Norwegian University of Life Sciences
Faculty of Environmental Science and Technology
Department of Ecology
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Philosophiae Doctor (PhD)
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Integrating life cycle assessment and forest modelling for environmental and economic assessment of forest based bioenergy in Norway

Integrering av livsløpsanalyse og skogmodellering for analyse av miljømessige og økonomiske konsekvenser av bruk av skogbasert bioenergi i Norge

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PhD Supervisors

Prof. Dr. Birger Solberg
Department of Ecology and Natural Resource Management (INA)
Norwegian University of Life Sciences (NMBU)

Prof. Dr. Ole Jørgen Hanssen
Department of Ecology and Natural Resource Management (INA)
Norwegian University of Life Sciences (NMBU)
and
Ostfold Research

PhD Evaluation Committee

Prof. Dr. Margareta Linnéa Wihersaari
Energy Technology, Åbo Akademi University
P.O. Box 311, FIN-65101 Vasa, Finland

Prof. Dr. Bart Muys
Department of Earth and Environmental Sciences, University of Leuven (KU Leuven)
Celestijnanlaan 200E, BE-4001 Leuven, Belgium

Prof. Dr. Hans Fredrik Hoen
Department of Ecology and Natural Resource Management, Norwegian University of Life Sciences
P.O. Box 5003, N-1432 Ås, Norway

Preface

This thesis is a partial fulfillment of the requirements for the PhD degree at the Department of Ecology and Natural Resource Management (INA) at the Norwegian University of Life Sciences (NMBU). Together with a program of formal courses and a dissertation, it completes the requirements for the degree of Philosophiae Doctor. The project was funded by the Norwegian Research Council through the strategic university program: “The future role of biomass energy in Norway – an interdisciplinary technological, economic and environmental research program".

The advisory group has consisted of main supervisor Prof. Birger Solberg (INA) and Prof. Ole Jørgen Hanssen (Ostfold Research and INA). I want to thank the supervisors for all the useful academic guidance. You have provided me insights, knowledge and inspiration that I highly appreciate!

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During my stay at NTNU my parents housed me and my family. Thank you for that, and for always being supportive! I also extend gratitude to my siblings, Marte, Magnus and Vegard, for your encouragement throughout this work.

A special appreciation to Terje for letting me embark upon this work and, not least, for following me the whole way. Thank you for all your efforts and pep talks!

Dear Sara, Johan and all my nieces and nephews! This work is dedicated to you with the hope that it can be a contribution (however small) to make the world a better place for your generation and those that follow.

Ellen Soldal

Moss, May 2015

Contents

Summary

There is a growing interest in bioenergy, both nationally and internationally, due to the increasing emissions of greenhouse gases. The Norwegian forest growing stock is increasing and can be used to produce a range of products which can replace fossil resources. Carbon dioxide (CO_2) is the most important of the anthropogenic greenhouse gases and the forest plays an important role in the carbon cycle; potentially acting as both source and sink of CO_2. Reflection of incoming solar radiation (albedo) is, together with carbon sequestration, one of the most important climate mitigation factor in boreal forest that can be influenced by forest management.

This study explores environmental and economic consequences of bioenergy and other wood-based products from Norwegian forest resources. Life cycle assessment (LCA) was applied in order to map the environmental impacts of wood use. Traditional LCA lacks time and space considerations, and these are important, in particular when assessing potential environmental effects of harvest and use of boreal forest. LCA was combined with a bio-economic forest management model (GAYA-J/LP) in an attempt to overcome these shortcomings and obtain a link to economic aspects of forest managed for climate change mitigation.

The results shows that use of forest resources can provide environmental benefits when replacing fossil and/or carbon intensive products. The forest products provide reduced emissions of greenhouse gases compared with other products filling the same functions, depending on the climate neutrality assumption of biomass and how it is used. With regard to other impact categories, like ozone depletion potential, acidification potential and eutrophication potential, the results are mixed. Important factors in analysis of climate change mitigation contribution identified are the climate neutrality assumption of bioenergy, the climate effect of changing albedo, substitution and sequestration and emissions of biogenic CO_2. In a forest case-study of a tax/subsidy system where the forest owner was credited for positive climate mitigation contribution, it was found that the harvest profile over time was influenced by albedo, substitution and carbon price assumptions, as well as the choice of discounting the climate contribution.

Conservation of biological diversity was included through restrictions on the forest management and harvest in the forest model. If the carbon flux in the forest was assumed to be neutral, a negative relationship between forest climate mitigation and conservation of biological diversity was identified, as the wood-products provide potential savings of GHG emissions compared to alternative products. However, when the climate mechanisms related to the forest were included, the relationship between biodiversity and climate change mitigation was both positive and negative, depending on assumptions on substitution and albedo.

Proposals for important future research are presented in Chapter 6.

The combination of the forest bio-economic model and LCA was found to be a valuable tool for assessment of environmental impacts of harvesting boreal forest. The main benefits of this method are inclusion of economic aspects and the possibilities for local adaption of the forest management. When the forest model and LCA are combined it can provide policy makers with site specific data that can contribute to a climate policy that is founded on important local factors that influence the mitigation potential of the forest.

Sammendrag

Interessen for bioenergi er økende, både i Norge og internasjonalt. Hovedgrunnen for interessen er de økende utslippene av klimagasser og frykten for klimaendringer. I Norge øker stående biomasse, og skogbiomassen kan brukes til å produsere en rekke produkter som kan erstatte fossile råstoff.

Karbondioksid (CO_2) er den viktigste menneskeskapte klimagassen og skogen spiller en viktig rolle i karbonets kretsløp. Skogen kan være en kilde til CO_2 og den kan absorbere CO_2 gjennom fotosyntesen. Refleksjon av solinnstråling (albedo) er, ved siden av CO_2, en av de viktigste klimadriverne knyttet til boreal skog som kan påvirkes av skogbehandling. Forvaltning av skogen påvirker økosystemtjenestene som skogen tilbyr. I tillegg til tømmerproduksjon, karbonopptak og –lagring, er det mange økosystemtjenester knyttet til boreal skog, som for eksempel biologisk mangfold og rekreasjon, som må tas hensyn til ved vurderinger av miljøkonsekvenser av bruk av skogbiomasse.

I denne avhandlingen analyseres miljømessige og økonomiske konsekvenser av bioenergi og andre produkter basert på norsk trevirke. Livsløpsanalyser (LCA) er benyttet for å vurdere de miljømessige effektene av bruk av trevirke. Tradisjonell LCA inkludere ikke forhold knyttet til tid og sted, men disse er viktige, spesielt når miljøkonsekvensene av høsting og bruk av trevirke skal vurderes. For å inkludere aspektene knyttet til tid og sted samt økonomiske virkninger, ble LCA kombinert med en bio-økonomisk skogmodell.

Alle treproduktene som er analysert i dette arbeidet har lavere utslipp av drivhusgasser enn sammenlignbare produkter som fyller samme funksjon, avhengig av om man regner bruk av biomasse som klimanøytralt. For andre miljøkategorier – som for eksempel forsuring, ozonnedbryting og eutrofiering, er resultatene blandet og mindre entydige. Ved analyser av klimafotavtrykk til bioenergi, er følgende faktorer funnet å være viktige: forutsetning om klimanøytralitet, klimaeffekten av endret albedo, substitusjon, og opptak og utslipp av biogent CO_2. I et skatte-/avgiftssystem hvor skogeieren ble belønnet for positive bidrag til å redusere klimaendringene, ble avvirkningen tydelig påvirket av antagelser om albedo, substitusjon og diskonteringsrente.

Bevaring av biologisk mangfold ble hensyntatt gjennom restriksjoner på skogbehandlingen, inkludert avvirkning, i skogmodellen. Dersom biomasse ble antatt å være klimanøytral, ble

det nødvendig å foreta en avveining mellom ulike miljøhensyn, med biologisk mangfold og rekreasjon på den ene siden og utslippsrelaterte miljøgevinster på den andre siden. Men da klimadriverne i skogen ble inkludert, fant vi både positivt og negativt forhold mellom bevaring av biologisk mangfold og klimabidrag, avhengig av antagelser om albedo og substitusjon av fossil produkter.

Framtidige sentrale forskningsoppgaver er identifisert i kapittel 6.

Kombinasjonen av skogmodellen og livsløpsanalyser kan være et nyttig verktøy ved vurderinger av miljøkonsekvensene av avvirkning og bruk av norsk trevirke. Hovedfordelene ved å kombinere bio-økonomisk modellering og livsløpsanalyser er at økonomiske aspekter blir inkludert i analysen og at analysene kan tilpasses lokal skogforvaltning. Denne metoden kan gi politikere og lokale forvaltere stedspesifikk data med informasjon om lokale faktorer som er viktige for klimatiltak i kommuner og regioner.

List of papers

This PhD thesis is based on the following papers referred to by their roman numerals (I-IV):

Paper I

Soldal, E. and Solberg B. 2014. Environmental impacts and costs of using wood from boreal forest for climate change mitigation: a review of recent studies in Scandinavia. Submitted February 2015.

Paper II

Soldal E. 2014. Life cycle assessment of bioethanol used for heavy-duty transport in Norway. Submitted September 2014, resubmitted after revision March 2015 and May 2015 (*Journal of Cleaner Production).*

Paper III

Soldal, E., Valente, C., Bergseng, E., Modahl, I. and Hanssen, O. J. 2014. Combining forest modeling and LCA: a case study of biodiversity and life cycle emissions for forest products. Submitted September 2014.

Paper IV

Soldal, E., Bergseng, E., Rørstad, P. K. and Solberg, B. 2015. Including forest carbon, albedo and product substitution in harvest decisions – a case study for a forest management unit in Norway. Submitted May 2015.

1 Introduction

Humans have always depended on the forest. The forest provided the early human settlements with heat, food, fodder and materials for housing, tools and weapons. As the human population has grown and our technology has developed, the ability to exploit and influence the forests and other ecosystems have strongly increased. According to Rockstrom et al. (2009) human activities are now the main driver of global environmental changes, climate change being one of the most important environmental challenges of today.

There are several observations that indicate that the Earth's climate is changing: increased average surface temperature, increased sea level and decreasing snow and ice cover (Cubasch et al., 2013). The International Panel on Climate Change (IPCC) was established in 1988 in order to provide knowledge about "human-induced climate change, its potential impacts and options for adaption and mitigation" (UNEP & WMO, 2013). The first assessment report was published in 1990 and it placed global climate change on the agenda. Since then the evidence of a changing climate has become strengthened and in the fifth assessment report, the IPCC states that "It is extremely likely that human influence has been the dominant cause of the observed warming since the mid-20th century" (IPCC, 2013, p. 17). There is high confidence that the observed climate changes affect both physical and biological systems (IPCC, 2013). The human influence which the IPCC points to as the dominant cause of climate change, is emissions of greenhouse gases (GHG). Despite the understanding of the relationship between emissions of GHG and climate change, and international agreements on reduction of these, the global emissions of GHG are increasing (Hartmann et al., 2013).

GHG capture radiative heat that is reflected from the Earth's surface (Forster et al., 2007, Le Treut et al., 2007). Life on Earth depends on the natural greenhouse effect, but the increased emissions of GHG after the industrial revolution have created an imbalance in the concentration of GHG in the atmosphere causing increased heat absorption (Hartmann et al., 2013).

Not only are the emissions of GHG increasing, the growth rate of emissions is also increasing. The United Nations (UN) has defined a 2°C target, which aims at keeping the global average temperature increase below 2°C. A substantial cut in global GHG emissions is called for in order to increase the likelihood of reaching this target. In terms of radiative forcing and anthropogenic emissions, carbon dioxide (CO_2) is the most important GHG. In 2011 the atmospheric concentration of CO_2 was 390.5 ppm (Hartmann et al., 2013).

The energy sector is the largest contributor to GHG emissions (Anderson et al., 2008). The increased concentration of CO_2 in the atmosphere can be directly linked to combustion of fossil fuel through analyses of isotopes, and burning of fossil fuel is found to be the most important contributor to human induced climate change (Forster et al., 2007, Le Treut et al., 2007, Blanco et al., 2014). Thus, a considerable change in the energy sector is necessary (Brandão et al., 2013). The key drivers of global CO_2 emissions are (Anderson et al., 2008):

- Carbon intensity (carbon released per unit of energy used): $\frac{CO_2}{Energy}$
- Energy intensity (amount of energy used in the production of goods and services): $\frac{Energy}{GDP}$
- Activity level per capita:: $\frac{GDP}{Capita}$

Reducing the activity level is controversial as governments want economic growth and reducing population growth is a sensitive subject. Reductions of emissions can then be obtained by increased energy efficiency and/or by increasing the share of renewable energy (Anderson et al., 2008, Brandão et al., 2013, Cubasch et al., 2013).

Bioenergy is globally the most used renewable energy, and IPCC has pointed to bioenergy as an important part of the mitigation strategy. There is political interest in bioenergy globally. Norway and many other countries have pronounced goals of increasing the share of bioenergy, together with other renewable energy sources (The European Parliament, 2009, Norwegian Ministry of Foreing Affairs, 2011, Norwegian Ministry of Environment, 2012, European Commission, 2014). IPCC has developed several scenarios to describe potential ways to decrease the dependency on fossil fuel. Bioenergy plays an important role in all these scenarios, and they predict that the bioenergy use will shift from the traditional use in small stoves to modern use for transportation, heat, and combined heat and power (Smith et al., 2014).

At the same as the Norwegian government has a stated goal of increasing the share of bioenergy, Norway is obligated to reduce the emissions of GHG through the Kyoto protocol.

Norway has limited supply of agricultural waste for bioenergy use and the main source for bioenergy in Norway is wood from forest. Because of limited possibilities of increasing bioenergy production from forest industry residues and waste wood, an increased bioenergy use will most likely have to be based on primary forest production (Bergseng et al., 2013). The annual harvest of forest in Norway has for a long time been about 10 mill m^3, which is less than half of the annual increment (Trømborg et al., 2011), so in that perspective forest biomass has the potential to contribute to increased use of bioenergy.

This synthesis aims at summarizing the background for the research questions asked and the obtained results in the four research papers. The four research papers constitutes the main parts of the thesis. The synthesis is structured as follows: In Chapter 2 the objectives of the study and the main research questions are presented, followed in Chapter 3 by a review of background literature and state-of-the art in relevant fields. In Chapter 4, theoretical basis, methods and data for the work are described. The main results are presented in Chapter 5. Finally, in Chapter 6 overall discussion, conclusions and future research tasks are presented. The four papers are included as Appendices I-IV.

2 Objectives and research questions

The overall objective of this thesis is to investigate environmental and economic impacts of using Norwegian forest resources for bioenergy. This is done by combining a bio-economic forest model with life cycle assessment (LCA). The main emphasis has been on integrating these two approaches in order to develop a tool for balancing and evaluating different forest management objectives, with particular reference to economic results, biodiversity and climate change contributions. Incorporating impacts of future climate change and changing atmospheric concentrations of CO_2 was outside the scope of this thesis.

The thesis focuses on boreal forest and forest products, and the literature review emphasizes Scandinavia literature because of similarities in tree species, growth conditions, silvicultural and forest management. Harvested forest biomass is of varying quality, dimensions and species, and several wood-based products compete over the same resources. In the European biomass market, bioenergy is still a co-product or by-product with low economic value and does not act as a driver for harvest (European Commission, 2014).

Consequently, analyses of the potential development of bioenergy based on forest resources must be seen in connection with other potential usages of the wood biomass.

Climate change and loss of biological diversity are among the most important environmental challenges related to forestry, and both are considered in the analysis. The environmental impacts of emissions from the forestry value chains are investigated by life cycle assessment. Because LCA normally does not include economic impacts nor the impacts of harvest on biological diversity, these two factors are included by combining LCA with a bio-economic forest management model.

The thesis is based on four papers. In the first paper, we have identified knowledge gaps in the existing scientific literature on environmental impacts, costs of providing wood products and abatement costs. As the work progressed, the main research questions which were explored in the remaining papers emerged to become as follows:

- What are the environmental effects of biomass used for a variety of wood-based products in Norway, and what are the trade-offs between ecosystem services and other environmental benefits provided by the wood products?
- What is the effect of including biogenic CO_2 and albedo on the estimated climate change mitigation potential of bioenergy based on Norwegian forest resources?
- How can forest management and biomass use be optimized for climate change mitigation?
- What are potential effects of biodiversity conservation on the climate change mitigation contribution from forestry?
- What are the costs trade-offs between biodiversity conservation and climate mitigation?

3 Background

Carbon cycle and forests

Carbon is the fundament for all living organisms on Earth (Lawrence, 2000) and CO_2 is the main atmospheric phase of carbon (Ciais et al., 2013). In 2010, 60 % of the anthropogenic GHG emissions were CO_2 (Cubasch et al., 2013).

Carbon is stored in several reservoirs, and human activity and natural processes lead to transport of carbon between these reservoirs (Ciais et al., 2013). The natural flux of carbon between the lithosphere, biosphere, soil, ocean and atmosphere is referred to as the carbon cycle (Figure 1). The carbon cycle can be divided into two parts, characterized by different turnover rates. The continuous natural flux of carbon between the atmosphere, the ocean and the biosphere through photosynthesis, respiration, decomposition and ocean surface exchange constitutes the part with a relatively fast turn-over (from a few years to a few thousand years). The part of the carbon cycle that includes the carbon stored in the lithosphere has slow reservoir turnover (>10,000 years). The natural flux between and within the fast and slow domains are more or less in balance, while the anthropogenic emissions of CO_2 adds to the flux, creating an imbalance (Denman et al., 2007).

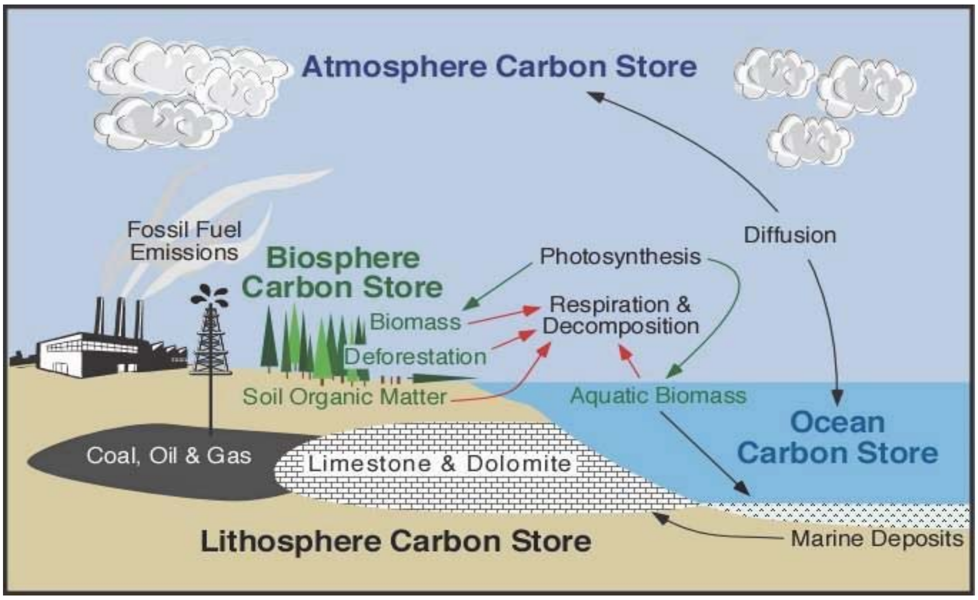

Figure 1: The carbon cycle with the main storage compartments (Pidwirny, 2006)

Growing biomass is an important part of the carbon cycle. Plants and trees sequester CO_2 from the atmosphere and store it as carbon in biomass. The carbon is oxidized and released to the atmosphere again as CO_2 when the biomass is decomposed or combusted. The CO_2 emitted through combustion or decay of biomass is referred to as biogenic CO_2. Because of the carbon sequestration and storage ability of biomass, biogenic CO_2 emitted through the combustion of biomass is often excluded from environmental assessments (The European Parliament, 2009, Cherubini et al., 2011b, Bowyer et al., 2012). Thus, the time span between emissions and sequestration is ignored (Cherubini et al., 2011b, Holtsmark, 2012, Matthews et al., 2014). There are several studies that focus on the effect of the timing of carbon flux between the atmosphere, biosphere and technosphere (see for example Cherubini et al., 2011a, Cherubini et al., 2012a, Cherubini et al., 2012b, 2013, Guest et al., 2013a, Guest et al., 2013b).

Cherubini et al. (2014) compare CO_2 from bioenergy with short-lived GHG (SLGHG), finding that the temperature response of biogenic CO_2 is constrained by the maximum emissions rates while the temperature response of long-lived GHG (LLGHG), like fossil CO_2, is proportional to the cumulative emissions. Some argue that a reduction in the SLGHG can mitigate a temperature increase in the short term, and that should be the chosen strategy in order to prevent the climate system to reach a tipping point (Bowerman et al., 2013). Others stress that the most important mean to reduce the climate change is to reduce the emissions of LLGHG, especially fossil CO_2 (Bowerman et al., 2013, Shoemaker et al., 2013). The temperature response from a pulse emission of biogenic CO_2 is characterized by an initial warming followed by a cooling effect and, in the long-term, the temperature response converge towards zero, while the temperature response of a corresponding quantity of fossil CO_2 will be sustained for centuries (Cherubini et al., 2014).

The forests play a key role in the carbon cycle. There are five primary carbon pools in the forest: aboveground biomass, belowground biomass, dead wood, litter and soil. Local conditions like type of forest ecosystem, site class, age of forest and forest management, including length of rotation, are factors that influence the flux of carbon between these pools and the atmosphere (Newell and Vos, 2012). In the boreal forest most of the carbon is stored in the soil (Newell and Vos, 2012). There are uncertainties in how large the carbon soil pool is and how it is influenced by harvest. de Wit et al. (2006) estimated a forest carbon budget for the productive forests in southeast Norway from 1971 to 2000 and found that the soil carbon increased by 4.5 % while the increase in carbon storage in biomass was almost 30 %.

The forest can contribute to mitigation of climate change in several ways. Most important, the forest sequesters large amounts of CO_2 through the photosynthesis, and until the tree is harvested or dies, this carbon is stored in the woody biomass and litter. Secondly, trees can replace fossil fuels and other energy- and/or carbon-intensive products, and thereby reducing production related emissions of GHG. When wood is being used for non-energy products, the carbon is stored in the technosphere until the product is discarded or combusted (Hoen and Solberg, 1994, Guest et al., 2013a, Smith et al., 2014). Bioenergy usually have higher emissions of CO_2 per energy unit than fossil fuels, but as mentioned, the biogenic CO_2 has been considered to be climate neutral (Schlamadinger et al., 1997, Cherubini et al., 2011a). Regardless of the assumption about the neutrality principle, the conversion efficiency (energy output per energy input) is important for the final results (Schlamadinger et al., 1997). Substitution of fossil-based products is a continuous option, while storage in soil, biomass and technosphere will, at some point, reach equilibrium (Schlamadinger et al., 1997, Gustavsson et al., 2010, Smith et al., 2014).

Forest biomass is a versatile raw material that has several energy applications, like heat, electricity and transportation fuel. In addition it can be used to produce construction material, paper and packaging, fibers for textiles and chemicals (Hoen and Solberg, 1994, Cherubini, 2010, Eriksson et al., 2012, Rødsrud et al., 2012). Matthews et al. (2014) expect that wood for materials, fibers and chemicals will increase in importance through the development of the new bio-economy and decouple the economy from fossil fuel.

Forest biomass is a renewable material, but also a limited resource. With many potential uses of woody biomass, the development of bioenergy based on forest resources increase the competition for fibers. Even though the demand for energy wood is expected to increase, it will not likely become the main driver of forest management in the future. Wood suitable for high-value applications like construction material will hardly be used for bioenergy initially (Trømborg and Solberg, 2010, Matthews et al., 2014). The different applications of wood creates different contributions to climate change mitigation, and a too narrow focus on bioenergy as a mitigation strategy can lead to a non-optimal use of the forest biomass (Moiseyev et al., 2014). When the environmental impacts of bioenergy are assessed, they need to be compared with the alternative fossil products that can be replaced as well as with other potential uses of the same biomass (Matthews et al., 2014).

In addition to the gas flux resulting from biomass use, changes in vegetation can also induce other impacts on local climate, like alteration of the hydrological cycle, shelter and changes in reflection of solar radiation, i.e. albedo (Solomon et al., 2007, Bright et al., 2011, Delucchi, 2011). Together with the flux of GHG, albedo is the most important human induced climate change mechanism (Delucchi, 2011). Especially in boreal forest with annual snow cover, the albedo effect can have a significant role because a snow-covered clearcutting reflects more of the incoming solar radiations than a dark forest cover (Betts, 2000, Bonan, 2008, Sjølie et al., 2013a). Bright et al. (2011) investigated how the albedo effect contributed to climate change resulting from harvest in Norway, and found that during the first decades the change in albedo offset the negative climatic effect of combustion of biofuel. Cherubini et al. (2012a) analyzed the effect of changes in albedo after harvest, and they found that the cooling effect in Norwegian forest was almost as large as the warming effect due to biogenic CO_2.

Cherubini et al. (2011a) argued that even though the same amount of CO_2 is being emitted and sequestered when using bioenergy, the biogenic CO_2 will stay in the atmosphere for a significant time, contributing to climate change. They launched the GWP_{bio} index in an attempt to capture the global warming potential (GWP) of biogenic CO_2 based on the atmospheric decay of CO_2 and re-growth. This characterization factor for biogenic CO_2 was further developed by including the impact of changes in albedo following a harvest (Bright et al., 2012, Cherubini et al., 2012a).

McKechnie et al. (2010) integrated forest carbon models and LCA in order to assess the total emissions of GHG including the carbon flux in the forest. They found that the impacts of forest dynamics are significant, and that the local factors that influence forest carbon dynamics should be included. Their study also reports an initial increase in GHG emissions from the bioenergy systems compared to the fossil energy pathways, but this increase is temporary and after some decades, the bioenergy pathways gives reduced net emissions of GHG.

Repo et al. (2012) assess the climate impact of using harvest residues for bioenergy. They include both a carbon budget for the forest and emissions associated with the production of bioenergy. Further, they compare the radiative forcing (RF, in Watt/m^2) as a result of these emissions, with the RF due to fossil fuels. All their bioenergy scenarios give smaller cumulative RF compared to fossil fuels.

Kilpeläinen et al. (2011) developed a methodology for calculation of CO_2 emissions and sequestrations in forest based on an ecosystem model, and included emissions from forest management operations and combustion of bioenergy. Under assumptions about a stable climate, they found that the net emissions of CO_2 for traditional timber scenario was -319 g CO_2/m^2/year. If timber production was integrated with bioenergy production (thinning and logging residue) net CO_2 emissions were -110 g CO_2/m^2/year. They did not include avoided emissions through substitution, so the main difference between these two results is the consumption of wood for energy (220 g CO_2/m^2/year in the latter scenario compared to zero g CO_2/m^2/year in the first) (Kilpeläinen et al., 2011). This methodology was used by Kilpeläinen et al. (2012) to find the net flux of CO_2 and the consequent radiative forcing impacts of bioenergy production and utilization under Finnish boreal conditions. Over a time frame of 90 years, the forest acted as both sink and source of carbon, but the cumulative radiative forcing was 19 % lower for bioenergy than for coal (Kilpeläinen et al., 2012). Albedo was not included.

There are several other examples of inclusion of biogenic CO_2 in climate change mitigation analysis of bioenergy and Matthews et al. (2014) have published a substantial literature review on this topic.

Other forest ecosystem services

In addition to the potential contribution to climate change mitigation, the forest provides many other services that are important. The term *ecosystem services* is used as a collected term for all provisioning, regulating, cultural and supporting services that nature provides for humans (Reid et al., 2005). Ecosystem services provided by boreal forests includes provisioning of timber, game, bioenergy and fibers for cellulose and bio-chemicals; regulating services which include flood control and erosion protection; supporting services which include biological diversity, sustainment of biochemical cycles, primary production and resilience to change; cultural services which include recreation, health and ethical values of biological diversity conservation (Lindhjem and Magnussen, 2012).

In addition to climate change, Rockstrom et al. (2009) name loss of biodiversity as one of the environmental problems of today that has crossed the boundaries for a safe operating stage. The term biodiversity includes variability within and between species and variability of ecosystems (Secretariat of the Convention on Biological Diversity, 2005). Through the Convention on Biological Diversity (CBD, ratified in 1992) and the Bern convention (ratified in 1986) Norway has obligations to conserve biological diversity and secure sustainable use of biological resources (Norwegian Ministry of Environment, 2001). Loss, degeneration and fragmenting of habitat and excessive harvest have been named two of the most important human influences on biodiversity (Lier-Hansen et al., 2013). In Norway, about 60 % of all terrestrial species are living in or in proximity of forests (Direktoratet for naturforvaltning, 2010). Of the Red listed species in Norway, 40 % is affiliated with forest biotopes. Old forests have a higher density of endangered species than young forests, but biodiversity also depends on a dynamic diversity of stand structures (Artsdatabanken, 2010, Oliver et al., 2013). In addition to support a variety of species, a diversity of stand structures can also increase the forests' resilience to catastrophic events (Oliver et al., 2013).

Land use and land use change (LULUC) have been included in LCA in different ways (Milà i Canals et al., 2007). Some studies include LULUC simply as land occupation (m^2/year), while others have attempted to qualitatively evaluate LULUC by classifying land areas (Antón et al., 2007). Suggested indicators for impact on biodiversity includes potential disappeared fraction of species, percent of threatened vascular plant species in region and red listed species (Milà i Canals et al., 2007). Milà i Canals et al. (2007) suggests a framework for inclusion of land use (occupation) and land use change (transformation) in LCA by linking LULUC impacts to biodiversity, biotic production potential and ecological soil quality, while Michelsen (2008) proposed a methodology to include the biodiversity aspects in accordance with this framework.

Oliver et al. (2013) examined the CO_2 and fossil fuels savings together with biodiversity protection through harvest and non-harvest scenarios. They found that the greatest CO_2 savings was achieved through substitution of concrete and steel. Wood for energy offers smaller savings, and according to them only residual wood should be used for energy (Oliver et al., 2013). Protecting biodiversity and maximizing forest CO_2 sequestration may not be compatible because biodiversity depends on a variety of different forest landscapes (open landscape, dense forest, understory forest and complex forest), while the highest amount of CO_2 savings are accomplished by keeping all forest as understory and complex forest structures (Oliver et al., 2013).

The role of forest management

How the forest is managed is essential to the services provided by the forest, and in Norway management of public and private forests are regulated by the Forest Act (Norwegian Ministry of Agriculture and Foodt, 2005). Through the Forest Act, the government wants to ensure that the forest owners take sufficient considerations to biological diversity, landscape, recreation and cultural heritage when managing the forest (Hoen and Svendsrud, 2014).

Forest management is an important tool in order to preserve the different qualities of the forest, and forest management in Norway is generally characterized by multipurpose management, i.e. a management regime designed to provide a range of products and services (Matthews et al., 2014). Maintenance of the different ecosystem services from the forests require a variety of management strategies and in many cases there will be a trade-off between at least some of the various ecosystem services (Lindhjem and Magnussen, 2012). Amongst mitigation options for forest that are mentioned by Smith et al. (2014), forest management is one of the most important for Norwegian forests. Lundmark et al. (2014) claim that the forest growth in Sweden can be increased by more than 50 % by changes in forest management. In addition to increased forest yield, forests can be managed in a way that can increase the sequestration of and the storage of carbon. Examples of possible management options to increase carbon sequestration and storage are higher regeneration densities, reduced thinning, forest fertilization, prolonged rotations, improved tree provenances, and choice of species combinations. Such management changes can give reduced provision of timber and wood fibers, and/or lower recreational value of forest. The trade-off between maximized biomass harvest and maximized biomass storage is an important consideration when assessing forest management mitigation strategies (Hoen and Solberg, 1994, Schlamadinger et al., 1997, Lundmark et al., 2014). Use of timber is important in that trade-off situation. Hoen and Solberg (1994) was a first attempt to combine those factors for boreal forest in a consistent bio-economic optimization framework in Norway.

There are also trade-offs between different climate change mechanisms. Recent research suggest that forest management strategies for climate change mitigation should focus on more than just GHG reduction, and that the albedo effect can be among the most important factors to consider in forest management (Betts, 2000, Gibbard et al., 2005, Bala et al., 2007, Betts et al., 2007, Bonan, 2008, Thompson et al., 2009, Schwaiger and Bird, 2010, Arora and Montenegro, 2011, Sjølie et al., 2013a). From a climate change mitigation perspective, including the albedo effect may imply shorter rotations, more mixed or broadleaved forests and less afforestation than what is optimal when only considering carbon sequestration (Bright et al., 2014, Sjølie et al., 2014).

Ter-Mikaelian et al. (2013) simulated future harvest scenarios in order to assess the change in carbon stock as a result of changing harvest levels in Ontario. In their analysis, they find that the projected carbon stock (in forest and harvested products) converge to within 2 % difference by 2100 for all scenarios, and they conclude that the sustainable harvest of boreal forest will have a small effect on the combined forest and wood products long-term carbon stock (Ter-Mikaelian et al., 2013). They do not include avoided emissions through substitution or effect of changed albedo after harvest.

Pingoud et al. (2010) integrated forest management and wood product substitution in a climate change perspective, and found that the largest stock of carbon in forest and products was achieved by increased rotation length and basal area. Use of saw logs for long-lived products instead of more energy-intense products, followed by cascading the material as bioenergy provided the largest climate benefit (Pingoud et al., 2010). Cascading of wood is in line with the industrial ecology concept, and the environmental and material benefits of cascading are confirmed by others (e.g. Dornburg and Faaij, 2005, Gustavsson et al., 2006, McKechnie et al., 2010).

Cost

The bio-economic forest model GAYA-J/LP has been used to analyze harvest and economic effects of biodiversity conservation (Hoen et al., 1998, Eid et al., 2002, Bergseng et al., 2011), as well as estimates of GHG balance and climate mitigation costs under different management regimes (Hoen, 1990, Hoen and Solberg, 1994, Solberg et al., 2008, Raymer et al., 2009). In Solberg et al. (2008) the model was used to quantify trade-offs between harvest income, climate mitigation and biodiversity protection at forest property level. Raymer et al. (2009) found that maximizing carbon benefits by forest management, reduced the net present value of the forest in Hedemark (a Norwegian region, 13 420 km^2) by 21 %. The corresponding carbon benefit, incl. substitution benefit, from the forests was 2.4 million ton CO_2-equivalents per year on average over 120 years.

Paper I gives an overview of cost studies in addition to environmental assessments. Several analyses indicate that bioenergy is not cost competitive with fossil energy (see for example

Hennig and Gawor, 2012, Bertrand et al., 2014, Gerssen-Gondelach et al., 2014) while others found that biomass based systems have lower costs than fossil reference (Kalt and Kranzl, 2011). Gerssen-Gondelach et al. (2014) found that bioenergy for heat and power generally has higher investment, operation and maintenance costs than fossil energy. However, they expected the price of fossil resources to increase in the future while the production cost of bioenergy is expected to decrease. Technology learning will increase the efficiency of bioenergy production and this will lead to decreased cost and less emissions per energy unit (Gerssen-Gondelach et al., 2014). According to the same study, biomaterials are already able to compete on price with other raw materials (Gerssen-Gondelach et al., 2014). Several studies found that bioenergy will contribute to reduced emissions of GHG, and a price on CO_2 can make bioenergy profitable (Bertrand et al., 2014).

The cost of bioenergy is important for the effect of policy instruments that aim at reduced GHG emissions or increased use of bioenergy. When EU and Norway want to increase the use of renewable energy, the reduction in GHG emissions is only one of several goals. In addition they want higher energy supply security and development of a competitive energy sector that provides employment (Bentsen et al., 2014). Policy instruments that are designed to reduce GHG emissions or to shift energy production from fossil fuels to renewables, can have different results and cost-effectiveness (Schmidt et al., 2011). Policy instruments designed to reduce the emissions of GHG, like CO_2 taxes and the EU emissions trading system (ETS), are expected to be the most cost-effective solutions because the market will allow an efficient allocation of reduction efforts among technologies. But the emissions reduction effect depends on available low carbon technologies (Schmidt et al., 2011). Direct promotion of selected energy technologies through feed-in tariffs, required share of biofuels and subsidies, may lead to development of only a part of the available technologies (Schmidt et al., 2011).

Location of biomass supply, plants for conversion and users of the energy are important factors for the cost of bioenergy, and this calls for spatial explicit modelling when analyzing the cost of bioenergy (Schmidt et al., 2011). According to Bergseng et al. (2013), the biomass supply costs in Norway are relatively stable for a large range of biomass demand; but when the demand approaches the supply limit, costs will increase rapidly. Again, bioenergy is rarely the driver of a forest biomass value chain, and like environmental analysis bioenergy concerns have to be a part of more comprehensive economic analyses (Sjølie and Solberg, 2011, Bergseng et al., 2013).

Biodiversity protection will in most cases reduce income from timber production for the forest owner. Bergseng et al. (2011) analyzed the relationship between biodiversity conservation and the cost associated with the increased conservation efforts. They found that increased restrictions on the forest management reduced the net present value of the forest by 10-45 % compared to a case with no restrictions on the forest treatment. Increased rotation length and minimum share of old-growth forest was considered to have high value for biodiversity, and were also the most costly forest management options. For the same forest area, Solberg et al. (2008) mapped the trade-offs between timber income, biodiversity protection, and carbon sequestration.

GHG abatement cost is the cost associated with reducing the emissions of GHG by shifting to an alternative system (Hennig and Gawor, 2012). It can be calculated as the fraction A/B were A is the additional cost associated with production of bioenergy instead of using fossil fuels and B is the corresponding reduction in GHG emissions because fossil fuels are replaced (Sterner and Fritsche, 2011). This is also referred to as cost-effectiveness. Many researchers have studied this, and results of selected studies done after 2003 are shown in Paper I.

4 Methodology

Theoretical basis

Systems theory is the theoretical basis for the analysis. General systems theory was already in the 1930s introduced as a concept by biologist von Bertalanffy as the scientific basis for a holistic exploration of systems (Von Bertalanffy, 1972). A system is a composition of elements and subsystems that is separated from the surroundings by fulfilling a common purpose, and a system has some characteristics that are more than just the sum of single elements (Von Bertalanffy, 1972, Dekkers, 2015). Systems theory is based on the realization that you need to understand the relationship between the system elements in order to regulate the system (Von Bertalanffy, 1972). In environmental analysis, the system theory is important because it describes the interaction between the system elements and between the system and its environment (Brattebø and Kjelstrup, 2011).

The inclusion of all relevant interactions are important also in economic analyses, but there human behavior, human welfare and the importance of human interactions and their influence on the environment are in focus. Environmental economics has during the last half century developed to include also environmental issues in economic theory (see for example Conrad (2010) for an overview). However, for simplicity, I have in this thesis chosen systems theory as theoretical basis.

During the 1970's and 80's, society gained knowledge of how large influence human activity has on the environment, and the field of environmental analysis shifted towards a systems-thinking approach (Brattebø et al., 2011). In order to avoid problem-shifting and non-optimal solutions, it was called for a more holistic problem solving. The scientific community saw the need for a systematic tool that integrated the material and energy flow of production systems with the outside world (Brattebø et al., 2011).

A number of analytical tools have been developed in order to systematically examine the energy and material flows and the impact on the environment. These methods include material flow analysis (MFA), energy and exergy analysis, environmental risk assessment (ERA), life cycle assessment (LCA), input-output analysis (IOA), cost benefit analysis (CBA), life cycle costing (LCC) and combinations of methods like IOA-LCA and LCA-MFA (Baumann and Tillman, 2004, Finnveden and Moberg, 2005, Heijungs et al., 2011).

Methods

This chapter describes the two main methods used in the thesis. Each of Paper I-IV have a methodological section, but the description there is by necessity rather short, and therefore enlarged in the following sections.

Life Cycle Assessment (LCA) is a well-established method for assessing the environmental impacts of production, consumption and disposal of goods and services and is used in three of the papers in this thesis.

Traditional LCA does not include analysis of effects of land use and land use change (LULUC), i.e. which effect does the harvest of forest biomass have on biological diversity, carbon storage and other ecosystem services. In order to overcome the weakness of LCA in relation to non-emission impacts and to have a strong link to economic impacts, an integration of LCA and a bio-economic forest model is applied in this thesis. GAYA –J/LP is a forest economic optimization model that can find the optimal treatment of forest stands, given pre-specified objective functions and forest management restrictions. Based on growth models, natural mortality models, economic objectives and pre-specified alternative forest management options, GAYA-J/LP simulates the development of the forest for a defined time horizon (Hoen, 1990, Hoen and Eid, 1990, Raymer et al., 2009). LCA is combined with GAYA-J/LP in two of the four thesis papers.

Life Cycle assessment (LCA)

LCA is an environmental systems analysis tool that is widely applied for investigations of potential impacts of products or services (Baumann and Tillman, 2004, Klöpffer and Grahl, 2014) that emerged in the late 1980's as a response to the increasing awareness of the human influence on the environment (Hanssen, 1999). LCA is a multi-disciplinary methodology that analyzes technical, natural and social systems, and the interface between these systems (Baumann and Tillman, 2004), as illustrated in Figure 2.

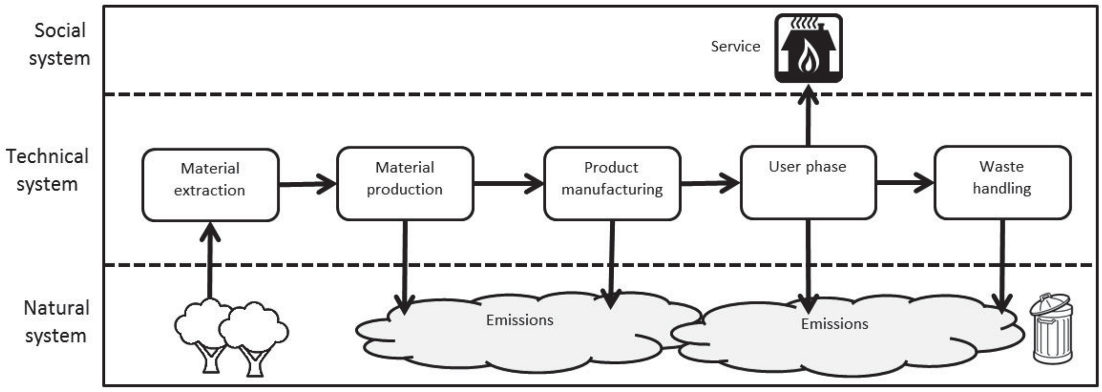

Figure 2: Illustration of analysis of environmental effects of bioenergy including interactions between natural systems, technical systems and social/economic systems.

LCA facilitates a quantitative analysis of potential environmental impacts across the life cycle of a product or service. A complete inventory of all material and energy requirements for production, use and disposal of the product of interest is gathered and the potential environmental impact resulting from the emissions are calculated. The potential impacts are related to a function that the technical system delivers to the social system based on resources from the natural system (ISO, 2006b). The method provides a tool for understanding the most important potential environmental impacts of the production system, and where in the production chain these impacts occur. It is used to compare product alternatives, and as a basis for strategic and political decision making (ISO, 2006b).

For example, in analyzing the environmental impacts associated with bioenergy, an essential question is whether biomass performs better or worse than alternative products that provide the same function and what is the best use of the available biomass. LCA facilitates the comparison between different raw materials and different uses of the raw material, using the same functional unit as point of reference. Matthews et al. (2014) found that LCA is a well suited tool for environmental analysis of wood products, and according to Agostini et al. (2013) there is a political and scientific agreement that LCA is a necessary methodology for these kinds of analysis.

An LCA study consists of four stages:

1. Goal and scope definition (including information on system boundaries, functional unit and allocation).
2. Inventory analysis (input and output of the product system).
3. Impact assessment (the results of the inventory are translated into contributions to relevant environmental impact categories).
4. Interpretation of the results.

The defined goal form the fundament for several important methodological choices. When the goal and scope of the analysis are defined, the modelling principle and decision context of the analysis are also defined. In LCA, there are two modelling principles: attributional and consequential (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2010b). I will get back to those later.

There are three archetypal decision contexts for LCA, illustrated in Figure 3. The production systems analyzed in the thesis are relatively small and will only have small-scale impacts in background systems or other systems of the economy. I have therefore used decision context A, which is referred to as “Micro-level decision support” and it is typically used for products or production systems with small-scale market consequences (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2010b).

Decision support?		Kind of process-changes in background system / other systems	
		None or small-scale	Large-scale
	Yes	Situation A "Micro-level decision support"	Situation B "Meso/macro-level decision support"
	No	Situation C "Accounting" (with C1: including interactions with other systems, C2: excluding interactions with other systems)	

Figure 3: The decision context of a LCA study depends on whether the study will be used to support decisions and, if so, will the decisions lead to small- or large-scale changes in background system or other systems Figure taken from European Commission-Joint Research Centre - Institute for Environment and Sustainability (2010b).

The system boundaries defines which processes that belong to the analyzed system (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2011). The system boundaries used for the analysis in the thesis are illustrated schematically in Figure 4, and described in detail in the individual papers. The top part of Figure 4 illustrates the system boundaries applied in Paper II, III and IV, while the bottom part (below the dashed line) illustrates how the carbon flux in the natural system is included in Paper II and IV.

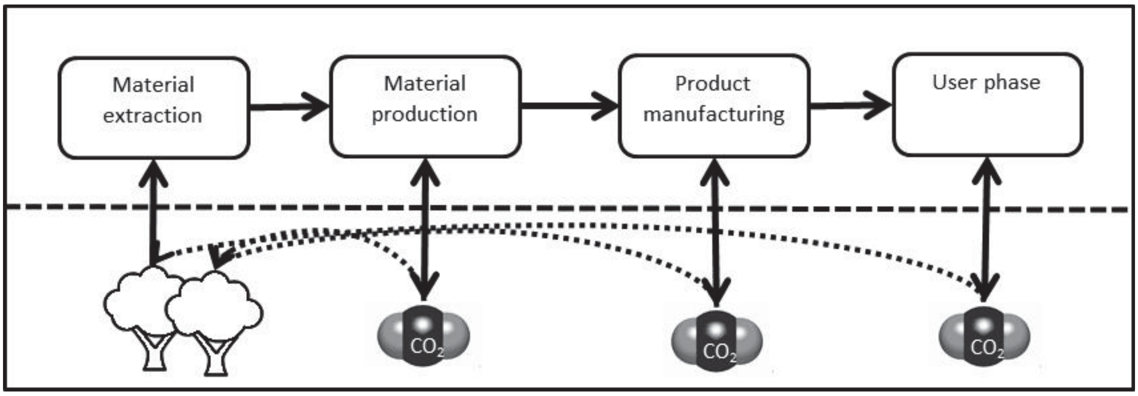

Figure 4: Illustration of system boundaries used in three of the four thesis papers. The flux of CO_2 in the forest is below the dashed line. The solid, double-sided arrows illustrate that the net flux of CO_2-equivalents between the forest value chain and the forest might positive or negative. The dotted arrows illustrate the carbon sequestration.

The environmental load associated with production and use of a product or service is distributed linearly to a unit of reference, a so-called functional unit (FU). The FU can be a unit of a product or even a service provided by the product (ISO, 2006b).Depending on the scope of the assessment, the FU can be related to input into the system or to output of the system. Potential environmental impact of wood products has two points of departure. The product can be compared to an alternative product that provides the same function or service (output), or it can be compared to another product that is based on the same resource (input) (Rivela et al., 2006). In this thesis, I have made use of both of those perspectives. In Paper II, I have compared bioethanol based on woody biomass with fossil diesel used for heavy-duty transport. In this context, it is interesting to analyze the potential environmental impacts of a service provided by the production system. Therefore, in this case, the FU was 1 km driven by a lorry. In Paper III, I have assessed the potential environmental impacts of the use of a forest property for different wood-based products including biodiversity conservation measures. Thus, in Paper III the FU is one km^2 of productive forest. A geographical functional unit was chosen to facilitate an up scaling of the analysis to a larger area.

When the production process that is being analyzed produce more than one product, the material and energy input and output as well as environmental burdens need to be allocated between different co-products (ISO, 2006b). There are many examples of such multifunctional processes in the forest value chain: residues from production of construction material are being used for production of energy, pulp and particle boards, biorefineries produce cellulose, bioethanol and biochemical at the same site with many processes in common (Cherubini et al., 2011c). The choice of allocation method has been shown to be important for the final result (Börjesson et al., 2010, Cherubini et al., 2011c, Kumar et al., 2012), and it is recommended to avoid allocation whenever possible (Baumann and Tillman, 2004). However, if allocation is necessary, there are two main methods: system expansion and partitioning. When using system expansion, the production process of interest is credited with avoided emissions from other production pathways of the co-products (i.e. emissions from production of co-products are subtracted from the total emissions of the multifunctional process) (Baumann and Tillman, 2004, Cherubini et al., 2011c) When using the partitioning method, the emissions from the multifunctional process are being divided amongst the products based on for example mass, energy content or economic value (Cherubini et al., 2011c). In this thesis I have applied mass (Papers II and III) and energy partitioning (Paper III).

The second step in LCA is life cycle inventory analysis (LCIA), i.e. the mapping of all material and energy flows required for production and emissions of substances connected with these flows. As mentioned, there are two main modelling principles when the life cycle data is collected: attributional and consequential (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2011). "Attributional" modelling is used when depicting an actual or forecasted specific or average value chain in a static technosphere, while "Consequential" is used to depict generic value-chains with expected changes in the foreground and background system (a dynamic technosphere) (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2010b). In this thesis, the investigated systems are existing systems, and therefore I have used attributional LCA in all papers. In Paper II specific data are collected for the foreground systems. In Paper III, there is a mix of generic and specific data that are described in detail in Paper III.

In the third step of an LCA, the life cycle impact assessment, the substance emissions that have been quantified in the inventory process are translated into environmental impact indicator results. During this stage the emissions are first assigned to the relevant impact categories according to the substances' ability to contribute in the specific impact categories (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2010a) The impacts of a product can be assessed on the midpoint level or end-point level (Figure 5). End-point categories are burdened with higher uncertainty than midpoint impact categories. The impact categories should reflect issues of direct environmental importance, and examples of midpoint impact categories include climate change, ozone depletion, eutrophication and eco toxicity. Examples of endpoint damages include damage to human health, damage to ecosystem diversity and resource availability (Goedkoop et al., 2009). In this thesis, I have not analyzed damages at end-point level.

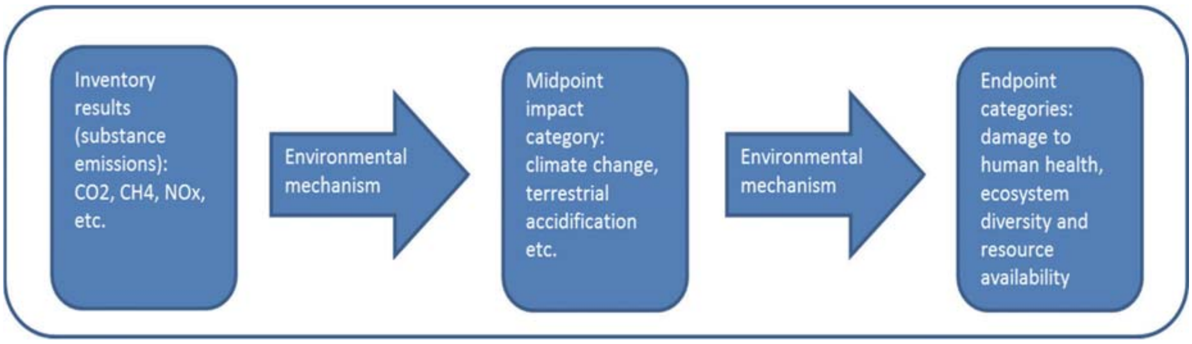

Figure 5: Emissions of substances have impacts in different defined impact categories, and the impact is modelled based on some environmental mechanisms. The further to the right we move in the process, the larger the uncertainty.

The next step of the impact assessment is characterization (European Commission-Joint Research Centre - Institute for Environment and Sustainability, 2010a). Emissions of different substances with the same type of environmental effect are converted into equivalents based on environmental mechanisms (Figure 5) (ISO, 2006a). There have been developed several methods for impact assessment. In Paper II, ReCiPe (Goedkoop et al., 2009) was used, and in Paper III CML-IA baseline v3.00 was used. In addition, Paper II contains a comparison of different characterization factors for biogenic CO_2 and the effect of change in solar radiation reflection by the Earth surface following a harvest (Table 1). For global warming potential (GWP) the unit is CO_2-equivalents, and IPCC usually give the GWP over three time horizons; 20, 100 and 500 years, and they are referred to as GWP_{20}, GWP_{100} and GWP_{500}, respectively. The characterization factor for fossil CO_2 is 1 (Solomon et al., 2007). The characterization factor for biogenic CO_2, GWP_{bio}, was launched by Cherubini et al. (2011a) in order to include the temporal increase in atmospheric concentration of CO_2 following combustion of bioenergy. In Bright et al. (2012), and Cherubini et al. (2012a) GWP_{bio} was further developed to include the effect of change in reflection of solar radiation (albedo) following harvest at Northern latitudes.

Table 1: Different characterization factors for biogenic CO_2 and albedo (CO_2-eq.) used in Paper II.

Method	GWP_{100} Biogenic CO_2	GWP_{100} Albedo	Net GWP_{100} (Biogenic CO_2+albedo)
GWP=1	1	-	1
GWP=0	0	-	0
GWP_{bio}	0.62	-0.42	0.2
GWP_{bio} incl. forest residue	0.51	-0.38	0.12

In Paper II and III, several impact categories are included: global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), photochemical oxidation formation potential (POFP) and ozone depletion potential (ODP). These impacts are the most common environmental impact categories assessed in the forest fuel supply chains (see e.g. Berg and Lindholm 2005 and Cherubini and Strømman 2011). In addition, particulate matter formation potential (PMFP) was included in Paper II, as this is important when assessing transportation fuel used in densely populated areas.

A limitation of LCA is that it is mainly constructed to assess the potential impacts of emissions and that it lacks the time dimension (ISO, 2006b, Michelsen, 2008). Production of a product or service can also have environmental effects that are not related to emissions, and that is particularly true for biomass based value chains that can have large impacts on land use and biodiversity (Milà i Canals et al., 2007, Michelsen, 2008). Still, a review of LCAs on bioenergy by Cherubini and Strømman (2011) revealed that only 9 % of the studies included land use impact in their assessment and none included assessment of the potential impacts on biodiversity.

GAYA-J/LP

In order to overcome some of the issues regarding space and time mentioned above and to include economic effects, LCA is here combined with a forest economic model, GAYA-J/LP, in two of the thesis papers (III and IV). GAYA-J/LP is a long-term bio-economic forest management optimization model that consists of two parts; a forest stand model (GAYA) and an optimization part (J/LP). It was developed and used for the first time for Norway in Hoen (1990), and later applied in several studies like Hoen and Eid (1990), Hoen and Solberg (1994), Eid et al. (2002), Raymer (2005), Raymer et al. (2005), Solberg et al. (2008), Bergseng et al. (2011). GAYA-J/LP combines biological and economical aspects of forest management in order to find optimal forest management solutions, assuming exogenously determined objective function, costs, prices, constraints and forest growth parameters.

The simulation part of the model (GAYA) simulates numerous possible development paths for the forest based on the initial state of the forest, obtained by forest inventory. The simulations are based on basal mean diameter, mean height weighted by basal area and number of trees, and growth is estimated on a 5 year basis (Hoen et al., 1998, Raymer et al., 2009). Simulation of the forest uses the functions for diameter increment, height and natural mortality as documented in Hoen et al. (1998). In addition to possible biological constraints regarding growth and mortality, the development of the forest stand is influenced by constraints that exclude forest management alternatives which are clearly unrealistic. Thus, GAYA simulates all realistic stand treatments and the corresponding developments of the forest stands, based on the ex-ante specified biological and management restrictions (Hoen et al., 1998, Raymer et al., 2009).

The output from GAYA is input to the linear programming, J/LP. This part of the model optimizes the management of the forest stands and selects the optimal set of treatment options among the numerous alternatives that is simulated by GAYA. The optimal solution is found, given the objective function. The objective function can be related to the economy for the forest owner (net present value), harvest volumes or qualities (share of sawn- and pulpwood, keeping harvest at certain levels), standing stock after harvest, or to specified forest management options with respect to regeneration, final felling and thinning (Hoen and Eid, 1990). In this thesis the objective function is maximization of the net present value (NPV).

In Paper III and IV, GAYA-J/LP is combined with LCA (Figure 6). The figure gives a schematic illustration of the integration between GAYA-J/LP and LCA and important building blocks in the different parts. GAYA-J/LP finds the optimal forest management, given the management restrictions and objective function. The resulting harvest is used in forestry value chains that are modelled in the LCA software, SimaPro (PRé Consultants 2013). In Paper III, GAYA-J/LP and LCA are combined in order to find environmental impacts of biomass use from a forest property in Østfold, Norway. In Paper IV, GAYA-J/LP and LCA are combined in order to evaluate potential climate change contribution of the same forest as in Paper III. The results from Paper III are used to calculate the net GHG emissions from production and substitution of wood-products (kg CO_2-eq./m^3) which are used as input to GAYA-J/LP in Paper IV (Figure 6). In addition, the carbon flux and albedo change in the forest following harvest are included in the analysis of the harvest and forest climate impacts.

GAYA J/LP use empirical data in order to predict future forest situations, while LCA use empirical data to describe the current situation. LCA is static in time while GAYA-J/LP models development over time. This combination provides some challenges when the two are integrated. In this thesis, this has been tackled in two different ways. In Paper III, the simulation output from GAYA-J/LP are summarized over two different time horizons and used as input to LCA (dry matter/km^2). In Paper IV, the output from LCA (kg CO_2-eq./m^3) are one of several inputs to GAYA-J/LP. In this case, we have tested the effect of discounting the climate effects alongside with timber profit. LCA is a useful tool for assessment of environmental impacts related to emissions while GAYA-J/LP provides a model for inclusion of environmental impacts related to land use. GAYA-J/LP keeps track of carbon while LCA supplement the analysis with other substances.

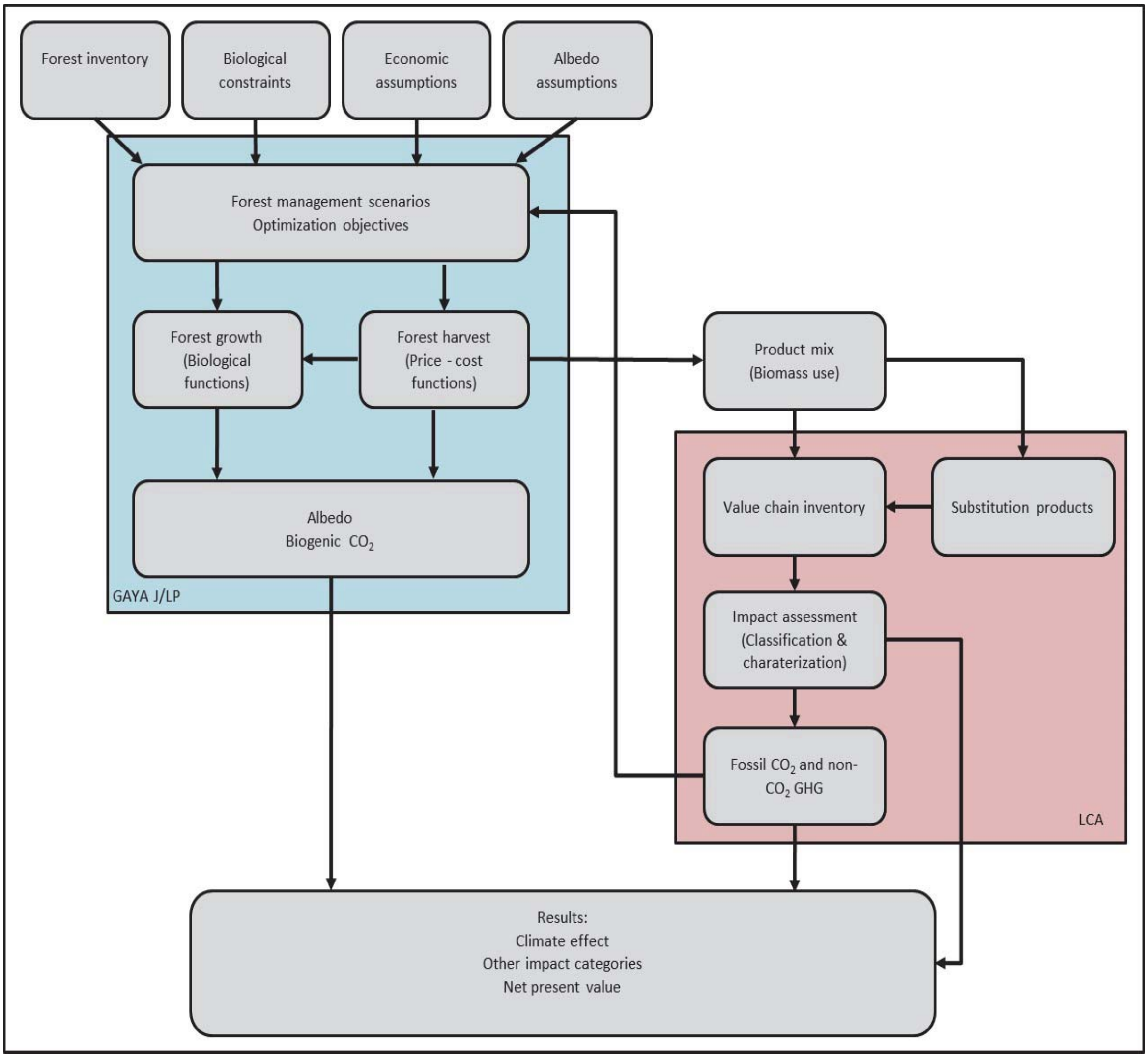

Figure 6 Structure of the integration of GAYA J/LP and LCA. The arrows indicate information flow.

In Paper III, three management scenarios are defined in GAYA-J/LP and the objective function is to maximize the net present value of the forest. Two of the forest management scenarios that form the basis for the analysis contain restrictions on the forest treatment in order to preserve biological diversity, cultural heritage sites and recreational values, while one scenario does not have any restrictions on the treatment. The latter is used as a reference.

The three scenarios in Paper III are:

1) A reference scenario (REF): a base scenario for comparison; without restrictions on forest management.
2) A scenario representing Program for the Endorsement of Forest Certification (PEFC), with constraints on management to preserve biodiversity. This scenario represents the current certification regime for sustainable forestry in Norway and the

constraints are an operationalization of the restrictions given by the certification organization. The Norwegian Forestry Law of 2005 legally authorizes the PEFC (LOV-2005-05-27-31).

3) A biodiversity scenario (BD), characterized by constraints on forest management with extensive care taken to preserve biological diversity and maintain the forests' recreational value. The constraints exceed the PEFC scenario with explicit local adaptations as defined by the local authorities as a basis for further multipurpose planning of the area, which enabled inclusion of specific considerations like important recreational areas. The municipality assigned areas to four different categories ranging from normal forestry with no restrictions (category 1) to full preservation of forest (i.e. no harvest at all, category 4). Category 2 and 3 are gradients between these two extremes.

Other methodological issues

In this thesis, case studies are used to assess the potential economic and environmental impacts of wood bioenergy. All the studies are limited to boreal coniferous forest dominated by Norway spruce (*Picea abies* (L.) Karst.) and Scots pine (*Pinus sylvestris* L.), with elements of deciduous species, like Birch (*Betula*) (FAO, 2001). In Paper II, production of bioethanol from lignocellulosic biomass used for heavy-duty transport is analyzed. In Paper III and IV, the LCA is combined with a forest model that works with specific spatial and time dimensions. For demonstrations on how the integration of these two can work, we chose a publicly owned forest in Fredrikstad municipality, southeast parts of Norway, as fundament for the analysis. According to Pulla et al. (2013), the "public forest […] play a key role in sustaining forest ecosystems, ensuring biodiversity protection, mitigating climate change, enhancing rural development and supplying timber and non-wood goods and services". GAYA-J/LP simulates harvest of different species (spruce, pine and birch), qualities and dimensions (sawn wood, pulp wood and logging residue) that forms the basis for a comparison between different uses of the harvest. Several of the modelled production technologies have restrictions on what kind of biomass (species and dimensions) they can utilize. Thus, it was necessary to create a production mix that defined the share of harvest allocated to the different value chains. Paper II, III and IV are therefore analyses based on local data, applying a bottom-up perspective. Nabuurs et al. (2007) reported large differences in estimates of potential climate change contribution by forest, depending on the perspective (bottom-up vs. top-down), and that generally, bottom-up studies have less radical estimates, more detailed data and better knowledge about important assumptions.

5 Results

In this chapter, results are presented that can contribute to answer the research questions asked in Chapter 2, based on the analysis in the four papers.

Environmental assessment

What are the environmental effects of biomass use for a variety of wood products in Norway, and is there a trade-off between ecosystem services and other environmental benefits provided by the wood products?

A qualitative assessment of biomass products compared to other products that provide the same service, show that with regard to GWP, 16 of 17 wood products perform better than the alternative product (Paper I). In other impact categories, the results are mixed (Paper I, II and III).

For bioethanol used as transportation fuel, the analysis show that fossil diesel performs better with regard to acidification potential (AD), eutrophication potential (EP) and particular matter formation (PMFP). The differences between the two fuels are in the range 3-11 %. In the impact categories photochemical oxidant formation (POFP), ozone depletion (ODP) and global warming (GWP), the bioethanol performs better than the fossil diesel. The savings are 19 %, 82 % and 80 %, respectively (Paper II).

When the woody biomass is used for a range of products described by a product mix, the emissions from the processing of the wood products are smaller than the emissions related to production of alternative products (replacement). The products in the product mix have different environmental impacts depending on production methods and replacement products (Paper III). In two (eutrophication and ozone depletion potential) of the five impact categories investigated, wood based packaging have larger value chain emissions than the plastic packaging it is replacing, while construction material provides benefits in all impact categories assessed. For most of the products and impact categories, the product processing is the most influential life cycle stage (Paper III). With regard to global warming potential, all products in the assessment provide GHG savings compared to the alternative products, with construction material and biorefinery providing the largest benefits (Paper III). The savings per m^3 of harvested wood, assuming a production mix which represents local use of the forest resources, varies between 568-614 kg CO_2-eq./m^3, depending on the forest management scenario (Paper III). The savings per km^2 of productive forest vary between 18.4-56.7 ton CO_2-eq/km^2/20 years and 91.4-275.5 ton CO_2-eq/km^2/100 years (Paper III).

As the product mix provides environmental benefits in all impact categories, there is a trade-off between conservation of biological diversity and other environmental impacts. The harvest is limited by the forest management restrictions, and in the biodiversity-scenario, the harvest is more than 60 % lower than in the other scenarios (Paper III). The environmental impact characterizations follow more or less the same trend (Table 2).

Table 2: Relative harvest level, environmental benefits and net present value (NPV) (percentage) of the forest management scenarios when the product mix is applied.

Impact category	REF	PEFC	BD
Harvest	*100 %*	*92 %*	*33 %*
GWP	100 %	95 %	32 %
ODP	100 %	92 %	37 %
POFP	100 %	98 %	28 %
AP	100 %	98 %	39 %
EP	96 %	100 %	35 %
NPV	100 %	94 %	52 %

GWP=global warming potential, ODP=ozon depletion potential, POFP=photochemical oxidant formation potential, AP=acidification potential, EP=eutrophication potential.

In some cases, there will be a trade-off between local and global environmental impacts. The bioethanol in Paper II performs better than fossil diesel with regard to the globally important climate change, but worse regarding the local/regional important eutrophication, acidification and formation of particles.

Forest climate contribution

What is the effect of including biogenic CO_2 and albedo on the estimated climate change mitigation potential of bioenergy based on Norwegian forest resources?

When assessing the global warming potential of bioenergy, the climate neutrality assumption is important as the emissions of biogenic CO_2 from production and use of bioethanol dominates the emissions of GHG (Paper II). If biogenic CO_2 is included in the GHG accounting, it constitutes 84 % of the total GHG emissions in Paper II. Fermenting of sugar and combustion of ethanol are the two most important sources of biogenic CO_2. The warming effect of biogenic CO_2 and the cooling effect of changed albedo have been included in the analysis of the bioethanol by the GWP_{bio}, a characterization factor for biogenic CO_2 (Table 1). Figure 7 illustrates the total emissions of CO_2-eq./km under different accounting strategies for biogenic CO_2.

If the biogenic CO_2 is assumed climate neutral (i.e. GWP=0), the bioethanol provides 80 % lower emissions of CO_2-equvalents per km (Paper II). If the bioenergy is not credited for sequestration of CO_2 by growing biomass, and is assumed to have the same characterization factor as fossil CO_2 (i.e. GWP=1), the bioethanol used for heavy-duty transport produce 33 % more emissions of CO_2-eq./km than fossil diesel.

When the warming effect of biogenic CO_2 and the cooling effect of albedo is included (Figure 7), the savings of CO_2-eq. for bioethanol is 57 % compared to fossil diesel (Paper III). If logging residues are included in the harvesting, both the warming effect of biogenic CO_2 and the cooling effect of albedo is smaller (i.e. closer to zero) assuming the same amount of bioenergy harvested. This means that the climate effect of bioenergy is smaller when harvest residues are collected. The savings of GHG emissions per km driven by bioethanol compared to fossil diesel is 65 % (Paper II).

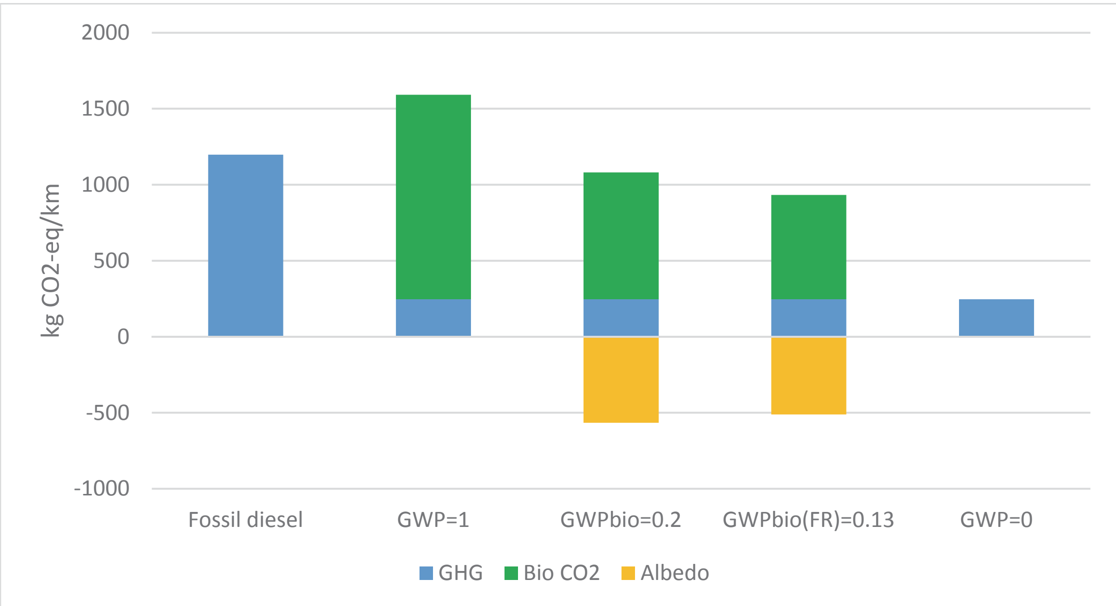

Figure 7: Total amount of CO_2-eq. per km driven by a truck fueled by fossil diesel or bioethanol. For the bioethanol, different assumptions about the climate effect of biogenic CO_2 and albedo are illustrated by GWP values between zero and one. The blue columns represent the emissions of all GHG minus biogenic CO_2 . GWP_{bio} values are found in Table 1. FR=forest residues collected in addition.

In paper IV, the biogenic CO_2 and albedo were included through the forest growth model. The forest owner was credited for carbon sequestration, replacement and climate cooling due to increased albedo and debited for emissions of carbon (harvest). The value of climate contribution was embedded as a price of carbon (0-500 NOK[1]/ton CO_2-eq.) and included in the net present value of the forest. Three levels of potential GHG-savings and albedo temperature response were applied in the optimization (low, medium and high). Figure 8 summarizes the changes in harvest under different assumptions about the potential GHG-savings and temperature effects of albedo when climate change mitigation was valued. The results are shown for both discounting of only timber income and for discounting of all income. When only the timber income is discounted and the potential GHG-savings of wood products (substitution) is sufficiently high and the climatic effect of changing albedo increases (low→high), more wood will be harvested when the forest owner can profit from CO_2 price, in addition to timber sale. If the albedo effect is left out or its potential effect is small, the forest owner will reduce the harvest and the standing stock will increase. At low substitution and albedo effect, the amount of carbon in standing stock is larger and removal of trees will lead to net emissions of CO_2 under the assumptions made. When all income are discounted, the storage of carbon in the forest is less important, and the harvest increases in almost all scenarios. The exception is for the scenarios with low substitution combined with low and medium albedo. Those combinations leads to reduced or stable accumulated harvest as the carbon price increase (Paper IV).

		Only timber income discounted Albedo			All income discounted Albedo		
		Low	Medium	High	Low	Medium	High
Substitution	Low	↓	↓	↑	↓	–	–
	Medium	↓	↓	↑	↑	↑	↑
	High	↓	–	↑	↑	↑	↑

Figure 8: Changes in total harvest relative to no consideration of albedo and substitution effect. The accumulated harvest are different in the scenarios with varying assumptions regarding the climate change mitigation potential of substitution and albedo effect, in addition to the difference between discounting all income or only timber income. The results are presented for 3 % p.a. discount rate, as 2 and 4 % p.a. discount rates show the same trend. The exception is for 2 and 4 % discount rate when all income is discounted. At 2 % p.a. discount where both low-low and low-medium scenario give reduced accumulated harvest. At 4 % p.a. discount rate, all scenarios give increased harvest. The harvest trends, indicated by the arrows, are the same for all levels of CO_2-price (0-500 NOK/ton CO_2-eq.). ↓ indicates reduced harvest, – indicate stable harvest level, ↑ indicate increased harvest.

[1] 1 €=8.4 NOK

How can the forest management and biomass use be optimized for climate change mitigation?

Based on the combination of GAYA-J/LP and LCA, we were able to calculate the GHG emissions saved per m^3 of spruce, pine and birch harvested and used in the value chains described in Paper III. The value chains ranked according to climate change mitigation potential are (from high to low):

1. Construction material (replacement: steel beams).
2. Biorefinery products (replacement: ethanol, guaiacol, cotton linter and superplaticizer).
3. District heating (replacement: light oil).
4. Combined heat and power (replacement: Nordic el-mix and light oil).
5. Cardboard packaging (replacement: plastic packaging).

The potential climate change mitigation contribution of the forest was calculated using a mix of these products. Three production-mix scenarios were constructed; low, medium and high GHG-savings due to replacement. In the "Low" scenario, the biomass harvested is used for production of cardboard packaging and this scenario has the smallest potential GHG-savings. In the "Medium" scenario the biomass is used for all the products in the model (1-5), and the potential GHG-savings are higher than in the "Low" scenario. In the "High" scenario, the biomass is directed to the three products that have the largest potential GHG-savings, resulting in the largest potential GHG-savings per m^3 of wood harvested (Table 3).

Table 3: Three production scenarios that influence the climate change mitigation potential by the forest. The amount of GHG savings for the species varies with the production mix.

Scenario	Construction material	Cogeneration	District heating	Packaging	Biorefinery products	kg CO_2-eq./m^3		
						Spruce	Pine	Birch
Low	0 %	0 %	0 %	100 %	0 %	-112	-112	-112
Medium	17 %	24 %	24 %	13 %	22 %	-639	-595	-339
High	29 %	0 %	29 %	0 %	41 %	-922	-763	-558

What is the potential effect of biodiversity conservation on climate change mitigation contribution from the forest?

In paper III, the climate impact of biogenic CO_2 and albedo were not included, and this resulted in a conflict of interest between climate change mitigation and biological diversity.

In Paper IV, the forest owner was credited for storing carbon and a mutual relationship between carbon storage and conservation of biological diversity as the introduction of a CO_2 price lead to decreased harvest volumes if the substitution effect of forest products were low. In a climate perspective, the value of a m^3 standing stock in the forest was higher than the value of a m^3 harvested. Change in albedo after harvest has the opposite effect, and at high climate impact of albedo, harvest increased as the price of CO_2 increased (Figure 8).

Cost

What are the costs trade-offs between biodiversity conservation and climate mitigation?

In Paper III the forest management were restricted with the goal of conserving biological diversity. In the scenario with the strongest restrictions on forest management, harvest is 8 % lower in the forest certification scenario (PEFC) compared to the reference (REF), and the harvest in the biodiversity scenario (BD) is 63 % lower. The net present value (NPV) of the timber follow the same trend: the PEFC scenario has 6 % lower NPV while the BD scenario has 48 % lower NPV compared to the REF scenario. The NPV at 3% p.a. discount rate was 2 090 000 NOK/km^2 lower in the BD scenario than in the PEFC scenario and 2 366 000 NOK/km^2 lower than in the REF scenario. Assuming that the reduction in NPV represents the cost of biodiversity conservation, the cost vary between 2 090 000 and 2 366 000 NOK/km^2 for this area and forest situation.

Paper I shows that there are large variations in reported abatement costs; varying between -45 and 560 €/ton CO_2. The negative abatement cost was reported for wood chips used for heat production in Austria, where a wood chips boiler replaced an oil-fired boiler. The highest abatement price were reported for wood chips used in a steam turbine to produce electricity for transportation, replacing diesel.

In Paper IV, introduction of a carbon price reduces the NPV of the timber income due to reduced harvest. The reduction in NPV is, in relative terms, smaller than the reduction in harvest levels, as the price of CO_2 increases. The reduction in NPV and harvest are dependent on the choice of discounting and of the discount rate. The harvest level is most influenced by the discounting decision. When only the timber income is discounted, the scenarios with the largest reduction in NPV of timber, reduce the accumulated harvest with almost 100 % at CO_2-price of 500 NOK/ton CO_2 compared to 0 NOK/ton CO_2 (Paper IV). When only harvest income is discounted, the largest reduction in NPV of timber coin sides with the largest reduction in accumulated harvest, i.e. the same combinations of substitution and albedo show largest reduction in both NPV of timber and in accumulated harvest volumes. When all incomes are discounted, the reduction in NPV of timber is associated with increased accumulated harvest when 2 % and 4 % discount rates are applied, and it is not the same combinations of substitution and albedo effect that show largest reduction in accumulated harvest volumes and largest reduction in NPV of timber (Paper IV).

6 Discussions and conclusions

Climate change is a global challenge that all nations should contribute to reduce. Bioenergy can play a critical role for mitigation if applied in a sustainable manner (IPCC, 2013). Together with the agricultural sector, the forestry sector has a unique position, as it can contribute both to reduced emissions and removal of atmospheric carbon (Smith et al., 2014). Norway has unused forest resources that could help to reduce the global warming by decreasing the atmospheric concentration of CO_2 and increasing the reflection of solar radiation. In this study, we have seen that wood can be used to produce several products and services, with high variations regarding environmental impacts. The forest provides a possible climate mitigation option at low marginal cost. Inclusion of biogenic CO_2 and albedo are important for the assessed climate footprint of forest products and services, and forest management can be used to improve forests' contribution to reduce climate change, through carbon sequestration and storage, product replacement and albedo. Depending on the radiative forcing response of albedo and the net carbon flux related to wood products, there will be a positive or negative correlation between climate change mitigation and other ecosystem services, in this thesis represented by conservation of biological diversity. IPCC (2013) states that if bioenergy systems are well-managed, climate mitigation and other societal goals, like conservation of biodiversity, can co-benefit.

Methods

Through the literature review, it became clear that even though the LCA principles and framework are standardized, there are large degrees of freedom within the analysis. There is a vast amount of LCAs available, and there is a wide range of results because of large differences in system boundaries, allocation method and reference system, in addition to underlying assumptions about production system and data. In the four papers in this thesis, it is evident that the assumption of climate neutral bioenergy is important, in addition to the choice of reference system. Cherubini and Strømman (2011) identify functional unit, reference system, inclusion of LULUC and allocation as the methodological choices that are most important for the results.

When analyzing biomass based systems, the environmental reporting usually include contribution to global warming, and some studies also include other impact categories. In Paper I, 13 out of the 25 studies reviewed include information on other impacts besides global warming. All of the 25 studies included GWP. There is an increasing interest in the climate change mitigation potential of wood products. However, in many cases there could be a conflict of interest between local and global effects and this need to be included to a greater extent in environmental assessments of wood products. Like other authors (Bright and Strømman, 2009, Bright et al., 2010, Kilpeläinen et al., 2012, Repo et al., 2012) I found in Paper II that the bioenergy had smaller emissions of GHG than fossil energy. On the other side, the bioenergy had higher emissions in other impact categories that are important for local environment. Many of the bioenergy technologies are still under development and the efficiency and energy balance are continuously being improved (Schlamadinger et al., 1997). This will lead to improved environmental profile of bioenergy products in the future. In addition to the environmental and economic motivation for use of forest, the use of wood products also contribute to other needs, like rural development, energy security and easy storage that can help balancing energy supply. For sustainable forestry, these factors should be included in the analysis.

In Paper III, we combined a forest model with LCA in order to analyze the potential environmental and economic impacts of forest product mix, including biodiversity and recreational value. The results indicate that the CO_2 savings are greatest when the biomass is used for construction rather than bioenergy, and this is in line with other studies (Eriksson et al., 2012, Oliver et al., 2013). In Paper IV, the integration of the forest model and LCA is used to keep track of carbon in a forest value chain and we also included the radiative forcing effect of changing albedo after harvest. The main advantage of this methodology is that the local factors plus time and space aspect are included. This forest model builds on the most used functions for growth, natural mortality and regeneration in Norway, and local factors that are important for economy, biological diversity, carbon sequestration and storage are included. Another strength of the model is that it can be used to assess relevant environmental impacts, not only keeping track of the carbon, alongside with cost and income.

Including albedo and biogenic CO_2

In two of the thesis papers, biogenic CO_2 and albedo have been included and the papers demonstrate that the choice about including these factors is very important for the analysis of the forest's potential contribution to reduce global warming. In Paper III, the biogenic CO_2 is considered climate neutral. In that case, the use of wood products has a large climate change mitigation effects. In Paper II, I explore different GWP characterization factors for biogenic CO_2, and find that even though the climate benefits of the bioethanol is smaller than it is if the bioenergy is assumed climate neutral, bioethanol may still provide GHG-savings compared to fossil diesel. In Paper IV, all CO_2 emissions irrespective of source, and changing albedo are included in the analysis. There are three different levels of GHG-savings due to replacement and three different levels of radiative forcing effect of changing albedo being explored. For variations in these levels, the modelling resulted in both increased and decreased harvest as a CO_2-tax was introduced. If the climate effect of albedo was small, the climate balance of the forest value chain was better if the CO_2 was stored in the forest rather than used for replacement of fossil products, for the 50-year period considered.

The climatic effect of biogenic CO_2 and albedo were in Paper II and IV assessed in two different ways: with a characterization factor for biogenic CO_2 (GWP_{bio}) and with a forest model (GAYA-J/LP) combined with LCA. Both methods have their advantages and disadvantages. The GWP_{bio} has the advantages that it is easy to apply for LCA practitioners and it facilitates a direct comparison with fossil fuel. The combination of LCA and GAYA-J/LP is not as easy to apply as the GWP_{bio} but it provides a site-specific and time-specific carbon accounting, and includes the economic aspects. This makes the method well suited for policy-making purposes, analyses of future scenarios and when the analyst wants to include considerations of ecosystem services and economy.

Forest management

Different wood products provide different environmental benefits, and depending on the environmental goals specified, the forest can be managed in ways that increase the production of raw material for certain products. There is a significant difference between the potential GHG-savings of the production mix scenarios in Paper IV. In order to optimize the climate change mitigation, the forest management can be aimed at producing sawn wood timber rather than pulpwood. The product mix can be modified by e.g. legislation (for example blend-in tariffs) or by introducing taxes and/or subsidies. GAYA-J/LP in combination with LCA can facilitate analyses of the environmental and economic effects of varying policy scenarios. Moreover, with this method, biological diversity and other ecosystem services can be included as premise providers for the biomass harvest, rather than impact categories affected by harvest. By using the forest model with space specific considerations, other aspects of biological diversity, like diversity of ecosystems, can be included. Biological diversity is not the only ecosystem service affected by forest activity, and by combining LCA and GAYA-J/LP, other land use considerations can be included. For example, an area with low biodiversity can be highly valued as a recreational area, and if land use is included in LCA only as an impact on biodiversity, the forest managers may overlook such aspects.

Uncertainties

Models are simplifications of the real world and will always yield some uncertainty because the real world is never as straight forward as models are. However, models facilitate comparison of possible developments and, most importantly, comparison of the impacts of alternative assumptions. In this sense, the simplicity creates more transparent and clear analysis, with possibilities for improved and consistent discussions, compared to what would be the case without modelling.

In this study, there are possible sources of error both in the forest modelling and in the value chain modelling that forms the fundament for the LCA. Development of the forest and, hence, the possible carbon storage and timber yield depend on the growth, mortality and regeneration in the forest model. These functions have been developed for historical climate, stand densities and stand ages. In analysis of future scenario, modelled stand densities and age could go far beyond the range that the applied models were created for and this could give overestimations of the possible timber yields and carbon storage capacities. When forest is set aside and the stand age and density increase, risk of disturbances (forest fire, wind damage, pest, climate change, and pollution) also increases and this creates uncertainty. If disturbances leads to loss of live biomass, the carbon in the biomass will be emitted as CO_2 without any substitution benefit. Lundmark et al. (2014) analyzed the carbon balance of Swedish forest, and they found that a storm-felling in the dataset resulted in significant loss of biomass and emissions of CO_2.

The radiative forcing effect of albedo is not fully understood yet. Snow cover is important for the albedo effect and a changing climate could mean less snow cover in the areas covered in this thesis. In the LCA model, the replacement product is very important for the results.

The assumptions about timber prices and forest operational costs are based on historical data, and their relative development are assumed constant in the modelling. An increase in demand for wood may easily result in increased prices of timber. This is also true for fossil resources, as the global production of fossil oil has probably peaked (Chapman, 2014). In Paper III and IV, the forest investigated is relatively small, so it is fair to assume that a change in the harvest level will not influence the market price of timber. However, if the model would be applied for a larger area, the forest model and LCA should be extended with a market model.

In Paper IV, we tested different levels of CO_2 prices, from 0 to 500 NOK/ton CO_2. Part of Norway's emissions of CO_2 is already subject to carbon tax. There are large uncertainties about how the carbon price will develop, but according to Bjørkum et al. (2009) the price will most likely increase. Moreover, if the carbon price is to function as a tool to mitigate GHG emissions, it must increase. In the most ambitious climate policy, Bjørkum et al. (2009) expect the CO_2 price to exceed 100 €/ton CO_2 by 2030, but this is not likely in the political landscape of today..

In all analyses of the thesis, the soil carbon has been left out of the forest carbon budget. This is due to two considerations. First, the largest change in carbon stock is expected to be in the above-ground biomass (Hoen and Solberg, 1994, de Wit et al., 2006) Second, there are large uncertainties regarding the relationship between forest management and soil carbon storage (de Wit et al., 2006, Lundmark et al., 2014). The soil carbon has been included in earlier studies with GAYA-J/LP (for example in Hoen and Solberg, 1994, Raymer et al., 2009), and the carbon models should be included again when they are improved.

The main emphasis in this work has been on the integration of LCA and GAYA-J/LP, while the underlying assumptions in these tools have not been intensively analyzed. For example, the choice of using only 50 years as optimization period and the assumption that the economic behavior of the forest owner is profit maximizing are given as prevailing condition and not challenged. The importance of these assumption can easily be checked in further analyses with this model, by changing the optimization objective functions (including model constraints) and by extending the optimization period.

The results in a wider perspective

Depending on assumptions in the modelling, the results indicate both increased and decreased harvest in order to optimize the climate change mitigation potential of the forest. This finding leads us to one of the main questions concerning biomass use, regardless if it is used for energy or other products: should we use fossil carbon or biological carbon? We depend on carbon, and as mentioned, carbon is stored in the lithosphere, biosphere, soil, ocean and atmosphere. If we want to reduce the storage in the atmosphere, we need to remove carbon from the atmosphere, and store it in some of the other compartments. So, how can we manage the carbon storage in order to reduce the atmospheric concentration of CO_2? An increase in the carbon storage in the biosphere by reducing harvest means that we will harvest and consume carbon from another compartment. The technology to harvest from the lithosphere is highly developed, but the transport of carbon to the lithosphere is part of the long-term carbon cycle, and today's carbon capture and storage technology have not developed enough to make us able to move large quantities of carbon back into the lithosphere. However, fossil carbon is readily available and bioenergy will in most cases produce more atmospheric CO_2 per energy unit than fossil, creating a temporary carbon debt (Schlamadinger et al., 1997). Therefore, to answer the above stated question, the assessments of bioenergy and biomass use must take into account the distribution of all near-term and long-term effects for both fossil and biogenic CO_2. Assumptions about substitution, carbon storage in biomass and the albedo effect as well as the time scale and time preferences considered, are essential in order to answer the questions regarding use of carbon from biomass versus carbon from fossil resources.

Future research

Already two decades ago, Hoen and Solberg (1994) emphasized the need for more research related to regeneration, fertilization, growth and stability in old stands. Still, this need is of high importance for Norwegian forestry in relation to forest climate mitigation. A collapse of forest stands due to wind, insects, snow or fungi will decrease the proportion of wood going to construction and drastically reduce the economic surplus from forestry, as well as most likely lead to higher GHG emissions from the forests.

In order to study a proper balance between biodiversity, albedo and timber growth effects, more sophisticated growth functions should be applied than the basal area growth functions used in this study. New functions have been developed (Bollandsås and Næsset, 2009), but have not been incorporated in the forest model yet. The new functions should be tried out in future studies like this thesis. Also, climate dependent growth and mortality functions should be developed to make possible inclusion of climate change. Also there are others factors of the forest's potential climate change contribution depend on future climate that could be embedded in future analysis.

As there is large uncertainty about the soil carbon under different harvest levels, it is important to further improve our understanding of this relationship. More accurate functions for the albedo impacts and improved knowledge about development of albedo under climate change with less snow, is also needed.

Fertilization of forests and extraction of stumps for increased bioenergy production are controversial forest measures seen from biodiversity and most other environmental points of view. However, fertilization is widely used in Sweden for increasing the forest growth, and stump extraction is practiced in Finland for bioenergy production. Both production systems should be evaluated using the combination of LCA and forest modelling applied in this thesis.

It is important to test the method used in Papers III and IV on other geographical areas than done in this thesis, both in the same region with similar forestry value chains, and in other regions with different value chains, forest growth and climate/albedo conditions. The choice of analysis period should also be tested by prolonging the optimization period to 150 years or longer, in order to capture the long-term climate impacts.

The method should also be tested for larger regions, taking into considerations the carbon leakage effect – i.e. that a harvest increase in one area will affect the harvest quantities in other areas, all other factors equal. This impact is often left out in forest climate mitigation analyses, but is likely to be significant. To include carbon leakage effects one needs to apply partial equilibrium models which include trade, like NorFor (Sjølie et al., 2013a, b) or EFI-GTM (Moiseyev et al., 2014).

The data quality in the LCA model should be under continuous improvement. One should continue to develop LCA methodology in order to capture the time and space issues that are important in environmental assessments, particularly when assessing use of biological resources. Increased emphasize should be given to all aspects that could improve the inclusion of the end-of-life phase and the effect of cascading. Here, questions related to expected future technology changes are important, as well as applying clearly defined system boundaries.

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PAPER I

Environmental impacts and costs of using wood from boreal forest for climate change mitigation: a review of recent studies in Scandinavia

Ellen Soldal and Birger Solberg

Abstract

This paper has as main goal to (i) review the results of recent Scandinavian studies of environmental and economic impacts of using wood from boreal forests for climate change mitigation; and (ii) identify main possibilities for improving such analyses. The reviewed environmental studies show that woody biomass can contribute to global climate change mitigation. However, regarding locally important environmental impact categories like waste handling, acidification and eutrophication, the results are mixed. Few of the reviewed studies include considerations to ecosystem services. There are large variations within the life cycle assessment (LCA) results, depending on system boundaries, type of feedstock, energy conversion technology, and type of fossil fuel replaced. The reported costs in the economic studies vary from -45 €/ton CO_2-eq. up to 400 €/ton CO_2-eq. The price of energy produced by bioenergy varies between types of energy, from 12 €/MWh to 311 €/MWh. The study show that geographical specific data, like forest yield and management, are important for the economic as well as the environmental results.

Future research should look at how the LCA methodology could be made more uniform for easier comparisons, at the same time as analysis should be based on local data as far as possible. The transparency of data collection and assumptions should be increased in order to facilitate comparisons. Potential tradeoffs between benefits in global issues and disadvantages in local issues are important to document. Attention should be given to better include the spatial and timing elements, costs and cost efficiency estimates, wood cascading, carbon leakage, climate change impacts on net forest carbon sequestration, and how harvest influences carbon storage in the soil. Further, more research should be done on the optimal forest management, harvesting and use of wood in a climate change mitigation regime.

1. Introduction

There are strong evidence that increased level of atmospheric carbon dioxide (CO_2) and other greenhouse gases (GHG) is the main reason for observed climate change (IPCC, 2013). CO_2 is the most significant GHG, with a lifetime of several thousand years (Forster et al., 2007), and is of particular importance in mitigating climate change (Bowerman et al., 2013).

In order to reduce climate change through low carbon emission solutions, biomass is being explored as an alternative to non-renewable sources of energy and material. Woody biomass is a widely applicable and available resource that can substitute many products based on fossil resources, like energy, chemicals and construction material. Forest can contribute to climate change mitigation through carbon sequestration and storing of CO_2, land use management and through avoided emissions by substituting non-renewable energy and materials (Smith et al., 2014). Wood-based products can improve the carbon balance in several ways; by demanding lower input of fossil energy, avoiding industrial process emissions from cement production, increasing supply of by-products that can be used for energy production, and by storing carbon in long-lived products (Eriksson et al., 2012).

Life cycle assessment (LCA) is a tool to quantify environmental impacts across the life cycle of a product or service. Material and energy requirements for production, use and disposal of the product of interest are gathered and resulting emissions are allocated to the product as described by the ISO 14044 standardization (ISO, 2006). The method provides a tool for understanding the most important environmental impacts of production systems, and where in the production chain these impacts occur. It is used to compare product alternatives, and as a basis for strategic and policy decision making. Environmental load associated with production and use of a product or service is allocated to a unit of input or output. This is referred to as functional unit (FU). Depending on the aim of the study, analysis of environmental impact of wood products usually has two points of departure. The product can be compared to an alternative product that provides the same service (output approach), or it can be compared to an alternative product that is based on the same resource (input approach) (Rivela et al., 2006). In forestry related research, functional unit is often 1 m^3 of solid wood (González-García et al., 2009a).

Traditionally, costs have not been reported in LCAs. However, production costs (including costs of non-marketed goods and services) are strong indications of use of scarce resources and are important in judging the possibilities for implementing a given production. Costs may also give valuable indications for policy making regarding for example how large subsidies or taxes need to be in order to achieve political goals, and for making it possible to combine and compare consistently different types of goods and services in bio-economic modeling, like in Hoen & Solberg (1994), Raymer et al. (2009) and Sjølie et al. (2014).

This paper has as main goal to (i) review the results of recent Scandinavian studies of environmental and economic impacts of using wood from boreal forests for climate mitigation; and (ii) identify main possibilities for improving such analyses. By wood, we here refer to roundwood, forest residues, chips and forest products. The reviewed environmental studies can be classified as life cycle analysis, and additional justification for this study, besides meeting the above stated goals, is to identify main possibilities for making it easier than at present to combine LCAs with bio-economic modelling.

The review focuses on analysis from Norway, Sweden and Finland. However, because very few cost studies were found from these countries, a few studies from Austria and Germany are also included. To make results representative of today's forest practices and processing of biomass, we have limited the review to papers published during the last decade. We have emphasized to obtain results from all major value chains in the forest sector: the pulp and paper, stem wood and construction, bio-energy and bio-chemical value chains (González-García et al., 2011).

In the following, methods used for comparison and main assumptions are described, then results are presented and discussed, and conclusions drawn.

2. Methodology

The following criteria for selection of studies have been used:

- Include processing of woody biomass.
- Include information on global warming potential (GWP).
- Published in 2005 or later for environmental studies and 2004 or later for economic studies.

- Geographically located in Norway, Sweden or Finland (with the already mentioned exception regarding cost analyses).

The reviewed LCA studies use different FU, like kWh, MJ, m^3 or m^2 floor area. The reviewed studies include mainly articles published in peer-reviewed journals, but also a few reports. To facilitate the use of the results in forest bio-economic modelling using m^3 harvest as link between LCA results and traditional forest modelling, we have recalculated the carbon footprint to a common FU, namely harvested m^3 solid wood over bark (m^3 s.o.b.) where this seemed possible and relevant. The recalculations are based on heating values, density and information given in the studies.

Choice of system boundaries in LCA defines which unit processes are included in the analysis (ISO, 2006). In the reviewed studies, system boundaries vary. Most studies include natural resource extraction, processing and transport, while many also include use/operation and end-of-life treatment. Most of the studies assume that biogenic CO_2 is climate neutral, while others also include carbon sequestration of forest biomass and changes in soil carbon. Our review identifies the main variations in system boundary assumptions.

When assessing environmental impacts of biomass use, the two most common indicators which have been used in the reviewed studies are energy consumption and GWP – the latter measured with the unit CO_2-equivalents per functional unit. Therefore, not all of the reviewed studies give information about other environmental impact categories. Environmental impacts are listed in two separate tables, one that includes only GWP estimates (Table 1) and one that includes only a qualitative assessment of other environmental impact categories than GWP (Table 2). A more exhaustive overview of environmental impacts (excluding GWP) than Table 2, is presented in the appendix (Table A1), focusing on quantifying these environmental impacts. In Table 3, the recalculated GHG emissions for a common unit, m^3 s.o.b., are listed. Finally, Table 4 summarizes the production and abatement costs of using woody biomass as reported in the few studies found to include cost estimates.

3. Results

3.1. LCAs of wood use

The main environmental results from the reviewed studies are summarized in Table 1 and Table 2. In the following sections, the results are outlined further for each of the product groups: pulp and paper, stem wood and construction material, energy and biorefinery products.

3.1.1. Pulp and paper

Pulp and paper production demands large quantities of energy and chemicals, and production of biomass (raw material) contributes to only a small part of the total carbon emissions (Ghose and Chinga-Carrasco, 2013; González-García et al., 2011; Judl et al., 2011). Judl et al. (2011) found that chemicals used in pulp production have rather strong environmental impacts. González-García et al. (2009b) report that also the transport of biomass from forest to pulp mill has considerable environmental impacts in paper production.

Because production of pulp is energy intensive, assumptions about the energy mix used is important for the result (Ghose and Chinga-Carrasco, 2013). Judl et al. (2011) report 441 kg CO_2-eq./air dried ton of pulp, while González-García et al. (2011) find that one air dried ton of pulp causes 415 kg CO_2-eq. The system boundaries in the two studies are similar, but there are some differences between them that can explain the variations in results. In Judl et al. (2011) the pulp is based on birch, while in González-García et al. (2011) the production is based on a mix of spruce (20 %) and pine (80 %) and the difference in fiber properties in softwood and hardwood can influence the use of chemicals in the delignification process (Judl et al. 2011). Moreover, the production of pulp in González-García et al. (2011) takes place at a biorefinery which also produces ethanol and lignosulfonates, and the environmental burdens are allocated using economic allocation. In this case, all the black liquor produced is recovered, either as by-products, energy or chemical recovery. In Judl et al. (2011) the black liquor is combusted and produce electricity that replace Finnish grid electricity. The environmental burdens are allocated using system expansion.

Both Nors et al. (2009) and Ghose and Chinga-Carrasco (2013) study newsprint, respectively from Finland and Norway. The Finnish newsprint production emits 760 kg CO_2-eq./ton paper compared to 512 kg CO_2-eq./ton for the Norwegian paper. The Finnish system includes more life cycle phases, amongst them acquisition of the raw materials while this has been left out of the Norwegian analysis. However, the fiber supply only accounts for 3 % of the carbon footprint. The most important emittance comes from the energy use, and in the Finnish case, the domestic electricity supply includes a significant share of coal electricity. Ghose and Chinga-Carrasco (2013) assume NORDEL electricity mix, which has a smaller share of coal and a larger share of hydro power. They demonstrate how important the electricity mix is by assuming Norwegian electricity and thereby reducing the global warming potential from 512 to 211 kg CO_2-eq./ton newsprint.

None of the reviewed studies compares pulp production with a reference product based on fossil resources.

3.1.2. Stem wood and construction material

Valente et al. (2011) and Michelsen et al. (2008) investigate two similar systems, namely wood logs delivered to customer, and the GHG emissions reported are 11.8 kg CO_2-eq./m^3 and 25.1 kg CO_2-eq./m^3, respectively. The two cases have similar system boundaries, and include regeneration, silviculture, logging, terrain transport and road transport to customer. However, the system described in Michelsen et al. (2008), also includes planning of forest operation and construction and maintenance of forest roads. The consumption of diesel (liters per m^3) in the two cases are different, with a variation of 0.17 l/m^3 for the processes they have in common. The diesel consumption is highest in Valente et al. (2011), but Michelsen et al. (2008) have the highest emissions of CO_2-equivalents per m^3. The two studies use different functional unit. Valente et al. (2011) used m^3 over bark, while Michelsen et al. (2008) used m^3 under bark. The bark generally constitutes about 10 % of the stem volume.

Production of cement is the largest source of non-energy related industrial emissions of anthropogenic CO_2. Hence, replacement of cement can be a significant climate change mitigation action (Gustavsson et al., 2006). As shown in Table 1 many studies have concluded that there are large climate benefits connected with replacing concrete or steel constructions with wood and this finding has been discovered by others earlier (see for example Petersen and Solberg, 2002; 2005). Gustavsson et al. (2006) find that replacing concrete buildings with wood-framed buildings provide GHG-savings between 25 and 129 kg CO_2-eq./m^2 floor area. Wærp et al. (2009) analyzed the impact of sawn wood, and expanded the system boundaries to include assembly and maintenance for the expected life time. The impact per m^3 of product was 19 kg CO_2-eq., and also this number is lower than what has been reported by Michelsen et al. (2008), even though the system includes more processes. This could be explained by a difference in analysis method. Michelsen et al. (2008) use a hybrid LCA, while Wærp et al. (2009) and Valente et al. (2011) use traditional LCA. Literature on hybrid LCA state that this method generally capture more of the total emissions because it has a more complete inclusion of upstream activities (Michelsen et al. 2008).

Wærp et al. (2009) found that for production of untreated construction materials the largest share of impacts are related to energy used in the production, while for treated materials, there are significant environmental burdens related to the use of chemicals. For example the GWP of wood paneling increases more than ten times if treated with paint – i.e. from 0.4 kg CO_2-eq./m^2 to 5.6 kg CO_2-eq./m^2 (Wærp et al., 2009).

In analysis of buildings that include the whole lifetime of the building, the user phase is the most energy demanding stage (Gustavsson and Joelsson, 2010; Gustavsson et al., 2010). This can be counteracted by adding more insulation, but that could again lead to increased use of energy and material in production (Gustavsson and Joelsson, 2010). Gustavsson et al. (2006) compared two different 4-storey buildings based on wood with similar buildings based on concrete, and found that wood-framed buildings produce less emissions of CO_2 to the atmosphere and they demand less energy. Over a 100 year time horizon, they found that wood-framed buildings have negative net emissions of CO_2-equivalents if forest residues produced in the process replace fossil fuel and construction material replace fossil fuel at end of life (Gustavsson et al., 2006). As shown in Table 1, GHG-savings appear to be about 20 % greater in the case where coal is replaced by biomass for energy production, rather than natural gas, even though emissions due to fossil fuel during production are smaller in the latter case. The wooden buildings demands less energy for production, and the share of biofuels is larger for these buildings. The potential GHG-savings occur due to replacement of fossil fuels by biomass. Change in both forest and building carbon stock is equal to zero after 100 years (Gustavsson et al., 2006). The greatest contribution to climate change mitigation is made when wood is used for construction first, and then for energy at end-of-life (Gustavsson et al., 2006).

3.1.3. Energy

Biomass can be used to produce different kinds of energy, like heat, electricity and transportation fuel. Production of these energy types vary greatly by production method and scale, varying from firewood in households to large industrial processes for production of ethanol or biodiesel. There is a consensus in the reviewed literature that technology used in combustion is the most important factor in analyses of energy systems, while transport is one of the least important (Raymer, 2006; Solli et al., 2009). If the share of import increases, transport becomes more important. Solli et al. (2009) found that if transport were increased by a factor of four, the transport became the most important stage regarding GWP in the life cycle of firewood. Other important factors are what product the bioenergy replaces and the characteristics of the bioenergy, like wood density and heating values (Raymer, 2006). In a comprehensive comparison of wood as heating source, Raymer (2006) found that for one m^3 of wood used for energy, avoided emissions varied from 210 to 640 kg CO_2-eq./m^3 or from 250 to 360 ton CO_2-eq./GWh energy produced. Greatest savings are evident when wood replaces fuel oil.

Wood can be used to produce electricity and heat in combined heat and power plants (CHP). Depending on system boundaries, transport distance, raw material and efficiency of energy conversion GWP for this type of electricity varies between 2.4 to 30.6 g CO_2-eq./kWh (Table 1, Brekke et al. 2008; Guest et al. 2011; Buonocore et al. 2012). For heat production, the GWP varies between 0.7 and 242 g CO_2-eq./kWh. Buonocore et al. (2012) reported 370 g CO_2-eq./kWh, and this is compared to a coal power plant which emitted 1109 g CO_2-eq./kWh and electricity produced by natural gas with emittance of 759 g CO_2-eq./kWh. Brekke et al. (2008) and Guest et al. (2011) analyzed different sized CHP in Norway based on Norwegian forest resources. Both studies include a CHP size of 1 MW. Brekke et al. (2008) found that the emissions of CO_2-equivalents were 12.3 and 14.3 g CO_2-eq./kWh_{energy}, depending on the share of available biomass harvested and, hence, the transport distance. In Guest et al. (2011) the emissions from the same sized CHP, were 11.2 g CO_2-eq./MJ_{energy}, or 40.3 g CO_2-eq./kWh_{energy}. Guest et al. (2011) have, contrary to Brekke et al. (2008) included biomass production and infrastructure for delivery of energy to customer, including energy loss in the infrastructure. In addition the assumed transport is longer in Guest et al. (2011). The two studies operate with different efficiencies, Guest et al. (2011) have assume higher efficiency for the electricity, while Brekke et al. (2008) assume higher thermal efficiency.

Jäppinen et al. (2014) found that when soil carbon changes were included in their analyses, it dominated the emissions of CO_2 from the value chain, especially if stumps were harvested. If the soil carbon changes were disregarded, the biofuels provided potential GHG-savings ranging from 94-98 %, and the savings decreased to the range 48-91 % when soil carbon were included (Jäppinen et al., 2014).

From Table 2 we see that regarding acidification potential (AP), González-Garcia et al. (2012a) find that electricity produced by gasification performs better than EU electricity mix, while Swedish electricity mix has lower impact in this category. According to Brekke et al. (2008), electricity production from two sizes of CHP (1 MW and 10 MW) fueled with forest resources performed better with regard to AP and photochemical oxidant formation potential (POFP) than Nordic electricity mix. With regard to eutrophication (EP), the comparison op.cit. with Nordic electricity mix depends on production size, where the smaller (1 MW) CHP has the highest impact in this category (per kWh).

González-Garcia et al. (2012b) modelled two energy crop production systems growing short rotation willow fertilized and non-fertilized. The non-fertilized system have lower yield, 4 tons/ha/year versus 6.7 tons/ha/year, but the production of fertilizer makes the non-fertilized case better in all impact categories except global warming potential. This case illustrates the conflict which may arise between climate change and other environmental issues. Per hectar, the difference between the fertilized and non-fertilized case is 123 kg CO_2-equivalents, but per MJ energy produced the difference is 0.005 g in favor of the non-fertilized case.

3.1.4. Biorefinery

A biorefinery produces several products, based on biomass input, that can replace non-wood products (Cherubini, 2010), like chemicals, transportation fuel and cellulose (Rødsrud et al., 2012). In a comparison of bioethanol from spruce with bioethanol from ethylene or sugar cane, lignocellulosic ethanol has a significantly lower carbon footprint (Rødsrud et al., 2012). From Table 1 and 2 it is seen that production and use of bioethanol or biodiesel compared with conventional petrol or diesel, show improvements in GWP, human toxicity potential

(HTP), photochemical oxidation formation potential (POFP), ozone depletion potential (ODP) and acidification potential (AP). Regarding eutrophication (EP), some find improvements while others find that bio- based products have larger impacts than fossil-based products. For the bioethanol reviewed, the GWP varies between 14.2 and 21.7 g CO_2-eq./MJ, and the biodiesel has a reported GWP of 12 g CO_2-eq./MJ when soil carbon impacts are not included. The European Union has decided on a fossil fuel standard emission value of 83.8 g CO_2-eq./MJ (Koponen et al., 2013).

Table 1 : Carbon footprint of wood products. Some of the wood products are compared with alternative production systems. In those cases (+) indicates that the wood based system performed better than the reference system and (÷) indicates that the wood based system performed worse than the reference system. Eight System boundaries are used: I Production of raw material, II Processing of raw material, III Transport, IV Infrastructure use, V Operation/use, VI End of life, VII Substitution, VIII Forest carbon. The studies are listed chronologically.

Study	Input	Product	Description of system		System	GWP	Climate mitigation comparison
Wihersaari (2005)	Logging residue - Norway spruce	CHP	Nitrogen loss compensated for by fertilization		I-III, V	13-16 kg CO_2-eq./MWh_{chips}	
		CHP	Nitrogen loss not compensated for by fertilization		I-III, V	6-9 kg CO_2-eq./MWh_{chips}	
Gustavsson et al. (2006)	Spruce and pine	4-story wood building, 16 apts. 1190 m^2 floor area, compared to concrete building	Net CO_2 emission for the 100-year lifecycle of the buildings includes carbon stock changes occurring during the building's lifecycle (incl. re-growth of the forest, uptake of CO_2 by cement carbonation reactions, demolition of the buildings, and substitution of fossil fuel by wood-based demolition waste).	Coal used to generate electricity for material production.	I-IV, VI-VIII	-41.4 tonnes CO_2-eq. (-34.8 kg CO_2-eq./m^2)	**(+)** 29-56 kg CO_2-eq./m^2
				Natural gas used to generate electricity for material production.	I-IV, VI-VIII	-12.2 tonnes CO_2-eq. (-10.3 kg CO_2-eq./m^2)	**(+)** 25-41 kg CO_2-eq./m^2
		4-story wood building, 21 apts. 1175 m^2 floor area, compared to concrete building	Same as above.	Coal used to generate electricity for material production.	I-IV, VI-VIII	-76.2 tonnes CO_2-eq. (-64.9 kg CO_2-eq./m^2)	**(+)** 34-129 kg CO_2-eq./m^2
				Natural gas used to generate electricity for material production.	I-IV, VI-VIII	-30.6 tonnes CO_2-eq. (-26 kg CO_2-eq./m^2)	**(+)** 35-88 kg CO_2-eq/.m^2

Study	Input	Product	Description of system	System	GWP	Climate mitigation comparison
Raymer (2006)	Pine fuel wood	Heat - dwellings		I-III, V	61 kg CO_2-eq./m^3	
	Birch fuel wood	Heat - dwellings		I-III, V	73 kg CO_2-eq./m^3	
	Sawdust from spruce	Heat - Industry		I-III, V	25 kg CO_2-eq./m^3	
	Bark	Heat - Industry		I-III, V	20 kg CO_2-eq./m^3	
	Briquettes	Heat - Industry		I-III, V	54 kg CO_2-eq./ton	
	Pellets	Heat - Industry		I-III, V	59 kg CO_2-eq./ton	
	Demolition wood	Heat - Industry		III, V	12 kg CO_2-eq./m^3	
Brekke et al. (2008)	Chips based on forest biomass + residues 50 % of available biomass harvested	CHP - 1 MW facility	Electricity prod: 4 GWh, heat prod: 16 GWh (input: 11 691 fm3) Energy balance (In/out): 1.25. 2138 kWh/fm^3. Transport distance: 17 km	II-IV	14.3 g CO_2-eq./kWh_{energy}	
	Chips based on forest biomass + residues 100 % of available biomass harvested	CHP - 1 MW facility	Electricity prod: 4 GWh, heat prod: 16 GWh. Allocation: energy. Energy balance (In/out): 1.25. 2138 kWh/fm^3. Transport distance: 6 km	II-IV	12.3 g CO_2-eq./kWh_{energy}	
	Chips based on forest biomass + residues 50 % of available biomass harvested	CHP - 10 MW facility	Electricity prod: 40 GWh, heat prod: 120 GWh (input 93 528 fm3). Allocation: energy. Energy balance (In/out): 1.25. 2138 kWh/fm^3. Transport distance: 47 km	II-IV	11.9 g CO_2-eq./kWh_{energy}	

	Chips based on forest biomass + residues 100 % of available biomass harvested	CHP - 10 MW facility	Electricity prod: 40 GWh, heat prod: 120 GWh. Allocation: energy. Energy balance (In/out): 1.25. 2138 kWh/fm3. Transport distance: 17 km	II-IV	9.8 g CO_2-eq./kWh_{energy}
Michelsen et al. (2008)		Roundwood logs	FU= 1 m^3 of roundwood logs transported to customer	I, III	25.1 kg CO_2-eq./m^3 s.u.b
González-García et al. (2009a)	Spruce and pine	Solid wood for pulp production	Assume 100 % Swedish wood.	I, III	36.1 kg CO_2-eq./m^3 s.u.b
Hagberg et al. (2009)	Sawdust	Pellets -heating plant 100 MW	Swedish electricity mix. Biomass used for drying.	I-III, V	3.4 g CO_2-eq./MJ
	Cutter dust/dry chips	Pellets -heating plant 100 MW	Swedish electricity mix.	I-III, V	3.7 g CO_2-eq./MJ
	Roundwood chips	Pellets -heating plant 100 MW	Swedish electricity mix. Biomass used for drying.	I-III, V	3.6 g CO_2-eq./MJ
Nors et al. (2009)		Newsprint	Incl. printing.	I-III, V	760 kg CO_2-eq./ton
		Magazine paper	Incl. printing.	I-III, V	1130 kg CO_2-eq./ton
Solli et al. (2009)	Birch fuel wood	Heat, household – old stove	Assume LHV: 2650 kWh/m^3. Efficiency: 50 %	I-V	110 g CO_2-eq./kWh
		Heat, household – new stove	Assume LHV: 2650 kWh/m^3. Efficiency: 70 %	I-V	77 g CO_2-eq./kWh
Wærp et al. (2009)	Spruce and pine	Sawn wood	Untreated material, FU= 1 m^3 assembled and maintained for 60 years. Does not incl. energy consumption in use phase. All of the waste goes to energy- or material recovery. More than 80 % of energy consumed is renewable.	I-II	19.1 kg CO_2-eq./m^3

Study	Input	Product	Description of system	System	GWP	Climate mitigation comparison
Wærp et al. (2009)	Spruce and pine	Planed timber	The construction material is untreated, FU= 1 m^3 assembled and maintained for 60 years. Does not incl. energy use in use phase. All of the waste goes to energy- or material recovery. Almost 80 % renewable energy.	I-VII	28.9 kg CO_2-eq./m^3	
		Wood paneling	Untreated paneling for indoor use, lifetime 30 years. About 80 % of energy renewable. 1 m2 of wood paneling needs 0.0162 m^3 planned timber.	I-VII	0.4 kg CO_2-eq./m^2	
		Planed timber, steeped with copper	Copper steeped. Expected life span: 60 years. 64 % renewable energy.	I-VII	52.6 kg CO_2-eq./m^3	
		Wood paneling, treated with paint	Expected life span: 50 years. Almost 90 % renewable energy. 1 m^2 of wood paneling needs 0.0162 m^3 planned timber.	I-VII	5.6 kg CO_2-eq./m^2	
		Massive beams	1 m^3 massive beams based on sawn wood, assembled and maintained for 60 years. Density: 500 kg/m^3. 1.8 % glue. 70 % renewable energy.	I-VII	103 kg CO_2-eq./m^3	
		Gluelam	1 m^3 gluelam beams, based on sawn wood, assembled and maintained for 60 years. Density: 470 kg/m^3. 1.7 % glue (=8.3 kg/m^3). 76 % renewable energy. Input: sawn timber 493 kg/^{m3}	I-VII	79 kg CO_2-eq./m^3	
Bright and Strømman (2009)	Mainly spruce.	Ethanol E100 - best case biochemical	Assumption: LHV=21.5 MJ/kg DM. Yield: 261 liters/ton feed. E85: 74.6 g CO2-eq./km. Reference: gasoline.	I-IV	E100 (at gate): 21.7 g CO_2-eq./MJ	**(+)** 83 g CO_2-eq./km
		Ethanol E100 - best case thermochemical	Assumption: LHV=21.5 MJ/kg DM. Yield: 276 liters/ton feed. E85: 60.3 g CO2-eq./km. Reference: gasoline.	I-IV	E100 (at gate): 14.2 g CO_2-eq./MJ	**(+)** 98 g CO_2-eq./km

Gustavsson et al. (2010)	Lumber (spruce, pine, oak), glulam, particle board, plywood.	construction material	Material and energy recovery of construction material included. Use phase also included. 0.611 tons dry matter biomass/m^2 floor area.		I-VIII	251 kg CO_2-eq./m^2 floor area
Lindholm et al. (2010)	Logging residue - Norway spruce	Chips			II-III	1.5-3.5 g CO_2-eq./MJ_{chips}
	Stumps	Chips			II-III	2.6-3.1 g CO_2-eq./MJ_{chips}
Modahl and Vold (2010)	Spruce	Cellulose	Biorefinery. Allocation: mass. Incl. transport to customer, 100 km. Incl. infrastructure of biorefinery and all operations.	FU= 1 ton cellulose	I-III	1160 kg CO_2-eq./ton DM
		Ethanol 96 %		FU= 1 m^3 ethanol	I-III	324 kg CO_2-eq./m^3 100 % ethanol
		Ethanol 99 %		FU= 1 m^3 ethanol	I-III	666 kg CO_2-eq./m^3 100 % ethanol
		Lignin (liquid)		FU= 1 ton lignin.	I-III	666 kg CO_2-eq./ton DM
		Lignin (powder)		FU= 1 ton lignin	I-III	1120 kg CO_2-eq./ton DM
		Vanillin		FU= 1 ton vanillin	I-III	1090 kg CO_2-eq./ton DM

Study	Input	Product	Description of system	System	GWP	Climate mitigation comparison
Pyörälä et al. (2012)	Norway spruce, pine, birch	Heat from wood stoves compared to coal	Carbon flux in forest included. 80 years rotation length, medium fertile ground. Thinning is based in Finnish recommendations. Carbon footprint varies with different management regimes. Depending on the management, varies from 80-191 kg CO_2-eq./MWh. Reference: coal (341 kg CO_2/MWh).	I-V, VIII	185 kg CO_2-eq./MWh	**(+)** 156 kg CO_2-eq./MWh
		As above	Carbon flux in forest included. 80 years rotation length, fertile ground. Thinning is based in Finnish recommendations. Carbon footprint varies with different management regimes. Depending on the management, varies from 49-242 kg CO_2-eq./MWh. Reference: coal (341 kg CO_2/MWh).	I-V, VIII	242 kg CO_2-eq./MWh	**(+)** 99 kg CO_2-eq./MWh
González-García et al. (2011)	Pine (20 %) and Spruce (80 %)	Pulp	FU= 1 air dried ton pulp (10 % water). Byproducts: ethanol (59.52 kg/ton cellulose) and lignosulfonates (23.81 kg/ton cellulose). Allocation: economic - price per ton. Pulp: 94.7 %, ethanol: 4 %, lignosulfonate: 1.3 %. Integrated biorefinery.	I-III	415 kg CO_2-eq./AD ton pulp	
Guest et al. (2011)	Forest residue and sawmill residue	CHP - 0.1 MW electricity	FU= 1 MJ electricity delivered to customer. Allocation: exergy (0.69)	I-IV, VI	11 g CO_2-eq./MJ	
		CHP - 0.1 MW heat	FU= 1 MJ heat delivered to customer. Allocation: exergy (0.31)	I-IV, VI	2.4 g CO_2-eq./MJ	
		CHP - 1 MW electricity	FU= 1 MJ electricity delivered to customer. Allocation: exergy (0.72)	I-IV, VI	8.8 g CO_2-eq./MJ	
		CHP - 1 MW heat	FU= 1 MJ heat delivered to customer. Allocation: exergy (0.28)	I-IV, VI	2.4 g CO_2-eq./MJ	
		CHP - 50 MW electricity	FU= 1 MJ electricity delivered to customer. Allocation: exergy (0.78)	I-IV, VI	10 g CO_2-eq./MJ	

		CHP - 50 MW heat	FU= 1 MJ heat delivered to customer. Allocation: exergy (0.22).	I-IV, VI	2.8 g CO_2-eq./MJ	
Judl et al. (2011)	Birch (Betula pendula, Betula pubescens)	Pulp	FU= 1 air dried ton pulp delivered at paper factory gate. Credited for byproduct: electricity. Input wood per FU 5.2 m^3	I-III	441 kg CO_2-eq./AD ton pulp	
Valente et al. (2011)	Mountain forest - birch	Energy wood	Total GWP= 16907.818 g CO2-eq./m^3 s.o.b. 30 % of emissions allocated to energy, and 70 % allocated to stem wood.	I-III	5072 g CO_2-eq./m^3 s.o.b.	
		Stem wood		I-III	11835 g CO_2-eq./s.o.b m^3	
Buonocore et al. (2012)	Timber + short rotation willow	CHP: 24 MW electricity	Willow plantations, sludge from septic tanks spread across plantation, harvest 3-5 year, main load in CHP: conventional willow, forest and wood industry residue, agricultural waste. Willow from "sludge-plantations": 15-20 % of total biomass combusted.	I-IV	0.00011 g CO_2-eq./$J_{electricity}$	
	Timber + short rotation willow	CHP: 45 MW heat	Willow plantations, sludge from septic tanks spread across plantation, harvest 3-5 year, main load in CHP: conventional willow, forest and wood industry residue, agricultural waste. Willow from "sludge-plantations": 15-20 % of total biomass combusted.	I-IV	0.00021 g CO_2-eq./J_{heat}	
González-García et al. (2012a)	Short-rotation coppice (willow)	Bioethanol (enzymatic)	Byproducts: electricity and gypsum. Allocation: economic. Net energy balance: 264.6 GJ/FU. FU: use of biomass chips coming from 1 ha of short rotation coppice willow (Salix spp.). Production of chips on an oven dry basis was 10 tons/ha/year over 20 time frame. Production of willow included (uptake of carbon). Reference: conventional petrol.	I-III, V	34.64 ton CO_2-eq. /ha SRC willow	**(+)** 79 %

Study	Input	Product	Description of system	System	GWP	Climate mitigation comparison
González-García et al. (2012a)	Short-rotation coppice (willow)	Electricity (by gasification)	Net energy balance: 102 GJ/FU.FU: use of biomass chips coming from 1 ha of short rotation coppice willow (Salix spp.). Production of chips on an oven dry basis was 10 ton/ha/year over 20 time frame. Production of willow included (uptake of carbon). Reference: Swedish electricity mix.	I-III, V	30.6 ton CO_2-eq. /ha SRC willow	**(÷)** ~50 %
			Reference: EU electricity mix	I-III, V		**(+)** >80 %
			Reference: Bioethanol (from same system)	I-III, V		**(+)** 4 ton CO_2-eq./ha
González-García et al. (2012b)	Short-rotation coppice (willow) - fertilized	Willow chips	Energy crop production 2 771 208 MJ/ha. GWP includes uptake of carbon in growing biomass. Yield: 6.7 ton/ha/year.	I, VIII	- 324 kg CO_2-eq./ha	
	Short-rotation coppice (willow) - non-fertilized		Energy crop production 1 697 652 MJ/ha. GWP includes uptake of carbon in growing biomass. Yield: 4 ton/ha/year.	I VIII	- 207.01 kg CO_2-eq./ha	
Ghose and Chinga-Carrasco (2013)		Newsprint	FU: 1 ton. Input per ton paper: 0.83 ton TMP, 0.0045 ton kraft pulp, 0.0672 ton fillers, 0.4 ton recovered paper. Elmix: Nordel	II-III	512 kg CO_2-eq./ton	
		Super calendared paper	FU: 1 ton. Input per ton paper: 0.56 ton TMP, 0.1 ton kraft pulp, 0.34 ton fillers. Elmix: Nordel	II-III	626 kg CO_2-eq./ton	
Holma et al. (2013)	Forest residue	Biodiesel (FTD) - stand-alone FTD plant	Changes in soil carbon not included. Reference: fossil fuel	I-III	12g CO_2-eq./MJ	**(+)** >80 %
			Soil carbon included.	I-III	50 g CO_2-eq./MJ	**(+)**

			Reference: fossil fuel			40 %
Cherubini and Strømman (2013)	Forest biomass	Heat (district heating)	Biogenic carbon and albedo included. Reference: heat by natural gas (73.1 g CO_2-eq./MJ)			**(+)**
Jäppinen et al. (2014)	Harvesting residues	Wood supplied to various energy plants/systems	Heating value: 7.49 GJ/m^3 solid. Processing of raw materials includes only chipping (i.e. supply chain emissions).	I-III	2.4 g CO_2-eq./MJ	**(+)**
			Reference: EU fossil comparator (when energy produced).			>90%
	Small-diameter energy wood		Heating value: 7.63 GJ/m^3 solid. Processing of raw materials includes only chipping (i.e. supply chain emissions).	I-III	3 g CO_2-eq./MJ	**(+)**
			Reference: EU fossil comparator (when energy produced).			>90%
	Stumps		Heating value: 7.67 GJ/m^3 solid. Processing of raw materials includes only chipping (i.e. supply chain emissions).	I-III	2.6 g CO_2-eq./MJ	**(+)**
			Reference: EU fossil comparator (when energy produced).			>90%

Table 2 Qualitative assessment of environmental impacts other than GWP of wood products relative to a reference system. (+) indicates that the wood based system performed better than the reference system. (÷) indicates that the wood-based system performed worse than the reference system. Eight System boundaries are used: I Production of raw material, II Processing of raw material, III Transport, IV Infrastructure for use, V Operation/use, VI end of life, VII Substitution, VIII Forest carbon. AP=Acidification potential, EP=Eutrophication potential, POFP=Photochemical oxidant formation potential, ODP=Ozone depletion potential, ADP=Abiotic depletion potential, CED=Cumulative energy demand, HTP=Human toxicity potential

Study	Input	Product	Comments	System boundaries	AP	EP	POFP	ODP	ADP	CED	HTP
Bright et al. (2010)	Spruce	Ethanol E85 - biochemical	Reference: Diesel	I-V	(+)	(÷)					(+)
		Ethanol E85 - thermochemical	Reference: Diesel	I-V	(+)	(+)					(+)
González-García et al. (2012a)	Short-rotation coppice (willow)	Bioethanol (enzymatic)	Reference: Petrol	I-III, V	(+)	(÷)	(+)	(+)	(+)	(+)	
		Electricity (by gasification)	Reference: Swedish el-mix	I-III, V	(÷)	(÷)	(÷)	(÷)	(÷)	(+)	
			Reference: EU el-mix	I-III, V	(+)	(+)	(+)	(+)	(+)	(+)	
			Reference: Bioethanol (enzymatic)	I-III, V	(+)	(÷)	(+)	(÷)	(÷)	(÷)	
Brekke et al. (2008)	Chips based on forest biomass (+) residues 50 % of available biomass harvested	CHP - 1 MW facility	Reference: Nordic el-mix	II-VI	(+)	(÷)	(+)				
		CHP - 10 MW facility		II-VI	(+)	(+)	(+)				
	Chips based on forest biomass (+) residues 100 %	CHP - 1 MW facility		II-VI	(+)	(÷)	(+)				
		CHP - 10 MW facility		II-VI	(+)	(+)	(+)				

	of available biomass harvested				
Gustavsson et al. (2006)	Spruce and pine	4-story building, 16 apts. 1190 m^2 floor area	Electricity production for material production: coal. Reference: concrete	I-IV, VI-VIII	**(+)**
			Electricity production for material production: natural gas. Reference: concrete	I-IV, VI-VIII	**(+)**
		4-story building, 21 apts. 1175 m^2 floor area	Electricity production for material production: coal. Reference: concrete	I-IV, VI-VIII	**(+)**
			Electricity production for material production: natural gas. Reference: concrete	I-IV, VI-VIII	**(+)**

Table 3: Carbon footprint of wood products recalculated to a common unit, kg CO_2-eq./m^3 s.o.b. Main assumptions important for the recalculation are shown in the last column. Eight System boundaries are used: I Production of raw material, II Processing of raw material, III Transport, IV Infrastructure for use, V Operation/use, VI end of life, VII Substitution, VIII Forest carbon. The studies are listed chronologically within the forest product categories: Pulp & paper, Energy and Stem wood and construction material.

Study	Input	Product	System bound.	Emission kg CO_2-eq./m^3 s.o.b.	Assumptions
Pulp & paper					
González-García et al. (2009a)	Spruce and pine	Solid wood for pulp production	I, III	36	
Judl et al. (2011)	Birch	Pulp	I-III	85	Input wood per ton pulp: 5.2 m^3
Energy					
Raymer (2006)	Pine fuel wood	Heat - dwellings	I-III, V	153	1m^3 chopped fire wood = 0.4 m^3 s.o.b
	Birch fuel wood	Heat - dwellings	I-III, V	183	1m^3 chopped fire wood = 0.4 m^3 s.o.b
Brekke et al. (2008)	Chips based on forest biomass + residues 50 % of available biomass harvested	CHP - 1 MW facility	II-VI	25	20 GWh/11 691 m^3= 1 710 kWh/m^3 s.o.b.
	Chips based on forest biomass + residues 100 % of available biomass harvested	CHP - 1 MW facility	II-VI	21	20 GWh/11 691 m^3= 1 710 kWh/m^3 s.o.b.
	Chips based on forest biomass + residues 50 % of available biomass harvested	CHP - 10 MW facility	II-VI	20	160 GWh/93 528 m^3= 1 710 kWh/m^3 s.o.b.
	Chips based on forest biomass + residues 100 % of available biomass harvested	CHP - 10 MW facility	II-VI	17	160 GWh/93 528 m^3= 1 710 kWh/m^3 s.o.b.
Hagberg et al. (2009)	Roundwood chips	Pellets - heating plant 100 MW	I-III, V	27	19.3 MJ/kg, 390 kg/m^3
Solli et al. (2009)	Birch fuel wood	Heat, household – old stove	I-V	292	LHV: 2650 kWh/m3
		Heat, household – new stove	I-V	204	LHV: 2650 kWh/m3

Study	Input	Product	System bound.	Emission kg CO_2-eq./m^3 s.o.b.	Assumptions
Pyörälä et al. (2012)	Norway spruce, pine, birch	Heat from wood stoves	I-V, VIII	252	Energy content: 3.24 MWh/ton 420 kg/m^3 s.o.b. 1.36 MWh/m^3 s.o.b.
			I-V, VIII	329	Energy content: 3.24 MWh/ton 420 kg/m^3 s.o.b. 1.36 MWh/m^3 s.o.b.
Valente et al. (2011)	Mountain forest - birch	Energy wood	I-III	5	Total GWP= 16 907.818 g CO_2-eq./m^3 s.o.b. 30 % of emissions allocated to energy wood
González-García et al. (2012a)	Short-rotation coppice (willow)	Bioethanol (enzymatic)	I-III, V	509	6.8 m^3/ton (Henriksson & Neumeister 2013)
		Electricity (by gasification)	I-III, V	450	6.8 m^3/ton (Henriksson & Neumeister 2013)
González-García et al. (2012b)	Short-rotation coppice (willow) - fertilized	Willow chips	I	-7	6.8 m^3/ton (Henriksson & Neumeister 2013)
	Short-rotation coppice (willow) - non-fertilized		I	-8	6.8 m^3/ton (Henriksson & Neumeister 2013)
Jäppinen et al. (2014)	Small-diameter energy wood		I-III	23	Heating value: 7.63 GJ/m^3 solid
Stem wood and construction material					
Gustavsson et al. (2006)	Spruce and pine	4-story wood building, 16 apts. 1190 m^2 floor area compared to concrete building	I-IV, VI-VIII	-22	0.611 tonnes dry matter biomass/m^2 floor area
				-7	0.611 tonnes dry matter biomass/m^2 floor area
		4-story wood building, 21 apts. 1175 m^2 floor area compared to concrete building	I-IV, VI-VIII	-42	0.611 tonnes dry matter biomass/m^2 floor area
				-16	0.611 tonnes dry matter biomass/m^2 floor area

Study	Input	Product	System bound.	Emission kg CO_2-eq./m^3 s.o.b.	Assumptions
Michelsen et al. (2008)		Roundwood logs	I, III	25	
Wærp et al. (2009)	Spruce and pine	Sawn wood	I-III	48	60 %by-products (Langerud et al. 2007)
Wærp et al. (2009)	Spruce and pine	Planed timber	I-VII	76	5 % cuttings
		Gluelam	I-VII	243	Input sawn timber: 493 kg/m^3 Density: 400 kg/m^3 s.o.b.
Gustavsson et al. (2010)	Lumber (spruce, pine, oak), glulam, particle board, plywood.	Construction material	I-VIII	160	0.611 tonnes dry matter biomass/m^2 floor area
Valente et al. (2011)	Mountain forest - birch	Stem wood	I-III	12	Total GWP= 16 907.818 g CO_2-eq./m^3 s.o.b. 70 % of emissions allocated to stem wood

3.2 Costs

As mentioned in Chapter 1, costs are strong indications of use of scarce resources and are important in judging the possibilities for implementing a given production. In order for wood to be an effective replacement measure in climate change mitigation, the market costs have to be in the range of complementary products when environmental improvements alone are not sufficient justification for replacement. If the environmental benefits are especially large and not reflected by market prices, use of biomass can be increased through political incentives like subsidies or taxes (Milazzo et al., 2013), influencing the market costs.

When assessing cost of biomass use, there are two different approaches that are commonly used: cost of production and abatement cost. Depending on the system boundaries, cost of production include investment cost, biomass procurement cost, transportation cost, and cost of operation and maintenance of plant or mill. Abatement cost is the additional cost that is associated with reducing the emissions of GHG by shifting from one production system to an alternative system. Often, we use the term marginal (or average) abatement cost, which is the abatement cost divided by the corresponding marginal or average reduction in GHG emissions caused by the change of production systems (Hennig and Gawor, 2012; Sterner and Fritsche, 2011).

According to Petersen and Solberg (2005), wood as building material is competitive on price compared to fossil-based alternatives, while Dornburg and Faaij (2005) claim that use of biomass in general is more expensive than the non-renewable alternatives. Biorefinery produces several products that compete in different markets, and this can stabilize the economy of the company (Rødsrud et al., 2012). Compared with the pulp and paper industry, Borregaard biorefinery has a more stable and higher return on capital employed (Rødsrud et al., 2012).

Transport is often the most costly stage in analyses of energy systems based on woody biomass (Valente et al. 2011; Michelsen et al. 2008). Michelsen et al. (2008) reported a total cost of 327 NOK[1]/m^3 delivered to user, while Valente et al. found a total cost of 463 NOK/m^3 for mountain birch (Table 4).

Production of bioenergy depends on the price of competing energy sources (Trømborg et al., 2013). In Norway, the main energy source is electricity, and low price of electricity has led to an energy market based on hydropower, whereas the standing stock of forest biomass increases. Solli et al. (2009) found that the price of heat produced by conventional household stoves using birch fuelwood costs between 0.5-0.7 NOK/kWh, depending on the efficiency of the stove. Modelling of the bio heat market in Norway shows that if price of energy is below 550 NOK/MWh (excl. VAT) district heating based on forest fuel is not profitable, and in order for the market to invest in conversion from oil boilers to wood based boilers, the price needs to be higher than 600 NOK/MWh excl. VAT (Trømborg et al., 2013). For comparison, in 2013 the price of electricity was 350 NOK/MWh excl. VAT for households in Norway (Statistics Norway, 2014).

There will be a trade-off between economy of scale and biomass procurement at nearly any wood processing plant, regardless of type of production. Total processing costs consists of

[1] 1 NOK=0.123 €

some parts that will increases as the size of the plant increases, like transportation of raw material, while other costs will decrease as the size increases, like operating cost and capital cost (Searcy and Flynn, 2009). At some point, the increase in biomass procurement cost will be higher than the decrease in capital and processing cost, and optimum plant size is below this point (Searcy and Flynn, 2009). When it comes to energy production, most often there is also a trade-off between higher efficiency of larger plants and lower infrastructure costs in smaller plants (Persson et al., 2006).

Wood-based productions compete over the same natural resource – forest fiber. If the production of bioenergy increases, the price of woody biomass will increase, all other factors equal. Rather few studies exist on quantifying such price impacts. Trømborg et al. (2013) found for Norway that if the energy price increased by 40 % (from 500 to 700 NOK/MWh), the biomass harvest would increase by 18 % and, at the same time, the price of pulpwood would increase by 20-30 %. The modelled increase in roundwood price can be counteracted by increasing the use of harvest residues (Trømborg et al., 2013).

There are large variations in abatement cost, depending on feedstock, energy conversion technology and which type of fossil fuel is replaced (Table 4). Negative values mean that the bioenergy has lower cost than the fossil reference (Sterner and Fritsche, 2011). Among the reviewed studies, heat generation in Austria by wood chips boiler has the lowest abatement cost, namely negative 45 €/ton CO_2-equivalents (Kalt and Kranzl, 2011). The highest abatement costs are associated with power generation in Germany, reporting costs as high as 400 €/ton CO_2-equivalents (Hennig and Gawor, 2012).

Table 4: Cost and abatement cost of various bioenergy feedstock and energy production types

Study	Region	Feedstock/production types	Cost (€/MWh)	Abatement cost (€/ton CO_2-eq.)	Substitution
Wahlund et al. (2004)	Sweden	Production of pellets		15-30[2]	Coal
		Wood used for electricity production		15[2]	Coal
		Wood for production of transportation fuel		5-86[2]	Fossil fuel
Solli et al. (2009)	Norway	Heat, household – old stove	86[3]		
		Heat, household – new stove	62[3]		
Gustavsson et al. (2011)	Sweden	Logging residue	12-15		
Kalt and Kranzl (2011)	Austria	Heat generation	48-135	-45-93	Oil boilers, gas-fired heating plants
		CHP	69-311	5-201	Gas turbines
		Transportation fuel	80-110	47-240	Fossil fuel
Sterner and Fritsche (2011)	Germany	Wood chips for transportation fuel		75-560	Gasoline and diesel
		Wood chips and pellets for heating		130-155	Fuel oil and natural gas
		Wood residue for power		20-380	Coal and natural gas
Hennig and Gawor (2012)	Germany	Forest residue and waste wood for power generation	120-177	240-400	Coal
Bergseng et al. (2013)	Norway	Logging residues for heat	44		
Bertrand et al. (2014)	EU	Torrified pellets from wood residues	30-32		Coal price: 11 €/kWh
		Wood pellets	25-31		
		Wood chips	13-27		
Gerssen-Gondelach et al. (2014)	Global	Wood chips and pellets - heat		61[2]	Natural gas and coal
		Wood chips and pellets – power co-fire		63[2]	Natural gas and coal

4. Discussion and conclusion

In this chapter we discuss in a more overall context the results presented from the reviewed studies. Two main questions are whether wood used for substitution can contribute to climate change mitigation, and what is the best use of the wood to reach that goal? Even though wood is a renewable resource, it is limited in availability. Hence, from a climate mitigation point of

[2] 2005 US $=0.787 €

[3] 1 NOK=0.123 €

view it should be used where it can produce the greatest GWP benefits (Koukkari and Nors, 2009). The reviewed studies show that woody biomass can contribute to climate change mitigation as most of the studies found that the GWP of forest biomass value chains was lower than the non-renewable alternatives compared with. This agrees with the findings of Eriksson et al. (2007), who studied the net emissions of carbon from substitution of different non-wood products, and found the largest reductions when biomass was used for construction material. This implies that assortments of wood that economically can be used for construction material, should primarily be used in construction, and only after that period be used for energy or pulpwood (Eriksson et al., 2007; Gustavsson et al., 2006).

The reviewed studies show that for other LCA-impact categories like eutrophication and acidification, results are mixed, as some show benefits and other find disadvantages with forest-based systems. In such cases, it will be a tradeoff between benefits in global issues, like climate change, and disadvantages in local issues like waste handling, acidification and eutrophication. In addition to climate change mitigation contribution, forest provides other services, like erosion protection, recreation and harvesting of non-wood products like game, mushrooms and berries. Moreover, it is home of many species and important for conservation of biological diversity. All these factors have to be considered in complete environmental and economic impact analyses.

There is a growing understanding that the carbon stored in products is significant, and this can now be included in nations' carbon accounting (Marland et al., 2010; Norwegian Ministry of Environment, 2012). If we use carbon storage in the forest or in products as a climate change mitigation measure, these storages must be kept constant over time or increased, otherwise they will lead to net emissions of the stored carbon (Gustavsson et al., 2010). Some have pointed out that the stock will eventually reach saturation, and storage in these compartments is a finite alternative (Gustavsson et al., 2010; Smith et al., 2014, p. 29). Others find that the carbon sequestration in soil and dead organic material continues, even though the biomass increment is small (Smith et al., 2014, p. 29).

Substitution of carbon- and/or energy intensive products is seen as a continuous option for atmospheric carbon reduction in the reviewed studies. Both quality and quantity of biomass available for substitution depend upon the forest management (Eriksson et al., 2007). Today's management is aimed at timber production because sawn wood gives the highest profit for the forest owners (Pyörälä et al., 2012). By increasing the rotation length, timber production, and, thus, the substitution potential may increase, while shorter rotation length may increase the production of energy wood. However, new sawmill technologies seem to give preferences for medium-sized diameter saw logs, and the use of glulam beams makes possible utilizing also small-diameter logs for construction wood.

Fertilization can also increase the production of both timber and energy wood (Pyörälä et al., 2012). Wihersaari (2005) found that when the nitrogen loss was compensated for by fertilizing, the emissions of GHG increased with 7 kg CO_2-eq./MWh_{chips}, in particular NOx in the forests and the emission of GHGs during fertilization production. The study shows the importance of including the net impact of fertilization in LCAs.

Use of biomass generally demands more land resources than the fossil alternative (Dornburg and Faaij, 2005). This may create conflicts with environmental services like recreation, biodiversity and water catchment. Very few of the reviewed studies discuss this topic. Such conflicts can be resolved through appropriate policy means or forest certification schemes.

The environmental profile of wood products can be improved through better utilization of the resources e.g. through cascading or use of biorefineries. Cascading refers to recycling of products that leads to a quality degradation in each round of recycling (Kim et al., 1997). Even though the potential use of the biomass are more limited as the quality degrades (Rivela et al., 2006), cascading of biomass reduces the need for virgin material, energy and land resources (Dornburg and Faaij, 2005). When biomass is recycled several times and finally burned, wood fibers can each time substitute for other, more energy demanding products or fossil fuels, thus, contributing to even more climate change mitigation than without cascading. In addition, the emissions of the carbon in the products are postponed (Dornburg and Faaij, 2005; Sikkema et al., 2013). In spite of these advantages, we have found very few studies where cascading has been applied, except where sawn wood, fiberboards or paper has been burned for fossil fuel substitution. For example, it would have been interesting to see analyses where sawn wood has been used for construction, then particleboards or fiberboards before being burned.

Even though the LCA methodology is standardized, there are large degrees of freedom when performing an LCA. In the process of data collection and analysis the LCA-practitioner has to make some assumptions that are subjectively founded regarding for example allocation of environmental burden, electricity mix, system boundaries and end-of-life treatment. These assumptions can have significant influence on the results (Werner et al., 2007). In addition, the LCAs show great variations regarding choice of FU, some related to input and some related to output. One should therefore be careful in comparing LCA results. However, they provide important information when it comes to environmental performance. The prime issue here is that the main assumptions underlying the results are clearly stated and understood.

The results in this review are geographically limited to the Nordic region, except for some of the reviewed studies on costs. Geographical characteristics like species composition, harvesting techniques and forest management regimes produce significant differences in LCA of forest value chains (see for example González-García et al., 2014; Jäppinen et al., 2014) and it is important to consider this when comparing LCA results.

Most of the reviewed studies assume that the biogenic CO_2 from biomass is climate neutral. If the biogenic CO_2 is included, the results will change. Cherubini and Strømman (2013) analyzed a future energy scenario where they compared district heating (DH) based on biomass with a system based on natural gas. They included biogenic CO_2 and albedo, and found a stronger climate change mitigation by DH based on biomass compared to DH based on natural gas. The study illustrates the importance of including albedo in LCAs dealing with climate mitigation in boreal forests.

Just like reported in Petersen & Solberg (2005), very few LCA studies include costs or cost efficiency estimates, or empirical analysis or reasoning about substitution possibilities in actual markets. However, LCA data of wood products are increasingly being used in forest sector modeling where economics play an important role in getting more realistic analyses of the climate mitigation impacts of changes in forest management and policy implementations - for example for Norway in Hoen and Solberg (1994); Raymer et al., (2011); Raymer et al., (2009); Sjølie et al. (2014); Soldal et al. (2014).

Also in line with previous reviews on LCAs of wood bioenergy (Cherubini and Stromman, 2011; Petersen and Solberg, 2005, Klein et al., 2015) we find that wood can have rather strong climate mitigation impacts, but also that the studies vary greatly with respect to system boundaries, applied technologies, geographical locations, and choice of FU. These factors make it rather difficult to compare LCA results even if the technologies compared may look similar.

Carbon leakage (i.e. that changed GHG emission in one place leads to changed GHG emission another place) is of considerable interest in analyzing climate mitigation strategies in the forest sector, as international trade is rather important in this sector. Surprisingly, few LCA studies includes carbon leakage considerations. Also, few of the reviewed studies have included impacts of the actual climate change over time.

Future research should look at how the LCA methodology could be made uniform for easier comparisons at the same time as analysis should be based on local data as far as possible. The transparency of data collection and assumptions should be increased in order to facilitate comparisons. Potential tradeoffs between benefits in global issues and disadvantages in local issues like waste handling, acidification and eutrophication are important to document. Attention should be given to better include the spatial and timing elements, costs and cost efficiency estimates, wood cascading, carbon leakage, climate change impacts on forest net carbon sequestration, and how harvest influences carbon storage in the soil. Further, more research should be done on the optimal forest management, harvesting and use of wood in a climate change mitigation regime.

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Appendix

Table A1: Quantitatively assessment of environmental impacts of wood based systems. Acidification potential (AP), toxicity potential (TP), eutrophication potential (EP), land competition (LC), photochemical oxidation formation potential (POFP), ozone depletion potential (OD), abiotic depletion potential (ADP) and cumulative energy demand (CED). System boundaries: I Production of raw material, II Processing of raw material, III Transport, IV Infrastructure for use, V Operation/use, VI end of life, VII Substitution, VIII Forest carbon.

Study	Input	Product	System boundary	AP	TP	EP	LC	POFP	ODP	ADP	CED
(Michelsen et al., 2008)		Roundwood logs	I, III	111 g SO_2-eq/m³		24 g PO_4-eq/m³		76 g C_2H_2-eq/m³			
(González-García et al., 2009a)	Spruce and pine	Solid wood for pulp production	I, III	10.2 mol H+-eq/m³ s.o.b		2.61 kg O_2-eq/m³ s.o.b					370.2 MJ/m³ s.o.b
(Solli et al., 2009)	Birch fuel wood	Heat, household – old stove	I-V	0.52 g SO_2-eq/kWh		0.1 g PO_4-eq/kWh		5.5 g C_2H_2-eq/kWh			
		Heat, household – new stove		0.37 g SO_2-eq/kWh		0.072 g PO_4-eq/kWh		3 g C_2H_2-eq/kWh			
(Wærp et al., 2009)	Spruce and pine	Sawn wood	I-II	267 g SO_2-eq/m³		41 g PO_4-eq/m³		24 g C_2H_4-eq/m³	$4{,}2 \times 10^{-4}$ g R11-eq/m³		1709 MJ/m³
		Planned timber	I-VII	344 g SO_2-eq/m³		0.054 kg PO_4-eq/m³		30 g C_2H_4-eq/m³	5.2×10^{-4} g R11-eq/m³		1964 MJ/m³
		Wood paneling	I-VII	5 g SO_2-eq/m²		1 g PO_4-eq/m²		0.4 C_2H_4-eq/m²	1.1×10^{-5} g R11-eq/m²		32 MJ/m²
		Planned timber, steeped with copper	I-VII	538 g SO_2-eq/m³		66 g PO_4-eq/m³		41 g C_2H_4-eq/m³	1.6×10^{-3} g R11-eq/m³		2403 MJ/m³
		Wood paneling, treated with paint	I-VII	40 g SO_2-eq/m²		6 g PO_4-eq/m²		8 g C_2H_4-eq/m²	1.7×10^{-5} g R11-eq/m²		166 MJ/m²
		Massive beams	I-VII	670 g SO_2-eq/m³		110 g PO_4-eq/m³		66 g C_2H_4-eq/m³	2.8×10^{-3} g R11-eq/m³		5176 MJ/m³

		Gluelam	I-VII	770 g SO_2-eq/m^3		120 g PO_4-eq/m^3		75 g C_2H_4-eq/m^3	1.7×10^{-3} g R11-eq/m^3		5144 MJ/m^3
(Bright and Strømman, 2009)	Mainly spruce.	Ethanol E85 - best case biochemical	I-V	446 mg SO_2-eq/km		59.5 mg PO_4-eq/km					
		Ethanol E85 - best case thermochemical		371 mg SO_2-eq/km		53.2 mg PO_4-eq/km					
(Lindholm et al., 2010)	Logging residue - Norway spruce	Chips	II-III	6.6-16 mg SO_2-eq/MJ_{chips}		1.2-2.8 mg PO_4^{-3}-eq/MJ_{chips}					
	Stumps	Chips		12-14 mg SO_2-eq/MJ_{chips}		2.1-2.5 mg PO_4^{-3}-eq/MJ_{chips}					
(Modahl and Vold, 2010)	Spruce	Cellulose	I-III	10.6 kg SO_2-eq/tons DM		3.56 kg PO_4^{-3}-eq/tons DM		770 g C_2H_4-eq/tons DM	9.3×10^{-2} g CFC11-eq/tons DM		34347 MJ LHV/tons DM
		ethanol 96 %		4.5 kg SO_2-eq/m^3 100% ethanol		2.17 kg PO_4^{-3}-eq/m^3 100% ethanol		290 g C_2H_4-eq/m^3 100% ethanol	2.6×10^{-2} g CFC11-eq/m^3 100% ethanol		9085 MJ LHV/m^3 100% ethanol
		ethanol 99 %		7.9 kg SO_2-eq/m^3 100% ethanol		2.68 kg PO_4^{-3}-eq/m^3 100% ethanol		490 kg C_2H_4-eq/m^3 100% ethanol	5.1×10^{-2} g CFC11-eq/m^3 100% ethanol		19080 MJ LHV/m^3 100% ethanol
		Lignin (liquid)		7.9 kg SO_2-eq/tons DM		3.04kg PO_4^{-3}-eq/tons DM		500 g C_2H_4-eq/tons DM	4.9×10^{-2} g CFC11-eq/tons DM		18403 MJ LHV/tons DM
		Lignin (powder)		10.8 kg SO_2-eq/tons DM		5.14 kg PO_4^{-3}-eq/tons DM		780 g C_2H_4-eq/tons DM	0.11 g CFC11-eq/tons DM		30657 MJ LHV/tons DM
		Vanillin		10.5 kg SO_2-eq/tons DM		3.12 kg PO_4^{-3}-eq/tons DM		750 g C_2H_4-eq/tons DM	8.9×10^{-2} g CFC11-eq/tons DM		32200 MJ LHV/tons DM
(González-García et al., 2011)	Pine (20 %) and Spruce (80 %)	Pulp	I-III	5.55 kg SO_2-eq/ton pulp		1.74 kg PO_4^{-3}-eq/ton pulp		170 g C_2H_2-eq/ton pulp	6.22×10^{-2} g CFC-11-eg/ton pulp		

(Guest et al., 2011)	Forest residue	CHP - 0.1 MW electricity	I-IV	0.27 g SO_2-eq/MJ	1706.4 g 1,4-DB-eq/MJ	0.07 g PO_4-eq/MJ		0.013 g C_2H_2-eq/MJ	1.9×10^{-7} g CFC-11-eq/MJ	0.03 g Sb-eq/MJ	
		CHP - 0.1 MW heat		0.06 g SO_2-eq/MJ	381.31 g 1,4-DB-eq/MJ	0.015 g PO_4-eq/MJ		2.9×10^{-3} g C_2H_2-eq/MJ	0.015 g CFC-11-eq/MJ	8.1×10^{-3} g Sb-eq/MJ	
		CHP - 1 MW electricity		0.12 g SO_2-eq/MJ	1405.34 g 1,4-DB-eq/MJ	0.03 g PO_4-eq/MJ		7.9×10^{-3} g C_2H_2-eq/MJ	1.5×10^{-7} g CFC-11-eq/MJ	0.025 g Sb-eq/MJ	
		CHP - 1 MW heat		0.027 g SO_2-eq/MJ	351 g 1,4-DB-eq/MJ	6.9×10^{-3} g PO_4-eq/MJ		1.8×10^{-3} g C_2H_2-eq/MJ	6.9×10^{-3} g CFC-11-eq/MJ	0.012 g Sb-eq/MJ	
		CHP - 50 MW electricity		0.056 g SO_2-eq/MJ	1505.36 g 1,4-DB-eq/MJ	0.014 g PO_4-eq/MJ		5.1×10^{-3} g C_2H_2-eq/MJ	2.5×10^{-7} g CFC-11-eq/MJ	0.041 g Sb-eq/MJ	
		CHP - 50 MW heat		0.014 g SO_2-eq/MJ	381 g 1,4-DB-eq/MJ	0.0033 g PO_4-eq/MJ		1.2×10^{-3} g C_2H_2-eq/MJ	3.3×10^{-3} g CFC-11-eq/MJ	0.017 g Sb-eq/MJ	
(González-García et al., 2012a)	Short-rotation coppice (willow)	Bioethanol (enzymatic)	I-III, V	219.7 kg SO_2-eq/ha		92.6 kg PO_4^{-3}-eq/ha	503.6 m^2a/ha	50.6 kg C_2H_4-eq/ha	2.4 g CFC-11-eq/ha	130.1 kg Sb eq/ha	292.7 GJ/ha
	Short-rotation coppice (willow)	Electricity (by gasification)		186.3 kg SO_2-eq/ha		116.9 kg PO_4^{-3}-eq/ha	1807 m^2/ha	5.7 kg C_2H_4-eq/ha	4.3 g CFC-11-eq/ha	258.3 kg Sb eq/ha	1171.5 GJ/ha
(González-García et al., 2012b)	Short-rotation coppice (willow) - fertilized	Willow chips	I	73.3 kg SO_2-eq/ha	2591.3 t 1,4-DB-eq/ha	159.5 kg PO_4^{-3}-eq/ha		1.2 kg C_2H_4-eq/ha	1.02 g CFC-11-eq/ha	52.5 kg Sb eq/ha	8777 MJ/ha
	Short-rotation coppice (willow) - non-fertilized			20.9 kg SO_2-eq/ha	624 t 1,4-DB-eq/ha	5.9 kg PO_4^{-3}-eq/ha		520 g C_2H_4-eq/ha	0.578 g CFC-11-eq/ha	20.9 kg Sb eq/ha	49552 MJ/ha
(Buonocore et al., 2012)	waste from agriculture, forest industry and logging + short rotation willow	Electricity (size: 24 MW of electricity)	I-III, IV	1.6×10^{-7} g SO_2-eq/$J_{electricity}$	3.2×10^{-7} g 1.4-DB-eq/$J_{electricity}$	3.6×10^{-8} g PO_4-eq/$J_{electricity}$		9×10^{-8} g C_2H_4-eq/$J_{electricity}$			

	waste from agriculture, forest industry and logging + short rotation willow	Heat (size: 45 MW of heat)		3.1×10^{-7} g SO_2-eq/J_{heat}	6.3×10^{-7} g 1.4-DB-eq/J_{heat}	7.1×10^{-8} g PO_4-eq/J_{heat}		1.8×10^{-7} g C_2H_4-eq/J_{heat}			
(Ghose and Chinga-Carrasco, 2013)		Newsprint	II-III	3.39 kg SO_2-eq/ton		60 g P-eq/ton		3.32 kg NMVOC-eq/ton	4.84×10^{-2} g CFC11-eq/ton		0.350 MWh/ton
		super calendered paper		3.45 kg SO_2-eq/ton		110 g P-eq/ton		3.21 kg NMVOC-eq/ton	5.96×10^{-2} g CFC11-eq/ton		0.470 MWh/ton

PAPER II

Life cycle assessment of bioethanol used for heavy-duty transport in Norway

Ellen Soldal

Abstract

The goal of this study is to evaluate environmental impacts of bioethanol used for heavy-duty transport. The raw material for the ethanol is woody biomass from Norwegian forest, and through a life cycle assessment (LCA), it is compared to conventional fossil diesel. The LCA indicate that the bioethanol may provide environmental benefits with regard to global warming potential, photochemical oxidant formation potential and ozone depletion potential, while the conventional diesel performs better regarding acidification potential, freshwater eutrophication potential and particulate matter formation potential. The impacts on local air quality are mixed. For the biogenic CO_2 emissions, different characterization factors are applied, and the analysis demonstrate that the accounting procedure for biogenic CO_2 is very important for the reported GWP and that the change in albedo after harvest should be included when analyzing the potential effect on global climate. Because the biogenic CO_2 is so important for the climate impact, local factors that influence the carbon flux in the forest needs to be included in future analysis.

1. Introduction

Concerns for climate change, future energy security and rural development have increased the interest for renewable energy (Bentsen et al., 2014), and bioenergy in particular (The European Parliament, 2009). Liquid biofuels for transportation are being promoted because they possibly provide both energy security and reduced greenhouse gas (GHG) emissions, and at the same time it is one of few options available for replacement of fossil fuel in the short-term (González-García et al., 2009b). Transportation accounts for 24 % of the global emissions of GHG and the transport sector depends heavily on fossil fuels (Gnansounou, 2010; United Nations, 2011). Mitigation strategies for the transport sector can include increased use of biofuels alongside with more efficient engine technologies (Norwegian Climate and Pollution Agency, 2010).

Worldwide, bioethanol is the most used renewable liquid fuel and the production has been increasing, from 17.3 billion liters in 2000 to 46 billion liters in 2007 (Balat, 2011; Balat and Balat, 2009; Quirin et al., 2004). Bioethanol is an alcohol that is produced by fermentation of sugars (Gode et al., 2011; Rødsrud et al., 2012). It can be made from woody biomass, municipal waste, annual agricultural crops and agricultural residue (Balat, 2011). Besides being renewable, the main advantages of using woody feedstock are that the use does not compete with food production and it offers flexibility with regard to harvesting time, compared to annual plants (Limayem and Ricke, 2012; Timilsina and Shrestha, 2011). Also, non-edible feedstock can often be grown on land that is less fertile, have relatively low harvesting costs and do not need very much maintenance for production (Wiloso et al., 2012). In order to increase the renewable energy share in the transportation sector and mitigate emissions of GHG, liquid fuels based on woody biomass is an interesting alternative for Norway, as the country has substantial forest biomass that can be used for bioenergy. Today, less than half of the annual increment is harvested (Trømborg et al., 2011; Trømborg and Solberg, 2010).

Many studies find that 2nd generation bioethanol has potential to reduce GHG emissions compared to fossil alternatives. Climate change mitigation potential of biofuels varies across feedstock, management regimes, conversion technologies, fossil energy input and local conditions (carbon sequestration in soil, harvesting yield) (Beer and Grant, 2007; Timilsina and Shrestha, 2011; von Blottnitz and Curran, 2007; Wiloso et al., 2012). In a well-to-wheel analysis of bioethanol (E85, 85 % bioethanol) used in a flexi-fuel vehicle based on Norway spruce, Bright and Strømman (2009) report GHG savings of 44-62% per km, compared to a gasoline fueled vehicle. In several studies González-García and colleagues have reported GHG savings of 50 % per km driven by a middle-sized passenger car fueled with E85 based on poplar and >60 % km for E100 compared to gasoline (González-García et al., 2009b; González-García et al., 2012; González-García et al., 2010).

Even though bioethanol performs better than conventional diesel or gasoline in many impact categories, several studies show that bioethanol typically performs worse than the fossil reference with regard to acidification and eutrophication (Cherubini and Ulgiati, 2010; González-García et al., 2009b; González-García et al., 2012). González-García et al. (2009b) show that when comparing blends of bioethanol, the fuel performs significantly worse in the impact categories acidification, eutrophication and photochemical oxidant formation (summer smog) as the share of ethanol increases (more than 100 % for acidification and eutrophication). They found that through the life cycle, bioethanol fuelled vehicles emit more NO_x, NMVOC, CO, NH_3 and N_2O than gasoline fuelled vehicles, up-stream agricultural activities like fertilization being the main contributor (González-García et al., 2009b). In Bright and Strømman (2009) the bioethanol performed better than gasoline in acidification (savings of 6-22 %), except for the worst case-modeling, and human toxicity (savings of 6-20 %). For eutrophication, the gasoline performed best (saving 2-24 %), except for the best case-modeling (Bright and Strømman, 2009).

Even though Timilsina and Shrestha (2011) found that bioethanol is better for local air quality than fossil fuels, increasing the amount of bioethanol in order to decrease the emissions of GHG may induce problem-shifting, as combustion of bioethanol can lead to increased emissions of harmful pollutants that have a negative effect on local air quality (Beer and Grant, 2007; López-Aparicio and Hak, 2013). Bioethanol is oxidized to acetaldehyde, and use of bioethanol is associated with increased levels of acetaldehyde and acetic acid. The former possibly carcinogenic and respiratory toxic and irritant, while the latter is corrosive and is connected with acid rain (López-Aparicio and Hak, 2013). But these substances belong to the category un-regulated substances, i.e. they are not regulated by legislation (Kytö et al., 2009). Thus, exhaust emission measurements does not include measurements of these. In Oslo, Norway, there are several buses running on bioethanol, and López-Aparicio and Hak (2013) performed air pollution screening along routes of bioethanol buses and along routes served by conventional diesel buses. They found the highest concentrations of acetaldehyde along the route of the bioethanol buses, while they could not detect a clear pattern regarding acetic acid. The concentration of acetaldehyde was above threshold limits in vicinity of the buses (López-Aparicio and Hak, 2013). Timilsina and Shrestha (2011) state that biofuels generally produce less particulate matter (PM) and volatile organic compounds (VOC) than their fossil counter parts. They also claim that ethanol lowers the sulfur and CO emissions, which are two threats to local air quality (Timilsina and Shrestha, 2011).

The bioethanol that is assessed in this study is produced by a biorefinery. The biorefinery concept of making use of most of the biomass input provides an efficient production pathway for biomass to liquid biofuels (Cherubini, 2010). The biorefinery in this study produces cellulose and bio chemicals in addition to bioethanol. Allocation is an important part of LCA and choices on the allocation are important for the results (Cherubini and Stromman, 2011). The different parts of the biorefinery are tightly connected but allocation of environmental burdens between the products is avoided when possible. When allocation was unavoidable, energy allocation was applied (Modahl et al., 2015). A comparative study of the GWP of the biorefinery products indicate that all products from this biorefinery perform better than the major competitive products (Rødsrud et al., 2012). The bioethanol is produced by a biochemical pathway and is based on Norway spruce (Rødsrud et al., 2012). In order to make the bioethanol approved as transportation fuel it is mixed with an additive (AD95). The abbreviation for bioethanol used is E95 and it follows the definition of Rehnlund et al. (2007): "Hydrous ethanol with low levels of additive such as denaturants, can be used in ethanol adapted ethanol diesel engines (today with ignition improver added to the ethanol)" (page VII).

Life cycle assessment (LCA) has been the prevailing tool to investigate the environmental impacts of bioenergy. Most of the LCA studies pay special attention to the biomass products' impact on global climate but the emissions of CO_2 from combustion of biomass has been assumed climate neutral based on the assumption that the same amount of CO_2 is sequestered by re-growth of biomass (Bowyer et al., 2012; Cherubini et al., 2011). Therefore, the global warming potential (GWP) of bioenergy in many LCAs only includes emissions of fossil CO_2 and non-CO_2 GHGs along the value chain. Lately, there has been a growing interest in the question on how to correctly account for CO_2 from bioenergy, and governments, NGOs and researchers have made proposals on how to solve this. In the US, the Environmental Protection Agency (EPA) has defined CO_2 as a pollutant under the Clean Air Act but because of uncertainties on how to separate between fossil and biogenic CO_2, they issued a three year deferral for biogenic CO_2 accounting (Environmental Protection Agency, 2011). Cherubini et al. (2011) argued that even though the same amount of CO_2 is being emitted and sequestered when using bioenergy, the biogenic CO_2 will stay in the atmosphere for a significant time, contributing to climate change. They launched the GWP_{bio} index in an attempt to capture the global warming potential of biogenic CO_2 based on the atmospheric decay of CO_2. The atmospheric decay of fossil CO_2 is described by the Bern CC model (Joos et al., 2012), and in the GWP_{bio} index removal of CO_2 from atmosphere due to regrowth of biomass is included (Cherubini et al., 2011). The longer the time horizon, the less is the impact from biomass use and the GWP_{bio} approaches zero as the time frame is extended (Cherubini et al., 2012).

According to Delucchi (2011) albedo is the most important human-induced climate change mechanism, together with emissions of GHG. The GWP_{bio} characterization factor has been expanded to include the effect of change in albedo (Bright et al., 2012; Cherubini et al., 2012). Changes in reflection of solar radiation (albedo) following a harvest has been shown to be important for the climate change mitigation potential of bioenergy, especially in the northern regions, where there is snow cover in the winter and early spring (Bright et al., 2013; Bright et al., 2011; Sjølie et al., 2013). In the latest report, the International Panel on Climate Change (IPCC) have medium confidence that change in surface albedo has lowered the radiative forcing (Watt/m^2) relative to 1750 by -0.15 W/m^2. In comparison, there is very high confidence that the emissions of CO_2 has increased the radiative forcing by 1.68 W/m^2 (Hartmann et al., 2013).

The GWP_{bio} is developed for single stand level, but Cherubini et al. (2013) investigated how a single stand level analysis relates to a landscape level approach. They find that the results for climate impact of biogenic CO_2 from single stand and landscape level are concurrent and that the spatial scale does not influence the conclusions (Cherubini et al., 2013).

In order to avoid problem shifting from global warming potential, other impact categories have been included. In addition to GWP, authors have pointed to acidification potential (AP), eutrophication potential (EP), photochemical oxidant formation potential (POFP) and ozone depletion potential (ODP) as important when assessing use of forest biomass (see for example Berg and Lindholm, 2005). Studies show diverging results in these impact categories for biofuels (see for example Bright and Strømman, 2009; Cherubini and Ulgiati, 2010; González-García et al., 2009a), and therefore I have included AP, EP, POFP and ODP in this analysis, in addition to GWP. Emissions of particles are important in fuel analysis because it is important for local air quality and is valuable information in strategic decisions on fuel choice in densely populated areas. Therefore, particulate matter formation potential (PMFP) has been included in the analysis.

There are few existing studies that evaluate high-blend lignocellulosic bioethanol used for heavy-duty transport (Beer and Grant, 2007; Beer et al., 2002; Rehnlund et al., 2007). Beer et al. (2002) does not include emissions of biogenic CO_2 in the carbon footprint. Beer and Grant (2007) investigated several impact categories, but like Beer et al. (2002) and Rehnlund et al. (2007), they did not include the potential climate effect of biogenic CO_2 and albedo. The aim of this study is to evaluate environmental impacts of bioethanol used for heavy-duty transport and compare it to conventional diesel through a well-to-wheel analysis, covering all life cycle stages from cradle to grave.

The research questions asked are:

1. What are the environmental effects of bioethanol (E95) based on Norwegian forest resources in selected impact categories important for bioenergy, compared to conventional diesel used for heavy-duty transport?
2. How are the exhaust emissions from combustion of E95 compared to conventional diesel?
3. What is the climatic impact of including the temporal change in albedo due to harvest and the temporal increase in atmospheric concentration of CO_2 due to combustion of bioethanol based on forest biomass harvested in Norway?

2. Material and methods

2.1. Goal and scope

The goal of this study is to assess environmental burdens associated with road transportation of goods for two fuel systems, with special attention given to the impact catgory global warming potential. The two systems compared are one fueled by bioethanol (E95) and one by conventional diesel (CD), both used for regional heavy duty-transport in Norway.

The environmental impacts of the production of lignocellulose bioethanol (cradle to gate) have been thoroughly investigated by Modahl and Vold (2010) and is also documented in (Modahl et al., 2015). Their analysis stop at the gate, and in the current study the system boundaries have been expanded to include transport to customer, production of additive, construction and maintenance of truck chassis, and user phase of the fuel (Figure 1).

The two fuels and the belonging vehicles are in use today by a transportation company in Norway. They are used for local and regional transport of groceries from a central storage to facilities where the groceries are sold or consumed. There is no difference in transported load between the two systems (Personal communication, Engen, O., 2012). Thus, the functional unit can be related to the distance travelled. The functional unit is 1 km driven by the truck. By making the functional unit related to the output of the two systems, the difference in the fuel efficiency of the vehicles is also accounted for, in addition to fuel consumption related to empty returns.

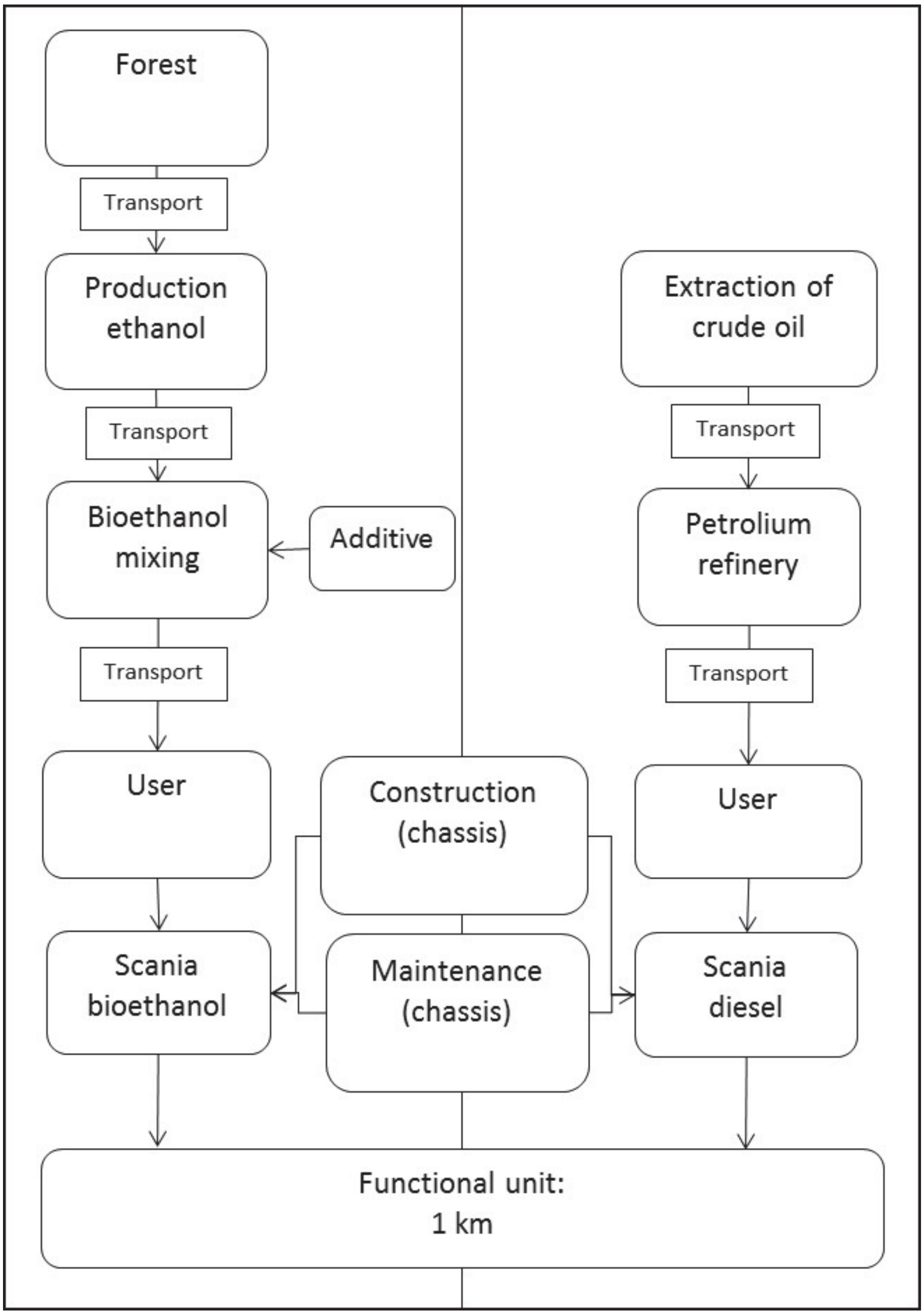

Figure 1: System boundaries for the transportation systems that are being compared in the analysis. Bioethanol (E95) on the left hand and conventional diesel (CD) on the right hand side.

2.2. Methodology

Results for the environmental impact categories of the E95 and the CD reported for this study are global warming potential (GWP), photochemical oxidant formation potential (POFP), ozone depletion potential (ODP), acidification potential (AP), eutrophication potential (represented by freshwater eutrophication potential, FEP) and particulate matter formation potential (PMFP) due to afore mentioned reasons. The impact categories are assessed using the ReCiPe midpoint method version 1.07, hierarchist version (Goedkoop et al., 2009). The analyses were carried out using the SimaPro 7.2.4 software (Pré Consultants, 2011) together with data from the Ecoinvent 2.2 database (Ecoinvent Centre, 2010).

IPCC normally reports GWP for time horizons of 20, 100 and 500 years (Solomon et al., 2007), and the GWP_{100} is used in this study. The potential climate impact of biogenic CO_2 and albedo are investigated through the application of the GWP_{bio} characterization factor (Bright et al., 2012; Cherubini et al., 2011).

The evaluation of exhaust emissions is based on measurements of emissions from the vehicle engine together with fuel consumption.

2.3. Life cycle inventory

The data for the life cycle inventory was collected from peer-reviewed literature, reports, Ecoinvent database and from personal communication.

2.3.1. Transportation fuel

The two fuels are used by a Norwegain transportation company, which has reported fuel consumption, distance travelled and volume load for both kinds of vehicles fueled by E95 and by CD over 12 months. There are five E95-vehicles and four CD-vehicles that have reported numbers for the full 12 months. For each moth, the average fuel consumption per km is given. The total average consumption of E95 is 0.55 l per kilometer, while the consumption of CD is 0.36 l/km (Personal communication, Engen, O., 2012). Table 1 provides information on fuel consumption (average, max and min), distance travelled and volume transported. The mean fuel consumption used in the study is not differentiated between turn and return because the vehicles often transport goods back to the terminal. Table 1 provides assumptions about fuel densities and fuel consumption.

Table 1: Fuel consumption for bioethanol and conventional diesel used for transportation of goods that have been applied in the analysis, in addition to distance travelled and volume transported. Assumed density of the fuels are also given.

	E95	**CD**
Average fuel consumption	0.55 l/km	0.36 l/km (0.302 kg/km)
Max fuel consumption per km	1.1 l/km	1.2 l/km
Min fuel consumption per km	0.12 l/km	0.12 l/km
Average km/month	2 416 km	2 685 km
Average load per month (m^3)	1 321 m^3	964 m^3
Density	0.789 kg/l	0.83 kg/l

The bioethanol is produced at a biorefinery located in Sarpsborg, Norway. The raw material is regional supplied Norway spruce (*Picea abies* (L), Karst.). Modahl et al. (2015) describes the production of bioethanol at the biorefinery in detail, and I refer the reader to this article for further details on the production and chemical use. Table 2 provides information on the LCA results for the bioethanol obtained by Modahl and Vold (2010). These results includes biomass procurement, infrastructure and production of ethanol. The biorefinery produces several products based on the same raw material and environmental burdens are allocated using mass allocation.

Table 2: LCA results for the bioethanol per m^3, from gradle to gate (Modahl & Vold, 2010).

Impact category	**Value**
Global warming potential (GWP)	324 kg CO_2-eq./m^3
Acidification potential (AP)	4.5 kg SO_2-eq./m^3
Eutrophication potential (EP)	2.17 kg PO_4^{-3}-eq./m^3
Photochemical oxidant formation potential (POFP)	0.29 kg C_2H_4-eq./m^3,
Ozone depletion potential (ODP)	5.6E-5 kg CFC-11-eq./m^3

The bioethanol is consumed by a transportation company situated 67 km away from the production site. In order to make the bioethanol accepted as transportation fuel, it is mixed with an additive. For the mixing, the ethanol is transported 17 km from the biorefinery, by road transport to the site where it is mixed (Personal communication, Enget, S., 2012). Then the ready-to-use bioethanol is assumed transported 50 km by road transport to the filling station. All transportation are assumed by truck in the Euro 5-class.

The additive constitutes about 5 % of the total weight of the fuel, and contains ignition improver, lubricant, corrosion protection and denaturants (SEKAB, 2009). The production of the additive is partly classified, and so the available information is somewhat limited. In Table 3 the relative amount of the chemicals used are listed. The relative amount of each chemical is provided by SEKAB (2009). I have used the mean amount of each chemical in the analysis. For production of the diesel, generic data from the Ecoinvent database is used (Ecoinvent Center 2010).

Table 3: Content (%) of the chemicals that are found in the additive that the bioethanol is mixed with in order to make it accepted as transportation fuel. The mean value is used in the analysis.

Additive ED95	Conc. weight %	Mean	Density	Data
Ethanol	20-25	22.5	0.789 l/kg	Bioethanol (96 %) from biorefinery (Modahl and Vold, 2010)
Methyl-t-butyl ether	17-21	19	0.74 l/kg	Methyl tert-butyl ether, at plant/RER U (Ecoinvent Center 2010)
Isobutanol	3-5	4	0.803 l/kg	Isobutanol, at plant/RER U (Ecoinvent Center 2010)
Lubricant	7-9	8	0.917 l/kg	Lubricating oil, at plant/RER U (Ecoinvent Center 2010)

2.3.2. Vehicle

There are two vehicles of interest, a bioethanol fueled truck and a fossil diesel fueled truck, both produced by Scania, Sweden. The trucks are assumed identical when it comes to construction and use (maintenance, distance traveled, refueling, loading capacity and loading routines). Expected life time is 540 000 km (Ecoinvent Center 2010). The powers of the vehicles are 270 horsepower. The diesel vehicle is assumed to be in the EEV Euro-class[1].

Scania has provided data on measurements of exhaust emissions per liter fuel consumed (Table 4) and construction of the vehicles (Table 5) (Bard, 2010; Klingberg, 2010). For the

[1] EEV stands for Enhanced environmentally friendly vehicle, and is a term used in the European emission standards for diesel vehicles >3.5 tons. EEV lies between Euro V and Euro VI.

production of the vehicles, an Ecoinvent process has been used as a basis, and modified in relation to the production details provided by Scania (Table 5).

Table 4: Composition of exhaust emissions due to combustion of bioethanol and conventional diesel, measured by Scania. (Bard, 2010; Klingberg, 2010).

	NOx	PM	HC	Biogenic CO_2	Fossil CO_2
E95	4	0,02	0,12	1480	
CD	7	0,05	0,02		2700

Table 5: Amount (kg) of material used to construct a Scania chassis (Bard, 2010; Klingberg, 2010).

Material	**Amount (kg)**
Scania chassis	*10 705*
Steel	3 960.9
Forging steel	2 355.1
Cast iron	2141
Aluminum	535.3
Plastics	535.3
Rubber (tyres)	481.7
Lead	160.6
Copper	107
Glass	107
Brass	53.5
Electronics	53.5
Paint	53.5
Oil and grease	53.5
Textile	53.5

For maintenance, the Ecoinvent process “Maintenance, lorry 28T/CH/I U” has been used. Energy consumption during production of the vehicles are reported by Scania (Bard, 2010; Klingberg, 2010), and listed in Table 6.

Table 6: Energy consumption at the Scania factory per produced chassis.

Energy	**Amount**	**Ecoinvent process**
Electricity	5.3 MWh	Electricity, medium voltage, production SE, at grid/SE U
District heating	1 MWh	0.95 MWh Wood chips, from industry, softwood, burned in furnace 1000 kW/CH U and 0.05 MWh Heat, light fuel oil, at industrial furnace 1MW/RER U
Diesel	0.5 MWh	Diesel, burned in building machine/GLO U
Natural gas	0.7 MWh	Natural gas, burned in industrial furnace >100kW/RER U
Heating oil	0.4 MWh	Heat, light fuel oil, at industrial furnace 1MW/RER U
LPG	0.3 MWh	Heat, natural gas, at industrial furnace >100kW/RER U
Total	*8.6 MWh*	

2.4. Biogenic CO_2 and albedo

In order to include the uncertainty regarding climate change mitigation impact of emissions of biogenic CO_2 and albedo, I have applied the GWP_{bio}, described by Cherubini et al. (2012). They have modelled removal of CO_2 from the atmosphere due to forest re-growth, and it varies with length of rotation and time horizon of analysis. The albedo is modelled to develop linearly from harvest and return to the pre-harvest level after 30 years. In this paper, I include the net GWP_{bio} (biogenic CO_2 and albedo) for time horizon 100 years for Norway. The GWP_{bio} changes when the system includes collection of forest residue, and this is included in the analysis. For detail on the GWP_{bio}, I refer the reader to Bright et al. (2012) and Cherubini et al. (2012). Characterization factors for biogenic CO_2 applied in this study are listed in Table 7.

Table 7: Different levels of characterization factors for biogenic CO2 and albedo included in order to assess the importance of the climate neutrality principle for bioenergy.

Method	GWP biogenic CO_2	GWP albedo	Characterization factor
GWP=0	0	-	0
GWP=1	1	-	1
GWP_{bio} (Bright et al., 2012)	0.62	-0.42	0.2
GWP_{bio} (incl. forest residue) (Bright et al., 2012)	0.51	-0.38	0.13

I have included all emissions of biogenic CO_2 through the life cycle of the bioethanol based on energy consumption during production, CO_2 formation during the fermentation process leading to bioethanol and the emissions of CO_2 from the combustion of bioethanol in the vehicle engine. The assumptions about biogenic CO_2 formation are found in Table 8.

Table 8: Formation of biogenic CO2 takes place in many of the life cycle stages of the bioethanol (E95). Listed in the table are source of biogenic CO2, assumptions and references.

Life cycle stage	Assumptions	Reference
Combustion of bioenergy at biorefinery:	1 100 MJ LHV/m^3 ethanol, LHV: 19.29 MJ/kg	Modahl and Vold, 2010
	Assumed carbon content: 50 %	
	Molecular weight ratio carbon to CO2: 44/12	
Combustion of waste at biorefinery	2 200 MJ LHV/ m^3 ethanol, LHV: 10.9 MJ/kg	Modahl and Vold, 2010
	Renewable fraction: 0.6	Marthinsen et al., 2010
	Organic fraction: 0.33	Marthinsen et al., 2008
	Assumed carbon content: 50 %	
	Molecular weight ratio carbon to CO_2: 44/12	
Complete fermentation	789 g CO_2/l	Rødsrud et al., 2012
Exhaust emissions	1480 g CO_2/l	Klingberg, 2010

3. Results

3.1. General environmental impact assessment

Detailed results for the E95 are presented in Table 9. The table does not include the absolute emissions from the CD system because it is based on the generic data in Ecoinvent and, hence quite uncertain, but a relationship between the impacts are given.

The E95 performs better than CD in three out of six impact categories, including GWP that will be covered more in depth in section 3.3. E95 performs better with regard to photochemical oxidant formation potential (POFP) and ozone depletion potential (ODP), while the CD performs better regarding terrestrial acidification potential (AP), freshwater eutrophication potential (FEP) and particulate matter formation potential (PMFP).

Table 9: Environmental burdens (g/km) related to use of bioethanol (E95) as transportation fuel, from well-to-wheel (functional unit 1 km). The total emissions are compared to that of 1 km driven by conventional diesel (CD) in the same system. Negative values indicate that the CD performs better than the E95.

	GWP ($g\ CO_2$ eq)	ODP (g CFC-11 eq)	AP ($g\ SO_2$ eq)	EP (g P eq)	POFP (g NMVOC)	PMFP (g PM10 eq)
Production	167	1.6E-05	1.73	3.0E-02	0.75	0.51
Transport	4	6.3E-07	0.01	3.8E-04	0.02	0.01
Additive	11	2.0E-06	0.06	1.9E-03	0.06	0.02
Production and maintenance chassis	65	7.6E-06	0.30	4.6E-02	0.23	0.15
Total	**246**	**2.6E-05**	**2.09**	**7.8E-02**	**1.07**	**0.68**
Comparison CD	**80 %**	**82 %**	**-7 %**	**-3 %**	**19 %**	**-11 %**

For all of the impact categories, the production of the bioethanol is the most important life cycle stage, except for eutrophication, where production and maintenance of the chassis is most important. This stage is identical for both vehicles, irrespective of fuel.

The difference between the E95 and CD are largest for ODP (82 %), while E95 has 19 % less potential for photochemical oxidant formation. The largest difference in favor of the CD is for the PMFP, where the value for E95 is 11 % larger. The value chain phase that is most influential regarding AP for the E95 is production and transport of timber, wood chips and chemicals. This is also the most important stage regarding POFP. According to Modahl and Vold (2010) all emissions from the biological effluent plant are allocated to the ethanol, and this may overestimate the FEP for E95. For ODP, production of energy carriers is the most important life cycle stage (Modahl and Vold, 2010).

The construction of the chassis contributes typically around 12 % of the life cycle emissions for E95. The exception is FEP, where it contributes about 50 % to the P-eq. emissions. The fuel consumption varies significantly, and therefore there are some uncertainty related to the absolute results. However, the variation in fuel consumption are in the same range for both fuels, so the relative relationship between the fuels may be burdened with less uncertainty.

3.2. Exhaust emissions

Scania has performed measurements of exhaust emissions on their vehicles (Bard, 2010; Klingberg, 2010), and together with fuel consumption, I calculated the emissions per km driven (Table 10).

With regard to NOx, the exhaust emissions from the E95 vehicle is 13 % lower compared to the CD vehicle. For particulate matter, the reduction is 25 %. For emissions of hydrocarbons (HC) the emissions from the E95 are 33 % higher than the CD.

Table 10: Emissions of substances (g/km) through the exhaust pipe due to combustion of bioetnaol (E95) and conventional diesel (CD), based on measurements of exhaust composition (g/l) (Bard, 2010; Klingberg, 2010) combined with fuel consumption (l/km).

Fuel	NO_x	PM	HC	Fossil CO_2	Biogenic CO_2
E95	2.2	0.011	0.07	0	814
CD	2.5	0.014	0.02	975	0

3.3. Global warming potential (GWP) of bioethanol

The total emissions of CO_2-eq (excluding biogenic CO_2) per km driven with E95 and CD, including production and all upstream activities, transport, additive for the E95 and production and maintenance of the chassis, are 246 g CO_2-eq/km and 1198 g CO_2-eq/km, respectively (Table 11). In these calculations biogenic CO_2 is assumed climate neutral. Thus, the CO_2-equivalents include emissions of fossil CO_2 and non-CO_2 GHG like methane and nitrous oxide. The main contributor for E95 is combustion of oil at the biorefinery, then production and transport of energy carriers, timber and chemicals follows (Modahl et al., 2015; Modahl and Vold, 2010).

Table 11: Emissions of CO_2-eq. (fossil CO_2 and non-CO_2 GHG) from well-to-wheel for 1 km driven by a truck fueled with E95 or CD (g CO_2-eq/km.

Life cycle stage	**E95** **g CO_2-eq/km**	**CD** **g CO_2-eq/km**
Production	169	
Transport	4	158
Additive	11	-
Production and maintenance chassis	65	65
Exhaust emissions	-	975
Total	**249**	**1198**

During the production and use of bioethanol, biogenic CO_2 is formed in several of the life cycle stages, not only the combustion of E95 in the vehicle engine (Table 8). The emissions of biogenic CO_2 are larger than the emissions of other GHG along the life time of E95 (Table 12).

Table 12: Total emissions of biogenic CO_2 from well-to-wheel for the ED96. For details on the factors of the calculations and assumptions, I refer the reader to Table 8.

Life cycle stage	**Calculation**	**g biogenic CO_2/km**
Combustion of bioenergy at biorefinery:	(1100/19.29)*0.5*(44/12)*0.55	58 g
Combustion of waste at biorefinery	(2200/10.9)*0.6*0.33*0.5*(44/12)*0.55	40 g
Fermentation	789 g/l * 0.55 l/km	434 g
Exhaust emissions	1480*0.55	814 g
Total biogenic CO_2 per km		**1346 g**

A comparison of the total CO_2-equivalents per km driven is conducted through the application of different characterization factors appointed to the biogenic CO_2. The analysis shows that, depending on the characterization factor for biogenic CO_2, the global warming potential of E95 varies form 246 g CO_2-eq/km to 1 592 g CO_2-eq/km (Table 13). For the GWP_{bio}, the net climate impact is resulting from a warming effect of biogenic CO_2 and a cooling effect of albedo. For comparison, the GWP of CD is also included.

Table 13: Total GWP (g CO_2-eq/km) per km driven with the bioethanol (E95) using different characterization factors for biogenic CO_2 and albedo. For comparison, the GWP of CD (g CO_2-eq/km) is included.

	Characterization factor	**Total amount of CO2-eq./km**
E95 (GWP=0)	0	246
E95 (GWP=1)	1	1592
E95	0.2	515
E95 (incl. forest residue)	0.13	421
CD	1	1 198

4. Discussion

Several environmental impacts of two transportation fuels were compared through a life cycle assessment, with emphasis on the potential climate change mitigation impact of bioethanol replacing conventional diesel. The analyses show that bioethanol used as transportation fuel may contribute to reduced GHG emissions when replacing fossil diesel, depending heavily on assumptions about climate impact of biogenic CO_2 relative to that of fossil CO_2. In this study, 84 % of the GHG emissions from the production and use of bioethanol is biogenic CO_2. If the biogenic CO_2 is assumed to have the same climate impact as fossil CO_2, the bioethanol emits more GHG than the conventional diesel. In the other impact categories included in the analysis, the results are mixed. Regarding eutrophication potential, acidification potential and particulate matter formation potential the conventional diesel performs better than the bioethanol. When it comes to ozone depletion potential and photochemical oxidant formation potential, the bioethanol has less impact than the conventional diesel.

Both vehicles in this study have exhaust emissions that are below the limits set by the European emissions standards. Even though the average fuel consumption per kilometer is higher for ethanol, the exhaust emissions per kilometer are lower for many of the substances important for local air quality. The exception is emissions of HC. According to Rehnlund et al. (2007) the measurements of HC for ethanol may be misleading because they include some ethanol which should not be included as HC. Because emissions of acetaldehyde and formaldehyde are not regulated by legislation the exhaust emissions of these substances are not measured, but the emissions of acetaldehyde are expected to be higher for E95 than CD, while CD usually has higher emissions of formaldehyde which represents a larger health risk (Rehnlund et al., 2007). Especially in densely populated areas, the improvements related to air quality is important.

Concerns about climate change is the main driver for the promotion of bioenergy in the developed world and this analysis shows that there is still uncertainty about how bioethanol used as transportation fuel can contribute. There are other studies that find that bioethanol reduces emissions of GHG, but most do not include the climate impact of biogenic CO_2 and albedo. Moreover, it is important to keep in mind other environmental aspects. In this study the bioethanol is less environmental friendly than conventional diesel with regard to EP, AP and PMFP, and this is consistent with other studies (Cherubini and Ulgiati, 2010; González-García et al., 2009b; González-García et al., 2012; von Blottnitz and Curran, 2007). In addition, issues regarding land use are important if the forest harvest is increasing. Several species depend on the forest, and there is a possible conflict of interest between climate change mitigation and conservation of biological diversity. In this study, land use change (LUC) has not been included because the productive forest in Norway is increasing, both with regard to area and volume. Nevertheless, if the production of bioethanol increases drastically, the effect of LUC must be included.

There are some sources of uncertainty in this study. Using GWP as a unit requires us to make some assumptions that add uncertainty to the results. Measured atmospheric concentration of CO_2 in January 2013 was 394.97 ppm (Dlugokencky and Tans, 2014) and this number is increasing. As the atmospheric concentration increases, the radiative efficiency of CO_2 decreases, i.e. the impact of one unit CO_2 is smaller when the background concentration is higher. The cooling effect of albedo represents some degree of uncertainty, as this relatively recent has been introduced in assessments of climate change. A changing climate will possibly alter the amount of snow and the period of snow cover. This will affect the impact of albedo. Changing climate may also have an impact on growth rates and mortality in the forest. It is difficult to get measures of actual exhaust emissions during use, because they depend on several factors like temperature, driving conditions, speed and traffic flow.

When analyzing the effect of bioenergy use, the time horizon is important. We need to take into account both short term and long term impacts. Climate change mitigation policies strive to reach goals on the long term, but the decisions and investments that need to be taken are near term. In this study a time horizon of 100 years is often applied but "There is certainly no conclusive scientific argument that can defend 100 years compared to other choices, and in the end the choice is a value-laden one" (Shine, 2009). The GWP_{bio} factor that has been introduced varies as time horizon of the study varies but it is also dependent on the rotation period of the biomass investigated. In this study, the biorefinery uses spruce in boreal forest, where the re-growth can take up to more than 100 years, but for other species the re-growth is faster. The albedo effect will also vary greatly between different geographical regions. Therefore, a characterization factor like this must to be locally adapted.

To conclude, this study shows that the crucial point in the evaluation of potential climate contribution by bioenergy systems is how to account for the increased emissions of biogenic CO_2 compared to reduced emissions of fossil CO_2. The assumptions about climate neutrality for bioenergy significantly influenced the results for global warming potential of bioethanol based on woody biomass. GWP_{bio} is a characterization factor for biomass-based energy that can be put in to use relatively easy by the LCA community, but there are several local factors that are important for the carbon flux in the forest, like forest management regimes, soil quality and type of forest ecosystem that is not included in a characterization factor like GWP_{bio}. The analysis indicate that there might be a conflict of interest between global environmental challenges, like climate change, and local environmental impacts, like eutrophication and acidification, when bioethanol is being promoted.

Future research

Wiloso et al. (2012) and Budsberg et al. (2012) conclude that carbon sequestration can offset value chain emissions of GHG for high share of ethanol blends. Thus, carbon sequestration and storage should be included in future analysis of climate change impact assessments of bioethanol. Application of an ecosystem model can also include biodiversity and landscape issues that are not covered by a traditional LCA. Future research on the climate change mitigation potential of bioenergy should include local factors that influence the flux of carbon in the forest in order to get a more complete picture of the climate impact of wood use. Further, analysis of bioenergy should also assess the impact on local air quality more in depth, including analysis of non-regulated substances, and impacts on ecosystems by increased harvest.

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PAPER III

Combining forest modeling and LCA: a case study of biodiversity and life cycle emissions for forest products

Ellen Soldal, Clara Valente, Even Bergseng, Ingunn Saur Modahl and Ole Jørgen Hanssen

Abstract

The forest plays an important role in climate change mitigation strategies, and wood has several applications with different environmental impacts. This study seeks to investigate the environmental effects of a mix of wood products and the conflict of interest between timber harvest and biological diversity. Potential environmental impacts are assessed through investigations of several forest value chains, a mix of products produced and the effect of different forest management scenarios.

Land use related impacts of forestry and potential environmental impacts of forest value chains were assessed by combining a forest model (GAYA-J) with LCA, including replacement of fossil-based products. Three management scenarios for a forest property in East Norway were defined in GAYA-J and the life cycle impact assessment for forest value chains, based on raw material from the current forest, were conducted. Considerations to biological diversity conservation and recreational value of the forest were included as management restrictions. A product mix was created, based on the forest value chains requirements regarding raw material. The functional unit was 1 km^2 productive forest.

Harvest yield (amount and other specifications) vary in the scenarios. Simulated harvest in the most restricted scenario is >60 % lower than in the other scenarios. However, as all the modelled value chains provide benefit in the impact category global warming potential compared to replaced products, it calls for a balancing between biological diversity conservation and climate change mitigation. In the other impact categories, the results are mixed. The production phase is the hot spot in most of the value chains.

This work demonstrates that a forest model can be combined with LCA. Forest products can potentially contribute to reduction of emissions of polluting substances. But there is a conflict of interest between conservation of biodiversity and climate change mitigation when the carbon storage and sequestration in the forest is not accounted for.

1 Introduction

There is increasing scientific agreement that anthropogenic emissions of greenhouse gases (GHG) cause global climate change (Forster et al. 2007). In order to mitigate climate change, there is growing attention towards increased use of biomass, particularly for energy purposes. Biomass is a promising renewable alternative to fossil resources, and woody biomass has numerous applications that can replace other non-renewable materials: transportation fuel, construction material, energy production and chemical production (Cherubini and Stromman 2011; Eriksson et al. 2012; Rødsrud et al. 2012).

Forests can contribute to mitigate human-driven climate change in several ways: for example through carbon sequestration and storage and through replacement of more carbon-intensive products (Smith et al. 2014). In Norway, 25% of the total land area is covered by productive forest, and the standing stock is increasing as the annual increment exceeds the annual harvest (de Wit et al. 2006; Granhus 2013). According to the International Panel on Climate Change (IPCC) forest management is an important mitigation strategy for boreal forest in the European region. Bioenergy has an essential role to play in their mitigation scenarios, and they predict that traditional, low efficient use of biomass for energy will decline at the same time as modern, high efficient use for heat, power, combined heat and power (CHP), industry and transportation will increase (Smith et al. 2014).

Forests are also important for a range of ecosystem services that can be affected (in a positive or negative way) when the use of biomass for energy and material increase (Smith et al. 2014). Rockstrom et al. (2009) name loss of biodiversity as one of the main three environmental challenges of today, where Nature's threshold limit has been exceeded. Through the Convention on Biological Diversity (CBD) Norway has obligations to conserve biological diversity and secure sustainable use of biological resources (Ministry of Environment 2001). In Norway, about 60 % of all terrestrial species are living in or in the proximity of forests (Direktoratet for naturforvaltning 2010). Of all the red listed species in Norway, 40 % is affiliated with forest (Artsdatabanken 2010). For many of the endangered species in forests, forestry and forest operations are the most influential factors (Artsdatabanken 2010; Direktoratet for naturforvaltning 2010). Because a large share of the endangered species is threatened by forestry activity, biodiversity assessment should be an important part of any environmental analysis of forest products. Even though land use, land use change and forestry (LULUCF) has been named the most important threat towards biological diversity, an assessment methodology for effects of LULUCF on biological diversity is lacking (Michelsen 2008). Forest management is important for the biodiversity, and sustainable forest certification schemes have been developed in order to secure that forests are managed in line with the current ecological, ethical and social standards. Nearly all productive forest in Norway is certified through the Program for the Endorsement of Forest Certification (PEFC 2013). In Norway, forest inventories include mapping of areas that entail qualities important for the preservation of biological diversity and biological diversity is conserved through constraints on the exploitation of these areas. These areas are registered as key habitats in the forest inventory.

Forests close to cities are important for the inhabitants in the vicinity and Norwegians have a tradition for using the local forest for recreational activities. In 2011, 81 % of the population stated that they had been hiking in forested areas or mountains at least once during the last year (Statistisk sentralbyrå 2011). The use of forests vary between regions and user groups (Sverdrup-Thygeson and Lie 2001). Local management is therefore important in order to reduce the level of conflicts between different users and between users and the forest owner.

Forest management can be used as a tool to increase the forests contribution to climate change mitigation, and there are several examples of studies that investigate the combined climatic effect of forest management and use of forest products (see for example Hoen and Solberg 1994; Raymer et al. 2009; Pingoud et al. 2010; Helin et al. 2012; Newell and Vos 2012). Raymer et al. (2009) optimized management in order to maximize the carbon benefits of forests in a Norwegian county (Hedmark). They used the stand simulator GAYA combined with an optimization model for forest management, called J/C (Raymer et al. 2009). Their model included all carbon flows in the forest, end-use of wood and replacement. Life cycle data for production of products were included in the forest model. The harvest was restricted to be at the same level as present (Raymer et al. 2009). In 2011, Raymer et al. used the same model, and assigned a value to carbon, to see how the value of carbon influenced the forest management. Eriksson et al. (2012) integrated a stand-level model, regional forest model, a partial equilibrium model and a wood product replacement model in order to find the potential climate change mitigation contribution by increased wood use in Europe. They found that increased demand for wood products had small effects on market and forest management. The CO_2 reductions were high per unit of wood used but low in total (Eriksson et al. 2012). Exceptions were when they modelled an extreme demand for wood all over Europe. Pingoud et al. (2010) assessed the trade-off between the temporary increase in carbon stock in the forest by increasing rotation length and the effect of replacement. They applied a displacement factor for wood which describes the reduction in fossil C in proportion to the wood used (Pingoud et al. 2010). They assumed a steady-state system, where forest growth and removal are identical, and so are inflow of wood products and decay of wood products (i.e. the product carbon stock is constant). They combined a wood supply model which included forest management with a wood use model, where the biomass yield (in ton C biomass/hectare) is divided in to three products, sawlogs, pulpwood and energywood and multiplied with the displacement factor (Pingoud et al. 2010). Raymer et al. (2011) included a product mix that is based on historical use of wood in Norway.

As mentioned, there is an interest in using forest biomass for climate change mitigation, and at the same time, biodiversity is threatened because of forestry activities. The main objective of this paper is to study the environmental effects of biomass used for a variety of wood products and how conservation of biodiversity influence potential climate change mitigation contribution. This will be explored through combining a forest model (GAYA-J) with life cycle assessment (LCA). The research questions asked in order to assess the potential environmental impacts of the biomass use are:

- What are the effects of the different biodiversity conservation measures on the environmental burdens and benefits of the biomass?
- Which parts of the forest value chains are most important in the cradle to grave assessment of forest products?
- What are the total environmental impacts of the biomass use for a variety of regionally relevant products?

GAYA-J and LCA are combined in order to analyze the environmental impacts of different use of forest biomass and how increased conservation measures influence the net present value of the forest, the harvest and the potential environmental cost and benefits by wood use. We investigate a case with a forest located in a Norwegian municipality (Fredrikstad), and define different forest management scenarios as well as alternative uses of biomass that are regionally relevant. Conservation of biological diversity is included in the model through the forest management scenarios, defined by restrictions on forest treatment, like type of harvest (clear-cut, selective logging), thinning regime and regeneration.

As mentioned earlier, several studies have investigated the combined effect of forest modelling and life cycle data. Unlike our study, they have included the life cycle data in the forest model (Raymer et al. 2009; Raymer et al. 2011) or used pre-defined values for the replacement (Pingoud et al. 2010; Eriksson et al. 2012). In this study we use a product mix that is determined by the harvest simulated by the forest model, and the output from the forest model (kg dry matter per km^2) is used as input to the life cycle assessment. This study distinguishes itself from the other studies by including biological diversity and recreational considerations in the analysis. Bergseng et al. (2011) included biological diversity in forest modeling but not in combination with life cycle assessment (LCA). Contrary to other studies mentioned, our study includes other impact categories in addition to global warming potential (GWP).

2 Material and method

2.1 Study area

The study area, the Fredrikstad Municipality forest, is located in the south-eastern part of Norway. The forest was inventoried in 2006 aided by aerial photographs. All GAYA-J simulations are based on this inventory. Total productive forest area is 4454 hectare. The forest is dominated by Norway spruce (*Picea abies* (L.) Karst.) and Scots pine (*Pinus sylvestris* L.) combined with some deciduous forest. The deciduous forest consists of mainly birch (*Betula*), and in the analysis all broad-leaved harvest is assumed to be birch. Forest area distributed on age classes and species are listed in Table 1.

The forest is located in the close vicinity of the cities Fredrikstad and Sarpsborg with more than 125,000 inhabitants, which utilize the forest for recreational activities like exercise, hiking and skiing.

Table 1: Forest area distributed on tree species and age class (20 years groups).

Age class (20 yrs)	Spruce	Pine	Broad leaves	Total
1	9.2%	3.5%	1.2%	14.0%
2	13.3%	5.7%	1.2%	20.2%
3	9.4%	6.3%	3.4%	19.2%
4	10.0%	2.5%	2.3%	14.8%
5	2.3%	5.4%	0.9%	8.5%
6	1.5%	7.7%	0.1%	9.3%
7	0.4%	9.6%	0.0%	10.1%
8	0.0%	3.1%	0.1%	3.2%
9	0.0%	0.8%	0.0%	0.8%
Total	46.2%	44.6%	9.2%	100.0%

2.2 Models and tools

In this study, two models are combined in order to analyze the environmental impacts of using forest biomass and how different forest management restrictions influence the emissions related environmental effects of forest products. The two models that we combine are a forest management model with optimization (GAYA-J) and an LCA model developed in SimaPro, version 8 (PRé Consultants 2013).

2.2.1 GAYA-J

The forest model GAYA-J consists of two tools: GAYA (Hoen and Eid 1990; Eid et al. 2002; Raymer et al. 2005; Bergseng et al. 2011) simulates numerous possible development paths for the forest based on the state of the forest, models for growth, mortality and regeneration and a set of treatment options; J (Lappi 2005) applies linear programming to select the optimal set of treatment options among the possible set simulated by GAYA given the object function (maximizing the forest’s net present value) and restrictions (on harvest flow, preservation of biodiversity etc.). Climate change is not taken into account in the simulations of treatment or harvest. GAYA -J is linked to a GIS and the inventory, all treatments, restrictions and results are spatially explicit. For example, areas of importance for biological diversity are designated and given restrictions on treatment.

GAYA provides simulation results for harvest (m^3, number of trees, species, share of sawn wood) and logging residues for three scenarios defined and two time horizons (20 and 100 years) given a number of restrictions. The time horizons have been selected because 20 year time frame is relevant for the plan periods in the municipality and the 100 year time frame is consistent with the IPCC time frame for LCA (European Commission-Joint Research Centre - Institute for Environment and Sustainability 2011). After GAYA has calculated a number of possible developments of the forest, J finds the optimal forest treatment, given the restrictions and an objective function of maximizing the net present value of the timber production.

Based on simulated harvest volumes, biomass dry weight in tonnes was calculated as a product of stem volume (V) using the biomass expansion factors (BEF) for spruce, pine and birch, as a proxy for broadleaved species. BEF are from Lehtonen et al. (2004):

$$W_i(V) = aV^b$$

We assume a 70% harvesting rate for logging residues (Nurmi 2007). The rest of the residues are left in the forest in order to avoid nutrient loss (Clarke 2012). Harvested logging residues were assumed left in the forest to dry, thus all results for logging residues are without leafs/needles.

2.2.2 LCA

LCA is a well-established and known methodology for assessing natural resource requirements and environmental impacts of the processes involved in the manufacture of a product, service or activity (ISO 2006a; ISO 2006b), from raw material extraction, through materials processing, use and disposal at the end of the product's life (from "cradle to grave") including all transportation steps involved. LCA assesses the environmental impacts of the system in the areas of ecological systems, human health and resource depletion. It does not report economic or social impacts.

The environmental impact categories assessed are climate change, acidification, eutrophication, photochemical oxidation, and ozone depletion, with their corresponding indicators global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), photochemical oxidation potential (POFP) and ozone depletion potential (ODP). These impacts are the most common environmental impact categories assessed in the forest fuel supply chains (see e.g. Berg and Lindholm 2005). The calculations of GWP include fossil CO_2 and all non-CO_2 GHG emissions and not emissions from land use, following the methodology of Giuntoli et al. (2014). Results for the environmental impacts were calculated by the LCA software SimaPro, version 8.02 (PRé Consultants 2013) together with the Ecoinvent 3 database (Ecoinvent Center 2013), a data source for life cycle inventory (LCI) data.

The system boundary of the study is the forest biomass supply chains are illustrated in Figure 1. These chains include six specific stages in the forest product chain: forestry operations, harvest treatment (when present), transport, production, user phase (when it causes emissions) and avoided products/energy. The disposal stage is not included in the study. In this LCA, the inputs and outputs referrers to a functional unit (FU) of 1 km^2 of average productive forest in Fredrikstad municipality. 1 km^2 was chosen as FU because the goal of the study was to find the environmental impact connected with wooden products from a forested area given different management regimes (scenarios described below) over different time periods (20 and 100 years). Furthermore, an area based unit was chosen because the biomass yield and forest management depends on the geographical location, and it provides the possibility to scale up the results to larger geographical areas. The analysis is carried out on a dry basis, i.e. the amounts of the various products are calculated in tonne dry matter (DM). The biomass output from the forest (as dry matter per km^2) depends on the selected time horizon and forest management scenario. The analysis includes all the major value chains relevant for the region's forest sector; energy (ethanol, heat (DH) and cogeneration of heat and electricity (CHP)), materials (construction material and packaging) and bio-chemicals (special cellulose, vanillin and lignin) (Figure 1). All products are based on the same raw material, i.e. timber from Fredrikstad municipality's forest.

An attributional approach has been used, because the results will be used for "accounting" purposes and "micro-level decision support" only. These decision context situations are called situation C and A according to the ILCD Handbook (European Commission-Joint Research Centre - Institute for Environment and Sustainability 2010).

The LCA model for the selected value chains was based on data collected from several peer review studies, documents and reports (incl. not public internal reports at Ostfold Research). Where specific data were missing, generic data were supplied by the Ecoinvent 3 database (Ecoinvent Center 2013). Data, sources and assumptions related to the examined processes and products are listed in table A1 in the Appendix.

The "Forestry operations" (Figure 1) includes the activities from the forest stand up to the forest road. This stage includes regeneration, thinning, final harvest and forwarding (terrain transport). In accordance with Flæte (2009), we assumed 43 % regeneration and diesel consumption 0.00937 l/m^3. Diesel consumption associated with pre-commercial thinning was 0.0594 l/m^3 (Flæte 2009). Harvesting and forwarding (990 m terrain transport) use 2.822 l of diesel per m^3 (Flæte 2009).

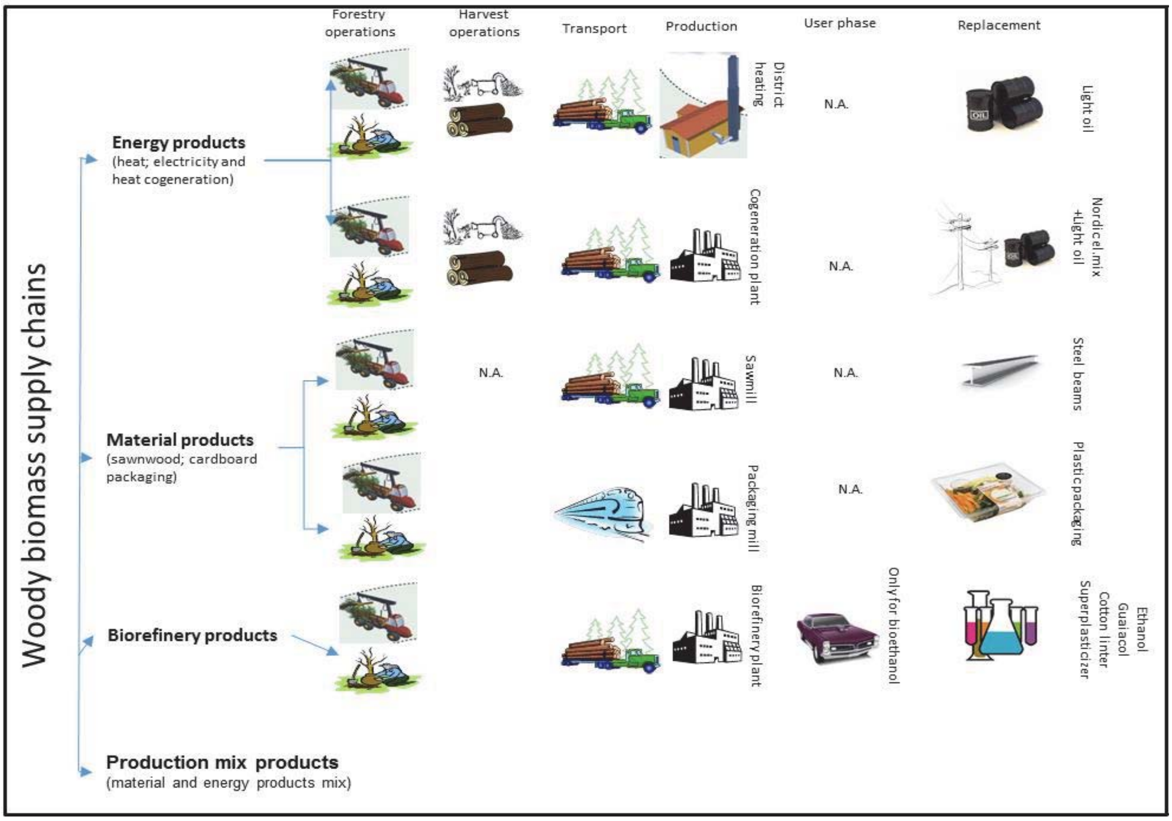

Figure 1: Forest biomass value chains: energy, material, biorefinery products, production mix products and replaced products

The “Harvest operations” (Figure 1) includes drying and chipping of harvested biomass. Chipping use 1.2 l of diesel per m^3 (Valente et al. 2011), while it was assumed natural drying and no emissions related to that (Pers.com E. Trømborg 2013).

“Transport” (Figure 1) from the forest stand to the production site takes place by train or truck depending on the distance (Table 2). In Norway 130 km is the shortest economically viable transport distance for railway transport of timber (Statens landbruksforvaltning 2010). Thus, if the distance is longer than 130 km, the biomass was assumed transported by train.

The “Production” (Figure 1) includes the materials, chemicals and energy related to the production site (mill or plant). The data for DH are based on a chips-fueled combustion facility in Ås, Norway (Bjerk 2007; Hafslund 2007). The CHP is assumed to be similar to the 10 MW-el plant producing 40 GWh electricity and 120 GWh heat, described in Brekke et al. (2008). Energy allocation was used to apportion burdens between heat (75%) and electricity (25%). In the packaging case, data for paper and pulp production (infrastructure, electricity, heat assumed from light oil, and emissions to water) come from an integrated pulp and paper plant (Norske skog 2012). The production of paper is used as a proxy, and the value chain is completed with data for a packaging mill (infrastructure, electricity use, water and waste), based on an Environmental Product Declaration (EPD) of cardboard packaging (Korsnäs AB 2011). For the construction material case, the data source is an EPD of Norwegian sawn dried timber (Sørnes 2012). The biorefinery case is based on an LCA for several products from a biorefinery plant (Modahl and Vold 2010). For special cellulose, vanillin and lignin powder the avoided emissions due to replacement were only available for GHG. Hence, GWP results are shown for the biorefinery value chain while the results in the other impact categories includes only bioethanol from the biorefinery value chain. Lignin powder was selected instead of lignin liquid, based on a conservative approach.

The “User phase” (Figure 1) includes the operational stage only in cases where this causes additional emissions. For the bioenergy products DH and CHP, the combustion of the biomass takes place in the “Production” stage and not in the “User” stage. The use phase was relevant only for the bioethanol case, and includes emissions from combustion of bioethanol in the engine assumed as in a passenger car, and not the infrastructure for the production of the car or the road etc. For the other products, there are no emissions from the use phase, and this stage is omitted.

The stage "Replacement" (Figure 1) is the stage when a product based on woody biomass substitutes a reference product produced by other raw materials and methods. Inclusion of this stage facilitates the calculation of avoided emissions (the amount of emission savings) due to the replacement effect. Assumptions regarding replaced products are listed in Table A1.

Table 2: Data and data sources for transport of biomass to production site. Empty returns has been assumed for transport of biomass.

Transport	Distance (km)	Type	Data sources
To sawmill	69	Truck	Average transport distance of sawn timber in Østfold (Flæte, P. O. 2009)
To CHP plant	17	Truck	Brekke et al. (2008)
To district heating	62	Truck	Google maps: Fredrikstad-Ås.
To packaging mill	130	Train	130 km is the shortest economically viable distance for railway transport in Norway (Statens landbruksforvaltning 2010)
To biorefinery	14	Truck	Google maps: Fredrikstad-Sarpsborg.

2.3 Scenarios

Three scenarios have been defined as a basis for analyses. Two of them include a set of constraints on forest management in order to preserve biological diversity, cultural heritage sites and recreational values, while one scenario do not include any constraints on the forest management to maximize economic output for the forest owner:

1. Reference scenario (REF): for comparison; a base scenario without restrictions on forest management.
2. Program for the Endorsement of Forest Certification (PEFC): constraints on management given in order to preserve biodiversity. This scenario is equal to the

current certification regime for sustainable forestry in Norway and the constraints are an operationalization of the restrictions given by the certification organization. The PEFC is legally authorized by the Norwegian Forest Act of 2005 (Landbruks- og matdepartementet 2005, LOV-2005-05-27-31).

3. Biodiversity (BD): constraints on forest management with extensive care taken to preserve biological diversity and maintain the forests' value as recreational area. The constraints exceed the PEFC scenario with explicit local adaptations as defined by the local authorities as a basis for further multipurpose planning of the area, which enabled inclusion of specific considerations like important recreational areas. The municipality assigned areas to four different categories ranging from normal forestry with no restrictions (cat. 1) to full preservation of forest (i.e. no harvest at all, cat. 4). Category 2 and 3 are gradients between these two extremes.

The main categories of management restrictions in the model are "No restrictions", "No clear-cutting" and "No harvesting". In the BD scenario, 24.3 % of the productive forest has the highest level of restrictions (no harvest), while in the PEFC scenario only 1.4 % of the productive forest has this strict level of management. At the other end of the scale, the PEFC scenario has 90.4 % of the forest area without restrictions, and the equivalent number for the BD scenario is 12.3 %. In the BD scenario, clear cutting is not allowed on 63.4 % of the area, while in the PEFC scenario, clear-cutting is not allowed on 8.2 % of the area.

Other restrictions that have been incorporated in the PEFC scenario are:

- No treatment in key habitats, as defined by the certification scheme.
- Minimum 10 % of the forest area must be deciduous forest.
- Conserve minimum 10 retention trees/ha at final harvest.
- Restricted treatment in areas with cultural heritage sites.

A fundamental assumption in GAYA-J is that the forest owner maximizes utility, here represented by net present value of timber production. By defining the three scenarios we explore how the harvest change in the same forested area, given weight to other factors than pure economic considerations. The PEFC scenario can be used as a proxy for a regular, private forest manager maximizing profit, while the BD scenario let the local government define restrictions on the management that may ensure other management goals, like biodiversity and recreational value, and not only the economic value.

In Norway municipalities are obligated to develop climate and energy plans, in order to contribute to reduction of GHG emissions. Thus, further motivation for choosing a public forest was to create a tool that could assist the local government in making climate and energy plans that were founded on local data and aiding them in assessment of locally available biomass resources.

2.4 Production mix

The simulated harvest provide timber and logging residues of different species (spruce, pine and birch), qualities and dimensions (sawn wood, pulp wood and logging residue), using price functions from Billingsmo and Veidahl (1992). Several of the modelled production technologies have restrictions on what kind of biomass (species and dimensions) they can utilize (Table 3). Therefore it was necessary to create a production mix which defined the share of harvest appointed to the different value chains. Each scenario and time horizon gives different output regarding species composition and quality, and consequently different production mixes. Some assumptions apply when the harvest where appointed to the different products:

- Only spruce and pine were used as construction material.
- The biorefinery only uses spruce.
- Production of construction material produced 50 % by-products that was assigned to district heating, CHP and packaging.

The distribution of the harvest between the different products is displayed in Table 3.

The biorefinery modelled uses Norway spruce and produces multiple outputs. Hence, it is not useful to separate the different products in the analysis, but rather create one product, namely “Biorefinery product”, that represents the weighted mix of products mentioned in Figure 1. The environmental impacts of the biorefinery products are allocated based on mass of output from production because it is tightly connected with mass of input. Mass allocation provides the following distribution between products: ethanol 5.86 %, vanillin 0.35 %, lignin and special cellulose 46.89 % each (Borregaard 2014). This allocation is only applicable to the distribution between the different products produced by the biorefinery. In the report by Modahl and Vold (2010) allocation has been avoided as far as possible, but when allocation could not be avoided, they have used mass allocation.

Table 3: The harvested wood is divided between the different products according to demands regarding species and quality. DH=district heating, CHP= combined heat and power.

Species	Logging residue	Sawn wood	Pulp wood
Spruce	DH (50 %), CHP (50 %)	Construction (25 %), DH (8 %), CHP (8 %), packaging (8 %), biorefinery (50 %)	Biorefinery (25 %), packaging (25 %), CHP (25 %), DH (25 %)
Pine	DH (50 %), CHP (50 %)	Construction (50 %), DH (17 %), CHP (17 %), packaging (17 %).	Packaging (33 %), CHP (33 %), DH (33 %)
Birch	DH (50 %), CHP (50 %)	-	Packaging (33 %), CHP (33 %), DH (33 %)

3 Results

3.1 Scenarios

The simulated output from harvest differs in the three scenarios (for complete results, see Table B1 in the Appendix). The trends in the results for the two TH are similar, and therefore results only for the TH=20 years are presented in Figure 2.

As a consequence of the areal restrictions on treatment, the simulated harvest from the scenarios varies. The restrictions connected with the PEFC scenario leads to a smaller harvest than what is the case in the REF scenario, the difference in harvested dry matter per km^2 is 8 %. The species compositions in the harvests in these two scenarios are not very different (Figure 2). The difference between BD and PEFC scenarios is large both regarding amount of harvest and species composition. In BD, the share of pine is larger while the share of spruce and birch are smaller than in PEFC. The amount of biomass harvested in BD is more than 60 % lower than in PEFC (Figure 2).

The net present value (NPV) of the timber follow the same trend as the harvest. In the PEFC scenario, the NPV is reduced by 6 % compared to the REF scenario, and in the BD scenario, the reduction in NPV is 48 %.

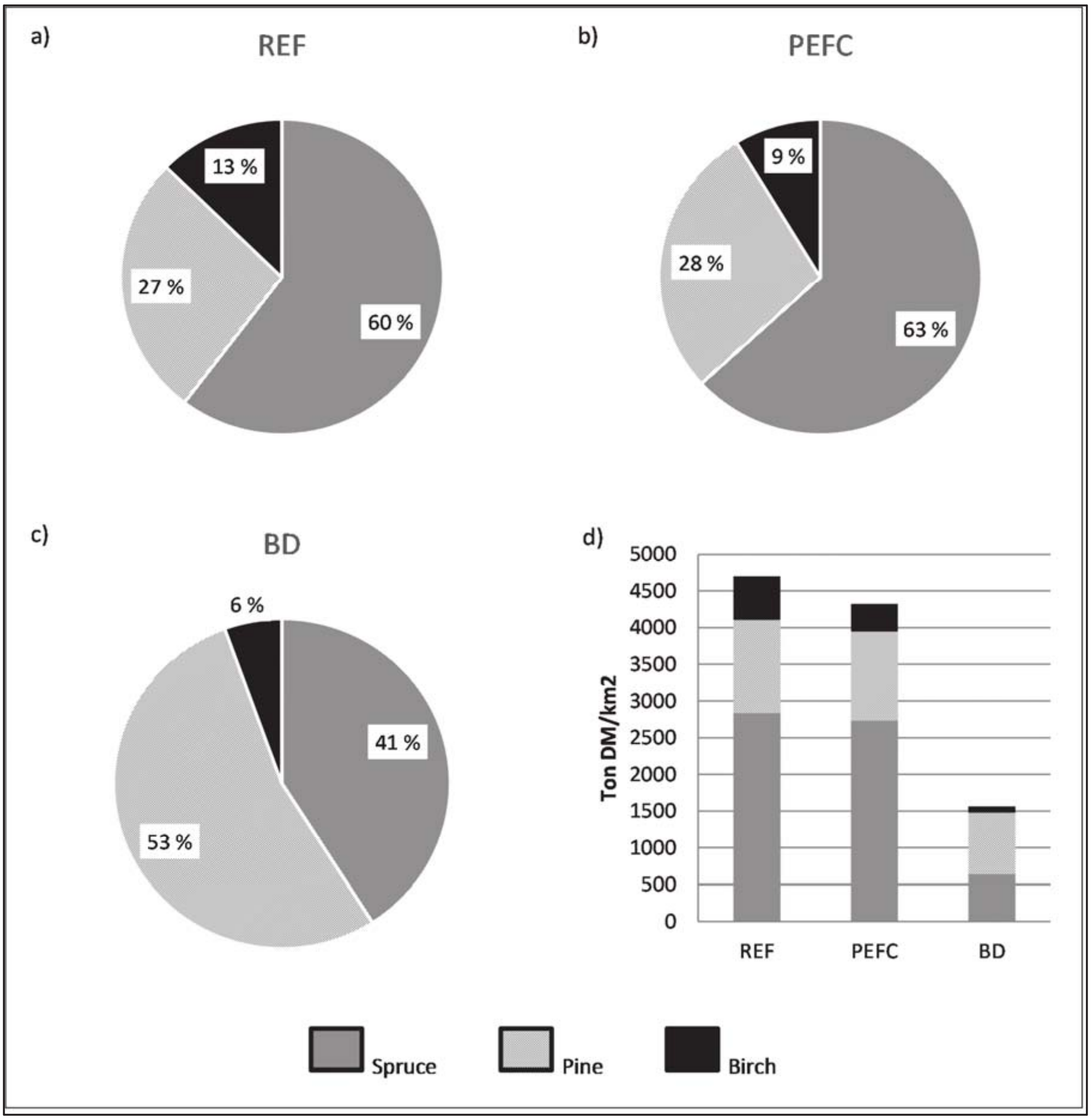

Figure 2: Species composition of harvest in percent (a-c) and total amount of dry matter harvested (ton DM/km2) (d) in the different scenarios. Time horizon 20 years. REF=reference scenario, PEFC= forest certification scenario, BD= biodiversity scenario.

3.2 Value chain

When considering only the emissions occurring within the biomass value chains (i.e. excluding replacement), the production phase has the greatest impacts in almost all the product categories (Figure 3 and 4). The stages along the value chain, ranked from largest to smallest environmental burdens are (the stages in parenthesis are not common to all products):

Production → (User) → Forestry operations → (Chipping) → Transport to production

This ranking of product stages is common for most value chains and environmental impact categories except for cogeneration of heat and power (CHP) and district heating (DH), where the forestry operations are the most important processes in the impact category ODP (CHP, Figure 4) and GWP (CHP and DH, Figure 3).

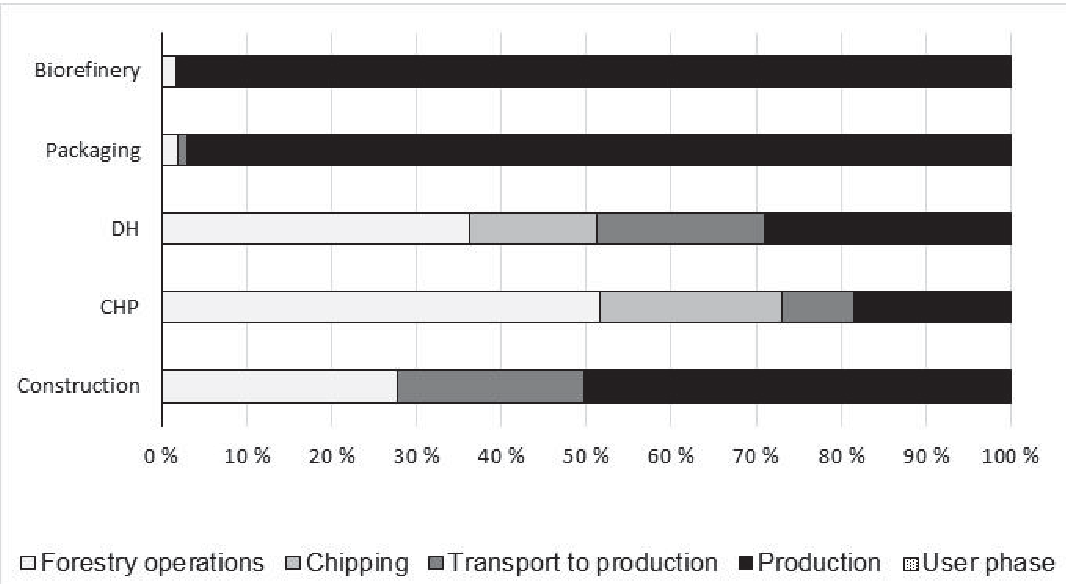

Figure 3: Percentage distribution of impacts in potential global warming from the life cycle stages in the forest value chains (not including replacement). DH= district heating, CHP= combined heat and power.

In the EP impact category, all products but packaging and CHP gave benefits (Figure 5). The emissions of PO_4^{-3}-eq. along the packaging value chain are more than 2.5 times larger than the emissions from the replaced plastic packaging. The hot spot in packaging production in EP is use of sodium hydroxide (NaOH) in the production of the pulp. Second most influential process is the electricity used at the paper mill and the packaging mill. The greatest benefits in the EP impact category are produced by construction material. The replacement of steel beams saves 25 times more emissions than the emissions along the value chain for construction material. Actually, the replacement effect of steel beams is large and the emissions from the wooden construction material are relatively small, thus the benefits from the construction material are larger than the benefit from other analysed products in three (EP, AP and GWP) out of the five impact categories analysed.

In the impact category acidification (AD), all products but bioethanol give benefits. Construction materials give the greatest benefits because of the large emissions connected with production of steel beams. Emissions of SO_2-equvalents from the steel beam value chain are more than 20 times higher than the emissions of SO_2-eq. from the wood based construction materials. For bioethanol, the production stage was the the main contributor – in the PEFC-scenario more than half of the emissions of SO_2-equivalents came from this stage (22,042 kg SO_2-eq) with the user phase being the second largest contributor. However, the bioethanol replaced 20,502 kg SO_2-eq emitted by fossil ethanol, the production stage being the main contributor. At the biorefinery, all emissions of SO_2-equivalents from the biological purification plant are allocated to the production of ethanol (Modahl and Vold 2010).

In all product categories and scenarios, the biomass based products provide benefits regarding POFP, i.e. the benefits from replacement are higher than the emissions related to the supply chain. The greatest benefits come from bioethanol. Bioethanol has the largest emissions during the production, but at the same time it has the largest savings when replacing fossil ethanol. Also construction material provides large benefits in this impact category.

In ODP impact category, all the products but bioethanol and packaging give benefits. Construction material shows the greatest benefits while packaging has the worst impact in this category. Packaging performs worst in this category due to large emissions in the production. Electricity used in the production phase is a hot spot. Also here, the use of NaOH is important. At the biorefinery, the production and transportation of energy carriers are the most important process regarding ODP. The results for the other products are characterized by high replacement effect in ODP. This is especially true for district heating (DH). DH has larger benefits than CHP and this is because it replaces only light fuel oil which is associated with high emissions, while CHP also replaces electricity from NordEl which has lower impact regarding ODP.

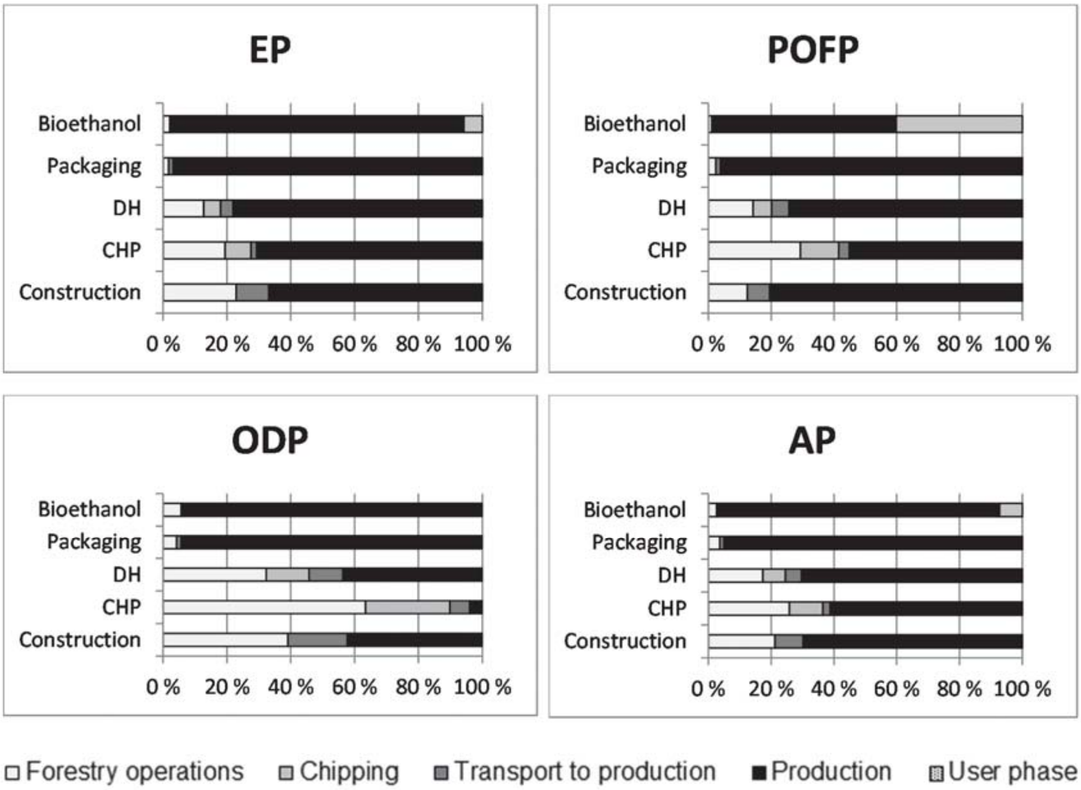

Figure 4: Percentage contribution to eutrophication potential (EP), photochemical oxidant formation potential (POFP), ozone depletion potential (ODP) and acidification potential (AP) at different life cycle stages of the forest products. The relationship between the shares of emissions is the same for all scenarios and time horizons when the product mix is not included. Values for biorefinery are reported for bioethanol only. DH= district heating, CHP= combined heat and power.

In the GWP impact category, there are benefits from all products in the assessment. The greatest benefits are evident for construction (35-40 %) and biorefinery products (18-28 %). Construction has low production emissions and large replacement effect, while the biorefinery products have high product emissions and large replacement effect. The most important process regarding GWP of biorefinery products is combustion of oil at the production site. Production and transport of energy carriers is the second most important process.

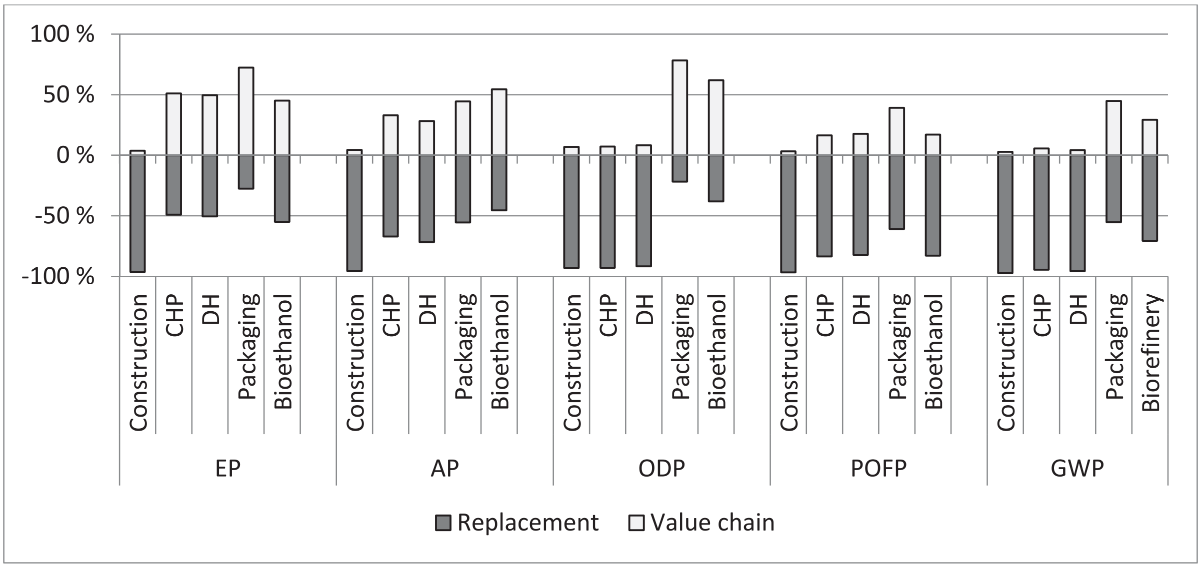

Figure 5: Relationship (in %) between forest value chains emissions and emissions from the replacement product in the impact categories eutrophication potential (EP), acidification potential (AP), ozone depletion potential (ODP) photochemical oxidant formation potential (POFP) and global warming potential (GWP). Values below zero are avoided emissions because of replacement of fossil products. DH= district heating, CHP= combined heat and power.

3.3 Production mix

The harvest varies over the different scenarios (Table B1), and the species composition and the share of sawn wood are two factors that influence the production mix. Within the 20 years TH, the share of sawn wood varies from 65 % in the BD scenario to 75 % in the PEFC scenario. In the PEFC scenario the share of spruce is 63 % while in the BD scenario it is 41 % (Figure 2). This is important for the production mix, as the biorefinery only uses spruce and construction material only used sawn wood. Even though the share of sawn wood is smaller in the BD scenario, the share of biomass going to construction is larger because a larger share of the sawn wood is pine that the biorefinery cannot use (see Table B2). When this product mix is applied, the sum of emissions from a km^2 of forest is smaller than the emissions from the modelled replacement products in all impact categories (Figure 6).

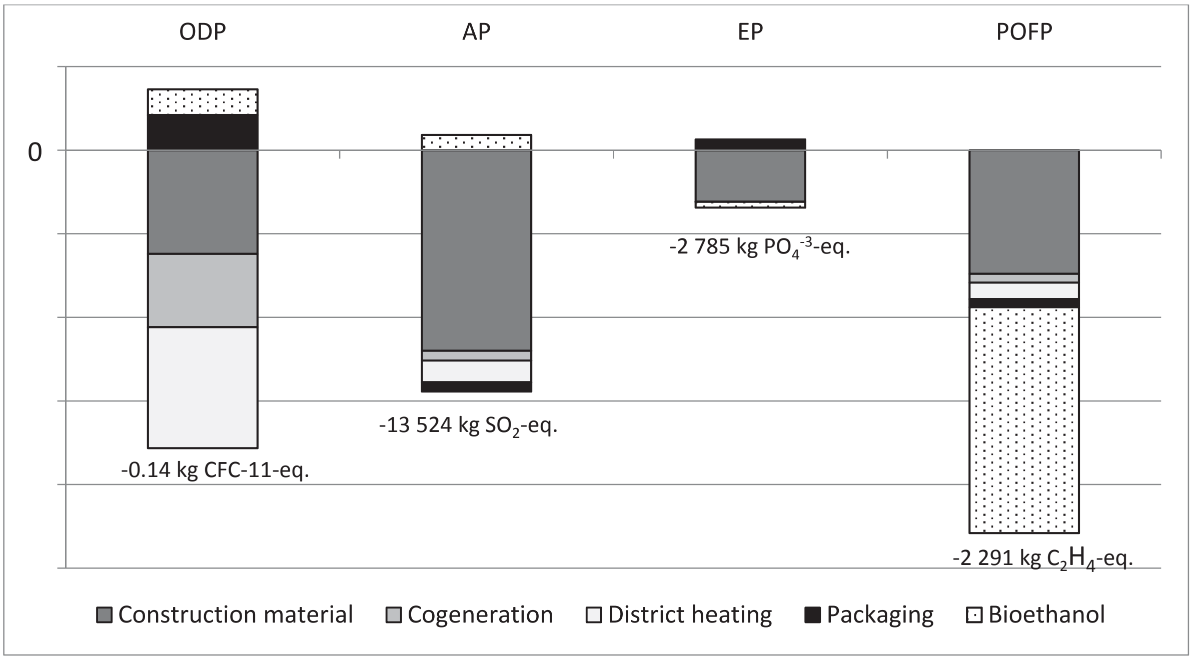

Figure 6: The absolute impact of different products in the product mix in the PEFC scenario (kg/km^2). The impacts in the other scenarios follow the same pattern. Values below zero are benefits of the wood based product compared to the fossil reference. In order to make the figure readable for all the impact categories, some of the values in the graph are given in thousands or hundreds. Absolute value given below bar. Values for biorefinery are reported for bioethanol only. ODP= ozone depletion potential, AP= acidification potential, EP= eutrophication potential, POFP= photochemical oxidant formation potential.

Figure 7 shows the relative contribution of the product mix in the impact categories GWP, ODP, POFP, AP and EP. The environmental benefits follow the same pattern as the harvest, except for EP. Because of the product mix, a smaller share of the total harvested biomass enters the packaging value chain in scenario PEFC than in BD (Table B2). In BD a larger share of the harvest is directed to the packaging value chain, and this creates a different trend for EP compared to the other impact categories (Figure 7).

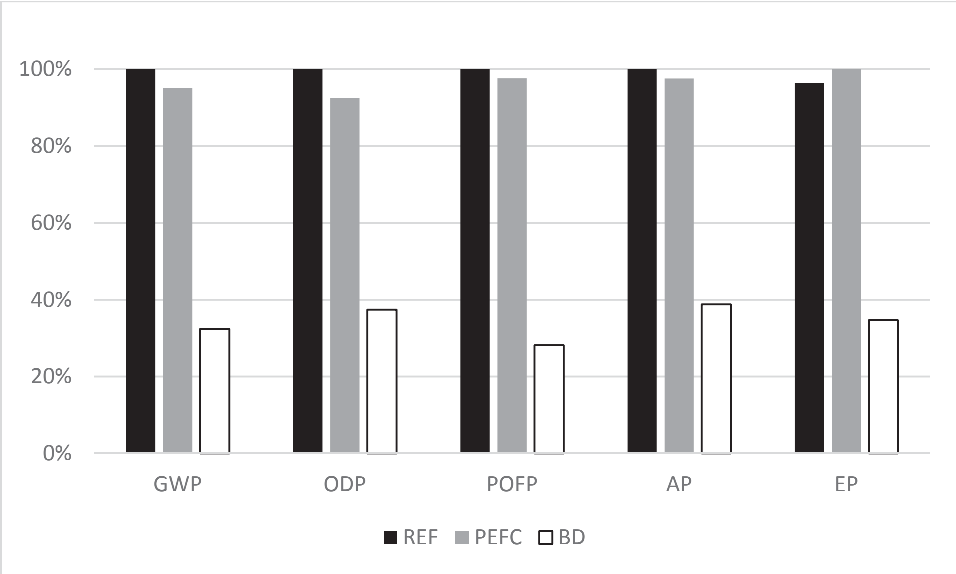

Figure 7: Relative (%) environmental benefits by the product mix for all impact categories and scenarios.

The net emissions of CO_2-eq/km^2 are shown in Figure 8. When the product mix is applied 19 % of the harvested biomass goes to construction material in PEFC (TH=20), but the relative contribution to the GWP benefits is greater; it produces 37 % of the GHG savings. The GHG-savings per m^3 of harvested wood (kg CO_2-eq/m^3) vary between 568 and 614 kg CO_2-eq/m^3 for the different scenarios and time horizons, with PEFC over 20 years providing the greatest savings per m^3, and BD over 100 years the smallest savings. The differences in savings per m^3 are related to the product mix for the different scenarios. When the savings are related to total area of productive forest, the savings vary between 18.4 and 56.7 tons of CO_2-eq/ha productive forest for the 20 years TH. For TH=100, the savings vary between 91.4 and 275.5 tons of CO_2-eq/ha. Here, the BD scenario has the smallest savings due to the fact that carbon storage and sequestration is not credited.

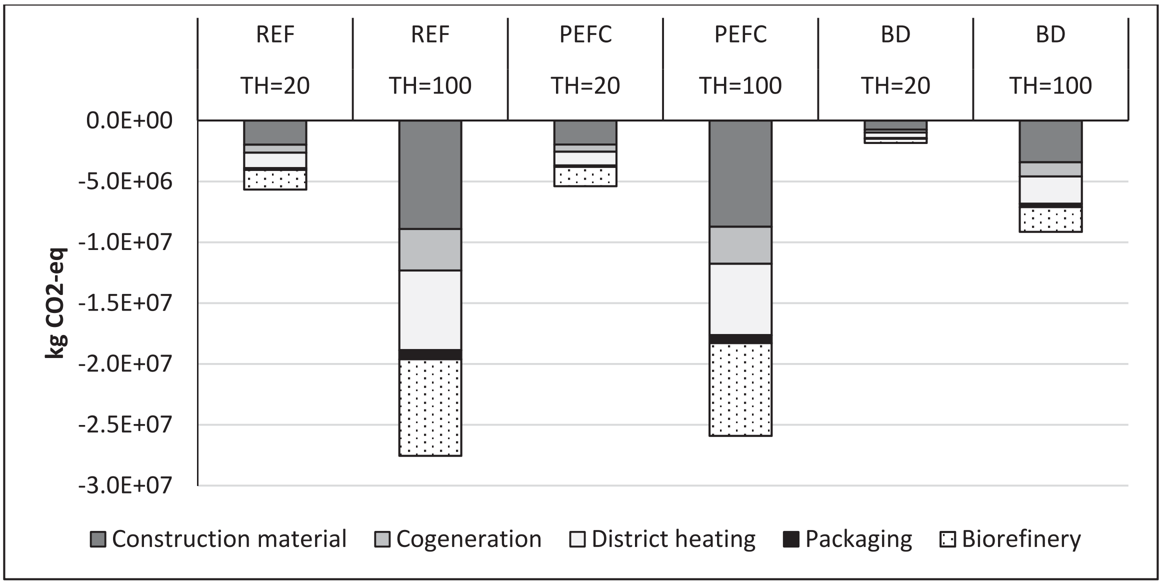

Figure 8 Net emissions of kg CO_2-eq/km^2 of productive forest from the different value chains in the management scenarios when the production mix is applied.

4 Discussion

Our work has demonstrated that a forest model can be combined with LCA in order to assess the conflict of interest between climate change mitigation, other environmental impacts, biological diversity and landscape considerations. This study shows that there is a potential conflict between objectives for forest contributions to mitigating climate change and a more holistic approach for forest management.

In this model, use of biomass as replacement for other, more carbon-intense materials and energy, is beneficial regarding emissions of GHG and C_2H_4-equivalents. In the other environmental impact categories analysed, the results are not so clear. Regarding ODP, both woodbased packaging and bioethanol have higher impacts than the fossil reference. This is also true for AP for bioethanol and EP for packaging. Construction material, DH and CHP have environmental benefits in all analysed impact categories compared to the fossil reference. In the scenario with the most restricted treatment (BD), the harvest is significantly smaller, and the environmental benefits coincides with this because when the production mix is applied, the use of woody biomass instead of other products have environmental benefits in all impact categories analysed. Thus, a trade-off between conservation of biological diversity and recreational areas on one side and the replacement effect in several impact categories on the other side. The harvest is reduced by more than

60 % in the BD-scenario compared with the PEFC-scenario. The benefits of the biomass value chain are reduced by values in the same range as the harvest reduction. The BD scenario is not credited for sequestration and storage of carbon. Inclusion of carbon sequestration and storage will probably improve the benefits of the forest regarding climate change mitigation (Raymer et al. 2011). According to Trumper at al. (2009) the boreal forest is a carbon sink and the storage of carbon in forest vegetation is in the range between 61-93 ton C/ha. The PEFC-scenario is not very different from the baseline scenario when it comes to harvest output; the reduction in harvest is less than 10 %. Bergseng et al. (2011) found that restrictions similar to our PEFC scenario gave a 5-10 % lower harvest than the reference for the first 30 years, after that the harvest was almost the same. For their biodiversity scenario, they found a 50-80 % lower annual harvest during the first 60 years, and after that the annual harvest was 25-60 % higher than in the reference scenario (Bergseng et al. 2011). We found a reduction in NPV of 5.6 % when comparing the REF and PEFC scenario, and the BD scenario had a 48.2 % lower NPV than the REF scenario. In Bergseng et al. (2011) the NPV of the forest was 5.8 % lower in the scenario that was similar to our PEFC scenario, and the NPV of the forest in the BD scenario was 43 % lower compared to the reference.

Timmermann and Dibdiakova (2014) have calculated the GHG emissions from forestry in East Norway, from seedling to factory gate. For a functional unit of 1 m^3 of wood delivered at industry gate, they report 17.893 kg CO_2-eq and almost half of the emissions are attributed to road transport (Timmermann and Dibdiakova 2014). In our study, the equivalent emissions are 17.5 kg CO_2-eq/m^3 in the PEFC scenario, TH=20. Timmermann and Dibdiakova (2014) have included construction and maintenance of forest road in their analysis. In addition they have a larger share of pulpwood (48 %) which is modelled with longer transport distances. Amongst the modelled value chains in our study, production of packaging and biorefinery products have the largest emissions, and this is consistent with earlier studies that show that these processes have large energy demands, and that the use of chemicals are important (González-García et al. 2011; Judl et al. 2011; Ghose and Chinga-Carrasco 2013). Due to modelling issues, wood used as energy at the biorefinery plant is not included and this leads to slightly overestimating environmental burdens for the biorefinery products. For ethanol, all emissions from the biological effluent plant are allocated to the production of ethanol and this can produce a higher impact in especially EP, AP, POFP. This allocation was done because of production related issues during the year of data collection and the producer of the bioethanol meant that this allocation best described the real situation. The other value chains in this study involves less processing, reflected by the lower environmental impacts. The greatest climate change mitigation benefit from wood use is when biomass is used for construction materials, which supports previous studies (Petersen and Solberg 2002; Petersen and Solberg 2005; Gustavsson et al. 2006). The emissions from production of construction materials are in the same range as cogeneration and district heating because of the relatively small energy demand during processing (Gustavsson and Joelsson 2010). The replacement of steel beams gives great GWP benefits as production of steel is energy-intensive and based on fossil energy, and several studies confirm that steel has greater impact on the environment than construction materials based on wood (Gustavsson and Joelsson 2010; Cabeza et al. 2014). The user phase and the end-of-life treatment were not included in the analysis, and those stages can be influential, especially regarding construction materials with a long expected life-time (Sandin et al. 2014). Steel constructions generally have lower maintenance demand than wood based construction material and assumptions about waste handling and recycling are important for the environmental profile of both steel and wood (Petersen and Solberg 2002).

Even though production emissions from the biorefinery are large, the net result shows that there are climate benefits tied to the use of biomass for these types of chemicals. The production mix that was constructed is important for the result in terms of potential environmental benefits per km^2. It gives a realistic description of the situation today, but the mix will change in the future. The waste handling of paper packaging from 1960 to 2005, indicates that the level of packaging based on paper has stabilized (Marsh and Bugusu 2007), and therefore it is a fair assumption that the share of packaging will stay at the same level as today. The biorefinery product has a positive influence on the net emissions of GHG per km^2 (that is, a greater CO_2-eq savings per km^2 than the product mix), using only spruce as raw material. In the BD-scenario, the share of spruce harvested is reduced in return for increased harvest of pine. This can contribute to a worsened result for the BD scenario regarding climate change mitigation. In addition to market forces at play, the production mix can be used as a political instrument for mitigation strategies. Because of the positive environmental benefits of wood use compared to other construction materials, the

Norwegian government wants to increase the use of wood in buildings, with public building projects moving in front (Landbruks- og matdepartementet 2011). During the Durban negotiations in 2011, countries agreed that carbon stored in harvested wood products can be included in the carbon accounting, and the country of origin will have climate benefits if the biomass is used for construction materials (Norwegian Ministry of Environment 2012). Thus, the use of wood for construction materials is expected to increase. In the EU, it is expected to increase by 35 % by 2020 (Landbruks- og matdepartementet 2011).

The choice of an area based FU was done because the aim of the study was to find the contribution, positive and negative, by the forest to a set of environmental impact categories. The amount of harvested biomass has shown to be important for the results, and both biomass yield and forest management are related to the geographical location of the forest. Using area as FU provides the opportunity to upscale the results in order to find the environmental impacts of forest products for a larger area; the forest owned by the municipality can be used as a proxy for the whole municipality and maybe for larger areas in the region. An advantage of this model is that the results can easily be related to other points of reference, like m^3 of wood. But because we used local data, general conclusions based on the results are uncertain. For example, the PEFC scenario had a small effect on the harvest and generally one would expect a certification scheme like that to impose larger restrictions. An analysis like this can be used as a fundament for a discussion about the effect of the forest certification scheme on biodiversity conservation.

We chose a public forest because according to Pulla et al. (2013), public forests play a key role in sustaining forest ecosystems and ensuring other management goals than just economic value. Municipalities are required to make climate and energy plans, and identification of current and future biomass resources can be a valuable tool for the local government when considering its possible contribution to climate change mitigation. The BD scenario was defined in cooperation with the local authorities, and because it is a public forest the municipality can decide on other management goals besides economic value. The local government possess knowledge of important recreational areas and can choose to take extensive biodiversity measures. In the model, the reason for restricting the management in one area, is not important, and by inviting the local government to define areas with restricted forest treatment, local important areas have been included in the analysis.

The choice of time horizon in LCA is a critical element. Conventionally, a time horizon of 100 years is used in the LCA calculations for GWP. There is in fact a temporal inconsistency between time horizon used in GWP and the life inventory results, assuming that the flux of emissions happens at time zero. This is not true for living products like trees, which can use decades to sequester the CO_2 emitted at a different point in time. The 20-year time frame for the forest model reflects the planning period for the municipalities' climate and energy work.

The production mix is important for the results in terms of GHG and other emissions savings per km^2, and it can produce some uncertainty as we do not know how the market for wood products will develop over the 20 years period. In the net emissions, assumptions on which product the biomass products replace are crucial. In this sort of analysis, the production technology and carbon balance of the substituted material is fundamental and it is important to find the product that the biomass product most likely will replace. If replacement is not realistic, the results are useless. In both energy production supply chains, the amount of heat and electricity produced depends on the moisture content of the wood and the energy efficiency of the plant. Assumptions about these factors can create some uncertainty.

This work has demonstrated that forest models can be combined with LCA in order to get a more holistic picture of the environmental impacts following exploitation of forests if the products substitute products based on fossil resources and energy. There are environmental benefits of using wood products instead of fossil products, but there is a negative relationship between the benefits related to emissions on one side and biological diversity and other landscape related impacts on the other side. Even though wood is a renewable resources, it is still a limited resource and there is competition over the biomass between different value chains. Several of the analyzed value chains use wood of similar quality. In order to find the products that can make the largest contribution to mitigation, a comparison between wood products, and between wood products and fossil products are necessary. The IPCC has pointed to bioenergy as an important factor for climate change mitigation, and both the EU and Norway has explicit goals of increasing the use of bioenergy. According to our model, the energy products are not the products that produce the greatest benefits. The political focus on bioenergy can lead to an increase in the bioenergy production at the expense of other products with greater climate change benefits. Cascading of wood is not included in our analysis, but other studies indicate that the use of wood for energy at the end of life, can further increase the environmental benefits if wood use (McKechnie et al. 2010).

The carbon cycle is complex, and in addition to the complexity of the bio-chemical processes that takes place in nature, the processes that takes place in the techno sphere adds further complexity to the total analysis of the life cycle emissions associated with production and use of products or services. This model includes the different processes in a systematic manner and provides a complete picture of the processes in the value chain. The model includes local adaptions to forest management and biological diversity. In the future, the model should include data for the other environmental impact categories for the biorefinery products that lacked this information. Further, this model should in the future include other climate change mitigation processes related to the forest in order to get a complete picture of the climate contributions by the forest. The model should include carbon storage, albedo and replacement in order to find the optimal use of the forest resources in a climate change mitigation context. In addition, future work should include analyses of up scaling of the model for larger geographical areas.

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Appendix

Appendix A

This appendix shows details about data and assumptions used for the LCA (Table A1)

Appendix B

This appendix shows detailed results for the forest model simulations and the calculations of dry matter output (Table B1), and for the production mix in the different scenarios (Table B2)

Table A1: Data, data sources and assumption for products, user phase and avoided energy and products.

Product from forest biomass supply chain	Specified product	User phase	Replaced product	Replaced amount	Data, data sources and assumption
Biological construction material	Sawn dried timber	N/A	Steel beams	60 kg steel replaced by 77 kg sawn dried timber.	Input data for sawn dried timber is 0,14 m^3 and density of dried sawn wood 550 kg/m^3. The composition of steel 50% virgin and 50% recycled. Dry matter content in sawn dried timber: 88%. Data from Petersen and Solberg (2002) and Lyng et al. (2010).
Cogenerated energy products	Electricity	N/A	Nordic electricity mix	1 kWh Nordic el mix replaced by 1 kWh electricity produced in CHP	Effective heat value of wood is calculated using the formula of Belbo and Gjølsjø (2008). Heat value of dry matter in wood of 5,28 kWh/kg DM and moisture content: 40%. Assumed combustion efficiency for heat and electricity is 80% (Brekke et al. 2008).
	Heat	N/A	Heat produced from light oil	1 kWh heat from light oil replaced by 1 kWh heat produced in CHP	
District heating	Heat	N/A	Heat produced from light oil	1 kWh heat from light oil replaced by 1 kWh heat produced from biomass	Effective heat value of wood is calculated using the formula of Belbo and Gjølsjø (2008). Heat value of dry matter in wood of 5,28 kWh/kg DM and moisture content: 35% Calculated combustion efficiency for heat production with vapour condensating is 110% (Hafslund 2007).
Packaging product	Carton board packaging	N/A	Plastic packaging (LDPE)	0.028 kg plastic packaging replaced by	Data for packaging for fish products.from: Møller and Schakenda (2012). Dry matter

				0.076 kg cardboard packaging	content for cardboard: 89.5% (Transport Information Service 2014)
Biorefinery	Speciality cellulose	N/A	Cotton linter	1 kg cotton linter replaced by 1 kg specialitycellulose	(Brekke et al. 2009; Modahl and Vold 2010; Ecoinvent Center 2013)
	Vanillin	N/A	Guaiacol	1 kg guaiacol replaced by 1 kg vanillin	(Brekke et al. 2009)
	Bioethanol (96%)	In passenger car	Ethanol from ethylene	1 kg ethylene replaced by 0,789 m^3 bioethanol	Density bioethanol: 0.789 kg/m3. Fuel consumption: 0.12746 l bioethanol/km (Brekke et al. 2009; Modahl and Vold 2010; Ecoinvent Center 2013).
	Lignin powder	N/A	Superplasticizers	1 kg superplasticizer replaced by 1 kg lignin powder	Brekke et al. (2009) based on original data from EFCA (2002). (Modahl and Vold 2010; Ecoinvent Center 2013).

Table B4: Simulated harvest in the different scenarios and time horizons. The total dry matter content harvested was used as input into the LCA model.

	REF		PEFC		BD	
	TH= 20	TH= 100	TH= 20	TH= 100	TH= 20	TH= 100
Harvested area (km^2)	1.88	9.56	1.75	9.06	1.20	5.83
Share of productive forest harvested	0.42	2.15	0.39	2.04	0.27	1.31
Share of sawn wood (%)	0.69	0.63	0.75	0.67	0.65	0.63
Stem wood output ('000 m^3)	42.50	212.48	39.08	195.39	14.33	71.62
Spruce ('000 m^3)	25.58	127.63	24.66	121.81	5.94	33.74
Pine ('000 m^3)	11.47	51.03	10.99	49.25	7.59	29.71
Birch ('000 m^3)	5.45	33.82	3.43	24.33	0.80	8.18

Output logging residue (ton DM)	4 459.10	22 002.31	4 101.77	20 206.81	1 422.16	7 137.77
Output stem wood ('000 m^3/km^2)	9.55	47.73	8.78	43.89	3.22	16.09
Spruce ('000 m^3/km^2)	5.74	28.67	5.54	27.36	1.33	7.58
Pine ('000 m^3/km^2)	2.58	11.46	2.47	11.06	1.71	6.67
Birch ('000 m^3/km^2)	1.22	7.60	0.77	5.46	0.18	1.84
Output DM stem wood (ton DM/km^2)	3 698.40	18 550.74	3 402.09	17 059.15	1 242.27	6 222.64
Spruce (ton DM/km^2)	2 231.67	11 197.16	2 149.80	10 678.21	508.88	2 928.23
Pine (ton DM/km^2)	996.42	4 429.03	956.47	4 280.25	664.84	2 588.65
Birch (ton DM/km^2)	470.30	2 924.55	295.82	2 100.69	68.56	705.76
Share Spruce DM/km^2	0.60	0.60	0.63	0.63	0.41	0.47
Share Pine DM/km^2	0.27	0.24	0.28	0.25	0.54	0.42
Share Birch DM/km^2	0.13	0.16	0.09	0.12	0.06	0.11
Logging residues (ton DM/km^2)	1 001.57	4 942.01	921.31	4 538.72	319.43	1 603.24
Total DM output (stem wood + logging residue, ton/km^2)	**4 699.97**	**23 492.75**	**4 323.40**	**21 597.86**	**1 561.71**	**7 825.88**

Table B2: The harvested biomass is distributed differently over the value chains depending on the composition of the harvest in the scenarios and time horizons. The limitations of the value chains, is the basis for the production mix.

TH	Scenario	Construction	CHP	DH	Packaging	Biorefinery
20	REF	0.18	0.24	0.24	0.13	0.21
	PEFC	0.19	0.23	0.23	0.12	0.23
	BD	0.20	0.25	0.25	0.15	0.12
100	REF	0.16	0.25	0.25	0.14	0.21
	PEFC	0.17	0.24	0.24	0.13	0.22
	BD	0.18	0.25	0.25	0.15	0.16

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PAPER IV

Including forest carbon, albedo and product substitution in harvest decisions – a case study for a forest management unit in Norway

Ellen Soldal, Even Bergseng, Per Kristian Rørstad and Birger Solberg

Abstract

This paper analyses how a profit maximizing forest owner would change harvest over time if forest income came from timber harvest and a tax/subsidy scheme that values the net climate change mitigation contribution of carbon sequestration and storage in forest, albedo, and product substitution. A forest bio-economic optimization model is combined with life cycle assessment. It is shown that harvest levels are strongly dependent on the importance given to albedo, substitution effects and how strongly carbon costs and benefits are discounted. The results also depend on a range of site-specific forestry factors and assumptions regarding the forestry value chain and forest owner behavior. As such, the study is explorative and one should be careful in drawing general conclusions based on this single case. However, the results provide insight into the absolute and relative harvest impacts of albedo and substitution assumptions in boreal forest management for climate change mitigation, and the importance of behavior assumptions in this kind of bio-economic modelling. The study also demonstrates how LCA and traditional forest optimization can be combined in order to get spatial and time specific analysis of climate change mitigation strategies in boreal forests

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1. Introduction

As climate change emerge as one of the greatest environmental challenges of our time (Rockstrom et al., 2009), the interest in biomass as a renewable source of energy and material increase. The ongoing climate change can be counteracted through reduced emissions of greenhouse gases (GHG) to the atmosphere, increased removal of carbon dioxide (CO_2) from the atmosphere or adaption to other climate drivers like albedo and evapotranspiration. Bonan (2008) names carbon flux and albedo as the two most important climate drivers in boreal forest, and in the following we will concentrate on these two mechanisms.

Albedo is the proportion of incoming solar radiation that is reflected by a surface. A change in vegetation cover can alter the albedo and, depending on the color and brightness of the surface, the change can have a positive (cooling) or negative (warming) effect on climate change. Several authors have pointed to the climate effect of changed albedo post-harvest, and the albedo effect should be included in analysis of climate impact of forest management in addition to GHG emissions (Betts, 2000, Gibbard et al., 2005, Bala et al., 2007, Betts et al., 2007, Bonan, 2008, Thompson et al., 2009, Schwaiger and Bird, 2010, Arora and Montenegro, 2011, Bright et al., 2011, Bright et al., 2012, Cherubini et al., 2012).

In their latest report, the International Panel on Climate Change (IPCC) emphasizes the importance of the part of the carbon cycle that takes place in forest, where CO_2 is sequestered, stored and emitted, and the forest has been appointed a leading role in international climate change mitigation strategies (Ciais et al., 2013). An example of the forest's role in climate change mitigation strategies is the United Nations Reducing emissions from deforestation and forest degradation (UN-REDD) program where developing countries are offered incentives to avoid deforestation and forest degradation (Allwood et al., 2014). Another example is the increase in bioenergy use that the European Union and Norway aim for (The European Parliament, 2009, Norwegian Ministry of Environment, 2012). These examples illustrate the two main climate change mitigation strategies that relates to forests (Colombo et al., 2012): Either, a) decrease atmospheric concentration of CO_2 by maximizing the carbon stock in forest through afforestation and protection from disturbances (human and/or natural disturbances), or b) harvest forest biomass in order to increase the carbon stock in the technosphere and substitute fossil resources.

Following the first strategy, i.e. forest conservation, the forest will continue to sequester and store carbon until it reaches saturation. Nabuurs et al. (2013) claim that we see the first signs that European forests are approaching carbon sink saturation. If this strategy is chosen, the demand for raw material that forest biomass would otherwise supply, must be covered by other materials and feedstock like fossil oil and gas from long-term lithospheric storages. In this case, the albedo effect will not contribute to a cooling of the atmosphere.

In the second strategy, emissions of biogenic CO_2 will contribute to climate warming while an increase in solar radiation reflection due to harvest will contribute to cooling of the atmosphere (Bonan, 2008). Similar to carbon storage in forest, carbon storage in the technosphere can reach saturation (Gustavsson et al., 2010). Contrary, substituting fossil resources with biomass is a continuous alternative (Smith et al., 2014), as it exploits the short- to medium-term carbon cycle instead of removal of fossil carbon from long-term underground storage. Cascading by re-using the wood products at their end-of-life may further increase climate benefits of wood utilization: For example, using wasted wood products as input in secondary wood productions (e.g. using wood from demolished constructions as fiber input in production of particleboard), and then burning secondary wood-based product for energy at their end-of-life.

The climate benefit of the boreal forest will be a trade-off between carbon storage, product substitution and albedo impacts. Climate benefit of changing albedo, carbon sequestration and storage are ecosystem services provided by the forest (Reid et al., 2005). In addition, forests provide a number of other ecosystem services, and optimal mitigation strategy will in most cases be somewhere in between the two extreme strategies mentioned above, in consideration of these other ecosystem aspects. Traditionally, timber production has been the goal of forest management because that is most profitable for forest owners (Pyörälä et al., 2012). However, from the mid 1970's other aims of forest management have been included and after the Millennium Ecosystem Assessment in 2005, there has been an increasing understanding of the importance of the other ecosystem services provided by forest (Reid et al., 2005). When payment for ecosystem services is introduced, private optimal rotation age can change (Olschewski and Benítez, 2010).

The selected mitigation strategy can be executed by putting a price on carbon or climate effects. Subsidies can encourage planting of trees and forest management that enhance tree growth, while a tax can discourage harvesting, depending on the mitigation strategy chosen (Kooten et al., 1995). Kooten et al. (1995) examined at a single forest stand level how the optimal rotation period would change when carbon uptake benefits were taken into account by valuing change in forest biomass with a carbon price. They did not consider the end-use of harvested timber, and found that when the value of carbon increased, the optimal rotation age increased. Hoen and Solberg (1994), using the GAYA-J forest model, analyzed optimal forest management at forestry level (i.e. for an area consisting of many single forest stands having different site classes, ages and tree species) how varying carbon prices influenced optimal rotation lengths as well as optimal choices of tree species and silvicultural intensities for all stands simultaneously. Their model included end-use of the harvested timber, so emissions of carbon over the whole life cycle were accounted for. The main results were that silvicultural intensities and optimal rotation lengths increased with increasing carbon prices. Raymer et al. (2011) applied the same model for analyzing optimal forest management at stand level for different carbon prices. Also in their analysis, the value of a forest site increased with the price of carbon, and timber revenue decreased as the carbon price increased. At medium site quality (G14), timber revenue was negative for a carbon price of 41 €/ton CO_2. Price and Willis (2011) found that optimal rotation length increased with increasing carbon price, but only to a certain level and was relatively stable around the age of maximum biomass productivity, regardless of carbon price.

Haus et al. (2014) compared cumulative radiative forcing of wood-based systems with a reference system. They concluded that the biomass system provided climate benefit, and intensification of forest management increased climate benefit. Kallio et al. (2013) examined the trade-off between the two mitigation strategies (sequester or substitute) for Finland by combining a market model, a forestry model and a soil model. Biomass was assumed utilized for energy replacing fossil fuels, in order to reach Finland's renewable energy targets. During the 20-year period of the assessment, increased harvests resulted in a decrease in forest carbon sink. This decrease was larger than avoided emissions due to replacement (Kallio et al., 2013). In Sweden, the relation between forest management, use of forest products and the carbon balance of the forest was investigated by combining models on forest growth, soil carbon, wood flow, carbon balance of wood products and substitution (Lundmark et al., 2014). The scenario with highest biomass yields from Swedish forestry provides the largest climate benefit, varying from 60 to 103 ton CO_2-eq/year (Lundmark et al., 2014). None of these studies included the effect of post-harvest change in albedo. Thompson et al. (2009) included both carbon and albedo in studies at single forest stand level, and found that valuation of albedo has the opposite effect of carbon valuation, i.e. including albedo leads to reduced optimal rotation age. Sjølie et al. (2013) obtained the same results, in a study covering the whole of Norway and utilizing a dynamic partial equilibrium forest sector model.

Life cycle assessment (LCA) is one of the most prevailing methods for analyzes of climate change effect of products and services. Nevertheless, traditional LCA is static and the use of biomass leads to temporal removal and addition of CO_2 to the atmosphere. Because LCA ignores the timing of emissions and removal, potential climate impact of this temporal change in atmospheric concentration of CO_2 is omitted (Cherubini et al., 2011, Brandão et al., 2013). Still, it is a valuable tool for environmental analysis, and when linked to a forest model the issues regarding time and space can be addressed. Combined with forest bio-economic modelling, LCA provides an opportunity to include albedo and other local factors like growth and natural mortality alongside with other ecosystem service and economic aspects.

Soldal et al. (2014) combined the bio-economic forest model GAYA-J with LCA in order to evaluate an optimized forest management for a forest property in the Southeastern part of Norway. They evaluated environmental effects of harvest, production and replacement under three forest management scenarios (reference scenario – no restrictions; forest certification scenario; and biodiversity conservation scenario). They found that forest biomass harvested, processed and used for substitution of non-wood products provided a net reduction of emissions of GHG. However, neither net flux of CO_2 in the forest nor the effects of changed albedo due to harvest were included.

Matthews et al. (2014) stress that biogenic CO_2 needs to be included in assessments of biomass use in order to make the results relevant for policy making and strategic planning, and Agostini et al. (2013) claim that "The assumption of biogenic carbon neutrality is not valid under policy relevant time horizons if carbon stock changes in the forest are not accounted for" (p. 18).

In this paper we explore how a profit maximizing forest owner, with a forest as described in Chapter 2.1, would change her harvest profile over time if the forest income came from timber harvest and a tax/subsidy scheme that values the climate mitigation contribution of forest growth, albedo, and product substitution, as defined in Chapter 2.2. The aim is to investigate the effects on forest management of internalizing costs and benefits related to climate change. This is done by valuing CO_2-flux in the same way as roundwood in the optimizations model's objective function. This mimics a tax/subsidy scheme where the forest owner is credited or debited for the net climate mitigation effects of both forest growth, forest harvest and changing albedo. According to Cubasch et al. (2013) policies to mitigate GHG emissions are "extremely complex and uncertain". Our tax/subsidy scheme represents a simplification of the allocation of climate change benefits within the forest sector, but can aid in investigating how wood producers/suppliers respond when costs and benefits of albedo, GHG emissions and sequestration of GHG are accounted on par with regular costs and benefits.

We combine a forest bio-economic model (GAYA-J) with LCA results and include in the analysis (i) carbon sequestration in forest biomass by forest growth, (ii) removal of carbon from the forest (iii) emissions of GHG from harvest operations and manufacturing of forest industry products (or bioenergy), (iv) avoided emissions of GHG by substitution due to replacement of non-wood products, and (v) changes in albedo due to biomass removal. Importance of climate contribution is included in the optimization through the magnitude of the assumed carbon prices.

2. Methodology

2.1. Study area

The forest modeling is based on a publicly owned forest property situated in the southeastern part of Norway (Fredrikstad municipality, 59°23'N 10°96'E, 0-120 m a.s.l). The local climate is dominated by proximity to the sea, and the middle winter temperature is -4 to -2°C, while average precipitation during winter (December-February) is 2.5 mm/day. The maximum depth of snow in the area is less than 50 cm, and the average number of days with snow cover is less than 150. Most days with snow cover have less than 25 cm snow depth (Norwegian Water Resources and Energy Directorate, 2013).

The total productive forest area included in the analysis is 4 454 hectares and is dominated by Norway spruce (*Picea abies* (L.) Karst.) and Scots pine (*Pinus sylvestris* L.) combined with some deciduous forest. The deciduous forest consists of mainly Birch (*Betula*). The forest was inventoried in 2006, and all simulations with GAYA are based on this inventory. Table 1 show forest area distributed on tree species and age classes.

Table 1: Forest area distributed on tree species and age class (20 years groups).

Age class (20 yrs.)	Spruce	Pine	Broad leaves	Total
1	9.2%	3.5%	1.2%	14.0%
2	13.3%	5.7%	1.2%	20.2%
3	9.4%	6.3%	3.4%	19.2%
4	10.0%	2.5%	2.3%	14.8%
5	2.3%	5.4%	0.9%	8.5%
6	1.5%	7.7%	0.1%	9.3%
7	0.4%	9.6%	0.0%	10.1%
8	0.0%	3.1%	0.1%	3.2%
9	0.0%	0.8%	0.0%	0.8%
Total	46.2%	44.6%	9.2%	100.0%

2.2. Forest model

GAYA-J is a dynamic, age-structured forest optimization model that bases the simulations on initial forest inventory and forest growth and mortality functions. The forest inventory provides information on the initial state of the forest stands (species, age and site index etc.) and GAYA simulates a range of feasible treatment schedules for each of the stand, including the option of no treatment. The simulation provides a wide matrix of possible treatments, and the optimization tool (J) optimizes treatment for the entire forest area assuming that the forest owner is a price-taking agent maximizing expected profit, represented by net present value (NPV), and subject to forest management constraints to secure environmental objectives besides climate mitigation. In this case, both timber revenue and climate (CO_2 + albedo) revenue contribute to the net present value.

The forest model produce period specific (5 year time periods) flux of carbon in and out of the forest. In this study, the forest owner is credited for carbon sequestration, cooling albedo effect after harvest and the net carbon effects of substitution of non-wood products, and debited for carbon emissions, according to assumed carbon prices. Different levels of carbon prices are tested, going from 0 NOK/ton CO_2–eq. up to 500 NOK/ton CO_2–eq. (1 €≈8 NOK). When wood is harvested, the forest owner is credited for avoided emissions due to substitution of non-wood products and for the cooling effect of albedo change. All carbon and albedo impacts except the forest growth are assumed to take place at the time of harvests.

GAYA-J estimates the harvesting income and costs of all forest operations, and NPV of timber harvest is calculated as discounted revenues from timber harvest minus costs of logging, forest transport and silvicultural activities for pre-specified timber prices and discount rates.

The time horizon of the analysis is in this study 50 years. Storage of carbon in wood products is not considered in this paper. We have used costs and timber prices for 2014, and 2 %, 3 % and 4 % p.a. real term discount rates. The analysis includes two alternatives regarding optimization objectives: one alternative where only timber costs and incomes are discounted, and one alternative where both timber and climate costs and benefits are discounted. After the 50 year optimization period, only the timber costs and incomes are included – i.e. climate mitigation benefits and costs are not included after 50 years.

The formal model is specified in Appendix 1.

2.3. Carbon

Carbon sequestration is assumed proportional with forest growth, and the forest owner is credited or debited for changes in standing stock. The amount of biomass is calculated using biomass expansion factors from Lehtonen (2004). Harvested wood contains 50 % carbon, and the amount of carbon is multiplied with the molecular weight ratio of CO_2 to carbon (44:12) in order to find the amount of CO_2.

Soldal et al. (2014) calculated net GHG-emissions for a range of wood products (GHG emissions from production of wood products minus the emissions from production of alternative products), and based on these numbers we calculated the GHG-savings due to replacement (CO_2-equivalents/m^3 harvested biomass). This was done for the three dominating tree species in the study area, namely spruce, pine and birch.

Five forestry value chains with replacement products are included in the analysis. Harvested biomass is allocated to different value chains based on each value chains specifications regarding wood dimensions and species. The harvested biomass is assumed used for production of heat in district heating (DH), heat and power in a combined heat and power plant (CHP), construction material, cardboard packaging and biochemicals in a biorefinery. The value chains and product mix are described in more detail in Soldal et al. (2014).

The products in the production mix have different GHG emissions, and ranking based on potential climate change mitigation impact are (from best to worst): (1) Construction material, (2) Biorefinery products, (3) District heating, (4) Combined heat and power, (5) Packaging (Soldal et al., 2014). We defined three scenarios based on this ranking: low, medium and high GHG-savings from wood use (Table 2). In the high scenario, wood is distributed to the products ranked with the highest climate change mitigation potential (29 % construction material, 41 % biorefinery and 29 % district heating). In the medium scenario, wood is distributed to the product mix defined in Soldal et al. (2014, see Appendix 2 for details), which is based on present market situation, assuming that future division among products is stable. In the low scenario, all wood is used for the lowest ranking product; namely packaging.

In Soldal et al. (2014), the final products provide potential climate benefits through substitution, so the net emissions of GHG per m^3 are all negative. Assumed net savings given in kg CO_2-equivalents per m^3 of Norway spruce, Scots pine and Birch harvested and processed through the forest value chain, are listed in Table 2.

2.4. Albedo

For Hedmark County in Norway, Bright et al. (2012) estimate an overall climate effect of albedo change after harvest, converted to CO_2-equivalents (GWP_{100}), of −3992, −4824 and −5656 g CO_2-eq./m^2 for high, medium and low site index, respectively. Albedo effect is converted to CO_2-equivalents by applying Equation 20 in Bright et al. (2012), to which we refer the reader for further information on the methodology. The Fredrikstad area experiences less snow than Hedmark, which implies a lower albedo effect. In Ås, Akershus County, with similar climatic conditions to Fredrikstad, there is an extensive registration of the local climate, and this is used as a proxy to calculate albedo for Fredrikstad Municipality. Using albedo functions from Bright et al. (2013) combined with the climate data for Ås (Thue-Hansen and Grimenes, 2010), we find a potential GWP_{100} of -790 g CO_2-eq./m^2 due to increased albedo after harvest. With a climate efficacy of albedo equal to 1.94 (Cherubini et al., 2012), this yields -15.3 ton CO_2-eq/ha. The climate efficacy of a climate forcing describes how effective the forcing agent is at altering the global surface temperature compared to CO_2. By definition, CO_2 has a climate efficacy of one (IPCC, 2013). The numbers from Bright et al. (2012) and our estimates based on meteorological data from Ås, form two levels of albedo temperature response (Table 2); low (climate data Ås) and medium (Bright et al., 2012), as described above. There is still uncertainty related to how large the temperature response of albedo is and how the climate impacts of albedo changes should be compared to a pulse CO_2 emission. For example, Sjølie at al. (2013) applied an albedo effect many times higher than Bright et al. (2012). The snow cover is also varying across the country. We therefore tested the effect of a higher level of albedo response by using an albedo impact ten times higher than used in Bright et al. (2012). Albedo effects for three levels of site index are included, because vegetation will return to pre-harvest levels faster on a high site index than a low site index, and this implies a smaller albedo effect. The exception for this is the low albedo scenario based on the function in Bright et al. (2013), where the functions are not differentiated according to site index.

Table 2: Potential net savings in ton CO_2-equivalents/m^3 harvested wood and CO_2-eq./ha under three levels of climate change mitigation contributions for substitution and albedo: low, medium and high. Low indicates the smallest contribution to climate change mitigation, medium indicates a mid-level contribution and high indicates a large mitigation contribution by the forest. H_{40}: site index system used in Norway, described by the height of dominant trees at breast-height age 40 years.

Scenario	Net substitution savings by replacement (CO_2-eq. tons/m^3)			Albedo effect (ton CO_2-eq. /ha)		
	Spruce	Pine	Broad leaves	$15.5 < H_{40}$	$9.5 < H_{40} < 15.5$	$H_{40} < 9.5$
Low	0.112	0.112	0.112	15.3[1]	15.3[1]	15.3[1]
Medium	0.639	0.595	0.339	39.92[2]	48.24[2]	56.56[2]
High	0.922	0.763	0.558	399.2[3]	482.4[3]	565.6[3]

[1] Based on meteorological data, Ås. [2] Bright et al. (2012). [3] Ten times higher than Bright et al. (2012).

Birch stands have a higher albedo than spruce and pine due to lighter colored canopy and the lack of leaves in winter when the ground is covered by snow (Bright et al. 2013). The share of birch is low in the area (<10%, Table 1), and therefore we have not differentiated the albedo values for the different species, as the dominating tree species spruce and pine stands have similar albedo (Bright et al. 2013). In general, we note that there is lack of knowledge on the size of albedo from different types of boreal forests. The albedo assumptions are shown in Table 2.

3. Results

Changes in carbon prices lead to changes in harvest levels. The effect varies with different carbon prices, albedo and substitution scenario and how strongly the climate impact benefits are discounted. In the following, we present the results for accumulated harvest, harvest over time at CO_2-price of 150 NOK/ton CO_2-eq. and net present value of timber harvest. All results are presented for different annual discount rates (2 %, 3 % and 4 %), and considering whether carbon impacts are discounted or not.

3.1. Accumulated harvest

Figure 1 shows total harvest for the first 50-year period for different combinations of substitution and albedo effects and as a function of the carbon price. To the left, the combined effect of carbon and albedo is not discounted, while income from roundwood is discounted with 2, 3 and 4 % p.a. real interest rates. To the right, both timber income and the combined effect of carbon and albedo are discounted with 2, 3 and 4 % p.a. real interest rates.

3.2. Harvest over time at 150 NOK/ton CO_2-eq.

Figure 2 shows the harvest per 5-year interval over the 50-year period at CO_2-price of 150 NOK/ton CO_2-eq. for different combination of substitution and albedo effects, compared to a baseline harvest assuming 0 NOK/ton CO_2-eq.. To the left, the combined effect of carbon and albedo is not discounted, while income from roundwood is discounted with 2, 3 and 4 % p.a. real interest rates. To the right, both timber income and the combined effect of carbon and albedo are discounted.

3.3. Net present value of timber income

Figure 3 shows the net present value of timber income for different combinations of substitution and albedo effects and as a function of the carbon price. To the left, the combined effect of carbon and albedo is not discounted, while income from roundwood is discounted with 2, 3 and 4 % p.a. real interest rates. To the right, both timber income and the combined effect of carbon and albedo are discounted with 2, 3 and 4 % p.a. real interest rates.

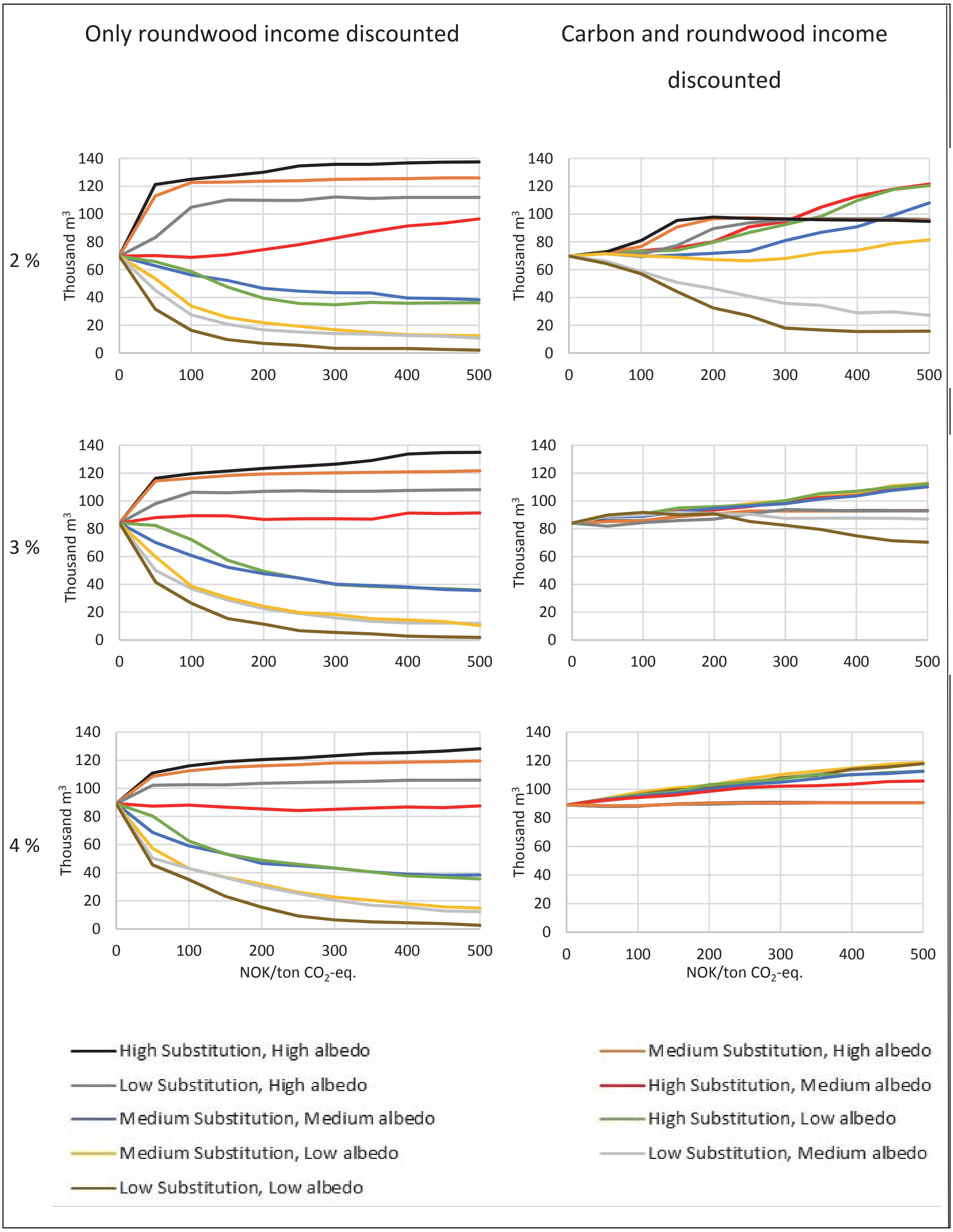

Figure 1: Summarized harvest (m^3) over the period (50 years) for different carbon prices (0-500 NOK/ton CO_2-eq.) and varying assumptions about the climate effect of substitution and albedo, at three levels of annual discount rates. To the left, only the roundwood income is discounted, while on the right hand side, both carbon and timber income are discounted. 1 €≈8 NOK.

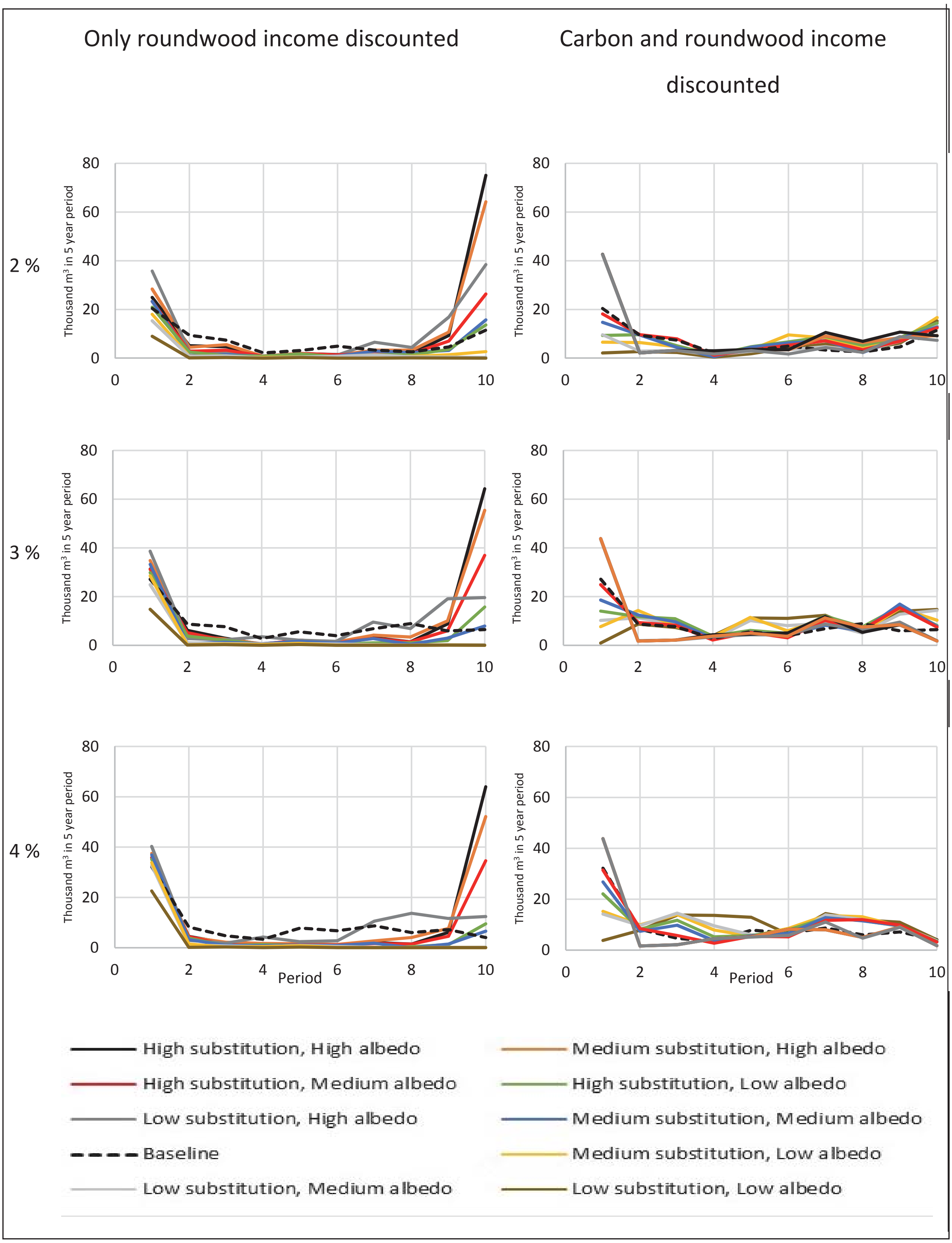

Figure 2: Harvest (m^3) per 5 year period for the various substitution and albedo scenarios for CO_2-price 150 NOK/ton CO_2, at three levels of annual discount rates. To the left, only the roundwood income is discounted, while on the right hand side, both carbon and timber income are discounted. 1 €≈8 NOK.

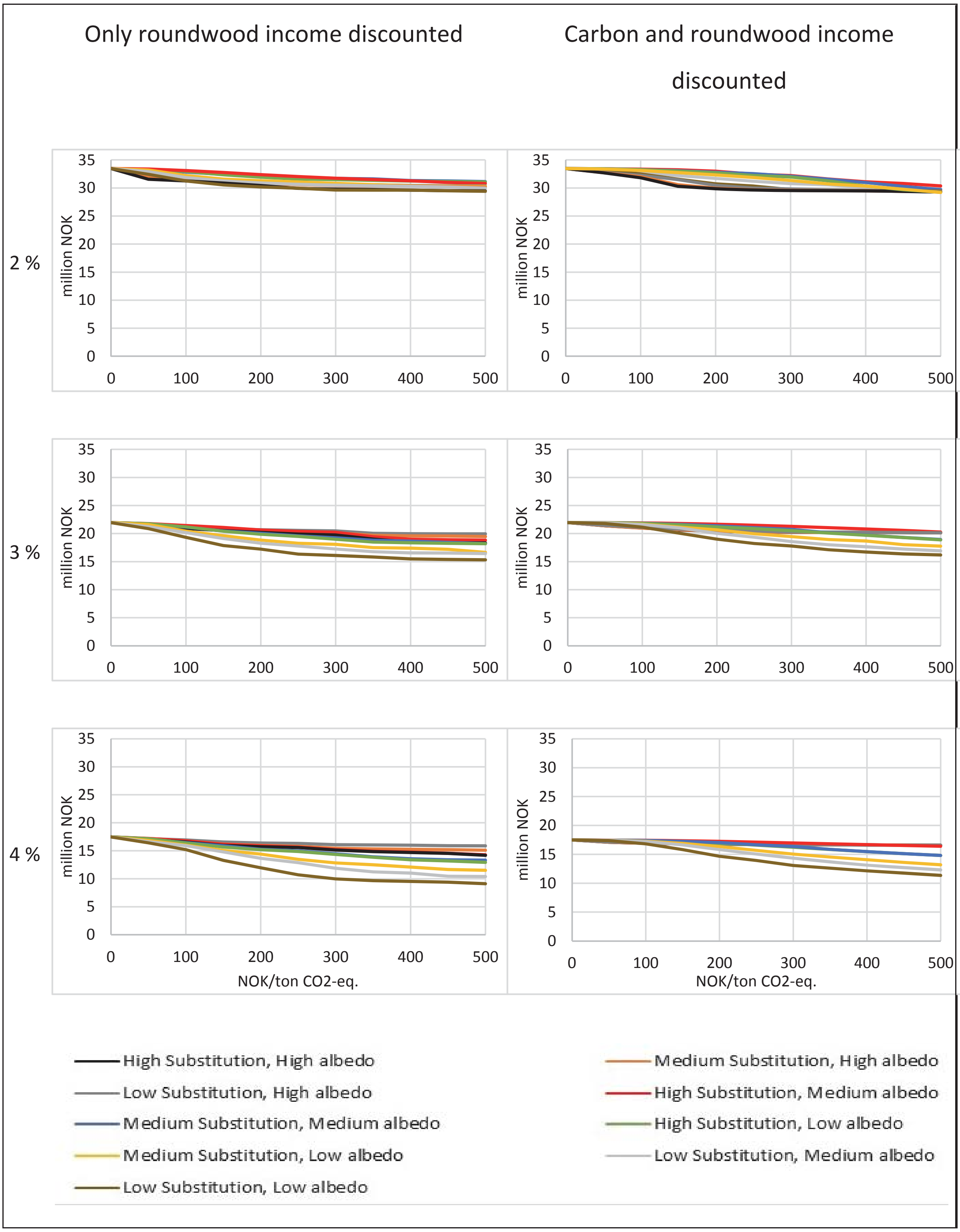

Figure 3: Net present value (million NOK) of timber income under different assumptions about albedo and substitution effects for different carbon prices and annual discount rates. To the left, only the roundwood income is discounted, while on the right hand side both carbon and timber income are discounted. 1 €≈8 NOK.

4. Discussion and conclusions

This study shows how a profit maximizing forest manager will adjust timber harvests if she receives – in addition to the usual timber income - a monetary value for climate effects of carbon sequestration, albedo and substitution. Figures 1 and 2 show that both albedo and substitution assumptions have strong impacts on harvest levels when the climate change contribution is given an economic value. The choice of discounting is also crucial for the simulated harvest pattern.

When only timber income is discounted, the largest change in total harvest levels is observed for carbon prices between 0 and 100. Further increases in CO_2-price produce diminishing change and harvest levels stabilize (Figure 1, left). The main reason for this is that with the assumed preference function (i.e. optimization objectives), the optimal strategy is to postpone the harvest as much as possible to the last period in order to gain both the albedo effect and the carbon sequestration effect caused by increased forest growth the first 50 years. These opportunities are taken even at a relatively low carbon price. At higher carbon prices the opportunities to obtain further climate benefits are limited. This explanation is supported by Figure 2 (left), where high harvests in the last period are clearly seen for the High and Medium albedo and substitution alternatives.

When all incomes are discounted (Figure 1 right) the total harvests become lower than the baseline harvest only for the alternatives “Low substitution Low albedo” and “Low substitution Medium albedo” at 2% and 3% p.a. discount rates. Otherwise the total harvest is always higher than in the baseline, where the carbon price is zero. The main reason for this is mentioned above - i.e. that when discounting also the climate effect, it becomes optimal to get the albedo and substitution effects as early as possible and is less preferable to harvest in later periods. This is also seen in Figure 2 (right) where the harvest is not increased in the last periods – i.e. in contrast to the harvests shown in Figure 2 (left). The results shown in Figure 2 (right) reflects that in the last period, the benefits from timber and climate mitigation are very low because of the discounting, and it is in fact more profitable for the forest owner to let more of the forest grow for later timber benefits than to harvest in period 45-50 years.

Figure 1 also shows that when all incomes are discounted, the accumulated harvest is less dispersed for the different assumptions about climate effect of substitution and albedo, compared to when only timber income is discounted. The main reason for this is that discounting also climate benefits, reduces the optimal possibilities compared to no discounting.

When all incomes are discounted, Figure 1 shows that the combinations with the largest total harvest includes both the Low and Medium albedo effects, contrary to when only timber income is discounted. At 2 % p.a. discount rate, the high albedo combinations produce larger changes in harvest levels at lower CO_2-prices than the medium and low albedo combinations.

Figure 2 shows that the baseline harvests during the first periods increases with increasing discount rates, in line with economic theory. This figure also shows that most of the scenarios have relatively large harvests in the first period, which mainly is caused by the forest having initially a relatively large share of mature stands. The low initial harvest shown in Figure 2 (right) at 3 % and 4 % p.a. discount rate are caused by combined effects of albedo, forest growth and substitution, which makes it profitable to postpone some of the harvest.

Figure 3 shows that timber income (NPV) declines with increasing carbon prices, and with decreasing substitution and albedo effects. As the CO_2-price increases, the income from climate change contribution becomes more important and dominates the NPV from timber revenues. High CO_2-price will favor forest management that increase the climate mitigation contribution. Management options for climate change mitigation are increasingly expensive as the incentives to reduce GHG emissions increase. This is in line with Hoen and Solberg (1994) who found that a large part of the increase in climate benefits were achieved by marginal cost of altered management. The largest reduction of NPV of timber revenues occurs for the Low substitution - Low albedo scenario for all analyses, except for discounting of all incomes at 2 % discount rate and carbon-price of 500 NOK/ton CO_2-eq. (Figure 3). In that case, it is the Medium substitution - Low albedo that gives the largest reduction in NPV of timber harvest. The largest percent reduction in NPV is 48 % (Figure 3 at 4 % p.a. discount rate, only timber income discounted). Raymer et al. (2009) reported a reduction in NPV of 21 % in the case where they maximized the carbon benefits assuming 2.5 % p.a. discount rate and no albedo effects.

In both of the two main compensation schemes (or objective functions) applied regarding discounting climate impacts, we assume that the forest owner is credited for product substitution. This simplification may lead to a bias toward harvesting relative to a compensation scheme where the forest owner is not credited for product replacement. However, if the forest industry was credited for product replacement, one could expect the willingness to pay for sawlogs to increase, resulting in higher sawlog prices. This could lead to increased harvest levels for a profit maximizing forest owner. In future studies with this model, this mechanism should be further analyzed by including the production process and the time effect of storing carbon in wood products, for example like in Hoen and Solberg (1994).

The potential GHG-savings due to replacement in this study vary from 112 kg CO_2-eq./m^3 to 922 kg CO_2-eq./m^3. Lundmark et al. (2014) reports avoided emissions of CO_2 equivalents varying from 466-719 kg CO_2-eq./m^3 of harvested wood. For the Swiss forestry sector, Werner et al. (2010) reports emissions savings of 600 and 700 kg CO_2-eq/m^3, depending on the end use. These numbers are in the same range as our medium substitution scenario.

When forests are not harvested, rotation ages increase and mortality increases. In this study, mortality is kept constant as a certain percentage of the number of standing trees, using mortality functions based on empirical data from relatively young forests (Braastad, 1982). The carbon accumulation in the low-harvest scenarios might therefore be overestimated.

Both albedo and carbon sequestration and storage are dependent on the climate. Depending on how future climate will change from today's situation, the forest growth and mortality may change, and alter the climate mitigation contribution by the forest. If the climate becomes warmer, the growth and yield may increase, but at the same time, the mortality may also increase because of pests and forest fires. The surface albedo is dependent on snow cover, and as the climate change, the snow cover may also change. Warmer climate could make evapotranspiration more important in boreal forest. In principle, the same model framework could be used to include potential future climate changes by including climate change dependent growth and mortality functions. However, because of the present uncertainty in how the climate will change and how these changes may effect forest growth and mortality, this has not been included in the modelling work in this paper. The albedo also varies with species, and deciduous forest has a higher albedo than coniferous forest. This means that when assuming the same albedo effect of harvest, the albedo contribution by deciduous forest will be overestimated.

We have not included soil carbon in the analysis because of large uncertainties regarding the magnitude of the soil carbon storage and how it develops after harvest. Monitoring of soil carbon stock in managed forests in Sweden shows that the stock is increasing and it is expected to continue to increase (Lundmark et al., 2014). Other authors conclude that the changes in soil carbon is small compared to the changes in the above-ground biomass (de Wit et al., 2006, Kallio et al., 2013). As a result, we may be underestimating the emissions of carbon during the first 10-20 years after harvest, but, if so, also underestimating the carbon sequestration after this period if we assume that the soil carbon stock is equal over each rotation period.

In our model, carbon leaves storage when wood is harvested, even though materials may be used for construction and, potentially, store carbon for decades after harvest. This is a conservative assumption that will not overestimate the climate contribution by forests.

The studied forest is a small forest, and carbon leakage impacts are not considered in this analysis. By carbon leakage impacts, we mean the carbon emission effects that occurs when a harvest change in one region lead to changed harvest in other regions. At the local level studied in this paper, the carbon leakage impacts will probably be marginal. However, for an upscaling of the study, to national level, carbon leakage should be included via a market model. It should also be emphasized that the analyses do not include forest fertilization and cascading of wood products. One should therefore be careful in using this study to draw conclusions to forest mitigation strategies at national (or global) level.

The choice of whether or not to discount the albedo and substitution effects is important for the results. There are arguments both for and against discounting of environmental impacts in the scientific literature, and the choice of discounting and discount rates is a value-laden choice (Hellweg et al. 2003, Levasseur et al. 2013). When climate benefits are discounted, future emissions are less important than sequestration in the near future. Our study clearly shows that the assumptions made regarding discounting influence strongly the harvest distribution over time.

By choosing different interest rates and discounting scenarios, we have used the model to show the implications of various forest owner behavioral assumptions. Raymer et al. (2011) discounted climate benefits, and got similar results as us regarding increased rotation periods, small changes in NPV of timber revenues, and increase in standing stock as the price of CO_2 increases. If 100 or 150 years instead of 50 years were chosen as length for the analysis period, the harvest and silvicultural investment over time would most likely be different, The impacts of different formulations of the objective function and longer analysis periods are of high interest to be investigated in future analyses.

Our results are in line with previous analyses in Norway of forest climate mitigation impacts. Hoen and Solberg (1994) and Raymer et al. (2009) assumed constant harvest level over time independent of carbon prices in order to eliminate carbon leakage impacts and highlight silvicultural impacts. They did not incorporate albedo, but included on the other side fertilization and soil impacts. Fertilization was found to increase the climate mitigation impacts considerably, even if the GHG emission in the production of fertilization was included. However, impacts on nitrogen sequestration was not included. The GHG impacts of soil inclusion were small. Sjølie et al. (2013) included albedo and had harvest as endogenous variable for the whole of Norway. Their main result was that incorporating albedo impacts increased harvest strongly in the first 10-years period when carbon prices increased.

The behavioral assumptions underlying the optimization model used in our study assumes that the forest owner is utility maximizing with perfect information, here represented by maximizing net present value of income as defined in Chapter 2.2. In real life, forest owners will not act as a perfect utility maximizer, as they lack full information and they may have other motivations than economic outcome. Many of these motivations could be included by assuming certain constraints as shown in Appendix 1. Further analyses, for example by introducing stronger constraint assumptions regarding forest management possibilities, would be interesting to explore.

Both the albedo effect and the net carbon flux differ depending on site index; sites with high index provide higher carbon benefits and lower albedo benefits, while the opposite is true for stands with low site index. A site-specific model like this can be used to improve the diverse management that is optimal for each site.

The results of this analysis are case specific as they depend on a range of case specific factors like initial state of the forest, productivity, effect of albedo, tree mortality, and the economic assumptions related to the forestry value chain, including assumed forest owner behavior. The generality of the results is thus uncertain. However, they provide information about the absolute and relative importance of albedo and substitution impacts, and as such improved knowledge for evaluating the full climate effects of forest management changes in boreal forests, and for choosing appropriate forest climate change mitigation strategies. Our study also demonstrates how LCA and a bio-economic optimization model like GAYA-J can be combined in order to get spatial and time specific analysis of forest climate change mitigation strategies.

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Appendix I: Model description

Formally, the optimization problem consists of the objective function [1], the area constraints [2], the non-negativity constraints [3], and the constraints covering possible environmental restrictions [4]:

[1] $$\max\ Z = \sum_{i=1}^{n} \sum_{j=1}^{J_i} NPV_{ij} \times w_{ij} + A + S$$

subject to

[2] $$\sum_{j=1}^{J_i} w_{ij} = 1, \forall i$$

[3] $w_{ij} \geq 0$ for all *i* and *j*

[4] $$\sum_{i=1}^{n} \sum_{j=1}^{J_i} of_{ijt} \times w_{ij} \geq OF \quad \forall\ t = 1,2,\ldots,T$$

where:

i = forest management unit *i* =1, *n*

j = treatment schedule $j=1, J_i$

t = time period *t*=1, T

NPV_{ij} = the net present value of management unit *i* if assigned treatment schedule *j* calculated as the discounted revenues from timber harvest subtracted costs for logging, off-road transportation, and silviculture. In the case of carbon sequestration, NPV_{ij} is the net present value of management unit *i* if assigned treatment schedule *j* calculated as the discounted revenues from the quantity of CO_2 (in tons) multiplied with the assumed economic value of net fixation of 1 ton of atmospheric CO_2, and subtracted silviculture costs. When both timber harvest and carbon sequestration in the forest are considered simultaneously, $\sum_{i=1}^{n}\sum_{j=1}^{J_i} NPV_{ij} \times w_{ij}$ equals the sum of these two objective function components.

w_{ij} = the weight (proportion) of management unit *i* assigned treatment schedule *j*

A = the economic value of the albedo impacts – i.e. the albedo impact in ton $CO_{2\text{-eq}}$/ha shown in Table 2 multiplied with the chosen CO_2 price. This value is discounted according to the assumptions made in Chapter 2 and is included in Z only for the first 50 years.

B = the economic value of the substitution impacts – i.e. the substitution impact in

ton $CO_{2\text{-eq}}$/m3 shown in Table 2 multiplied with the chosen CO_2 price. This value is discounted according to the assumptions made in Chapter 2 and is included in Z only for the first 50 years.$of_{i,j,t}$ = 0 if a management unit *i* does not satisfy a certain prespecified environmental requirements (e.g. regarding keeping old growth forest) in period *t* for treatment schedule *j*, otherwise equal to the area of the management unit

OF = the minimum required share of forest to fulfill a certain prespecified environmental requirements (e.g. regarding keeping a minimum area of old growth forest).

Appendix II: Production mix

Table A2 shows the division of harvest between the different products.

Table A2 The harvest divsion between the different products according to demands regarding species and quality. DH=district heating, CHP= combined heat and power.

Species	Logging residue	Sawn wood	Pulp wood
Spruce	DH (50 %), CHP (50 %)	Construction (25 %), DH (8 %), CHP (8 %), packaging (8 %), biorefinery (50 %)	Biorefinery (25 %), packaging (25 %), CHP (25 %), DH (25 %)
Pine	DH (50 %), CHP (50 %)	Construction (50 %), DH (17 %), CHP (17 %), packaging (17 %).	Packaging (33 %), CHP (33 %), DH (33 %)
Birch	DH (50 %), CHP (50 %)	-	Packaging (33 %), CHP (33 %), DH (33 %)

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Østfoldforskning

Stadion 4
NO-1671 Kråkerøy
+47 69 35 11 00
ostfoldforskning.no

Norwegian University
of Life Sciences

Postboks 5003
NO-1432 Ås, Norway
+47 67 23 00 00
www.nmbu.no