. some metals are used in the first place as semiconductors in the electronic industry not as energy sources.

Hence, we regard this CEENE method as a good ad interim solution and further research is needed to obtain a method which does cover final damage related to resource depletion, possibly integrating the good aspects of methods such as CEENE.

Another difficulty in this topic is the **aggregation of damage done to different areas of protection** (human health, natural resources and natural system, Figure 1.2, pg. 4) into one indicator. There already exist approaches of environmental sustainability assessment which do so, as is the case for Eco-indicator 99 method (Goedkoop and Spriensma, 2001) and ReCiPe methodology (Goedkoop et al., 2009), introduced by Sleeswijk et al. (2008). In essence, in these approaches, the resulting endpoint indicator values, the different damage values to the areas of protection, of a part of the human/industrial system are divided by those of another (part of) the human/industrial system and summed up (optionally after weighting). This gives in the strict sense only a comparison between both systems. As an example we normalized the environmental impact of 1 m^3 sawn timber on human health (1.40E-02 DALY) and ecosystem quality (1.60E-04 species*yr lost), described in chapter 2, in a manner done according to ReCiPe methodology. More precisely, we compared our system with the environmental impact of Europe. The latest estimated emission values for Europe as a whole in the year 2000 are 2.02E-02 DALY $capita^{-1}$ yr^{-1} and 1.81E-04 species*yr $capita^{-1}$ yr^{-1}, for human health and ecosystem quality respectively, this according to the Hierarchist approach (“Downloads - ReCiPe,” 2014). Dividing the values of the case study by these European level values and summing them up, results in the normalized value of 1.57 capita*yr. In practice, some additional weighting is then applied and the unit of the resulting value is named as ‘points’, while it should just be ‘capita*yr’. In general, this resulting value only expresses how the impact of 1 m^3 sawn timber compares itself to that of an average European citizen in the year 2000.

As a matter of fact, this type of approach does not aggregate for example damage done to mankind and nature into a single one by expressing damage done to nature in terms of mankind or vice versa. When applying such a methodology, the user should be well aware of this. Dahlbo et al. (2013) already discuss that this confuses the audience. We go a step further and even discourage the usage of this approach of normalizing to another part of the human/industrial system, as the audience is easily misguided by it and the added value of it is slim. For these reasons, this normalization possibility for ReCiPe was also not used in chapters 2 and 5.

A better solution is needed in which for example damage to mankind or nature are weighted and/or expressed in one unit. This is addressed in the next section.

6.2.4 A (need for a) revised idealistic/ethical backbone for sustainability assessment and an associated methodological framework

Next to methodological improvements, we should revise our goal for environmental sustainability assessment of which the main question is: "Which has priority to be saved: mankind or nature?". It is appropriate to first answer this ethical question before furthering research in this field. Otherwise, it will not be possible to develop one single methodology with one outcome for each case and different methodologies (life cycle analysis, ecosystem services assessment, emergy calculations,...) with different viewpoints, leading to possibly different outcomes, will continue to be used.

Bourdeau (2004) reviews different environmental ethical visions/codes, with other priorities, on the ethics of mankind and nature: from absolute anthropocentrism (moral rights for mankind) to biocentrism (moral rights for all living entities) and the most radical ecocentrism (moral rights for all living and non-living entities). When in fact considering the word 'sustainability' one needs to associate to it a certain entity which needs to be maintained/sustained/beheld. This is another dimension/aspect of this term which is relevant to address. Likewise to the different levels introduced by Bourdeau (2004), different types of sustainability may exist regarding what to priorily sustain, of which from a human point of view the most interesting are given in Figure 6.1. The ideology of 'egocentrism' represents the possible interest in only priorly sustaining oneself. Note that a spectrum exists across the presented ones in Figure 1. For example, when only considering the priorily sustaining of humans of a certain region (e.g. Europe), this approach is somewhere between egocentric and anthropocentric. Which matters to protect/sustain besides the priorily addressed entities, is not fixed by this dimension. An anthropocentric sustainability approach may exist which takes into account impact on animals but this aspect will be less important than impact on mankind. In anthropocentrism the goal one wants to reach by protection of the environment is to sustain humans as nature provides vital services to us, though mankind's survival has priority over nature's maintenance.

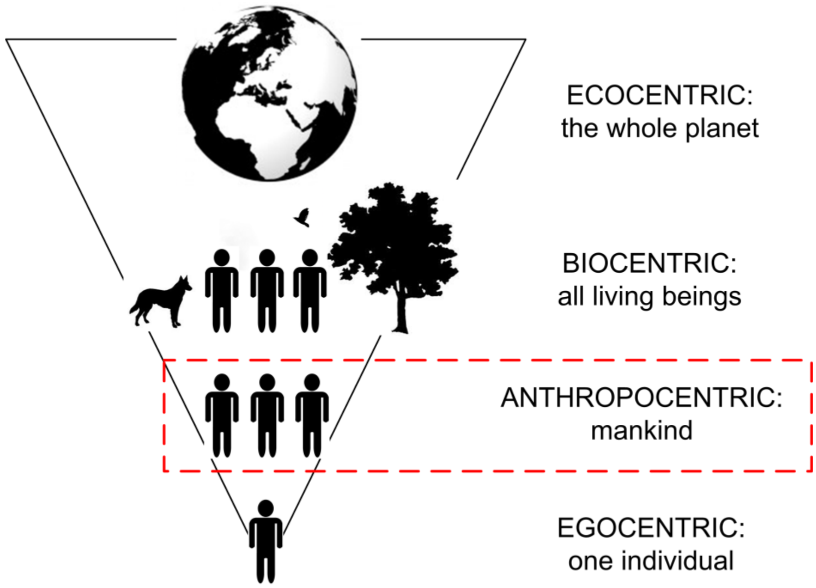

Figure 6.1. Different things can be chosen to have priority to be protected/maintained/sustained. (Indirectly this means that entities of the same level have an identical intrinsic value.) This leads to a spectrum of sustainability ideologies/ethics and associated assessment methodologies. The most common things we want to protect and the corresponding type of sustainability assessment are given. except the egocentric one. The original definition of sustainability/sustainable development corresponds with the anthropocentric one (red dotted line) (WCED, 1987).

In the strict sense, the original definition of sustainable development, sustainability, by Brundtland (WCED, 1987),"the development that meets the needs of the present without compromising the ability of future generations to meet their own", corresponds to an anthropocentric point of view, thus priorly maintaining human beings, irrespective of their differences (Moldan et al., 2012). Article 1 of the Rio Declaration confirms this (UNCED, 1992): "Human beings are at the centre of concerns for sustainable development. They are entitled to a healthy and productive life in harmony with nature.".

On the other hand, the Brundtland definition does not address only the survival/maintenance/protection of mankind but all needs of humankind. The term 'human needs' in the definition of sustainable development by Brundtland (WCED, 1987) is too unclear and needs further elaboration. The famous pyramid of Maslow (1943) gives an overview and hierarchy of human needs (Figure 6.2). A main distinction can be made between primary human needs (at the bottom) which lead to human survival, maintenance of human health, and the other human needs, which we may call here secondary, leading to prosperity.

Figure 6.2. The human needs presented as the pyramid of Maslow (1943), starting from primary (for survival) at the bottom.

It is clear that according to the original definition of sustainability, primary human needs, here considered as the ones leading to maintenance of human lives, have priority. Above that, secondary human needs can only be fulfilled if one is alive. In fact this approach is based on the assumption that one may split human needs in a fundamental part for a healthy life and another part representing secondary needs, addressed by welfare or prosperity. Bourdeau (2004) suggests an ethical code by which these primary human needs are therefore met first and foremost whereas the need for other living organisms and ecosystems are allowed to prevail over secondary human needs. From another perspective, the desire to protect these other living organisms and ecosystems, is in fact a possible secondary human need. By consequence, in line with latter reasoning, an overlapping sustainability framework needs thus to be developed in which human primary and secondary needs are quantitified and in which the primary have priority over the secondary.

To the contrary, some sustainability assessment methodologies address ecocentric sustainability, dealing with the maintenance of the planet as a whole, e.g. the framework of Muys (2013), thermodynamic-based methodologies such as the CEENE methodology (Dewulf et al., 2007) and the emergy framework (Brown and Ulgiati, 2010), and LCA in general when covering other endpoint areas of protection besides human well-being (Figure 1.2, pg. 4).

Coming back to our framework, we will here represent a preliminary conceptual version of it. Firstly an indicator is needed which expresses gain/loss for human health, the primary human needs, such as the Disability Adjusted Life Years (DALY) indicator (Arnesen and Nord, 1999). This implies only one area of protection namely human health and discards the others (Figure 1.2, pg. 4). However, these others act as midpoints and should be expressed in loss of human health. This is a major scientific challenge. Concerning damage to ecosystems, this might lead to a loss in ecosystem services, which then optionally leads to damage of human health, e.g. toxic compounds in the sea, kill fishes, then reduces fish catch and finally lead to loss in human survival due to lack of food. In fact, loss in ecosystem services is recently proposed as another indicator to represent damage to the natural environment besides loss in species diversity (Koellner and Geyer, 2013). This has already been worked out for land use impact on some services, in addition to biotic production (Saad et al., 2013). On top of that, a framework has been presented in which even the impact of change in ecosystem services via economic system on mankind is assessed (Cordier et al., n.d.). To express the effect of natural resource consumption in human health loss, the methodology of Pfister et al. (2011, 2009) in which water consumption is expressed in DALY should be considered as a good example to do so. The concept of our proposed new method is shown in Figure 6.3.

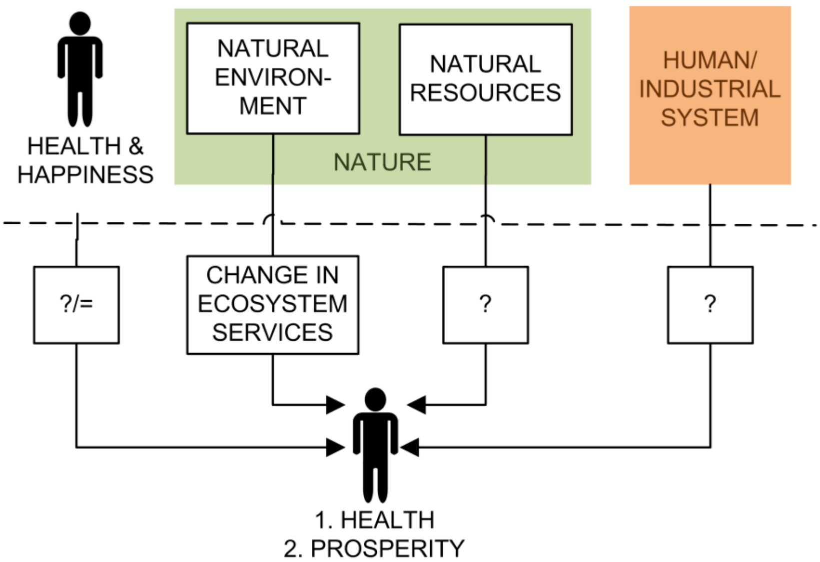

Figure 6.3. The concept for the newly proposed approach wherein only human well-being is a final area of protection, in accordance with the original definition of sustainability/sustainable development (WCED, 1987). Herein, the aspect of human health has priority over prosperity. Only damage or benefit to these should be considered. The areas of protection (de Haes et al., 1999) serve as midpoint indicators and are shown above. These should be expressed in human well-being via methodological approaches. For natural environment this could possibly be done via the ecosystem services approach. Economic sustainability is in fact already included in the response of the human/industrial system. Social sustainability is accounted for via human happiness.

The already existing methodologies of environmental sustainability can be addressed to develop this overlapping methodology. As such, the framework introduced in chapter 2 is already a small step towards achieving this development since it already combines the environmental benefit of uptake of harmful compounds (included in ecosystem service assessment) with the environmental impact assessment of human/industrial systems (included in LCA), this for an integrated human/industrial-natural system.

For prosperity, a respective indicator needs to be developed or selected. Overall, this could not be only regarded from an environmental point of view, but also from a social and an economic one. In fact, the economic aspect is already partially integrated through the response of the human/industrial system. For example, an economic loss leads to less possibilities in maintaining human health. For social sustainability, one should address also human happiness at a midpoint level. This is important as a depressed person (low happiness) might work less, creating less goods for mankind and even commit suicide (loss of human health). Well-being of animals is then also accounted for since a share of mankind can become depressed/unhappy if this is not attended to. Human happiness is after all in a certain degree needed for human survival, it can be partially regarded as a primary need. How to integrate this aspect, should be further researched in the field of social sustainability assessment. Next to that, higher economic profits may lead to more happiness and prosperity. To conclude, this total new concept has two indicators representing benefit/damage to human-health and provisioning of secondary human needs, addressed here as 'prosperity', of which the first has priority over the second. The economic aspect is indirectly included in both previous aspects. This new framework is in complete accordance with the original anthropocentric sustainability/sustainable development definition. Note that this framework is still immature and in need of considerable more refinement.

We will shortly compare our framework with the already existing frameworks for sustainability assessment, see Table 6.1. The Prosuite-framework introduces an alternative five-pillar concept compared to the traditional three-pillar concept consisting of an economic, social and environmental pillar. Latter pillar was further split up into four areas of protection by de Haes et al. (1999). The five-pillar concept does not appear to be that innovative as it is quite similar to the three-pillar concept in combination with the areas of protection approach. The area of protection human/industrial system is however not considered but it is often neglected when addressing the areas of protection. The Prosuite-framework does however give a good qualitative and quantitative description of its pillars. Next to that, there is no complete anthropocentric focus in any of the approaches as in ours, which is the essence of this framework. The Prosuite framework can however be considered to approach the economic pillar from a more anthropocentric point of view by expressing it as prosperity in gross domestic product, though on the other hand this is still expressed in money. Besides human health, the endpoints/pillars of the other approaches serve as midpoints, impact at midway, for the final impact on human health and prosperity in our approach.

Table 6.1. Different endpoints/pillars of existing sustainability assessment frameworks and the one we introduce. We advise to use this approach after being completely worked out.

Three-pillar (Elkington, 1999)	**Areas of protection (de Haes et al., 1999)**	**Five-pillar (Gaasbeek and Meijer, 2013)**	**Our presented approach**
Environmental	Human health	Human health	Human health & prosperity
	Natural resources	Exhaustible resources	
	Natural environment/ Ecosystem (quality)	Natural environment	
	Human/Industrial system	/	
Social		Social well-being	
Economic		Prosperity	

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Curriculum Vitae

1. General information

Personal details

Full name	Thomas Schaubroeck
Date of birth	24/01/1986
Gender	Male
Nationality	Belgian
Affiliation	Ghent University, Faculty of Bioscience Engineering, Environmental Organic Chemistry and Technology (EnVOC; http://www.ugent.be/bw/doct/en/research/envoc)
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E-mail	thomas.schaubroeck@ugent.be

Professional position

2010-present Ghent University, EnVOC: **PhD candidate**
Project title: 'Towards a thermodynamics-based integrative human/industrial – eco-system analysis for sustainable resource management'
Promotors: Prof. Jo Dewulf, Prof. Kris Verheyen and Prof. Bart Muys

Education

2004-2009 Ghent University, Faculty of Bioscience engineering: **Master in Bioscience Engineering, option Chemistry and Bioprocess Technology (MSc),** distinction
Thesis: 'Functional equilibrium in OLAND aggregates for sustainable nitrogen removal'; Promotor: Prof. Willy Verstraete; tutor: Prof. Siegfried Vlaeminck

1998-2004 Instituut O.L.V. van Vreugde, Roeselare (Belgium): Science-mathematics (8 hr)

Expertise

- Environmental sustainability, more specifically life cycle assessment (LCA) and ecosystem service assessment, their application and methodology
- Resource usage and efficiency of production systems using exergy analysis and Exergetic LCA (ELCA)
- LCA on wastewater treatment plants, forestry and aquaculture

- ecosystem service assessment of forest and the influence of forest operations
- Ecological study of flux network in systems, with a focus on forest ecosystems, via Ecological Network Analysis (ENA)
- Ecological modelling
- Particulate matter removal, deposition and washoff by plant vegetation, specifically the modelling of these processes
- Microbial technology and ecology of a novel nitrogen removal treatment process (nitritation with anammox)

2. Research activities

Publications

Overview: https://biblio.ugent.be/person/802000753404

<u>*Peer-reviewed articles (A1)*</u>
IF: Impact Factor of year closest to publication year

ACCEPTED

1. Taelman, S.E., De Meester, S., **Schaubroeck, T.**, Sakshaug, E., Alvarenga, R. A. F., Dewulf, J., 2014. Accounting for the occupation of the marine environment as a natural resource in life cycle assessment: an exergy based approach. *Resourc. Conserv. Recycl.* 91, 1-10. (IF: 2.3)
2. Huysman, S., **Schaubroeck, T.**, Dewulf, J., 2014. Quantification of spatially differentiated resource footprints for products and services through a macro-economic and thermodynamic approach. *Environ. Sci. Technol.* doi: 10.1021/es500777k (IF: 5.5)
3. **Schaubroeck, T.**, Deckmyn, G., Neirynck, J., Staelens, J., Adriaenssens, S., Dewulf, J., Muys, B., Verheyen, K., Accepted. Multilayered modeling of particulate matter removal by a growing forest over time, from plant surface deposition to washoff via rainfall. *Environ. Sci. Technol.* doi: 10.1021/es5019724 (IF: 5.5)
4. Huysveld, S., **Schaubroeck, T.**, De Meester, S., Sorgeloos, P., Van Langenhove, H., Van linden V., Dewulf, J., 2013. Resource use analysis of pangasius aquaculture in the mekong delta in Vietnam using exergetic life cycle assessment. *Journal of Cleaner Production* 51, 225–233. (IF: 3.40)
5. **Schaubroeck, T.**, Alvarenga, R. A. F., Verheyen, K., Muys, B., Dewulf, J., 2013. Quantifying the Environmental Impact of an Integrated Human/Industrial-Natural System Using Life Cycle Assessment; A Case study on a Forest and Wood Processing Chain. *Environ. Sci. Technol.* 47, 13578-13586. (IF: 5.5)
6. **Schaubroeck, T.**, Bagchi, S., De Clippeleir, H., Carballa, M., Verstraete, W., Vlaeminck, S.E., 2012. Successful hydraulic strategies to start up OLAND sequencing batch reactors at lab scale. *Microbial Biotechnology* 5, 403–414. (IF: 3.2)
7. **Schaubroeck, T.**, Staelens, J., Verheyen, K., Muys, B., Dewulf, J., 2012. Improved ecological network analysis for environmental sustainability assessment; a case study on a forest ecosystem. *Ecological Modelling* 247, 144–156. (IF: 2.1)
8. Vlaeminck, S.E., Terada, A., Smets, B.F., De Clippeleir, H., **Schaubroeck, T.**, Bolca, S., Demeestere, L., Mast, J., Boon, N., Carballa, M., Verstraete, W., 2010. Aggregate size and architecture determine microbial activity balance for one-stage partial nitritation and anammox. *Applied and Environmental Microbiology* 76, 900–909. (IF: 3.8)

UNDER REVIEW OR REVISION

1. Trang, N.T., **Schaubroeck, T.**, De Meester, S., Duyvejonck, M., Sorgeloos, P., Dewulf, J., 2013. Resource consumption assessment of Pangasius fillet products from Vietnamese aquaculture to European retailers. *Journal of Cleaner Production* (IF: 3.4)

National articles with reading committee (A3)

1. **Schaubroeck, T.**, Bagchi, S., De Clippeleir, H., Carballa, M., Verstraete, W., Vlaeminck, S.E., 2012. Improved start-up of OLAND sequencing batch reactors by means of hydraulic strategies. Communications in Agricultural And Applied Biological Sciences 77.

Complete papers in conference proceedings (C1)

1. Vlaeminck, S.E., Terada, A., Smets, B.F., De Clippeleir, H., **Schaubroeck, T.**, Carballa, M., Verstraete W. 2010. Small aggregates can cause nitrite accumulation in one-stage partial nitritation and anammox. IWA World Water Congress and Exhibition. Montréal, 19-24 September 2010.
2. De Clippeleir, H., Weissenbacher, N., **Schaubroeck, T.**, Boeckx, P., Boon, N., Wett, B.. Mainstream partial nitritation/anammox: Balancing overall sustainability with energy savings. WEFTEC.12. New Orleans, September 29-October 3 2012.

Presentations on conferences

Oral presentations

1. **Schaubroeck, T.**, Staelens, J., Verheyen, K., Muys, B. & Dewulf, J. (2012). Analyse van massafluxen in een bosecosysteem; casestudie van een grove dennenbestand in de Kempen. Symposium voor Starters in het natuur- en bosonderzoek, Brussel, 16 maart 2012.
2. **Schaubroeck, T.**, Staelens, J., Verheyen, K., Muys, B., Dewulf, J., (2013). Improved Ecological Network Analysis for Environmental Sustainability Assessment; a Case Study on a Forest Ecosystem. Presentation abstract: Ecological Modelling for Ecosystem Sustainability in the context of Global Change, 19th Biennial ISEM conference, Abstracts, Toulouse, 28-31/10/2013.
3. **Schaubroeck, T.**, Alvarenga, R.A.F., Verheyen, K., Muys, B., Dewulf, J., (2013). Environmental sustainability of integrated human/industrial-natural systems. Presentation abstract: Ecological Modelling for Ecosystem Sustainability in the context of Global Change, 19th Biennial ISEM conference, Abstracts, Toulouse, 28-31/10/2013.

Poster presentations

1. **Schaubroeck, T.**, Bagchi, S., De Clippeleir, H., Carballa, M., Verstraete, W. & Vlaeminck S.E. Improved start-up of OLAND sequencing batch reactors by means of hydraulic strategies. 17th Symposium on Applied Biological Sciences. Leuven, 10 February 2012.
2. **Schaubroeck, T.**, Staelens, J., Verheyen, K., Muys, B., Dewulf, J., 2012. Accounting for ecosystem functioning using an improved ecological network analysis methodology; a case study on a forest ecosystem. 17th Symposium on Applied Biological Sciences. Leuven, 10 February 2012.
3. **Schaubroeck, T.**, Bagchi, S., De Clippeleir, H., Carballa, M., Verstraete, W. & Vlaeminck S.E. Successful hydraulic strategies to start up OLAND sequencing batch reactors at lab scale. First international symposium on Microbial resource management in biotechnology: Concepts & applications'. Ghent, 30 June – 1 July 2011. Awarded with excellent poster prize

Conference organisation

Member of the organizing committee of the '16th Symposium on Applied Biological Sciences'.

3. Teaching and tutoring activities

Teaching at UGent, FBE

2010
Chemical analytical methods – partim Organic, **practicum** Column Chromatography (**20 hrs**)
Analysis of organic micro-pollutants, **tasks** (**10 hrs**)
Process engineering 2, **exercises** (**12 hrs**)
Environmental organic chemistry, **practicum** Olfactometry (**15 hrs**)
2011
Chemical analytical methods – partim Organic, **practicum** Column Chromatography (**20 hrs**)
Analysis of organic micro-pollutants, **tasks** (**10 hrs**)
Process engineering 2, **exercises** (**12 hrs**)
Environmental organic chemistry, **practicum** Olfactometry (**15 hrs**)
2012
Chemical analytical methods – partim Organic, **practicum** Column Chromatography (**20 hrs**)
Process engineering 2, **exercises** (**12 hrs**)
Environmental organic chemistry, **practicum** Olfactometry (**15 hrs**)
2013
Process engineering 2, **exercises** (**12 hrs**)
Environmental organic chemistry, **practicum** Olfactometry (**15 hrs**)

Supervision of master theses

MSc theses *supervised as* ***tutor***
1. Sophie Huysveld (2010-2011, Bio-engineer option Environmental Technology):
Life Cycle Assessment of Pangasius production in the Mekong Delta: a cradle to farm gate study
2. Lluisa Garcia Marti (2011-2013, Bio-engineer option Environmental Technology):
Life Cycle Assessment of Shrimp Aquaculture in Nicaragua
3. Leen Sturtewagen (2013-present, Bio-engineer option Nutrition):
Resource use profile and nutritional value assessment of a canteen meal; a case study on pork vs. quorn
4. Pieterjan Serruys (2013-present, Bio-engineer option Chemistry and Bioprocestechnology):
Resource use analysis of an integrated farming system in Vietnam, an exergetic life cycle assessment

Dankwoord

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Veel ontspanning vond ik in mijn muzikale hobbies. De eerste jaren van mijn doctoraat heb ik gemusiceerd bij Siegfried en zijn Brugse Cantorij Jubilate. Als Benjamin werd ik er goed ontvangen en heb ik er mooie momenten beleefd. Ook bedankt aan gezin Vlaeminck waar ik mijn buik mocht vullen met lekkers. *Muchas Gracias* aan de band UltravioOlet (nee, dit is geen spellingsfout) waar ik als percussionist voor de *beat* mag zorgen. Ik kan me uitleven bij jullie! Bedankt Elise, Kathleen, Karen, Graciela, Francis, Jozefien en Isabelle!

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Last but not least, Sigrid, je geeft kleur aan mijn leven. Bedankt voor je steun en liefde. Je zet deze dromer af en toe met de voeten terug op de grond, wat nodig is (dit zeker 's ochtendsvroeg).

September 2014,

Thomas Schaubroeck