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Smart land management for bio-based economies: simultaneous optimization of biomass production and other ecosystem services in forests

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Pieter

Content

Summary

Our planet faces different grand challenges, such as climate change, resource depletion and biodiversity losses, all of which put pressure on the ecosystem services on which life depends. In forest ecosystems, these three challenges collide and contest the current management approaches. On the one hand, decarbonizing our economy by shifting towards a bio-based economy increases the demand for woody biomass from forests for material and energy purposes. Harvesting additional biomass from forests thus stipulates new questions on technical, economic and ecological constraints. On the other hand, the magnitude of biodiversity loss and decline of ecosystem services is unprecedented and requires urgent action. In this thesis we aim to understand how forest and nature managers are to answer the simultaneously arising challenges of a strong demand for woody biomass, a need for biodiversity conservation and the provision of a multitude of ecosystem services while adhering to new models of participation, governance and management. We selected 'Bosland' (Belgium), a nature and forest area mainly covered by pine plantations on former heathland, as a case study. Bosland is pioneering novel governance settings, has a high demand for woody biomass and multiple other ecosystem services and holds important biodiversity values. Bosland allowed us (i) to investigate how novel governance arrangements were developed and implemented, (ii) to empirically quantify the potential of additional woody biomass harvest for a bio-based economy and (iii) to investigate how biomass production, biodiversity conservation and recreation are interrelated and whether trade-offs can be minimised.

To uncover novelties in forest management and governance, we adopted a learning history approach to study the coming into existence of the Bosland project and used transition analysis to reveal innovative aspects. The Bosland project originated as a collaboration between public forest owners and non-profit organisations, after a change in legislation that increased the administrative workload. After extensive public participation a long-term vision was co-created to guide the short term management actions. The project went into an implementation phase with the launch of a master plan that institutionalized stakeholder participation and consolidated the collaboration between the partners. In general, we found many striking differences between traditional forest management and the Bosland approach, which can be of inspiration for both policy makers and practitioners that are exploring more appropriate approaches to deal with the grand challenges. Many of the novelties introduced and piloted in Bosland, align with the relative new concept of ecosystem stewardship. This style of ecosystem governance is specifically targeted at answering landscape changes and uses an adaptive management style in which scientists and managers closely collaborate to constantly adjust management targets according to what is needed.

To investigate the potential of Bosland as a producer of additional woody biomass for a bio-based economy, we compared the technical and economic constraints of different harvest strategies for whole tree harvesting (WTH) in clear-cuts and thinnings. On the clear-cuts we found that the use of a mobile chipper yielded better results than the currently used road-side chipper on fuel consumption, chip quality and time and cost-effectiveness. In the thinnings, an excavator, a forwarder and a road-side chipper were more cost-efficient than a harvester, a tractor with trailer and a mobile chipper respectively. Both in the clear-cuts (40% of the crowns) and in the thinnings (46% of the crowns) substantial harvest losses occurred. The major conclusion was however that the margin of profit on harvesting additional biomass as wood chips was very limited under the current circumstances. It was much more profitable to harvest logs separately, even in early thinnings and to minimize top bucking diameters to maximize the share of logs compared to the amount of wood chips. We also determined sustainability constraints of harvesting additional biomass, by inventorying nutrient stocks in biomass and soil before and after WTH. With the help of a nutrient budget model, the long term effect of WTH and stem-only harvesting (SOH) on soil fertility was determined. The results showed a sharp decline of base cations and phosphorus when WTH was applied in an intensive way in Bosland. This would most likely cause growth reductions in the near future and to guarantee long term sustainability, we recommend to apply SOH under the given circumstances.

On a landscape scale, we demonstrated trade-offs between biodiversity conservation and both wood harvesting and recreation but we also presented smart solutions to integrate these different management goals. By primarily clear-cutting forest stands adjacent to existing open patches, habitat networks of species of open landscapes were reinforced, while damage to populations of forest species was limited. Recreation had a negative impact on some of the focal species in our study, but a smart trail design, avoiding the core of the study area, could host a higher number of visitors with a very limited impact on the vulnerable focal species.

In summary, we found a very limited potential to harvest additional woody biomass from pine stands due to ecological, economic and technical constraints (in decreasing order of importance). On a landscape scale, wood harvest can be combined with recreation and biodiversity conservation in a robust forest and nature area such as Bosland when applying a smart land management, spatially optimizing synergies between services. There is a clear need of more empirical research on a stand and a landscape scale, also in other systems and other regions. We formulated different recommendations for forest managers and policy makers and stressed the need for different management models. The Bosland approach can be considered as an innovative example of ecosystem stewardship, specifically aimed at collaboration, participation and explicitly embracing transformation. To accelerate the transition towards novel management and governance styles that are better fit at dealing with grand challenges, there is a need for more examples on how to design change processes and implement transformative ways of doing and organizing in governance models.

Samenvatting

Onze planeet staat voor de grote uitdaging om grote bedreigingen zoals klimaatverandering, de uitputting van hulpbronnen en biodiversiteitsverlies te stoppen. Dit heeft direct en indirect impact op het beheer van bos- en natuurgebieden waar die uitdagingen samenkomen. Zo is een transitie naar een “bio-based economy” nodig om klimaatopwarming te stoppen, wat de vraag naar houtige biomassa uit bossen doet stijgen, zowel voor materiaal- en energietoepassing. De technische, economische en ecologische beperkingen van bijkomende biomassa-oogst uit bossen zijn nog grotendeels onbekend. Intussen hebben biodiversiteitsverlies en de afname in ecosysteemdiensten geleid tot een sterke nood om de resterende biodiversiteitswaarden in de bossen goed te beschermen. Integratie van verschillende ecosysteemdiensten en biodiversiteitsbehoud kan enkel met een slim landbeheer. In deze thesis willen we kennis aanleveren aan bos- en natuurbeheerders en beleidsmakers over nieuwe beheerstrategieën en bestuursmethoden die beter geschikt zijn om bossen te beheren onder de huidige bedreigingen en op het leveren van een waaier aan ecosysteemdiensten aan een veelvoud van belanghebbenden. We selecteerden Bosland (België), een natuur- en bosgebied met dennenplantages op voormalige heide, als studiegebied. Bosland is een pionier op vlak van bestuursmethoden, er is een grote vraag naar houtige biomassa en andere ecosysteemdiensten en het gebied herbergt belangrijke biodiversiteitswaarden. In Bosland (i) onderzochten we hoe nieuwe bestuursmethoden werden ontwikkeld en toegepast, (ii) bepaalden we empirisch hoeveel houtige biomassa er bijkomend kan geoogst worden en (iii) bekeken we hoe de trade-offs tussen biomassaproductie, biodiversiteitsbehoud en recreatie kunnen geminimaliseerd worden.

We bestudeerden het Boslandproject met een “learning history” en met een transitieanalyse legden we innovatieve aspecten bloot. Het project ontstond als een samenwerking tussen publieke boseigenaars en ngo’s, na een verandering in de wetgeving die de administratieve last verhoogde. Na een uitgebreid participatietraject werd een gemeenschappelijke lange termijnvisie opgesteld om beheeracties op kortere termijn te sturen. Een masterplan integreerde participatie in de beheerstructuur en legde de samenwerking tussen de partners vast. We vonden veel verschillen tussen het klassieke bosbeheer en de aanpak in Bosland, die kan beschouwd worden als een voorloper die veel gelijkenissen vertoont met “ecosystem stewardship”. Deze vrij nieuwe stijl van bestuur is ontwikkeld om om te gaan met grote veranderingen en hanteert een adaptief beheer waarin onderzoeker en beheerders nauw samenwerken om de beheerdoelen constant bij te stellen.

Om het houtige biomassapotentieel van Bosland te bepalen, vergeleken we de technische en economische beperkingen van verschillende oogststrategieën voor het oogsten van volledige bomen (WTH) in kaalslagen en dunningen. Bij een kaalslag vonden we dat een mobiele hakselaar beter was dan een hakselaar aan de perceelsrand (zoals nu courant gebruikt wordt), zowel qua brandstofgebruik, houtsnipperkwaliteit en tijds- en kostenefficiëntie. Bij de dunningen waren een rupskraan met knipkop, een forwarder en een hakselaar aan de perceelsrand kostenefficiënter dan een harvester, een tractor met uitrijkar en een mobiele hakselaar respectievelijk. Zowel in de kaalslagen (40%) als in de dunningen (46%) bleef een groot deel van de kruinen in het bestand als oogstverlies. De belangrijkste conclusie was echter dat de winstmarge op de oogst van bijkomende biomassa als houtsnippers heel beperkt was onder de huidige omstandigheden. Het was veel rendabeler om rondhout apart te oogsten (zelfs in vroege dunningen) en om de aftopdiameter zo klein mogelijk te houden om het aandeel rondhout zo groot mogelijk te maken t.o.v. het aandeel houtsnippers. We bepaalden ook duurzaamheidsbeperkingen van bijkomende biomassaoogst, door de nutriëntenvoorraden in de biomassa en de bodem voor en na de oogst op te meten. Met behulp van een model werden de langetermijneffecten van WTH en het oogsten van enkel stammen (SOH) op bodemvruchtbaarheid bepaald. Onder een intensief beheer met WTH vonden we een sterke afname van de voorraad van basische kationen en fosfor in Bosland. Dit leidt hoogstwaarschijnlijk tot een afname van de groei in de nabije toekomst, we raden dus aan om SOH toe te passen.

Op een landschapsschaal vonden we trade-offs tussen het beschermen van biodiversiteit en zowel houtoogst als recreatie, maar we stelden ook slimme oplossingen voor om de verschillende beheerdoelen te integreren. Door prioritair bestanden te oogsten naast bestaande open plekken, wordt het habitat van open-plek-soorten versterkt, terwijl de schade voor typische bossoorten beperkt blijft. Verstoring door wandelaars had een negatief effect op enkele indicatorsoorten uit de studie, maar een slim ontwerp van de wandelpaden dat de kern van het gebied vrijwaart, kan een stijgend aantal wandelaars opvangen met een heel beperkte impact op de kwetsbare soorten.

Er zijn dus weinig mogelijkheden om bijkomend biomassa te oogsten uit de dennenbestanden, door ecologische, economische en technische beperkingen (in dalende mate van belangrijkheid). Op een landschapsschaal is het mogelijk om recreatie en houtoogst te combineren met biodiversiteitsbehoud in een robuust natuur- en bosgebied zoals Bosland, als er een slim landbeheer wordt toegepast, die synergiën tussen diensten ruimtelijk optimaliseert. Er is duidelijk nood aan meer empirisch onderzoek op bestands- en op landschapsniveau in andere (bos)ecosystemen en andere regio's. We formuleerden verschillende aanbevelingen voor bosbeheerders en beleidsmakers en benadrukten de nood aan andere beheermodellen. De Bosland-aanpak is een innovatief voorbeeld van ecosystem stewardship, gericht op samenwerking, participatie en het omgaan met veranderingen. Om de transitie naar nieuwe beheerstrategieën en bestuurmethoden te versnellen is er nood aan meer voorbeelden die een soortgelijk transformatieproces toepassen.

List of abbreviations

AIC	Akaike's Information Criterion
B_{econ}	Economic biomass harvest potential
B_{sust}	Sustainable biomass harvest potential
B_{tech}	Technical biomass harvest potential
B_{theo}	Theoretical biomass harvest potential
Ca	Calcium
CEC_e	Effective Cation Exchange Capacity
dbh	diameter at breast height
E_H	Export by Harvesting
E_L	Export by Leaching
GHS species	Fauna and flora associated with Grassland, Heathland and Sandy habitats
GLM	Generalized Linear Model
GMt	Green Metric ton
GWh	Giga Watt hour
Ha	Hectares
I_D	Input by Deposition
I_W	Input by Weathering
K	Potassium
L F and H layer	Litter, Fragmentation and Humus layer
Mg	Magnesium
Mj	Mega joule
NCV_0	Net calorific value
N	Nitrogen
OSB	Oriented Strand Board
P	Phosphorus
sd	standard deviation
SOH	Stem Only Harvesting
SNM	Strategic Niche Management
SMH	Scheduled Machine Hours
TM	Transition Management
WTH	Whole Tree Harvesting

1. General introduction

1.1. Changing ecosystems and a changing society

During the past millennia, the planet's environment has been relatively stable. This period of stability, called the Holocene, seems to have come to an end since the Industrial Revolution. Human actions have since become the main driver of global environmental change and this new era has been called the Anthropocene (Crutzen, 2002). These human based changes could push the Earths system outside the stable environmental state of the Holocene. Rockstrom *et al*. (2009) proposed a framework based on "planetary boundaries" to maintain the Holocene state. These planetary boundaries define the safe operating space for humanity. Nine crucial processes were defined and their state was evaluated against a threshold that should not be crossed to avoid a shift to a new state with potentially disastrous consequences for humans (Rockstrom *et al.*, 2009). Three of the processes that were evaluated had already moved beyond the safe operating space, namely climate change, disruption of the biogeochemical cycles of nitrogen and phosphorus and mostly biodiversity loss; these processes have the potential on its own to drive the Earths system into a new state (Rockstrom *et al.*, 2009; Steffen *et al.*, 2015). Although the planetary boundaries are defined for separate processes, the different boundaries are tightly coupled, crossing the climate change boundary for example will also accelerate biodiversity loss.

Anthropogenic climate change is caused by an increased concentration of greenhouse gasses in the atmosphere (IPCC, 2014). To keep climate change within safe boundaries, future greenhouse gas emission should be strongly reduced, which calls for a drastic change in current practices worldwide (IPCC, 2014). The 21st United Nations Conference of Parties has led to the Paris agreement, an internationally recognized agreement governing greenhouse gas emissions from 2020, a hopeful step forward. However translating the overall goals to practical action remains an enormous challenge. To decarbonize our economy multiple strategies and solutions will need to be developed (EU 20 20 20). One of these strategies is to replace fossil resources with renewable biological resources (Jenkins, 2008). This so-called bio-based economy could offer a sustainable alternative for the production of energy, chemicals and materials from bio-renewable feedstocks from agriculture, forestry and aquatic resources. The transition towards a bio-based economy will increase the demand for these bio-renewable feedstocks, including the demand for woody biomass from forests (see §1.2).

At the same time, biodiversity losses are occurring on a very fast rate, which is illustrated by a species extinction rate which is up to one thousand times faster than the fossil record (MEA, 2005).

The main direct drivers causing this biodiversity crisis are habitat loss and fragmentation, pollution with mainly nitrogen and phosphorus, overexploitation, introduction of invasive species and climate change (MEA, 2005). All of these five processes are caused by humans and these drivers are reinforcing each other (Brook *et al.*, 2008). In the Convention on Biological Diversity, signed in Rio de Janeiro in 1992, representatives of almost every country on the globe spoke out the ambition to conserve biodiversity and exploit it in a fair and sustainable way. The target to halt the decline of biodiversity by 2010 however has largely failed and remains a big challenge for the next decennia (Gilbert, 2010). One of the main reasons for the failure to stop biodiversity loss is the fact that we fail to recognize and anchor the value of biodiversity in our current economic models, illustrated by the lack of money set aside for conservation projects (Gilbert, 2010). To increase the willingness to invest in biodiversity, it might be useful to demonstrate the link between human well-being and conservation and use principles of the emerging field of ecological economics (Roman *et al.*, 2009)(see §1.3).

Parallel to the challenges arising from the changing environment also societal changes are occurring. Citizenship norms are shifting from duty-based citizenship to a more engaged citizenship that seeks to place more control over political activity in the hands of the citizenry (Dalton, 2008). Over the last decades there has been a rising call for participation in research, policy and practice of natural resource management and biodiversity conservation (Schultz *et al.*, 2011). Involvement of stakeholders could strengthen legitimacy of decision making, could improve accuracy as a more diverse knowledge base is utilized and could increase overall efficiency (Schultz *et al.*, 2011). In the meantime, participation has become a key consideration in environmental policy-making and was institutionalized in the Aarhus Convention 1998 (Collins & Ison, 2009). The Aarhus Convention grants the public rights regarding access to information and public participation on matters concerning the environment. More and more, the perception that ecosystems and societies are interdependent gets wide acceptance. This interdependence implies that people-oriented management and conservation of ecosystems are more likely to succeed than protectionism based on authoritarian practices (Schultz *et al.*, 2011). The complexity of socio-ecological systems asks for another style of decision making, much more process based, where stakeholders are continuously learning from each other (Garmendia & Stagl, 2010). Integration of these new insights in practice is not easy and requires transforming policy making which first asks for a mind shift of the decision makers. This sometimes results in frictions and critics on the participation paradigm such as that it would slow down decision making or would dilute the impact of scientific knowledge (Schultz *et al.*, 2011).

These changes, both in ecosystems and in societies, challenge the current practices in many fields. Sustaining a society within safe planetary boundaries asks for new paradigms and systemic changes. It is clear that forest and nature management will be strongly influenced by the current climate and biodiversity crisis and the shift towards engaged citizenship and increased demand for stakeholder participation. In the next paragraphs we look into the relation between forest management and climate mitigation (§1.2), conservation of biodiversity and its values (§1.3) and the societal transition (§1.4). We aim to identify knowledge gaps that should be filled to inform a future management that can better deal with the current challenges.

1.2. The bio-based economy and its impact on forests

An important strategy for mitigating climate change is to shift from fossil-based to bio-based resources, often referred to as the transition towards a bio-based economy (Jenkins, 2008). This results in an increasing demand of all kinds of biomass, from agriculture, from aquaculture, but also from forests. Forests produce woody biomass that can be used both as a material and as a source of energy. Production of woody biomass yields a large direct economic value to forest owners. Every year 485 million m³ of wood are felled in Europe (Eurostat, 2011a). Next to the use of woody biomass for material purposes, the use of woody biomass for bioenergy has increased with almost 80% in the 27 European Union member states between 1990 and 2008 (Eurostat, 2011a). Moreover, the demand is expected to keep rising and to double by 2030, mainly as a result of the EU 20-20 objectives (Mantau *et al.*, 2010)(Figure 1.1). For more than two-thirds, this woody biomass originates from forests (Mantau *et al.*, 2010).

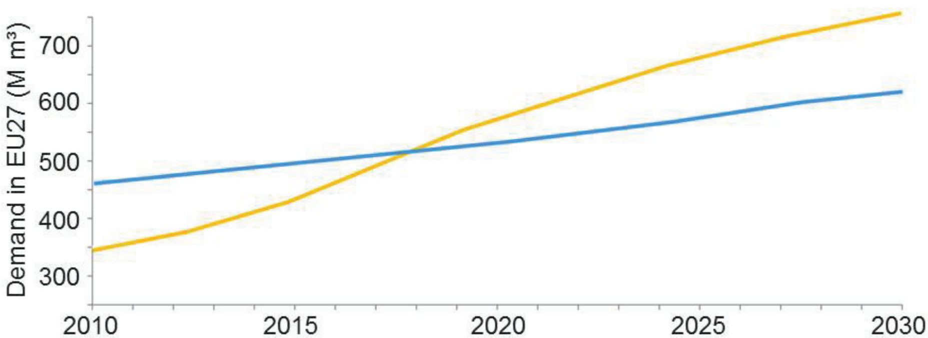

Figure 1.1: Expected change of demand of wood for material (blue) and energy (orange) purposes in EU27 (after Mantau *et al.* (2010)*)*

Parallel to the rising demand for woody biomass an increase in the price can be expected. Raunikar *et al.* (2010) used a global model on forest products under different future IPCC scenarios and found that the price of fuel wood would rise and converge towards the price of pulpwood by about 2025. This would lead to an increase of the use of pulpwood for energy purposes. At that point the price of all wood (fuel wood, pulp wood, but also quality wood) would then continue to rise steadily. Härtl & Knoke (2014) elaborated on the same issues but also included influence of future (rising) oil prices and found very similar results. There is thus a clear trade-off between the use as wood for energy purposes and for material purposes. The modelled rising trend in the price of woody biomass can already be seen in the current prices. In Germany for example, the price of wood chips has steadily risen and has doubled between 2003 and 2013 (Figure 1.2)(Lutz, 2013). However the German data show that woody biomass is a more cost-effective source of energy than fossil fuels.

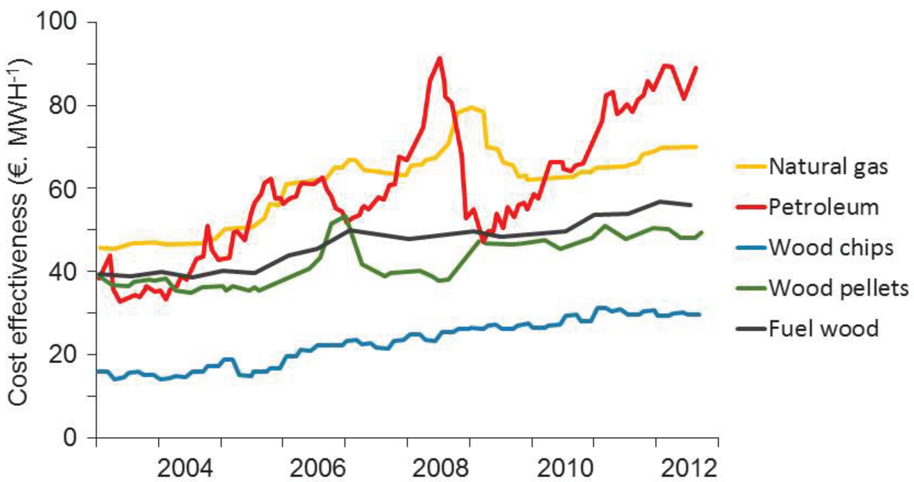

Figure 1.2: Price trend of different fuels in Germany per unit of energy (Lutz, 2013).

In Belgium, about 4 million m³ of wood are harvested yearly (Eurostat, 2011a). About 14 000 people are working in the wood and wood products sector, which represents an output of 3,7 billion euro (and there is still three times as much people and money in wood related sectors such as paper, paper products and printing (Eurostat, 2011a)). Large parts of the sector are working with imported, manufactured wood. The product group on wooden fibreboard and packaging material, however, handles a lot of round wood, also from Flemish forests. Companies within this product group represent 3766 jobs and an output of 1.7 billion (Bosbode 2015). A large part of the Belgian wood production is located in Wallonia. Flanders has a low forest cover of about 11%. The total forest area in Flanders is about 150 000 ha, of which about 30 % are public forests (Waterinckx & Roelandt, 2001). The average stand volume in Flemish forests is about 216 m³/ha and annual yearly increment is estimated as 5 m³/ha. Through the official wood sales from the public forests, roughly 200 000 m³ of wood is sold every year. About 72 000 m³ of wood from private forests was sold through the forest groups in 2011, of which about 63 000 m³ as industry wood (OVAM, 2013). There is also a considerable number of private forest owners that sell their wood on their own initiative, numbers on this are largely lacking.

The rising demand for woody biomass for energy purposes resulted in an increased import, for Belgium and the Netherlands mostly as pellets from North-America (Sikkema *et al.*, 2010). This intercontinental transport is controversial because of sustainability issues (Greenpeace, 2011) and has been issue of recent debate in local newspapers (Figure 1.3). The rising demand also stimulated the interest in local production of wood chips and pellets, stipulating new questions for the forestry sector about the cost-effectiveness of different harvest strategies. In Flanders, the legislation only allows the production of renewable energy from smaller assortments of woody biomass that cannot be used as a material (Vlaamse Regering, 2004). Such a cascaded use is a logical choice from a sustainability point of view, it maximizes efficiency of biomass use and stimulates a circular economy (Keegan *et al.*, 2012). For this reason, the newly applied forestry methods to produce wood chips and pellets in Flanders mainly include whole-tree harvesting in early thinnings and additional harvest of biomass that was previously left in the forest floor after roundwood harvest. In recent years the harvest residues from exploitations executed by the Agency of Forest and Nature from the Flemish community themselves have been sold for energetic valorisation (2046 tons in 2011 (OVAM, 2013)). There are no numbers to be found of the harvest of residues in private forest in Flanders. It is clear that harvest of additional woody biomass, so on top of the harvest of pulp and industry wood, is still a small but emerging business in Flanders and neighbouring regions. Currently the scientific and practical knowledge on harvesting additional woody biomass is mainly concentrated in regions such as Canada and Scandinavia. However, the emerging patterns from these studies are very hard to transfer to other regions, as harvest of additional woody biomass is species-, site- and practice specific (Helmisaari *et al.*, 2014). Flanders and neighbouring regions are for example characterized by a low total forest area and a very disintegrated forest ownership (Van Gossum *et al.*, 2011). *Knowledge on the technically harvestable amount of biomass from different forest types is partly lacking. Moreover the practical feasibility and the cost-effectiveness of different harvest strategies for additional biomass harvest is unknown for Flanders and neighbouring regions.*

Figure 1.3: Overview of headlines of recent Flemish newspapers. (DM = De Morgen, DS = De Standaard)

The large scale utilization of woody biomass for bioenergy also raises serious questions on sustainability aspects (Schulze *et al.*, 2012). Biomass plays an important role in several ecological processes and extracting additional biomass from forests could have several unintentional negative effects. For example, by extracting additional biomass also more nutrients are exported from the forest, as the nutrient concentrations in the crown are much higher than in the logs. Depending on the forest and soil type additional biomass harvesting could thus impact the long term soil fertility and the future productivity of forest stands (Walmsley *et al.*, 2009; Wall, 2012). Additional harvest of woody biomass can also result in a loss of biodiversity (Berger *et al.*, 2013). For instance, when harvest residues remain in the stand these form valuable micro-habitats for different species, such as small mammals (Carey & Harrington, 2001), saproxylic beetles (Jonsell *et al.*, 2007) and fungi (Nordén *et al.*, 2004). However, also positive effects of additional biomass removal on some species occur. For instance, for insects that prefer warm and sunny conditions (Vandekerkhove *et al.*, 2012). Additional harvest of woody biomass could also influence the preference of recreationists and, for instance, lead to a decreased number of visitors (Verkerk *et al.*, 2014). Ecosystem impact assessment of additional biomass harvest is thus a complex issue, with different aspects and sometimes contrasting results (Riffell *et al.*, 2011). Most of the previous research on the impact of biomass harvesting was executed in different regions or based on large-scale models. *To determine a sustainable harvesting schedule for additional biomass for Flanders and neighbouring regions there is a strong need for empirical studies looking at the impact of additional biomass harvesting on a stand scale.*

1.3. Biodiversity loss and ecosystem services

Globally it is estimated that more than half of the known terrestrial plant and animal species live in forests (MEA, 2005). Consequently, habitat loss through deforestation is one of the major causes of biodiversity loss (Brockerhoff *et al.*, 2008) and forest remnants in cleared or urbanized landscapes form important biodiversity hotspots (Godefroid & Koedam, 2003). Safeguarding remaining biodiversity in forests should be part of any conservation strategy. Biodiversity also plays an important role in different ecological processes that deliver benefits to society (Cardinale *et al.*, 2012). These benefits consist of goods and services that people obtain from nature and are called ecosystem services (MEA, 2005). Ecosystem services have a certain value to society and contribute to human well-being (MEA, 2005). Ecosystem services can be categorized in many ways, a common approach is the functional grouping of services in four categories: provisioning services, regulating services, cultural services and supporting services. Figure 1.4 demonstrates the link between the different categories of ecosystem services and different constituents of human wellbeing and lists some examples of every service category.

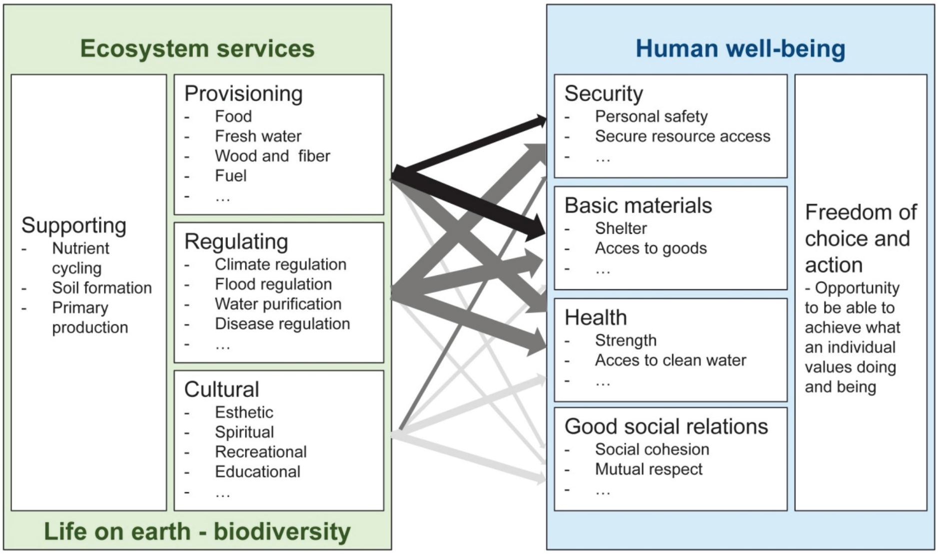

Figure 1.4: The different categories of ecosystem services with examples and their linkage to different constituents of well-being. Arrows width depicts the intensity of the linkage and the darkness of the arrow colour depicts the potential for mediation by socio-economic factors (MEA, 2005).

A continued supply of ecosystem services is threatened due to the ongoing global environmental changes and ecosystem degradation, while, at the same time, the demand for ecosystem services is increasing with human population growth (Cardinale *et al.*, 2012). The current biodiversity crisis is directly affecting ecosystem service provision and is thus a threat to human well-being. Unravelling the link between biodiversity loss, ecosystem services and human well-being can help to raise awareness about the gravity of the current crisis and provide insights on effective levers to halt and/or reverse biodiversity loss. The ecosystem services concept can be helpful in communication and can underpin biodiversity conservation. To achieve success in conservation, the ecosystem service concept needs to be integrated throughout the decision making process (Figure 1.5)(Daily *et al.*, 2009). A certain ecosystem will deliver services to society, that represent a certain value. With the help of biophysical models it is possible to quantify the services delivered by ecosystems. The value for society can be estimated with the help of economic and cultural models. Note that this process of valuation does not necessarily have to lead to monetary values (see box 1). The translation of the ecosystem services to values provides useful information to policymakers and managers that can take incentives and decisions about different management scenarios that will at their turn influence biodiversity and the ecosystems. In this way, the policy cycle can optimize management of ecosystems for a desired and sustainable ecosystem service delivery while conserving biodiversity.

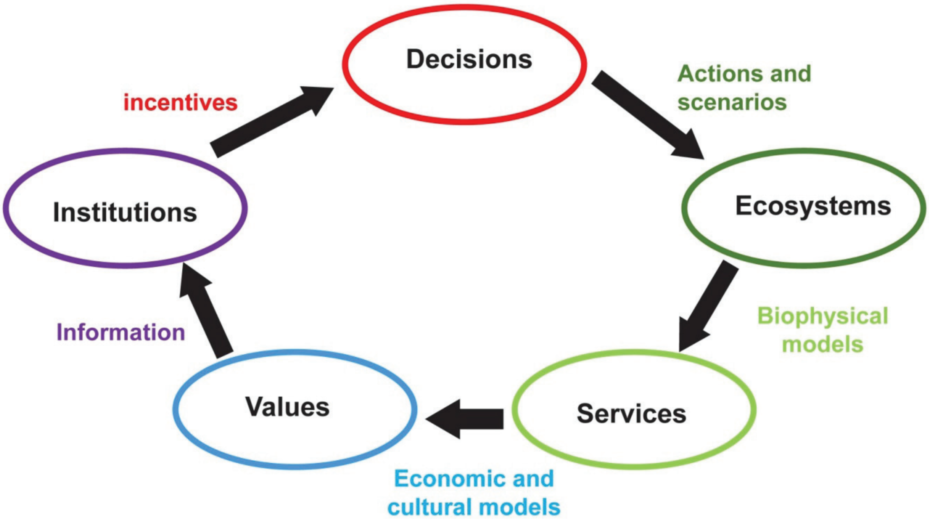

Figure 1.5: Integration of ecosystem services concept in decision making (Daily *et al.*, 2009).

A good functioning of the policy cycle asks for a close interaction and a good collaboration between scientists (mainly working on the biophysical, economic and cultural models), the policy-makers (translating information to incentives) and the managers (deciding on management actions and scenario's). Throughout the policy cycle, communication between scientists, policy makers and managers is necessary, combined with a participatory approach with the stakeholders and the general public (Daily *et al.*, 2009). Ecosystem science and practice has not yet fully embraced this approach (Mace *et al.*, 2012). *Currently there is an urgent need to develop the interdisciplinary science of ecosystem management integrating knowledge from ecology, conservation biology, resource economy and other fields* (Mace *et al.*, 2012) .

Biophysical science often focusses on the link between management actions and scenarios on ecosystems and between ecosystems and the delivered services. A certain management scenario could promote a certain ecosystem service while having a negative effect on other ecosystem services. We talk about synergies and trade-offs between ecosystem services if services reinforce or counteract each other respectively. Trade-offs and synergies between services can be very context dependent and hard to predict. However, some more general trends can be detected between ecosystem services categories on a range from natural landscapes with a high biodiversity to urban landscapes with a low biodiversity (Figure 1.6)((Braat & ten Brink, 2008).

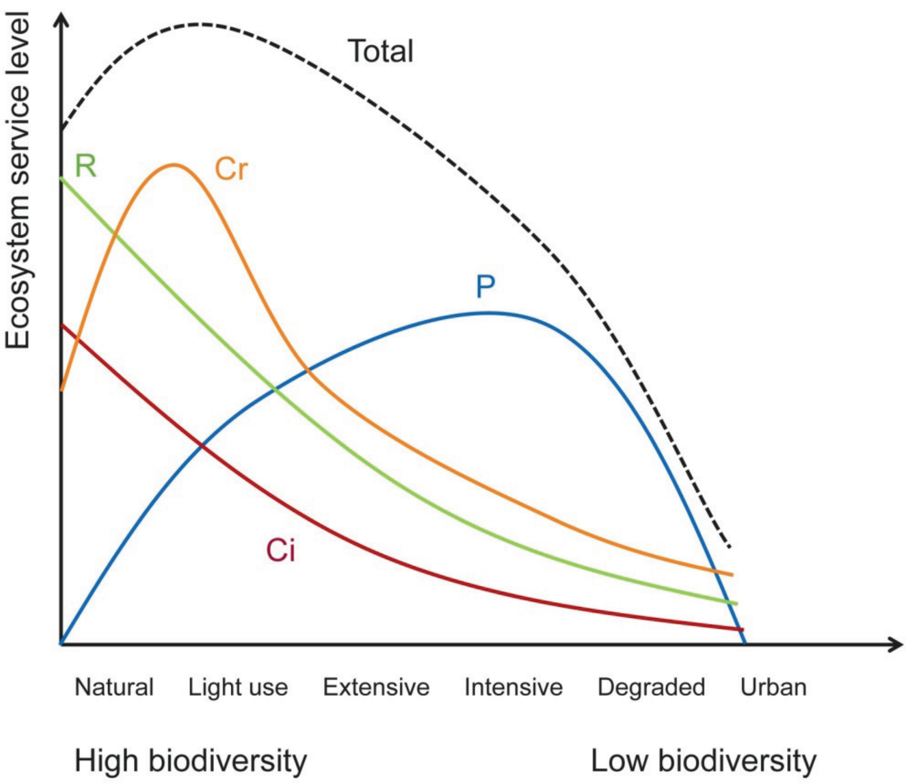

Figure 1.6: Conceptual representation of ecosystem services delivery of different categories on a range from natural to urban landscapes (Braat & ten Brink, 2008)*. R, regulating services; P, provisioning services; Ci, cultural information services; Cr cultural recreation services.*

We observe a trade-off between provisioning services that optimally deliver under intensive management and regulating and cultural services that have a higher value under more natural situations. For example the provisioning value of intensive agriculture will be higher than under extensive management, but the trade off with regulating services can still result in a lower total ecosystem service value (Power, 2010). Trade-offs and synergies also exist between different services within one category and are often depending on the context and the spatial and temporal scale (Rodriguez *et al.*, 2006; Gamfeldt *et al.*, 2013). *More examples of studies analysing trade-offs and synergies between biodiversity and ecosystem services, and among ecosystem services themselves, are needed to foster an effective management, answering the demands of different stakeholders and safeguarding the future ecosystem services delivery* (Mace *et al.*, 2012).

Box 1: Monetary valuation: strong argument or nature for sale?

Valuation of ecosystem services strives to quantify the value of ecosystem services. These values do not necessarily have to be monetary values, however this happens often and monetary valuation is strongly linked to the ecosystem services concept, certainly in general perception. Of course, without fresh air and pure water for example, the economies of the Earth would no longer function. So in one sense their total value to the economy is infinite and it makes no sense to economically quantify the value of ecosystems (Costanza *et al.*, 1998). However it can be useful to estimate the 'marginal' value of ecosystem services, (i.e. the estimated change in economic value compared to the change in ecosystem services from the current level)(Costanza *et al.*, 1998).

Monetary valuation yields several benefits:

- Comparing economic values of services can be very relevant information to decide upon management scenarios (e.g. mangrove conservation vs shrimp farming in Sathirathai & Barbier (2001))
- The monetary value of ecosystem services can be an extra argument to protect ecosystems. (e.g. the enormous monetary values of ecosystem services in the Leuser national park (Indonesia) as an extra argument against deforestation (van Beukering *et al.*, 2003))
- Monetary valuation can be an eye-opener, stressing the importance of nature to policy makers and to a general public (Posner *et al.*, 2016). (e.g. every year 260 ton of fine particulate matter is captured in Flemish pine forests, resulting in a health gain of 40 million euro (Schaubroeck *et al.*, 2014))
- Monetary valuation can be a method to integrate external environmental costs in the price of products and services. (e.g. organic vs conventional apples (Reganold *et al.*, 2001))

However monetary valuation also holds several pitfalls:

- One could perceive that forest and nature is a negotiable good (MEA, 2005). According to this reasoning, one could destroy all forests in Belgium if simply paying enough.
- Monetary valuation is mostly unsure, sometimes unprecise and often depending on the valuation method. This can cause confusion, resistance and even abuse (MEA, 2005).
- One could think that our ecosystems only need protection for the sake of the economy. This:
    - Could lead to unethical choices, such as no longer protecting species and ecosystems that are economically unimportant (Deliège & Neuteleers, 2011);
    - Could reduce public support for nature and forest, due to a predominantly frigid and functionalistic vision (Deliège & Neuteleers, 2011).
    - Neglects the subjective intrinsic value of nature and forest to people (Deliège & Neuteleers, 2011; Sandler, 2012);
    - Neglects the objective intrinsic value of nature (a concept under debate), the fact that species have a good for their own and that species extinction is a loss, independently of the subjective value awarded by people (cf. Sandler (2012)).

Therefore it is very important to rightly use the ecosystem services concept. When applying monetary valuation, it is very important to:

- Mention that this price is rather a shadow price than a market price;
- Mention how this value was estimated, how accurate and how certain the value is. A range seems more appropriate than a fixed number;
- Realize that the ecosystem service value is not the only reason why forest and nature need management and protection. The intrinsic value of forest and nature and the ethics concerning human induced species extinctions stand apart from the estimated monetary value.

1.4. Towards forest stewardship?

Over the last decades there has been a rising call for participation in natural resource management and biodiversity conservation and this has also influenced forest management (Schultz *et al.*, 2011). Since the 1960s and 1970s, there has been an increased appreciation of local knowledge, leading to some pioneering projects in which experts and local stakeholders worked complimentary (Bruña-

García & Marey-Pérez, 2014). Public participation is thus clearly not a new concept, but integration in forest planning long lagged behind (Bruña-García & Marey-Pérez, 2014). Gradually people are getting more interested in influencing the decision-making process and in changing forest management practices. This is also the reason why professionals in forestry need new communication styles, also addressing a higher number of non-professionals (Tyrväinen *et al.*, 2006). Recently there have been different examples of forest management planning efforts including participation, such as stakeholder participation in the final phases of forest zoning (Sugimura & Howard, 2008), the use of public participation GIS to integrate stakeholders priorities in forest planning (De Meo *et al.*, 2013) and participatory approaches to develop alternative scenarios for forest resource management (Haatanen *et al.*, 2014). An important trend can also be observed in the way forest are governed. In the 1960s and 1970s forests were mostly managed under hierarchical governance and closed co-governance (Arnouts *et al.*, 2012). This means that governing was mainly the domain of the government, with non-governmental actors in a subservient or a very restricted role. Gradually a shift is at least partially occurring towards new modes of governance, including open co-governance and self-governance (Arnouts *et al.*, 2012). This implies that non-governmental actors hold an autonomous position next to governmental actors (open co-governance) or that the governmental actors keep distance and allow a predominance of non-governmental actors (self-governance)(Arnouts *et al.*, 2012).

More and more the complexity of interacting ecosystems and social systems is acknowledged (Elbakidze *et al.*, 2010). The challenge of accommodating multiple users' claims and interests is addressed in different methodological approaches. Examples include ecosystem management (e.g. Dekker *et al.* (2007), Cosens (2013)), adaptive (forest) management (e.g. Temperli *et al.* (2012)) and more recently ecosystem stewardship (Folke *et al.*, 2009; Chapin *et al.*, 2010). Ecosystem stewardship is an action oriented framework that was developed with specific attention to the rapid changes threatening ecosystems and aims to foster the socio-ecological sustainability of the earth (Chapin *et al.*, 2010)(Table 1.1).

Uncertainty and changes have always been characterizing social-ecological systems and according to ecosystem stewardship, this uncertainty should not be an obstruction to action (Folke *et al.*, 2009). Ecosystem stewardship explicitly endorse the integration of ecological sustainability and socio-economic sustainability of human well-being, recognizing that people are integral components of social-ecological systems (Chapin *et al.*, 2010). Three overlapping sustainability approaches are integrated in ecosystem stewardship (Chapin *et al.*, 2010): (i) reducing vulnerability to expected changes (Turner *et al.*, 2003); (ii) resilience to sustain desirable conditions despite changes (Folke, 2006); (iii) leaving undesirable change trajectories when windows of opportunity open (Folke *et al.*, 2005). By building on previous knowledge from these three approaches, ecosystem stewardship provides a perspective that better allows to manage the grand challenges that are threatening society (Chapin *et al.*, 2010).

Table 1.1: Characteristics of ecosystem stewardship, compared to steady-state resource management (table from Chapin *et al.* (2010)*)*

Characteristic	**Steady-state resource management**	**Ecosystem stewardship**
Reference point	Historic condition	Trajectory of change
Central goal	Ecological integrity	Sustain social-ecological systems and delivery of ecosystem services
Predominant approach	Manage resource stocks and condition	Manage stabilizing and amplifying feedbacks
Role of uncertainty	Reduce uncertainty before taking action	Embrace uncertainty: maximize flexibility to adapt an uncertain future
Role of research	Researchers transfer findings to managers who take action	Researchers and managers collaborate through adaptive management to create continuous learning loops
Response to disturbance	Minimize disturbance probability and impacts	Disturbance cycles used to provide windows of opportunity
Resources of primary concern	Species composition and ecosystem structure	Biodiversity, well-being and adaptive capacity

Ecosystem stewardship can be seen as the next step in an evolution in Western resource management, from exploitation, where sustainability was not an important consideration, to paradigms targeting maximum sustainable yield of one resource to recent approaches of ecosystem management (Chapin *et al.*, 2010)(Figure 1.7). Maximum sustained yield aims at maximizing the yield in a sustainable way, but often tends to overexploit targeted resources for different reasons (listed in Holling & Meffe (1996)). Ecosystem management overcomes most of these problems and aims to sustain multiple ecosystem services. However ecosystem management often uses static, historic reference points that are not achievable under the current challenges (Chapin *et al.*, 2010). The transition from ecosystem exploitation towards ecosystem stewardship has been running in parallel with the higher described changes in participation and governance in forest management.

Despite the fact that ecosystem stewardship was stated to be “sufficiently mature to make important contributions to all social-ecological systems” (Chapin *et al.*, 2010), practical examples, integrating social and ecological sustainability have rarely been described. As stressed by Power &

Chapin (2010), most people learn from examples and new, inspiring examples will add to our knowledge of the paths toward a renewed and sustainable relationship with our planet's working and natural ecosystems. *Examples of forest management projects with a participation and governance style that are challenging the traditional approach, also adopting change trajectories and thus following the ecosystem stewardship principles are highly needed. Good practices of more holistic management approaches need to be replicated, scaled up and embedded in governance to accelerate the transition towards ecosystem stewardship (see* Gorissen *et al.* (in progress)*).*

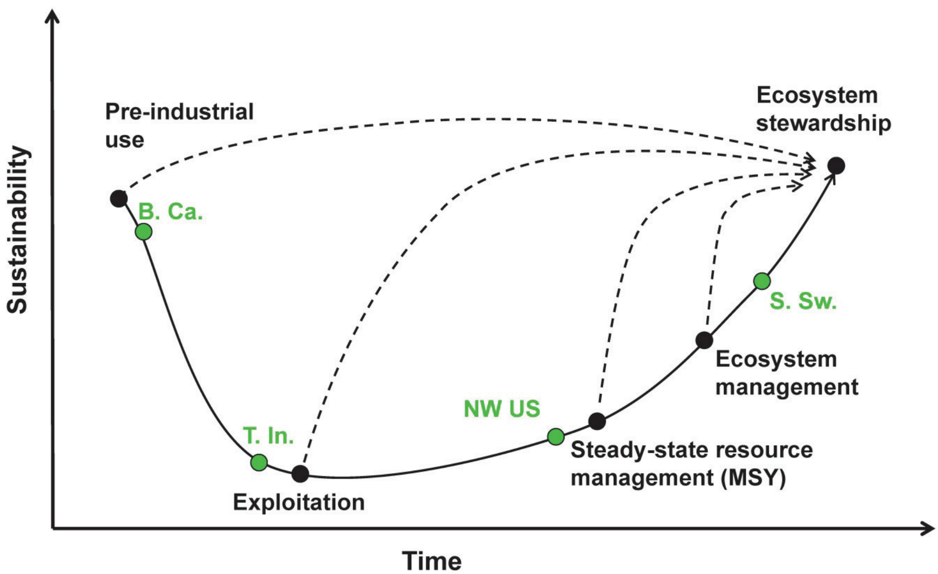

Figure 1.7: The temporal change of resource management regimes (black dots) observed in many Western nations. Green dots show selected locations: B. Ca.: Boreal Canada; T. In.: Tropical Indonesia; NW US: North-western USA; S. Sw.: Southern Sweden. Dashed arrows show opportunities for developing countries to accelerate the transition and evolve directly towards ecosystem stewardship (figure adapted from Chapin *et al.* (2010)*).*

1.5. Objectives and outline of this thesis

We are facing systemic challenges that cannot be resolved by simple interventions or optimisation of the current system. This requires also a new kind of research that promotes systemic thinking to effectively overcome silo thinking and compartmentalisation. Therefore, in this thesis we set out to combine several research approaches, including empirical, exploratory and solution-oriented approaches, linking basic research to applied research. By doing so, we hope to promote a more integrative, generalistic perspective that is valuable and actionable in terms of implementation for practitioners on the field.

The main goal of this thesis is to understand how forest and nature managers are to answer the simultaneously arising challenges of a strong demand for woody biomass, a need for biodiversity conservation and the provision of a multitude of ecosystem services while adhering to new models of participation, governance and management. In the **introduction** we sketched these grand challenges and their influence on forest management. We identified several knowledge gaps and a need for integrated examples of research and practice that couple an ecosystem based approach with innovative socio-economic aspects. To study some of the more concrete knowledge gaps in ecosystem management we adopted a case study area (Bosland) pioneering an innovative management approach and that faces the challenges listed above. Within this case study area we performed different interrelated work packages that are described in the next chapters (Figure 1.8).

In **chapter 2** we describe the case study area, called Bosland and located in north-western Belgium. We describe the history of forest management within the project, by studying policy and management documents and by interviewing key stakeholders. As the project uses innovative methods of participation and governance we conducted an analysis based on the transition management theory to find out if: (i) in what ways the Bosland approach differs from the classical forest management regime as observed in most other forests; (ii) which governance strategies, methods and instruments were successful in Bosland and how can these be scaled up and replicated to accelerate the transition towards ecosystem stewardship.

Bosland primarily consists of monoculture pine stands on nutrient poor, sandy soils. In Flanders and neighbouring temperate regions, pine stands make up a very large part of the forests and are thus a very relevant study system (e.g., 39% in Flanders (Waterinckx & Roelandt, 2001), 33% in the Netherlands (Dirkse *et al.*, 2007)). The pine trees in Bosland were planted on heathland to produce wood that was used in the coal mine industry. However, after the closure of the mines, different functions of the forest became more prominent, such as biodiversity conservation, recreational value and several regulating ecosystem services such as air and water purification and carbon sequestration. Currently, the transition to a more sustainable society with renewable sources of energy and material triggers the development of a bio-based economy. In the same forests, that were planted originally to sustain a fossil industry, the demand for woody biomass is thus currently rising once again. However, this rising demand should be fit in the current forest management transition aimed at converting monotone pine plantations towards more diverse and multifunctional ecosystems. The challenges in forest management are thus not only limited to the social aspects (chapter 2), but also to the biophysical components aspects to deliver sustainable biomass in harmony with the other forest ecosystem services (chapter 3, 4 & 5).

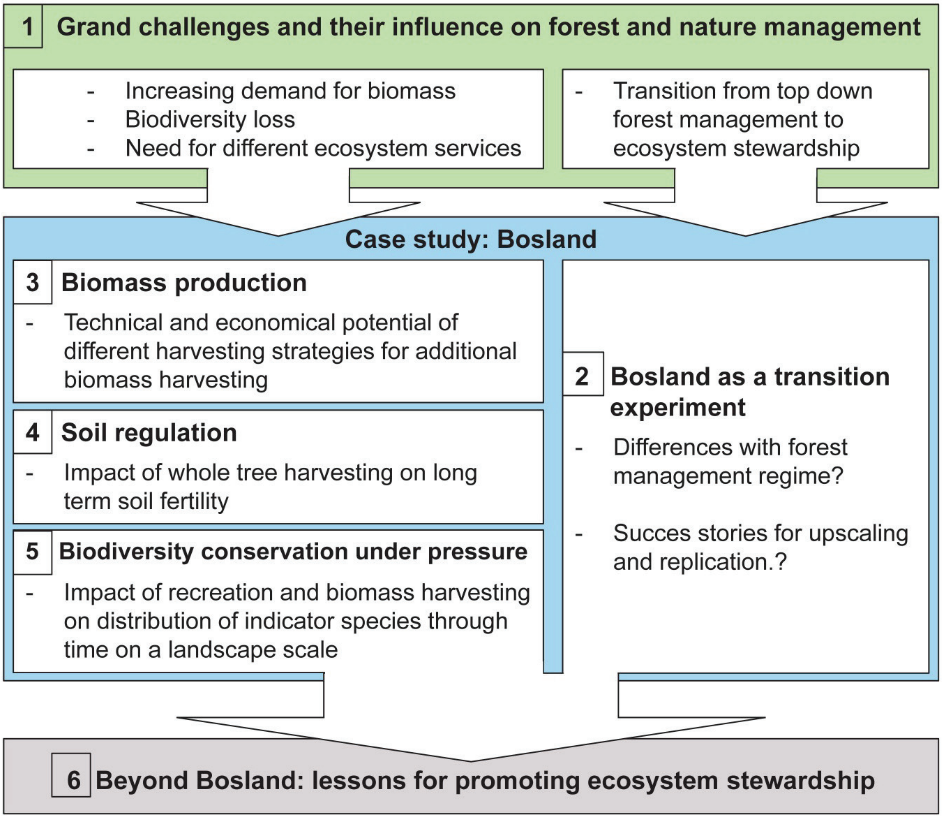

Figure 1.8: Lay-out of the thesis, illustrating the relation between the different chapters.

Whole tree harvesting is a common strategy to increase biomass harvest from forests in Scandinavia, knowledge in our region however remains limited. In **chapter 3** we investigate the techno-economical potential of additional biomass harvest for pine stands in North-Western Europe. We compare different harvesting strategies for whole tree harvesting in clear-cuts and early thinnings in pine stands in Bosland. The main research questions of this chapter are: (i) what harvest strategy is the most suited for harvest of additional biomass in thinnings and clear-cuts; (ii) what are the costs and earnings of different strategies for additional biomass harvesting.

In **chapter 4** we take a closer look at the impact of whole tree harvesting on soil fertility under an intensive management scenario in the same pine stands as in chapter 3. A transition to a sustainable bio-based economy asks for harvest regimes that are able to safeguard soil fertility on the long term. The soil nutrient concentrations of different ecosystem pools were measured before and after harvest and the long term nutrient stocks were modelled. We meant to find out (i) if the sandy soils were able to sustain an intensive management regime with whole tree harvesting; (ii) what the long term differences in nutrient stock would be if whole tree harvesting was applied instead of stem only harvesting.

In **chapter 5** we focus on the importance of Bosland for biodiversity conservation and we study the impact of recreation and wood harvest on different indicator species through time on a landscape scale. A forest is more than a biomass production plant and we meant to find out which place wood provisioning can take in the socio-ecological system. We mapped the distribution of the indicator species and analysed the impact of current pressure by recreation and wood harvesting to predict how species will react on varying future scenarios. The main research questions are: (i) what is the impact of recreation of different species and how can this be integrated in forest management; (ii) is it possible to spatio-temporally optimise wood harvesting to sustain populations of the studied species; (iii) if trade-offs are found, is it possible to integrate these three management objectives in one forest and at what cost.

Finally, in **chapter 6,** we first draw some conclusions for the further management of Bosland in relation to biomass harvesting, ecosystem service provision and biodiversity conservation. Next we evaluate to what extent the knowledge we gathered in Bosland is applicable outside the area. Given the universality of the challenges we studied, we are able to formulate some recommendations for managers of other forest and nature areas and for policy makers. We also looked into the socio-economical innovations of the Bosland project and its role in the ongoing transition in natural resource management. We evaluate different aspects of the current management styles applied and give recommendations for steering the most successful methods into the forest management regime throughout north-western Europe.

2. Towards co-ownership in forest management: analysis of a pioneering case 'Bosland' (Flanders, Belgium) through transition lenses

After: Vangansbeke, P., Gorissen, L., Nevens, F., Verheyen, K. 2015. Towards co-ownership in forest management: Analysis of a pioneering case Bosland (Flanders, Belgium) through transition lenses. Forest Policy and Economics 50: 98-109. IF 2015: 1.856.

2.1. Abstract

Forest management in Western-Europe is evolving towards multifunctionality and higher levels of sustainability. Co-owned forest managing models, where different owners collaborate and forest users participate however, are still rather an exception than a rule. Bosland (literally forest-land) in Flanders (Belgium) is a statutory partnership of several public forest owners and stakeholders, managing an area of about 22000 ha of previously fragmented forest relicts. By looking at this case through transition lenses we describe a pioneering case in forest management where a new way of management is adopted more geared toward management for coherence across multiple ecosystem services and across a multitude of stakeholders. By use of a learning history we were able to reconstruct the change trajectory of Bosland. Analysis of this change trajectory through transition lenses aided to identify essential key features in which Bosland differs from 'management as usual' approaches:

(i) a distinctive paradigm shift towards management for coherence;
(ii) a long term vision that informs and guides the short-term action agenda;
(iii) a bottom up approach focusing on participation and co-creation.

The methods used and lessons learnt in Bosland can thus be highly interesting for the wider community involved in forest and nature management.

2.2. Introduction

Belgium is one of the most densely populated countries in Europe, with a population density of 364.3 inhabitants per square kilometre (Eurostat, 2011b) and it has a relatively low forest cover of 23 % (Eurostat, 2011a) compared to the European average of 111.92 p/km² and 47% respectively. The European Environment Agency assessed the country on its land use and recommended that: ‘Belgium must manage land use carefully in the future. The challenge is on the one hand to allow for the development of social and economic activities based on land, and on the other hand, to protect the integrity of natural resource systems and the output of ecosystem goods and services which can also bring economic and social benefits in the long term.’ (EEA, 2010). This advice seems especially legitimate for Flanders, the northern part of Belgium, where forested land is scarce and severely fragmented. With a forest land cover of less than 11% (Van Herzele, 2006; INBO, 2012), the forest surface per capita of the region is smaller than any country in Europe (Eurostat, 2011a). The remaining forest relicts are of value in multiple ways, as they provide several ecosystem services, such as natural habitats for biodiversity, green refuges and open spaces for recreation, flood regulation, purification of water and air, carbon sequestration and provision of wood and biomass (Hermy *et al.*, 2008; Liekens *et al.*, 2013).

Effectively and coherently deploying the diverse forest-related services involves a wide range of societal actors and thus requires a land management style that is fit to deal with complexity and participation of stakeholders. In that perspective, the ‘established’ forest management approaches are not well-equipped to deal with these issues in the most effective way. More recently, several tools have been developed that allow forest management (planning) that unites multiple services (Pukkala & Kangas, 1993; Pukkala & Miina, 1997; Wolfslehner *et al.*, 2005). Implementation however lags behind, especially in cases where a broad variety of stakeholders is involved. In addition, land management and planning approaches should go beyond management of one ecosystem and collaborate on a landscape scale, especially in highly urbanised regions such as Flanders. To evolve towards a new kind of multifunctional and actor supported forest management, an approach appropriate to unite the diversity of potential values, services and stakeholders desires or claims needs to be enrolled.

This switch is quite a challenge for Flanders, because of the current largely disintegrated forest ownership and management: 70% of forest is divided among more than 100.000 private owners (Serbruyns & Luyssaert, 2006). The Flemish government is encouraging cooperation by stimulating private forest owners to unite in forest groups (Van Gossum *et al.*, 2005), organized as subsidized non-profit organizations. Despite these good intentions, co-owned forests supporting multiple purpose management remain scarce in Flanders indicating that ‘traditional’ top-down policy instruments are not well-suited to achieve that very objective (Van Gossum *et al.*, 2005; 2012).

The current challenges in forest management call for a new approach that actively includes stakeholders in the decision making process by combining bottom up and top down methods. In other words, a change process with a specified direction targeting the culture, structure and practice components of society concurrently. An approach that addresses such kind of challenges is the one of transitions and transition management (Grin et al., 2010). A transition is defined as “a radical, structural change of a societal (sub)system that is the result of a co-evolution of economic, cultural, technological, ecological and institutional developments at different scale levels” (Kemp et al., 2001). A number of anticipated transitions regarding energy, resources, biodiversity, etc. will require new practices, institutions and policy frameworks to deal with the limited space in a smarter and more sustainable manner. In this chapter we reconstruct the change trajectory of ‘Bosland’ using a learning history like approach. Subsequently we examine the history of Bosland by the semantics of transition theory to support identification of innovative aspects and key features that go beyond innovation as usual and which may be of inspiration for a wider public involved in forest management.

2.3. Material and methods

2.3.1. Bosland

Bosland (51.17°N 5.34°E) covers the area of three municipalities (Hechtel-Eksel, Overpelt and Lommel) in the North-West of the Limburg province (Figure 2.1). Currently the project is managed by a partnership of the four different owners (the three municipalities and the Agency for Forest and Nature Management of the Flemish region (Agentschap voor Natuur en Bos, ANB)) and two non-profit organizations (Regionaal Landschap Lage Kempen, a local organization for landscape conservation and Tourisme Limburg , a regional organization promoting tourism). Both non-profit organizations work independently, but the local and regional government respectively has a member in the board of directors, so both can be considered as public-private organizations.

Bosland lies on the border of the Campine plateau and almost all soils are characteristically sandy and poor. Until the middle of the 19th century, Bosland was mainly covered by an extensive heath land (Coordination cell Bosland, 2012). Gradually afforestation with conifers took place, *Pinus* *sylvestris* and *Pinus nigra* are the main tree species. Bosland has a total surface of 22 000 ha of which approximately 17 000 ha consists of non-constructed area (Coordination cell Bosland, 2012), containing almost 10 000 ha of nature- and forest area. Public forest covers more than 4500 ha and ownership is divided between the municipalities and the Flemish region. The Flemish region owns about 2260 ha, while the municipalities own about 1850 ha (Lommel), 630 ha (Hechtel-Eksel) and 40 ha (Overpelt). Privately owned forests account for approximately 2250 ha, of which approximately 180 owners with a total of 515 ha are member of the local forest group. Nature outside forests is mainly heathland and grassland and is owned by the Federal state (1497 ha, inaccessible military domain), Natuurpunt, a non-governmental organization on nature protection (356 ha in management) and the Flemish region (66 ha).

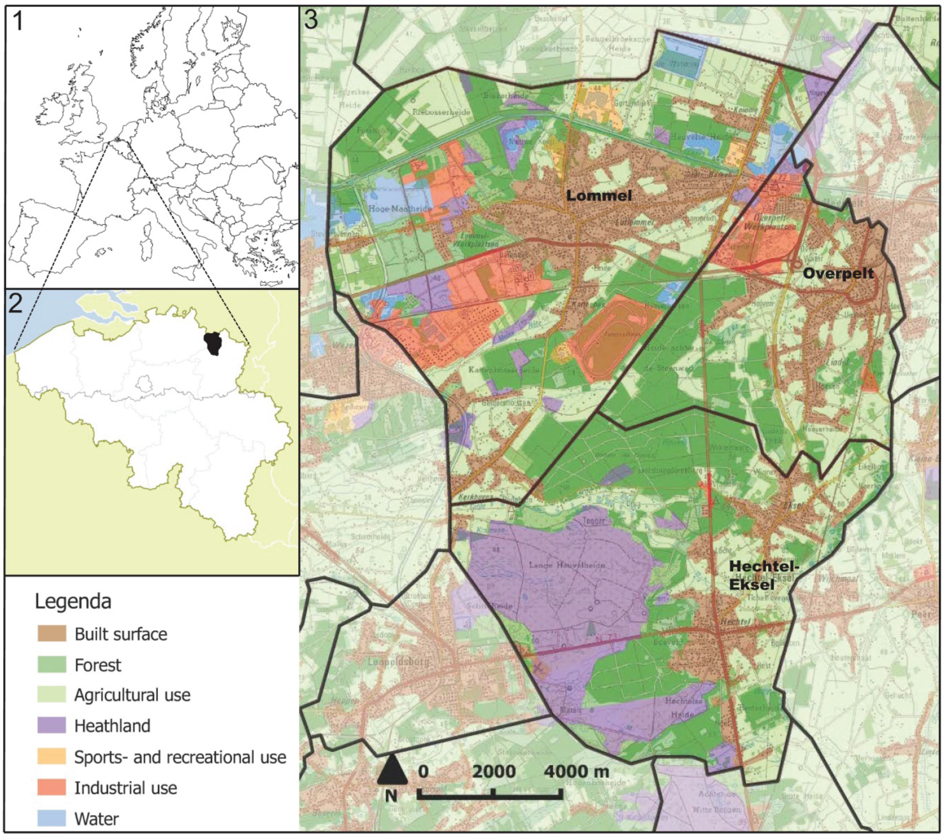

Figure 2.1: Situation of Belgium in Europe (left top) and Bosland in Belgium (left middle) and a land-cover map of Bosland (right). The heart of Bosland exists of forests that used to be managed by the different owners and are now managed together.

2.3.2. Learning history

In order to reconstruct the change trajectory that preceded the realization of Bosland, we adopted a learning history approach that we tailored to our specific objective since learning in transition trajectories exceeds the level of an individual organization. Instead it focuses on changes in the wider system: i.e. changes in the collaboration between organization and across networks, the prerequisites for this to happen and how a multitude of stakeholders is involved. The traditional approach of a learning history is to help organizations to learn from their own change and innovation processes (Kleiner & Roth, 1996). Currently, learning histories are also used in policy, for example to evaluate transitions (Willems *et al.*, 2009). Typically three levels of information are used to construct a learning history. In a first step the facts are listed. Secondly main stakeholders are asked to tell their account and give their opinion on the listed facts. In the final step a deeper analysis is made by an external researcher, combining the information of the first two levels (Kleiner & Roth, 1996). Since our focus is on the wider system, we adjusted the learning history approach for our reconstruction to include the following steps (see also Roelofs (2011)):

1. Focus determination

Elaborate discussion of the case with a forest expert and the initiator. This gave us insight in the stakeholders involved and the relevant documents to study.

2. Document analysis

Analysis of all relevant documents. This allowed us to draw up a timeline of the change trajectory.

3. Interviews

Interviews with at least 1 representative of the key stakeholders. This allowed us to include also the perceptions of the stakeholders.

4. Analysis using transition lenses

Analysis of the learning history outcome through the adoption of transition lenses. This allowed us to identify the features where the change trajectory delineated from innovation as usual trajectories.

For the documents analysis we collected all the policy documents related to the project: the first long term vision documents (Indeherberg *et al.*, 2006; Andriessen *et al.*, 2007), the extensive management plans (Gorissen, 2006; ABO NV, 2010; Econnection, 2012) and the Bosland master plan (Coordination cell Bosland, 2012). During the creation of these documents several participative processes had taken place: a survey of 200 forest visitors, discussion walks (8) and workshops (10) with all stakeholders, feedback sessions in the municipalities (3) and informative walks for the extensive management plans; brainstorming sessions (4) for the master plan. Reports of these events were available and have been reviewed as well. From these documents, we distilled a timeline that represents the important steps in the change trajectory that preceded the realization of Bosland.

Step 1 and 2 allowed us to identify key individuals from each important stakeholder group with whom semi-structured interviews were conducted. Interviewees were all closely related to the project, from different types of involved parties (public vs private; municipalities vs Flemish region; profit vs non-profit) and from different 'levels of action' (political; administrative; management). Each interviewee was also asked which other person(s) from other organisations they would suggest for us to interview to validate whether our selection was appropriate. According to the methodology of semi-structured interviewing, we determined key questions beforehand, but gave space and opportunity to the interviewee to bring up new issues. Roughly we asked all of the interviewees to report the history of the Bosland project, to indicate their role in the process and to point out which factors they experienced as facilitating/opposing the transition (the interview guide can be found in Appendix §1.1). All interviews were conducted in 2012 and lasted for about 45 minutes.

The interviewed stakeholders were the project leader from the governmental Agency for Nature and Forest (ANB) (1); the major of one of the municipalities (2); the head of the public service department for environment of another municipality (3); the manager of the landscape conservation non-profit organization, "Regionaal Landschap Lage Kempen" (4); the regional co-coordinator of the non-profit organization for touristic promotion "Tourisme Limburg" (5); a wood purchaser of a major wood processor in the region (6); "Natuurpunt", a non-governmental organization on nature protection working in the municipalities (7). "Natuurpunt" was represented by the chairman and the treasurer of the Hechtel-Eksel branch and the two chairmen of the Noord-Limburg branch in a group interview. They wanted to be interviewed together and reached a consensus for every answer. For this reason, their input has been handled as one perception in the learning history.

All interviews were recorded and completely transcribed afterwards. Key messages and features returning in at least 2 of the interviews were retained and used for learning history. We will further on refer to the cited stakeholder with the corresponding number (X) (cf. Kern & Smith (2008)).The information collected in step 1-3 was then analysed through the lenses of transition theory (see §2.3.3). This aided the identification of novelties and important features in the change trajectory. Combined with a learning history like approach this allows us to represent the Bosland case in a manner that can be useful for the wider community involved in forest and nature management and inspire future change trajectories in forest management.

2.3.3. Transition theory

2.3.3.1. The essentials

Transition thinking originated in research focusing on socio-technical systems (Rip & Kemp, 1998; Geels, 2002; Hoogma *et al.*, 2002; Geels, 2004), reflexive modernization (Grin *et al.*, 2006), social practices and societal governance (Rotmans *et al.*, 2000; Loorbach, 2007). Transitions are radical shifts from one system to another, implying structural and systemic changes; they encompass co-evolutionary processes where interactions between societal subsystems influence the dynamics of individual subsystems (Grin *et al.*, 2010). Hence, transitions are complex processes that involve multiple actors and different fields and typically span a long time frame (in terms of multiple decades) (Martens & Rotmans, 2005; Raven *et al.*, 2010).

The transition framework has been developed to understand transitions, to solve persistent problems and to promote sustainable development. Persistent problems are complex problems, deeply entrenched in societal structures and difficult to manage given the diversity of actors and vested interest involved (Loorbach, 2007). The transition framework combines four ‘archetypical’ phases (Rotmans *et al.*, 2005) and three interacting levels (Geels, 2005)(Figure 2.2). During a predevelopment phase no visible changes occur, but a lot of experiments take place, actually preparing the transition by making drastically innovative systemic configurations work on a limited scale. During a subsequent take-off phase, the first societal changes gradually become more visible. Actual up scaling and out scaling are the core of the acceleration phase in which changes in different areas reinforce each other into a broader dynamic/momentum. Finally in the stabilization phase the societal change comes to a rest and the system is in a new but dynamic equilibrium (Martens & Rotmans, 2005).

The societal changes that transitions imply, only take place under certain, favorable circumstances with interactions of changes at three different scale-levels. The meso-level subsystem of society that is undergoing the transition is called the regime. A term that indicates elements of inertia and resistance to change, caused by typical elements such as (technological) lock-ins, standing (infra)structures, institutions, vested power relations, etc. The transition multi-level perspective assumes that changes in the regime occur if supported by pressure inducing changes/events on the landscape- or macro-level and at the same time inspired by different successful experiments on the niche- or micro-level (Geels & Kemp, 2000; Geels, 2002). There are many definitions of the regime, but in general we can distinguish two different conceptualizations. The first is used to describe socio-technical systems (Nelson & Winter, 1977; Dosi, 1982; Rip & Kemp, 1998; Schot, 1998; Geels, 2002; Elzen *et al.*, 2004) and the second is used to describe societal systems (i.e. sectors or regional entities) (Rotmans *et al.*, 2005; Van Raak, 2006; Loorbach, 2007). These two schools of thought do not exclude each other, rather their differences are merely in focus and tradition. The landscape forms the societal background to the transition, it consists of social values, political cultures, environmental and economic trend; evolutions on which there is little or no possibility to 'interfere' on an individual basis. The niches are the micro-level of innovation, where, in an experimental and protected environment, shielded from regime pressure and change inertia, novelties are created, tested and diffused (Loorbach & Rotmans, 2010; Raven *et al.*, 2010).

Interaction of the three levels (co-evolution) is needed and niche emergence or development is one of the crucial steps in a transition. Moreover, the niche is the only level that can be steered by individual practitioners with the help of approaches/conceptual frameworks like transition management (TM) or strategic niche management (SNM) (Raven *et al.*, 2008). Thus a transition experiment in a niche can be one of the multiple starting points that can induce a transition (Raven *et al.*, 2010; van den Bosch, 2010).

SNM originated as a new policy perspective on how to modulate transition experiments and the emergence of niches with a high potential for sustainable development. According to SNM, it is possible to facilitate innovation journeys by executing experiments for the creation of technological niches: protected spaces that allow maturing of technologies through co-evolution with user practices and regulatory structures. SNM builds on three internal niche processes: (i) voicing and shaping of expectations and visions, (ii) building of social networks and (iii) an explicit learning process (Raven *et al.*, 2010).

TM is a governance mode that attempts to resolve persistent societal problems. It is an iterative process consisting of four steps: (i) problem structuring and organization of a transition arena; (ii) drafting a transition agenda, visioning and the identification of transition pathways; (iii) defining and performing transition experiments through mobilizing networks; (iv) monitoring, evaluating and lesson drawing, to be fed back in the other steps (Loorbach, 2007). TM focuses more on the regime actors, next to actors in the niche or experiments. Also within the regime some innovations occur that differ from the current culture. These so-called traditional innovation experiments can cause gradual change within the regime, but they will not cause the structural changes as observed under a transition (Grin *et al.*, 2010). Traditional innovation experiments predominantly focus on incremental change and depend on self-referential systems that promote path dependency which tend to reproduce already existing systems and worldviews (cf. Unruh (2000)). Transition experiments on the other hands are focused on radical innovation that supports system innovation, which also includes critically scrutinizing existing structures and institutions which are often not questioned in more traditional innovation experiments (Grin *et al.,* 2010)

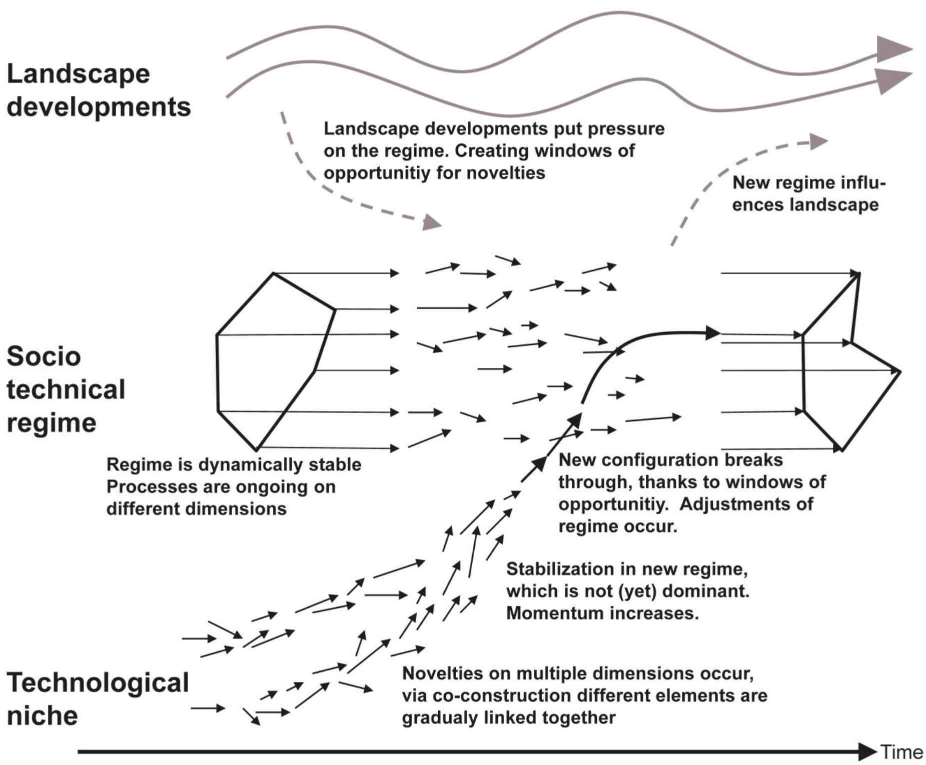

Figure 2.2: The multilevel transition framework, with an added time component. Pressure from the landscape opens windows of opportunities and innovations in niches can influence the regime (after Geels (2005)*). See §2.3.3 for further explanations.*

A transition experiment mainly differs from a traditional innovation-experiment by the goal, the time frame, the methods, the context and the learning process; where the former is socially broader, systemic, long-term oriented and characterized by a different learning process of the actors (Table 2.1) (Raven *et al.*, 2008). In transition experiments, the learning process is more elaborate, including involvement of multiple, divergent fields, social learning (i.e. actors learning from each other through interactive processes about values, norms and goals (Kemp & Weehuizen, 2005) and double loop learning (i.e. learning questioning the fundamental design, goals and activities)(Argyris, 1976; Kemp & Weehuizen, 2005). Generally, learning in traditional innovation-experiments is more limited to individual learning, to a few fields and to single loop learning (i.e. more technical learning about the use of certain means and instruments within the actual framework (Kemp & Weehuizen, 2005)).

Table 2.1: : Overview of the characteristics that allow to distinguish a transition experiment from a traditional innovation-experiment (translated from Raven *et al.* (2008)*)*

	Traditional innovation-experiment	**Transition experiment**
Starting point	Learning related to solutions (marketing of innovation)	Learning related to social issues
Problem features	Prescribed and simple	Complex and uncertain
Goal	Identification of optimal solution	Contribution to societal change
Time frame	Short and middle term	Long term
Methods	Testing, modifying, demonstrating	Exploring, searching, learning
Learning process	Single-loop, individually, a few fields	Double-loop, social, multiple fields
Actors	Specialized staff (researchers/engineers)	Socially complete alliance
Experimenting context	Controlled context (lab/simulation/ testing environment)	Social context
Management context	Traditional project management Command and control	Transition management; Strategic Niche Management Influencing, steering, facilitating

2.3.3.2. The transition perspective and forest management

Our society today is facing many sustainability problems. According to Rockstrom *et al.* (2009), we already crossed the sustainable boundaries of our planet for at least three processes: climate change, the nitrogen cycle and biodiversity loss. What is also worrying is that these sustainability problems are often interlinked and influencing each other (Rockstrom *et al.*, 2009). Many of these sustainability problems are grounded in land use. Biodiversity loss for example, is mainly driven by land use changes, such as the conversion of natural ecosystems into agriculture and urban areas (Sala *et al.*, 2000). Given the scale and scope of the challenges in land use today, traditional sectorial thinking is inappropriate to dealing with these systemic problems. The transition (management) approach is strongly focussing on integrated persistent problems (Loorbach & Rotmans, 2010) and has been proposed to steer the needed changes in land use planning (UNEP, 2014).

Forest management in Europe has shifted in recent decades towards multifunctionality (Puettmann *et al.*, 2009). The felling rate has increased from 58% to 62% of increment in Europe between 1990 and 2010 and in the meantime forest management practices increasingly include biodiversity protection (Forest Europe, 2011). Already more than one fifth of European forests are managed primarily to protect water, soil and infrastructure (MCPFE, 2007). The idea that a forest should be managed as a complex adaptive system is gradually getting wider accepted (Puettmann *et al.*, 2009). The ecosystem services concept (MEA, 2005; TEEB, 2011) has strengthened this idea. The ecosystem services framework adopts a more holistic landscape view in which the interconnections between services and with other land-uses are made more explicit. Moreover, the 2011 TEEB-study helped to draw attention to all different services provided by forest and nature areas and emphasizes the importance of forest and nature on society and vice versa.

This growing perception that ecosystems and societies are interdependent, forming complex social-ecological systems, has promoted the idea that stakeholder participation is a necessity in ecosystem management (Schultz *et al.*, 2011). It was stated that results of management and assessment of social-ecological systems are improved when the full range of stakeholders is involved (Walker *et al.*, 2002). Sometimes, critique against this vision have been put forward, arguing that involving all stakeholders could for example slow-down decision making or decrease ecosystem management efficiency by hindering the application of scientific knowledge (du Toit *et al.*, 2004). However, most studies that have empirically tested the impact of stakeholder participation on ecosystem management show a positive relationship (Brody, 2003; Lebel *et al.*, 2006; Schultz *et al.*, 2011). So it is broadly accepted that involvement of stakeholders throughout the management process is a good way to increase local support (TEEB, 2011), legitimacy (Treffny & Beilin, 2011) and societal learning (Borowski *et al.*, 2008; Garmendia & Stagl, 2010). However, well-functioning coordination mechanisms between different levels of government and stakeholder groups are still rare (MCPFE, 2007).

Traditionally forest and nature areas have been managed with an expert-driven top-down approach with little attention for broad, local stakeholder input (For an example from Germany, see Maier *et al.* (2014)). Recent developments are more oriented towards involvement of stakeholders, but are not always perceived as very successful (Maier *et al.*, 2014). Also in Flanders there are examples of a trend towards more collaboration and stakeholder input. For instance, all public forest owners (and some private forest owners, depending on the spatial planning) with a forest larger than 5 ha need to elaborate an “extensive forest management plan” (Flemish Community, 2003). For this plan, forest owners have to make an extensive inventory of their forest (both on a dendrometical and on an ecological basis), to start up a social participation project to involve all forest stakeholders and to make a projection of future management measures in function of the current situation and the stakeholders view. The costs involved in making up the management plan are largely paid back by means of a subsidy of €200 per ha. In this way forest owners are forced to consult stakeholders. Moreover collaboration between forest owners was stimulated with an added subsidy of €20 (for more than 3 collaborating forest owners) or €50 (for more than 10 collaborating forest owners) per ha (Flemish Community, 2003).

However, knowledge, perceptions and viewpoints vary greatly among societal stakeholders and forest owners, reflecting the tension between different interests (Van Gossum *et al.*, 2011). In addition, the perceptions co-evolve with the modernization of the social structure of private forest owners (Ziegenspeck *et al.*, 2004). Gradually private forest ownership is changing from the typical agricultural forest owners to people living in cities and shifting the focus towards enjoyment and utilization of timber for own needs, the so called ‘non-agricultural forest owners’ (Kvarda, 2004). Van Gossum *et al.* (2011) classified Flemish forest owners (public and private) according to their perception towards sustainable forest management and differed between a private property coalition, an economic coalition, a local use coalition, a sustainable forest management coalition and a nature coalition. The introduction of sustainable forest management and collaboration on a landscape scale is thus still hampered due to the differences in viewpoints between the forest owners (Van Gossum *et al.*, 2011).

Up till now, participation processes in forest management are predominantly information and consultation processes, described as one of the lower types of participation (Arnstein, 1969; Edelenbos & Monnikenhof, 2001). Informing stakeholders occurs only after decisions have been made, offering no chance to the public to influence the agenda or to express their viewpoints (Edelenbos & Monnikenhof, 2001). Consultation allows stakeholders to present their opinion, but still only at the end of a development process (e.g. policy making) and in most cases this does not includes active support of stakeholders, cross sectorial collaboration, empowerment and ownership (Edelenbos & Monnikenhof, 2001). Higher types of participation such as advising, co-creation and self-management involve stakeholders from the beginning of a process and deliver more in terms of legitimacy and social learning (Edelenbos & Monnikenhof, 2001).

Forest managers are thus faced with enacting a transition from a rather monofunctional, expert-driven, and science-based system to a more inclusive and socially responsive model of decision-making (Beckley *et al.*, 2005). To achieve this, well-functioning coordination mechanisms between different levels of government and stakeholder groups, which require shifting mindsets of forest managers, may prove to be necessary. To study which features have played an important role in the development of the co-owned Bosland forest, we analysed the history of Bosland through the lenses of transition theory, since this framework is especially well suited to study transition trajectories and to identify which features are going beyond traditional innovation approaches.

2.4. Results

2.4.1. The change trajectory of Bosland

The history of Bosland is presented in Table 2.2 and the most important steps are summarised in Figure 2.3. Our analysis shows that several elements highlighted in transition approaches (Grin *et al.*, 2010; Loorbach & Rotmans, 2010; Nevens *et al.*, 2013) are present in the Bosland case: problem structuring or system analysis, envisioning, transition pathways or scenario development, experimenting and anchoring.

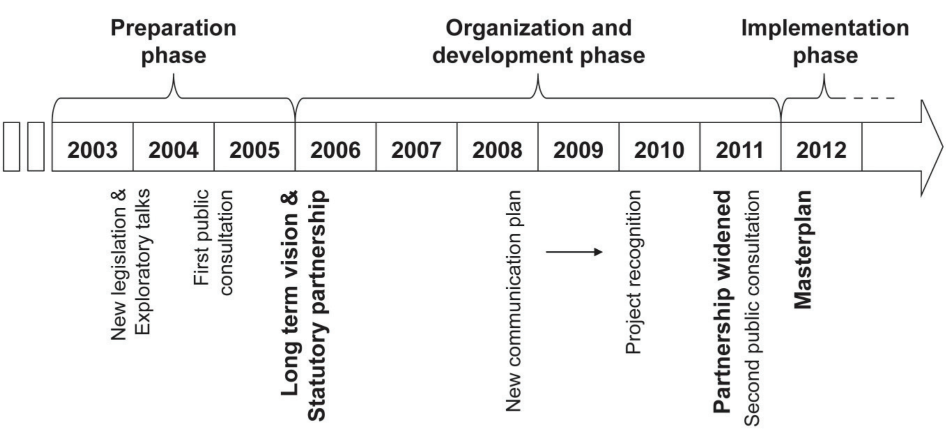

Figure 2.3: Flowchart on the formation of Bosland. The development phases, as defined by the partners, are marked above the timeline, events below; the most important events are marked in bold. See Table 2.2 for more information about the events.

Table 2.2: Learning history about the change trajectory of Bosland, combining facts from policy documents in column 1, the story of Bosland through semi-structured interviews of key stakeholders (numbers between brackets refer to different interviewees, see §2.3.2) sporadically complemented with info from the policy documents in column 2 and a transition analysis of the change trajectory of Bosland in column 3.

Bosland timeline	The story of Bosland by the interviewees	Transition analysis of change trajectory of Bosland
- **New legislation (2003) leads to exploratory talks (2004)** In 2003 a new legislation on forest management was launched. From now on an extensive management plan had to be made up for all public forests (Flemish Community, 2003). In 2004 exploratory talks between the two municipalities, the town and ANB, started.	- The extensive management plan required by the new legislation entailed a significant increase in workload. This fact triggered the governmental Agency for Nature and Forest (ANB) to rethink the traditional top down approach in favour of a more collaborative approach with other forest owners. That way, a better integrated management plan could be developed across different stakeholders while increasing local support for the plan. (1) - This idea was first discussed between ANB and the Hechtel-Eksel municipality. Not only a division of the workload was possible but also the idea came up to co-develop a long term vision for the area on a landscape scale (1,2) - Soon, the city of Lommel joined the discussion (1,2,3). - At first there was a kind of disbelief at the level of some municipalities (1,2). - Previously, a unidirectional collaboration between the municipalities and ANB was common practice. ANB, as forest management experts, imposed rules and legislations about forest management on the municipalities (1,2). Participation of the municipalities was restricted to an information and consultation level. This top-down approach resulted in often disillusioned municipalities: the feeling existed that they couldn't decide what to do in their own forests and that ANB had always an authoritarian finger raised in warning (1,2,5). - But existing prejudices were put aside and the collaboration was started (1,2).	- The new legislation induced a **process of reflexivity** within certain actors from ANB allowing them to rethink their current practices and strategies. This opened up space for the emergence of new solutions and new roles to be adopted: this first phase indicated the first steps of letting go of the traditional management style focussed on 'demand and control' towards a new style more geared towards '**facilitation of collaboration'**. - At first this was met with scepticism from the approached partners since they were not used to this new role of ANB. - **Dialogues** between partners 1-3 and the possibility of co-creating a long term vision between the different partners gradually built trust and the collaboration was initiated. ➔ This coincides with what is described as a first phase in transition management processes: setting the scene and exploring transition dynamics. It is also in line with the notion that (mental, financial, temporal) space is an essential feature to rethink common practice.
- **First public**	- The partners agreed that coordination and cooperation needed to be	- Together the partnership organised an elaborate

consultation (2004-2005): A survey of 200 forest visitors was made. Eight discussion walks, ten discussion workshops and three feedback sessions in the municipalities with all stakeholders were organized. This information was complemented by many forest inventory and biodiversity monitoring studies.	underpinned by a long term vision for forest management. To achieve this, an intensive consultation process of the local public was set up (1,2,3). - During the first envisioning process several events took place to involve local people (1,2,7). The resulting vision was brought back to the broad stakeholder groups by means of discussion sessions and informative tours in the forest where people could give feedback. - Also in the elaboration of the vision in management plans, the view of the public was crucial. For each of the five management plans, a public session was organized in which the vision was presented and a translation in tangible measures was proposed (Gorissen, 2006; ABO NV, 2010; Econnection, 2012). - Many reactions from the public came, during and after these sessions and they were discussed and included in the management plans or refused. In the latter case a justification was provided (1,2).	participatory trajectory to enrol an **envisioning exercise** across a multitude of stakeholders. - The outcomes of this exercise was **instrumental to** inform the **management plans** and different discussion rounds have been organised to fine-tune these while creating support for and ownership of the vision and management plan. ➔ This coincides with the phase of 'envisioning' in transition while the translation of the vision into management plans is what is referred to as 'anchoring' in transition literature (Nevens *et al*. 2013). This means that new roles and new activities are anchored into the regime.
- **First long term vision developed and statutory partnership founded (2006)** The first long term vision documents were made (Indeherberg *et al.*, 2006; Andriessen *et al.*, 2007) and based on this the extensive management plans were developed (Gorissen, 2006; ABO NV, 2010; Econnection, 2012). The statutory partnership was officially founded in 2006.	- The insights were bundled into a long term vision which lay the foundation of the partnership with the four public partners in 2006. In these vision documents the different forest functions (ecological, economical and recreational functions) were described and an equilibrium between the functions was presented on a landscape scale. Where possible, different functions were combined in a certain forest patch, but often the necessary choice for a certain prioritized function was made (Indeherberg *et al.*, 2006; Andriessen *et al.*, 2007). The vision documents consisted of a description of the projected forest functions for every forest patch and sketch maps of the most important ecological, economical and recreational 'network infrastructure' on a landscape scale. - For the first time forest managers of ANB seemed to acknowledge that a forest is more than an isolated area of trees and nature (1,2,5) and other forest users were actively thinking about preserving the common values of the forest, such as certain animal or plant species that need protection (1,2). Moreover, the focus widened from "trees-management" to "landscape-management" (2,3).	- The joint envisioning process induced a **shift** from managing forest patches to managing on a landscape scale. Such paradigm shifts are necessary for the enactment of sustainable development. Rotmans (2013) writes 'A persistent problem is that we tend to try to innovate public systems starting from old values and old paradigms. These attempts are doomed to fail'. - The envisioning exercise gave rise to a **long term vision** co-created by a multitude of stakeholders. This led to a vision that superseded the traditional forestry view and incorporated also functions deemed important by the general public and other stakeholders. In smaller forest patches different functions may compete (trade-offs) but on a landscape scale, these different functions can be developed or conserved next to

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- Widening the focus was not always easy to cope with in the beginning (1,3).

- The involved Bosland actors believed that a co-owned forest would increase legitimacy and active support and would also offer chances on a management perspective to integrate management goals on a landscape scale (1,2,3,7).

- This long term vision was instrumental for setting up the action agenda and drawing up interim objectives, captured in subsequent Bosland forest management plans (Gorissen, 2006; ABO NV, 2010; Econnection, 2012).

- To concretize the vision, extended sectorial long term visions were developed for tourism and wood production. On the basis of an extensive inventory of standing stock and an empirical growth model, a prognosis was made of wood stocks and harvest between 2010 and 2070 under different management scenario's (Moonen et al., 2011). The results under different scenarios were then discussed with relevant stakeholders and after an intense voting procedure, a consensus was found and a long term scenario for management for wood provision was selected. This long term vision is now reflected in the management plans that are implemented in the forest (Moonen et al., 2011)(1,2,6). This experimental approach of strategic forest management planning by long term scenarios was never used in Flemish forest management before (1,6).

- Some actors felt excluded from this partnership (4,7) and still feel that they were not able to put a foot down on how the outline would look like.
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each other.
➔ This process is also referred to as reframing in the problem structuring phase: by bringing together a diversity of views and aspirations, more holistic approaches can be developed. In transition literature, system thinking is often formulated as a requisite to overcome persistent problems (Nevens et al., 2013) while development of a shared, long term vision is defined as strategic transition management (Loorbach, 2007).

- The collaboration between the partners was made official by the foundation of a statutory partnership. This highlights the different role of ANB.
➔ In transition literature, forming new collaborations with unusual partners is often referred to as a catalyst to give rise to new (often more radical) solutions (Rotmans, 2013). This is what Loorbach defined as tactical transition management (Loorbach, 2007).

- Envisioning as a point of departure for setting interim objectives (backcasting) is characteristic for transition management (Grin et al., 2010), but it is only genuinely instrumental if it is actually coupled to effective strategy and action development (a short term action agenda).
➔ This refers to operational transition management (Loorbach, 2007).

- Even though participation was central to the approach, not all stakeholders were involved from the beginning. This coincides with the
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		selective participation approach in transition management where in the beginning a select group of people is brought together: not representatives but open-minded, visionary individuals that are able to look beyond their stake (Loorbach & Rotmans, 2010). ➔ A transition management approach aims to give rise to a new discourse with higher ambition level that is fuelled by an appealing long term vision
- **A new communication plan (2008) resulted in project recognition (2010)** In 2008, a new communication plan was launched, including a new project name: "Bosland". In 2010 the project received a funding of 2,13 million euro of the "Limburg Sterk Merk" program, supporting projects that promote a balanced development of the province of Limburg.	- The name 'Bosland' was chosen since it captured the essence of the long term vision and the project wanted the forest to be the homeland of all forest users (1). - The project gained momentum in 2010 when it was recognized as a strategic project in the framework of spatial planning by the responsible minister of the Flemish government. This recognition was essential in securing political support for 'Bosland' and in securing financial support for further enrolment (1,2). The project office was started and the general vision was translated to management plans. - The interviewees linked to a partner organisation (1-5) confirmed a feeling of equal standing and co-ownership.	- To reflect the paradigm shift and truly **anchor** the partnership a new name was chosen to strengthen the bonds but also give a 'face' (recognition) to the project that has a more inclusive connotation. ➔ This is in line with the observation that developing a new language is important in transition processes. - By facilitating the change trajectory, ANB adopted a **new role** that better marries a top down and bottom up approach. Recognition of the project as a strategic project by the Flemish Government and funding allow further development of Bosland.
- **Partnership widened (2011)** In 2011 two non-profit partners joined the project, "Regionaal Landschap Lage Kempen" and "Tourisme Limburg".	- The two new partners are not forest land owners in the strict sense, but allowing them as co-owners in the project resulted in additional specific expertise and in new dynamics, such as new touristic brochures and a new project website (1, 2) - The focus slightly broadened and with six partners it became more complex to reach consensus (3,5).	- Since the vision focussed on multiple services of Bosland, two new non-profit partners joined the partnership. This probably increased **legitimacy** of the partnership. ➔ This reflects the cyclic character of change processes. A vision is not regarded as an end point but as a cyclic (and reflexive) continuation of thinking - acting - assessing - (re) thinking- acting- assessing … (Sondeijker *et*

		al., 2006; Nevens *et al.*, 2013).
- **Second public consultation (2011)** An independent innovation centre was asked to organize a new participatory process in preparation of the masterplan. The “Bosland parliament” was founded.	- Five strategic goals for Bosland were proposed by the partners and discussed and evaluated on four brainstorm sessions in preparation of the masterplan (1,2,3,4,7). - However, some interviewees argued that four two hours brainstorm sessions are insufficient to be called structural participation (4,7) and that invitation of participants was quite ad hoc and not per definition representative for all relevant stakes (7). - The biggest challenge faced in the participation process was to keep stakeholders involved in forest management after the planning phases (1,2). A need to institutionalize the different participatory groups originated and eventually resulted in “the Bosland parliament”, consisting of three equal “participative houses”. These houses were filled up in parallel with the pillars of sustainable forest management (Flemish Community, 2003): an ecological house, a social house and an economical house. This innovative approach for participation was established to allow stakeholders’ close and active involvement in the matters of Boland’s forest management. In the management structure of the project, the parliament is placed parallel to the different management bodies acting as a permanent sounding board (Figure 2.4). Whenever a concrete project is started, a working group is founded with members of the steering committee, the management committee and of the relevant house. ○ The economic house consists of people that are dependent on Bosland for their income. Two distinct groups can be distinguished: the wood sector and the tourism business. As this house consists mainly of professionals, meetings are not periodical, but the members only come together in directly relevant working groups (1,2,6). ○ The ecological house includes people interested in biodiversity in Bosland, it unites volunteers and foresters. They come together periodically and have played a major role in inventories and research (1,2,3,7). The local branches of “Natuurpunt” argue that it is unfair that they are not included	- To keep stakeholders involved in the further development of Bosland, an **innovative and experimental approach** was adopted in the form of the Bosland parliament. This approach works well for the more specialised houses: the economic and ecological house, where focus and stakes are quite apparent. More difficulty arises with the establishment of a social house where focus and stakes are less clear. The establishment of a Bosland Parliament is an example of social innovation and reflects the **governance dimension** of the change trajectory (Grin *et al.*, 2010). Hereby it contributes to **empowerment** of the various stakeholders. ➔ This aligns with the next phase of a transition trajectory: to set up transition experiments to learn about barriers and windows of opportunity for the new system setting (Grin *et al.*, 2010). ➔ The establishment of a Bosland Parliament and its recognition within the management structure highlights the issue of agency and structure within transition literature (Grin *et al.*, 2010) and the recognition that empowerment contributes to sustainability governance (Avelino, 2011).

	as a partner and only individuals are allowed in the ecological house. They feel neglected as an organization (7). ○ The implementation of the social house however, is a big challenge. Every forest user, local or visiting should be able to join the social house. A regular meeting is hard to organize, because it is impossible to invite all users and the house is kept alive as different instruments involving all interested users. For now, follow-up guided tours for every action in the forest, information sessions on large-scale public activities and communication through the municipalities, in the newsletter and via the website offer the possibility for everyone to contribute (1,2). - Critics highlight that the separation of the houses is contradictory to integrated forest management, where a balance should be found between economical, ecological and social values through direct dialogue (7). Despite the good intentions of most partners, saying that they want to involve all stakeholders and that they do so (1,2,5), criticism thus remains on the participatory approach (7). Some partners admit as well that there is still room for improvement on participation but that these processes are now often limited by time and budget (2,4).	
- **Masterplan launched (2012)** The final version of the master plan was presented and launched in June 2012 (Coordination cell Bosland, 2012).	- After the participatory process, five strategic goals were included and elaborated in the masterplan ○ Bosland as a producer of ecosystem goods and services ○ Bosland as a hotspot for biodiversity ○ Bosland as a participatory factor ○ Bosland as a touristic and recreational pole of attraction ○ Bosland as local socio-cultural heritage - These strategic goals form the basis of further concrete planning (Coordination cell Bosland, 2012) and are the start of what the partners name the implementation phase.	- The original vision has been updated with the information of the new partners and the outcomes of the participatory exercises has been **consolidated** through the second public consultation. The master plan **links the long term vision to the short and medium term action agenda** and **embeds** the outcomes of the change trajectory **institutionally**. ➔ This represents the last phase in transition management approaches: that of operational transition management (Loorbach, 2007).

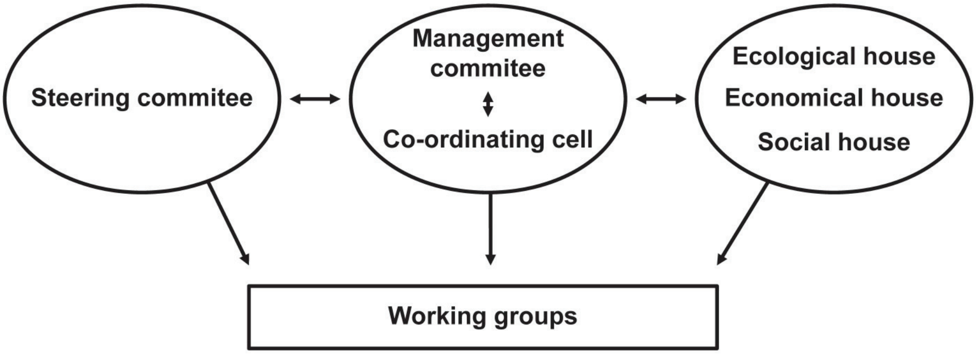

Figure 2.4: Management structure of the Bosland project.

2.4.2. The way ahead

According to the partners the development of the master plan introduced an implementation phase in which the co-produced long term vision is translated in concrete management actions and in which the collaboration and unity will also be made visible in the forest (1-5). Participation of the public by means of the Bosland parliament remains crucial in this implementation. All partners have an optimistic and confident eye on the future of the project (1-5), despite the limits on time and budget and some criticisms on the project. At the moment the focus lies thus on collaboration with the current partners and on concrete management actions in the field (2,3,4). In line with adaptive management there will however be future moments of project evaluation and renewed broadening of the focus. Perhaps in the future, the collaboration with other forest and nature owners could be expanded. For instance collaboration with “Natuurpunt”, a non-governmental organization on nature protection working in the municipalities (managing 356 ha in the area) and with the “forest group”, a public-private organisation supporting forest owners (uniting 515 ha of approximately 180 private forest owners) could be strengthened to sustain and increase legitimacy and carrying capacity of the project and to work towards the desired outcomes.

2.5. Discussion and conclusion

In recent years, forest management in Western Europe is transitioning towards multifunctionality, combining principles of traditional silviculture and ecology with complexity and adaptation (Puettmann *et al.*, 2009). Concerning participation and co-ownership principles, there is still a long way to go (Bruña-García & Marey-Pérez, 2014). Gradually however, the importance of public support for forest is settling in (Bruña-García & Marey-Pérez, 2014) and leading international studies (MEA, 2005; TEEB, 2011) make the link between citizens and forest (management) in both directions. To manage a forest coherently across a multitude of stakeholders and across multiple ecosystems requires a new management approach. It is especially in this regard that Bosland is an interesting case. The change trajectory towards Bosland gave rise to a new discourse with a higher ambition level inspired by a long term vision and fuelled by a new collaboration between different partners. In addition, it induced experimentation with new governance settings. The learning history approach allowed us to reconstruct the history of the development of Bosland. Analysing this change trajectory through transition lenses enabled us to structure the change process and to identify essential steps and innovative features that have been developed through a collective search and learning process of the new partnership and to relate these to the transition framework.

First, a distinctive feature of Bosland is that the traditional style and role of ANB changed from a modus of ‘command and control’ to a modus of ‘facilitation for co-creation and collaboration’ across different partners. This reflects a paradigm shift from fragmented management responsibilities (each partner manages own fragment of forest) to co-management for coherence on a landscape scale. To enable this shift, the traditional top-down approach gave way for a more bottom-up approach. From a transition perspective the following features are regarded as positive aspects in change trajectories: adoption of a facilitating style and role, co-creation of a shared vision through selective participation, initiation of new collaborations (Grin *et al.*, 2010) and these aspects were all present in the change trajectory preceding Bosland.

Second, to go beyond ‘innovation as usual’, a new discourse needs to be developed with a higher ambition level (Rotmans, 2013). In transition approaches, this is achieved by linking a shared and co-created long term vision to a short term action agenda (back casting) (Grin *et al.*, 2010). Our learning history showed that this was the case in Bosland and that the vision helped to unite the different stakeholders and to give direction to the management plans and the masterplan. In general, every short term action in Bosland is in alignment with the long term vision and the concurrent strategic headlines.

Third, Bosland was co-constructed by a multitude of actors by means of a considerable focus on participation of stakeholders and forest users. Furthermore, this participatory approach will continue to play an important role in the future management of Bosland by means of the forest parliament and houses. As the learning history showed, participation started on a small scale involving a selected set of participants and gradually broadened to include more stakeholders and forest uses. The establishment of the Bosland parliament in parallel with other management structures is an example par excellence of broad social network building and can be regarded as first steps towards a governance approach within forest management. The evolution within participation is also in line with what Loorbach & Rotmans (2010) define as selective participation. The difference with traditional approaches is that selective participation does not aim to reach a consensual vision that gains wide support (but usually also leads to suboptimal solutions). Instead it is aimed to gain a deeply shared and owned vision with a high ambition level in a select group of key participants that is later on widened to include more actors.

Fourth, from a governance point of view, three different types of activity and new roles have been distinguished and conceptualized as strategic, tactical and operational transition management in transition literature (Loorbach, 2007). If we look at the history of Bosland, we can recognize these iterative steps: building a long term vision aligns with strategic TM, the formation of a new collaboration and the establishment of the Bosland parliament aligns with tactical TM and the vision inspired masterplan of Bosland aligns with operational TM.

Taken together these features are closely aligned to what is described as the outcome of a successful transition process (Rotmans, 2013). Our results also illustrate that the change trajectory of Bosland goes beyond what is considered as traditional innovation (see Table 2.1). Many aspects show that Bosland reflects a transition trajectory illustrating more fundamental innovation features such as:

- The starting point of setting up a collaboration to deal with the issues of complexity are more focused on learning in terms of social issues than learning related to ‘restricted’ solutions of forest management;
- The time frame clearly focused on the long term and the long term was coupled to a short term action agenda;
- The change trajectory described illustrates the process as a joint search and learning process with a high degree of exploration;
- The role of ANB shifted towards a role more focused on facilitation and co-creation;
- Innovative governance settings were introduced (e.g. the establishment of the Bosland parliament) that are more socially inclusive.

Furthermore, the collective search and learning process was fundamental for building reflexive capacity which is a necessary precondition to support a long term process of sustainable development (Grin *et al.*, 2010). Such search and learning processes can also be described as multi-actor social learning processes which are an important feature of governance in transition literature (Grin *et al.*, 2010). Because of this innovative approach, we conclude that Bosland is a pioneering initiative, a frontrunner that put into practice a new way of forest management. This reconstruction and analysis of Bosland using novel frameworks to highlight the distinctive features might be of interest and of inspiration for the wider community involved in forest and nature management.

Of course, the change trajectory demonstrated in Bosland is still ongoing and is only a first step towards a possibly new mode of forest management. A more elaborate strategy (defined as deepening in transition literature) is needed to capture the lessons learnt and to document the change trajectory so that the information can be instrumental for repetition in other contexts (defined as broadening in transition literature). This learning history can contribute to this process and the transition framework proved to be very useful in this respect. However, more research is highly welcome to investigate such management practices further and to study the conditions that need to be fulfilled to scale up this new manner of forest management. To influence organization and management approaches on the regime level, more Bosland-like approaches are needed in other instances and contexts. It is of course hard to predict the future evolutions in the Flemish forest management regime. A strong focus on collaboration and participation seems to be a point of particular interest and in this respect some aspects of the Bosland approach seem valuable. However, in general Flemish forests are even more disintegrated physically and based on ownership. It is clear that an increased number of stakeholders for a reduced forest area will make the described approach more complex to implement. We believe that also in these situations a common narrative and a strong collaboration can increase involvement of all forest users. More time and more experimenting will be needed to develop similar approaches, to evaluate the specific strengths of different methods and to observe possible entry in the regime. For other projects to succeed, there are some constraints and similar circumstances as for Bosland (changing legislation and pressure on the landscape scale, enthusiastic people and a willingness to challenge the current culture) or other drivers might be needed to get over the threshold to start a trajectory of change.

Anyhow, the governmental Agency for Nature and Forest (ANB) has acquired a taste for the approach and is currently setting up a similar project in another forest and nature area in the province of Limburg (Duinengordel, 2012). With an eye to the ongoing transition in forest and nature management it will be highly interesting to observe the course of this project and to learn from the differences between the projects. Finally, more and mutually reinforcing success stories are needed for such novel management approaches to be scaled up. We conclude that Bosland can be regarded as a pioneering frontrunner case, not free of growing pains. Nevertheless, such pioneering cases as Bosland need to be described and analysed since they could be an essential stepping stone in the transition to more sustainable forest management systems.

2.6. Acknowledgements

We would like to express our gratitude to the interviewees for their time and personal account. Extra thanks to the people at the Bosland project office who provided us with policy documents and minutes of all participatory events.

3. Logging operations in pine stands in Belgium with additional harvest of woody biomass: yield, economics and energy balance

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3.1. Abstract

Due to the enhanced demands for woody biomass, it is increasingly relevant to assess possibilities to harvest forest harvest residues in addition to logs. Here, eight strategies for whole tree harvesting from clear-cuts and early thinnings of pine (*Pinus nigra*) stands in northern Belgium are evaluated. A detailed cost analysis using the machine rate method was conducted along with scenario and sensitivity analyses of the variables affecting the harvesting cost. On average, we found a much higher revenue for logs than for wood chips from forest harvest residues. In clear-cuts, a mobile chipper was more profitable than a road-side chipper. In early thinnings, on the other hand, the harvesting cost of logs was higher than for clear-cuts. However, the revenue remained higher than for chips, making the separate harvesting of logs and chips more cost-effective than chipping whole trees. In the latter case, an excavator, a forwarder and a road-side chipper were more cost-effective than a harvester, a tractor with trailer and a mobile chipper, respectively. Harvest of additional woody biomass required limited energy input compared to processing and intercontinental transport of wood pellets. However, at present we find very small profits from local additional biomass harvests. The low and fragmented forest cover and important sustainability issues further impede the development of a viable production sector in this region.

3.2. Introduction

The use of woody biomass for bioenergy has increased with almost 80% in the 27 European Union (EU) member states between 1990 and 2008 (Eurostat, 2011a). Moreover, the demand is expected to keep rising and to double by 2030, mainly as a result of the EU 20-20 objectives (Mantau *et al.*, 2010). For more than two-thirds, this woody biomass originates from forests (Mantau *et al.*, 2010). On the one hand, this rising demand resulted in an increased import of woody biomass, for Belgium and the Netherlands mostly as pellets from North-America (Sikkema *et al.*, 2010). On the other hand this also stimulated the interest in local production of wood chips and pellets, stipulating new questions for the forestry sector about the cost-effectiveness of different harvest strategies. The large scale utilization of woody biomass for bioenergy also raises serious questions on sustainability aspects (Schulze *et al.*, 2012).

In Flanders (the northern part of Belgium), the legislation only allows the production of renewable energy from smaller assortments of woody biomass that cannot be used as a material (Vlaamse Regering, 2004). For this reason, the newly applied forestry methods to produce wood chips and pellets in Flanders mainly include whole harvesting of trees in early thinnings and additional harvest of biomass that was previously left in the forest floor after roundwood harvest. Traditional logging operations for roundwood production in coniferous forests are highly mechanized and elaborative studies comparing productivity and economic return for different harvest strategies have been published for different regions (e.g. North-America (Adebayo *et al.*, 2007), Fennoscandia (Ovaskainen *et al.*, 2011) and Central-Europe (Mederski, 2006; Visser & Spinelli, 2012)). Harvest of woody biomass from early thinnings and from clear-cut harvest residues is also a highly mechanized and emerging practice while empirical evidence is more scarce (but see Spinelli & Magagnotti (2010), Lehtimaki & Nurmi (2011) and Walsh & Strandgard (2014)). Studies focussing on the economic aspects of energy wood harvest are even more scarce and coming from different regions, for different forest operations and for different tree species: clear-cuts in pine stands in Italy (Marchi *et al.*, 2011), clear-cuts in pine stands in USA (Conrad IV *et al.*, 2013), clear-cut in poplar stands in Italy (Spinelli *et al.*, 2012), clear-cut and heavy thinning in mixed stands of pine and cypress in an Italian mountain region (Spinelli *et al.*, 2014). The emerging patterns from these studies, are not always comparable and very hard to transfer to other systems and other regions, since harvest of woody biomass for bioenergy is species-, site- and practice-specific (Helmisaari *et al.*, 2014). Flanders and neighbouring regions for example are characterized by a low total forest area of 10-20% (Hermy *et al.*, 2008), disintegrated forest ownership with a mean size of the forest property of less than 1 ha (Van Gossum *et al.*, 2011) and a very high urbanisation rate (built-up areas amounted to 15% in 2005) (Hermy *et al.*, 2008), resulting in short transport distances for forest products. Harvesting costs for different harvest strategies for roundwood and additional biomass have, to our knowledge, never been investigated in this region. However, harvesting costs are extremely important, because together with transportation cost they often represent about 70% of the total biomass cost (Panichelli & Gnansounou, 2008).

Here we report the results of a large-scale field experiment in Corsican pine (*Pinus nigra*) stands in the Bosland region in Flanders, comparing several harvest strategies for roundwood production and additional wood chip production from clear-cuts and thinnings. We specifically investigated (i) whether the currently applied roadside chipping strategy was more cost-effective than on-site chipping both for clear-cuts and thinnings, (ii) how variation in the top bucking diameter (i.e., the diameter of the stem where the tree is separated for roundwood and for wood chip production) influenced the total harvest income and the quality of roundwood and wood chips in clear-cuts, (iii) what the cost efficiency was of separately harvesting the stem for roundwood and the crown for wood chips compared to whole-tree chipping in early thinnings, (iv) whether a simpler combination of an excavator with a shear harvester head and a tractor with a trailer had a similar efficiency as a typical harvester-forwarder combination in harvesting whole trees for wood chip production in thinnings. Moreover we examined the energy input in the production process of the locally produced wood chips as one aspect of sustainability and compared it with pellets imported from North-America.

3.3. Materials and methods

3.3.1. Study site

The study was executed in Bosland (see chapter 2 for more information on the study area). In 2012, eight monoculture Corsican pine stands of similar size (average 1.14 ha) were selected for a field trial (Table 3.1). In Lommel, we selected four stands of an older stand type for a clear-cut (47 years old, median diameter at breast height (dbh) of the trees was 26 cm). These stands had been thinned once, at an age of about 30 year. In Overpelt, we sampled four stands of a younger stand type (33 year old, median diameter at breast height (dbh) was 15 cm) for early thinning. Trees within these stands were harvested as roundwood for a factory producing orientated strand board (OSB) and as wood chips for combustion. All stands were equally accessible for the various forest machines and a place for stocking of logs and wood chips was available within 500 m of all stands.

The dbh of all trees in three randomly located square plots of 400 m² per stand was measured before and after the harvest. The standing stocks of the old stand type (average 355 m³/ha) differed significantly from the young stand type (305.29 m³/ha) (analysis of variance and a Tukey post hoc test with stand as a blocking factor; overall p-value < 0.01). Within each stand type no significant differences were found between the stands (p-value for the four older stands = 0.162; for the four younger stands = 0.483). Therefore, the selected stands were suitable for our analysis since the circumstances were comparable for all stands within each stand type and it was presumed that terrain circumstances are no explanation for possible differences between harvesting efficiencies.

Table 3.1: Characteristics of the four older Corsican pine stands that were clear-cut in Lommel (L1-4) and the four younger stands that were thinned in Overpelt (O1-4). More information about the harvest strategy can be found in Table 3.2.

	Area (ha)	Year of planting	Standing stock (m³/ha)	Thinning intensity (% stem number)	Harvest strategy
L1	1.15	1965	349.3	/	C1
L2	1.17	1965	364.4	/	C2
L3	0.89	1965	341.8	/	C3
L4	0.92	1965	365.5	/	C4
O1	1.05	1979	272.5	20.1	T1
O2	1.00	1979	315.8	24.9	T2
O3	1.35	1979	327.8	21.2	T3
O4	1.55	1979	305.0	15.8	T4

3.3.1. Tested harvest strategies

A literature review was performed and the possible strategies for combined harvest of roundwood and wood chips were listed (Spinelli & Hartsough, 2001; Spinelli & Magagnotti, 2010; do Canto *et al.*, 2011; Lehtimaki & Nurmi, 2011; Marchi *et al.*, 2011; Conrad IV *et al.*, 2013; Walsh & Strandgard, 2014). In order to increase the practical relevance of our empirical study, we invited local policymakers, forest harvesting experts and stakeholders from the Belgian wood industry and from the bio energy sector to take part in a board of experts. On 14 May 2012, 12 experts discussed the different options and jointly selected the 8 most promising harvest strategies from the list, based on criteria of technical and economic suitability and practical knowledge gaps (Table 3.2, Table 8.1 in Appendix).

Table 3.2: Selected strategies for combined harvest of roundwood and wood chips for clear-cuts (C1-4) and early thinnings (T1-4) in Flanders.

Strategy	Explanation of strategy	Motivation of selection by board of experts
C1	Harvester + forwarder for stem-wood (Ø** 12 cm) + Forwarder for crown wood + chipper on roadside	Interesting to see what influence an increase in top bucking diameter has (more biomass – less stem wood).
C2	Harvester + forwarder for stem-wood (Ø 7 cm) + Forwarder for crown wood + chipper on roadside	The actual standard approach, the board of experts believes this to be the most efficient strategy.
C3	Harvester + forwarder for stem-wood (Ø 12 cm) + Crown wood chipped in stand by mobile chipper behind tractor	Interesting to see what influence an increase in top bucking diameter has + if terrain chipping can be economically feasible.
C4	Harvester + forwarder for stem-wood (Ø 7 cm) + Crown wood chipped in stand by mobile chipper behind tractor	Highly interesting to test if terrain chipping can be economically feasible.
T1	Harvester + forwarder for whole trees + chipper on roadside	The actual standard approach for early thinnings
T2	Harvester + whole trees chipped in stand by mobile chipper behind tractor	Highly interesting to test whether terrain chipping can be economically feasible in early thinning operations
T3*	Harvester + forwarder for stem-wood + forwarder for crown wood + chipper on roadside	The actual standard approach for early thinnings in older stands. Interesting to see if it is economically more feasible to harvest stem wood separately
T4	Excavator with shear harvester head + tractor with trailer for whole trees + chipper on roadside	Interesting to test what economic outcome will be of this low-tech variation of the actual standard approach

* Not in original shortlist, but added by board of experts because of the relatively high age of stands for early thinning in Overpelt

** Ø = top bucking diameter

The specifications of the harvesting were outlined and sent to different forest harvesting companies. Three companies sent in an offer and, as usually done in Flanders, the company proposing the best financial conditions was selected. We expected that this market based selection would result in a cost-efficiency driven and close-to-reality harvesting approach. Before the start of the harvest, a meeting was set up with the operator to outline the conditions for the experiment in detail (different harvest strategy for every stand and presence of scientists during operations).

The board of experts deliberately selected simple harvest strategies, involving relatively basic forestry equipment (Figure 3.1, Table 8.1 in Appendix). The high-tech harvest strategies (e.g. T5, T6) are probably not economically feasible for the Flemish and Western European forestry context with low forest area, small stands and short hauling distances. The harvest strategy including the mobile terrain chipper behind a tractor was perhaps the only exception since, to the best of our knowledge, this combination was never used in Flemish forestry before. The mobile terrain chipper used in the experiment was mostly used for chipping operations on trees along public roads and was not equipped with forestry tires. Before every operation with the mobile chipper, a mulcher was used to flatten the terrain. By simply equipping the mobile chipper with forestry tires, the use of the mulcher could have been avoided and for this reason the costs of the mulcher were not included in the cost comparison.

Each machine was always operated by the same operator in the different stands, avoiding operator training bias. Nonetheless, each machine was operated by a different operator, to enhance machine efficiency due to operator skills. To minimize operator bias, the harvesting company selected experienced operators with more than three years working experience for each machine. Note, however, that the operator for the mobile chipper had an equal amount of experience with the machine as the other operators, but mostly from harvesting of tree lines on roadsides and less in forest harvesting.

3.3.2. Data collection

Machine costs were calculated using the machine rate method (Miyata, 1980), separating fixed costs, variable costs and labour cost. We used a stopwatch to measure the time of every separate step in the harvest and the breaks, also the reason for a break was registered (i.e., operator break vs. technical break). The total fuel consumption for every machine for each of the harvest strategies was measured as well. Each machine started with a full fuel tank and was refilled after each operation by means of a field fuel pump which registered the amount of fuel that was tanked up. Most of the data about the machinery (e.g., purchase price, economic life, salvage value, annual use, repair and maintenance cost, fuel cost) were provided through the harvesting companies. For estimating the utilization rate (i.e., the ratio between productive hours and scheduled machine hours, SMH) we first determined the ratio between all breaks and productive hours in the field trial. To compensate for the transport of the machinery this value was then decreased with 10% for our final estimate of the utilization rate (inferred from Mederski (2006)). Data about interest rate, insurances and taxes (do Canto *et al.*, 2011), lubricant cost (Conrad IV *et al.*, 2013; Adebayo *et al.*, 2007), overhead and labour cost (Marchi *et al.*, 2011) were obtained from literature and double-checked with the harvesting companies for accuracy.

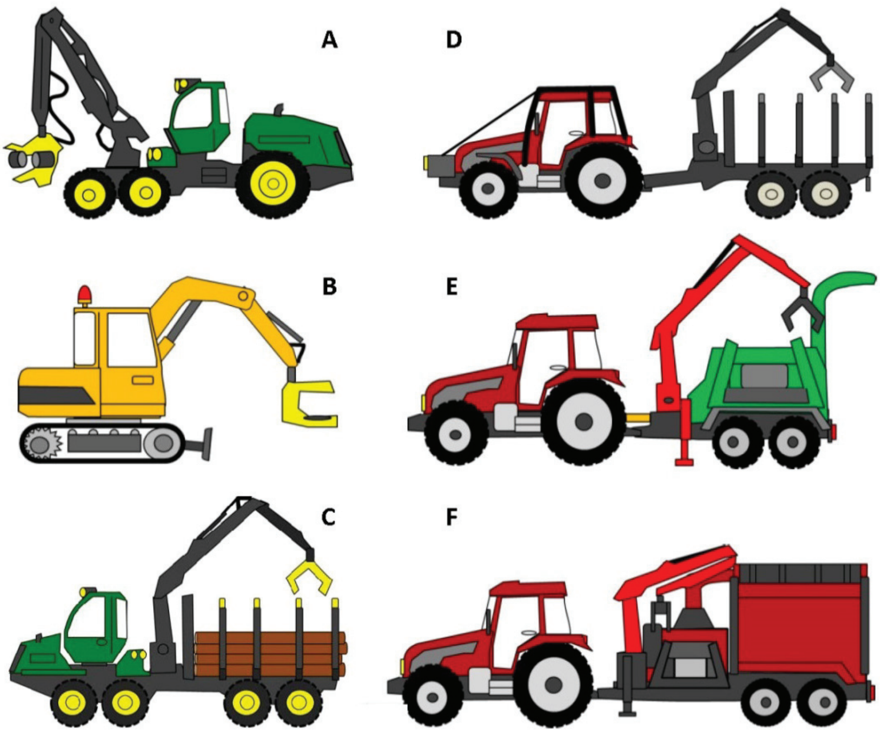

Figure 3.1: Drawings of the machines used in the experiment: a harvester (A), an excavator (B), a forwarder (C), a tractor with trailer (D), a roadside chipper (E) and a terrain chipper (F) (drawings by INVERDE after Osselaere & Vangansbeke (2013)*).*

The figures of fresh mass of the wood chips harvested in each stand and the total mass of the roundwood of the clear-cuts and of the early thinning (strategy T3) were obtained from the OSB factory and the volume of the harvested stem wood from every stand was obtained from the operator. Also one pooled sample of the harvested wood chips was obtained for every treatment by taking 10 subsamples from every container. The sampled chips from every stand were dried in the oven at 105 °C for two days to determine the moisture content on wet basis and the dry mass (according to the NEN-EN 14774-2 norm). The particle distribution of the chips was determined with sieves according to the NEN-EN 15149-1 norm and the ash content was determined through gradually heating a grinded subsample of the chips to 550 °C according to the NEN-EN 14775 norm.

3.3.3. Data analysis

For every harvest strategy, the total cost was calculated by combining the machine cost per SMH, calculated using the machine rate method (Miyata, 1980), with the productive time and utilization rate for each machine. The harvesting cost per green metric ton (GMt) roundwood and wood chips at the edge of the stand for each strategy was then calculated by dividing the total cost for each strategy by the fresh mass of the harvest.

The variables used to determine the harvesting cost of wood and biomass were mostly obtained by interviews and literature and are, therefore, deterministic rather than stochastic. A sensitivity analysis was carried out to determine the variables that have the highest impact on the harvesting cost. A Monte Carlo simulation (50,000 trials) was performed for the harvesting cost of roundwood and wood chips for each strategy, varying the variables following a normal distribution with a standard deviation of 10% of the estimated value (given in Table 3.4). The sensitivity of the harvesting cost for a certain variation of each variable was determined as the amount of the harvesting cost variance that was explained by the variance of that variable in a linear model (R^2 value)(Van Dael *et al.*, 2013). All analyses were performed in R 3.0.1 (R Core Team, 2013).

We calculated the ratio between the total fossil energy consumed during the additional harvest of the biomass and the energy output of the harvested wood chips under the different strategies as a sustainability criterion (Marchi *et al.*, 2011). The total fossil energy consumed was estimated by multiplying the energy content of 37 MJ/L (Bailey *et al.*, 2003) for diesel with the measured consumption for additional harvest and by first increasing this value by 20 % to account for the production and transport of the fuel and then by 30 % for manufacturing, repair and maintenance of the machines (following Mikkola & Ahokas (2010)). The theoretical energy output of oven dry wood chips was estimated using a net calorific value (NCV_0) of 18,5 MJ/kg (Francescato *et al.*, 2008).

3.4. Results

3.4.1. Amount of harvest

An average of 355.4 green metric ton (GMt) of roundwood was harvested per hectare from the clear-cuts (Table 3.3A). As expected, a higher amount of roundwood was found for smaller top bucking diameters (average 365.2 GMt/ha vs. 345.3 GMt/ha). The extra biomass from the clear-cuts, harvested as wood chips from the tree tops, amounted to an average of 89.5 GMt/ha. The amount of wood chips from the clear-cut stands where a 12 cm top bucking diameter was used was higher vs. a 7 cm top diameter (average 92.6 GMt/ha vs. 86.4 GMt/ha). For the thinned stands where whole trees were chipped, the average harvest was 113.74 GMt/ha wood chips. In the other thinned stand, we harvested 60.5 GMt/ha roundwood and 42.3 GMt/ha wood chips. In both the thinned and the clear-cuts stand some harvest residuals were left on the site, even after the additional biomass harvest, but were not measured in this study.

3.4.2. Harvesting cost of logs and wood chips

The cost per SMH was highest for the mobile chipper (€ 130.28), followed by the road-side chipper (€ 96.62), the harvester (€ 64.76) and the forwarder (€ 52.07) (Table 8.2 in Appendix). The cost is also determined by the effective working time of the machines in each strategy, which was generally highest for the harvesters (Table 3.3B). A higher wood harvesting cost was found for the logs in the thinning operation (€12.09/GMt) in comparison to the clear-cut operation (average of €6.19/GMt), due to the more difficult harvesting conditions due to the remaining stand (Table 3.3C). In the clear-cuts, no difference was found between the harvesting cost of the logs in relation to the top bucking diameter. However, a lower wood chip harvesting cost was found under strategies with the mobile chipper (average €12.76/GMt) and with a larger top bucking diameter (average €14.17/GMt), compared to the strategies with a roadside chipper (average €16.19/GMt) and a smaller top bucking diameter (average €14.78/GMt), respectively. The lower wood chip harvesting cost for a larger top bucking diameter was caused by the larger dimensions and higher cohesion and density of the biomass which made chipping easier and more efficient. The better result for the on-site mobile chipper was explained by the shorter waiting breaks and the resulting higher utilization rate.

Table 3.3: A. Total harvest of logs and wood chips of a forest stand for the different strategies. B. Productive time and total cost for each machine under the different strategies. C. Calculated harvesting cost per green metric ton logs and wood chips at the edge of the stand. For the 'additional wood chips', we did not account for the cost of felling, which was included in the cost of log production. For the whole tree chips (T1, T2 and T4) on the contrary we did assign the felling cost in the cost for wood chip production . D. Moisture content and ash residue of the wood chips from the different harvest strategies.

		C1 1.15 ha	**C2 1.17 ha**	**C3 0.89 ha**	**C4 0.92 ha**	**T3 1.35 ha**	**T1 1.05 ha**	**T2 1.00 ha**	**T4 1.55 ha**
A	Total harvest of logs (GMt)	401.91	426.58	304.37	336.44	81.74	/	/	/
	Total harvest of wood chips (GMt)	107.78	100.33	81.44	80.06	57.12	99.72	136.52	170.08
B	Harvester productive time (h)	19.73	17.38	12.13	14.16	8.05	6.89	9.72	/
	Harvester cost (€)	1728.97	1523.03	1062.97	1240.86	705.43	603.78	851.44	/
	Excavator productive time (h)	/	/	/	/	/	/	/	11.65
	Excavator cost (€)	/	/	/	/	/	/	/	802.53
	Forwarder wood productive time (h)	12.95	15.87	10.53	13.77	4.23	/	/	/
	Forwarder wood cost (€)	866.30	1061.45	704.57	920.85	283.17	/	/	/
	Forwarder biomass productive time (h)	5.81	5.78	/	/	5.00	4.60	/	/
	Forwarder biomass cost (€)	388.84	386.84	/	/	334.44	307.68	/	/
	Tractor + Trailer productive time (h)	/	/	/	/	/	/	/	11.94
	Tractor + Trailer cost (€)	/	/	/	/	/	/	/	743.23
	Tractor + Roadside chipper productive time (h)	5.08	4.83	/	/	2.22	3.25	/	4.58
	Tractor + Roadside chipper cost (€)	1328.79	1263.47	/	/	579.43	849.57	/	1198.08
	Tractor + Mobile chipper productive time (h)	/	/	6.83	7.09	/	/	11.49	/
	Tractor + Mobile chipper cost (€)	/	/	1010.44	1049.07	/	/	1699.13	/
C	Harvesting cost of logs (€.GMt^{-1})	6.46	6.06	5.81	6.43	12.09	/	/	/
	Harvesting cost of additional wood chips (€.GMt^{-1})	15.94	16.45	12.41	13.10	16.00	/	/	/
	Harvesting cost of whole tree wood chips (€.GMt^{-1})	/	/	/	/	/	17.66	18.68	16.13
D	Moisture content of the wood chips (%)	58.25	59.58	60.66	61.05	55.41	57.33	60.72	58.74
	Ash residue of the wood chips (%)	2.77	3.90	1.12	1.42	8.10	2.04	0.63	0.74

The thinnings where whole trees were chipped resulted in the highest total harvesting cost for wood chips of all strategies, mainly due to the inclusion of the cost for felling. Among these three strategies, the combination of an excavator, tractor with trailer and road-side chipper (€16.13/GMt) led to the lowest harvesting cost and the harvester-forwarder-road-side chipper combination (€17.66/GMt) scored slightly better than the harvester-mobile chipper combination (€18.68/GMt). The lowest harvesting cost for whole tree chips under strategy T4 was due to the use of the excavator, that had a lower cost per SMH and a similar utilization rate and productivity (GMt/h) as a harvester in thinnings. The harvesting cost under this strategy could even have been lower if a forwarder was used in this scenario, as the use of the tractor and trailer had a lower cost-efficiency, because of the lower productivity (GMt/h) for a similar cost per SMH and utilization rate. The highest harvesting cost for wood chips under strategy T2 was due to the more pronounced drawbacks of the on-site mobile chipper in thinnings: the machine and operator had less experience in real forest operations and manoeuvring the tractor with mobile chipper (including a chip container) through the thinning corridors cost extra time. In T3, where logs were produced, the harvesting cost of wood chips was comparable with the clear-cut strategies with the roadside chipper.

3.4.3. Sensitivity and scenario analysis

The sensitivity analysis revealed that, for every wood harvest strategy, the harvesting cost of logs mainly depended on the utilization rate (on average explaining 30.3% of the variation in harvesting cost), the purchase price (11.2%) and the annual use of the harvester (9.9%) (Table 3.4). The labour cost (16%) and the utilization rate of the forwarder (10.9%) were also important.

For the harvesting cost of additional wood chips, the utilization rate of the chipper (both mobile and road-side in the respective scenarios) was by far the most important variable (on average explaining 51.2% of the variation in harvesting cost). Other important variables were the purchase price (8.1%) and the annual use of the chippers (6.8%), the labour (only for the roadside chipper, 7.33%), the utilization rate of the tractor of the mobile chipper (6.9%) and the repair and maintenance of the mobile chipper (5.7%). Looking at the harvesting cost for whole tree chips, the utilization rate of the chippers remained the most important variable (accounting for 33.3% of the variation in harvesting cost), but was more closely followed by different variables for the different scenarios, i.e., the labour cost (T1, T4; 14.5%), the utilization rate of the harvester (T1, T2; 10%) and the utilization rate of the trailer (T4; 12.7%).

*Table 3.4: Sensitivity analysis of the cost calculations. For every harvesting cost the R^2 values of the five most important variables influencing the costs are shown, indicating the importance of the variable in the variation of the cost price after 50,000 trials. The R^2 value of the most important variable is highlighted. If a variable was not relevant in a harvesting cost calculation it was represented with a /, R^2 values for variables that were less important for a strategy are represented with a *.*

	Harvesting cost of roundwood					**Harvesting cost of additional wood chips**					**Harvesting cost of whole tree wood chips**		
	C1	**C2**	**C3**	**C4**	**T3**	**C1**	**C2**	**C3**	**C4**	**T3**	**T1**	**T2**	**T4**
Utilization rate harvester	**0.330**	**0.277**	**0.286**	**0.266**	**0.358**	* *	* *	*	*	*	0.107	0.092	/
Labour	0.153	0.168	0.166	0.171	0.142	0.063	0.064	*	*	0.093	0.126	0.068	0.164
Purchase price harvester	0.121	0.102	0.105	0.098	0.132			*	*	*	*	*	/
Utilization rate forwarder	0.084	0.134	0.125	0.145	0.059			*	*	0.091	*	*	/
Annual use harvester	0.108	0.091	0.093	0.087	0.117	*	*	*	*	*	*	*	/ *
Utilization rate roadside chipper	/	/	/	/	/	**0.497**	**0.493**	/	/	**0.409**	**0.314**	/	**0.271**
Purchase price roadside chipper	/	/	/	/	/	0.104	0.103	/	/	0.086	0.064	/	0.056
Annual use roadside chipper	/	/	/	/	/	0.092	0.091	/	/	0.077	0.058	/	
Economic life roadside chipper	/	/	/	/	/	0.055	0.055	/	/	*	*	/	
Utilization rate mobile chipper	/	/	/	/	/	/	/	**0.581**	**0.581**	/	/	**0.415**	/
Utilization rate tractor of mobile chipper	/	/	/	/	/	/	/	0.069	0.069	/	/	0.050	/
Purchase price mobile chipper	/	/	/	/	/	/	/	0.057	0.057	/	/	0.043	/
Repair and maintenance mobile chipper	/	/	/	/	/	/	/	0.057	0.057	/	/	*	/
Annual use mobile chipper	/	/	/	/	/	/	/	0.039	0.039	/	/	*	/
Utilization rate trailer	/	/	/	/	/	/	/	/	/	/	/	/	0.127
Fuel price	*	*	*	*	*	*	*	*	*	*	*	*	0.063

To illustrate the importance of the difference in utilization rate between the chippers a scenario analysis was conducted, varying the utilization rate of the road-side and mobile chipper for respectively strategy C2 and C4 (Figure 3.2). For a similar utilization rate, the harvesting cost of the wood chips of the road-side chipper was always lower, even for a 10% higher purchase price for the road-side chipper and a 10% lower purchase price for the mobile chipper (the second most influential variable). Currently, the harvesting cost of the wood chips of the mobile chipper was lower due to the much higher utilization rate. The utilization rate of the road-side chipper should increase to at least 56% to compete with the mobile chipper under current purchase prices.

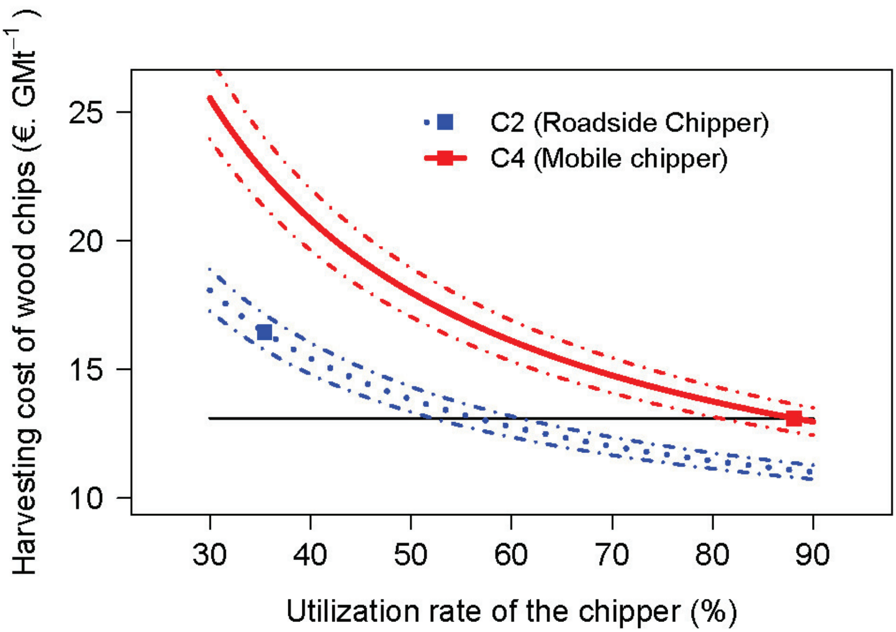

Figure 3.2: Scenario-analysis on the impact of utilization rate and purchase price of road-side and mobile chipper on the harvesting cost of a green metric ton of wood chips for harvest strategies C2 and C4. The striped-dashed lines show the harvesting cost with a 10% reduction or a 10% increase of the purchase price. The squares show the current situation, the horizontal line shows that the utilization rate of the road-side chipper should increase to 56% to compete with the mobile chipper.

3.4.4. Wood chip quality

The analysis of the wood chip quality showed several differences between the harvest strategies. For the clear-cut strategies, a difference between the location of chipping was observed. When the crowns were chipped in the stand (strategies C4 and C3), a larger share of the smaller chip fractions, a lower ash residue and a slightly higher moisture content was found (Figure 3.3, Table 3.3D). This smaller average fraction was due to the very low degree of large chips (>32 mm) caused by a smaller mesh size of the screen of the mobile chipper. In a chipper the woody biomass is comminuted until the particles can permeate through a screen. A smaller mesh size thus results in smaller particles and also a lower efficiency of the chipper, because of the longer chipping process (Nati *et al.*, 2010). A high fraction of very small particles (< 3 mm) lowers the overall quality of the wood chips. The fraction of very small particles is quite high under all clear-cut strategies, and definitely under the strategies with a lower bucking diameter (C2 and C4). The lower quality of the chips from the road-side chipper (higher ash residue) and the lower moisture content was due to the extra handling under these strategies, which increased the chance on pollution with soil and the extra possibility to dry at the air. We also found a higher ash residue and a slightly higher moisture content in strategies C2 and C4 compared to, respectively, C1 and C3. This lower chip quality and large amount of very small particles under the strategies with a small top bucking diameter was related to the relatively higher share of green material than wood.

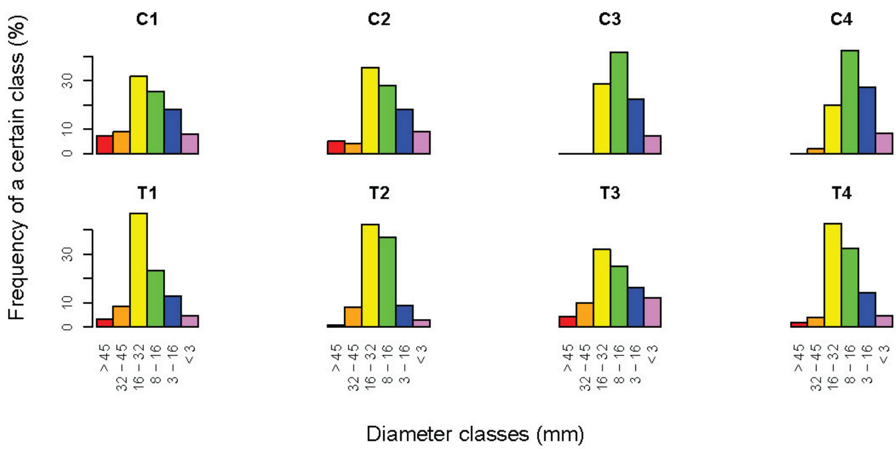

Figure 3.3: Distribution of the wood chips from each of the harvest strategies in the clear-cuts in diameter classes.

The analysis of the wood chip quality showed that the chips from strategy T3, where logs were harvested separately, had the highest ash residue, the lowest moisture content and the largest share of small particles (<3 mm), because of the relatively lower share of wood than green material. The chips from strategy T2, involving the on-site mobile chipper had the lowest ash residue and the highest moisture content of the thinned stands, for the same reasons as raised for the clear-cuts. The chips from strategy T4 had a lower ash residue and a higher moisture content than the other thinning strategies with the road-side chipper (T3 and T1). This higher chip quality was due to the use of an excavator instead of a harvester. The excavator lifted the trees after felling and was better suited to put the trees softly on the ground, reducing the pollution with soil particles.

3.4.5. Energy balance

The ratio between the extra fossil energy input to harvest the additional biomass as wood chips on the one hand and the theoretical energy output from the wood chips on the other hand varied between 0.71% and 1.16% under the harvest strategies where roundwood was harvested separately. In the clear-cuts a lower ratio was found under the harvest strategies with the mobile chipper (average 0.75%) and with a smaller top bucking diameter (average 0.91%) compared to strategies including the road-side chipper (average 1.14%) and a larger top bucking diameter (average 0.98%), respectively. For the whole tree wood chips from the thinnings, the ratio was higher and amounted to an average of 1.29%, because all used fuel was accounted for.

3.5. Discussion

In Flanders and neighbouring temperate regions, pine stands make up a large part of the forests (e.g., 39% in Flanders (Waterinckx & Roelandt, 2001), 33% in the Netherlands (Dirkse *et al.*, 2007)). Traditionally, these stands are thinned after 30 years and clear-cut at the end of the rotation period, which mostly varies between 40-110 years (Pussinen *et al.*, 2002). Pihlainen *et al.* (2014) reported on longer rotation periods if carbon storage was co-included as a management target, while Dwivedi & Khanna (2014) evaluated much shorter rotation periods when focussing on biomass production. Thus, the two tested forestry operations, thinning and clear-cut of pine stands of 33 and 47 years old, can be considered as quite characteristic for pine stand management with a short to average rotation period. Given the importance of pine stands in Flanders and neighbouring regions and the silvicultural system applied in these stands, the comparison between different harvest strategies for these forestry operations is probably the most relevant forestry experiment for the woody biomass industry in this region. Below we elaborate on the results and try to draw relevant conclusions for the forestry sector in the region.

3.5.1. Harvesting cost and economic balance

In the clear-cuts, the lowest wood chip harvesting cost was found for the strategy involving an on-site mobile chipper and using a larger top bucking diameter of 12 cm. In the thinnings the cheapest strategy to produce wood chips was from the crowns of trees where stems were harvested separately as logs (which were, however, much more expensive to harvest than in the clear-cuts). Marchi *et al.* (2011) used a similar set-up for clear-cuts in pine stands, but found contrasting results: a harvesting cost of €18.3/GMt for a terrain chipper and €12.3/GMt for a roadside chipper. However, in this study, the roadside chipper had a utilization rate of 67.6%. Our scenario analysis showed a similar harvesting cost at this utilization rate. The costs using a terrain chipper are harder to compare between the studies because a different type of machine, without a built-in container, was used. The contrast with our results remains striking, certainly considering the limited experience with the mobile chipper in forest stand. However, photographic material from (Marchi *et al.*, 2011) also shows that the harvest residuals for terrain chipping were sloppily left all over the stand, making it less accessible. Spinelli *et al.* (2012) also made a comparison between roadside and terrain chipping. Parallel to our results, they found a lower harvesting cost for terrain chipping (€16.3/GMt and €17.1/GMt for two different poplar clones) than for roadside chipping (€19.7/GMt and €23.2/GMt). However, these results were found for whole tree chips from easily accessible stands with a short rotation period (Spinelli *et al.*, 2012). It is thus speculative to draw conclusions from these three diverging studies, but terrain accessibility seems a key factor in explaining success of terrain chipping (note also the much higher harvesting costs for terrain chipping in the less accessible thinnings in this study) .

As mentioned earlier, the harvesting cost calculated in this study covers only the process from the standing stock to the fresh logs and chips at the stocking place on the roadside. Afterwards logs and chips were sold and transported to the OSB factory and associated energy plant. We assumed a cost of €8/GMt (the average price according to the operators) for the transport of the chips and logs and a resale price of €30/GMt and €50/GMt for the wood chips and the logs respectively, as was paid by the customer in the experiment. We calculated an economic balance including resale value and transport costs to obtain an overview and to make a complete comparison between the strategies (Table 3.5).

Table 3.5: Balance and revenue of the production of logs, additional wood chips and whole tree wood chips under the different harvest strategies, given the current resale price (€30 per MGt woodchips; €50 per MGt logs) and transport cost (€8 per MGt).

	C1	C2	C3	C4	T1	T2	T3	T4
Balance logs (€.MGt^{-1})	35.54	35.94	36.19	35.57	/	/	29.91	/
Balance additional wood chips (€.MGt^{-1})	6.06	5.55	9.59	8.90	/	/	6.00	/
Balance whole tree wood chips (€.MGt^{-1})	/	/	/	/	4.34	3.32	/	5.87
Revenue logs (€.ha^{-1})	12421.70	13104.17	12377.53	13009.54	/	/	1810.73	/
Revenue additional wood chips (€.ha^{-1})	568.29	476.00	877.80	774.19	/	/	253.90	/
Revenue whole tree wood chips (€.ha^{-1})	/	/	/	/	412.19	452.87	/	643.82
Total revenue (€.ha^{-1})	12989.99	13580.17	13255.34	13783.72	412.19	452.87	2064.63	643.82

Under the current circumstances, using a mobile chipper and a small top bucking diameter (e.g. 7 cm) was the most interesting clear-cut strategy from an economic point of view. In the thinnings, it was found that harvesting logs separately was - by far - the most beneficial. The strategies in which the whole trees were chipped were less favourable. The best of these strategies was the one where the trees were felled by an excavator, moved on by a tractor and trailer and chipped by a road-side chipper. The strategy using a mobile chipper was by far the least cost-effective, but this result might be biased by the limited experience of the operator in harvesting in forest stands. The main conclusion from this economic analysis is that the revenue from the wood was much higher than the revenue from the wood chips, because of the lower harvesting cost and the higher selling price. In the clear-cuts, strategies using a smaller top bucking diameter resulted in a larger share of logs and less wood chips. This was much more profitable, because the extra income of the higher share of logs exceeded by far the extra harvesting cost of the wood chips under the strategies using a smaller top bucking diameter. Moreover, the hypothetical price shift for the wood chips should be large to compensate for the lower income from logs under the scenarios with a large top bucking diameter. Using larger top bucking diameters could indeed have a positive impact on the large-scale bioenergy potentials, as stated in Räisänen & Nurmi (2014), however this seems economic unfeasible. A case study from pine plantations in the southern Coastal Plain, USA, (Conrad IV *et al.*, 2013) also compared the economical balance of harvesting wood for material and for energy purposes and came to the same conclusion: “until energy wood prices appreciate substantially, loggers are unlikely to sacrifice roundwood production to increase energy wood production”.

According to the economic balance it was profitable to harvest additional biomass under the form of wood chips. However, the revenue was very small and forest management costs and the potential cost of the loss of other ecosystem services due to this additional biomass harvesting were not yet included. Moreover we did not investigate the fact that subsequent biomass harvesting could reduce productivity in roundwood harvesting and extraction. These productivity losses can have a significant impact on the unit cost of roundwood harvest and extraction (e.g. Walsh & Strandgard (2014) found a 4.9% increase in cost in Australian *Pinus radiata* monocultures on flat terrain). Future income losses should, in theory be discounted to evaluate the profitability of this biomass harvest. It is questionable whether a profitable business model can be developed for this additional biomass harvest in Flanders under current price conditions. Generally, unit production costs decline as fixed costs are spread over increasing production volume (Mansfield, 1988), a large-scale harvest could maybe be more profitable. However, this is hard to realise in the Flemish forestry context with limited forest cover.

The revenue was no direct profit for the exploitation company, that paid a price to the forest owner to execute the harvesting and to buy the logs and the wood chips. In our case-study, the harvesting company had to use a different harvesting strategy for each stand, leading to higher costs and a lower, thus not representative, price that was paid to the forest owner. It is, however, clear that the harvesting company could pay more to the forest owner for the clear-cut than for the thinning and that there is hardly negotiation space to pay for additional biomass harvest, due to the limited revenue. From the position of the forest owner, the total price paid for the harvest must at least compensate for the cost of managing the stand (e.g., for forest regeneration in 1965/1997). Moreover, the harvest of logs and wood chips could lead to a decrease in biodiversity (Berger *et al.*, 2013), nutrient cycling (Schulze *et al.*, 2012; Berger *et al.*, 2013), carbon sequestration (Berger *et al.*, 2013; Schulze *et al.*, 2012; Helmisaari *et al.*, 2014) and some other ecosystem services of the stands, which might have an economic consequence for the forest owner (e.g. by reducing stand productivity for next rotations (Walmsley *et al.*, 2009; Wall, 2012)). So, for the system to be economically sustainable, the money the forest owner receives must also compensate for this potential economic loss.

3.5.2. Sensitivity and scenario analysis

The sensitivity analysis indicated that the utilization rate of the chipper is the single most important variable affecting the harvesting cost of the chips. For the road-side chipper, a utilization rate of only 35% is found, which is a clear explanation for the higher harvesting cost. The very low utilization rate of the road-side chipper in our study, was also evident on the field by the high frequency of forced technical breaks because of the limited transport capacity. Whenever the containers were filled with wood chips, the mobile chipper had to wait for the containers to be transported and emptied at the energy plant. Spinelli & Visser (2009) found an average utilization rate of 73.8% for 36 different chipping machines and described two studies with comparable utilization rates also due to organizational delay. Our scenario analysis revealed that increasing the utilization rate of the road-side chipper could be a way to reduce harvesting cost of the wood chips. This asks for a better alignment of the truck transportation strategy to the productivity of the road-side chipper meaning that more trucks for transport and thus more personnel are required to keep up with the roadside chipper. This, in turn, would require a larger scale of harvesting of additional biomass, reducing viability in Flanders and neighbouring regions to a (possibly very) limited number of companies. We expected a realistic and cost-efficiency driven harvesting approach from the harvesting company. The results of our scenario analysis showed that better equipment balancing (finetuning different steps of the harvest chain with each other, in this case increasing the number of trucks) could easily increase the utilization rate of the roadside chipper and, consequently, reduce the wood chip production cost. It is clear that mobile chipping holds some potential under these circumstances, but more research with a control for equipment balance and operator training level could further answer these remaining questions.

3.5.3. Wood chip quality

Good quality wood chips include a small share of chips that are too big (>63 mm) or too fine (<3mm) and a low degree of pollution (i.e. a low ash residue) (Spinelli *et al.*, 2011). Spinelli *et al.* (2011) compared wood chips from four different feedstock types in Italy and concluded that quality of wood chips from forest residues is generally lower than wood chips from sawmill residue and from small whole trees. The amount of fines in the clear-cuts in our experiment varied between 7% and 9%, which seemed acceptable and in line with the results from the road-side chipper in Marchi *et al.* (2011). However, the relatively high ash residue from the chips chipped at the road-side made the quality of this biomass inferior to the chips from the terrain chipper. The whole tree chips from the thinnings had a relatively high quality, confirming the findings of Spinelli *et al.* (2011). Especially the trees harvested with an excavator and the terrain chipped biomass showed a very low degree of pollution. A really inferior quality was found for the chips from the thinning where round wood was extracted first.

For small installations, wood chips with a lot of small particles and a high ash residue (such as the wood chips from treatment T3) are unsuitable and thus in need of a pre-treatment, such as sieving. When the wood chips are used in a more robust, large energy plant, this is less important. In our case, the customer paid an equal price (€30/GMt) for all chips, in spite of the significant differences in chip quality. Production of higher quality wood chips (involving a higher share of stem wood) is not promoted. So, from an economic point of view it is definitely more interesting to harvest as much of the trees as possible as logs, of course respecting the lower margin of 7 cm imposed by the particle board company.

3.5.4. Woody biomass: an efficient source of renewable energy?

Application of woody biomass for the generation of bioenergy is subject to fierce discussion. On the one hand, bioenergy from woody biomass strongly reduces greenhouse gas emissions compared to non-renewable energy (Njakou Djomo *et al.*, 2013). On the other hand, woody biomass left in the forest aids carbon sequestration and climate mitigation (Schulze *et al.*, 2012). A good quantification of the greenhouse gas balance of forestry operation asks for a life cycle analysis including all direct and indirect emissions and falls beyond the scope of this paper. However, Njakou Djomo *et al.* (2011) demonstrated a significant positive relationship between the greenhouse gas emissions and the energy efficiency (ratio between energy input and theoretical energy output) of the harvesting and production process, which is easier to calculate. We calculated the energy efficiency for wood chips from clear-cut harvest residues, harvested with an on-site mobile chipper (0.75%) and with a road-side chipper (1.14%) and for whole tree chips from thinnings (1.29%). On-site chipping of harvest residues in clear-cuts led to the highest energy efficiency, but in general the amount of energy used during harvesting and chipping biomass was limited. Other processes in the production chain of, for example, imported pellets are much more important to calculate the total energy balance: drying (e.g., 10.71 % dry mass loss when dried in a terminal), pelletizing (e.g., 24.6% of internal energy used) and intercontinental shipping (e.g., 6% of internal energy consumed for transport by bulk container ship across the Atlantic Ocean)(Edwards *et al.*, 2012).

3.5.5. Towards sustainable biomass

Sustainable development was defined by the United Nations (1987) as 'a development that meets the needs of the present without compromising the ability of future generations to meet their own needs'. Sustainability is commonly represented as a set of triangular concepts, with three pillars: economy, environment and society or with a triple-bottom-line: people, planet, profit. Above we have extensively discussed the economic aspect of sustainability of local woody biomass production for Flanders and neighbouring regions. The harvest of additional woody biomass also rises additional questions on the ecological aspect of sustainability. For example, during whole tree harvesting more nutrients are exported from the forest then under conventional harvest as the nutrient concentrations (e.g., nitrogen, phosphorus, base cations) in the crown is much higher than in the logs (Olsson *et al.*, 1996b). Depending on forest and soil type and the studied period, whole tree harvesting sometimes has an impact on the future productivity of a stand (Walmsley *et al.*, 2009; Wall, 2012; Fleming *et al.*, 2014; Olsson *et al.*, 1996b; Phillips & Watmough, 2012). Additional harvest of biomass in forests might also have an impact on biodiversity, on the functioning of associated aquatic ecosystems and on carbon sequestration (Berger *et al.*, 2013; Helmisaari *et al.*, 2014). It is clear that ecosystem impact assessment of additional biomass harvest is a complex issue, with sometimes contrasting results (Riffell *et al.*, 2011).

The revenue of the additional biomass harvest from our experiments turned out to be very small. A larger scale would be needed to reduce harvesting cost of wood chips and to make this process economically more attractive. Within the limited Flemish forestry context this is, however, hard to achieve. With the rising demands, mainly for bioenergy, prices may rise in the near future. However, material use of logs will remain more profitable than chipping of logs, unless the price for (good quality) wood chips raises dramatically. This supports, also from an economic point of view, a cascaded use for biomass giving priority to material application and future reuse and recycling over energy production.

Meanwhile, large amounts of wood pellets are imported, mainly from North-America. In the production and transport process of the imported pellets a higher share of fossil energy is used. From an energy perspective local biomass is preferred, but local sustainable yield is limited. Sustainable harvest of additional biomass from forest ecosystems encompasses more than economic and energy balances and takes into account social and ecological factors. Strong criteria for local and imported biomass are needed to safeguard forest ecosystems from the possible impact of overharvesting on biodiversity and soil fertility, carbon sequestration and other ecosystem services. We believe that more research and a scientifically supported policy is needed for safely implementing additional biomass harvest, independent of the economic feasibility.

3.6. Conclusion

We investigated the technical possibilities and the cost-effectiveness of different harvesting strategies in pine stands in Belgium. These stands include a potentially important source of biomass in the temperate and boreal regions of Europe and North-America. The currently 'conventional' harvest of logs could be expanded by harvesting additional biomass for bioenergy from leftovers. However, we found a very limited economic benefit for harvesting this additional biomass under the current circumstances. The harvesting of logs is much more profitable and should be maximized to obtain the highest profit. This is translated in a small top bucking diameter in clear-cuts and in avoiding whole tree chipping, even in early thinnings. In general, we found that a mobile chipper can achieve better results in cost-effectiveness, energy balance and chip quality than the currently used road-side chipper in clear-cuts. However, the cost-effectiveness of a mobile chipper seems highly dependent on terrain accessibility. Another very important factor in evaluating the cost-effectiveness of the harvesting strategy is the equipment balancing. In our study, poorly coordinated timing of the road-side chipper with the chip transport was the main reason for the lower cost-effectiveness in these strategies. Therefore, an important recommendation is to optimize equipment balancing to reduce harvesting costs and for future studies to control for equipment balancing in the set-up. More studies on the economics of additional biomass harvesting in this, and other regions, will further our understanding on how best to extract woody biomass from forests.

3.7. Acknowledgements

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4. Strong negative impacts of whole tree harvesting in pine stands on poor, sandy soils: a long-term nutrient budget modelling approach

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4.1. Abstract

Global environmental changes such as climate change, overexploitation and human population growth increase the interest in woody biomass from forests as a resource for green energy, chemistry and materials. Whole Tree Harvesting (WTH) can provide additional woody biomass, mainly for bioenergy, by harvesting parts of the crown not harvested under conventional Stem-Only Harvesting (SOH). However, WTH also increases nutrient export, potentially depleting soil nutrients and threatening future stand productivity. Here we assess the impacts of WTH in Corsican pine stands (*Pinus nigra* ssp. *laricio* var. *Corsicana* Loud.) with a rotation period of 48 years on poor, sandy soils in Belgium. We performed a detailed nutrient budget assessment before and after thinnings and clear-cuts under scenarios of WTH and modelled the long-term changes in ecosystem nutrients under both WTH and SOH. Our results demonstrate a strong immediate impact of WTH on nutrient stocks (mainly in clear-cuts). In clear-cuts with WTH, half of the base cations (calcium, potassium, magnesium) in the trees and forest floor were exported. The amount of available cations in the soil is not sufficient to immediately compensate for this export. Only one fourth of the amount exported were available for biota in the top 50 cm of the soil. We also modelled long-term development of ecosystem nutrients (available nutrients in the soil and nutrients in trees and forest floor) and found that the available soil calcium, potassium and phosphorus stocks are insufficiently replenished by deposition and weathering to sustain WTH on the long term. We found no indications of potential depletion of ecosystem cations and phosphorus for the next ten rotation periods under SOH management. Our results thus support a less intensive management in pine stands on poor, sandy soils, for instance, by adopting SOH and/or longer rotation periods.

4.2. Introduction

Enhanced utilization and harvest of whole trees raises questions about the sustainability of this practice and the impact on ecosystem services delivered by forests (Schulze *et al.*, 2012). For example, the additional harvest of biomass in forests on top of the harvest of logs might negatively affect forest biodiversity of saproxylics, small mammals and birds and the functioning of associated aquatic ecosystems by increasing acidifying potential and reducing stream productivity (Berger *et al.*, 2013). Also soil microbial properties and activity and related soil productivity and functioning can be influenced (Smaill *et al.*, 2008b). During Whole Tree Harvesting (WTH) more nutrients are exported from the forest than under Stem-Only Harvesting (SOH) (Achat *et al.*, 2015). The additional export could be significant, despite the lower crown biomass compared to stem biomass, because the nutrient concentrations in these tree parts are much higher than in logs (Neirynck *et al.*, 1998). Jorgensen *et al.* (1975) found that the export of N, P and K under WTH, including the larger roots, was about three times bigger than under SOH in a 16 year old pine plantation. Depending on the forest and soil type, WTH might have a negative impact on the soil fertility of a stand (Olsson *et al.*, 1996a; Jorgensen *et al.*, 1975) and its future productivity (Johnson, 1994; Walmsley *et al.*, 2009; Wall, 2012). A harvesting regime can be considered unsustainable when the ratio between the imports (mainly through deposition and weathering) and exports (mainly through harvest, leaching and run-off) of nutrients is smaller than 0.9, and if the remaining ecosystem nutrient stock is not sufficient for the next ten rotation periods (Gottlein *et al.*, 2011).The ecosystem nutrient stock consists in the nutrients in trees, forest floor and the available soil nutrients (Figure 4.1).

Studying the effects of contrasting harvesting scenarios on soil nutrient development can be performed (1) by empirically comparing pre- and post-harvest nutrient stocks, (2) by modelling the long-term impact or (3) by quantifying growth reductions in the stand. Here we give a short literature overview of different studies on the impact of WTH on nutrient status of forest stands.

A first type of WTH nutrient studies focused on the empirical identification of immediate or long-term effects of harvesting intensity on nutrient stocks. For example, Klockow *et al*. (2013) studied the effect of slash and live-tree retention in *Populus tremuloides* dominated forests in the USA. They found that a lower harvesting intensity (i.e. SOH vs. two intermediate scenarios retaining some slash on the stand vs. WTH) positively influenced the total nutrient stocks of the stand. Most remarkably, this study mentioned a slash retention of almost 50 % under WTH due to harvest losses (Klockow *et al.*, 2013). Olsson *et al.* (1996b) found a significant effect of harvesting intensity

(SOH vs WTH) on base saturation, especially in the litter layer (L, F and H layer), 16 years after harvest in spruce and pine stands in Sweden. (Phillips & Watmough, 2012) found a decrease in available soil stocks of calcium (Ca) and potassium (K), by making a detailed nutrient budget before and after stem-only selection cutting in sugar maple stands (*Acer saccharum* Marsh.) in Ontario, Canada. Jorgensen *et al.* (1975) found a significant decrease in available soil nutrient pools when WTH was applied instead of SOH. Vanguelova *et al.* (2010) found an increase in acidity and a decrease of available soil K and phosphorus (P) stocks under WTH in comparison to SOH in Sitka spruce stands in the UK after 28 years and Smaill *et al.* (2008b) detected a significantly lower biomass and nitrogen content of the litter layer under WTH compared to SOH, 8-16 year after harvest in pine stands in New Zealand. On the other hand, some studies reported little significant differences in nutrient stocks between stands after WTH and SOH. Wall & Hytonen (2011), for example, studied Norway spruce stands 30 years after SOH and WTH, with needles left on site, in Finland. They found no significant differences between the stands in stocks in forest floor and concentration in foliage of nitrogen (N), magnesium (Mg), P, Ca and K. Wilhelm *et al.* (2013) compared nutrient budgets and fluxes before and after harvest for 3 harvesting intensities (WTH and treatments leaving most of the crown in the stand) in oak dominated stands on poor, sandy soils in Wisconsin, USA. Only little differences were detected between the treatments in the first two years after harvest. In general, these empirical studies offer excellent insights into the immediate impact of different harvest regimes and can be used to test results from modelling work. However, this type of studies does not directly evaluate the long-term perspective of possible soil depletion, making it harder to extrapolate the results to longer time frames.

A second type of studies used models to estimate the long-term impact of different harvesting intensities on nutrient stocks. Aherne *et al.* (2012), for instance, modelled the soil nutrient status under different harvesting intensities and under projected climate change scenarios for Scots pine (*Pinus sylvestris*), birch (*Betula pendula*) and Norway spruce (*Picea abies*) on contrasting soils in Finland. According to the model, WTH (with crowns, excluding stumps) in pine stands increased the removal of biomass by only 24 %. Yet, the removal of base cations more than tripled and nitrogen was removed six times more than under SOH. Palviainen & Finér (2012) developed equations to estimate the nutrient content of crowns and stems based on the stand volume for pine, spruce and birch in Fennoscandia. Based on these equations they modelled nutrient exports under SOH and WTH for thinnings and clear-cuts. Generally they found negative nutrient balances under WTH for most nutrients and most tree species. The study of Phillips & Watmough (2012) estimated the long term impact of a stem only selection harvest by starting from an empirical dataset on the impact of harvesting and modelling the nutrient import by weathering and atmospheric depositions and the nutrient export by leaching. They found a net loss and a high long-term risk of depletion for bioavailable K and mainly Ca. Zanchi *et al.* (2014) modelled responses of spruce stands to increased biomass extraction (by residue removal, intensifying thinnings and shortening rotation periods) in southern Sweden. By assuming a fixed harvest loss of 40% of the foliage under all scenarios, they found significant changes in aboveground and belowground stocks and fluxes of carbon. In sum, modelling studies give an interesting overview of impact on a larger space and time scale. Moreover, a well performing model, tested on field data, such as the NuBalM model for nitrogen and biomass pools in pine stands, has the potential of being a useful management tool (Smaill *et al.*, 2011). The drawback is that the data is mostly not empirically generated and sometimes lacking terrain validity, e.g. poorly accounting for the fact that only part of the crowns are exported and that significant harvest losses occur on site.

A third kind of studies directly assessed the impact of different harvesting intensities on future productivity of forest stands. Egnell (2011) found a significant decrease of productivity over 31 years in planted spruce after WTH in northern Sweden. Fleming *et al.* (2014) compared total aboveground biomass 15 years after harvest in pine stands in Ontario, Canada. The aboveground biomass decreased significantly under WTH with removal of the forest floor. Stands under SOH had a higher aboveground biomass than stands under WTH, but this difference was not significant and mainly caused by a higher natural regeneration. Kaarakka *et al.* (2014), found no effect of harvesting intensity on growth of the next generation ten years after clear-cutting in spruce stands in Finland. However, in this study a clear effect of treatment was found on the stocks in mineral soil and litter layer, suggesting that on the longer term WTH could have negative effects on site productivity. Wall & Hytonen (2011) found no decrease in spruce stem volume production between stands under WTH (with crowns left on site for one year after harvest, so that needles were not exported) and SOH in Finland, even after 30 years. However, the total site productivity was higher in the sites where only stems were harvested, because of the higher density of the naturally regenerated seedlings. Ponder *et al.* (2012) compared growth in 45 long-term soil productivity experiments across several climate regions and soil types throughout North-America 10 year after harvest and also found little consistent effects of planted tree biomass in stands under WTH and even under WTH with forest floor removal compared to SOH. Finally, Walmsley *et al.* (2009) found a reduction in diameter at breast height in *Picea sitchensis* stands in the UK 23 years after WTH in comparison to SOH. This type of study yields very interesting results reporting on direct change in ecosystem service delivery. Possible drawbacks are the delay between WTH and final results and the many possible confounding factors that can cause growth differences, other than the management regime (Burger, 1996). For a good understanding of the ecological causes of a possible growth reduction, there is a strong need to combine the growth reduction results with a thorough study of the ecosystem nutrient stocks.

Some of the studies from the three types mentioned above detected a reduced ecosystem nutrient stock and tree growth after WTH. A recent meta-analysis by Achat *et al.* (2015) based on 749 case studies also demonstrated a clear impact of a higher harvesting intensity (removing branches and foliage) on nutrient export leading to reduced available and total nutrient stocks in soils and, subsequently, growth reductions in the short or medium term.

Different management practices have been described to remediate this. The first, most straightforward decision could be to reduce intensity of harvesting. This could be done by adopting SOH, by leaving the foliage in the stand (Wall & Hytonen, 2011; Achat *et al.*, 2015), by exporting only part of the harvestable crown (Klockow *et al.*, 2013) or by switching between WTH and SOH in consecutive rotations. Another possibility is to change the forest management type by adopting a so-called ecological length of rotation, a longer rotation period that gives enough time to natural processes such as weathering and deposition to compensate for the export of nutrients through harvesting (Achat *et al.*, 2015). Another option is to adopt other harvesting systems, such as selection cutting instead of clear-cutting (Phillips & Watmough, 2012). A last method to compensate for the increased nutrient exports is to apply specific fertilization (Brandtberg & Olsson, 2012). N and K fertilization has been put forward to sustain forest growth under WTH in Finland (including stump extraction) (Aherne *et al.*, 2012). However, Smaill *et al.* (2008a) found that the N fertilization effect was strictly additive to the effects of increased organic matter removal and thus that fertilization did not appear to counteract all the effects of additional biomass harvest. Moreover N fertilization leads to a lower pH and a possible increase in leaching of base cations (Ballard, 2000).

As mentioned earlier, impact assessments of additional biomass harvests on ecosystem nutrient stocks and long-term soil fertility are complex and have resulted in contrasting findings (Riffell *et al.*, 2011). Results are dependent on forest stand type, soil type, climate and amount of atmospheric deposition. Therefore, it is important to synthesize relevant knowledge of each geographic area where biomass is extracted and to draw more general conclusions whenever possible (Abbas *et al.*, 2011).

Here we investigate, for the first time, the impact of WTH on nutrient budgets of pine stands on poor, sandy soils in North-Western Europe. This is a highly relevant study system for numerous reasons. The poor sandy soil type we studied is widespread in the region and typically contains low stocks of exchangeable base cations (Neirynck *et al.*, 1998). Moreover the study region has a strong history of acidifying deposition, in contrast with Scandinavia where most studies to date were performed. The total N-deposition levels in Belgium for example were on average 5.3 times higher than N-deposition levels in Sweden in 2013 (data obtained from the European Monitoring and Evaluation Programme database (EMEP; http://www.emep.int)). These high levels of acidifying deposition can result in a strong leaching of base cations, further depleting the available soil stocks (Verstraeten *et al.*, 2012). Hence, our study system represents almost a worst-case scenario. In addition, the demand for renewable energy sources in this densely populated and strongly industrialized region is especially high. Pine stands make up a large part of the forests in this region (e.g. 39 % in Flanders (Waterinckx & Roelandt, 2001) and 33 % in the Netherlands (Dirkse *et al.*, 2007)), especially on the sandy soil types. Hence, currently there is already a very high interest to harvest additional biomass from these stands. Our understanding of the consequences on nutrient budgets, however, is still incomplete.

We performed a detailed inventory of nutrient exports and stocks before and after a thinning and a clear-cut, taking away whole trees. We used these empirical data in a long-term nutrient budget modelling. We thus combined the first and the second type of WTH impact studies described above, building on an empirical basis to maximize ecological understanding and estimate long-term impact. We hypothesized that WTH depletes ecosystem stocks of base cations and possibly phosphorus on the long term under the studied circumstances (short rotation period, poor soils and high acidifying deposition loads) significantly more than SOH. Based on the results, we formulated recommendations for sustainable forest management.

4.3. Methods

4.3.1. Study region

The study was performed in Bosland, where the soils are characteristically dry, sandy and nutrient poor and were classified as Carbic Podzols (IUSS Working Group WRB, 2007). The forest is located at the edge of the Campine plateau, which originated from a mixture of tertiary sands and gravel-rich sands deposited by the Meuse river. During the Pleistocene these sands were covered by aeolian sand deposits. Locally drift sand dunes occurred in the area. The soil consists of very coarse sands with up to 83% (substrate) and 97% (drift sands) of particles with a diameter larger than 50 micrometer. Until the middle of the 19th century, Bosland was mainly covered by an extensive heathland. Afterwards, gradual afforestation with conifers took place with Scots pine (*Pinus* *sylvestris*) and Corsican pine (*Pinus nigra* ssp. l*aricio* var. *Corsicana* Loud.) as dominant tree species. More information on the study area can be found in Chapter two.

4.3.2. Management of pine stands

To develop our harvest scenarios, we interviewed two Bosland forest managers about the standard management of pine stands for wood production. In these stands, pines are left alone for about 30 years after planting or natural regeneration. Then harvester passages are created and a first thinning is executed, taking away about 20% of the total volume. Subsequently, every six or nine years after the previous thinning a large part of the stand increment is taken away by a new thinning (Jansen *et al.*, 1996). The rotation period is classically ended by a clear-cut at a stand age between 40 and 100 years, depending on the management regime. We chose to study a management regime with a relatively short rotation period of 48 years. A shorter rotation period is most suited to optimize biomass production (Dwivedi & Khanna, 2014). The management applied in Bosland, as described above, is comparable with the management of pine stands in other countries (thinned after around 30 years and clear-cut after 40-110 years in Finland (Pussinen *et al.*, 2002); 10-40 years in USA when focussing on biomass production (Dwivedi & Khanna, 2014) and 77 years in Finland when focussing on timber and additional biomass to 82-118 years when carbon storage was adopted as one of the management goals (Pihlainen *et al.*, 2014).

4.3.3. Stand selection

We used the same eight monoculture stands of Corsican pine as for the whole tree harvesting experiment of chapter three. This harvesting was closely monitored and slightly different harvesting practices were used in the different stands to compare efficiency and cost-effectiveness (Chapter three). The stands had a similar size (1.13 ± S.D. 0.22 ha; Table 4.1) and were selected based on their similarity in soil type, tree species and management and were chosen to be representative for the region.

All stands occurred on typically dry to very dry sandy soils and were situated in Overpelt and Lommel. Four stands with a stand age of 33 years were selected in Overpelt (stands O1 to O4; centre of stands 51.21°N, 5.36°E). These stands were originally planted on former heathland in 1922, but destroyed by fire in 1976 and replanted in 1979 with a planting density of 6666 trees per ha. Four older stands with an age of 48 years were selected in Lommel (stands L1 to L4; centre of stands 51.18°N, 5.30°E), also planted on former heathland with the same planting density. The stands in Lommel had been thinned twice. In stands O1-O4 we executed a thinning, stands L1-L4 were clear-cut.

Table 4.1: Stand and soil characteristics of the thinned stands in Overpelt (O1-O4) and the clear-cut stands in Lommel (L1-L4).

	Area (ha)	Year of planting	Standing stock (m³/ha)	Thinning intensity (% of number of trees)	Average soil pH-$H_2$0 (0-50 cm)
O1	1.05	1979	272.5	20.1	4.4
O2	1.00	1979	315.8	24.9	4.3
O3	1.35	1979	327.8	21.2	4.2
O4	1.55	1979	305.0	15.8	4.4
L1	1.15	1965	349.3	Not applicable	4.3
L2	1.17	1965	364.4	Not applicable	4.4
L3	0.89	1965	341.8	Not applicable	4.4
L4	0.92	1965	365.5	Not applicable	4.3

All clear-cuts were performed with a harvester and logs were extracted using a forwarder. The tree-tops were chipped inside the stand with a mobile chipper for stands L3 and L4 and were extracted with a forwarder to a roadside chipper for stands L1 and L2. The top bucking diameter, the diameter at which the logs are separated from the tree tops, was set at 7 cm for stands L2 and L4 and at 12 cm for stands L1 and L3. Three of the stands in Overpelt were thinned by a harvester, stand O4 was thinned by an excavator with a pinching head. In three of the thinned stands, whole trees were chipped: in stand O1 the trees were extracted with help of a forwarder and chipped at the roadside; in stand O2 the trees were chipped in the stand by a mobile chipper; in O4 the trees were extracted by a tractor with a trailer and chipped at the roadside. In stand O3 the logs and tree tops were extracted separately by a forwarder and the tree tops were chipped at the roadside (for more details on the different harvesting practices, see chapter three).

4.3.4. Data collection

In every stand, samples were taken from different ecosystem compartments before and after the harvest. We randomly laid out 3 square plots of 400 m² in every stand in which we measured the diameter of all trees before and in the thinnings also after harvest. The fresh mass of all lying coarse dead wood with a diameter over 5 cm was determined before and after harvest and subsamples were taken. Within every plot, we systematically laid out 5 square subplots of 1 m² in which we collected all fine dead wood (with a diameter under 5 cm) before and immediately after harvest and determined the fresh weight. To avoid the impact of previous sampling, we altered the exact location of the subplots sampled after the harvest from the subplots sampled before the harvest. In the middle of each of the subplots, we took a sample of the mineral soil until 50 cm depth before the harvest, separated in five subsamples of 10 cm layers. Additionally, before and immediately after the harvest, we collected all species present in the understorey (woody and non-woody species) and a 0.25 × 0.25 m sample of the whole litter layer (L F and H layer) in the middle of each subplot. The samples for each soil layer were pooled at the plot level, resulting in five mixed soil samples per plot, one for each 10 cm layer. To quantify the standing stocks of trees, we cut five trees in both regions selected with a stratified sampling design: three trees with an average diameter of the stands were selected, plus one tree having the first and one tree having the third quartile diameter (after Neirynck *et al.* (1998). We randomly selected trees with the desired diameter, keeping a distance of more than 10m from the forest edge. For these trees, the exact height was determined with a measuring tape and stem discs were sampled at 1 m height and of every third meter higher (1 m, 4 m, 7 m, etc.). Of each of these model trees 20 fresh grams of the current needles were sampled to assess the nutrient status of the trees (Rautio *et al.*, 2010). Finally we also collected a sample of the harvested wood chips and pooled ten subsamples of 0.5 dm^3 for each exported chip container. These wood chips consist of crown material (sticks, twigs, bark and needles) for the clear-cuts and for T3. The wood chips of T1, T2 and T4 originate of whole trees and also contain stem wood material. A total of 79 containers were sampled.

4.3.5. Soil and wood chemical analyses

Soil samples were dried at 40°C until a constant weight was obtained and passed through a 1 mm sieve. pH-H2O was measured using a glass electrode (Orion, Orion Europe, Cambridge, England, model 920A) following the procedure described in ISO 10390:1994(E). Total N and C contents were measured by dry combustion using an elemental analyser (Vario MAX CNS, Elementar, Germany). Exchangeable K, Ca, Mg, Na and Al content was measured by atomic absorption spectrophotometry (AA240FS, Fast Sequential AAS) after extraction in $BaCl_2$ (NEN 5738:1996). This method was used as an estimation of the available cation concentrations in the soil. For calculation of effective cation exchange capacity (CEC_e) of the soils, all extracted exchangeable cations (K, Ca, Mg, Na and Al in meq.kg-1) were summed. Total P concentrations (P_{Total}) were measured after complete destruction with HClO4 (65 %), HNO_3 (70 %) and H_2SO4 (98 %) in Teflon bombs for 4 h at 150 °C. Concentrations of P were measured according to the malachite green procedure (Lajtha *et al.*, 1999). Available inorganic soil P within one growing season was measured by extraction in NaHCO3 (Olsen-P according to ISO 11263:1994(E) and colorimetric measurement according to the malachite green procedure (Lajtha *et al*. 1999)). This directly available soil P pool is replenished by the slowly cycling active P pool (Richter *et al.*, 2006), consisting of phosphate that reacted with aluminium (Al^{3+}) and iron (Fe^{3+}). The slowly cycling P pool was calculated based on the relationship: slowly cycling P = Olsen-P × 3.0736. This relationship was revealed from a database of sandy soil measurements of both Olsen-P and slowly cycling P, measured as oxalate-P according to NEN 5776:2006. This database consisted of 68 different soils under grassland and heathland and a very strong relation (linear regression, R^2 = 0,92) was observed.

Samples of wood chips, needles, dead wood, understorey and litter layer were dried at 65°C to constant weight and the dry weight was determined. Subsamples of the coarse and small dead wood and the stem discs were dried at 65°C to constant weight, weighed and ground to particles <0.5 mm (Retsch, SM200). Total N and C concentrations were measured by high temperature combustion using an elemental analyser (Vario MACRO cube CNS, Elementar, Germany). P, K, Ca and Mg concentrations were obtained after digesting 100 mg of sample with 0.4 ml $HClO_4$ (65 %) and 2 ml HNO_3 (70 %) in Teflon bombs for 4 h at 140 ºC. P was measured colorimetrically according to the malachite green procedure (Lajtha *et al*., 1999). Concentrations of K, Ca and Mg were measured by atomic absorption spectrophotometry (AA240FS, Fast Sequential AAS).

4.3.6. Data analysis

4.3.6.1. Differences between stands within locations

To test for differences between the stands and harvest practices within both locations (Lommel and Overpelt) we applied mixed-effect models for each location with *stand* as a fixed effect term and *plot* (and *subplot* nested within *plot*, if applicable) as random-effect terms for each response variable using the nlme package in R 3.0.1 (R Core Team, 2013). The response variables were biomass and nutrient stocks for the different elements of the ecosystem compartments in the forest floor and mineral soil. The standing stock did not differ significantly between stands within one location (Chapter three). Differences in soil characteristics and nutrient pools of the forest floor between the different stands of each location were small (Table 8.3 in Appendix). However, there was significant variation between stands in soil C in deeper soil layers in Lommel and of soil pH and exchangeable Mg stocks in soils in Overpelt. These initial differences might confound results of the impact assessment. Here we expect a limited impact, as the stands within one location were quite uniform in general. Moreover we found no other significant differences between stands after harvest than those present before harvest. The small differences between the harvest practices did thus not affect the nutrient pools and nutrient export. The four stands within each location were therefore considered as replicates.

4.3.6.2. Differences between locations

Since the stands in the two locations had contrasting stand age and density, significant differences existed between the nutrient pools in trees and forest floor e.g., more biomass in the stems and thicker litter layer in the older stands. We thus mainly focused on the soil differences between locations (Lommel and Overpelt)(Table 4.3). As all stands in both locations were classified within the same sandy soil type on the soil map we expected very similar soil conditions, a prerequisite to estimate future ecosystem nutrient stocks with a space for time substitution. To check this hypothesis we first made a pedological description up to 50 cm depth (cf. Davis *et al.* (2004)). The soil profiles were very similar in all stands of both locations and typical for carbic podzols with an obvious E-horizon on sandy parent material. To further test for differences between locations, we applied mixed-effect models with *location* as a fixed effect term and *stand* and *plot,* nested within *stand*, as random-effect terms for different response variables using the *nlme* package in R. The tested response variables were the stock of C, the stock of exchangeable Al and base cations, the CEC_e and the ratio between base cations (Ca, Mg, K) and Al.

We found significantly higher concentrations of exchangeable Al and base cations in the stands located in Lommel, resulting in a much higher CEC_e in the top soil, in comparison with the younger stands in Overpelt ($p < 0.001$). Yet, the CEC_e was strongly correlated with the amount of soil organic material (analysed as % C as measure for % organic material) ($r = 0.94$, $n = 24$, $P < 0.001$). We also found a much higher C content in the older stands located in Lommel. Moreover, the ratio between exchangeable base cations (Ca, K, Mg) and Al was not significantly different between both locations ($p = 0.16$). As soils in both locations had a very similar texture, history and total nutrient stock, it can be expected that a large part of the difference in organic matter content and related CEC_e might disappear with the ageing of the Overpelt stands. In this respect, the Overpelt stands can be considered as a younger version of the Lommel stand.

Studying long-term changes in soil productivity always implies some uncertainties. When using permanent plots, diverging growth patterns due to differing management regimes can easily be confounded by other factors (Burger, 1996). Inappropriate use of space-for-time substitution procedures on the other hand can lead to false conclusions about ecological processes. Space-for-time substitutions procedures remain an important tool for studying temporal dynamics of soil development (Walker *et al.*, 2010) and are most appropriate for studying simple systems following temporally linear trajectories (Walker *et al.*, 2010), such as the pine stands we studied.

4.3.6.3. Impact of harvest on nutrient stocks in trees and forest floor

The volume of the stems that were harvested was obtained from the operator for every stand. To estimate the amount of nutrients in the stems we harvested ten model trees, following studies by Neirynck *et al.* (1998) and Berben *et al.* (1983) in the same area. The nutrient concentrations in the stem discs of the model trees were determined in the lab (§4.3.5). The total nutrient content of the model trees was then estimated based on the nutrient concentration of the stem discs. The total amount of nutrients exported with the stems per stand was then calculated with the nutrient content and volume of the model trees and the harvested volume per stand. The export of crown nutrients in the clear-cuts was calculated with the weight and the nutrient concentration of the wood chips. The export under WTH in the clear-cuts was then calculated by summing the stem export and the crown export. The amount of nutrients that would have been exported under SOH in the clear-cuts was only based on the stem nutrients. In thinning T3 we used the same method as for the clear-cuts both for WTH and SOH. In T1, T2 and T4 we calculated total export under WTH with the weight and the concentration of the whole tree chips. SOH in these thinnings was not actually executed, but we calculated the theoretical export with the volume and the nutrient content of the model trees and with the harvested stem volume per stand (as obtained from the operator).

To estimate the initial standing stock in the clear-cuts we used the estimate of the exported stem and the crown biomass increased with the biomass of the assumed harvest losses. To estimate these harvest losses, we calculated the difference between the biomass of the litter layer and the fine and coarse dead wood before and after the harvest. The amount of nutrients in the crown was then calculated using the estimated crown biomass and the concentrations of the wood chips. For the thinnings, the total stock of the stems before harvest was estimated by dividing the harvested stem stock with the harvesting intensity (Table 3.1). For the crowns, the total stock before harvest was estimated by dividing the sum of the exported crown stock and the assumed harvest losses with the harvesting intensity. The harvest losses were estimated in the same was as for the clear-cuts.

The root biomass and root nutrient stocks were estimated using the ratio of aboveground to belowground biomass and nutrient amount of the trees harvested in the study of Neirynck *et al.* (1998). The nutrient stocks of the understorey, the coarse and fine dead wood and the litter layer were calculated as their nutrient concentration times their dry mass. The amount of nutrients in the mineral soil (0-50 cm) was estimated using the measured nutrient content and the bulk density. The bulk density of each 10 cm layer from a nearby plot of the ICP intensive monitoring network (Level II) plot (less than 3 km away) was used, which is reasonable since the variation in bulk densities in this region is very low (coefficient of variation <5% for every layer from 4 Level II plots in the Campine region).

To evaluate the magnitude and immediate impact of the export by harvest we compared the amount of exported nutrients with the nutrient stocks in the trees and in the forest floor and the available nutrients in soil that together make up the ecosystem nutrient stock (Figure 4.1). Different methods exist to analyse the available nutrient stocks in soils. Here we used the term available soil P for the slow cycling P pool, available soil cations were measured after $BaCl_2$-extraction and the total N pool in the soil was considered as available soil N (see §4.3.5).

4.3.6.4. Nutrient budget modelling

To model the future impact of WTH and SOH on ecosystem nutrient stocks we considered all nutrients included in trees and in the forest floor as bio-available. Moreover, we neglected all the internal fluxes (fine root turnover, growth, litterfall and decomposition), as these do not change the amount of ecosystem nutrients (Figure 4.1).

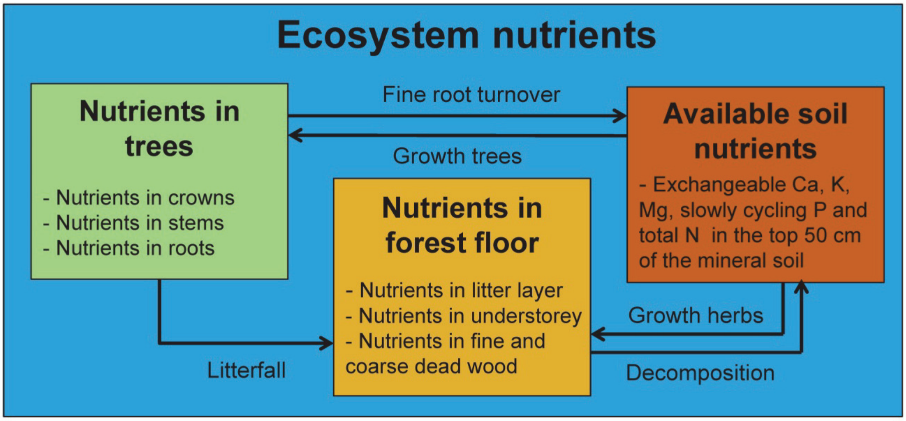

Figure 4.1: Overview of the different stocks of ecosystem nutrients and the internal fluxes between these stocks.

To model the future ecosystem nutrient budgets, we defined a simple standard management scenario with a first thinning creating harvester passages at 33 years, a second thinning at 39 years and a clear-cut at 48 years (based on the interviews with the forest managers, see above). To evaluate the total impact of this management regime, we estimated the stem volume of the thinning at 39 years stand age using the yield table of (Jansen *et al.*, 1996). We determined the yield class based on the tree height and age of the model trees and found that the growth of the stands in Bosland followed the yield curve of the highest yield class (16) for inland Corsican pine (Jansen *et al.*, 1996). Second, we estimated the nutrient concentration of the 39 year old stems as the linear interpolation of the concentration of the stems at 33 and 48 years, assuming that the change in stem nutrient concentration between 33 and 48 years is a linear process. Finally, the export of WTH in the thinning at 39 years was calculated by multiplying the SOH export with the linear interpolation of the ratio between WTH and SOH export from both studied cases (at 33 and at 48 years). The underlying assumption here is that the decreasing biomass of the crown compared to the stem between 33 and 48 years is a linear process. The modelled exports of the future thinnings at 33 and 39 year of stand age and of clear-cuts were kept identical to the current values, and thus independent of the future nutrient budget.

In addition to export by harvest (E_H), other processes also influence the ecosystem nutrient stocks in a stand. These ecosystem nutrient stocks are further depleted by leaching with percolating soil water (E_L) and replenished by weathering of mineral soil (I_W) and deposition (I_D) (Figure 4.2). Nitrogen fixation, run-off and NH_3 volatilization were not included, since they are of minor importance for the pine trees and the sandy, dry soils in our study area (Wilhelm *et al.*, 2013).

We used data on nutrient leaching and deposition from the nearby ICP forests intensive forest monitoring (Level II) plot. This forest is a very similar Corsican pine stand situated in Ravels (51,40°N, 5.05°E ; 30 km from study area) (Verstraeten *et al.*, 2012; Verstraeten *et al.*, 2014). Bulk and through fall depositions of nutrients were measured using rainfall collectors in the open field and the forest stand, respectively. We calculated dry deposition values using the canopy budget model of Ulrich (1983). The canopy budget model simulates the interaction of major ions within forest canopies based on through fall and bulk deposition measurements. The model is used for estimating dry deposition and canopy exchange fluxes in a wide range of forests (Staelens *et al.*, 2008). Leaching of nutrients under the rooting zone was determined by multiplying nutrient concentrations of the soil solution with the amount of the water percolation flux on a depth of 0.75 m. Rates of nutrient deposition and leaching in Flanders have strongly decreased during the past two decennia (Verstraeten *et al.*, 2012). This decrease stabilized; therefore the average value of the last four years has been used for the future deposition and leaching rates as no further decrease is to be expected (Figure 8.1 in Appendix). Deposition and leaching for P were below the detection

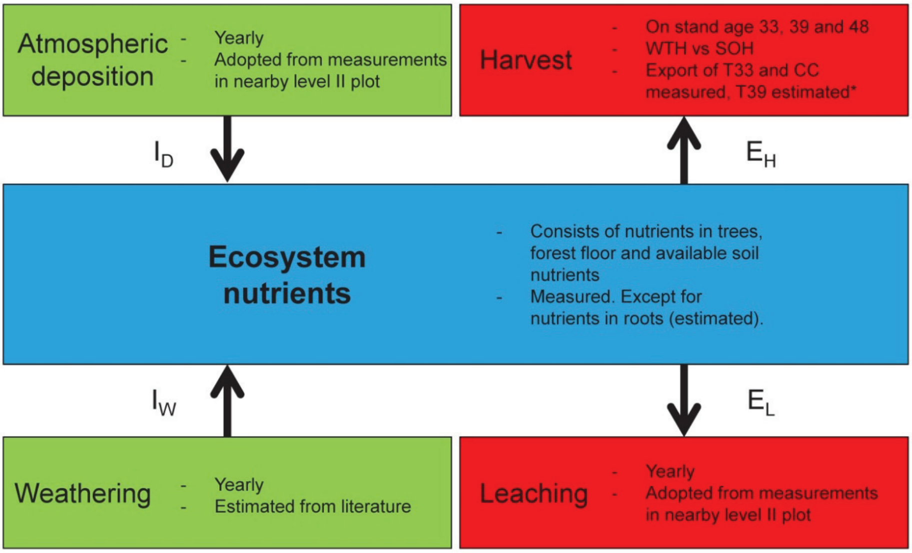

Figure 4.2: Scheme of the nutrient budget modelling approach. The amount of ecosystem nutrients is influenced by the balance of four external fluxes: imports (in green) by atmospheric deposition (I_D) and weathering (I_W) and exports (in red) by Harvest (E_H) and Leaching (E_L). The period and the source of the data is given for each flux.(T33 = thinning at stand age 33, T39 = thinning at stand age 39, CC = clear-cut at stand age 48)*

rate in the level II plots and were neglected in the modelling. Weathering rates were based on a geochemical model applied to sandy soils in the Netherlands with similar characteristics as the soils in the studied area (van der Salm *et al.*, 1999). Weathering for N was considered to be negligible, definitely compared to the high input by deposition. All external fluxes (deposition, leaching, weathering and export) were considered as a constant in our future model (Table 4.2).

Table 4.2: External fluxes of the nutrients used in the modelling (kg.ha^{-1}.yr^{-1}). The data on leaching and deposition was adopted from measurements in a nearby level II plot, data from weathering was obtained from literature.

	Weathering	Deposition	Leaching
Ca	0.27 +- 0.08	4.8 +- 0.6	0.9 +- 0.16
Mg	0.23 +- 0.17	3.5 +- 0.75	0.5 +- 0.13
K	2.57 +- 0.83	2.2 +- 0.56	0.7 +- 0.2
N	0	27.2 +- 0.58	7.8 +- 1.79
P	0.12 +- 0.11	0	0

Future nutrient budgets are modelled by summing the yearly fluxes for weathering, deposition and leaching and the exports of thinnings and clear-cuts. The nutrient budget modelling was executed for a period of 100 years (2011-2111). The situation in the clear-cut stand just before the harvest in 2011 (harvest was in 2012) was adopted. Afterwards thinnings were modelled at a stand age of 33 and 39 and a next clear-cut at a stand age of 48, thus in 2060 and repeated through each subsequent rotation. We also took part of the uncertainty of the model into account based on the best available S.D. of the respective fluxes. For example the S.D. of the amount of ecosystem potassium on time i is calculated as follows: $S.D.(K_i) = \sqrt{S.D.(K_{i-1})^2 + S.D.(I_W K)^2 + S.D.(I_D K)^2 + S.D.(E_L K)^2 + S.D.(E_H K)^2}$ (abbreviations given in caption of Figure 4.2).

4.4. Results

4.4.1. Pre-harvest nutrient status

The soils at both locations were relatively acidic (average pH H_2O 4.33). The amount of base cations in the soil was low, especially in the Overpelt stands (0.27 $meq.kg^{-1}$) compared to Lommel (0.63 $meq.kg^{-1}$). Both soils had a similar ratio of base cations to Al (0.054 $meq.meq^{-1}$). To estimate the pre-harvest nutrients status, the available and total stocks in the soils of both locations were determined (Table 4.3). The available soil stock of base cations and P was relatively small.

Corsican pine is well adapted to these nutrient poor soil conditions, but not to very acidic situations (Hill *et al.*, 1999). To estimate the current nutrient status of the stands, we compared the needle nutrient concentrations to the concentrations described as “low” and “high” in the ICP Forests manual (Rautio *et al.*, 2010). The observed Mg concentrations in both Lommel and Overpelt were below the 5 percentile of the ICP Forests Level II dataset. Also for Ca (mainly in Lommel) and K (in Overpelt) the observed needle concentrations were on the lower side of the plausible interval, suggesting that base cation concentrations at our study sites were close to the lower limit of the species.

4.4.2. Immediate impact of harvest on nutrients in trees and forest floor

In the clear-cut stands L1-L4, stocks in trees and forest floor amounted to 396.4 $ton.ha^{-1}$ before the harvest; whole tree harvesting reduced this to less than half of the initial stock with an export of 206.4 $ton.ha^{-1}$ (Figure 4.3 and Table 8.4 in Appendix). In the thinned stands O1-O4, the initial stock in trees and forest floor was 349.3 ton.ha^{-1} and only 43.5 ton.ha^{-1} was exported. Not all material from stems and crowns was exported: the increase in the litter layer after harvest is predominantly related to harvest losses of needles and small branches from the crowns.

Table 4.3: Available and total nutrient stocks in soils (0-50 cm) of the stands in Overpelt and Lommel before harvest (kg.ha^{-1}). For meaning of available soil Ca, Mg, K, Al and P see main text, soil N and C content was considered as available.

	Overpelt		Lommel	
	Available	Total	Available	Total
Ca	13.9 (5.8)	404.1 (31.2)	65.3 (16.5)	517.7 (388.1)
Mg	5.2 (0.6)	988.5 (70.7)	9.6 (2.7)	894.4 (186.9)
K	31.7 (3.6)	1747.8 (122.1)	32.2 (4.5)	1778.1 (302.1)
Al	350.2 (32.2)	11796.6 (962.4)	704.6 (118.3)	11483 (1540.9)
P	63.8 (20.1)	250.1 (17.9)	87.5 (23.6)	351.3 (46.1)
N	3237.2 (215)		4270.3 (804.1)	
C	40307.2 (3631.1)		80474.3 (19377.6)	

Table 4.4: Needle nutrient concentrations (µg.g^{-1}) in the stands in Overpelt and Lommel before harvest and concentrations for Corsican pine, based on the ICP forest dataset (Rautio *et al.*, 2010). *Nutrients that differed significantly between locations are marked with a * (p <0.05); nutrient concentrations that were below the lower boundary of the ICP values were shaded in red.*

	This study				ICP manual	
	Overpelt		Lommel		5%ile, lower limit	95%ile, upper limit
	Mean	S.D.	Mean	S.D.		
Ca	1631	413	1141	344	970	4420
Mg	520	104	506	116	560	2080
K	4730*	857	7881*	1980	3880	8300
N	14998	1853	16518	2181	8420	21180
P	1021	105	1098	70	810	1570

By only considering the stem export, we estimated the impact of SOH in which crowns are left in the forest stands. In the clear-cut stands the difference in biomass export between WTH and SOH was proportionally small, with an export of 170.5 ton.ha^{-1} under SOH, which is 82 % of the biomass exported under WTH. In the thinned stands, the difference was proportionally larger, with an export of 26.2 ton.ha^{-1} under SOH, which is only 60 % of the biomass exported under WTH. The trees in the younger thinned stands had deeper crowns relative to tree height than the mature pines in the clear-cut stand. In general, when solely looking at the mass of the stocks, a clear-cut had a strong impact on the stocks in trees and forest floor but the extra impact of WTH seemed relatively small.

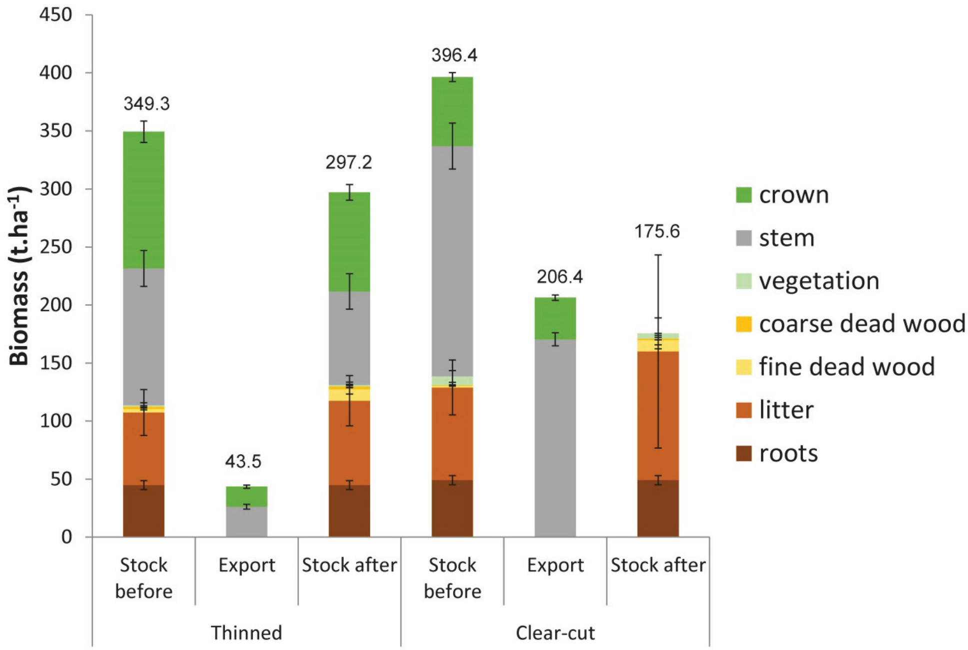

Figure 4.3: Impact of harvest on different compartments of trees and forest floor and export under WTH of the thinned vs. clear-cut stands (error bars mark standard deviations for every compartment).

To evaluate the direct impact on the nutrient stocks in trees and forest floor we looked into the export of the base cations, and N and P (Table 4.5 and Table 8.5 in Appendix). In the thinned stands about 11 % of the base cations in trees and forest floor was exported and about 8 % of N and P. The reduction of the export of nutrients under SOH was quite similar to the reduction of biomass export (variation between 40 % and 70 % of export compared to WTH for different nutrients and 60 % for biomass).

Under clear-cuts, again the heavy impact of WTH on the stocks in trees and forest floor was evident. For base cations, half of the pool in trees and forest floor was exported under WTH, and for N and P one third. This relatively large export of nutrients could easily affect future tree growth and site productivity, when available stocks in the soil are small and/or insufficiently replenished. The export of base cations under WTH in the clear-cuts exceeded the available stock in the top 50 cm of the soil more than fourfold (compared to a little less than threefold under SOH). Under WTH, for P the export in clear-cuts was about equal to the slow cycling soil stock and for N to one fifth of the soil N (compared to 58 % for P and 15 % for N under SOH).

Leaving the crowns in the stand after clear-cut had a significant reduction on impact as on average only 67 % of the base cations, 69 % of the N and 55 % of the P was exported in comparison with WTH, while 82 % of the biomass under WTH was taken away under SOH.

4.4.3. Long-term impact on ecosystem nutrient stocks

The modelling showed that the clear-cut reduced the stocks of all nutrients, both under SOH but more strongly under WTH (Figure 4.4). In the next 33 years, the different stocks become replenished by deposition and weathering while the stand matures until the first thinning. After a modelled second thinning and a new clear-cut in 2060, the first rotation is finished and it is possible to evaluate the evolution of the ecosystem stocks for the different nutrients. The ecosystem stock of Mg is predicted to increase over the rotation period while the ecosystem stock of P and Ca is expected to decrease under both WTH and SOH. Differences between SOH and WTH were most obvious for K and N with a long-term decrease under WTH and an increase under SOH.

Table 4.5: Nutrient stocks in trees and forest floor and export from the thinned and clear-cut stands under whole tree harvesting (kg.ha^{-1}). The stock after harvest under stem-only harvesting was calculated by adding the export of crowns to the stock of the forest floor, as the crowns would stay in the forest floor after stem-only harvesting.

		Stock before		Whole tree harvesting			Stem-only harvesting		
		Trees	Forest floor	Export	Stock after		Export	Stock after	
					Trees	Forest floor		Trees	Forest floor
Thinned	Ca	185.7	134.1	32.3	135.3	146.6	22.7	135.3	156.3
	Mg	57.2	24.7	9.2	42.3	27.7	5.1	42.3	31.8
	K	318.1	83.7	48.1	238.7	83.8	23.7	238.7	108.3
	N	1096.1	1091.7	169.7	822.4	1134.3	97.3	822.4	1206.7
	P	40.2	44.3	6.1	30.0	44.1	2.5	30.0	47.7
Clear-cut	Ca	340.2	208.1	238.2	32.7	257.4	176.3	32.7	319.3
	Mg	63.0	32.0	42.0	7.0	42.6	27.6	7.0	57.0
	K	268.0	98.5	169.3	38.1	108.4	102.1	38.1	175.5
	N	1398.9	1531.0	903.9	211.2	1689.7	627.1	211.2	1966.5
	P	46.9	54.7	30.1	5.2	64.3	16.6	5.2	77.8

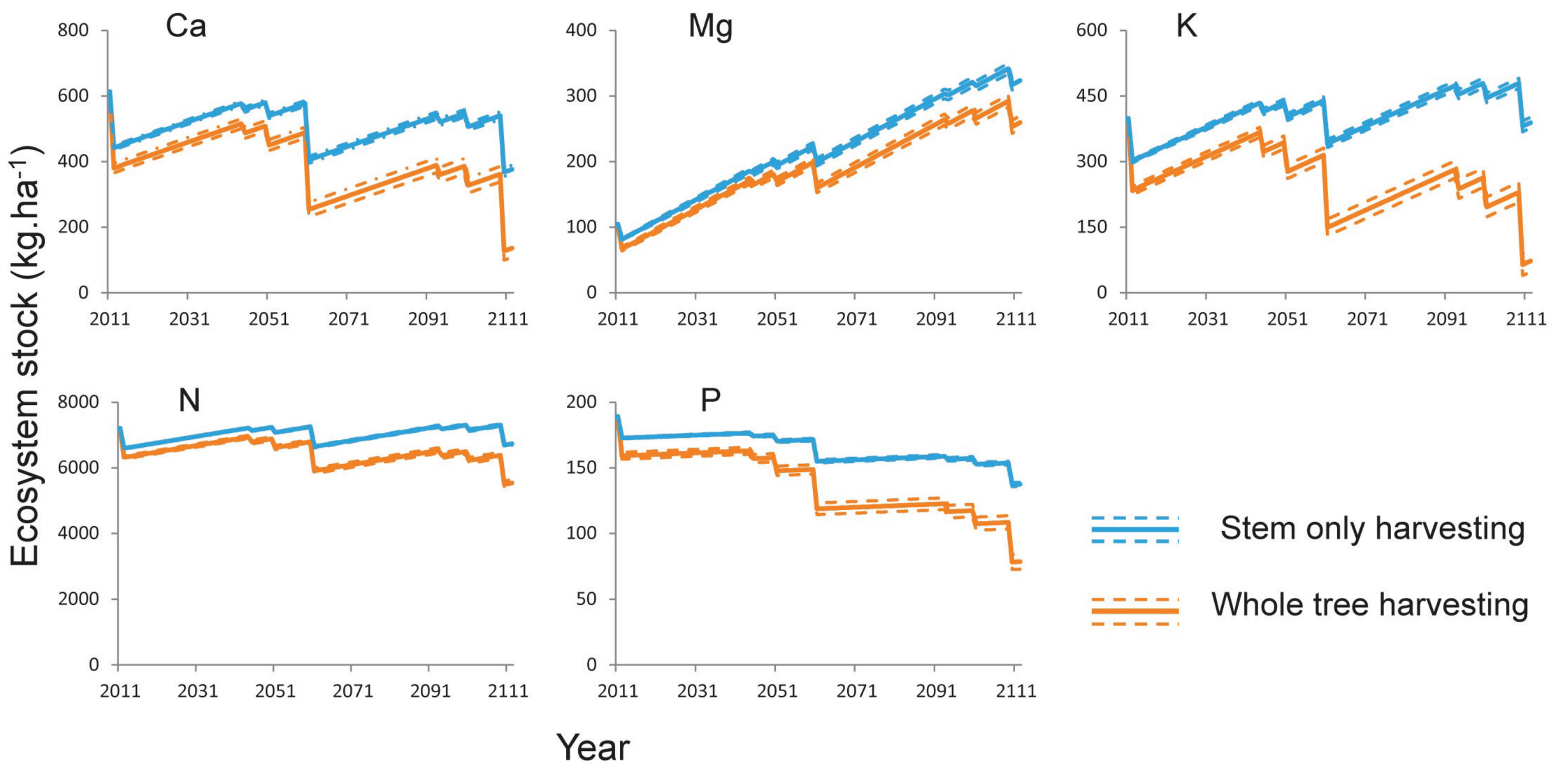

Figure 4.4: Modelled temporal development of the ecosystem nutrient stocks in Corsican pine stands on poor, sandy soils under a 48-year rotation period with two thinnings at 33 and 39 years of stand age and a clear-cut at 48 years of stand age under stem-only or whole tree harvesting, based on the data from the experiment. Dashed lines denote S.D. based on current uncertainties in the fluxes.

4.5. Discussion

4.5.1. Nutrient exports

It is clear that, except for Mg an N, the stocks in the soil were insufficient to sustain the same growth levels under WTH (and SOH) if not sufficiently replenished by deposition and weathering. Compared to the literature, the differences between WTH and SOH were much smaller than described for pine stands in Finland (Aherne *et al.*, 2012). In their study, the export of cations under WTH was more than three times higher than with SOH, while we found a ratio of 1.5. This difference was probably due to the fact that the models used in Aherne *et al.* (2012) did not take harvest losses into account. We found harvest losses of 40 % of the crown in the clear-cuts and 46 % in thinning. These harvest losses probably contained more twigs and needles and thus represented an even larger share of the nutrients. When comparing our results to the findings of Palviainen & Finér (2012) we found higher ranges for export of most nutrients for SOH, probably because of the higher productivity under a Belgian climate compared to the situation in Fennoscandia. We found strikingly higher N exports in our study. This difference was most likely related to the high N availability in Belgium, with a history of very much higher N deposition rates than in Fennoscandia (see Waldner *et al.* (2014)). The modelled export through WTH of Palviainen & Finér (2012) was within the same range as in our study. The difference between WTH and SOH in this study was thus again larger than the observed difference in our study. This might be explained because the harvest losses were not included in the modelling study of Palviainen & Finér (2012).

The impact of thinnings was smaller and less drastic than the impact of clear-cuts, with only ca. 12 % of the base cations and 8 % of the N and P stock in trees and forest floors exported under WTH (6 % and 4 % respectively under SOH). Nonetheless, export of base cations under WTH (with only 20 % of the trees removed) equalled the available soil stock for base cations in soil (see also Palviainen & Finér (2012)).

4.5.2. Future modelling and impact on long-term soil fertility

4.5.2.1. Long-term impact

The modelled long-term changes in ecosystem nutrient stocks varied greatly among nutrients and treatments. For example, ecosystem Mg stocks tend to strongly increase, while P and Ca stocks always decreased. On the other hand, ecosystem K and P stocks only increased under SOH. When ecosystem nutrient stocks are decreasing, there is a risk of a shortage on the shorter or longer term. In these cases it is important to estimate the possible impact and time-frame and to evaluate if current harvesting regime could be continued. Göttlein *et al.* (2007) defined a harvesting regime as “problematic” when the ratio between the imports (mainly through deposition and weathering) and exports (mainly through harvest, leaching and run-off) of nutrients is smaller than 0.9, and if the remaining ecosystem nutrient stock is not sufficient for the next ten rotation periods. Under WTH, the ratio between imports and exports was smaller than 0.9 for each nutrient except Mg, indicating a possible significant decrease in stock (Göttlein *et al.*, 2007) . For Ca, K and P, the current ecosystem stock was only sufficient to support four future rotation periods under the current circumstances. The ratio of N import/export is also smaller than 0.9 for WTH, but the current ecosystem stock is sufficient to sustain 16 more rotation periods under current circumstances. Under SOH, the ratio import/export is only smaller than 0.9 for Ca and P. However, current ecosystem stocks suffice for fourteen and ten future rotation periods respectively, making the situation less critical than under WTH. These results largely coincide with the findings of (Palviainen & Finér, 2012), who also found deficiencies of P, K and Ca under WTH for pine or spruce stands. In addition they also found shortages of N for spruce and birch stands. As mentioned earlier, the deposition of N in Belgium is and has been larger than for Fennoscandia, resulting in a build-up of N in the forest floor and in soils in the former region. Under a system of SOH, however, Palviainen & Finér (2012) did not detect a decrease in ecosystem nutrients, except for P and K under some circumstances. Hence, our results stress the strong negative impacts of WTH on ecosystem stocks of Ca, K, and P and the possible drawbacks on future productivity.

The modelled increase of Mg in time is somewhat contradictory to the low levels of ecosystem Mg in soils and to the low levels in the needles, indicating a possible deficit. One possible explanation could be that the weathering rate for Mg is an overestimate. Another explanation could be that the current Mg status reflects the situation of the previous decades with even higher acidifying deposition and leaching of base cations, such as Mg (Figure 8.1 in Appendix).

4.5.2.2. Uncertainties

For the modelling we used the best available data and methodology, but some uncertainties and assumptions were inevitable, as described in the methods section. One of the most important assumptions was that the standard deviations of the data on the fluxes reflect the uncertainty of the fluxes. Determining this uncertainty in budget closure, including external fluxes such as weathering, leaching and deposition remains very challenging (Yanai *et al.*, 2012). Another uncertainty is the transfer from data of nearby stands to our study area. However, these stands were very nearby and very similar, which should limit this spatial variability. Extrapolating the results to other pine stands, other regions and other stand types implies higher uncertainty.

Moreover, we only considered the top 50 cm of the soil, while most trees might root deeper and can use available soil nutrients from deeper layers. However, we found a sharp decrease in available soil nutrients with depth and Cermak *et al.* (1998) demonstrated a paraboloid root architecture for pine trees (with a decreasing amount of roots with depth). Based on these arguments we believe that the uptake below 50 cm is very limited.

When extrapolating current fluxes to future situations, not only the current variation in fluxes but also possible future changes may need to be taken into account. Yet, these estimates are extremely difficult to quantify and were thus not included in our simple model. For example, new technologies might cause harvest losses to decrease and exports to increase. Increasing tree growth (McMahon *et al.*, 2010; Pretzsch *et al.*, 2014) under influence of a changing climate or decreasing tree growth under decreasing available nutrient stocks in soil could also influence exports. New legislations or expansion of agriculture and industry might cause a decrease or an increase in N deposition rates, respectively, which is directly linked to changes in leaching rates. In turn, weathering rates can be affected by climate change (Sverdrup & Warfvinge, 1993). As another example, it has also been demonstrated that fluxes could be influenced by the event of harvesting itself, for example increased nitrogen leaching after harvest (Devine *et al.*, 2012). Thus, the modelling result after 100 years is an indication of the evolution in ecosystem stocks when continuing on the current management path rather than a precise prediction of the ecosystem stocks in each year.

4.5.3. Other sustainability issues

Apart from soil nutrient depletion, intensified forest management with short rotation periods and WTH cause other sustainability issues. Also soil microbial properties and activity and related soil productivity and functioning can be influenced (Smaill *et al.*, 2008b). From an economic point of view, we demonstrated already in Chapter three that WTH is hardly profitable in this region under current market conditions. Moreover there are different studies that demonstrate a negative impact of WTH on biodiversity of saproxylics, small mammals and birds (Berger *et al.*, 2013). Other studies challenge the idea of bioenergy from forestry biomass as a carbon neutral alternative (Schulze *et al.*, 2012). These issues are beyond the scope of the current study, but should also be kept in mind when applying WTH.

4.5.4. Management recommendations

According to our long-term modelling, poor, sandy soils cannot sustain a WTH system of Corsican pine in this region without intervention. Based on our data, we thus recommend to apply SOH, under the current circumstances to reduce impacts on soil fertility. In addition, longer rotation periods can lower the impact on available soil nutrient stocks (Zanchi *et al.*, 2014; Achat *et al.*, 2015). Older trees have slower growth rates and a larger stem to crown ratio, thereby reducing export of base cations and nutrients per unit of time with harvesting. Moreover under longer rotation periods leaching and deposition can more sufficiently replenish ecosystem nutrient stocks (Achat *et al.*, 2015). Currently most stands in Bosland are managed under longer rotation periods and thus with a less narrow focus on production. Under these longer rotation periods, WTH might be considered in some thinnings or clear-cuts, for example, once every three to four rotation periods. Another measure to reduce nutrient export with WTH is to leave the crowns in the stand for one year such that the majority of the needles are shed before the crowns are exported (Wall & Hytonen, 2011). This is also beneficial for the energy content due to lower loss in dry mass in comparison with drying at the terminal (Edwards *et al.*, 2012). In the near future, about half of the pine stands in Bosland will be transformed to native broadleaf species such as oak and birch (Moonen *et al.*, 2011). This conversion will cause the nutrient fertility and the cycling of nutrients to change. Recently, Augusto *et al.* (2015) reviewed scientific literature to compare effects of evergreen gymnosperms and deciduous angiosperms on ecosystem functioning. When converting coniferous to broadleaved stands there will be a decrease in inputs of potentially acidifying atmospheric depositions Augusto *et al.* (2015). Under high levels of atmospheric deposition, this will also lead to a decrease in leaching of base cations (De Schrijver *et al.,* 2012). Augusto *et al*.

(2015) found that conversion to broadleaved stands could result in a slight increase of pH and base saturation of the soil. Concerning tree growth, it was found that a conversion to broadleaved trees could slightly decrease biomass production. On the other hand, broadleaved trees demand more nutrients (Augusto *et al*., 2015). Most of these nutrients, however, are in ephemeral tree parts (mostly leafs) and are quickly recycled within the ecosystem (Augusto *et al.,* 2015). Consequently, concerning whole tree harvesting in deciduous stands in winter, a decrease of nutrient export could be expected compared to coniferous trees, because leaves will be shed.

In general, a conversion of coniferous to broadleaved stands is thus expected to increase the rate of nutrient cycling and also the soil fertility, while the export of nutrients under whole tree harvesting could slightly decrease . It can thus be expected that this type of forest conversion will have positive effects on soil fertility (De Schrijver *et al.,* 2002). It has also been stated that the conversion to broadleaf stands is best executed gradually to limit disturbance of the forest microclimate; shelter cutting has been proposed as a good management practice (De Schrijver *et al.,* 2002). However, many knowledge gaps remain and there are also big differences between different deciduous and coniferous species (Augusto *et al.*, 2015). It would be definitely highly interesting to execute a similar study in a mixed broadleaf stand in Bosland and to compare the results of the different nutrient stocks with the current study.

This new management context for these stands also opens up possibilities for different sylvicultural systems, such as selective cutting instead of clear-cutting with possibly less profound implications on nutrient cycling (Phillips & Watmough, 2012). Apart from reducing the export of nutrients, one could also compensate nutrient exports through fertilization to sustain WTH and short rotation periods. However, a well-balanced (different element concentrations in relation to local shortages), slowly releasing and stand-wide application would be necessary to avoid an increase in leaching and a possible shift in soil biota and vegetation (Hedwall *et al.*, 2014). Some past studies also demonstrated that fertilization cannot replace nutrient loss from greater harvest exports and leads to a lower pH (Smaill *et al.*, 2008a; Ballard, 2000). Moreover, it is very difficult to predict the specific fertilization requirements without thoroughly screening soil or needle nutrient levels and fertilization is expensive (Eisenbies *et al.*, 2009). It is thus very questionable if WTH including this remediation measure could be cost-efficient in the Flemish forest context, given the current small margin of profit (see Chapter three).

4.6. Conclusions

Our results reveal a strong negative impact of WTH on ecosystem nutrient stocks, definitely for clear-cuts. According to our knowledge of the fluxes that influence the available nutrient stocks in the sandy soils in our study area, an intense harvesting regime with WTH cannot be sustained. Shortages of Ca, K and P will most likely occur, decreasing soil fertility and reducing tree growth. The uncertainty associated with ecosystem future stocks adds to the conclusion that a less intensive system with longer rotation periods and (mostly) SOH is more suitable for pine stands on poor sandy soils. This study also highlighted the limited scientific knowledge available on important processes, such as mineral weathering. More research on site-specific fluxes and stocks is therefore needed before large-scale WTH is considered.

4.7. Acknowledgements

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5. Spatially combining wood production and recreation with biodiversity conservation

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5.1. Abstract

Pine plantations established on former heathland are common throughout Western-Europe and North-America. Such areas can continue to support high biodiversity values of the former heathlands in the more open areas, while simultaneously delivering ecosystem services such as wood production and recreation in the forested areas. Spatially optimizing wood harvest and recreation without threatening the biodiversity values, however, is challenging. Demand for woody biomass is increasing but other pressures on biodiversity including climate change, habitat fragmentation and air pollution are intensifying too. However, strategies to spatially optimize different ecosystem services with biodiversity conservation are still underexplored in research literature. Here we explore optimization scenarios for advancing ecosystem stewardship in a pine plantation in Belgium. Point observations of seven key indicator species were used to estimate habitat suitability using generalized linear models. Based on the habitat suitability and species' characteristics, the spatially explicit conservation value of different forested and open patches was determined with the help a spatially-explicit conservation planning tool. Recreational pressure was quantified by interviewing forest managers and with automated trail counters. The impact of wood production and recreation on the conservation of the indicator species was evaluated. We found trade-offs between biodiversity conservation and both wood production and recreation, but were able to present a final scenario that combines biodiversity conservation with a restricted impact on both services. This case study illustrates that innovative forest management planning can achieve better integration of the delivery of different forest ecosystem services such as wood production and recreation with biodiversity conservation.

5.2. Introduction

5.2.1. Pine plantations on heathland

Since the 19th century heathland has been converted to pine plantations in order to increase wood production and economic profit of these areas in both Europe and North-America, (Foster *et al.,* 2002; Bertoncelj & Dolman 2013a; Bieling *et al.,* 2013; Moran-Ordonez *et al*., 2013). However this has often led to a loss of fauna and flora associated with grassland, heathland and sandy habitats (GHS species)(Andres & Ojeda 2002; Bertoncelj & Dolman 2013a; Farren *et al*., 2010). Heathland is now considered a rare and threatened habitat which is eligible for protection under e.g. the European Habitat Directive (Walker *et al*., 2004). The resulting landscape type is widespread throughout Europe and North-America, combines open and closed habitats and holds important values for biodiversity conservation, wood production and recreation.

To restore biodiversity values in these pine-heathland systems many efforts have focused on re-converting plantations and restoring heathland (Eycott *et al*., 2006; De Valck *et al*., 2014). Nevertheless, while this can be a valuable and practical strategy in terms of biodiversity of the GHS species (Walker *et al*., 2004), recovery can be slow and results in a loss of other species that are linked to aggrading (pine) forests (Ozanne *et al*., 2000; Burton 2007). Moreover, re-conversion to heathland is not always possible and desirable for all stakeholders, because forest plantations also offer other key ecosystem services such as wood production, soil protection, water regulation and recreation (Zipper *et al*., 2011; Vihervaara *et al*., 2012; Jacobs *et al*., 2013; De Valck *et al*., 2014). The demand for woody biomass for example is high and rapidly increasing (Mantau *et al*., 2010) and forests in densely populated regions such as Flanders face a very high recreational demand (Hermy *et al*., 2008).

While trade-offs between pine plantations and GHS species conservation are obvious, it has often been overlooked that benefits could be non-exclusive (Bertoncelj & Dolman 2013a). Viable populations of some GHS species persist in the pine plantation matrix, thanks to the network of temporal (e.g. clear-cut areas) and permanent open patches (e.g. remnant heathland, forest rides)(Bertoncelj & Dolman 2013a; Pedley *et al*., 2013). In addition to these GHS species, also typical species of pine forests are hosted in these landscapes. These forest specialists, such as forest carabid beetles, are often negatively affected by increasing open areas (Barbaro *et al*., 2005; 2007). Hence, we argue that forest management in these systems, with a focus on wood harvest and recreation, definitely has certain trade-offs with biodiversity conservation. However, benefits of wood harvest and recreation could be non-exclusive, also leading to some synergies with biodiversity conservation. Additional quantitative data could help to further unravel the relation between the services of plantation forests and biodiversity conservation.

5.2.2. Recreation and biodiversity

There is a general consensus that recreation can have a direct negative impact on biodiversity (Steven *et al*., 2011), mainly by altering the ability of animals to exploit resources (Gill 2007). However effects of recreation vary across ecosystems, species, recreation forms and intensity levels (Liddle 1996; Ficetola *et al*., 2007). Some species groups are specifically vulnerable, such as ground-breeding birds (Mallord *et al*., 2007), ground-dwelling forest birds (Thompson 2015) and large mammals (George & Crooks 2006). Impact of recreation on ground dwelling arthropods is generally low (Zolotarev & Belskaya 2015), but butterflies were reported to be directly, negatively influenced by recreation (Bennett *et al*., 2013). There are also varying approaches to estimate the impact of recreation on species with divergent results (Gill 2007) and the relationship between the amount of recreational use and recreational impact is not always (curvi)linear (Monz *et al*., 2013). Mallord *et al*. (2007) found a clear negative effect of disturbance on the density of woodlarks in heathlands (*Lullula arborea*). George & Crooks (2006) found a lower density of large mammals along paths with more visitors in an urban nature reserve dominated by shrubs and open oak forests. Thompson (2015) underlines the need for trail-free refuge habitat for forest birds in deciduous forests. These examples show that there can be a strong impact of recreation on different species in different habitats. However, for the local context of our study area (pine plantations on former heathland), there is hardly any literature to be found. Only for the ‘flagship’ bird species, European nightjars (*Caprimulgus europaeus*), strong negative effects of visitors on nightjar populations were identified (Langston *et al*., 2007; Lowe *et al*., 2014).

To reduce the impact of recreation on biodiversity, a trail network can be designed to guide recreationists to spatiotemporally separate visitors from vulnerable species (Ferrarini *et al*., 2008). Standard trail design is already used to avoid vulnerable areas and to screen sensitive species from disturbance by recreation, but is sometimes too general for optimal results (Rodriguez-Prieto *et al*., 2014). A better way to design trails is based on empirical research (Fernandez-Juricic *et al*., 2007) and by the use of simulation models (Stillman & Goss-Custard 2010) to tailor the trail design to best fit the local context. However, Ficetola *et al*. (2007) and Rodriguez-Prieto *et al.* (2014) demonstrated that an appropriate design for one focal species is not necessarily appropriate for another species. Subsequently, adopting a multi-taxa approach might promote intelligent trail design to limit disturbance for a whole set of species. An example of such an intelligent steering of recreation pressure are the differing access rules for different users (walkers vs walkers with dogs vs boating activities) in the protection of bird colonies as proposed by Fernandez-Juricic *et al.* (2007).

5.2.3. Wood production and biodiversity

Wood harvest from clear-cuts can have a direct negative influence on forest species (Linden & Roloff 2013). Species dependent on shade, dead wood, old trees and cavities, such as shade-demanding woodland herbs, woodpeckers and saproxylic beetles are most vulnerable (Martin & Eadie 1999; Djupström *et al*., 2012). Clear-cuts also have a drastic influence on microclimatic environmental and biological conditions such as light, temperature and availability of food and shelter. However, species that are suited to more open conditions will use intensively managed forests and open, clear-cut areas as new valuable habitats (Bertoncelj & Dolman 2013b; Morris *et al*., 2013; Reidy *et al*., 2014). At landscape scale, the patchwork of open patches in a forest matrix can sustain viable metapopulations of GHS species. However, the success of these metapopulations will depend on the spatiotemporal lay-out of the clear-cuts and the dispersal capacity of the species (Johst *et al*., 2011).

Most programs to conserve forest biodiversity focus on setting aside protected areas and creating forest reserves (Lindenmayer *et al*., 2006). It has been stated that forest reserves alone are not enough because they generally only cover a limited area and are often isolated from each other (Daily *et al*., 2001; Lindenmayer, Franklin & Fischer 2006; Mönkkönen *et al*., 2014). Another biodiversity conservation measure is retaining mature forest habitat elements on clear-cuts, such as green trees or snags, to reduce the negative impacts of wood harvest in clear-cuts (Söderström 2009; Linden & Roloff 2013). Recent findings highlight that a green tree retention level of at least 10 - 15 % of all standing trees on large areas is needed to obtain a strong conservation effect on most forest bird species (Söderström 2009). This contrasts with current retention levels which are often around 2 % (Söderström 2009). Installing protected areas and retaining habitat elements could definitely be part of an effective forest biodiversity conservation strategy, but at the same time it is important to create structural diversity on different scales and to increase habitat connectivity for different species (Lindenmayer, Franklin & Fischer 2006; Brockerhoff *et al*., 2008; Gustafsson & Perhans 2010).

For the protection of the vulnerable GHS species, conservation managers often create permanent open patches with grassland, heathland or sand dunes (Walker *et al*., 2004). Another classical conservation measure, also to increase habitat connectivity, is the broadening of forest roads (Bertoncelj & Dolman 2013a).

All of the above mentioned conservation strategies are valuable, but all have a clear trade-off with wood production. Management scenarios that optimize spatial design of temporal open patches to sustain metapopulations of both GHS and forest dwelling species are less conventional. However, these innovative methods could be highly effective (definitely when combined with classical conservation strategies), while more or less safeguarding the important wood and biomass production function of forests (Mönkkönen *et al*., 2011; 2014).

5.2.4. Management challenges

Forest managers have the challenging task to balance management between biodiversity conservation, wood production and recreation among other ecosystem services. Classic land sparing approaches, such as setting aside protected area are well known. Under land sparing, biodiversity conservation is spatially separated from production and other services, while under land sharing both goals are integrated on the same land (Phalan *et al.*, 2011). Land sharing can be a very valuable conservation strategy if benefits of ecosystem services and biodiversity conservation are non-exclusive (Phalan *et al.*, 2011). However, information and data on innovative land sharing approaches in forests, combining these three management goals and optimizing spatio-temporal synergies, are lacking. Moreover, land sparing and land sharing are often treated as alternative strategies (Phalan *et al.*, 2011) but a combination of both approaches would likely be the most successful strategy since different actions benefit different species and ecosystem services (Rey Benayas & Bullock, 2012). We thus set out to investigate the trade-offs between wood production, recreation and biodiversity conservation in a pine plantation on former heathland and explored possible scenarios for improvement. We gathered empirical data for the different services and used spatially explicit analyses to study the synergies and trade-offs between biodiversity and the two ecosystem services. Based on the analyses we formulate future management and recreation scenarios with their impact on biodiversity for our case study area. We complement contemporary management approaches to advance smart(er) ecosystem stewardship that can both benefit policy makers and practitioners beyond our study area.

5.3. Material & Methods

5.3.1. Study area

The study was performed in north-eastern Belgium in Bosland (center of the study region: 51.17°N, 5.34°E). Bosland is a statutory partnership of four public owners and two non-profit organizations that used to work next to each other, but now closely collaborate to increase the impact and coherence of the management in their forest and nature areas (chapter two). Bosland covers a total surface area of 22 000 ha of which approximately 35% is forest, 7% heathland and 3% grassland. The soils are characteristically dry, sandy and nutrient poor and were classified as Carbic Podzols (IUSS Working Group WRB 2007). Until the middle of the 19th century, Bosland was mainly covered by an extensive heathland. Afterwards, gradual afforestation with conifers took place with Scots pine (*Pinus sylvestris*) and Corsican pine (*Pinus nigra* ssp. l*aricio* var. *Corsicana* Loud.) as dominant tree species (chapter 2). For this study we delimited a study area of 1347 ha in the heart of Bosland, commonly known as Pijnven and Slijkven. The study area is covered by a matrix of pine plantations and has traditionally been managed for wood production under a simple harvest regime including some thinnings (from 30 years stand age, ca each 6 – 9 years) and a final clear-cut after ca. 50 - 100 years. For biodiversity purposes certain areas have been set aside as forest reserves (26 ha) and as permanent open patches (77 ha). The forest matrix is interlaced with a network of forest rides that are both used for recreation (mostly walking, but also cycling and horseback riding) and for wood harvest and can also be a valuable habitat for the GHS species. The study area is also split up in two zones, one with a high recreational pressure (792 ha) and the other with a low recreational pressure (555 ha), without marked tracks.

5.3.2. Data collection

5.3.2.1. Biodiversity

We chose to use an indicator species approach to monitor the biodiversity of the study area. We organized a brain-storm session with the local platform on fauna and flora, formally grouping people that work on management and research of species in the area with volunteers of local nature conservation organizations, often involved in monitoring (chapter two). We asked them to make a credible selection of indicator species that are locally relevant. Indicator species needed to be medium widespread within the study area and reasonably detectable. To increase the ecological relevance we asked for a large range in species' habitat preference, mobility and home range. Ten indicator species were selected for the in-depth study, but only seven species were used in the analyses (Table 5.1), three other species (i.e., *Coronella austriaca*, *Formica spec.* and *Genista pilosa*), were removed from the analysis, because the total number of observations was below ten. The final indicator species pool consisted of two forest species (crested tit and coal tit; *Lophophanes cristatus* and *Periparus ater*), three GHS species (grayling, small heath and northern dune tiger beetle; *Hipparchia semele, Coenonympha pamphilus* and *Cicindela hybrida*) and two species that depend both on forest and open habitats (nightjar and common lizard; *Caprimulgus europaeus* and *Zootica vivipara*). A literature review was performed to double check the habitat preference and the species mobility (Table 5.1).

Table 5.1: Characteristics of the indicator species. A after Lens & Dhondt (1994)*; B after* Brotons (2000)*; C after* Sharps *et al.* (2015)*; D after* Clobert *et al.* (1994)*; E after* Simon-Reising *et al.* (1996) *; F after* Maes & Bonte (2006) *; G after* Cormont *et al.* (2011) *; H after* De Vos *et al.* (2004)*; I after* Jooris *et al.* (2012)*; J after* Desender *et al.* (2008)*; K after* Maes *et al.* (2011).

Common name	Scientific name	Class - order	Total no. of observa tions	Habitat preference	Dispersion distance (m)	Protection status (red list)
Crested tit	*Lophophane s cristatus*	Aves - Passeriformes	227	Forest	2000 [A]	Least concern [H]
Coal tit	*Periparus ater*	Aves - Passeriformes	145	Forest	370 [B]	Least concern [H]
Nightjar	*Caprimulgus europaeus*	Aves - Caprimulgifor mes	145	Forest/ Heathland	747 [C]	Vulnerable [H]
Common lizard	*Zootoca vivipara*	Reptilia - Squamata	14	Forest/ Heathland	30 [D]	Least concern/Near threatened [I]
Northern dune tiger beetle	*Cicindela hybrida*	Insecta - Coleoptera	32	Heathland/ Sand dune	40 [E]	Near threatened [J]
Grayling	*Hipparchia semele*	Insecta - Lepidoptera	52	Heathland/ Grassland	150 [F]	Endangered [K]
Small heath	*Coenonymp ha pamphilus*	Insecta - Lepidoptera	206	Grassland	150 [G]	Least concern [K]

An inventory of the butterflies and the tiger beetle was made three times along transects on the forest rides through the study area in June and August of 2013 and 2014 by bicycle or on foot (Fig. 1A). The insect inventory was done between 10 am and 4 pm and only on sunny days, binoculars were used for easier determination from a distance. The exact GPS location of each observation of an individual was registered. The crested tit and coal tit were also inventoried by walking the observation transects three times, in April 2014. The bird inventory was executed between sunrise and 11 am on non-rainy days only. The observations were auditory (recognition of vocal sounds) and the exact location was not determined, but a stand was marked as occupied or not. We alternated the direction of the transects between days to compensate for a possible time effect (e.g. highest bird activity just after sunrise). The nightjar inventory was based on the sound of its churring song on one warm summer evening (July 10th 2014) with the help of no less than 60 volunteers spread over the entire study area. Each churring individual was marked on a map. The distribution of common lizards was assessed based on presence under black corrugated sheets that served as artificial refuges (Busby & Parmelee 1996). Eighty of these sheets were laid out across the entire study area, left for one year and checked for presence of lizards three times in August 2014. Finally the rough data for the biodiversity inventory were compiled in a map with 821 point observations of the seven species (Figure 5.1A).

5.3.2.2. Recreation

Bosland is a very important touristic destination with more than 250 000 yearly arrivals and more than one million yearly overnight stays. To determine the spatial distribution of the numerous visitors we compiled quantitative visitor data with questionnaires and automated trail counters. We started with interviewing the forest managers about the number of visitors on different road segments. We used a map with all roads and tracks and asked them to mark them with five different colors based on the relative recreational intensity. We then made up a relative recreational intensity map with an average score from the interviews. Then we installed six automated infrared trail counters (TRAFx research ltd, Canmore, Alberta, Canada) to quantify the exact number of visitors. The location of the trail counters was decided in consultation with the forest guards and with the goal to survey varying recreation intensities. We only had the counters available during a period of seven months between October 2014 and May 2015. To interpret these counts and the possibility to extrapolate the data we investigated data of four other counters in Bosland, just outside our study area. These counters were all located within 7 km of our study area in similar habitat with counts for three consecutive years were available. We calculated a conversion factor as the ratio between the average daily number of visitors between October and May and the average daily number of visitors for a whole year. We found an average conversion factor of 1.05 (sd. 0.18) and used this to estimate an average daily number of visitors for a whole year for our own counter data.

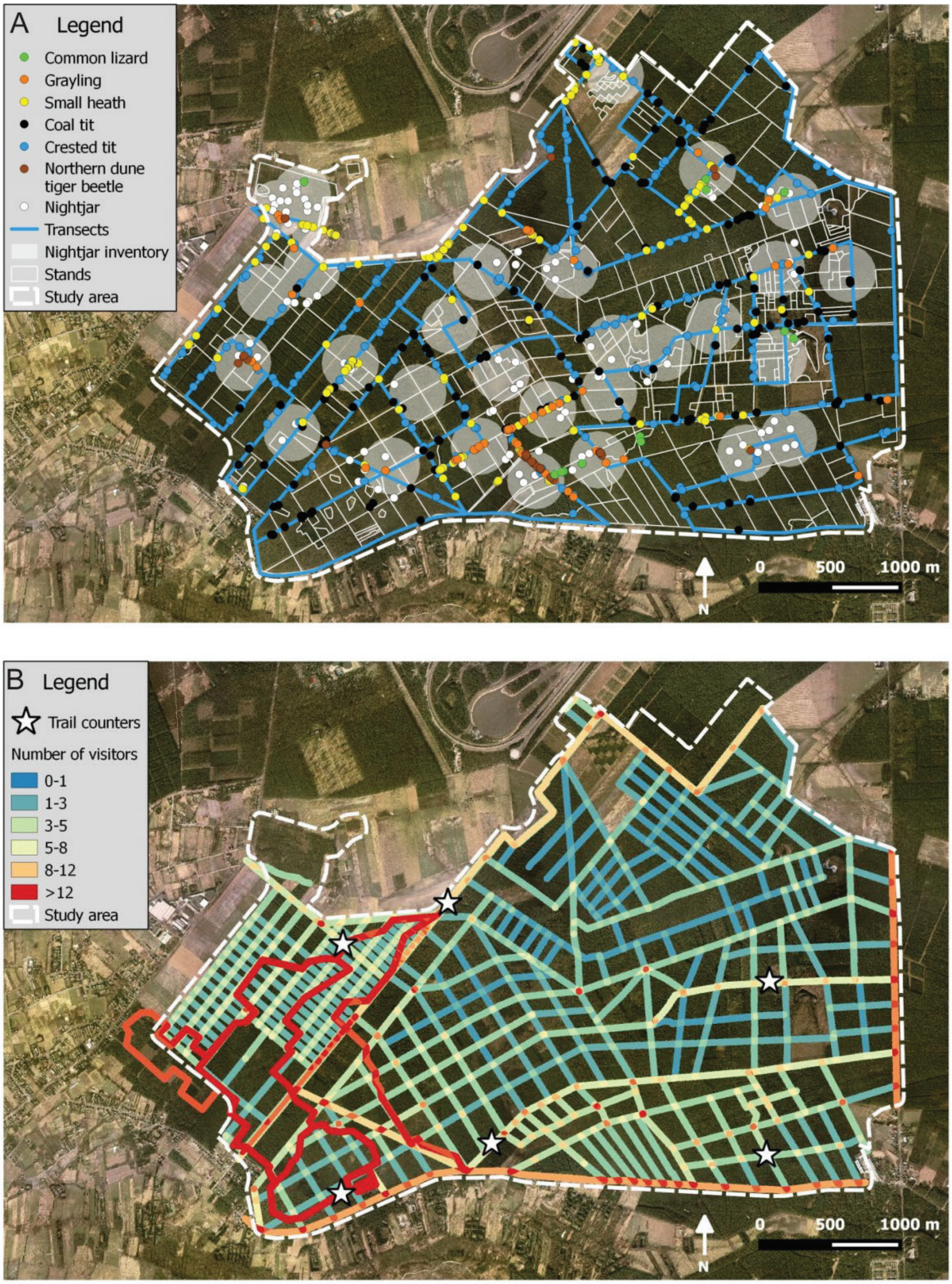

Figure 5.1: A. Total number of observations of the different indicator species within the study area (n = 821). B. Number of visitors on the different roads in the forests as deducted from interviews with forest guards and counts from trail counters.

Next we adjusted the relative recreational intensity map with the estimated average number of daily visitors for every road segment and obtained a map with an estimated recreational pressure for each road segment (Figure 5.1B). We also calculated a recreation pressure score for each forest stand with the following formula (Figure 8.2 in Appendix):

$$recreation\ score\ of\ forest\ stand = \frac{\sum[(length\ of\ adjacent\ road\ fragment) * (average\ daily\ number\ of\ visitors\ on\ adjacent\ road\ fragment)}{surface\ area\ of\ a\ forest\ stand}$$

5.3.2.3. Wood production

We estimated the mean annual increment for each stand, based on the stand age, the dominant tree species and the site quality which was deducted from the soil map (after Broekx *et al.* (2013)). We then calculated the standing stock of every stand by multiplying the stand age with the stand area, the mean annual increment for each tree species and a harvest factor. The harvest factor was included to compensate for the wood harvested in thinnings and thus to estimate current standing stock rather than total production of the stand since planting. This harvest factor was calculated as the ratio between the volume of the final harvest and the total volume of all thinnings and final harvest according to the growth table of Jansen *et al.* (1996).

The production of biomass from tree tops was calculated with the stem volume and a species-specific biomass expansion factor (after Vande Walle *et al.* (2005)) and with estimated harvest losses of 40 % (chapter four).

5.3.2.4. Habitat characteristics

We combined different data sources and layers to map the habitat in the study area. First of all, for the forest stands we used the map of the 2010 forest inventory. This map included all the forest stands and important habitat information such as dominant tree species and stand age. We added two important habitat features to this data layer, namely the recreational pressure of the stand (based on the pressure of the surrounding road segments) and the amount of neighboring open habitat.

We mapped the entire road network, based on aerial photographs and ground field data. The main habitat features for each road segment were the area, the orientation, the recreational pressure and the surface type (tracks on sand, grass, tree litter or paved with tarmac). A third information layer was a map with the non-forested patches within the area, the recreational pressure and the surface type (orchard, agriculture, sandy, heathland/grassland, clear-cut, plantation) as the main habitat features. We combined the road network patch layer with the non-forested patches layer in one layer for all open habitat patches.

Finally we come to a map with both forest patches and open patches, where a patch is defined as a more or less homogeneous habitat (i.e. a forest stand or an open patch). An overview of the different habitat features of both forest patches and open patches is given in Table 8.6 in Appendix.

5.3.3. Data analyses

5.3.3.1. Biodiversity

All spatial analyses were performed in QGIS 2.10.1 (QGIS Development Team 2015) and all statistical analyses were implemented in R 3.0.1 (R Core Team 2013), using the Multi-Model Inference package (MuMIn). Every point observation was assigned to a forest stand (Coal tit, Crested tit, Lizard and Nightjar) or to an open habitat element (i.e. a road segment or a permanent open patch) (Butterflies, Beetles, Lizard and Nightjar). For nightjars we used a circular buffer with a radius of 20 m, because the exact location of a churring individual is hard to locate exactly. The presence of a certain species in a patch was modelled with a logistic regression with the different habitat features as predictors for the patches (either forest stands or open habitat patches) that were part of the inventory for this species. Patches were considered as part of the inventory when lying adjacent to an observation route (Forest species and GHS species), containing a corrugated sheet (Lizard) or lying within 400 m of an observation point (Nightjar) (after Rebbeck *et al.* (2001)). Observation surface was included in the regression models as a covariate, to compensate for the fact that a higher observation surface automatically leads to a higher number of observations. We rescaled all numerical predictors by subtracting the mean value and dividing through the standard deviation to increase comparability. We ran generalized linear models (GLMs) using a binomial distribution for every possible combination of predictors (i.e. 256 models for forest patches, 16 for open patches). The models were ranked based on the AIC criterion, using the dredge function in the MuMIn package. Models with a delta AIC smaller than four were considered equivalent (Bolker 2008). These so-called top models were used to calculate an average model with the model averaging function in the MuMIn package (Symonds & Moussalli 2011). The R^2 was calculated for the model containing all predictors that appeared in the top models. The importance value was used to evaluate the relevance of the different predictors for species distribution. Habitat features that did not appear in the top models were left out of the analysis. The final average coefficients were used to predict probabilities of presence of the different species in all patches. The probability of occurrence was considered as a measure for habitat suitability and was mapped with a value between 0 and 1 for every habitat patch.

These habitat suitability maps were imported in *Zonation 4* (C-BIG, Helsinki), a framework and software tool for conservation prioritization and large-scale spatial conservation planning. It identifies areas that are important for retaining habitat quality and connectivity simultaneously for multiple species, thus providing a quantitative method for enhancing persistence of biodiversity in the long term (Moilanen *et al*., 2014). The software tool translated the habitat suitability maps to a raster with 5 m × 5 m grid cells and ranked these cells according to their importance for the maintenance of a species. We used the basic core-area cell removal rule algorithm to decide which cells were least important for a species (Moilanen *et al*., 2014). To evaluate habitat quality and connectivity, this algorithm depends on two species-specific biological parameters: the dispersal capacity and the kernel width. The dispersion capacity was calculated as the inverse of half the dispersion distance in meters (Moilanen *et al*., 2014)(Table 5.1). The kernel width was based on the mobility of a species through the forest matrix. For the species that depended on forest we set the kernel width to 50 m, the species of open habitats were assigned a smaller kernel width of 35 m (small copper and grayling) and 20 m (northern dune tiger beetle) depending on their dispersion distance. We grouped the forest species (coal and crested tit), the GHS species (butterflies and beetles) and the mixed species (nightjar and lizard). We thus obtained one rank of the different pixels in the study area for their suitability to sustain the current populations of the seven species under study.

5.3.3.2. Recreation and biodiversity

To evaluate trade-offs and synergies between recreation and biodiversity we selected the species for which recreation was an important variable in predicting the distribution (threshold set on an importance value larger than 0.5 (Lindtke *et al*., 2013)). These species were presumed to be vulnerable for recreation, as their distribution was negatively related to recreation intensity. The stands that had the highest average rank in Zonation for these vulnerable species were considered as the stands that were most vulnerable for recreation. The stands with the lowest average rank were considered as stands where recreation pressure has a lower impact on the distribution of the species involved. To test the trade-offs between recreation and biodiversity we calculated the impact of three recreation scenarios on the habitat suitability for the involved species. Scenario S1 doubles the amount of recreation everywhere; Scenario S2 doubles the amount of visitors in the least vulnerable areas and halves the amount of visitors in the most vulnerable areas, leading to an overall increase of 25% in the number of visitors; Scenario S3 increases recreation with 25% everywhere. We used these hypothetical recreation data to calculate the habitat suitability with the GLM for every species and compared the average habitat suitability score with the current reference.

5.3.3.3. Wood production and biodiversity

To evaluate the impact of harvesting on biodiversity we looked into the habitat preferences of the species. We considered a negative effect of clear-cuts on the habitat quality and connectivity for forest species and a positive effect on GHS species, that will profit from these new, temporal open patches. Nightjars depend on both forest stands and open patches and are very mobile, so spatial allocation of the harvested stands is probably less crucial to sustain populations. We next developed a harvesting plan for the next 20 years according to three different harvesting scenarios. First, in a wood production scenario, we followed the existing long-term vision on wood production (Moonen *et al*., 2011). Under this scenario, the oldest and most productive stands are harvested first. We ranked the stands by hand to a decreasing wood production score and harvested every year about 1% of the total area (average rotation period of 100 years). Second, in the biodiversity scenario, we first harvested the stands that have a low importance for forest species distribution and a high rank for the distribution of the GHS species. The third scenario is an integrated scenario that puts equal weights on the wood production rank and the biodiversity rank (as a low forest species rank and a high rank for GHS species). Finally, we calculated the output flow of harvested stem wood (and crown biomass) under these three scenarios.

5.4. Results

5.4.1. Habitat suitability

Our statistical models successfully explained the distribution of all but one of our study species (importance values and adjusted R squared in Table 5.2, average coefficients in Table 8.7 in Appendix). Only for the common lizard, we found that the best model was the intercept only model, without any environmental predictors. This is probably due to both the low number of patches in the inventory and the low number of observations. This species was left out of the analysis.

Table 5.2: Importance values of the different habitat features for predicting the presence of the species. A high importance value means a high presence of a habitat feature in the top (ΔAIC < 4) generalized linear models for a certain species. Habitat features that were absent in the top models are marked with light grey shading, habitat features that were absent for all species are not shown in the table. The adjusted R^2 values were calculated for models containing all relevant predictors. More info about the habitat features can be found in Table 8.6 in Appendix.

	Forest stands						Open patches			
	Area	**Age class**	**Recreation**	**Border**	**Management**	**Adjusted R^2**	**Patch type**	**Area**	**Recreation**	**Adjusted R^2**
Coal tit	0.31	0.45	0.94	0.93	0.20	0.19				
Crested tit	0.46	0.01	0.40	0.30	0.78	0.31				
Nightjar	1.00	0.97	0.36	1.00	0.12	0.39	1.00	0.96	0.29	0.30
Small heath							1.00	0.79	0.66	0.38
Grayling							1.00	0.39	0.78	0.35
Northern dune tiger beetle							1.00	0.67	0.31	0.45

The probability of occurrence of the coal tit was strongly negatively related to a higher recreation pressure and to the amount of adjacent open patches. Coal tits seemed to prefer closed high forest without open patches, without too much recreation and from age class 81-100. Crested tits had a higher probability of occurrence in large high forest stands, with a low recreational intensity and a limited amount of border with open habitat.

The probability of occurrence for churring nightjars was higher in smaller stands with a high amount of adjacent open habitat. Also some stand age classes had a much higher probability of occurrence for nightjars, particularly stands from age class 81-100, 21-40 and uneven aged stands. To a much lesser extent the probability to find churring nightjars was also negatively related to the amount of recreational intensity. In the open patches, probability of presence of churring nightjars was mainly related to patch type (high probability in young plantations and low probability in agricultural and orchard patches) and size (again higher probability in smaller patches). In general, there was a higher number of churring nightjars in forest stands (104) than in open habitats (41).

The probability of occurrence of small heath was positively related to large open patches with grassland, heathland or sandy habitats and to a low number of visitors. Grayling had a higher probability of occurrence in clear-cuts and plantations and to a lesser extent in grassland, heathland and sandy habitats. Also for grayling we found a negative relation between the recreation intensity and the probability of occurrence. The probability of occurrence of tiger beetle was highest in large open patches with a sandy surface and in grassland or heathland.

The average coefficients of the top models were then applied to predict probability of occurrence for the indicator species in stands and open patches (Figure 8.3 in Appendix). These habitat suitability maps were then imported in Zonation to evaluate the value of each grid cell for the conservation of a species groups (forest species, GHS species and species that depend both on stands and on forest), given the spatial distribution of the habitat patches and the mobility characteristics of the indicator species (Figure 8.4 in Appendix).

5.4.2. Recreation and biodiversity

The coal tit, the small heath and the grayling were most vulnerable for recreational pressure. We ranked all landscape cells according to their conservation priority for these species and made up a map with the most and the least vulnerable areas for these species concerning recreation (Figure 5.2).

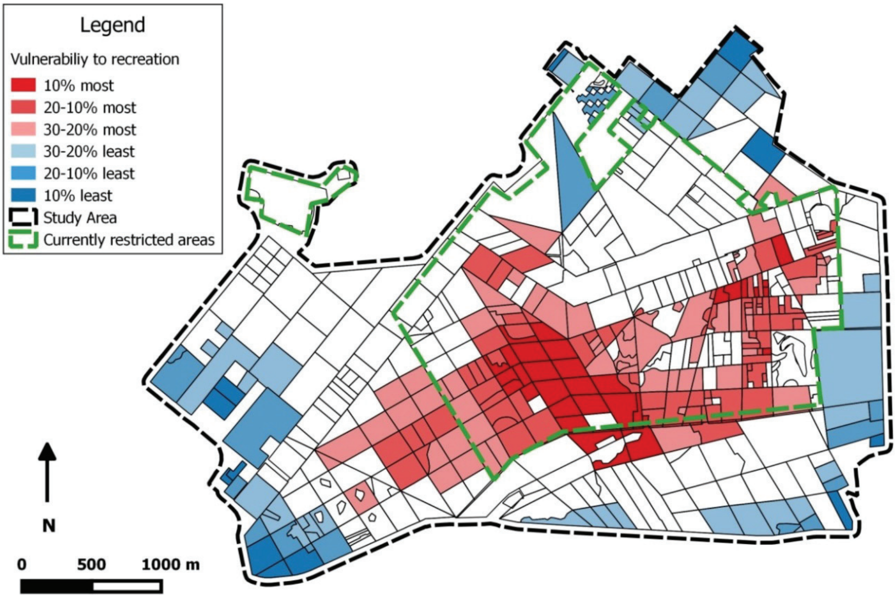

Figure 5.2: Vulnerability of forest stands to recreational pressure for coal tit, small heath and grayling.

Next we investigated the effect of different hypothetical recreation scenarios on the habitat suitability for coal tit, small heath and grayling (Table 5.3). We found a negative effect of increased recreation on habitat suitability, but the impact was much smaller if recreation in the most vulnerable patches was limited. There is thus a trade-off between recreation and biodiversity conservation, but it can be minimized if both goals are spatially separated.

Table 5.3: Impact of three hypothetical recreation scenarios on the habitat suitability for the three vulnerable species. Scenario S1 doubles the amount of recreation everywhere; Scenario S2 doubles the amount of visitors in the least vulnerable areas (blue in Fig.2) and halves the amount of visitors in the most vulnerable areas (red in Fig. 2), leading to an overall increase of 25% in the number of visitors; Scenario S3 increases recreation with 25% everywhere.

Scenario	Recreation (% of current situation)	Habitat suitability for the three vulnerable species (% of current situation): average (standard deviation)
S1	200	83.16 (1.36)
S2	125	97.82 (0.67)
S3	125	94.67 (0.27)

5.4.3. Wood harvest and biodiversity

Harvesting wood and biomass from final cuttings transforms mature stands to clear-cuts. This could have both negative (forest species) and positive (GHS species) effects on conservation of species. Depending on the developed scenarios, the management plan for the next twenty years diverge substantially. The biodiversity and wood production scenario share hardly any stands spatially, while under the integration scenario most harvested stands are also harvested under the biodiversity or wood production scenario (Figure 5.3). As expected the harvest of woody biomass is strongly determined by the chosen scenario (Table 5.4). In general, the more biodiversity conservation is included as a management target, the less wood is harvested, indicating a clear trade-off between these management goals.

Figure 5.3: Harvest schedule for the next twenty years under three different scenarios, a wood production scenario (WPS), a biodiversity scenario (BS) and an integrated scenario (IS). Some stands are only harvested under one scenario, some are harvested under two scenarios (brown and green dots) and some even under three scenarios (black dots).

Table 5.4: Mean (standard deviation between brackets) annual harvest of stem and crown wood and for the three scenarios.

	Wood harvest (m^3 $year^{-1}$)			
	Stem		Crown	
Biodiversity scenario	4422.4	(1330.4)	848.0	(254.4)
Wood production scenario	5645.8	(1820.2)	1083.2	(349.6)
Integrated scenario	4910.4	(964.0)	941.9	(185.5)

5.5. Discussion

5.5.1. Habitat suitability

Our results demonstrate that patch habitat features play an important role in the probability of occurrence of the indicator species. Only for the lizard, we found no significant relationship with any of the analysed habitat features. For the other species that use the forest matrix as a habitat, important features are the recreational pressure, the amount of forest border, the stand age or management type and the area. The contrast between coal tits and nightjars was interesting, with the former preferring large stands with limited borders and the latter preferring small stands with adjacent open space. This is in line with our expectations since coal tit is classified as a forest species (Brotons 2000) and nightjar as a mixed habitat species (Verstraeten *et al*., 2011). For the trail network and the open patches, we found a strong relation between the type of ground cover and the probability of occurrence of all indicator species. The butterfly species seemed to be more abundant when the number of visitors was lower. It is not surprising that the tiger beetle preferred large, sandy patches, however it is necessary to treat the results for this species with some caution, considering the limited number of observations. The probability of occurrence of small heath was bigger in larger open patches, while the opposite was true for nightjars. Nightjars thus occurred more in both smaller patches of forest and smaller open habitats, this links to its preference to a varied landscape. Nightjars were described to be vulnerable to recreational pressure (Langston *et al*., 2007; Lowe *et al*., 2014), however we did not detect a strong relation between recreational pressure. A possible explanation could be the mismatch between the location of a churring bird and the breeding location and the fact that we gathered data after sunset, when there is hardly any disturbance by recreation. Langston *et al*. (2007) mentioned that the main disturbance by recreationists on nightjars was related to a lower breeding success. Disturbance at the song posts after sunset will be much more limited.

5.5.2. Recreation and biodiversity

The stands that were important for the conservation of the populations of the coal tit and the butterflies were determined as the stands most vulnerable to recreational pressure. Most, but not all of the stands that were mapped as ‘vulnerable’ are already located in the actual zone with a low recreational pressure. This was expected, because the current distribution of the three species is the main parameter to determine the stand vulnerability and the distribution of these species is already influenced by the actual recreational pressure. The fact that we did not find a strong relationship between the number of visitors and the other indicator species does not necessarily mean that there is no such a negative effect of recreation on these species. However our data do not allow to assign certain stands for protecting these species. It is important to note that most of the stands that were mapped as tolerant to recreation were located at the edge of the study area. This is partly because of their habitat features that are less suited to sustain the indicator species. The effect is reinforced by the basic core-area cell removal rule algorithm implemented in Zonation, that promoted suitable habitat that is connected to other suitable habitat (Moilanen *et al.*, 2014). When looking into the results of the hypothetical recreation scenarios we observe a decrease of the habitat suitability for the three species with an increasing number of visitors. Protection of the most vulnerable patches seems indeed crucial to sustain populations. After all, we found a very low decrease in habitat suitability in scenario S2 compared to S3, where the reduction in habitat suitability is twice as big for the same total amount of visitors.

Our results thus seem to support the classical, land sparing approach to design the track network mostly in the border of a nature reserve, while safeguarding the core of the area from visitors for conservation purposes (Rodriguez-Prieto *et al.*, 2014).

5.5.3. Wood production and biodiversity

Depending on the management focus, the temporal lay out of the clear-cuts is almost entirely different, with the integrated scenario as an intermediate solution between both mono-functional scenarios. Stands harvested under the biodiversity scenario are mostly located closer to the edge of the study area where there is a low conservation value for the forest species and adjacent to existing open patches to increase habitat of GHS species. Adoption of the biodiversity scenario would reduce yearly stem harvest with ca. 22 %. When using an average resale price of 23 € m^{-3} stem wood and of 4 € m^{-3} crown wood (chapter two), subtracting a 33 % margin of profit for the harvesting company), the biodiversity scenario results in an income decrease of 29 000 euro per year compared to the wood production scenario over a planning period of 20 years. In the integrated scenario, annual harvest and total income declines by ca. 13 % (a loss of 17 000 € yr^{-1}). It is important to be cautious in interpreting these economic values which are based on a rough estimation of growth and for instance neglect possible positive biodiversity effects on tree growth. With the given data, forest managers can easily develop their own scenarios with a different weight for biodiversity or harvesting. Although including biodiversity conservation as a management goal negatively affects wood production, the results show that a land-sharing approach is possible without detrimental impact on either wood production and biodiversity conservation.

5.5.4. Integration of services

In order to better support complex ecosystem dynamics, we will need to develop a new kind of (planetary) stewardship (e.g. Power & Chapin (2010); von Heland *et al.* (2014)) which combines a systems approach with transformative action. The current study can be seen as a first stepping stone in this regard since we combine first notions of systems thinking (linking biomass production, biodiversity and recreation; using multi-species analysis; scenario development) with a more transformational approach (involving volunteers, action research design, focus on practical applicability and close cooperation with policy makers) in a real-life setting. We believe that this exploratory study furthers our understanding of what ecosystem stewardship entails by adding new insights on the synergies and trade-offs of different management scenarios which may be of particular interest for policy makers or practitioners on the field.

Developing a management scenario that includes recreation pressure, wood harvest and reaches biodiversity conservation goals is not easy. Comparing different management scenarios can help forest managers to identify knowledge gaps that need to be addressed for better ecosystem management and can help policy makers to develop adaptive management approaches that are more appropriate to support a multitude of ecosystem services. The different scenarios show how management can be focused locally on increasing either biodiversity or biomass harvest. By bringing these two together in an integrated scenario, an approach can be developed where the trade-offs can be minimized while optimizing the synergies. Installing the integrated harvest plan would increase the value of the landscape for biodiversity conservation, while safeguarding 87% of the current wood harvest. In combination with an intelligent trail design and conventional conservation strategies this could be an important step towards bringing into practice better stewardship management arrangements.

Scenarios such as the ones developed here can be very useful for forest managers since they provide first indications on the estimate of the income loss (or suspended income) when incorporating biodiversity conservation as a management goal. They can better balance installation of this scenario with the costs of other biodiversity conservation measures. Mönkkönen *et al.* (2011) modeled the cost-effectiveness of different biodiversity conservation measures: installation of a few permanent large reserves, of many permanent/temporary small reserves ('SLOSS dilemma'), and green tree retention. An important next step would be to investigate what additional costs might arise over a longer time period when choosing the wood optimization scenario.

When management is focused solely on biodiversity conservation, both recreation and harvest are restricted to the stands at the border of the study area (= land sparing). On the long term, this leads to a more homogeneous landscape with a forest core and a large area that is dominated by open habitats. While not included in our study, there also exist trade-offs and synergies between recreation and wood harvest. On the one hand, recreationists value structural variation at the landscape scale. On the other hand, clear-cuts can evoke strong objections by visitors (Brunson & Reiter 1996). Forest management measures such as thinning can also affect recreation. There is little information available, but Heyman *et al.* (2011), for example, studied the effect of openness in the understory of plantation forests and found a preference of visitors for a more open understory, but a slightly negative effect on bird biodiversity in more open plots. In order to develop better stewardship practices, more research is thus needed to cover a wider spectrum of ecosystem services and a more encompassing set of species.

5.5.5. Methodological remarks

Of course, studying multi-species habitat preferences on a landscape scale is susceptible to uncertainties such as parametrization of the modeling. First of all, the selection of the indicator species is an important a priori choice that will have an important influence on the results. Ideally, all biodiversity components across all taxa are included but this is virtually impossible. Therefore, indicator species are chosen that are assumed to well represent the biodiversity values of a patch. It is important to choose different indicator species from a wide range of taxa, habitat preferences and mobility (Heink & Kowarik 2010; Rodriguez-Prieto *et al*., 2014; Pakkala *et al*., 2014). With help of the local volunteers we succeeded in fulfilling these requirements. However, due to limited observations we had to exclude some species from the analysis, causing a slightly unbalanced distribution of indicator species with only insects as GHS species and only birds as forest species.

Second, we considered only presence/absence of a species in a habitat patch as an indicator for a suitable or unsuitable territory. We believe this to be quite accurate for the insects, the lizard and the singing coal and crested tit. Nightjar home ranges, however, are much bigger and absence of a churring bird is probably not a solid indicator of unsuitable habitat (Sharps *et al*., 2015). However, given the complex life strategy of nightjars and the difficulty in mapping nightjar territories (Rebbeck *et al*., 2001), our methodology seems a good compromise with practical feasibility. We also chose for a high spatial resolution with a high number of observers, but as a drawback we only used data from one night, which could distort the results. A third element that could distort the interpretation of the result is a possible mismatch between the scale of habitat mapping and preferences of the smaller indicator species. We worked at the landscape scale and performed analyses on the patch level (forest stands, forest road segments and open patches). Distribution of some indicator species will depend on micro-habitat features within patches, such as the presence of a host plant, microclimates or a smaller structure element (e.g. for grayling (Maes *et al*., 2006)), and bear little relationship with patch-level habitat features (Pakkala *et al*., 2014). However, only working at the patch level in such a large landscape was practically feasible. A fourth element that influenced our final results was the delineation of our study area. Our study area can more or less be considered as an ecological unity with sharp borders, agricultural areas in the north-west and main roads in the south and east. We thus considered every cell outside our study area as unsuitable habitat for the studied populations. Well-connected habitats occurred logically more in the center of our study area than at the border and were thus awarded a higher conservation value. Setting the importance value threshold on 0.5 (after Lindtke *et al.* (2013)) to evaluate vulnerability to recreation can also be subject to debate. Calcagno & de Mazancourt (2010) suggest a threshold of 0.8, which would give a different result in our analysis. A direct measurement of recreation pressure at the same time of species distribution mapping would maybe also have yielded better results. However we used the best available alternative, by estimating the year round recreation pressure with help of a conversion factor. Finally there are also lots of related issues that were not looked into in this study, but where supplementary research could be highly valuable. A next step could be a more formal trade-off analysis between wood harvest, recreation and biodiversity conservation. This can be achieved through a multi-criteria analysis of a set of alternative scenarios that combine different harvest and recreation regimes. However, to quantify the impact of each scenario on biodiversity conservation, absolute values are required instead of relative biodiversity conservation scores as those provided by the program 'Zonation'. Another interesting way to assess these trade-offs could be to execute the proposed management scenarios in the field and evaluate their impact on biodiversity. The future biodiversity surveys can then be used in future management plans, adopting a true adaptive management cycle (Lindenmayer *et al.*, 2006). Other aspects that urgently require further research include the relationship between other harvesting techniques than clear-cuts and biodiversity (e.g. Fuller (2013)), the biological interactions between species (Pakkala *et al*., 2014) and the economic (Schou *et al*., 2012) and ecological (Brown *et al*., 2015) impact of the planned conversion to broadleaves.

5.6. Conclusion

To conclude, the combined valuation of biodiversity conservation and wood production led to an integrated harvest plan increases the biodiversity conservation value of the landscape, while safeguarding 87 % of the current wood harvest. In addition, knowledge on the conservation value of stands can underpin an intelligent trail network design, guiding visitor streams and sheltering biodiversity hotspots. We showed that wood production and recreation have certain trade-offs with biodiversity conservation. However, with an intelligent spatiotemporal design, important biodiversity conservation gains can be made without greatly reducing the delivery of other services. The current study will help policy makers and practitioners to develop future management schedules, for Bosland and beyond. Moreover it demonstrates nicely that a combined land-sharing (for wood harvest) and land-sparing (for recreation) approach might lead to the greatest gains in simultaneous improving ecosystem services and biodiversity. There is an urgent need for additional research on the science-management interface, mainly on the interplay of different forest ecosystem services and the impacts for biodiversity.

5.7. Acknowledgements

We are very grateful to all people that have helped with the data collection: Lien Poelmans (butterflies); Ruben Evens, Eddy Ulenaers and volunteers (nightjars); Johan Agten, Eddy Ulenaers and Dries Gorissen (recreation). We also want to thank the biodiversity platform of Bosland for input on the indicator species and Lander Baeten, Leen Depauw, Renato Toledo, Sanne Van Den Berge and Laura Van Vooren on the data analyses. We are very grateful to E. Brockerhoff, H. Jactel and two anonymous reviewers for their comments and corrections on the previous version of the manuscript.

6. General discussion and conclusion

6.1. Main findings and implications for Bosland

6.1.1. Forest biomass harvesting: potential and limitations

As mentioned in chapter one, the demand for woody biomass from forests is increasing. Woody biomass is used for material and energy purposes and plays an important role in a transition to a bio-based economy. In our forests this increased demand leads to an interest in biomass that was not previously harvested, mainly left-overs on clear-cuts and trees from early thinnings. In chapter three and chapter four of this thesis we studied the possibilities to harvest additional biomass from pine stands in Bosland. The data from these chapters allow us to calculate a theoretical, maximal harvest potential of logs and wood chips for pine stands. However, we have also described different constraints for biomass harvesting. In the first place there are technical constraints, limiting the harvest potential because part of the biomass is not extractable with current technologies. There are also economic constraints that further limit harvest potential, because it is currently not profitable from an economic point of view to extract all technically harvestable woody biomass. Finally there are also several sustainability constraints that further limit the potential to harvest woody biomass. These different constraints lead to different, nested harvest potentials (Figure 6.1)(Vis *et al.*, 2010).

Based on the ecosystem stocks it is possible to calculate the theoretical potential biomass harvest under the intensive harvest regime described in chapter four (thinnings at a stand age of 30 years and 39 years and a clear-cut on a stand age of 48 years). In theory all woody biomass that is harvested in thinnings and clear-cuts could be used for energy purposes. However, Flemish legislation promotes the use of fully grown stems for material purposes and restricts its use for bio-energy. Such a cascaded use is a logical choice from a sustainability point of view, it maximizes efficiency of biomass use (after material use, application for bio-energy is still possible) and stimulates a circular economy (Keegan *et al.*, 2012). In this study we thus only looked at the biomass potential from additional sources (i.e. crowns from clear-cut leftovers (top bucking diameter 12 cm) and whole trees from early thinnings). This theoretical biomass harvest potential leads to a yearly harvest of 7.2 GMt logs and to 11.35 GMt wood chips per hectare in pine stands (Table 6.1).

In total, Bosland comprises about 6750 ha of forests (chapter two). So the total theoretical harvest potential of Bosland amounts to 48 619 GMt of logs and 76 628 GMt of additional biomass per

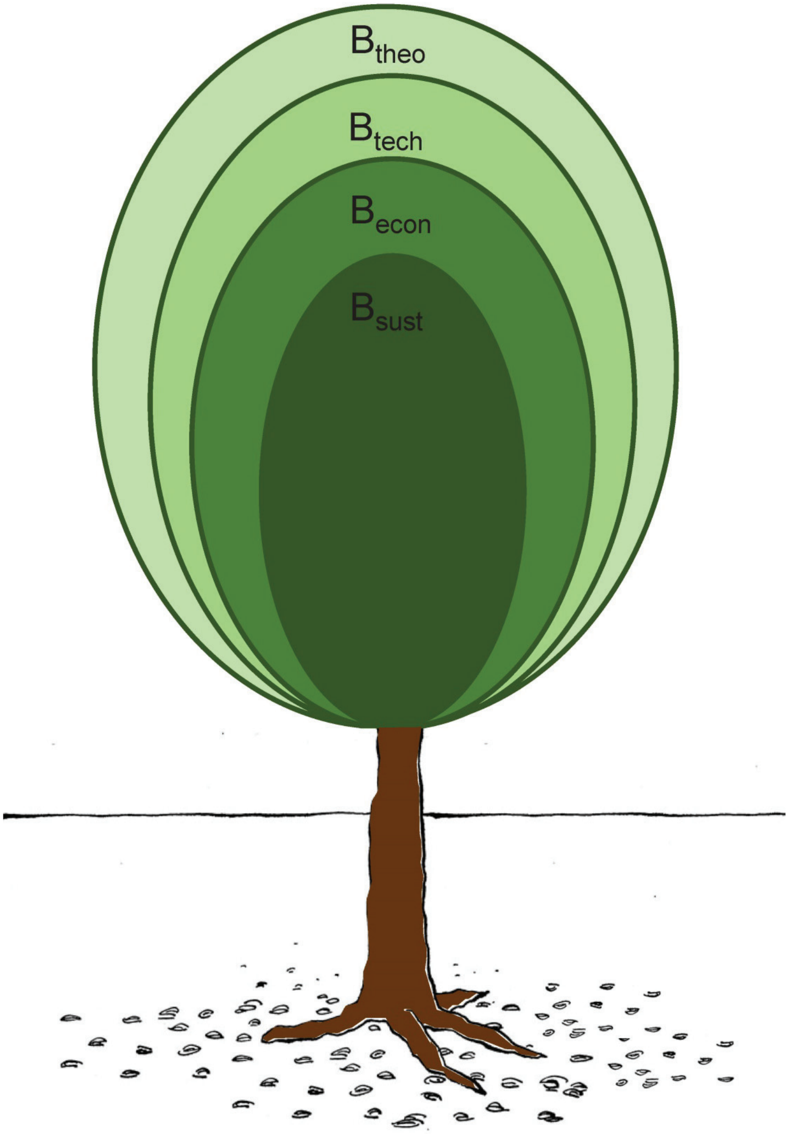

Figure 6.1: A conceptual figure illustrating the nested harvesting potentials of woody biomass in forests. The theoretical potential (B_{theo}) includes all biomass; the technical biomass potential (B_{tech}) includes all biomass that can be harvested with current technologies and excludes harvest losses; the economic potential (B_{econ}) includes all biomass that can be extracted in a cost-efficient way; the sustainable implementation potential (B_{sust}) includes only the biomass that can be extracted in a sustainable way, so without damaging the ecosystem.

year, for example as wood chips. If all wood chips would be used for bio-energy the total theoretical potential equals about 230 GWh (compensated for energy losses from drying and moisture content after (Edwards *et al.*, 2012; Francescato *et al.*, 2008)). This amount of energy could provide 66 000 average families with electricity (VREG, 2016)(under a theoretical 100 % conversion efficiency).

In chapter three we discussed the technical constraints of additional biomass harvesting in pine forests in Flanders. We compared different strategies for additional biomass harvesting in early thinnings and clear-cuts and found that a higher chip quality was achieved with a mobile chipper instead of the currently used roadside chipper. All strategies lead to significant harvest losses, mainly under the form of twigs and needles that broke off and remained on the site, adding to the litter layer after harvest. We found a difference between harvest losses in clear-cuts (40 %) compared to thinnings (46 %). If we incorporate these harvest losses it is possible to calculate a technical harvesting potential for biomass in Bosland and a technical bio-energy potential, based on the same assumptions as for the theoretical potential (Table 6.1). Incorporating harvest losses reduces the potential for bio-energy in Bosland to 162 GWh or electricity for 46 000 families. By improving technologies it would in theory be possible to reduce harvest losses and increase the technical potential. However, the fraction that is currently lost consists mainly of twigs and needles and is least interesting for bio-energy and material use. It has a low wood content and consists mainly of bark and needles, leading to a lower energy content and mainly a higher ash residue (chapter three). Moreover, these harvest losses contain high amounts of nutrients and increasing technical potential would also lead to an increased nutrient export and possible soil depletion (chapter four).

In chapter three we also looked at the economic potential of additional biomass harvest in Bosland. We found that it was economically profitable to harvest additional biomass from the pine stands in Bosland. However, the margin of profit was very narrow for most operations. We found that a mobile chipper can achieve a higher cost effectiveness than the currently used roadside chipper in clear-cuts. However, the cost effectiveness of a mobile chipper seems highly dependent on terrain accessibility and the cost effectiveness of the roadside chipper could be much improved with better equipment balancing. In thinnings we found a higher cost effectiveness of the roadside chipper, due to the lesser mobility of the mobile chipper. The most important finding from the economic analysis, however, was that harvesting logs is currently much more profitable than harvesting wood chips. For the forest exploitation company it was possible to increase income (i) by decreasing the top bucking diameter in clear-cuts from 12 cm to 8 cm, resulting in a higher amount of logs and a lower amount of wood chips (ii) by harvesting the stems separately in thinnings and sell them as logs instead of wood chips. If we apply these two measures (top bucking diameter of 8 cm instead of 12 cm and harvesting stems separately in early thinnings), we can calculate the economic potential for biomass harvesting in Bosland (Table 6.1). This reduces the potential for bio-energy in Bosland to 79 GWh or electricity for 23 000 families, at the same time the production of logs increases to 77 076 GMt per year. In the future, the prices of logs and mainly wood chips will probably keep rising and will make additional biomass harvesting increasingly profitable. Raunikar *et al.* (2010) modelled the price of different wood fractions and found that prices for energy wood could converge towards the prices of pulpwood by 2025. However, even if this occurs, harvesting costs for wood chips are still higher than for logs, so it would probably remain more profitable to extract logs than to chip them.

In chapter four we looked at the impact of additional biomass harvesting on the long term soil fertility. Soil fertility is an important sustainability criterion, not least because it directly influences future harvest of biomass. We defined a harvest regime as unsustainable if the ratio between the imports (mainly through deposition and weathering) and the exports (mainly through harvest and leaching) of nutrient was smaller than 0.9 and if the remaining ecosystem nutrient stock was not sufficient for the next ten rotation periods (after Göttlein *et al*. (2007)). We found very strong negative trends in long term nutrient concentrations under WTH for Ca, K and P with a depletion of the ecosystem nutrient stock after only four rotations. Under SOH we found slightly negative trends for Ca and P only and the current ecosystem nutrient stocks were sufficient for fourteen and a little more than ten future rotation periods. According to our modelling exercise, which has definitely some limits and comprises some uncertainties, P is the most limiting nutrient for sustaining an intensive harvesting regime (relates also to disturbances in the P biogeochemical flow as described in the work on planetary boundaries (§1.1)(Steffen *et al.*, 2015)). If we would maximize harvest until we reach the sustainability criterion (to sustain enough P for ten more rotation periods) we could harvest about one tenth of the current additional biomass from clear-cuts only. So this allows us to calculate a sustainable implementation harvest potential, only based on soil fertility (Table 6.1). This greatly reduces the bio-energy potential to 3.6 GWh, enough for the electricity of 1041 families if this harvesting regime would be installed in every stand of Bosland. However, this is of course impossible, as some stands in Bosland are for instance set-aside as forest reserves. Moreover, as mentioned in chapter four, the uncertainties in the modelling should be reason for precaution and it is probably not the best strategy to maximize harvest just to match a well-chosen but still arbitrary sustainability criterion. The general recommendation of chapter four was to limit harvest in these pine stands under such an intensive harvest regime to SOH for Bosland. This equals the sustainable implementation potential for additional biomass harvest in Bosland to zero. It is clear that the sustainability restrictions have the most severe impact on the harvest potential (compared to technical and economical restrictions). This is a very important finding that is of relevance for other pine stands in Western-Europe and probably also hold important lessons for other forest types.

However, the results from chapter four are reflecting almost a worst-case scenario and are not easily transferred to other regions. Most pine stands are currently already managed under longer rotation periods, which could decrease impact of WTH on long term soil fertility. Other soil types, even sandy soil types with only a slightly higher amount of nutrients, could possibly bear a higher export. Moreover, the results from our study were obtained in a region and time period with a very high (historical) acidifying deposition, possibly resulting in a decreased stock of base cations because of increased leaching (Verstraeten *et al.*, 2012). These depositions were lower for other regions and have decreased for Flanders in the last decades (Verstraeten *et al.*, 2012). A history of lower acidifying depositions could lead to a higher amount of base cations in soils and thus to a higher resilience to increased harvesting. Anyway, our findings show that long term soil fertility can be a very important limitation for additional biomass harvest in these forest types. This illustrates the need for precaution and a robust site-specific analysis of the risks. Harvesting too much undermines ecosystem integrity and resilience for the long term (as also mentioned in ecosystem stewardship literature (Chapin *et al.*, 2010))

To define the actual sustainable implementation potential for additional biomass, it is probably needed to look at more than one sustainability criterion (however, soil fertility is a very important one that should definitely be included). As mentioned in chapter one, an increased harvest of biomass could also affect biodiversity and even recreation. In the framework of this thesis we have not determined the impact of additional biomass harvest on these or other ecosystem services on a stand scale.

If the biomass from Bosland is to be the source of energy and material to fuel the transition towards a sustainable bio-based economy it is clear that only the sustainable harvest potential is available. This relates to ecosystem management or ecosystem stewardship (Chapin *et al*, 2010). When the different constraints are neglecting, this refers towards earlier management regimes such as (over)-exploitation or steady state resource management (Figure 1.7).

Table 6.1: A. Theoretical potential, technical potential, economic potential and sustainable implementation potential for logs and wood chips for a pine stand of one hectare in Bosland under thinnings at stand age 30 years (T30) and 39 years (T39) and a clear-cut at a stand age of 48 years (CC) and potential harvest for one rotation and per year (average +- sd). B. Yearly theoretical, technical, economic and sustainable potential energy production and number of households that could be supplied with electricity if all wood chips would be used for energy production and if all stands in Bosland would be used for biomass production (average +- sd). For more details on the assumptions and calculations see main text.

		Theoretical potential		Technical potential		Economic potential		Sustainable implementation	
A		**Logs (GMt)**	**Wood chips (GMt)**	**Logs (GMt)**	**Wood chips (GMt)**	**Logs (GMt)**	**Wood chips (GMt)**	**Logs (GMt)**	**Wood chips (GMt)**
per ha	**T30**	0	158.96 +- 33.4	0	111.02 +- 18.04	60.55 +- 4.76	42.31 +- 3.33	60.55 +- 4.76	0
	T39	0	231.59 +- 40.32	0	181.23 +- 22.99	122.4 +- 14.76	58.83 +- 7.61	122.4 +- 14.76	0
	CC	345.74 +- 5.30	154.36 +- 2.61	345.74 +- 5.30	92.61 +- 1.57	365.15 +- 0.78	86.39 +- 0.90	365.15 +- 0.78	8.64 +- 0.09
	Rotation	345.74 +- 5.30	544.91 +- 52.42	345.74 +- 5.30	384.87 +- 29.26	548.1 +- 15.53	187.53 +- 8.36	548.1 +- 15.53	8.64 +- 0.09
	Per year	7.20 +- 0.11	11.35 +- 1.09	7.20 +- 0.11	8.02 +- 0.61	11.42 +- 0.32	3.91 +- 0.17	11.42 +- 0.32	0.18 +- 0.00
B									
Bosland	**Energy (GWh/year)**		229.88 +- 22.12		162.36 +- 12.34		79.11 +- 3.53		3.64 +- 0.04
	Households' electricity (per year)		65681 +- 6319		46390 +- 3527		22604 +- 1007		1041 +- 11

6.1.2. Smart land management for different ecosystem services

In chapter five we investigated how biodiversity conservation in pine stands on former heathland was affected by wood and biomass harvest and recreation on a landscape scale. We found a trade-off between biodiversity conservation and both wood production and recreation, but with different implications for management.

Wood and biomass production had a negative effect on forest species by increasing the amount of forest edges and thus fragmenting forest habitat. However, at the same time, harvesting wood and biomass had a positive effect on the GHS species, that use the clear-cut areas as habitat patches. The impact of removal of clear-cut leftovers from clear-cuts on GHS species was not investigated in this research, but positive effects could be expected (Vandekerkhove *et al.*, 2012). Given the divergent effects of harvesting on different species groups, it seems that **land sharing** can be adopted, integrating both wood harvest and biodiversity conservation on the same land. Yields decreased when biodiversity conservation was included as a management goal next to wood and additional biomass production. Instead of only selecting the stands that would deliver optimal harvest, the selection was then also based on the habitat network and preferences of the indicator species. By harvesting a stand next to an existing open patch, the amount of forest edge increased less, so the impact on forest species was smaller, while at the same time there was new suitable habitat created for GHS species, adjacent to existing habitat. By applying this strategy we created a strong connected habitat of open patches that, so to speak, shifts through time and space. Applying this strategy should guarantee conservation of both forest species and GHS species and can be combined with wood and biomass harvesting (as also demonstrated by for instance (Marušák *et al.*, 2015). By awarding equal weights to biodiversity conservation and harvesting, we found, for instance, that the yield only decreased with 13% compared to a scenario with only harvest as a management goal.

For recreation on the other hand we found a clear negative effect on the distribution of some species. Coal tit, small heath and grayling were mainly found in areas with a lower recreation pressure. The negative effect of recreation asks for a **land sparing** approach, designating one part of the forest for recreation and other parts for conservation of the vulnerable species. If recreation pressure in core areas for the populations of these species was decreased, an overall increase of recreation pressure in other parts of the forest had hardly any impact. This means that recreation and biodiversity conservation are compatible management goals within the studied area. The zones that were least vulnerable for recreation were mostly located at the border of the studied area, this supports a track network design in the border of a nature reserve, while safeguarding the core of the area (Rodriguez-Prieto *et al.*, 2014). However, biodiversity conservation should go beyond the indicator species we selected and even the fact that we did not detect a clear effect of recreation on the other studied species does not per definition mean that there is no effect. For example, breeding nightjars are known to be vulnerable for recreation (Langston *et al.*, 2007). However, we found no clear effect of recreation, but we only mapped the distribution of nightjars based on churring locations and not on nesting locations. The proposed approach, shielding the core area from recreation could and should be complemented with species-specific measures. For example, if nesting locations of nightjars are exactly known (which is often the case in Bosland, thanks to the work of www.tracingnature.com) a diversion of a track could be installed during the breeding season.

In general, we demonstrated that a good inventory of indicator species distributions, recreation pressure and wood harvest can inform a smart land management approach that integrates management goals and delivers different services on a landscape scale. Our results support an approach that adopts both land-sharing (with wood harvest) and land-sparing (with recreation) as a biodiversity conservation strategy. These results could be combined in a map with direct suggestions for management both on recreation and harvest in the study area (Figure 6.2). These suggestions will be presented to and discussed with the forest managers in the area. They have the terrain expertise to evaluate the practical feasibility of these measures and to add other measures, such as species specific conservation actions. In a next stage the proposed measures will be discussed with a wider public, such as the platform of fauna and flora that gave input in the initial selection of the indicator species. As mentioned in chapter one, a constant dialogue between scientists, policy makers and field practitioners, combined with a participatory approach is essential for supporting an adaptive policy approach and successfully applying the ecosystem services framework (Daily *et al.*, 2009).

A well-balanced management of the different ecosystem services is a very important step towards ecosystem stewardship. However it is only a first step. Next to innovative models answering the biophysical complexity of ecosystem management, there is also a need for new management models that better unite the views of different stakeholders and the longing for participation. (§0).

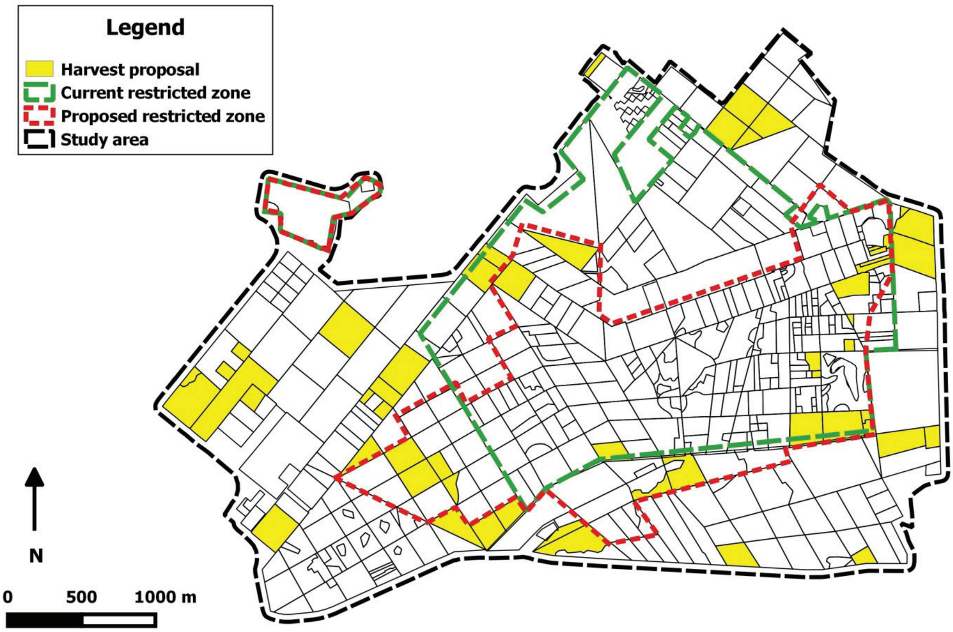

Figure 6.2: Map of the study area with proposed management actions based on the results of chapter five. The yellow stands are proposed for harvesting in the next twenty years and a new restricted zone is proposed, slightly diverging from the current restricted zone. These results could be expanded by adding data on other species and the practical feasibility should be checked with the forest managers.

Box 2: The value of nature recreation

In our analysis in chapter five we looked at the impact of recreation on biodiversity conservation and we found a clear trade-off that suggests a land sparing approach in which both goals are spatially separated. This could be interpreted as an advocacy to fence off all forests from recreationists to stop further biodiversity losses in forests. This was, however, not at all the main message we want to put forward, because this would ignore the different values of nature recreation.

Forest areas are among the most popular settings for outdoor recreation (Nielsen *et al.*, 2007). Access to nature and forest areas for recreation can benefit mental and physical health, by reducing stress and stimulating physical activity (Doctorman & Boman, 2016). Especially for children, access to nature and forests can make a big difference in well-being (Wells, 2000) and can also influence future behaviour towards forest and nature conservation. The economic value of recreation is often expressed by a willingness to pay of recreationists. This willingness to pay depends on different factors, such as the naturalness of the forest, the societal background of the recreationist and the access to alternative forest and nature areas (Nielsen *et al.*, 2007; De Valck *et al.*, 2014). Anyway, the economic value of recreation in a large forest and nature area, such as Bosland in an urbanized environment, such as Flanders is substantial. As mentioned in chapter five, Bosland is a major tourist destination with more than 300 000 overnight stays every year, with the resulting benefits for hotel and catering industry. The partners of the Bosland project have developed numerous routes for hikers, cyclists, horseback riders and mountain bikers and have adopted a special focus on children. Different educational tracks, adventurous routes and playing grounds have been installed, all with a close link to the forest (Figure 6.3).

Both recreation and biodiversity conservation are thus necessary and desired functions of forests. The results of our study, showed also that recreation and biodiversity conservation can go together in the same forest, but in different areas. This land sparing approach is perfectly feasible in large forests such as Bosland and even in the study area of chapter five, a core area with important nature conservation values. However, it has been demonstrated in similar ecosystems that habitat fragmentation intensifies trade-offs between biodiversity and recreation (Cordingley *et al.*, 2015).

The clear trade-off between recreation and biodiversity conservation, can certainly be considered as a threat for the fragmented forest area in Flanders. There is thus a strong need for larger forest areas, which will make biodiversity values more robust and resilient, so that this can go together with a high recreational pressure in certain less-vulnerable areas. Our results can thus better be interpreted as a policy call for forest and nature expansion than as an appeal to close off forest and nature remnants in an urbanized landscape. More robust populations of rare species in these larger forest patches could on their turn attract more eco-tourists and have a positive effect on recreation.

Figure 6.3: Nature recreation and education will have a positive influence on well-being of children (and adults). Several playgrounds have been installed in Bosland with for instance huge wooden forest insects that are used as toys for children (Photo: Bosland).

6.1.3. New management models

During the last decades, forest management in Western Europe is transitioning towards multifunctionality, combining principles of different traditional fields with complexity and adaptation (Puettmann *et al.*, 2009). Managing a forest across a multitude of stakeholders, under the pressure of several grand challenges and aimed at a range of ecosystem services requires new management approaches. In chapter two, we described the development of Bosland using a learning history approach and transition lenses. This allowed us to reconstruct the history of Bosland and enabled us to identify essential steps and innovative features that have been developed through the collective search and learning process of the new partnership. In contrast to the traditional top down management style that is directive, ANB shifted towards a more collaborative, catalyst style to promote collective and shared value creation across the involved stakeholders. Co-creation of a shared long term vision helped to unite the different stakeholders and to give direction to the management plans and masterplan. In general, every short term action in Bosland is in alignment with the long term vision and the concurrent strategic headlines. The participatory approach will continue to play an important role in the future management of Bosland by means of a forest parliament and houses to give space to and connect the variety of stakeholders and a forest laboratory to cross-pollinate between science and policy. Taken together, these innovative features also require new roles and different arrangements between the partners to realise the long term ambition. Because of this innovative features, we concluded that Bosland differs fundamentally from the forest management as usual and can be considered a pioneering case, putting into practice a new way of forest management.

But, is this new approach also better suited to face the grand challenges and the rapid changes that put pressure on the current practices in forest management? Different features of the Bosland project align with the characteristics of ecosystem stewardship as described in Table 1.1. One clear example was the shifting role of ANB as a "*resource manager from a decision-maker to a facilitator who engages stakeholder groups to respond to socio-ecological changes*" (Chapin *et al.*, 2010). Another, striking example was the origin, the development and the aim of the Bosland project: the project originated after a new, challenging legislation that induced collaboration or to put it in the words of Chapin *et al. (2010)*, "*a disturbance was used as a window of opportunity*", the project developed from a co-created long term vision that inspired short term actions "*guiding a change trajectory*" towards a management aimed at "*sustaining social-ecological system and the delivery of ecosystem services*". Another typical feature of ecosystem stewardship is the role of researchers, who work more interdisciplinary and who “*collaborate with managers through adaptive management to create continuous learning loops*” (Chapin *et al.*, 2010). The Bosland partners seem to realize that a close collaboration with scientists can yield great benefits to the project and have recently launched “Boslab” (www.boslab.be). Boslab should become a platform for researchers and managers to share and exchange information aimed at directly implementing this knowledge in an adaptive management-like approach. Together with the platform, a field research station and an “open air laboratory”, with different test sets that should allow visitors to really experience high tech forest research, will be installed in the field (Figure 6.4). Moreover, the results from scientific research in Bosland are not just put aside, but often really implemented in management. The findings of chapter 2 for example were communicated to the Bosland partners and have provoked action. The representatives of the Bosland partners have installed a structural dialogue with the formal representatives of the NGOs such as nature organisations, complementary to the participation of local volunteers and naturalists in the ecological house. This has already lead to interesting additional interaction and collaboration with Natuurpunt in concrete projects (guiding tours, adjustment of local management visions, …).

The Bosland project thus meets many of the typical features of ecosystem stewardship, a new mode of governance specifically aimed at answering changes in society and ecosystems. As mentioned already in chapter two, the Bosland model is not the one and only management of the future for forest and nature areas and there is certainly room for improvement in the approach. However, innovative and successful examples such as Bosland, could be replicated, up scaled and embedded in governance to help in accelerating the transition towards ecosystem stewardship (Gorissen *et al.*, in progress).

According to ecosystem stewardship theory (Chapin et al., 2010):

- Every system exhibits critical vulnerabilities that are worsened by environmental and social changes that stretch the socio-ecological system beyond its limits of adaptability.

- Every system has sources of diversity (socio-economic, biological and institutional) that provide building blocks for adapting to the changing future.

- Every system has thus opportunities for transition, following more desirable trajectories of social-ecological change.

This would mean that ecosystem stewardship could be used in every socio-ecological system to answer the environmental and social pressure. However, there are of course some conditions which make successful application easier (Chapin *et al.,* 2010).

- If changes/pressure are clear and managers realise the hazard of these rising pressure on the socio-ecological system.

- A larger socio-economic, biological and/or institutional diversity increases resilience of a system and makes it easier to adapt and to define more desirable transition trajectories.

For the Bosland case it is clear that the conditions were very well suited for a new management regime with elements of ecosystem stewardship. As described in chapter 1, there are multiple large environmental and social challenges that put pressure on forest ecosystems. These challenges are more manifest in the Flemish context with a low forest cover and high urbanization rates (Hermy *et al.*, 2008; Cordingley *et al.*, 2015). The new demands and rising pressure were also notified by some key people in Bosland. Moreover, Bosland partners were "encouraged" to work together by the changing legislation (Chapter 2). The Bosland partnership increased institutional diversity. The increasing demand for participation led to the development of the Bosland parliament that gave a voice to different socio-economic groups (increasing socio-economic diversity). Bosland is the largest forest of Flanders and hosts important biodiversity values in a diverse landscape of forests and more open habitats such as heathland and grassland (high biological diversity).

The high pressure on the system, combined with the high resilience of the system (high socio-economic, institutional and biological diversity) facilitated the transition to new management models.

This transition is of course also possible in other systems (and according to Chapin et al. (2010) in all socio-ecological systems) although conditions will not always be as favourable as in Bosland.

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HET BELANG VAN LIMBURG
ZATERDAG 17 EN ZONDAG 18 OKTOBER 2015

REPORTAGE

HBVL

GROOTSTE OPENLUCHTLABO VAN

Wetenschappers werken aan bos van de toekomst

HECHTEL-EKSEL - Een speeltuin voor kleine maar ook grote kinderen. Dit is het minste wat van Bosland, het grootste bos van Vlaanderen, kan worden gezegd. Maar Bosland is ook Boslab: het grootste openluchtlabo van Vlaanderen waar wetenschappers van nu aan het bos van de toekomst werken. Een bos met heel veel potentieel, met bomen die zware metalen in de bodem houden, met snelgroeiende boomsoorten om hout te winnen en dat allemaal zonder de veilige thuisbasis voor planten en dieren in het gedrang te brengen.

"Boslab is gelanceerd omdat we willen weten hoe een bos er volgens het ideale scenario moet uitzien. En omdat onze mensen en vrijwilligers daarvoor niet alle knowhow hebben, schakelden we de hulp van de wetenschappers in. Zij proberen antwoorden te zoeken op heel wat vragen en uitdagingen zoals hoe Bosland in te richten, te ontwikkelen en toegankelijk te maken. In ruil daarvoor krijgen ze een ontzettend grote proeftuin om te experimenteren", begint Dries Gorissen, een van de 'uitvinders' van Bosland. Voordeel van dit scenario is volgens alle partijen dat de bevindingen ook meteen in de praktijk worden omgezet. "Het ambitieuze plan op lange termijn is om een 'voorbeeld'-bos te ontwikkelen. En het verhaal gaat nog verder. Boslab wil ook de bezoekers betrekken bij al dat 'hightech'-groen. Wetenschapstoerisme waarmee we op een eenvoudige manier willen laten zien hoe een bos werkt, hoe bijzondere soorten leven, enzovoort. Een van de pistes is een soort 'boomriem' die meet hoe een boom krimpt of uitzet afhankelijk van de hoeveelheid water dat aanwezig is in de stam", licht Gorissen nog toe. Wij gingen al eens op vooronderzoek om te kijken wat een bos ons allemaal kan leren.

Bosland
Neerpelt
Lommel
Overpelt
Hechtel-Eksel
Peer
Leopoldsburg
Beringen

Tom Artois en Kris Verheyen, twee professoren die onderzoek doen in Bosland, en Dries Gorissen, de 'uitvinder' van bosland. FOTO MIST

Op zondag 18 oktober van 13.30 tot 17 uur is er een grote **slothappening** van de Week van het Bos in de Lommelse Sahara. Kom langs en maak kennis met de natuurhelden van Bosland.

Door Mireille Maes

Figure 6.4: Press article in a Belgian newspaper announcing the launch of Boslab.

HET BELANG VAN LIMBURG
ZATERDAG 17 EN ZONDAG 18 OKTOBER 2015

HBvLPLUS | 13

VLAANDEREN LIGT IN NOORD-LIMBURG

1 Eiken houden zware metalen in de grond

Onderzoek door Lotte van Nevel

In het Waaltjesbos in Lommel is onderzocht hoe de vervuiling in de grond vast te leggen zodat het geen verdere schade kan aanrichten FOTO HBVL

Noord-Limburg en dus ook een stukje Bosland, draagt een geschiedenis van vervuiling met zware metalen met zich mee. "Die zware metalen zitten in de grond. In het Waaltjesbos in Lommel is nu onderzocht hoe de vervuiling in de grond vast te leggen zodat het geen verdere schade kan aanrichten", vertelt Kris Verheyen, professor Bosecologie en Bosbeheer aan de UGent. Medio jaren 1990 werden in het bos meer dan 400.000 jonge bomen geplant. "Het zijn verschillende boomsoorten waardoor we onderzoeksmateriaal genoeg hadden. En ja, er blijkt duidelijk verschil te zitten in de bomen. De beste boom om de metalen in de grond te houden, is de eik. Het is een soort die heel weinig opneemt zodat metalen blijven waar ze zitten. En gelijktijdig weten we ook welke soort zeker niet op zwaar vervuilde gronden mag worden geplant: de trilpopulier. De reden: deze soort neemt zink en cadmium op in de bladeren van waaruit die opnieuw in het ecosysteem terechtkomen.

2 Combinatie inheemse en uitheemse bomen

Forbio-experiment

Klimaatsverandering en natuur, het blijft een heikel punt. Daarom is de UGent samen met de KULeuven – in het kader van het wereldwijde Forbio-experiment - aan het bekijken met welke boomsoorten in de toekomst wordt voortgegaan. Een van de Belgische sites is Pijnven in Eksel. "Intussen weten we al dat gemengde bossen het best functioneren. En onder gemengde bossen bedoelen we hier in Bosland een combinatie van de inheemse wintereik, grove den en ruwe berk samen met de uitheemse douglasspar en lariks", vat de Gentse professor samen. Dat er ook gewerkt wordt met uitheemse soorten hoeft niet per se fout te zijn. "Die twee uitheemse soorten leveren duurzaam hout op. Door ze op kleinere schaal te planten, kunnen ze voor de komende generaties geld en een milieuvriendelijke grondstof opleveren. Hoeveel precies? Dat kunnen we over enkele decennia onderzoeken. Want een boom heeft toch minstens 40 tot 50 jaar nodig voor hij gekapt kan worden."

Medewerkers van Forbio in actie tijdens de winter. FOTO HBVL

3 Gaat de Ekselse nachtzwaluw vreemd?

Onderzoek Ruben Evens

De nachtzwaluw. FOTO HBVL

Vier jaar zijn onderzoeker Ruben Evens en boswachter Eddy Ulenaers al bezig om het leven van de nachtzwaluw in de bossen van het Pijnven in kaart te brengen. Hij vergaarde een massa aan informatie voor ziin doctoraatsonderzoek waarmee hij 2 jaar geleden aan de UHasselt is gestart. Binnenkort start daar een tweede doctoraatsstudiel. En die gaat over niets minder dan het liefdesleven van de 'geitenmelker'. Tom Artois, professor Dierkunde, Biodiversiteit en Toxicologie aan de UHasselt: "We gaan kijken of en hoe de populaties van nachtzwaluwen - op het militair domein in Meeuwen en Helchteren zitten er bijvoorbeeld ook - met elkaar zijn gelinkt. Zoeken ze hun partner binnen dezelfde groep of durven ze al eens over de grenzen heen te kijken? Waar ontmoeten zij elkaar? Met de zendertjes die Ruben Evens aan de vogels hangt, weten we al heel veel. Vanaf 1 oktober komt daar dus het onderzoek naar de relaties tussen populaties bij. Afhankelijk van de bevindingen kunnen we opnieuw maatregelen nemen om de populatie nachtzwaluwen te helpen en vooral te versterken."

4 Balans tussen natuurbehoud, houtoogst en recreatie

Onderzoek door Pieter Vangansbeke

Een bos heeft verschillende functies: mensen komen er wandelen en genieten van de rust, het is een biotoop voor heel wat planten en dieren, maar er wordt ook hout gekapt. Een combinatie van de drie lijkt onrealistisch. "We zijn in Boslab aan het zoeken naar de perfecte balans tussen natuurbehoud, houtoogst en recreatie", legt professor Verheyen uit. "En er zijn veel mogelijkheden. Een van de denkpistes is de kruinen die bij het kappen van de bomen te verkopen als biomassa voor hernieuwbare energie. Tot nu toe bleven die gewoon achter in het bos. Hoeveel van dat houtafval - dat bijvoorbeeld kan dienen als schuilplaats voor vogels en insecten - kan een bos missen zonder de cycli te verstoren? Een tweede antwoord dat we zoeken is hoe recreatie zo goed mogelijk met de bescherming van kwetsbare natuur te combineren valt. Met wandeltellers proberen we nu te weten te komen hoeveel mensen de verschillende gebieden bezoeken en of er maatregelen moeten genomen worden om bepaalde soorten te beschermen. Uiteindelijk willen we komen tot een evenwicht tussen de verschillende functies van het bos."

6.2. Beyond Bosland: implications for management and policy

6.2.1. Biomass harvest potential and limitations

From the results described in chapters three and four we can formulate some recommendations on harvest of additional biomass from forests for forest management and policy. We have listed short, clear and management-oriented recommendations below and grouped these according to the different constraints that were defined in §6.1.1. Some of the findings are quite generally applicable, whilst other findings are highly context dependent and need additional research in other forest types/soil types/regions (see §6.3).

In chapter three we have studied the technical feasibility to harvest additional biomass in thinnings and clear cuts in pine stands. The results from the technical comparison can be reasonably well extrapolated to other pine stands to other regions. Exceptions are pine stands on steep slopes and stands that are not harvested in a clear-cut system. Moreover, we have not tested all possible harvesting strategies, in regions with a better developed forest exploitation sector, such as Fennoscandia or Canada, more options might be available, including specialized high tech machines.

Technical management recommendations:

- When harvesting additional biomass from clear-cuts in pine stands a mobile chipper could be a good alternative to the classically applied method of a road-side chipper, both in terms of time efficiency, energy efficiency and chip quality.

- When harvesting whole trees in early thinning in pine stands, an harvester, a forwarder and a road-side chipper are more time efficient than an excavator, a tractor with trailer and a mobile chipper respectively. The mobile chipper tested in our study was too big and not very manoeuvrable. Using this mobile chipper, took significantly more time making this option less suitable for thinnings, despite a higher chip quality.

- Harvesting crown wood, both from thinnings and clear-cuts in pine stands, resulted in substantial harvest losses, mainly from twigs and needles. We found harvest losses of about 40% in clear-cuts and 46% in early thinnings. The harvest losses will be lower in older stands and the exact harvest losses can differ according to exploitation strategy, stand characteristics and tree species, but the general conclusion seems also applicable to other forest types and should be kept in mind when estimating quantities of additional biomass.

- Good equipment balancing is very important for the time efficiency of a harvest strategy. If one step in a harvesting strategy is much slower than the other steps, this can slow down the whole harvesting chain (with also important economic consequences, see further).

In chapter three we also investigated the economic constraints for additional biomass harvesting in pine stands in Bosland. The economic constraints are influenced by the macro-economic trends such as price for logs and wood chips (and also for labour and fossil fuels) that could vary between regions and over time. Moreover, amount of forest cover (supply) and existence of bio-based companies (demand) can influence size of exploitation companies, which also has an impact on economics of harvesting. All of the above, makes the results about economics more context (and time) specific than the results on technical constraints. However, the results from our study are more or less generic for harvesting of additional biomass in pine stands in Flanders and neighbouring regions.

Management recommendations on economics

- It proofed more profitable to harvest wood as logs for material use than as wood chips for bio-energy. This means that (i) WTH in clear-cuts was most cost-effective with a low top bucking diameter, resulting in a higher share of logs and a lower share of wood chips; (ii) even in early thinning it was much more cost-effective to harvest logs separately than to chip whole trees. These results can probably be extrapolated to other forest types and regions.

- In clear-cuts, a mobile chipper was more cost-effective to harvest crowns than the more traditional system with a forwarder and a road-side chipper in our study. However, the opposite was true when the utilization rate of the road-side chipper would have been increased to the same level as the mobile chipper.

- When chipping whole trees in thinnings (which is less profitable than harvesting logs separately) a higher cost-effectiveness was reached when an excavator, a forwarder and a road-side chipper were used instead of a harvester, a tractor with trailer and a mobile chipper respectively. The harvester was more time-efficient, but had also a higher cost per SMH.

- Good equipment balancing is very important to keep the utilization rate of every machine high and in this way keeping the costs low. Good equipment balancing asks for a good internal organization and for a certain scale in the exploitation company. These results can be extrapolated to other stand types and regions.

- The margin of profit for harvesting additional biomass for the exploitation companies is very limited. This limits the income for forest owners/managers that are prepared to sell crown wood on top of logs and this limits the development of a viable exploitation sector specialized in harvesting additional biomass. The margin of profit might increase in the near future with the projected increase in price of energy wood.

- The indirect economic constraints of harvesting additional biomass were not included in our analysis, but do matter. If intensive WTH leads to a decrease in soil fertility (see further) this negative effect should be included in the economic analysis, which would most definitely make harvesting of additional biomass a net loss under the current circumstances.

In chapter four we studied the impact of additional biomass harvest on nutrient stocks and long term soil fertility in pine stands in Bosland. As mentioned several times before, we studied almost a worst-case scenario (i) involving an intensive management with a short rotation period (ii) on nutrient-poor sandy soils (iii) in an area with (a history of) high acidifying deposition, potentially leading to increased leaching of base cations. These three factors make the results of our study hard to extrapolate to other study systems. However for most pine stands in Flanders and neighbouring regions the situation is similar and the results give a clear indication of the impact of WTH on soil fertility under an intensive, short rotation management. However, the nature of the results we found calls for precaution in other systems and for additional research on nutrient budgets (Paré & Thiffault, 2016).

Management recommendations regarding soil fertility

- The soil nutrient status of the stands before harvest demonstrated that soils were already very acid and low in base-cations, which could have an effect on the growth of Corsican pines. So even under a business as usual management with stem only harvesting (SOH) the nutrient status of these stands should be evaluated from time to time to secure a sustainable growth.

- Whole tree harvesting (WTH) had a severe impact on ecosystem nutrient stocks in the studied pine stands, certainly in clear-cuts. The effect of harvesting crowns as additional biomass had a relatively limited impact on the exported biomass (22% more under clear-cuts), but a much higher impact on export of nutrients (50% more base cations and 81% more P under clear-cuts). In total about half of the nutrients in trees and forest floor were exported under WTH in clear-cuts. The main message of these results can probably largely be extrapolated to other regions and even to other stand types.

- The results of the modelling demonstrated a clear decrease in the ecosystem nutrient-stocks on the long term when applying WTH under the described circumstances. The major recommendation for management would be to not harvest additional biomass from these pine stands.

- In general, to sustain soil fertility nutrient inputs should equal nutrient exports on the long term. Different management measures could help in avoiding a long term depletion of nutrients by decreasing export through harvest, such as (i) longer rotation periods; (ii) leaving the crowns in the stand for one year, so that needles are shed; (iii) adoption of other silvicultural systems, such as selective cutting; (iv) only applying whole tree harvest on some occasions, such as every fourth rotation. Other management measures might avoid nutrient depletion by increasing nutrient import, such as (v) a well-balanced, stand wide, slow-releasing fertilization to compensate the losses through harvesting.

  Our results do not allow to estimate the effectiveness/cost of (a combination of) the different measures, but again we plead for precaution and a need for monitoring the nutrient status through time if WTH is applied.

Many of the management recommendations listed above are severely limiting the sustainable implementation potential of additional biomass from pine stand. Even though a large amount of additional biomass is theoretically available every year from these pine stands, the largest part is unavailable by technical, economic or mainly sustainability constraints. The findings from this research definitely need further investigation and testing in other forest systems, but some important messages for policy makers can already be formulated.

Policy recommendations on additional biomass harvesting

- A transition to a bio-based economy is desirable as part of the solution for mitigating climate change, but the amount of additional woody biomass that can be harvested from pine plantations in Bosland-like conditions is limited because of different technical, economic and mainly sustainability constraints.

- There is a strong need to inform forest owners and managers on the impact of additional biomass harvesting on soil nutrient depletion and other ecosystem processes and services that were not studied in the current work. For Flanders, a valuable first step was taken by the development of an online tool advising on the ecological constraints on additional biomass harvesting (www.ecopedia.be/biomassa/ecologische_randvoorwaarden_oogst_biomassa)(Cosyns *et al.*, 2015).

6.2.2. Smart land-use for multiple ecosystem services

The results of chapter five also allow to make recommendations for management and policy. Some of the results are very site specific and linked to the spatial lay-out of the study area, but some more general lessons can be drawn that are relevant for all pine plantations on former heathlands (a widespread habitat type) and even to other forest types and ecosystems.

Management recommendations

- Wood harvesting in pine stands on former heathland has mixed effects on conservation of different species groups, allowing a land sharing approach. With a smart harvesting plan based on suitable ecological models (cf. metapopulation based Zonation software) it is possible to reinforce and connect habitats for GHS species without threatening populations of forest species and while largely retaining yield from harvesting (87% of the yield retained in our study). The optimal spatial lay-out is highly context dependent and should be determined by mapping the distribution of focal species. However, from our study a more general valuable strategy can be drawn: primarily harvesting stands next to existing open patches.

- Recreation has a negative impact on the distribution of some species, asking for a land sparing approach to combine both management goals. The lay-out of the recreation infrastructure can be changed in function of the distribution of vulnerable focal species (context-dependent). More general, it seems a valuable strategy to concentrate recreation in the border of a forest and nature complex, safeguarding a central area for biodiversity conservation.

- Information on spatial distribution of focal species, recreation pressure and harvest can be valuable information for forest managers to adapt management plans to better integrate the three management goals. Future inventory of the focal species, recreation pressure and harvest will allow managers to constantly evaluate management actions and keep on adapting management plans to answer possible future changes.

Policy recommendations

- To combine the wood harvest, recreation and biodiversity conservation, a smart land management can help in better dealing with trade-offs and synergies on a landscape scale (see also Cordingley *et al.* (2016)). However, trade-offs are intensified when habitat is more fragmented (Cordingley *et al.*, 2015). This demonstrates the need for large areas of nature and forest to increase resilience and robustly deliver ecosystem services and maintain biodiversity.

6.2.3. Towards ecosystem stewardship

The way the Bosland project originated, has developed and is currently managed is a nice illustration of a change trajectory from classic resource management towards collaborative management, embracing different aspects of ecosystem stewardship. From the successes achieved in Bosland, different recommendations can be formulated for forest and nature managers and for policy makers.

Management recommendations

- A change in Flemish legislation opened a window of opportunity to rethink the traditional forest management approach and experiment with new participatory settings to better adhere to the dynamics of change. To achieve this, ANB shifted from a directive top down style to a role as facilitator to promote collaboration and a multi-perspective approach involving stakeholders from the start by co-creating a shared long term vision. Such a new role of catalyst and facilitator was key in the Bosland transition trajectory and other forest managers could learn from this example

- Interaction with (also citizen) scientists can yield valuable information that can be used for smart land management and adaptive management. Such an adaptive management approach with a closer collaboration between scientists and practitioners is better suited to face grand challenges such as climate change and biodiversity loss.

- A more general recommendation for forest managers is to adopt some of the principles of ecosystem stewardship in the management of the forest and nature areas and shift away from steady state resource management.

Policy recommendations

- Ecosystems operate on different spatial scales while forest management is mostly organised according to territorial borders. To fully maintain, promote and restore ecosystem services and biodiversity collaborations between forest and nature managers on a landscape scale should be stimulated. Co-managing larger areas can increase public support and cost-efficiency, while more management goals can be reached without threatening biodiversity conservation. If possible, also collaborations with private forest owners could be set up. This is a challenge, because few private forest owners are prepared to change their management practices (Sebruyns & Luyssaert, 2006). Good provision of information to private forest owners seems crucial for acceptance of policy instruments (Serbruyns & Luyssaert, 2006).

- New governance approaches are urgently needed to safeguard biodiversity and ecosystem services in a transition towards a bio-based economy. This requires resources for experimentation and embedding (space, time, money) and tools for replication and upscaling (Gorissen *et al.*, in progress).

- Successful, innovative examples of a more holistic management, such as Bosland, could be connected in a learning network on a larger (European/Global) scale to exchange good practices.

6.3. Perspectives for further research

6.3.1. Further research in Bosland

There remain some key questions for research in Bosland. In relation to chapter two, it would be interesting to see how the Bosland project further develops and how it interacts with other projects and with the forest management regime. Given the participatory background, Bosland is also a highly interesting case to further study the possibilities to collaborate with private forest owners to better reach management goals on a landscape scale (cf. Serbryns & Luyssaert (2006)). Concerning biomass harvest potential and technical, economic and soil limitations in Bosland, it would be interesting to study the effect of the planned large-scale conversion from pine plantations to mixed broadleaf stands. The conversion will definitely influence technical and economic harvest potential. For instance, crown wood from broadleaf trees is more desired by private households as fuel, which may strongly decrease the economic potential for industrial application as a biofuel. Also the impact of conversion and of whole tree harvesting of deciduous trees on soil fertility would be an interesting research question. As already mentioned in the discussion of chapter 4, conversion to broadleaf tree species, generally increases nutrient cycling and improves soil fertility. It would be specifically interesting to study this in Bosland, because results can be compared with the results of chapter 4 of this work. Concerning the integration of wood harvest and recreation with biodiversity conservation, there is a clear need for more data on some key species before the proposed measures can be applied. For nightjar for instance the current research by University of Hasselt, will provide adequate information concerning management and recreation pressure (Evens, 2011). For other species (and definitely for species protected by European directives such as woodlark (*Lullula arborea*)), further inventory would be necessary. When management measures on recreation and harvest would be adopted (after consideration of the forest managers and possible public consultation) it would be highly interesting to evaluate impact on the focal species to improve the models and to keep optimizing future management.

Of course there are also many other opportunities for further research in Bosland, independently of the work presented in this dissertation. The possibility to do research on forest and nature management on a landscape scale is already quite unique for Flanders. Moreover there is a good research infrastructure in Bosland, for instance with the presence of the FORBIO research plot, a large scale experiment on the functioning of tree species mixtures and monocultures (Verheyen *et al.*, 2013). The development of BOSLAB and a scientific board will further welcome scientists in Bosland and promote collaboration between researchers and managers in the future.

6.3.2. Biomass harvest potential and limitations

The central research question for the transition to the bio-based economy remains, how much woody biomass can be harvested in a sustainable way from forests? In our study we have given some answers and we shed light on the different constraints for harvesting additional biomass in pine stands. There is, however, a strong need to verify the results from our studies in other systems and in other regions to fully unravel how much woody biomass can be harvested on larger spatial scales without deteriorating ecosystem integrity and durability.

The technical constraints we identified in chapter three are probably quite generalizable to other pine stands, but other systems might need other technologies and might face different limitations. In broadleaved stands for instance, crown wood is traditionally often sold as fuel wood to individuals who harvest mostly manually (Jespers *et al.*, 2012). Given the rising energy wood prices it could be expected that exploitation companies will enlarge their share in this segment leading to a further mechanization. This asks for more research on technical constraints in these forest types, for instance on the use of tracked machines to reduce compaction of soils with a finer texture (Ampoorter *et al.*, 2012). Many research questions also remain on the technical constraints of harvesting woody biomass from outside forests. Recently there has been some research on different biomass harvesting systems in short rotation coppice (Wolbert-Haverkamp & Musshoff, 2014), but on the technical harvesting from hedgerows for instance knowledge is scarce and further research is needed (Van Den Berge, 2014).

Economic constraints are more variable over time and space. It would be highly relevant to further study the impact of forest fragmentation on economics of wood and biomass harvesting. To fuel the upcoming transition to a bio-based economy, there is also a need for more knowledge on the economic viability of harvesting different kinds of woody biomass from other forest and landscape types (e.g. broadleaf stands, hedgerows, short rotation coppice). Also constraints on other types of biomass should be investigated, such as herbaceous biomass from low-input, high-diversity systems (Van Meerbeek *et al.*, 2015a; Van Meerbeek *et al.*, 2015b).

As demonstrated in chapter four, soil fertility is a major concern when additional biomass is harvested from pine stands, at least in Bosland. The uncertainty that occurred in the modelling was partly due to the limited knowledge on important soil processes such as weathering. Also other important research should be done on the impact of different mitigation measures on long term soil fertility, such as (i) longer rotation periods; (ii) other silvicultural systems; (iii) fertilization ; (iv) leaving crowns in the stand for one year so needles are shed. There is also need for similar research in other forest systems, (i) with a history of lower acidifying deposition; (ii) on different soil types; (iii) and with different dominant tree species. Putting together the results from different types of research on impact of soil fertility can result in more general lessons for managers, so also in the future there will be need for review such as the one of Paré & Thiffault (2016).

Harvesting additional biomass also affects other ecosystem services and processes that were not investigated within the current research. It would be highly relevant to study the impact of additional biomass harvest on a stand scale (i) on biodiversity conservation of different species groups; (ii) on carbon sequestration; (iii) on visual preference of visitors; (iv) and on nutrient leaching.

In the meantime large biomass plants are often mainly provided with pellets from overseas, in Belgium mostly from North America (Sikkema *et al.*, 2010). Some critics exist on the fossil fuels used in international transport and on the sustainability of the harvest in the country of origin (Schulze *et al.*, 2012). Sustainable harvesting of additional woody biomass should be a prerequisite for the award of subsidies for bio-energy. There is a clear need for more research on the sustainability of the entire production chain of different sources of woody energy. The use of an environmental impact assessment combined with ecosystem service valuation seems a valuable research strategy to obtain this (cf. Schaubroeck *et al.* (2016)).

6.3.3. Smart land-use for multiple ecosystem services

Grand challenges, such as biodiversity loss and climate change ask for a smart land use that combines different functions in an efficient way on a landscape scale, definitely in highly urbanized regions such as Flanders. In chapter five we looked at the impact of wood harvesting and recreation on biodiversity conservation in a landscape dominated by pine stands and open heathland patches and we proposed management measures that smartly combine the different targets. However this is only one example and more research on different relations, on different ecosystem services and in different landscapes is needed. For instance, the impact of biodiversity on recreation, of biodiversity on wood production and of wood harvest on recreation could also be determined (even in the same study area). Or the impact of wood and biomass production could be determined on other ecosystem services on a landscape scale, such as pollination, capturing of fine particulate matter and pest control. The same questions should be answered in different landscapes to reach a smart land management on larger scales.

It would also be highly interesting to more specifically study the impact of additional biomass harvesting on a landscape level (so compare WTH with SOH). The results on ecological constraints on a stand level could be combined with an analysis of the impact of additional harvesting on different services on a landscape scale. When the impact on different services could satisfyingly be quantified and if it would be possible to monetary valuate the different services, the economic impact of additional harvesting could be determined and these costs could be internalized in the price of woody biomass. However, it should be stressed that our current knowledge is not sufficient to determine the exact cost, certainly given the different pitfalls in monetary valuation (see box 1).

6.3.4. Ecosystem stewardship

To effectively manage ecosystems, under the current challenges and given the increased demand for participation by stakeholders, there is a need for different governance models such as ecosystem stewardship. Given the urgency of grand challenges such as biodiversity loss and climate change, best practices need to be replicated, scaled up and embedded to accelerate the transition to more sustainable systems. Comparing different styles of participation and governance in forest and nature management and evaluating their effectivity can help to discover best practices. More systemic solutions are required to overcome silo policy and politics. Interdisciplinary research projects for instance, bring together researchers from different fields, such as forest and nature policy, ecological economics, ecology and resource management. When the principles of ecosystem stewardship are adopted, researchers will also take up new responsibilities and different roles (Chapin *et al.*, 2010). To evolve towards adaptive management (Temperli *et al.*, 2012; Williams & Brown, 2014) and locally attuned stewardship in forest and nature management, there will be need for a constant dialogue between researchers, managers and policy makers and the different stakeholder groups throughout the decision-making process.

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8. Appendices

8.1. General interview guide

The generalized version of the questions used in the interview are listed below. Before every interview the relevance of each question was evaluated, looking at the history of the organization of the interviewee. Every main question was posed, the secondary questions were posed if extra information was necessary. Personalized questions were added whenever the interviewee came up with relevant elements for the transition analysis.

1. Could you introduce yourself and your organization and explain how you got involved in the Bosland project?
    a. How was the situation and the relation with the forest of your organization before the formation of the partnership?
    b. What were your first thoughts on the idea of forming a partnership?
2. Could you explain about the role of your organization in the project? (with help of a picture of the management structure, figure 3)
3. To which extent was your organization able to participate?
    a. Do you feel like your organization has played a role in the development of the long term vision? To which extent?
    b. Do you feel like your organization has an impact on the actual management of the forests? To which extent?
4. How is the relation of your organization with the (other) partners? Minor, equal, superior?
5. According to you, did the Bosland parliament have an impact on the policy and management of the project?
    a. To which extent did they participate?
    b. How is the cooperation with the other bodies in the management structure?
6. What were strengths, weaknesses, opportunities and threats in the Bosland project?
    a. What were crucial factors/events/people… in the formation process?
    b. What are things you look different at nowadays? What did you learn? What would you do different?
7. How do you think the future of the Bosland project will look like?

*Table 8.1: Rejected strategies for combined harvest of logs and wood chips for clear-cuts (C5-10) and early thinnings (T5-7) in Flanders. * Ø = top bucking diameter*

Strategy	Explanation of strategy	Motivation of selection/rejection by board of experts
C5	Harvester + forwarder for stem-wood (Ø* 7 cm) + One year later: forwarder for crown wood + chipper on roadside	Economically less feasible to come back one year later; Drying of woody biomass expected to be only marginal.
C6	Harvester + forwarder for stem-wood (Ø 12 cm) + One year later: forwarder for crown wood + chipper on roadside	Economically less feasible to come back one year later; Drying of woody biomass expected to be only marginal.
C7	Harvester + forwarder for stem-wood (Ø 7 cm) + Bundler to collect crown wood + forwarder for crown wood + chipper on roadside/at energy plant	Bundler seems economically unfeasible for forestry in Flanders due to low forest area, small forest stands and short hauling distances.
C8	Harvester + forwarder for stem-wood (Ø 12 cm) + Bundler to collect crown wood + forwarder for bundles + chipper on roadside/at energy plant	Bundler seems economically unfeasible for forestry in Flanders due to low forest area, small forest stands and short hauling distances
C9	Harvester + forwarder for stem-wood (Ø 7cm) + Biobaler to collect crown wood + forwarder for bales + chipper on roadside/at energy plant	Interesting option, but the board of experts believes that the biobaler is not suited to operate in the rough terrain conditions of a clear-cut (stumps, terrain topography, …)
C10	Harvester + forwarder for stem-wood (Ø 12 cm) + Biobaler to collect crown wood + forwarder for bales + chipper on roadside/at energy plant	Interesting option, but the board of experts believes that the biobaler is not suited for operating in the rough terrain conditions of a clear-cut (stumps, terrain topography, …)
T5	Harvester + whole trees chipped in stand by integrated mobile chipper	Highly specialized integrated chipper appears to be economically unfeasible for forestry in Flanders due to low forest area and small forest stands + expensive to get machine in Flanders
T6	Whole tree harvested and chipped by integrated mobile chipper with harvester-head	Theoretically interesting because of probably lower chip contamination. Doubtful if this strategy can be made operational + Too specialized for Flemish forestry context.
T7	Harwarder for whole trees + chipper on roadside	Harwarder seems economically unfeasible for forestry in Flanders + expensive to get machine in Flanders

Table 8.2: Calculated machine cost for the forestry equipment used in the different harvest experiments (SMH = scheduled machine hours)

Machine	**Harvester**	**Excavator**	**Forwarder**	**Trailer**	**Tractor**	**Road-side chipper**	**Tractor**	**Mobile chipper**	**Tractor**
Type	**John Deere 1170E**	**Hyundai R145**	**John Deere 1010E**	**Own manu-facturing**	**Valtra 8950**	**Jenz HEM420**	**Valtra T191**	**Greentec 952**	**Valtra N141**
Used in harvest strategy	**C1, C2, C3, C4, T1, T2**	**T4**	**C1, C2, C3, C4, T1, T3**	**T4**		**C1, C2, T1, T3, T4**		**C3, C4, T2**	
Purchase price (k€)	375	110	235	110	85	220	100	180	130
Economic life (year)	10	7.5	10	7.5	7.5	5	7.5	5	5
Salvage value (k€)	75	30	30	70	15	30	25	54	39
Average value of yearly investment (k€)	240.00	75.33	142.75	92.67	54.67	144.00	67.50	129.60	93.60
Interest, insurance and taxes (k€.year^{-1})	24.00	7.53	14.28	9.27	5.47	14.40	6.75	12.96	9.36
Annual use (SMH)	2000	1500	2000	1500	1500	1500	1500	1500	1500
Total fixed cost (€.SMH^{-1})	**27.00**	**12.13**	**17.39**	**9.73**	**9.87**	**34.93**	**11.17**	**25.44**	**18.37**
Fuel consumption (L.h^{-1})	12.79	18.2	11.36	/	8.29	/	35.66	/	19.19
Fuel & lubricant (€.h^{-1})	11.91	16.94	10.58	/	7.72	/	33.20	/	17.87
Repair & maintenance (€.h^{-1})	8.12	4.48	5.14	2.01	2.41	23.52	5.64	30.00	15.00
Use efficiency (%)	73.90	74.42	77.84	83.04	83.04	35.43	35.43	88.10	88.10
Total variable cost (€.SMH^{-1})	**14.80**	**15.94**	**12.23**	**1.67**	**8.41**	**8.33**	**13.76**	**26.43**	**28.95**
Overhead on variable cost (20 %)	2.96	3.19	2.45	0.33	1.68	1.67	2.75	5.29	5.79
Total cost without labour (€.SMH^{-1})	**44.76**	**31.27**	**32.07**	**11.73**	**19.96**	**44.93**	**27.68**	**57.16**	**53.12**
Labour (€.SMH^{-1})	20	20	20	20		20		20	
Total cost (€.SMH^{-1})	**64.76**	**51.27**	**52.07**	**51.69**		**92.62**		**130.28**	

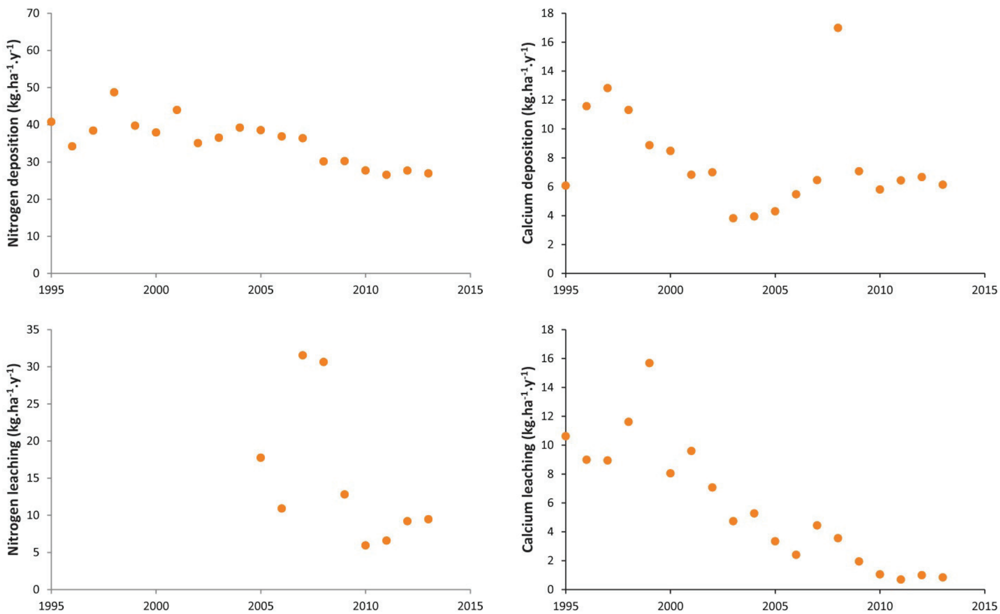

Figure 8.1: Illustration of changes in deposition and leaching of nitrogen and calcium in the Corsican pine dominated level II plot in Ravels. Similar data for deposition and leaching of all nutrients are available, but are not shown here.

Table 8.3: Differences in stock for different nutrients in different compartments of the forest floor and the mineral soil between stands within both Locations. (= significant differences between stands; n.s. = non-significant differences between stands; n.a. = not applicable for this compartment).*

		mass	C	N	pH	Total					Available				
						P	Ca	K	Mg	Al	P	Ca	K	Mg	Al
Lommel	**Litter layer**	n.s.	*	*	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Understorey	n.s.	n.s	n.s	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Fine dead wood	n.s.	n.s	n.s	n.a.	*	n.s	n.s	*	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Coarse dead wood	n.s.	n.s	n.s	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Soil 0 to 10 cm	n.a.	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s
	Soil 10 to 20 cm	n.a.	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s
	Soil 20 to 30 cm	n.a.	*	n.s	n.s	n.s.	n.s	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s
	Soil 30 to 40 cm	n.a.	*	*	n.s	n.s.	n.s	n.s	n.s	n.s	n.s.	n.s	n.s	n.s	n.s
	Soil 40 to 50 cm	n.a.	*	*	n.s	*	n.s	n.s	n.s	n.s	*	*	*	n.s	n.s
Overpelt	**Litter layer**	n.s.	n.s.	n.s	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Understorey	n.s.	n.s.	n.s	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Fine dead wood	n.s.	n.s.	n.s	n.a.	n.s.	n.s	n.s	*	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Coarse dead wood	n.s.	n.s.	n.s	n.a.	n.s.	n.s	n.s	n.s	n.s	n.a.	n.a.	n.a.	n.a.	n.a.
	Soil 0 to 10 cm	n.a.	n.s.	n.s.	n.s.	n.s.	*	*	n.s	n.s	n.s.	*	n.s	n.s	n.s
	Soil 10 to 20 cm	n.a.	n.s.	n.s.	*	n.s.	n.s	n.s	n.s	n.s	n.s.	*	n.s	*	n.s
	Soil 20 to 30 cm	n.a.	n.s.	n.s.	*	n.s.	n.s	n.s	n.s	n.s	n.s.	n.s	n.s	*	n.s
	Soil 30 to 40 cm	n.a.	n.s.	*	*	n.s.	n.s	n.s	n.s	n.s	*	n.s	n.s	*	*
	Soil 40 to 50 cm	n.a.	n.s.	n.s.	n.s.	n.s.	*	n.s	n.s	n.s	n.s.	n.s	n.s	*	n.s

Table 8.4 Biomass exported from the different clear-cut stands in Lommel (C1-C4) and the thinned stands in Overpelt (T1-T4)

	Export (t/ha)	
Stand	**Stems**	**Crowns**
C1	167.66	39.13
C2	174.91	34.66
C3	164.06	36.00
C4	175.44	33.89
T1	24.29	16.05
T2	24.65	16.29
T3	28.56	18.87
T4	27.26	18.01

Table 8.5 Average concentration (with standard deviation) of the different nutrients in the stems and crowns in the clear-cut stands in Lommel and in the thinned stands in Overpelt.

		Clear-cut		Thinning	
		Average	**sd**	**Average**	**sd**
Crowns	**K**	0.187%	0.019%	0.141%	0.043%
	Mg	0.040%	0.007%	0.024%	0.008%
	Ca	0.172%	0.025%	0.056%	0.025%
	Al	0.022%	0.008%	0.017%	0.014%
	P	0.038%	0.006%	0.021%	0.009%
	C	50.849%	3.351%	49.599%	3.625%
	N	0.771%	0.118%	0.419%	0.154%
	S	0.055%	0.012%	0.066%	0.031%
Stems	**K**	0.060%	0.006%	0.090%	0.012%
	Mg	0.016%	0.002%	0.020%	0.003%
	Ca	0.103%	0.010%	0.087%	0.011%
	Al	0.009%	0.001%	0.017%	0.002%
	P	0.010%	0.001%	0.009%	0.001%
	C	50.475%	5.020%	50.752%	6.646%
	N	0.368%	0.037%	0.371%	0.049%
	S	0.053%	0.005%	0.064%	0.008%

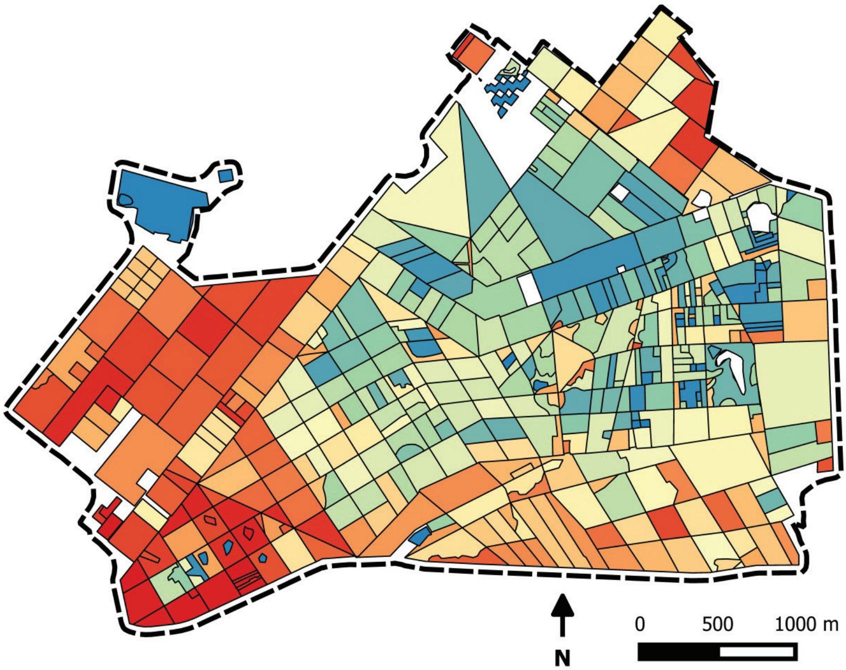

Figure 8.2: Classification of the forest stands based on recreational pressure. Stands in reds have the highest recreation pressure based on the number of visitors on adjacent roads (see text for details on calculation), stands in blues have the lowest recreation pressure.

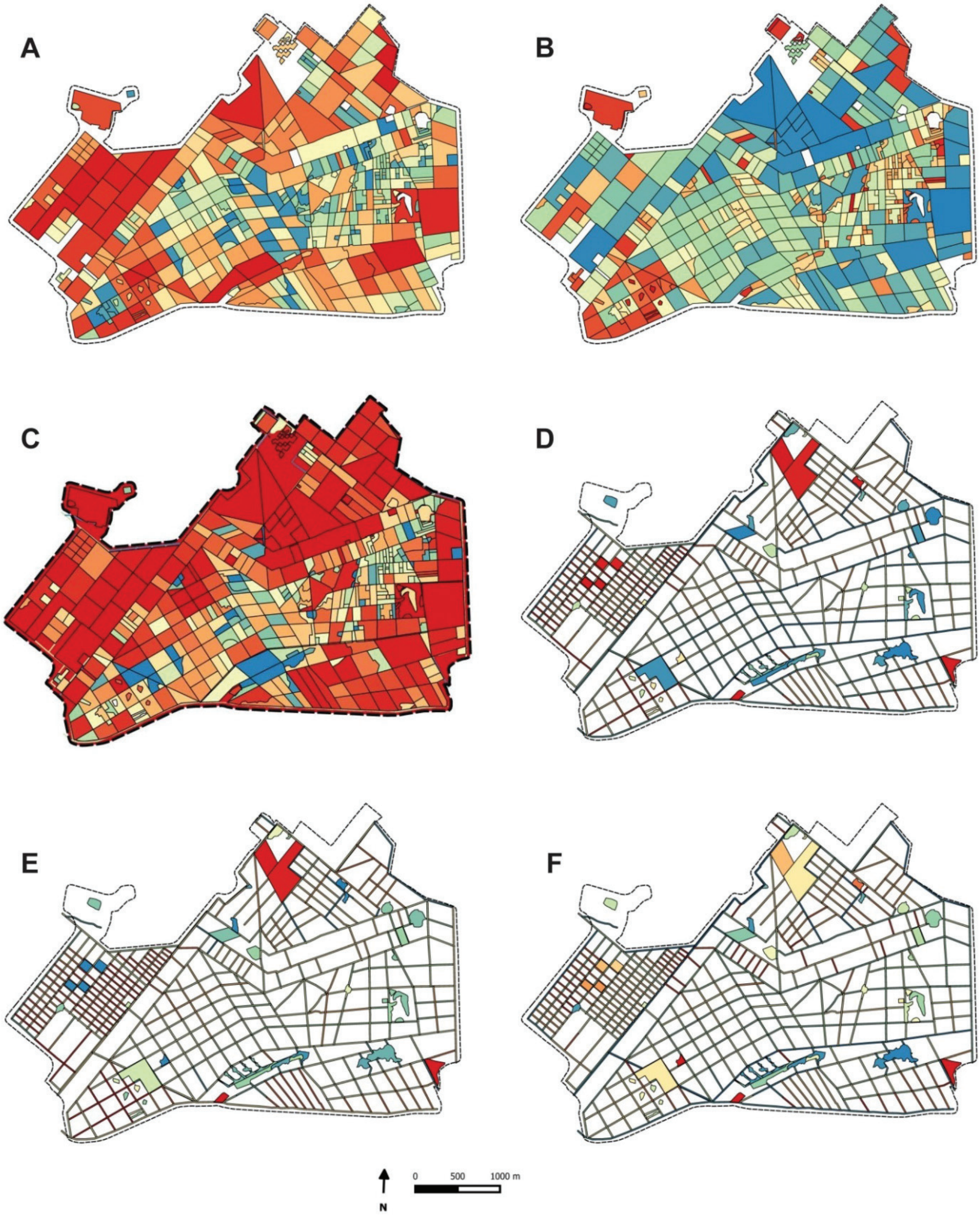

Figure 8.3: Habitat suitability maps for the indicator species, based on the GLMs, blues stand for a high habitat suitability, reds for a low habitat suitability. A Coal tit, B Crested tit, C Nightjar, D Small heath, E Grayling, F Northern dune tiger beetle.

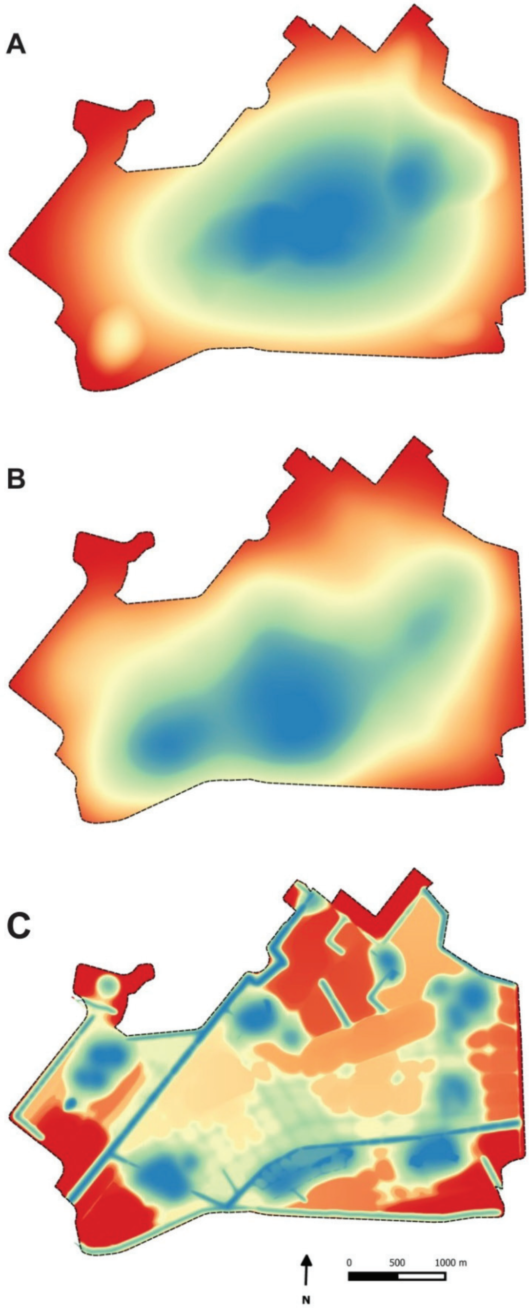

Figure 8.4: Zonation rank of the landscape for the different species groups, blues stand for a high conservation value for a species group, red for a low conservation value. A the forest species, B the species that depend on both forest and open habitat, C the GHS species.

Table 8.6: Overview of the habitat features of forest stands and open patches that were used as predictors in the habitat suitability analysis. For every species generalized linear models were built for every possible combination of habitat features for the relevant habitat type.

Numeric habitat features			
Forest stands		**Open habitat**	
Feature	Unit	Feature	Unit
Area	ha	Area	ha
Recreation	score	Recreation	score
Border with open habitat	m	Orientation	°

Categorical habitat features					
Forest stands					**Open**
Age class	Structure	Type	Mixture	Dominant tree species	Surface type
1-20	High forest	Coniferous	Uniform	*Pinus sylvestris*	Orchard
21-40	Coppice with standards	Broadleaves	Mixed groups	*Pinus corsicana*	Tree litter path
41-60	NA	Mix Coniferous - Broadleaves	Mixed on tree level	*Quercus robur*	Heathland/ grassland
61-80		Forest Reserve	NA	*Quercus rubra*	Clear-cut
81-100		Open		*Larix sp*	Agriculture
101-120				*Fagus sylvatica*	Tarmac road
NA				*Pseudotsuga menziessii*	Sandy
Uneven				Other	Plantation
				NA	

Table 8.7: Average coefficients from the top GLMs for the different species

	Forest stands								Open patches				
	Intercept	Border	Recreation	Area	Age class		Management		Intercept	Area	recreation	Patch type	
Coal tit	-1.100	-0.468	-0.540	0.044	1-20	-0.021	High forest	0					
					21-40	0.029							
					41-60	-0.018	Coppice with standards	-2.708					
					61-80	0.091							
					81-100	0.657							
					101-120	0	NA	-0.146					
					NA	-6.229							
					uneven	0.105							
Crested tit	-0.31	-0.024	-0.077	0.155			High forest	0					
							Coppice with standards	-11.957					
							NA	-2.04					
Nightjar	-2.192	1.040	-0.095	-2.685	1-20	-0.357			15.340	-1.737	0.044	Orchard	-33.664
					21-40	0.713						Tree litter path	-16.447
					41-60	-0.023						Heathland/ grassland	-16.386
					61-80	0.102						Clear-cut	-16.418
					81-100	1.843						Agriculture	-31.205
					101-120	0.000						Sandy	-14.770
					NA	-14.387						Plantation	0
					uneven	0.795							
Small									-2.3745	0.6985	-0.362	Orchard	-18.587

heath					Tree litter path	-0.692
					Heathland/ grassland	1.652
					Clear-cut	-15.266
					Agriculture	-23.361
					Tarmac road	1.186
					Sandy	1.954
					Plantation	0.000
Grayling		-0.7273	0.1621	-0.591	Orchard	-20.506
					Tree -litter path	-18.685
					Heathland/ grassland	-1.027
					Clear-cut	0.298
					Agriculture	-23.122
					Tarmac road	-1.028
					Sandy	-0.347
					Plantation	0.000
Northern dune tiger beetle		-22.693	0.4666	0.0103	Orchard	0.340
					Tree litter path	2.271
					Heathland/ grassland	19.162
					Clear-cut	2.081
					Agriculture	-1.061
					Tarmac road	19.422
					Sandy	21.985
					Plantation	0.000

9. Curriculum vitae

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2008-2010 MSc. In Bioscience Engineering, Forest and Nature Management, Ghent University (Belgium) . Including an Erasmus exchange in Umeå (Sweden) at Swedish University of Agricultural Science in 2009.

2005-2008 Bsc. in Bioscience Engineering: Land and Forest Management, Ghent University (Belgium)

1999-2005 Secondary School, Latin-Mathematics, Koninklijk Atheneum Mariakerke, Ghent.

Professional experience

2015-2016 Researcher on the LIFE Pays Mosan project: Restoration of species rich calcareous grassland on former agricultural areas. ForNaLab, Ghent University.

2011-2015 PhD research at Ghent University in collaboration with VITO

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Unit Transition Energy and Environment

Scientific activities

Areas of interest and expertise

Ecosystem services, woody biomass harvest, spatial conservation planning, landscape ecology, transition management, sustainable forest management, plant community ecology, biodiversity – ecosystem functioning, restoration ecology, global change impacts on biodiversity, temperate forests, species-rich grasslands.

Peer-reviewed scientific articles included in Web of Science

Published or in press

Vangansbeke, P., De Schrijver, A., De Frenne, P., Verstraeten, A., Gorissen, L., Verheyen, K. 2015. Strong negative impacts of whole tree harvesting in pine stands on poor, sandy soils: A long-term nutrient budget modelling approach. Forest Ecology and Management 356: 101-111. IF 2015: 2.660.

Vangansbeke, P., Osselaere, J., Van Dael, M., De Frenne, P., Gruwez, R., Pelkmans, L., Gorissen, L., Verheyen, K. 2015. Logging operations in pine stands in Belgium with additional harvest of woody biomass: yield, economics, and energy balance. Canadian Journal of Forest Research. 45: 987-997. IF 2015: 1.683.

Vangansbeke, P., Gorissen, L., Nevens, F., Verheyen, K. 2015. Towards co-ownership in forest management: Analysis of a pioneering case Bosland (Flanders, Belgium) through transition lenses. Forest Policy and Economics 50: 98-109. IF 2015: 1.856.

De Frenne, P., Rodríguez-Sánchez, F., De Schrijver, A., Coomes, D.A., Hermy, M., **Vangansbeke, P.**, Verheyen, K. 2015. Light accelerates plant responses to warming. Nature Plants, 1: 15110.

De Frenne, P., Coomes, D.A., De Schrijver, A., Staelens, J., Alexander, J.M., Bernhardt-Römermann, M., Brunet, J., Chabrerie, O., Chiarucci, A., den Ouden, J., Eckstein, R.L., Graae, B.J., Gruwez, R., Hédl, R., Hermy, M., Kolb, A., Marell, A., Mullender, S.M., Olsen, S.L., Orczewska, A., Peterken, G., Petrik, P., Plue, J., Simonson, W.D., Tomescu, C.V., **Vangansbeke, P.**, Verstraeten, G., Vesterdal, L., Wulf, M., Verheyen, K. 2014. Plant movements and climate warming: intraspecific variation in growth responses to nonlocal soils. New Phytologist 202: 431-441. IF 2013: 6.545.

Verheyen, K., Buggenhout, M., **Vangansbeke, P.**, De Dobbelaere, A., Verdonckt, P., Bonte, D. 2014. Potential of Short Rotation Coppice plantations to reinforce functional biodiversity in agricultural landscapes. Biomass and Bioenergy 67: 435-442. IF 2014: 3.394

Gruwez, R., De Frenne, P., De Schrijver, A., Leroux, O., **Vangansbeke, P.**, Verheyen, K. 2014. Negative effects of temperature and atmospheric depositions on the seed viability of common juniper (Juniperus communis). Annals of Botany 113: 489-500. IF 2014: 3.654

Baeten, L., **Vangansbeke, P.**, Hermy, M., Peterken, G., Vanhuyse, K., Verheyen, K. 2012. Distinguishing between turnover and nestedness in the quantification of biotic homogenization. Biodiversity and Conservation 21: 1399-1409. IF 2012: 2.264

Vangansbeke, P., Blondeel, H., Landuyt, D., De Frenne, P., Gorissen, L., Verheyen, K. In press. Spatially combining wood production and recreation with biodiversity conservation. Biodiversity and Conservation, in press: doi: 10.1007/s10531-016-1135-5.

Gruwez, R., De Frenne, P., Vander Mijnsbrugge, K., **Vangansbeke, P.**, Verheyen, K. 2016. Increased temperatures negatively affect Juniperus communis seeds: evidence from transplant experiments along a latitudinal gradient. Plant Biology, in press: doi: 10.1111/plb.12407.

Submitted

Gruwez, R., Hommel, P., De Frenne, P., De Schrijver, A., Huiskes, H., de Waal, R., **Vangansbeke, P.**, Verheyen, K. Submitted. Effects of management actions on the recruitment of threatened common juniper populations (*Juniperus communis*). Plant Ecology and Evolution. Accepted with revicisions.

Gruwez, R., De Frenne, P., De Schrijver, A., **Vangansbeke, P.**, Verheyen, K. Submitted. Climate warming and atmospheric deposition affect seed viability of common juniper (*Juniperis communis*) via their impact on the nutrient status of the plant. Ecological Research, accepted with revisions.

Peer-reviewed articles in other journals

Vangansbeke, P., Blondeel, H., Verheyen, K. 2015. Wood harvest, recreation and vulnuerable species together in a forest? It is possible! Natuur. focus 14: 158.

Vangansbeke, P., Osselaere, J., Gorissen, L., Verheyen, K. 2014. A comparison of different harvesting chains with additional biomass harvests in pine stands. Bosrevue 47: 9-16.

Vangansbeke, P., Verheyen, K., Van Beek, E., Muys, B. 2012. Forest Ecosystem Services: what forests deliver to people. Bosrevue 41: 11-16.

Vanhuyse, K., **Vangansbeke, P.**, Peterken, G. 2012. Lady Park Wood: The Loss of Ground Flora. Natur Cymru - Nature of Wales 43: 15-19.

Schauwvliege, W., De Smedt, P., **Vangansbeke, P**., Van Camp, B. 2016. Moth inventarisations in forests: do we sample the whole community? Natuur.focus, in press.

Blondeel, H., **Vangansbeke, P.** 2016. Biodiversity, recreation and biomass production in Flemish forests: trade-offs and synergies. Bosrevue, in press.

Book chapters

Vangansbeke, P., Gorissen, L., Verheyen, K. 2013. Bosland: Application of the Ecosystem Services Concept in a New Style of Forest Management. Chapter 41 in: Jacobs,S., Dendoncker, N., Keune, H. (Eds.), Ecosystem Services. Elsevier, Boston, pp. 397-404.

Scientific reports

Osselaere, J., **Vangansbeke, P. 2013**. Technics and strategies for the harvest of woody biomass: results from terrain experiments exectued in Bosland. Inverde, Brussels, Belgium.

MSc. Thesis

30 years of change (1979-2009) in the tree-layer of the forest reserve Lady Park Wood (UK). ForNaLab, Ghent University. Supervisors: Prof. Dr. Ir. Kris Verheyen & Prof. Dr. Ir. Martin Hermy, tutor: Dr.Ir. Lander Baeten.

Participation in congresses, symposia or workshops

Oral presentations

Vangansbeke, P. 2015. Biomassa-oogst in Bosland. Symposium: Bossen en hun groeiplaats: toop oor, to rich and too hard. Gontrode, Belgium.

Bracke, R., De Smedt, P., **Vangansbeke, P.**, Mertens,J., Verheyen, K. 2015. Ups and Downs of Moths. Starters in Het Natuur- En Bosonderzoek. Brussels, Belgium.

Vangansbeke, P., Gorissen, L., Nevens, F., Verheyen, K. 2014. Transition Towards Co-ownership in Forest Management: Bosland (Flanders, Belgium) as a Frontrunner. Resilience conference. Montpellier, France.

Vangansbeke, P., Vanhuyse, K. 2010. Thirty years of change in the herb and tree-layer in Lady Park Wood. Startersdag in Het Natuur- En Bosonderzoek. Brussels, Belgium.

Poster presentations

Vangansbeke, P., Landuyt, D. 2014. Integration of Ecosystem Services in Bosland. BEES Christmas Market. Gembloux, Belgium.

Osselaere, J., **Vangansbeke, P.**, Verheyen, K. 2013. Detirmining optimal technics and strategies for the harvest of woody biomass. Startersdag in Het Natuur- En Bosonderzoek. Brussels, Belgium.

Vangansbeke, P., Gorissen, L., Verheyen, K. 2012. Smart Land Use for Bio-based Economies: Simultaneous Optimization of Biomass Harvesting and Other Ecosystem Services in Bosland. TEEBelgium Conference. Brussels, Belgium.

Supervision of MSc thesis students as a tutor

Osselaere, J. 2013. Technics and strategies for biomass harvesting. Ghent University. Supervisor: Prof. Dr. Ir. Kris Verheyen.

De Vreese, S. 2014. Comparison of biomass production and cost-efficiency of short rotation forestry and poplar plantations. Ghent University. Supervisor: Prof. Dr. Ir. Jan Mertens.

Bracke, R. 2015. Ups and downs of moths. Ghent University. Supervisor: Prof. Dr. Ir. Jan Mertens, co-tutor: Ir. Pallieter De Smedt.

Blondeel, H. 2015. Biodiversity and biomass production in Flemish forests: synergies and trade-offs. Ghent University. Supervisor: Prof. Dr. Ir. Kris Verheyen.

Schauwvliege, W. in preparation. Vertical distribution of moth species richness in temperate forests. Ghent University. Supervisor Prof. Dr. Ir. Jan Mertens, co-tutor: Ir. Pallieter De Smedt.

Scientific side-activities

Courses

2014 Statistics – Multilevel Analysis for Grouped and Longitudinal Data. Ghent, Belgium.

2013 Linking Ecological Research and Natural Systems Management. Aarhus, Denmark.

2012 Statistics – Introduction to R. Ghent, Belgium.

2012 Integrated Assessment of Ecosystem Services – From Theory to Practice. Amsterdam, The Netherlands

2012 Communication and Negotiation Skills

2012 Creative thinking

2011 Scientific English and Presentation Skills

Projects

2012-2013 Co-coordinator of the KOBE B3 project: Technics and strategies for the harvest of biomass from forests. ForNaLab, Ghent University – Flemish Institute for Technological Research (VITO). In cooperation with the Flemish Agency of Forest and Nature Management (ANB) and Inverde

2012 Team member of the KOBE B1 project: Carrying capacity of forests for an increased harvest of biomass. ForNaLab, Ghent University– Flemish Institute for Technological Research (VITO). In cooperation with Flemish Agency of Forest and Nature Management (ANB) and Flemish Institute for Nature and Forest Research (INBO).

Review tasks for international journals

Forest Policy and Economics.
Ecography.