VISIONS • SCIENCE • TECHNOLOGY • RESEARCH HIGHLIGHTS

Dissertation
75

Evaluating land-use related environmental impacts of biomass value chains for decision-support

Comparison and testing of methodologies proposed for environmental life cycle assessment

Tuomas Helin

VTT

VTT SCIENCE 75

Evaluating land-use related environmental impacts of biomass value chains for decision-support

Comparison and testing of methodologiesproposed for environmental life cycle impact assessment

Tuomas Helin

VTT Technical Research Centre of Finland

Thesis for the degree of Doctor of of Science in Technology to be presented with due permission for public examination and criticism in lecture room 1382, at Lappeenranta University of Technology, on the Friday 6.3.2015 at 12:00.

ISBN 978-951-38-8208-2 (Soft back ed.)
ISBN 978-951-38-8209-9 (URL: http://www.vtt.fi/publications/index.jsp)

VTT Science 75

ISSN-L 2242-119X
ISSN 2242-119X (Print)
ISSN 2242-1203 (Online)

JULKAISIJA – UTGIVARE – PUBLISHER

VTT
PL 1000 (Tekniikantie 4 A, Espoo)
02044 VTT
Puh. 020 722 111, faksi 020 722 7001

VTT
PB 1000 (Teknikvägen 4 A, Esbo)
FI-02044 VTT
Tfn +358 20 722 111, telefax +358 20 722 7001

VTT Technical Research Centre of Finland
P.O. Box 1000 (Tekniikantie 4 A, Espoo)
FI-02044 VTT, Finland
Tel. +358 20 722 111, fax +358 20 722 7001

Grano Oy, Kuopio 2015

Preface

This thesis is the result of many interesting years of research work in research projects around the theme of environmental impacts of land use and their assessment with life cycle assessment methodology. I have been lucky enough to have been surrounded by very skilled and experienced colleagues all these years. You share the common enthusiastic attitude towards assessing and providing answers to the grand environmental problems and questions that are in focus in this thesis. Thank you to Tuomas Mattila and Riina Antikainen from Finnish Environment Institute SYKE and professor Risto Soukka from LUT for all the help in the first steep steps in learning how to write quality scientific articles. Special thanks to Sampo Soimakallio, Laura Sokka, Kim Pingoud, Marjukka Kujanpää, Tiina Pajula, Qianyu Li and Helena Wessman from VTT for the countless hours spent on intensive discussions, analyses and paper-writing sessions on climate impacts of forest bioenergy. This thesis would have never come to this point without your contribution, enthusiasm and research ideas. I would like to thank all the colleagues from VTT Sustainability Assessment and Energy Systems teams for all the support and all the fun in and outside the office on the course of last few years at VTT. Big thank you for Jari Hynynen, Hannu Salminen and Saija Huuskonen from Metla for all the collaboration. This thesis and the climate impact assessments within would not exist without your excellent work on forest modelling. Additionally, I would like to thank all those people from forest industry who have been partially funding and especially actively giving critical comments on our preliminary results in the research projects that made this thesis possible. All those discussions have taught me a lot on the necessity of clarity and continuous critical self-reflection in all the communication and dissemination of my scientific work. I thank Dr. Miguel Brandão and Dr. Assumpció Antón Vallejo for the valuable points raised in the pre-examination process that helped improve the quality of the thesis.

Of course the biggest gratitude goes to my lovely wife Tanja and our little son Joona-Aleksander for all the patience and support on the course of this lengthy dissertation process. It has required a lot of flexibility and patience from your side Tanja, especially in the last passing year with too many hours spent on finalising this thesis. Finally I want to thank the rest of my family, parents Lea and Tapio, sister Kirsi and all my friends for making life fun and worth living, and for supporting in building a world-view that respects the future generations' well-being, not only our own. This thesis is a tiny contribution in seeking that common goal.

In Helsinki, December 2014

Tuomas Helin

Academic dissertation

Supervising professor	Professor Risto Soukka School of Technology, LUT Energy Faculty, Laboratory of Life Cycle Modelling Lappeenranta University of Technology, Finland
Thesis advisor	Principal Scientist Sampo Soimakallio VTT Technical Research Centre of Finland
Reviewers	Dr Miguel Brandão Massey University, University of New Zealand Institute of Agriculture and Environment Private Bag 11-222 Palmerston North, New Zealand
	Dr. Assumpció Antón Vallejo IRTA, Institut of Research, Agriculture & Food Cabrils, Catalonia, Spain
Opponent	Dr. Jörg Schweinle Thünen Institute Institute of International Forestry and Forest Economics Hamburg-Bergedorf, Germany

List of publications

This thesis is based on the following original publications which are referred to in the text as Articles I–IV.The publications are reproduced with kind permission from the publishers.

I Mattila T, **Helin T**, Antikainen R (2012) Land use indicators in life cycle assessment—a case study on beer production. The International Journal of Life Cycle Assessment 17: 277–286.

II **Helin T**, Sokka L, Soimakallio S, Pingoud K, Pajula T (2013) Approaches for inclusion of forest carbon cycle in life cycle assessment—a review. Global Change Biology Bioenergy 5(5):475–486.

III **Helin T,** Holma A, Soimakallio S (2014) Is land use impact assessment in LCA applicable for forest biomass value chains? Findings from comparison of use of Scandinavian wood, agro-biomass and peat for energy. The International Journal of Life Cycle Assessment, 19, 770–785.

IV **Helin T,** Huuskonen S, Salminen H, Hynynen J, Soimakallio S, Pingoud K Global warming potentials of stemwood used for energy and materials in Southern Finland: Differentiation of impacts based on type of harvest and product lifetime. *Submitted manuscript.*

Author's contributions

The author was the corresponding author of all four articles included in this dissertation. More details are listed below for the author's contribution to the individual articles.

Article I. The author was responsible for the analysis of the impact indicators for soil quality and for carrying out the life cycle assessment case (LCA) study modelling. The analysis of the final results and the writing process of the article was a shared effort of all the three authors of the article.

Article II. The author of the thesis was the main author of the review article. The systematic review of selected literature was an equally shared effort by the author and Dr. Laura Sokka. All authors of the review article contributed equally to the identification and pre-screening of relevant literature, in formulating the review questions and in drawing the final conclusions. The author of this thesis was the main contributor on the analysis on reference situation, climate indicators and climate impacts of forest product use.

Article III. The author is the main author and responsible for the majority of the analysis and text in the article. Mrs. Holma contributed to the analysis of final results and discussion on biodiversity and Dr .Soimakallio gave valuable comments on the climate impacts and indicators. The rest of the contribution is from the author.

Article IV. The author is the main author and responsible for the majority of the analysis and text in the article. Dr. Soimakallio and Dr. Pingoud supported the main author in the definition of the research question and impact indicator metrics and commented on the paper. Dr. Salminen, Dr. Huuskonen and Dr. Hynynen are responsible for carrying out and writing the description of the forest growth modelling activities. The rest of the contribution is from the author.

Contents

1. Introduction

1.1 Background and the research environment

The rapid growth in both the global human population and the overall material well-being since the beginning of the industrial era has raised concern on the safe planetary boundaries on, for example, atmospheric greenhouse gas concentrations, rate of biodiversity loss and availability of resources such as energy carriers and productive land (MEA 2005; Rockström et al. 2009; IPCC 2013). Surpassing these ecological boundaries is likely to be in odds with the targets of sustainable development, that is, ensuring that the needs of today can be met without compromising the needs of future generations (WCED 1987). Some international treaties have been established in the international policy arena which state that there is a common need to mitigate these impacts (UNFCCC 1992; CBD 1993), and some propose targets for the levels of mitigating these impacts, such as Kyoto Protocol (UNFCCC 1997) and Copenhagen Accord (UNFCCC 2009) on limiting the atmospheric greenhouse gas concentrations and United Nations Millennium Development Goals (UNCSD 2012) to achieve significant reduction in the rate of biodiversity loss, among other jointly agreed targets.

The use of fossil fuels and minerals over the industrial era, and today, play a major role in the creation and persistence of the global environmental challenges of today. Bioeconomy, and the increased use of renewable energy, including bioenergy, have been considered as some of the possible means of mitigating human impacts on the environment, especially on climate (EC 2002; Directive 2009/28/EC; EC 2011). At the same time, anthropogenic land use has been identified to cause pressure to the environment (Foley et al. 2005). For example, past and ongoing clearing of the natural ecosystems and their continuous occupation and management for our purposes has had a significant contribution on the increase of greenhouse gas concentrations in the atmosphere (Houghton 2012). Transformation and occupation of vast land areas has been identified as the key contributor to high rates of biodiversity loss (MEA 2005; Rockström et al. 2009). Transformation and occupation of land areas for agriculture and grazing has been identified to cause degradation of soil in many areas (ISRIC & UNEP 1991; Oldeman 1992; MEA 2005). Land use interventions, thus, have been shown to have a significant contribution to the formulation of the global environmental challenges. Consequently, a discussion has arisen on the potential environmental impacts of increasing need for productive land area in a transformation towards bioeconomy (e.g. Searchinger et al. 2008; Havlík et al. 2011; Weiss et al. 2012; Bringezu et al. 2009; 2012; Pedroli et al. 2013; Immerzeel et al. 2014).

To manage the grand challenges of sustainable development, there is a need to be able to quantify the efficiency of the potential mitigation measures we plan to implement in the different levels of the society. Informed decisions need to be based on measurable, quantifiable criteria to secure that the mitigation targets can actually be met with the planned mitigation actions. Quantitative tools for environmental management and impact assessment can potentially provide the needed information. Industrial ecology (cf. Lifset 1997) is the field of research that aims to systematically examine local, regional and global materials and energy uses and flows in products, processes, industrial sectors and economies. Ness et al. (2007) have synthesised quantitative tools for environmental management and impact assessment under the term ‘tools for sustainability assessment’. They define the purpose of sustainability assessment as follows:

> “to provide decision-makers with an evaluation of global to local integrated nature-society systems in short and long term perspectives in order to assist them to determine which actions should or should not be taken in attempt to make society sustainable.” (Ness et al. 2007)

Ness et al. (2007) divide quantitative sustainability assessment tools into three main sub-categories (i) environmental indicators that allow measuring and tracking individual or integrated sustainability-related trends in retrospect, (ii) bottom-up product-related assessments that allow both retrospective and prospective assessment of environmental impacts of *specific goods and services* to support decision-making and (iii) top-down integrated assessment tools with aim to support decision-making related with policy or project in a *specific region* with a prospective temporal scope.

The tools of environmental management need to be effective in quantifying the actual environmental implications of different means of mitigation to support informed decisions in all levels of society. Life cycle assessment has been applied under the research area of industrial ecology in product-level environmental management (sub-category ii above) for more than 30 years (see Udo de Haes & Heijungs 2007 for an overview) and has proven to be an effective tool in identifying and comparing the environmental impacts that originate from the use and flow of fossil and mineral resources in different product systems. LCA is considered as the most established and well-developed tool in the product-related *environmental*[1] impact assessment perspective (Ness et al. 2007). A glance to the history of LCA (Udo de Haes & Heijungs 2007) shows that the methodology has been initially built around industrial systems to reflect the environmental impacts originating

---

[1] Not to be confused with overall sustainability assessment, which would also consider societal and economic aspects.

mainly from use of fossil and mineral resources and from the release of organic and inorganic compounds, i.e. emissions, from technosphere to ecosphere (cf. Figure 1, Schweinle et al. 2002; Heuvelmans et al. 2005). Nowadays, LCA methodology has been increasingly implemented for biomaterial and bioenergy value chains (see e.g. Cherubini & Strømman 2011; Cespi et al. 2013), product systems that likely influence land-use and management patterns, thus may have consequent environmental impacts. There is a need to augment the scope of LCA methodology, among other environmental management tools, to secure that land-use related environmental impacts are included and objectively described in product-related environmental impact assessment studies.

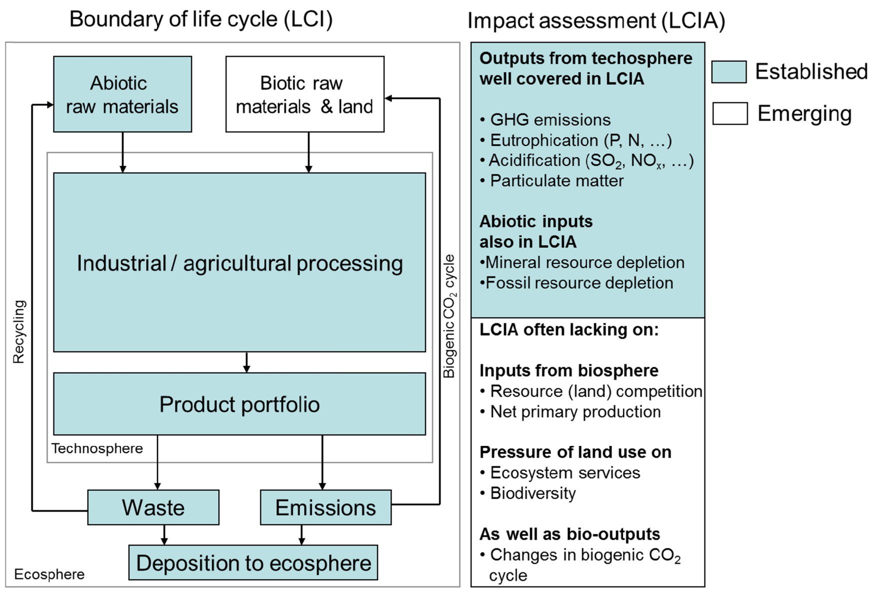

Figure 1. Illustration of techno-ecosphere interactions, system boundaries and impact assessment categories typically depicted and analysed with LCA methodology. Established life cycle inventory (LCI) and life cycle impact assessment (LCIA) items are illustrated with light blue colour and often missing, but emerging LCI and LCIA items are expressed with white background colour.

Initial approaches for including land use in LCA were made by Heijungs et al. (1992) by quantifying occupation of earth in m^2 with no distinction made between the different ways that the earth is used and no consideration given to the original state of the soil and ecosystem. Then, it was widely agreed that a qualitative evaluation of the soil quality changes is necessary for the activity considered. Many steps have been made since early 1990's and the recent work by UNEP-SETAC Life cycle initiative[2] has led to many developments in LCA. A framework on life cycle impact assessment (LCIA) for land use (Milà i Canals et al. 2007a) describes how to link land use interventions (land occupation and land transformation) to selected environmental midpoint indicators and damage categories. Three impact pathways for land use were defined: impacts on biodiversity, biotic production potential and ecological soil quality. Milà i Canals et al. (2007a) framework enabled consistent impact characterization of both land occupation (m^2a) and land transformation (m^2 from land use class to another) interventions with respect to dynamic reference land use situation. UNEP-SETAC Life cycle initiative updated the framework (Koellner et al. 2013a; Koellner & Geyer 2013) and the new framework includes, for example, an approach for bio-geographical differentiation of land-use impacts.

The UNEP-SETAC work has enabled the development of many midpoint and endpoint land use indicators for LCIA, for example Schmidt (2008) and de Baan et al. (2013a, b) on biodiversity, Brandão et al. (2010) and Brandão and Milà i Canals (2013) on soil quality and biotic production and Müller-Wenk and Brandão (2010) on climate regulation potential. Additionally, Ewing et al. (2010) and Haberl et al. (2007) have introduced independent ecological indicators that can be applied in LCA as midpoint indicators on competition over productive land and net primary production (NPP). These indicators, among others, have been operationalized in LCA case studies for example margarine production systems (Milà i Canals et al. 2013), based on the UNEP-SETAC framework on land use in LCA.

One of the most discussed environmental aspects of land use is its contribution to climate regulation and mitigation. Since the publication of widely cited studies that questioned the climate neutrality of use of agrobiofuels (Searchinger et al., 2008, 2009) and of forest bioenergy (Zanchi et al. 2010; Walker et al. 2010), a discussion has followed on the climate impacts of forest bioenergy both in the scientific literature (e.g. Lippke et al. 2011; Cherubini et al. 2011; Holtsmark 2012; Haberl et al. 2012a; Haberl et al. 2012b; Schulze et al. 2012a; Bright et al. 2012b; Lamers & Junginger 2013) and in the public media (BirdLife 2010; Miner 2010; Sedjo 2011; Mainville 2011; Cowie et al. 2013) with no evident consensus established on the related assessment methods nor the conclusions.

1.2 Objectives and scope

This dissertation focuses on the potential impact assessment methods, conceptual models and environmental indicators that have been proposed to be implemented into the LCA framework for the inclusion of land use related environmental impacts

[2] In 2002, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched an International Life Cycle Partnership, known as the Life Cycle Initiative, to enable users around the world to put life cycle thinking into effective practice. For more information: http://lcinitiative.unep.fr/

in the assessment of environmental impacts of product systems. Only a limited number of LCA case studies are available that have implemented and tested the land use impact assessment framework in LCA, thus there remained a need to critically test and analyse the methodology and give suggestions for future improvement needs from practitioners' perspective. The majority of LCA studies that include land use impact assessment have focused only on agricultural biomass product systems while there remains limited information on the applicability of the framework to forest biomass value chains, especially from resource, ecosystem service, climate, and biodiversity perspectives. Additionally, there existed only limited efforts to reflect and discuss the methodological considerations of land use in LCIA framework (Milà i Canals et al. 2007a; Koellner et al. 2013) in climate impact assessment of use of stemwood from managed forests. The application of land use in LCIA framework could potentially resolve some of the underlying methodological reasons that have led to the lack of consensus on climate impacts of bioenergy in scientific literature. Following these gaps in the existing research, the main research questions of this dissertation are:

- What is the applicability of existing land use impact indicators and impact assessment frameworks from LCA practitioners' perspective? Can they highlight meaningful differences in the environmental impacts of biomass value chains? More specifically, are the indicators and frameworks readily applicable for forest biomass value chains? (Articles I and III)
- How the land use impact assessment framework could be reflected in the assessment of climate impacts of the use of biomass from managed forests by considering the potential changes the value chain implies on the terrestrial carbon stocks? What could the global warming potential of use of stemwood from managed forests be from such a perspective? (Articles II and IV)
- What decision making situations the methodology present in this thesis can give support to and to which decision support situations are other modes of LCA required?

1.3 Research approach, process and dissertation structure

This dissertation is structured around four research articles. A schematic overview of the research process and the interrelationships between the individual research papers are presented in Figure 2. The scope and methodological approaches of the four research articles are summarized in Table 1 and discussed in the text below. Articles I–IV together form a research entity that, through critical review and testing, advances the inclusion of assessment of impacts of land use on climate change, land resource competition, ecosystem services and biodiversity in product LCA context.

Table 1. Summary of the scope and methodologies applied in the research articles I–IV.

	Article I	**Article II**	**Article III**	**Article IV**
Scope				
Forest biomass		x	x	x
Agricultural biomass	x		x	
Biodiversity	x		x	
Resource depletion	x		x	
Ecosystem services	x[a]		x	
Climate change [b]		x	x	x
Methodology				
Quantitative LCA study	x		x	
Literature review	x[c]	x		
Impact indicator development				x

[a] Limited to impacts on soil quality/productivity

[b] Climate regulation is among the ecosystem services that land provides, but this impact category is presented independently from the other ecosystem services, given its notable weight in the current environmental discourse.

[c] With a broad interpretation of literature review: Selection, summary and categorisation of a set land use impact indicators that the authors considered potentially applicable for LCA, based on a database search in the relevant scientific literature.

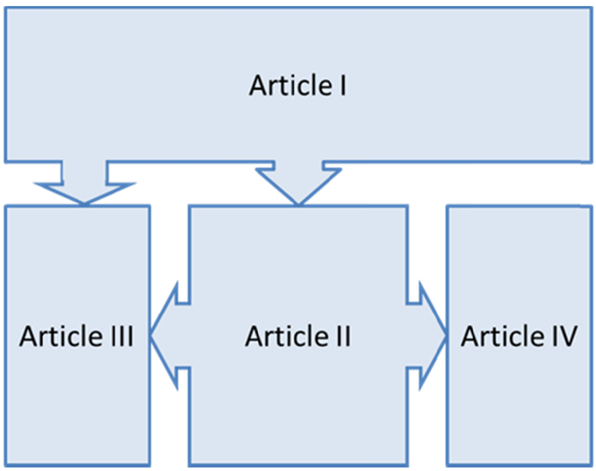

Figure 2. A schematic overview of the research process and the interrelationships between the individual research articles I-IV of this Thesis.

Article I is one of the first attempts to critically test the applicability of land use impact assessment framework (Mila i Canals et al. 2007a) and a broad set of land use impact indicators on several impact pathways in a comparative product LCA case study for agricultural biomass. Nine different land use impact indicators were selected based on a literature search, categorised into three different impact categories (resource depletion, soil quality and biodiversity) and their appliability was discussed in conjunction with a comparative LCA case study on beer and wine production. Several recommendations were given from LCA practitioners' perspective on applying and interpreting these land use impact indicators in the LCA context.

Article II includes an literature review and suggestions on how some key parameters and modelling decisions should be considered in future climate impact assessments of forest biomass utilisation in product LCA. The suggestions given in the Article II are based on the existing land use impact assessment framework and a review of existing literature on climate impact assessment of use of wood from managed forests. The aim was to understand the underlying methodological reasons for very different results and conclusions on climate impacts of forest bioenergy present in the existing literature, consider which methodological approaches are likely most suitable for product LCA, and to potentially help in harmonizing the impact assessment methods in future climate impact assessment studies. The suggestions given on how to deal with some key aspects in the future climate impact assessments are consistent with the land use impact assessment framework for LCA (tested in Articles I and III) and are based on the most suitable practices identified for product LCA context in the existing literature. Both Articles III and IV consider the methodological suggestions of Artice II in their methods of research.

Article III is the first LCA case study that critically tests the applicability of updated[3] land use impact assessment framework (Koellner et al. 2013) and a new set of land use impact indicators (cf. Koellner & Geyer 2013) for forest value chain. It expands the analysis made in Article I on land use impact assessment framework and impact indicators by considering the new framework and indicators and with the more detailed analysis on managed forest land use. Ten land use impact indicators were tested that aim to quantify the impacts of land use on climate regulation, resource depletion, several ecosystem services and biodiversity. They were applied in a comparative LCA study on energy use of stemwood, agricultural and peat biomass in Finland. Several research needs were identified to allow further application of the land use impact assessment framework for forest biomass value chains. Additionally, suggestions were given for further refinements of time considerations in the application of reference situation for land when the land use impact assessment framework is applied.

[3] Surrounding academic world, namely LULCIA project within the Phase 2 of the UNEP-SETAC Life Cycle Initiative, updated the land use impact assessment framework for LCA and proposed a new, selected set of land use indicators to be used for LCA (cf. Koellner & Geyer 2013) after the Article I was published.

Article IV focuses on the climate impact assessment of stemwood utilisation from managed forests. It is a direct application of the methodological suggestions given in Article II for climate impact assessment of forest biomass. It is the first research article that aims to differentiate the impacts of stemwood originating from either thinning activies or final fellings and helps in building understanding on the global warming impact of use of stemwood biomass for energy and long-lived wooden products.

2. Methodological foundation

2.1 Life cycle assessment

2.1.1 LCA and decision-support

LCA is the widely accepted methodology for the environmental impact assessment of products and systems (e.g. Finnveden et al. 2009). LCA addresses the environmental aspects and potential environmental impacts of product systems over their life cycle (ISO 14040:2006). LCA methodology comprises of four iterative phases: Goal and scope definition, life cycle inventory (LCI) modelling, life cycle impact assessment (LCIA) and, the interpretation of results (ISO 14040:2006, cf. Figure 4).

The initial goal and scope definition phase of LCA is of utmost importance. The intended application and audience, in other words the anticipated decision-support context of the study need to guide in the consistent definition of the research questions of the study. After all, LCA is generally described as a tool to support decision-making (Udo de Haes & Heijungs 2007; Ness et al. 2007; JRC-IES 2010; Plevin et al. 2013) either directly or indirectly (Tillman 2000; JRC-IES 2010). A decision-maker may be e.g. a policy maker, corporate manager or a consumer. The decision-context and respective research questions need to be carefully considered and defined, as these guide further modelling decisions in the latter phases of LCA methodology towards the ones that can give best support to the potential decision-making situation.

ILCD Handbook (JRC-IES 2010) lists three archetypes of decision-support contexts: (i) Micro-level decision support, (ii) meso to macro-level decision support and (iii) accounting with no decision-support relevance[4] . According to the division of JRC-IES (2010), in micro-level decisions the process-changes implied to sys-

[4] It can be questioned whether the third archetypal situation of JRC-IES (2010), accounting with no decision support, is a tool of product-related environmental management at all. It could potentially fall under the category of retrospective environmental indicators that allow measuring and tracking individual or integrated sustainability-related trends in retrospect (cf. three sub-categories of Ness et al. (2007) in the introduction of this dissertation).

tems external to the studied product system are considered insignificant or non-existent in magnitude. Examples of such could be decisions made by individual consumers or managers in small-to-medium sized organisations. For meso or macro-level decisions it should be anticipated that the decision is likely to have indirect (e.g. market-mediated) impacts to systems that are external to the studied product system. Examples of these could be political decision-making or strategic-decisions in large organisations. JRC-IES (2010) suggests that LCA modelling in micro-level decision-context should be carried out in isolation from external systems (so called ‘attributional approach’, see details below) and in meso-to-macro-level decision context the structural consequences to external systems shall be included in the assessment (so called ‘consequential approach’, see details below). Admittably, the differentiation of distinct decision-contexts is always a subjective choice, with no clear boundary when a decision is large enough to trigger structural changes in markets, thus implying consequences to external systems.

In a close relation with the issue of decision-support, the development of LCA has led to a division of different LCA types, out of which the most widely applied ones are attributional LCA (ALCA) and consequential LCA (CLCA) (Curran et al. 2005; Finnveden et al. 2009; Zamagni et al. 2012). ALCA approach has been defined as a method that describes the environmentally relevant physical flows of a past, current, or potential future product system as they occur (Curran et al. 2005). The CLCA approach has been defined as a method that aims to describe how environmentally relevant physical flows would have been or would be changed in response to possible decisions that would have been in the past or would be made in the future (Curran et al. 2005). In ALCA, average data depicting the actual physical flows are widely applied, as opposed to CLCA, in which marginal data are typically used when relevant for the purpose of assessing the consequences (Ekvall & Weidema 2004; Finnveden et al. 2009).

The results of an ALCA approach cannot answer questions related with, for example, impacts from changes in bioenergy production and land demand, and the CLCA approach cannot describe the impacts of unchanged, on-going land and biomass use activities. Thus the first step of the LCA methodology, concise and transparent definition of goal and scope, is of utmost importance. The decision-making context needs to be clear for the LCA practitioner to secure that the questions relevant for the decision-maker can actually be answered. Failure to identify the relevant question, and failure to distinguish the differences between the ALCA and CLCA approaches, can lead to an inappropriate method being applied, and misinterpretation of the results (Brander et al. 2009; Plevin et al. 2013). The selection of appropriate LCA mode, impact assessment metrics, and so forth is therefore essential when environmental impacts of land and biomass use are explored with the LCA methodology.

On the other hand, the division of LCA into two distinct taxonomic modes, ALCA and CLCA, has been challenged for not being unambiguous. For example, Suh & Yang (2014) argue that adopting a strict division of LCA modelling into two modes, and forcing studies into either category may hinder one from recognising relevant questions and potentially hampers a constructive dialog about the creative use of modelling frameworks. This discussion seems relevant for methodological questions around land use impact assessment and is reflected in the discussion of this thesis.

2.1.2 Land use in LCA

Initial approaches for including land use in LCA were made by Heijungs et al. (1992) by quantifying occupation of earth in m^2 with no distinction made between the different ways that the earth is used and no consideration given to the original state of the soil or ecosystem. Many steps have been made since early 1990's (see e.g. Schweinle et al. 2002; Alvarado et al. 2002; Wessman et al. 2003) and the recent work by UNEP-SETAC Life cycle initiative has led to many developments in land use impact assessment in LCA. Proposals for a framework on life cycle impact assessment (LCIA) for land use (Koellner & Scholz 2007; Milà i Canals et al. 2007a; Koellner et al. 2013) describe how to link land use interventions (land occupation and land transformation) to selected environmental midpoint indicators and damage categories. Some concerns have been raised during the process (e.g. Udo de Haes 2006) on whether the selected impacts fit and if all relevant impacts can ever fit into the methodological structure of LCA.

Milà i Canals et al. (2007a) presented an outline for combining the impacts of land occupation and transformation (Figure 3). Transformation changes the land quality, which may then restore towards the reference state in case no further occupation intervention is considered. The impact is then the integrated area between the reference state and the land quality development (blue area A_{trans} in Figure 3). The influence of occupation can be considered as a delay in the restoration process (green area A_{occ} in Figure 3). Over time, if the area had not been occupied, it would have initiated the natural regeneration process earlier in time. Therefore the impact of occupation can be seen as the time integrated loss of quality due to the delay.

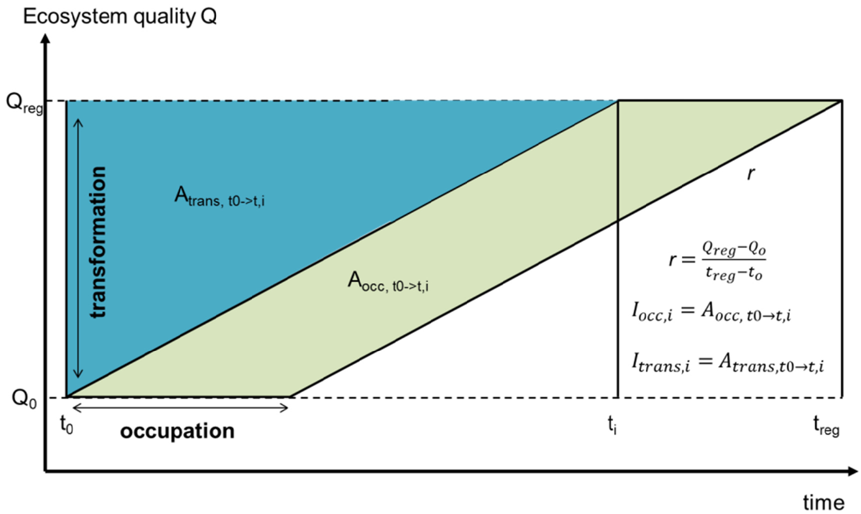

Figure 3. Impacts of land transformation and occupation interventions on ecosystem quality (Q) (redrawn based on Milà i Canals et al. 2007a, Figs 1 & 2). Blue area A_{trans} depicts the integrated impact of land transformation on Q over time and green area A_{occ} the integrated impact of land occupation.

The frameworks of LCIA for land use (Milà i Canals et al. 2007a; Koellner et al. 2013) include detailed discussion and suggestions on the modeling of so called “no use” reference situation, which is the evolution of the ecosystem quality of the land area in question *in the absence* of the studied activity. In ALCA, the “no use” reference situation is proposed to be the natural regeneration of the land area (Figure 3), while in CLCA, an alternative land-use situation would be applied as the reference. Alternative land use can be either natural vegetation or some human land use, depending on the situation in that specific area, and it can be derived, for example, from statistical time series or economic models (Milà i Canals et al. 2007a). JRC-IES (2010) gives guidance for modeling agro and forestry systems that are in line with Milà i Canals et al. (2007a) and Koellner et al. 2013:

> *“only the net interventions related to human land management activities shall be inventoried in LCI. Interventions that would occur also if the site was unused shall not be inventoried […] The "no use" reference system shall be the independent behaviour of the site, starting from the status of the land at that moment when the area of the analysed system is prepared for the modelled system” under ALCA modeling.*

Both natural regeneration and alternative land use can be considered as "the independent behaviour of the site," as stated in JRC-IES (2010), and the choice depends on the research question and the respective, most appropriate modeling approach selected (attributional or consequential).

This dissertation focuses on questions related with micro-level decision support, thus methods and modelling choices applied in the thesis (Articles I-IV) can most probably be considered to follow principles of attributional modelling, if such taxonomy is needed (cf. Suh & Yang 2014). Modelling of structural consequences implied by the studied product system to external systems through market-mechanisms is outside the scope of this thesis. Modelling of such market-mediated consequences could be carried out with economic partial equilibrium (PE) or general equilibrium (GE) models, such as GTAP[5] .

Figure 4 includes a general overview of the methodological steps and information needs of LCA case studies with focus on land-use impact assessment (Articles I and III).

[5] https://www.gtap.agecon.purdue.edu/

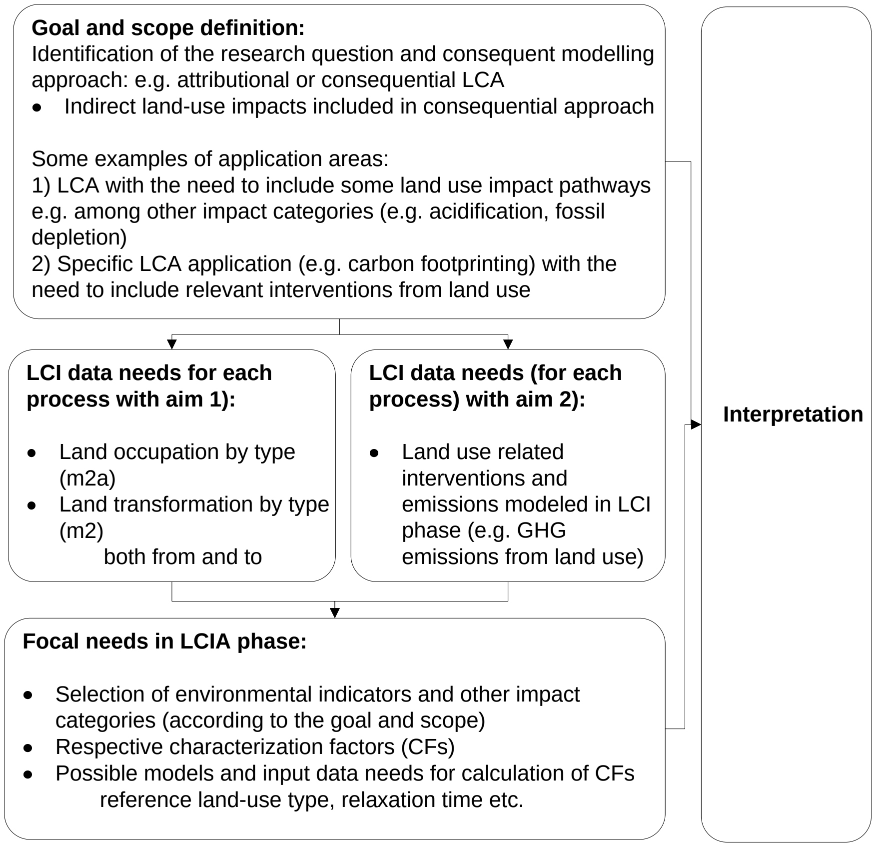

Figure 4. The four phases of LCA methodology (ISO 14040:2006) and examples of information needs in land-use impact assessment divided into distinct phases of LCA.

2.2 Global carbon cycle, land use and bio-based product systems

The atmosphere is a relatively small reservoir of carbon in the global carbon cycle (Figure 5, IPCC 2007). Anthropogenic activities, that is, fossil fuel combustion and cement production (244 GtC) and release of terrestrial carbon due to land transformations (140 GtC) over time period 1750–1994 have contributed to the increase of atmospheric CO_2 concentrations (IPCC 2007). The net increase of CO_2 (as GtC) over 1750-1995 was lower than the sum of release of CO_2 from 'anthropogenic' activities, approximately 165 GtC, as surface ocean and terrestrial biomass (including soil) function as natural carbon sinks. Between 2003 and 2013, terrestrial biomass (including soil) sequestered annually altogether over 25% of the carbon emitted from fossil fuel combustion and cement production, that is, functioning as a carbon sink of similar magnitude to the oceans (IGBP/GCP 2013). As the carbon emissions to the atmosphere and the sinks from the atmosphere are not in balance, the atmospheric carbon dioxide concentrations are growing.

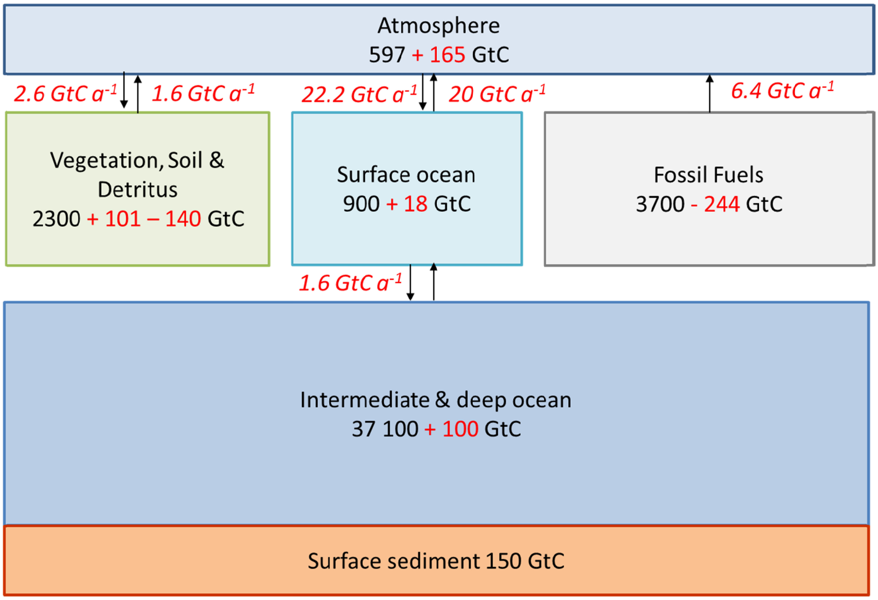

Figure 5. A simplified illustration of global carbon cycle for the 1990's (figure modified from Figure 7.3 in IPCC 2007, Chapter 7.3.1.1[6]). Pre-industrial 'natural' C reservoir sizes prior year 1750 in black and cumulative 'antropogenic' impacts on reservoir sizes over 1750–1994 time period in red. Reservoir sizes in GtC and fluxes in Gt Cyr^{-1}. Fluxes (red, italics) depict the annual 'anthropogenic' changes in reservoir sizes for the 1990's. Note the different timeframes in pool sizes (1750–1994) and fluxes (1990's). The net terrestrial loss of –39 GtC is inferred from cumulative fossil fuel emissions minus atmospheric increase minus ocean storage. The loss of –140 GtC from the 'vegetation, soil and detritus' compartment represents the cumulative emissions from land use change (Houghton 2003), and requires a terrestrial biosphere sink of 101 GtC. Please refer to the original figure in IPCC 2007 for a more complete illustration, including e.g. all natural C fluxes and discussion on uncertainties

Land use plays a significant role in the global carbon cycle. A loss of 140 GtC from the terrestrial C reservoirs (Figure 5, IPCC 2007) represents the cumulative emissions from land use change over years 1750–1994 (Houghton 2003). The terres-

---

[6] https://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch7s7-3.html

trial biosphere sink, 101 GtC in Figure 5, is not directly measured or observed, thus it is referred as ‘residual terrestrial sink’. It is determined indirectly from other terms of the global C budget as the sources and sinks of C must be in balance (IGBP/GCP 2013; Houghton 2012). This residual terrestrial sink does not include C sinks that result from management, only indirect and natural effects (Houghton 2012). The net terrestrial loss of –39 GtC over years 1750–1995 is thus inferred indirectly from cumulative fossil fuel emissions minus atmospheric increase minus ocean storage.

Land use influences atmospheric CO_2 concentrations through land transformation and land occupation interventions. Roughly 25% of the increased CO_2 concentration in the atmosphere is caused by land use change (40 GtC out of 165 GtC in Figure 5). Houghton (2012) has estimated that about 90% of the net release of carbon from terrestrial ecosystems to the atmosphere since 1850 has resulted from clearing (transformation) and management (occupation) of world’s forests. The differentiated contribution of forest clearing and forest management are described in Figure 6 (Houghton 2012). Forest clearing for agriculture and other anthropogenic land uses has decreased the forest area by circa one billion ha since 1850, releasing approximately 110 GtC to the atmosphere. The continued need to occupy this land for anthropogenic purposes prevents the natural regeneration (reforestation) process of these sites. Moreover, land areas remaining as forests have, on average, faced a decrease in their carbon density due to continuous harvesting of wood and other processes degrading forest (Houghton 2012). Wood harvesting and continuous forest management have released circa 40 GtC to the atmosphere, thus contributing to the increment in atmospheric CO_2 concentrations. Continued and on-going land occupation interventions for infrastructure, agriculture, grazing and managed forestry thus continue to postpone the natural regeneration process that would otherwise allow the average C densities to increase towards natural levels (cf. Luyssaert et al. 2008). Even if global net land transformations could be stopped today, the on-going land occupation activities continue to contribute to increased atmospheric CO_2 concentrations by affecting the net natural C sink capacity of terrestrial systems. Bio-based product systems are not an exception.

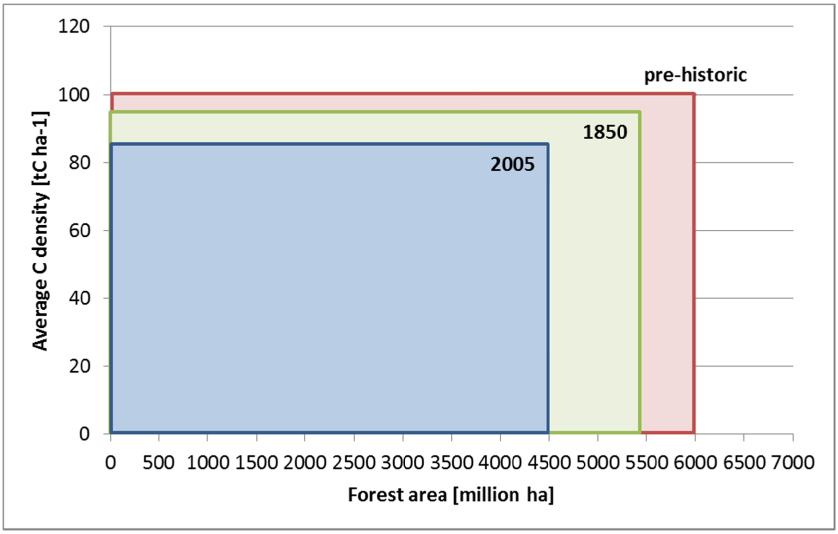

Figure 6. The area and average carbon density of the world's forests before human activity, in 1850, and in 2005 (redrawn from Houghton 2012).

Forests (including soils) are a significant pool of terrestrial carbon. They have been recognized to gain carbon in many areas (DeFries et al. 2002; Pan et al. 2011), with even an increased growth rate in some regions, for example in Finland (Kauppi et al. 2010). In Europe, the growth of forests was significantly higher compared to removals between 1990 and 2010, resulting in an annual average net carbon sink in the standing biomass (Köhl et al. 2011). The annual sink varies from year to year, mainly due to the intensity of removals. Thus, a change in land management (with no change in land-use class) contributes to a change in terrestrial C stocks. Nabuurs et al. (2013) have observed that in some areas of Western-Europe the carbon sink seems to be saturating, as these forests are likely reaching a dynamic equilibrium of C stocks with the current intensity of wood harvests. They find indications of sensitive balance between forest increment (growth) and intensity of harvests.

Handling of carbon emissions and sequestration of long rotation forest biomass remains as one of the controversial issues in LCA (e.g. Guinee et al. 2009). Traditionally, the use of sustainably grown forest biomass has been considered carbon neutral in LCA, based on the notion that the carbon that is released during combustion or decay of biomass was already sequestered to the growing biomass from the atmosphere, and, additinally will be sequestered back in future regrowth. Many biomass GHG accounting methods have been developed (e.g. IPCC 2006; WRI 2006; Eriksson et al. 2010; Pena et al. 2011; Cherubini et al. 2011; Lippke et al. 2011), resulting in notably different conclusions. A method of literature review is applied in Article II to discuss and identify which characteristics of the different approaches are most suitable for the objective treatment of forest carbon cycle and biomass carbon flows within LCA. The reviewed documents included 15 peer reviewed articles, five technical or other 'grey' reports, five guidance documents and Directive 2009/28/EC (Article II, supplementary material). The selected documents were reviewed through a set of 5 questions:

1. Does the approach apply some reference situation for land use? If it does, specify what.
2. Does the approach consider timing of emissions and sinks, and on what time horizons?
3. What indicators are used for measuring the GHG emissions or their warming impact?
4. Forest modeling: Does the approach consider the whole carbon stock (living biomass, dead biomass and soil) of the forest or only part of it? Is the approach based on the use of specific forest models or literature values?
5. Forest product use: Does the approach consider biomass carbon stored in a product? What about product substitution impacts?

The questions were chosen based on previous knowledge of the critical factors in the assessment of the forest biomass carbon flows.

2.3 Land use impact indicators for LCA

Land use impact indicators potentially applicable for LCA are discussed in this section. To be able to determine the environmental relevance of, for example, occupation of 1 ha of managed forest, cropland or mineral excavation site, assessment of differentiated impacts is needed. Impact indicators, that is, characterisation factors in the LCA-related terminology, are needed in the LCIA phase of the LCA methodology (cf. Figure 4). Several characterisation factors have been developed for LCIA for different land use classes and for different environmental mid- and endpoints and areas of protection (see e.g. review of indicators in Article I). Following the grouping proposed by Ridoutt et al. (2013) for midpoint land use impact characterisation, the impact indicators are first divided into resource-based indicators and ones indicating impacts on ecosystem services and biodiversity. Additionally, climate impact indicators are discussed independently from other ecosystem service impact indicators, due to the notable weight of climate change in the current environmental discourse. A resource-based approach to land use impact modelling considers that productive land is a scarce and limited resource (cf. Rockström et al. 2009) and that the production of almost any goods or services adds incrementally to the global demand for productive land, thus having an impact on loss of natural ecosystems (Ridoutt et al. 2013). In terms of driver–pressure–state–impact-response (DPSIR) framework (see e.g. Stanners et al.

2007), resource-based indicators would represent pressure on natural ecosystems, not direct environmental impacts. Impact indicators for ecosystem services and biodiversity can be considered representative of potential environmental impacts in terms of the DPSIR framework.

Land use impact indicators for LCA are part of a large entity of ecological indicators. In general, ecological indicators are used for assessing and reporting past trends and supporting future decision-making processes. Land use impact indicators in the LCA context can be considered the ones supporting decision-making processes. To be effective, an ecological indicator should provide relevant information about changes, be sensitive, be able to detect changes at the appropriate temporal and spatial scale, be based on well-understood and generally accepted conceptual models of the system, be based on reliable data that are available to assess trends and are collected in a relatively straightforward process, be based on data for which monitoring systems are in place and be easily understood by policymakers (MEA 2005). General criteria for effective ecological indicators should be reflected in the definition and selection of appropriate impact indicators in LCA.

2.3.1 Climate regulation

The scope of climate impact indicators in this thesis is on biogenic carbon flows only, although the author is aware of other potentially considerable climate impact pathways connected to land management; The direct surface albedo dynamics (e.g. Bright et al. 2012a) and the aerosol dynamics linked with indirect changes in cloud albedo (e.g. Spracklen et al. 2008, Paasonen et al. 2013) in forest land management and nitrous oxide (N_2O) emissions in agricultural land use (e.g. Crutzen et al. 2008; de Santi et al. 2008). Nitrous oxide emissions from fertilizing activities in agriculutural land use is not discussed in detail in as the main focus in climate impact assessment in this thesis is on managed forest lands. The surface and aerosol albedo impacts are left outside the scope of this thesis as their research is still in its early steps, thus significant uncertainties remain with the magnitude and even the sign of impacts (Bala et al. 2007; Spracklen et al. 2008; IPCC 2013. Technical Summary TS.3.4) and consequently, the scope of today's forest and climate policy is still limited to the impacts stemming from greenhouse gas emissions and carbon dynamics.

As LCA is typically applied as a static tool, in which the emissions are assumed to take place at the same time, it might not have traditionally been well suited to assessing the complexity of forest carbon dynamics (McKechnie et al. 2011; Bright et al. 2011). Some traditional approaches (e.g. Eriksson et al. 2010; Miner 2010) consider the annual average net change in the C stock of the regional forest system and ignore the timing of the sinks and emissions within the specific product system relative to a reference situation. Such static approaches likely reflect the guidelines for annual national greenhouse gas inventories in the agriculture, forestry, and other land use (AFOLU) sector (IPCC 2006), but may have limited applicability in decision-support with product-specific LCA. Therefore, Levasseur et al. (2010) have proposed a dynamic LCA approach for climate impact assessment, in which the temporal profile of emissions is included in the LCI results and time-dependent characterization factors are applied in the LCIA phase. The timing of emissions and sinks has an impact on the overall cumulative climatic impact of the activity over a certain timeframe (Levasseur et al. 2010, 2012; Moura-Costa & Wilson 2000; Fearnside et al. 2000). The carbon emission into the atmosphere has a warming impact (radiative forcing), whereas the sequestration has a cooling impact. The carbon debt between emission and sequestration results in a warming effect if sequestration lags emission. As a consequence, the result of the climate impact assessment is dependent on the time horizon of the assessment (Cherubini et al. 2011).

Müller-Wenk & Brandao (2010) and Kujanpää et al. (2010) have introduced climate indicators for land or forest biomass use that take timing of emissions and sinks into consideration and can be compared to a ton of fossil CO_2 emissions. Koellner et al. (2013) recommend applying the climate regulation potential (CRP) approach and respective characterisation factors proposed by Müller-Wenk and Brandão (2010) in climate impact assessment of land use. The drawback of these indicators is that they are only close approximations to the warming impact, because the fossil-combustion equivalent indicator (Müller-Wenk & Brandao 2010) is based on average literature data on terrestrial C stocks and the approach by Kujanpää et al. (2010) does not take the impact of CO_2 absorption by the top sea layer and other terrestrial carbon stocks into consideration. Moreover, Müller-Wenk and Brandão (2010) do not differentiate carbon stocks of natural and managed forests, out of which the latter in most areas have smaller carbon stocks (e.g. Luyssaert et al. 2008). Additionally, the characterisation factors present in Müller-Wenk and Brandão (2010) are modelled for a rather long timeframe (500 years) from the climate mitigation perspective, which potentially limits the relevance of the indicator results for today's decision-making purposes. The IPCC, for example, refrains from providing climate indicators for longer timeframes than 100 years in their latest fifth assessment report (IPCC 2013), due to the limited certainty in the state of future climate, thus in the numerical values of climate indicators in long time horizons.

Global warming potential (GWP) is a climate metric that has been initially introduced for climate policy purposes and is widely adopted in LCA for the comparison of climate impacts of different greenhouse gases. It is a relative climate metric that is based on the cumulative radiative forcing, i.e. energy imbalance of the climate system over specific timeframe, and is able to capture the temporal pattern of emissions and sinks and can be applied on multiple timeframes. The GWP coefficients of the individual greenhouse gases are different for different timeframes, because the atmospheric lifetimes and radiative efficiencies of distinct forcing agents differ from each other (IPCC 2007, Table 2.14). It has become a general practice within the LCA community to apply 100 year timeframe in LCIA.

The dynamic nature of the GWP coefficients makes it an interesting metric to be applied for climate impact assessment of activities that have an influence on the forest carbon cycle. A method for determining the GWP coefficients for energy use of long-rotation forest stemwood biomass, termed GWP_{bio}, was originally proposed in Cherubini et al. (2011). The GWP_{bio} factor was derived in this initial approach by approximating the atmospheric decay of carbon from long-rotation biomass with a simplified forest growth equation. Since then several studies have proposed additional modifications to the GWP_{bio} indicator by considering the climate impact relative to a no-use reference (baseline) situation (Pingoud et al., 2012), by the inclusion of the carbon dynamics of harvest residues (Guest et al., 2013), by considering the no-use reference situation and carbon dynamics of all carbon stocks in the forest (Holtsmark 2013) and by considering impacts of delayed release from long-lived products and/or product substitution (Cherubini et al. 2012; Pingoud et al. 2012). An advantage of the relative climate indicators such as GWP_{bio} and global temperature change potential (GTP) coefficients is that the climate impacts can be communicated in a unit familiar to LCA practitioners and the broad audience, fossil CO_2 equivalents (see IPCC 2013, Chapter 8.7 for more details on other climate metrics than GWP).

Direct climate impacts of stemwood use were modelled in Article IV in relation to no-use reference situation. Modelling of GWP_{bio} (Article IV) was initiated by the modelling of time-integral of radiative forcing (RF), also known as absolute global warming potential (AGWP) that is caused by the difference in forest carbon stocks due to the initial biomass harvesting (Pingoud et al., 2012). $AGWP_{bio}$ was defined as

$$AGWP_{bio} = \int_0^T RF\big(S_{bio}(t)\big)dt, \qquad \text{(Eq.1)}$$

where $S_{bio}(t)$ is the change in atmospheric CO_2 concentrations due to the difference in biomass carbon stocks in the harvesting and no-harvesting scenarios. To estimate the RF in time, an impulse response model REFUGE-3 (Pingoud *et al.,* 2012) which is based on the Bern Carbon Cycle Model 2.5CC (IPCC, 2007, p.213) was applied in Article IV.

AGWP for reference gas fossil CO_2 is formulated as

$$AGWP_{fos} = \int_0^T RF\left(S_{fos}(t)\right)dt, \qquad \text{(Eq.2)}$$

where $S_{fos}(t)$ is the atmospheric CO_2 concentration due to unit pulse of fossil CO_2 and RF is estimated with the same REFUGE-3 model (Pingoud et al. 2012).

Finally, a GWP_{bio} is defined as a ratio of AGWP for forest biomass over AGWP of reference gas, fossil CO_2 (Cherubini et al. 2011).

$$GWP_{bio}(T) = \frac{AGWP_{bio}(T)}{AGWP_{fos}(T)}. \tag{Eq.3}$$

The GWP_{bio} indicators include an inherent assumption that the carbon content in the harvested forest biomass is released to the atmosphere within the 1st year after harvest. In many harvested wood product value chains this assumption is counter-factual, and correction factors are needed in order to take impact of delayed release in long-lived products into consideration. Some previous estimates are present in the scientific literature, but they do not fully satisfy the need. Cherubini et al. (2012) includes GWP_{bio} factors that aggregate both results of their forest model and delayed release in product system into one value and Pingoud et al. (2012) include both substitution impacts and delayed emission in their GWP_{biouse} coefficients. New $GWP_{bio,product}$ correction factors were formulated in Article IV that allow transparent separation of impact of product storage stage (delayed release) from impacts of forest harvesting and/or product substitution. Modifying the approach present in Pingoud et al. (2012) by the exclusion of substitution impacts, $GWP_{bio,product}$ correction factor was defined as

$$GWP_{bio,product}(T) = \frac{AGWP_{bio,product}(T)}{AGWP_{fos}(T)} = \frac{\int_0^T RF\left(S_{seq}(t)\right) dt,}{\int_0^T RF\left(S_{fos}(t)\right) dt,} \tag{Eq.4}$$

where $S_{seq}(t)$ is the reduced CO_2 concentration due to delayed release (or permanent storage) of C in biomass products. $S_{seq}(t)$ is given a value -1 over the product lifetime τ (t: $0 \rightarrow \tau$) and instant release to the atmosphere is assumed in the end of product lifetime for simplicity. For forest biomass use with instant release to the atmosphere in $t = 0$, such as bioenergy, $S_{seq}(0 \rightarrow 100)$ equals 0, thus $GWP_{bio,product} = 0$.

A harvesting decision causes an instantaneous, observable change in the forest carbon stocks at the time of harvest, and the future evolution of carbon stocks can be modelled for both harvesting and no-harvesting scenarios. This difference in the evolution of future carbon stocks can be allocated to the wood products obtained in that specific harvest operation, in line with the methodological suggestions given in Milà i Canals et al. (2007a), JRC-IES (2010), Koellner et al. (2013), Article II. A method that aims for the isolation of impacts caused by the actual harvesting actions carried out for today's product systems, and at the same time focuses on the forest landscape level (a spatial boundary recommended by e.g. Sedjo 2011; Lamers & Junginger 2013; Jonker et al*.* 2013) was proposed and applied in Article IV. A forest growth model MOTTI (Hynynen et al. 2002; Matala et al. 2003; Salminen et al. 2005) was applied to be able to isolate the impact of wood harvested today from the impacts caused by past and future forest management activities. Two landscape level scenarios were constructed and compared: In the harvesting scenario the maximum level of annual harvests that still maintain the standing stock (and carbon stock) of the forest were carried out in the 1st year of the model run, in line with the principles of sustainable forest management. No further harvests were included in the years 2–100 in this harvesting scenario. In the no-harvest reference scenario no harvests occur over the 100 year modelling run. The comparison of the marginal difference in these two scenarios enables the isolation of the impact of harvested wood products of individual year from the impacts caused by wood products obtained in the past and future forest management activities.

Another approach would be to study a harvesting scenario in which the annual sustainable levels of harvests are carried out in every year over the modelling period, and compare this to a no-use, natural regeneration scenario. There would remain at least two drawbacks with such an approach: The time lag in between individual harvest and the following impacts on carbon pools is from decades to centuries. The impacts of a harvest carried out in, for example, year 95 of the modelling run would be fully covered only decades or a century after the 100 year modelling timeframe. On the other hand, from the perspective of impact assessment (LCA) of a product obtained today, allocating the impacts of potential future product systems to a product manufactured and used today does not seem relevant for the decision maker. Thus a method is proposed and applied in Article IV that aims for the isolation of impacts caused by the actual harvesting actions carried out for today's product systems, and at the same time focuses on the forest landscape level.

2.3.2 Resource (pressure) perspective

A resource-based approach to land use impact assessment views productive land as a scarce and limited resource (cf. Rockström et al. 2009) and that the production of any goods or services adds incrementally to the global demand for productive land, thus having an impact on loss of natural ecosystems (Ridoutt et al. 2013). Two indicators have been applied in this thesis from the resource or pressure perspective: ecological footprint (Ewing et al. 2010) and human appropriation of net primary production (HANPP) in Haberl et al. (2007).

Life cycle inventory results, land occupation (area × time) and land transformation (area of land converted from land use class to another) per studied functional unit can be considered as simple land use indicators from the pressure perspective. Goedkoop et al. (2008) have for example proposed to utilise these inventory results categorised into urban, agricultural and natural land within other midpoint LCA impact indicators. As there is no impact characterisation in the LCI results, that is, no consideration whether occupation of 1 hectare of mineral excavation site is more beneficial or detrimental than occupation of same area of organic cropland, thus the interpretation of environmental relevance of pure LCI results is challenging. Land use impact assessment methods that aim to differentiate the level of pressure with a characterisation model have been developed.

Ecological footprint indicator quantifies the demand that humans put on natural capital in terms of occupation of biological productive area (Wackernagel et al. 2002) and is measured in hectares normalised to the average productivity of all bioproductive hectares on Earth (Ewing et al. 2010). Ecological footprint has been operationalized in LCA for example in Huijbregts et al. (2008). Ecological footprint indicator is defined in Ewing et al. (2010) as the sum of direct land occupation (EF_{direct}) and indirect land occupation through the need for carbon uptake of fossil greenhouse gas emissions (EF_{CO2}). The EF_{direct} term of any studied system is measured in abstract units, global hectares, and is based on both the actual land area and the equivalence (weighting) factors of the bioproductivity of specific land use types occupied (see Ewing et al. 2010, Table 2). In this study, only EF_{direct} was considered to be relevant as resource indicator for products, following the notion made by Steen-Olsen et al. (2012) that the carbon uptake land can be considered to overlap with carbon footprint indicator. It should be stressed that EF_{direct} differs from life cycle inventory items such as 'agricultural land occupation' and 'urban land occupation' that have been proposed to be used as midpoint land use indicators without any characterisation (Goedkoop et al. 2008). EF_{direct} indicator differentiates the bioproductivity of distinct land use types. For example, the occupation of 1 ha of forest land is considered to put less pressure on the availability of productive land than the occupation of 1 ha of agricultural land.

Human appropriation of net primary production (HANPP) describes the difference in the free NPP left for ecosystems between the current land use and a reference natural state (Haberl et al. 2007). HANPP indicator highlights how much pressure we apply on ecosystems by indicating how large a share of NPP we appropriate for our uses. HANPP serves as a pressure indicator on the use of limited resource (bioproductivity). On average, mankind is using one fourth of the terrestrial NPP, the main surplus being in the tropical rainforests, in the boreal zone, and in western United States (Haberl et al. 2007). Additionally, the amount of free NPP left for the ecosystem has been found to correlate well with species diversity. HANPP indicator can be operationalised in product-level impact assessment (LCA) with the characterisation method and characterisation factors for specific land-use categories present in Mattila et al. (2011) and Article I.

2.3.3 Biodiversity and ecosystem services

A set of regionally differentiated LCIA impact indicators for several distinct ecosystem services and biodiversity has been published in a special issue of International Journal of Life Cycle Assessment, titled 'Global land use impacts on biodiversity and ecosystem services in LCA' (Koellner & Geyer 2013). They, and some other indicators of biodiversity are briefly presented here. Climate regulation potential indicator (Müller-Wenk & Brandão 2010) was presented in Section 2.3.1 together with other climate indicators.

Changes in soil organic carbon SOC and soil organic matter SOM has been proposed as an indicator for soil quality from future biomass production potential

(BPP) perspective for LCA (Milà i Canals et al. 2007b; Brandão & Milà i Canals 2013). It has been considered as suitable an indicator as SOC levels are often reported and are closely related to many other soil quality indicators, such as cation exchange capacity and soil life activity. Damage is modelled with BPP as the time-integrated difference in soil organic carbon (SOC) content between the studied land cover and a reference state. The SOC indicator does not cover all aspects of ecological soil quality: Soil erosion, compaction, build-up of toxic substances, acidification, salinization, and depletion of nutrients and ground water would need to be covered with other indicators in LCIA (Milà i Canals et al. 2007a).

LANCA (Beck et al. 2010) is a method for calculating several essential land functions (e.g., erosion resistance, filtration potential, and groundwater recharge) to determine the potential impacts of the studied activity on the ecological quality of land. Modelling of impacts with LANCA model can be based on detailed site-specific data if the impact modelling is applied in the LCI phase. To allow broader application Saad et al. (2013) have operationalised the method and published spatially-differentiated average characterisation factors for three different ecosystem services across 14 biomes: freshwater regulation, erosion regulation and water purification by both mechanical and physiochemical filtration.

Several indicators have been proposed for inclusion of impact of land use on biodiversity (see Articles I & III for details). Most recent UNEP-SETAC framework for land use impact assessment in LCA (Koellner et al. 2013) suggests the approach present in (de Baan et al. 2013a) to quantify land use impacts on biodiversity across different world regions applying species richness as the indicator. Species richness of different land use types is presented relative to the richness in reference land use, seminatural land cover (de Baan et al. 2013a). Potentially lost non-endemic species (PLNS) has been later proposed by the same authors as an indicator (de Baan et al. 2013b) that aims at describing the potential regional extinction of non-endemic species that are considered reversible. See Table 1 in Article I for short descriptions and analysis of some biodiversity indicators previously suggested to be applied in LCA.

3. Results

3.1 Practical applicability of resource depletion, ecosystem services and biodiversity indicators in LCA

Articles I and III both critically analysed the potential of land use impact assessment framework and several land use impact indicators. Their performance in highlighting the different impacts that land use in some selected biomass value chains may have on resource depletion, ecosystem services and biodiversity was in focus. Challenges in the practical application and in the interpretation of the results were discussed from a LCA practitioners' perspective. Both articles confirmed that land use impact indicators are applicable in LCIA and can highlight differences in impacts from distinct land use classes. Both concluded that results from impact assessment phase differ significantly from inventory results and impact characterisation in LCIA is necessary to highlight actual environmental relevance of different land uses. However, many open questions remain for both agricultural land use and especially occupation of managed forest land.

Article I included a comparative LCA case study on land use impacts of production of beer and wine. It was highlighted that the land use indicators that focus on an individual impact category, for example biodiversity, lead to consistent results on the relative environmental impact of one agricultural biomass product system over the other. It was found out that indicators exist and can be applied for all three studied impact categories of land use (resource depletion, ecosystem services and biodiversity), and that they all have positive features in highlighting actual impacts. However, limited certainty in the land use LCIA results remained due to several reasons.

LCI data on land occupation and transformation is consistently present only in one LCI database, thus cross-comparison and validation of inventory data remains difficult, if not impossible. This was found problematic, because some indicators on biodiversity impacts proved to be very sensitive to assumptions on land transformations far down the supply chain. There remain limited possibilities to validate that these transformations, and the respective environmental impacts, actually take place in the modelled value chain.

Additionally, none of the tested indicators fulfil the criteria for effective ecological indicators (see MEA 2005). Majority of them fall short in being able to detect changes in the appropriate temporal and spatial scale and not being very easy to comprehend by decision-makers. Moreover, the lack of reliable, regionally differentiated characterisation factors limited their credibility at the time of writing the Article I. There remained a need to compile such differentiated characterisation factors to increase the reliability and applicability of land use impact assessment of agro-biomass based value chains. Additionally it was concluded that no single indicator can describe all major environmental impact pathways of land use; all three impact pathways of land use need to be studied in LCIA with the respective land use indicators.

After the publication of Article I, an updated version of land use impact assessment framework (Koellner et al. 2013) and a set of regionally differentiated impact indicators for several distinct ecosystem services and biodiversity were published in Koellner & Geyer (2013), based on the work of LULCIA project within the Phase 2 of the UNEP-SETAC Life Cycle Initiative. It was found in the analysis present in Article III that the updated framework and these LCIA indicators could be applied in the comparison of solid bioenergy sources and that meaningful differences could be found in between distinct land use classes in the compared biomass value chains. However, caution needs to be applied in the interpretation of the results as limited certainty remains on whether the approach does highlight the actual land use impacts of managed forest value chains.

First reason for the limited certainty was that a disparity was observed in between the results of some applied LCIA indicators and the findings present in previous, non-LCA related literature on impacts of managed forestry on ecosystem services and biodiversity (cf. Raulund-Rasmussen et al. 2011). For example, according to Katzensteiner et al. (2011) managed forests provide the ecosystem service of provision of water in sufficient quality and quantity, while, conversely, the applied LCIA indicators on water related ecosystem services suggest most severe negative impacts in forest bioenergy value chains.

Additionally, challenges were faced in the inventory modelling for forest biomass: Should the forest land occupation be inventoried based on the actual average intensity of the forest biomass harvest in the region, or should a theoretical average annual growth of biomass be applied? This selection has major implications to further selections in characterisation modelling in the impact assessment phase and, thus, on the impact indicator results.

The characterisation factors, in their current state, do not differentiate varying management intensities, only distinct land use typologies (agriculture, forest, urban etc.). However, Scandinavian forest biomass value chains source their raw material from *managed* forests. The land occupation (m^2a) approach to inventory modelling is very challenging for wood, as the long-rotation biomass systems can be harvested with different intensities that almost never matches with the annual biomass growth or NPP on the site.

Moreover, the land use history of the site in question can, and most probably does, have a long-term impact on the characteristics and future evolution of the site in question. Attribution of land-use impacts to current use of a parcel of land, without considering the impact the past land uses implied, is problematic.

The methodological challenges that the Articles I and III highlight in land use impact assessment in LCA can be divided into two sub-categories: (i) Unresolved challenges connected with the whole land use inventory – impact characterisation – interpretation – decision support process within LCA framework and (ii) open questions related with the actual decision-support relevance of some individual impact indicators.

The challenges connected with the whole LCA process are potentially most difficult to resolve. As highlighted in Articles I and III, and previously by the majority of research articles that discuss the land use impact framework and indicators (e.g. Milá i Canals et al. 2007a; Koellner et al. 2013), the magnitude of environmental impacts of land occupation and transformation interventions are very case and site-specific. Conversely, the LCA methodology is, to a large extent, generalistic in terms of geographical scope, especially regarding the background systems in the studied product system.

Additionally, LCA methodology is constructed on a process where the system boundary definitions, assumptions and issues related with geographic scope and cut-offs (subjective value choices) in LCI modelling of a product system are decided by an LCA practitioner (cf. Figure 4). The time and monetary resource constraints that he or she has have an impact on these selections in inventory modelling.

LCIA phase, on the other hand, is based on existing characterisation models and previously published characterisation factors in the majority of LCA modelling exercises. The underlying subjective value choices in geographic representativeness, assumptions, cut-offs and timeframe considerations of characterisation factors are typically made by LCIA method developer, not the LCA practitioner who decides to apply some existing characterisation factors to apply in the LCIA phase of his or her study (cf. Figure 4). The resource constraints of LCIA indicator or method developer determine the extent that he or she can apply and publish geographically and temporally differentiated characterisation factors for the impact category in question.

At the same time, the resource constraints of an LCA practitioner determine the extent he or she can expand the LCA modelling into the field of modelling new, differentiated characterisation factors. In practice, the role of LCA practitioner is typically to focus on applying the modelling choices in LCI phase, but to rely on the academia on focusing on the modelling of characterisation factors needed in the LCIA phase. This means for land use modelling in LCA, in practice, that the LCA practitioners focus on collection and modelling of land occupation (m^2a), land transformation (m^2) interventions and geographic locations of individual unit processes in their LCI models. And in LCIA, the practitioners try to include the most representative published characterisation factors for different unit processes in the studied product system (Figure 4).

These resource constraints have led to the current situation where LCIA indicator and method developers have published generalistic, average level static characterisation factors available for some (large) geographic regions and gives the LCA practitioners suggestions to model case- and site-specific characterisation factors for the foreground system of the studied product system (cf. Koellner et al. 2013). LCA practitioners, in practice, rely almost entirely on LCIA method developers and the characterisation factors they can offer. This dilemma in land use impact assessment in the LCA context is very challenging, if not impossible, to overcome with the real-world resource constraints, thus discouraging broader application if not resolved (cf. Baitz et al. 2013).

The second sub-category of open questions is the actual decision-support relevance of aspects that some individual impact indicators actually measure. These are most probably easier to overcome with interaction in between LCIA method developers and LCA practitioners.

One example is the freshwater regulation potential indicator (FWPR, Saad et al. 2013), which is based on the groundwater recharge rate (millimeter groundwater recharged over time). The higher (faster) the recharge rate, the lower impact on freshwater regulation service. It is questionable whether the indicator highlights meaningful impact on water regulation ecosystem services specifically in boreal forests, which provide a water regulation ecosystem service of decreasing peak flow during rainy periods and increasing base flow during dry periods (Katzensteiner et al. 2011). It remains controversial whether the high FWRP indicator score for forest land signifies an increased or decreased impact on forest ecosystem services.

Similar examples are the water purification potential (WPP) indicators. They correctly indicate that WPP is similar for hectare of agricultural and managed forest land, and suggest the highest impact on WPP ecosystem service would occur in forest bioenergy chains with lowest yields (Article III). But is this indicator score actually relevant for supporting decision-making? The indicator does not consider whether there actually is a difference in the presence of unwanted compounds to be purified in forest and cropland soil, and consequently, a difference in the water quality of seepage water in the downstream water receptors.

Measuring impacts on biodiversity face similar difficulties from the viewpoint of decision maker. It is not clear to the decision-maker, nor the LCA analyst, whether the amount of species in general, the amount of *rare* species or the quality of habitats should be used as the basis of decision-making that aims to minimize impacts on biodiversity.

3.2 Climate impact assessment for stemwood from managed forests

Climate impacts assessments that have focused on the use of forest biomass for energy and long-lived products have yielded many, partially contradicting results on the direct climate impacts of forest management and use of forest biomass. Thus far, no evident consensus has been reached either on the assessment methods nor the conclusions on the climate implications of forest bioenergy.

Article II aimed to enhance the understanding on the underlying methodological reasons for the divergence in the previous results and included methodological suggestions for future studies. Both the land use impact assessment framework for LCA and existing climate impact assessment literature was reflected in the proposed method for product LCA. One of the main findings was that to capture the dynamic nature of forest carbon stocks, a reference situation for forest land use has to be defined appropriately, in line with the goal and scope of the study. The findings in the review (Article II) were that the reference situation should be natural regeneration of forest land in attributional LCA and alternative land use in consequential LCA. An illustrative example of the concept of selection of such dynamic 'no-use' reference for evolution of forest carbon stocks is presented in Figure 7. The studied activity in time t_0 has an impact on the forest carbon stocks, and more importantly, on the *potential evolution* of future forest carbon stocks. In attributional modelling approach for LCA, in which exogenous product systems, that is, alternative economic users of the studied parcel of land, are not considered. Thus the impact of studied activity on forest carbon stocks can be considered as the relative difference between evolution of the carbon stocks under the studied product system and under no economic use for the land (i.e. natural regeneration or relaxation) under attributional modelling. Under consequential modelling approach for LCA, in which impacts through exogenous product systems are included, an alternative economic user for the parcel of land is considered. In consequential modelling approach the impact of studied activity on forest carbon stocks can be considered as the relative difference between evolution of the carbon stocks under the studied product system and under alternative economic use for land. Different intensity of harvests is used as an example of an alternative use for the studied parcel of forest land in Figure 7.

It can also be appreciated that under the schemes requiring emission verification (e.g. Directive 2009/28/EC), the application of a virtual reference situation describing something that did not take place, is to some extent controversial. On the other hand, ignoring the 'no use' reference land-use situation results in conclusions that do not reflect the environmental impacts of the system studied.

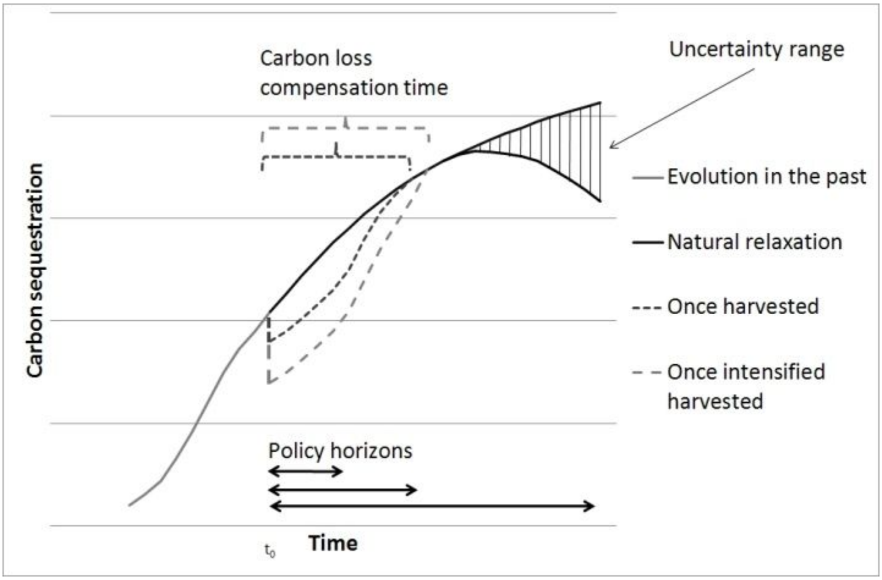

Figure 7. Illustrative example of evolution of carbon stocks in forest land of any spatial scope under various forest management options, including no harvesting and harvesting in t_o with different intensities. The relative difference in the future evolution of the forest carbon stocks in the presence of studied activity occurring in t_o and in the absence of the studied activity can be considered as the impact of the studied activity. Evolution in natural relaxation (regeneration) may be applied as the reference situation under attributional modelling approach in LCA.

Regarding the modeling of the evolution of carbon stocks in the studied forest system and the 'no use' reference situation, the use of dynamic forest models was found to be the recommended methodological approach (Article II). The forest models enable a more detailed analysis compared to the use of literature values, although more resources are needed from the LCA practitioner or the LCIA indicator developer. Changes in all the different forest C stocks, such as stemwood, branches, roots, litter and soil, need to be considered. Special attention should be paid to the consideration and transparent reporting of the uncertainties related to the modeling of future development of the biomass stocks (Articles II, IV).

Many reviewed studies (Article II) emphasize that there is no real scientific answer to what the studied timeframe should be, but that it depends on the (political) aims of the assessment (Schlamadinger et al. 1997; Kirkinen et al. 2008; Walker et al. 2010; Zanchi et al. 2010). As there is no scientifically correct timeframe, it was recommended that different timeframes should be considered. To be able to conduct the analysis under varying timeframes, a method that considers the timing of emissions and sinks, and that can be used within different timespans, seems most suitable. The indicator should take cumulative radiative forcing into account, but it is advisable that this is communicated relative to a pulse emission of one mass unit of fossil CO_2. In this way, dynamics in carbon stock changes will be included and the result can be communicated in a unit familiar to a broad audience (fossil CO_2 equivalent). The climate impact LCIA methods should allow flexible selection of case-specific inventory data on product use in the LCI phase, and biomass carbon stored in products should be taken into account. A climate indicator that is applied in LCA should not include any predefined assumptions regarding the substitution impacts of biomass use, because such an approach always includes uncertain and subjectively selected assumptions on what products would be substituted with biomass. Such impacts can be analyzed independently and separately with the selected climate impact indicator to ensure transparency.

To improve understanding and usability of the climate impact assessment results, it was concluded in Article II that it is crucial that the climate impacts of (1) forest biomass production and harvesting activities, (2) biomass carbon storage in long-lived products, and (3) product or energy substitution are considered independently and reported separately.

To operationalize the findings and suggestions of Article II, a method for climate impact assessment of stemwood use from either final fellings or different thinning operations was introduced in Article IV to study the potential global warming impact of use of stemwood from managed forests. Differentiated relative GWP coefficients were modelled for stemwood use from both final fellings and thinning operations that are potentially applicable in (attributional) LCA case studies, including $GWP_{bio,product}$ correction factors for long-lived wood products. The results of Article IV suggest that the climate impact of use of stemwood from commercial thinnings is lower than from stemwood from final fellings in the short term (some decades) but no significant difference was found in 60–100 year timespan (Figure 8). The results of Article IV suggest that the product lifetime has much higher relative influence on the climate impacts of wood-based value chains than the origin of stemwood either from thinnings or final fellings (Table 2). Additionally, the climate impact of energy use of stemwood from thinnings was found to be higher than the previous estimates on impacts of energy use of forest residues and stumps present in the scientific literature. Although the evolution of future C stocks in unmanaged boreal forests is uncertain, a sensitivity analysis suggests that the landscape-level model results on climate impacts are not sensitive to the assumptions made on the future evolution of C stocks in unmanaged forest (Figure 8).

Table 2. Relative GWP factors for timehorizons (TH) 20 a, 50 a and 100 a for (a) harvested stemwood from different harvest types, (b) $GWP_{bio,product}$ correction factors for selected carbon storage times and (c) Examples of net relative $GWP_{net\text{-}bio}$ factors for simplified cases. Min refers to minimum estimate and max to maximum estimate.

	TH =	20a	TH =	50a	TH =	100a
	min	max	min	max	min	max
(a) $GWP_{bio,forest}$						
First thinning	1.53	1.64	1.16	1.37	0.71	0.97
Commercial thinnings	0.86	0.92	0.75	0.90	0.51	0.73
Final fellings	1.15	1.24	1.13	1.36	0.67	0.90
All harvests	1.10	1.20	1.03	1.23	0.63	0.86
(b) $GWP_{bio,product}$						
Storage 20a	-1.0		-0.35		-0.17	
Storage 60a	-1.0		-1.0		-0.52	
Storage 100a	-1.0		-1.0		-1.0	
(c) GWP_{netbio}						
First thinning + instant combustion	1.53	1.63	1.16	1.37	0.71	0.97
Commercial thinning + 20a storage	-0.14	-0.08	0.40	0.55	0.34	0.56
Final felling + 100a storage	0.15	0.24	0.03	0.23	-0.33	-0.10

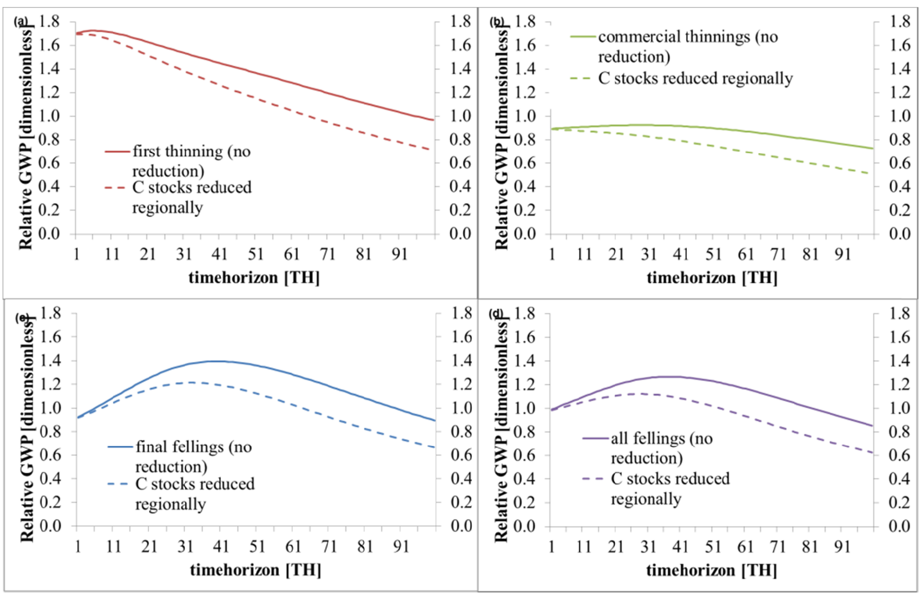

Figure 8. Relative GWP (GWP_{bio}) factors for stemwood from (a) first thinning (b) commercial thinnigs (c) final felling and (d) net for all harvests for timehorizons 1…100 a. Solid line depicts the high-end estimate and dashed line the low-end estimate from the sensitivity analysis for future evolution of forest C stocks.

The landscape level approach for climate impact assessment resulted in similar results with some stand-level approaches present in previous literature that included the same forest C pools and studied the impacts relative to no-harvest situation (Holtsmark 2013 in Table 3).

Table 3. comparison of estimates of GWP_{bio} for stemwood from final fellings in the previous literature and in Article IV. TH = time horizon. Note that only the values from previous literature that do not include the collection of forest residues are presented here to enable inter-model comparison.

	TH = 20a	TH = 50a	TH = 100a
Cherubini et al. 2011	1	n.a.	0.4
Pingoud et al. 2012	1	0.9	0.6
Guest et al. 2013	1.3	n.a.	0.6
Holtsmark 2013	1.6…1.9	>2	1.1…1.5
Soimakallio 2014	0.9…1.0	n.a.	0.2…0.7
Final fellings, Article IV	1.1…1.3	1.1…1.4	0.7…0.9

4. Discussion

4.1 Micro and macro-level decision support

An essential question regarding the validity and the practical applicability of the analysis and the results presented in this thesis is what decision making situations can these results give support to? And which decision support situations call for other approaches for modelling the potential environmental impacts.

The methods applied in this thesis are bottom-up modelling approaches and focus on studying existing (unchanged) product systems. It is important to remember that all models are only imperfect representations of the real world and the limitations in each modelling approach need to be identified in the interpretation of the results. The selected approach aims to:

i. Give information on environmental impacts endogenous to the studied product system, that is, excluding modelling of market mediated impacts. Latter are likely to occur, but cannot be described with this method.
ii. Give decision-support to micro- and meso-level decision making. Examples of such are the decisions made by private consumers and organisational managers in small-to-medium sized enterprises, where one may expect impacts to external systems to be limited or non-existent in magnitude.

The applied approach potentially allows comparisons of the magnitude of land-use induced environmental impacts of product systems with identical function. Examples are the comparison of beverages (Article I) and solid fuels for heat production (Article III). The results gained with the approach applied in the thesis cannot, however, be interpreted to depict actual changes in environmental interventions as a consequence of substitution of one product with another. Modelling of such changes would call for modelling framework that tries to include impacts exogenous to the studied product system through market-mediated impacts, including price elasticies, a modelling approach often referred as consequential in the LCA related literature. Such market-mechanisms are excluded in the modelling approach applied in this thesis, a limitation that should be considered if the results presented here were to be reflected in macro-level decision-making, such as ones made by policy-makers or strategic decision-makers in large international organisations. As discussed earlier on, the differentiation of distinct decision-contexts is always a subjective choice, with no clear boundary when a decision is large enough to trigger structural changes in the markets.

As the analysis in this thesis mainly aims to support micro or meso-level decisions, the applied approach is most probably considered to follow principles of so called attributional LCA, if such taxonomy is appreciated. There seems to be some potential disparities in the existing literature on whether a counterfactual baseline, that is, a reference situation for the land shall be considered in attributional approach for LCA (cf. Sections 2.1.1–2.1.2 of the thesis). Developments in the land use impact assessment framework within LCA have proposed an approach for traditional or attributional LCA that considers the natural regeneration of the land areas *directly* affected by the studied product system as the counterfactual 'no-use' reference baseline towards which the impacts of the studied system shall be assessed (e.g. Schweinle et al. 2002; Milà i Canals et al. 2007a; Koellner et al. 2013). The European ILCD Handbook guidance document for LCA (JRC-IES 2010) has adopted this suggestion for environmental impact modelling related with land use. On the other hand, the relevant literature on the developments around division of LCA approaches into two distinct taxonomic groups, attributional and consequential (Ekvall & Weidema 2004; Curran et al. 2005; Finnveden et al. 2009; Zamagni et al. 2012) imply that under an attributional modelling approach to LCA, no counterfactuals or "what ifs" would be considered. Following the latter subgroup of LCA related research, some have come into a conclusion that an attributional LCA does not aim to depict environmental impacts, only to attribute flows to different isolated product systems (e.g. Plevin et al. 2013).

In the veins of discussion in Suh & Yang (2014), the author of this dissertation is inclined to think that the driving force for modelling selections applied in an LCA study need to be mainly based on the goal definition phase of the study, not a predefined taxonomy for modelling. The goal of any quantitative environmental impact assessment with any modelling tool, including LCA, has to seek, by definition, to assess environmental impacts. The decision-context and a consequent, well defined research question should be the main driver for modelling decisions. An LCA practitioner should aim to answer a well-defined question as well as possible, not to fulfil a specific modelling approach, if the latter does not bring additional value nor enable identification of environmental impacts. If the division of LCA approaches into distinct taxonomic groups does not deliver clarity in a specific research context, then following one approach just for the sake of following one, does not seem very beneficial.

So what are the research questions one can be answering with the distinct possibilities for inclusion or exclusion of counterfactual baselines? There seems to be three potential approaches for the inclusion or exclusion of a counterfactual:

- Attributional LCA without any baseline for land use, that is, relating the land use interventions to a situation that prevails and is assumed to prevail in the future, or was the situation recently;
- Attributional LCA with 'no-economic use' baseline for land use, that is, natural regeneration towards natural potential vegetation; and
- Consequential LCA with alternative land use as the reference situation to our product system not occupying the land-area currently occupied.

Following the first option in line with the LCA literature that seems to imply that no counterfactual should be considered in an attributional approach, one is relating the studied activity to a situation that prevailed either before the introduction of the activity or, for example, last year. This change in the ecosystem quality indicator can be observed and measured and follows the approach of annual reporting for normative purposes such as reporting of national greenhouse gas balances for the Kyoto protocol. A concrete research question could be "*How did the environmental interventions change in comparison with the interventions that prevailed in this land area in a defined time t in the past?*". In an illustrative example from such a perspective, a parking lot that was a parking lot in the past, for example already last year, would have no impact through land use on the ecosystem service biotic productivity, because there was no biotic productivity on the site last year either. The results of such a research question are retrospective and do not try to identify relative *impacts* of the existence of the studied activity in relation to the absence of the studied activity. Thus one may argue that such an approach has limited use in decision-support.

Then following the second option in line with the land use related LCA literature that explicitly suggests that a 'no-use', natural regeneration reference situation for land use should be considered in an attributional approach, one is relating the existence of the studied activity to a counterfactual situation, the absence of the activity. The marginal difference in between the existence and the absence of the activity is considered as the endogenous impact of the studied activity. Such an approach leads to a concrete research question *"What are the direct land use related environmental impacts of the studied activity relative to the absence of the activity in isolation from other product systems potentially occupying the land?"*. This is in line with the research questions and modelling approaches applied in this thesis. Using the same example as above, the relative impact of an existing parking lot on biotic productivity would be the difference in evolution of the biotic productivity in continued use of the land as a parking lot versus the case of natural regeneration of the lot. Such a research question is future-oriented and tries to identify an isolated impact, thus implying support relevance for micro or meso-level decision-making which can be expected to have limited impacts to external systems. However, as the impact cannot be observed or measured due to the inclusion of counterfactuals in the analysis, thus there seems to be limited possibilities to apply results of such an approach in normative use which requires that the impacts can be verified. An example of such is the verification of emissions under the EU emission trading system. This has potential implications to the likely limited applicability of GWP_{bio} factors for forest bioenergy in the EU emission trading system.

Following the third option in line with all the relevant literature on consequential LCA, an alternative use for the land would be applied as the counterfactual reference situation under consequential modelling approach. The alternative would be the likely alternative economic use of the land that is determined by the markets. The identification of such impacts exogenous to the studied product system would require the modelling of market mediated responses with e.g. partial or general equilibrium economic models. The marginal difference in the evolution of ecosystem quality in the continued use in the studied product system and in the identified alternative economic use for land would be the impact of the studied product system. A concrete research question could be *"How would the ecosystem quality change as a consequence of a studied change in the demand of product X?"*. Using the same example of a parking lot, the impact on the biotic productivity would be the marginal difference in between the non-existent biotic productivity in the current land use versus the biotic production in the alternative land using system. The results of such a research question are prospective and try to depict the complete picture of impacts when the activity is studied in conjunction with the surrounding economic activities and markets for land. Modelling of impacts through market-mediated indirect land-use change (ILUC) would fall within such a consequential modelling framework. Such an research question aims to give support to macro-level decision making, which is outside the scope of this study.

Another issue regarding decision-support is whether to use midpoint or endpoint indicators in communicating the environmental impact assessment results to the interested audience. Midpoint level land-use impact indicators were applied in this thesis and no endpoint modelling was pursued. Endpoint indicators would have aimed to relate and reflect the weight of the studied individual environmental impact category results to the three environmental safeguard objects, namely human health, ecosystems and resources. An advantage of midpoint impact indicators is the smaller uncertainty and lower subjectivity in the indicator results, while a potential disadvantage is the lack of communication of relative environmental severeness of individual impacts in relation with (i) each other and (ii) other environmental aspects. The author considers that the decision-makers in the intended audience are allowed to (and will) apply their personal preferences and perceptions on the relative weight of individual decision-making criteria, that is, the weight and importance they want to give to individual environmental impact pathways in their decision making processes. The author is inclined to think that the decision-makers face, and are capable for, multicriterial decision-making in the presence of possible trade-offs in between individual decision-making criterion in their everyday life and work. The author's personal view is that they potentially rely more on non-aggregated environmental information in measurable units rather than aggregated results in non-physical units based on someone else's considerations on the weight of individual decision-making criterion in his or her decision making. If one perceives the conservation of biodiversity (or any other individual indicator or set of indicators) as the main decision-making criterion over the other environmental impact categories, he or she should be given the chance to find that information in the communication of the impact assessment results.

4.2 Considerations on baseline and timeframe

There remain a few considerations on the details in the application of the counterfactual ‘no-use’ baseline in relation with time and decision making that merit discussion. There seems to be a trade-off in between securing that all the far-reaching impacts are covered (long-term perspective) in assessment of impacts from land use and in maintaining sensitivity to short-term changes relevant to today’s decision-making purposes.

When one decides to apply the natural regeneration as the counterfactual reference level in micro or macro level decision support, one can apply either (i) the potential natural vegetation that was present in the area in the past or (ii) climax of potential natural vegetation to be reached potentially in distant future or (iii) consider regeneration *rate* over shorter timeframes. The author suggest to consider a forward-looking perspective in the determination of the reference situation, select an impact modelling timeframe that is relevant for today’s decision-making (e.g. 20, 50 or 100 years for climate impacts) and still respect the quasi-natural land cover as the reference situation (see Figure 3). To be able to give relevant support for decision-making, the LCA practitioner could select likely the most appropriate timeframe for impact characterisation in the goal definition stage and apply the characterisation factors modelled for both land occupation and transformation with the respective timeframe in the study (e.g. GWP-100 for climate impact assessment of land use). This can be carried out in full accordance with the proposed guidelines for land use impact assessment in LCA (Milà i Canals et al. 2007a; Koellner et al. 2013) if the reference situation is considered to be natural regeneration onwards from the current state, not the (quasi-)natural situation that was present in the history or can only be reached within centuries. If the most relevant impact modelling timeframe for the decision support in question is longer than the regeneration time, then the characterisation factors applied describe absolute distance to an idealistic natural vegetation state. This might be the case for some decision-making support situations for many of the land use impact pathways, but most probably is not relevant for decisions related e.g. with climate regulation and mitigation.

Articles II–IV applied a forward-looking baseline, focusing on the development of the ecological quality indicators from the present situation onwards. Some (e.g. Strauss 2011) argue in the context of climate impacts of forest value chains that the forward-looking view ignores the fact that forest investment decisions in recent decades were usually based on anticipation of wood use in the future. For example, Sedjo (2011) argues that past forest management, such as the planting of trees, was based on expectations of future use. Thus, from a broad forest-system perspective, burning biomass today would not release new carbon into the atmosphere, but would only emit carbon that was sequestered in the past in anticipation of this future use. Both perspectives can be appreciated, but the forward-looking approach seems to answer the most relevant questions from the 2 °C climate mitigation perspective. As mentioned earlier, there is no real scientific answer to what the studied timeframe should be, but that it depends on the (political) aims of the assessment (Schlamadinger et al. 1997; Kirkinen et al. 2008; Walker et al. 2010; Zanchi et al. 2010). It is possible that shorter time frames than 100 years will be emphasized in climate policy in the future, due to rapid emission reduction requirements in order to stabilize atmospheric greenhouse gas concentrations at a low level, for example in accordance with the 2 °C target, or to lower the temperature increase rate. For other impact categories, other timeframes may be preferred.

5. Conclusions

A significant challenge remains regarding land-use aspects and environmentally-aware decision making. The overarching aim of LCA modelling is to provide environmentally-relevant information to give support to decision making. Environmental impacts of land use in biomass value chains are likely of major significance, and can significantly influence whether a land-use intensive activity can help or hinder in meeting environmental mitigation targets.

The main aims of this thesis were (i) to test and analyse the applicability of land use impact indicators and assessment frameworks from LCA practitioners' perspective, especially for forest biomass value chains, (ii) to analyse how the land use impact assessment framework could be reflected in the climate impact assessments of use of forest biomass from managed forests and to compile respective characterisation factors potentially applicable for LCA, and (iii) to discuss which decision-making situations the approach and results of the thesis can give support to.

The omission of land-use impacts in quantitative modelling studies that aim to give objective information on environmental impacts, can, potentially, misinform the decision-maker. The exclusion of land use impact pathway in a quantitative environmental impact modelling with LCA seems to lead to giving incomplete environmental information to the decision maker, thus the driving aim for the LCA analysis is not met. If this then leads to the conclusion that land-use impacts need to be implemented in future LCA studies, then the core question on the remaining methodological challenges discussed in this thesis remains: Can we anticipate that the process-related difficulties in the assessment of land-use related environmental impacts in LCA modelling framework can be resolved?

It can be concluded that land use impact indicators are necessary in LCA in highlighting differences in impacts from distinct land use classes. However, many open questions remain on certainty of highlighting actual impacts of land use, especially regarding impacts of managed forest land use on biodiversity and ecosystem services such as water regulation and purification.

When the climate impacts of managed forestry are assessed, a reference situation for forest land use has to be defined appropriately, in line with the goal and scope of the study. Ignoring the 'no use' reference land-use situation results in conclusions that do not reflect the environmental impacts of the system studied.

The climate impact of energy use of boreal stemwood was found to be higher in the short term and lower in the long-term in comparison with fossil fuels that emit identical amount of CO_2 in combustion, due to changes implied to forest C stocks. The climate impact of energy use of stemwood from thinnings was found to be higher than the previous estimates on impacts of energy use of forest residues and stumps present in the scientific literature. The product lifetime has much higher relative influence on the climate impacts of wood-based value chains than the origin of stemwood either from thinnings or final fellings. Climate neutrality seems to be likely only in the case when almost all the carbon of harvested wood is stored in long-lived wooden products.

Bottom-up modelling approaches were adopted in this thesis which focuses on studying existing (unchanged) product systems. All models are only imperfect representations of the real world and their limitations need to be identified in the interpretation of the results. This thesis aims to support micro- and macro-level decision making. The applied approach potentially allows comparisons of the magnitude of land-use induced environmental impacts of product systems with identical function. The results presented in the thesis cannot, however, be interpreted to depict actual changes in environmental interventions as a consequence of substitution of one product with another.

It is obvious that, in the current form, the land use impacts cannot be modelled with a high degree of certainty nor communicated with adequate level of clarity to a decision maker, especially regarding impacts of forest biomass value chains on ecosystem services and biodiversity. Thus, in the current form of land use impact assessment in LCA, the core aim of LCA modelling, supporting actual, environmentally considerate decisions is not fully met. It seems that, for now, the decision makers need to make their decisions, with potentially far reaching implications, based on incomplete information on the environmental consequences of their decisions.

Meanwhile, it seems that the academia needs to keep on improving the modelling framework. The analysis in this thesis on land use impact assessment of forest value chains can help the academia in further development of land use impact assessment methods and respective characterisation factors. Additionally, the discussion on the selection of reference situation in relation with the actual decision-making context can deliver clarity and consistency in selecting appropriate reference levels in LCA modelling to best support the decision at hand.

Building on this background, it can be concluded that the academia needs to be clear in their discourse with decision-makers in communicating the limited certainty on whether land-use intensive activities can help in meeting the strict mitigation targets we are globally facing.

Acknowledgements

The author wants to express his gratitude to the following organizations for their funding, which made this Dissertation process possible:

VTT Technical Research Centre of Finland
FIBIC Finnish Bioeconomy Cluster
Tekes – the Finnish Funding Agency for Innovation
Academy of Finland
UPM
Metsä Group

References

Alvarado F, Backlund B, Berg S, Hohenthal C, Kaila S, Lindholm E-L, Wessman H (2002) Evaluation of land use oriented LCA methods and associated indicators.Scan Forsk-Rapport 739.

Baitz M, Albrecht S, Brauner E (2013) LCA's theory and practice: like ebony and ivory living in perfect harmony? Int J Life Cycle Assess 18:5–13

Bala G, Caldeira K, Wickett M, Phillips TJ, Lobell DB, Delire C, Mirin A (2007) Combined climate and carbon-cycle effects of large-scale deforestation. PNAS 104 (16): 6550-6555.

Beck T, Bos U, Wittstock B, Baitz M, Fischer M, Sedlbauer K (2010) LANCA-land use indicator value calculation in life cycle assessment. Fraunhofer, Stuttgart.

BirdLife (2010) Bioenergy. A carbon accounting time bomb. Birdlife International. Available at: www.birdlife.org/eu/pdfs/carbon_bomb_21_06_2010.pdf (Accessed on 20.9.2013)

Brandão M, Milà i Canals L (2013) Global characterisation factors to assess land use impacts on biotic production. Int J Life Cycle Assess 18: 1243–1252.

Brandão M, Milà i Canals L, Clift R (2010) Soil organic carbon changes in the cultivation of energy crops: implications for GHG balances and soil quality for use in LCA. Biomass and Bioenergy 35: 2323–2336.

Brander M, Tipper R, Hutchinson C, Davis G (2009) Consequential and attributional approaches to LCA: a guide to policy makers with specific reference to greenhouse gas LCA of biofuels. Econometrica Press Technical paper TP-090403-A.

Bright RM, Strømman AH, Peters GP (2011) Radiative forcing impacts of boreal forest biofuels: a scenario study for Norway in light of albedo. Environmental Science & Technology, 45, 7570–7580.

Bright RM, Cherubini F, Strømman AH (2012a) Climate impacts of bioenergy: inclusion of carbon cycle and albedo dynamics in life impact assessments. Environmental Impact Assessment Review, 37, 2–11.

Bright RM, Cherubini F, Astrup R et al., (2012b) A comment to "Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral": important insights beyond greenhouse gas accounting. Global Change Biology Bioenergy, 4, 617–619.

Bringezu S, Schütz H, Arnold K, Merten F, Kabaschi S, Borelbach P, Michels C, Reinhardt GA, Rettenmaier N (2009) Global implications of biomass and biofuel use in Germany – Recent trends and future scenarios for domestic and foreign agricultural land use and resulting GHG emissions. Journal of Cleaner Production 17: S57–S68.

Bringezu S, O'Brien M, Schütz H (2012) Beyond biofuels: Assessing global land use for domestic consumption of biomass: A conceptual and empirical contribution to sustainable management of global resources. Land Use Policy (29): 224–232.

CBD Convention on Biological Diversity (1993). Convention on Biological Diversity. Secretariat of the Convention of Biological Diversity. http://www.cbd.int/convention/

Cespi D, Passarini F, Ciacci L, Vassura I, Castellani V, Collina E, Piazzalunga A, Morselli L (2013) Heating systems LCA: comparison of biomass-based appliances. The International Journal of Life Cycle Assessment (in press). DOI 10.1007/s11367-013-0611-3

Cherubini F, Strømman AH (2011) Life cycle assessment of bioenergy systems: State of the art and future challenges. Bioresource technology 102 (2): 437–451. DOI 10.1016/j.biortech.2010.08.010

Cherubini F, Peters GP, Berntsen T, Strømman AH, Hertwich E (2011) CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. Global Change Biology Bioenergy, 3, 413–426.

Cherubini F, Guest G, Strømman AH (2012) Application of probability distributions to the modelling of biogenic CO2 fluxes in life cycle assessment. Global Change Biology Bioenergy, 4, 784–798.

Cowie A, Berndes G, Smith T (2013) On the timing of greenhouse gas mitigation benefits of forest-based bioenergy. IEA Bioenergy Executive Committee statement 2013:4. Available at: http://www.ieabioenergy.com/LibItem.aspx?id=7787 (Accessed on 20.9.2013)

Crutzen PJ, Mosier AR, Smith KA, Winiwarter W (2008) N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos Chem Phys 8: 389–395.

Curran MA, Mann M, Norris G (2005) The international workshop on electricity data for life cycle inventories. Journal of Cleaner Production, 8, 853–862.

de Baan L, Alkemade R, Koellner T (2013a) Land use impacts on biodiversity in LCA: a global approach. Int J Life Cycle Assess 18: 1216–1230.

de Baan L, Mutel CL, Curran M, Hellweg S, Koellner T (2013b) Land use in life cycle assessment: global characterization factors based on regional and global potential species extinction. Environ Sci Technol 47: 9281–9290.

DeFries R, Houghton RA, Hansen M et al. (2002) Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s. Proceedings of the National Academy of Sciences of the United States of America (PNAS), 99, 14256–14261.

de Santi G, Edwards R, Szekeres S, Neuwahl F, Mahieu V (Eds.), (2008) Biofuels in the European context: facts and uncertainties, European Commission Joint Research Centre, JRC.

Directive 2009/28/EC (2011) Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources. The Official Journal of the European Union.

EC European Commission (2002) Life sciences and biotechnology — A strategy for Europe. COM(2002) 27.

EC European Commission (2011) Energy roadmap 2050. COM(2011) 885, 15 December. URL http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2011:0885:FIN:EN:PDF

Ekvall T, Weidema BP (2004) System boundaries and input data in consequential life cycle inventory analysis. The International Journal of Life Cycle Assessment, 9, 161–171.

Eriksson E, Karlsson P-E, Hallberg L, Jelse K (2010) Carbon footprint of cartons in Europe – Carbon Footprint methodology and biogenic carbon sequestration. IVL Report B1924.

Ewing B, Goldfinger S, Wackernagel M, Stechbart M, Rizk SM, Reed A, Kitzes J (2010) Ecological footprint atlas 2010. Global Footprint Network, Oakland.

Fearnside PM, Lashof DA, Moura-Costa P (2000) Accounting for time in mitigating global warming through land-use change and forestry. Mitigation and Adaptation Strategies for Global Change, 5, 239–270.

Finnveden G, Hauschild MZ, Ekvall T et al., (2009) Recent developments in life cycle assessment. Journal of Environmental Management, 1, 1–21.

Foley JA, DeFries R, Asner GP et al. (2005) Global consequences of land use. Science 309 (5734): 570–574.

Goedkoop M, Heijungs R, Huijbregts MAJ, De Schryver A, Struijs J, van Zelm R (2008) ReCiPe 2008. A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. The Hague, The Netherlands

Guest G, Cherubini F, Strømman AH (2013) The role of forest residues in the accounting for the global warming potential of bioenergy. Global Change Biology Bioenergy, 5, 459–466.

Guinee JB, Heijungs R, van der Voet E (2009) A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. The International Journal of Life Cycle Assessment, 14, 328–339.

Haberl H, Erb KH, Krausmann F, Gaube V, Bondeau A, Plutzar C, Gingrich S, Lucht W, Fischer-Kowalski M (2007) Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. PNAS 104: 12942–12947.

Haberl H, Sprinz D, Bonazountas M et al. (2012a) Correcting a fundamental error in greenhouse gas accounting related to bioenergy. Energy Policy 45, 18–23.

Haberl H, Schultze E-D, Körner C, Law BE, Holtsmark B, Luyssaert S (2012b) Response: complexities of sustainable forest use. Global Change Biology Bioenergy, 4, doi: 10.1111/gcbb.12004

Havlík P, Schneider UA, Schmid E, et al. (2011) Global land-use implications of first and second generation biofuel targets. Energy Policy 38 (10): 5690–5702.

Heijungs R, Guinee JB, Huppes G, Lankreijer RM, Ansems AAM, Eggels PG, et al. (1992) Environmental life cycle assessment of products guide and backgrounds. Leiden: Centre of Environmental Science (CML).

Heuvelmans G, Muys B, Feyen J (2005) Extending the Life Cycle Methodology to Cover Impacts of Land Use Systems on the Water Balance. Int J LCA 10(2): 113–119.

Holtsmark B (2012) Harvesting in boreal forests and the biofuel carbon debt. Climatic Change, 112, 415–428.

Holtsmark B (2013) Quantifying the global warming potential of CO2 emissions from wood fuels. Global Change Biology Bioenergy (in press) doi: 10.1111/gcbb.12110

Houghton RA (2003) Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000. Tellus, 55B(2), 378–390.

Houghton RA (2012) Chapter 4: Historic changes in terrestrial carbon storage. In Lal R, Lorenz K, Hüttl RF, Schneider BU, van Braun J (eds). Recarbonization of the biosphere: Ecosystems and the global carbon cycle. Springer. 559 pp.

Huijbregts MAJ, Hellweg S, Frischknecht R, Hungerbühlerd K, Hendriks AJ (2008) Ecological footprint accounting in the life cycle assessment of products. Ecol Econ 64:798–807

Hynynen J, Ojansuu R, Hökkä H, Siipilehto J, Salminen H, Haapaka P (2002) Models for predicting stand development in MELA system. The Finnish Forest Research Institute. Research Papers 835, ISBN 951-40-1815-X, ISSN 0358-4283

IGBP/GCP (2013) Global carbon dioxide budget. The International Geosphere-Biosphere Programme, The Global Carbon Project. Available at: http://www.globalcarbonproject.org/carbonbudget/index.htm

Immerzeel DJ, Verweij PA, van der Hilst F, Faaij APC (2014) Biodiversity impacts of bioenergy crop production: a state-of-the-art review. Global Change Biology Bioenergy, doi:10.1111/gcbb.12067.

IPCC Intergovernmental Panel on Climate Change (2006) Agriculture, Forestry and Other Land Use. IPCC Guidelines for National Greenhouse Gas Inventories. Available at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

IPCC Intergovernmental Panel on Climate Change (2007) Fourth Assessment Report (AR 4). Working Group I Report 'The Physical Science Basis'. Cambridge University Press.

IPCC Intergovernmental Panel on Climate Change (2013) Climate change 2013 – The physical science basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

ISO 14040 (2006) Environmental management. Life cycle assessment. Principles and framework.

ISRIC World Soil Information, UNEP United Nations Environment Program (1991) World map of the status of human-induced soil degradation. International Soil Reference and Information Centre (ISRIC), United Nations Environment Programme (UNEP), April 1991

Jonker JGG, Junginger M, Faaij A (2013) Carbon payback period and carbon offset parity point of wood pellet production in the South-eastern United States. Global Change Biology Bioenergy (in press) doi: 10.1111/gcbb.12056

JRC-IES: Joint Research Centre – Institute for Environment and Sustainability (2010) International Reference Life Cycle Data System (ILCD) handbook. JRC-IES, Ispra.

Katzensteiner K, Kilmo E, Szukics U, Delaney CM (2011) Impact of forest management alternatives on water budgets and runoff processes. Paper 2 in Raulund-Rasmussen K, De Jong J, Humphrey JW et al. Papers on impacts of forest management on environmental services. EFI Technical Report 57, 2011

Kauppi P, Rautiainen A, Korhonen K et al. (2010) Changing stock of biomass carbon in a boreal forest over 93 years. Forest Ecology and Management, 259, 1239–1244.

Kirkinen J, Palosuo T, Holmgren K, Savolainen I (2008) Impact due to the use of combustible fuels: life cycle viewpoint and relative radiative forcing commitment. Environmental Management, 42, 458–469.

Koellner T, Geyer R (eds) (2013) Global land use impacts on biodiversity and ecosystem services in LCA. Int J Life CycleAssess 18(6):1185–1187

Koellner T, Scholz RW (2007) Assessment of land use impacts on the natural environment. Part 1: an analytical framework for pure land occupation and land use change. Int J Life Cycle Assess 12 (1):16–23

Koellner T, de Baan L, Beck T, Brandão M, Civit B, Margni M, Milà i Canals L, Saad R, de Souza DM, Müller-Wenk R (2013) UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. Int J Life Cycle Assess 18: 1188–1202.

Köhl M, Bastup-Birk A, Marchetti M et al. (2011) Criterion 3: maintenance and encouragement of productive functions of forests (wood and non-wood). In: State of Europe’s Forests 2011. Status and Trends in Sustainable Forest Management in Europe (eds FOREST EUROPE, UNECE and FAO). Pp. 51–64, Ministerial Conference on the Protection of Forests in Europe, Oslo.

Kujanpää M, Eggers J, Verkerk H, Helin T, Lindner M, Wessman H (2010) Carbon balance of forest residue collection and combustion in Southern-Finland. Proceedings of the 18th European Biomass Conference and Exhibition, Lyon.

Lamers P, Junginger M (2013) The ‘debt’ is in the detail: A synthesis of recent temporal forest carbon analyses on woody biomass for energy. Biofuels, Bioproducts and Biorefining 7 (4), 373–385.

Levasseur A, Lesage P, Margni M, Deschenes L, Samson R (2010) Considering time in LCA: dynamic LCA and its application to global warming impact assessments. Environmental Science & Technology, 44, 3169–3174.

Levasseur A, Lesage P, Margni M, Brandão M, Samson R (2012) Assessing temporary carbon sequestration and storage projects through land use, land-use change and forestry: comparison of dynamic life cycle assessment with ton-year approaches. Climatic Change, 115, 759–776.

Lifset R. (1997) A Metaphor, a Field, and a Journal. Journal of Industrial Ecology 1: 1–3.

Lippke B, Oneil E, Harrison R, Skog K, Gustavsson L, Sahre R (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. Carbon Management, 2, 303–333.

Luyssaert S, Schulze ED, Börner A, Knohl A, Hessenmöller D, Law BE, Ciais P, Grace J (2008) Old-growth forests as global carbon sinks. Nature 455:213–215.

Mainville N (2011) Fuelling a BioMess. Why burning trees for energy will harm people, the climate and forests. Greenpeace Canada. Available at: http://www.greenpeace.org/canada/Global/canada/report/2011/10/Forest Biomess_Eng.pdf (Accessed on 20.9.2013)

Matala J, Hynynen J, Miina J et al. (2003) Comparison of a physiological model and a statistical model for prediction of growth and yield in boreal forests. Ecological Modelling 161, 95–116.

Mattila T, Seppälä J, Nissinen A, Mäenpää I (2011) Land use impacts of industries and products in the Finnish economy: a comparison of three indicators. Biomass and Bioenergy 35: 4781–4787.

McKechnie J, Colombo S, Chen J, Mabee W, MacLean HL (2011) Forest bioenergy or forest carbon? assessing trade-offs in greenhouse gas mitigation with wood-based fuels. Environmental Science & Technology, 45, 789–795.

Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Freiermuth Knuchel R, Gaillard G, Michelsen O, Müller-Wenk R, Rydgren B (2007a) Key elements in a framework for land use impact assessment within LCA. Int J Life Cycle Assess 12(1):5–15.

Milà i Canals L, Romanyà J, Cowell SJ (2007b) Method for assessing impacts on life support functions (LSF) related to the use of ‘fertile land’ in life cycle assessment (LCA). J Clean Prod 15:1426–1440

Milà i Canals L, Rigarlsford G, SimS (2013) Land use impact assessment of margarine. Int J Life Cycle Assess 18: 1265–1277.

MEA, Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: Synthesis. Washington, DC: Island Press.

Miner R (2010) Biomass Carbon Neutrality. NCASI Discussion paper. Available at: http://nafoalliance.org/wp-content/uploads/NCASI-Biomass-carbon-neutrality.pdf (accessed 20 September 2013).

Moura-Costa P, Wilson C (2000) An equivalence factor between CO_2 avoided emissions and sequestration – description and applications in forestry. Mitigation and Adaptation Strategies for Global Change, 5, 51–60.

Müller-Wenk R, Brandão M (2010) Climatic impact of land use in LCA—carbon transfers between vegetation/soil and air. Int J Life Cycle Assess 15: 172–182.

Nabuurs G-J, Lindner M, Verkerk PJ, Gunia K, Deda P, Michalak R, Grassi G (2013) First signs of carbon sink saturation in European forest biomass. Nature climate change 3, 792–796

Ness B, Urbel-Piirsalu E, Anderberg S, Olsson L (2007) Categorising tools for sustainability assessment. Ecological Economics 60: 498–508.

Oldeman LR (1992) Global extent of soil degradation. ISRIC Bi-annual report 1991–1992. Pp. 19–36.

Paasonen P, Asmi A, Petäjä T et al. (2013) Warming-induced increase in aerosol number concentration likely to moderate climate change. Nature Geoscience (in press) DOI: 10.1038/NGEO1800

Pan Y, Birdsley RA, Fang Y et al. (2011) A large and persistent carbon sink in the World's forests. Science, 333, 988–993.

Pedroli B, Elbersen B, Frederiksen P et al. (2013) Is energy cropping in Europe compatible with biodiversity? opportunities and threats to biodiversity from land-based production of biomass for bioenergy purposes. Biomass and Bioenergy, 55: 73–86.

Pena N, Bird DN, Zanchi G, (2011) Improved methods for carbon accounting for bioenergy. Occasional paper 64. CIFOR, Bogor, Indonesia.

Pingoud K, Ekholm T, Savolainen I (2012) Global warming potential (GWP) factors and warming payback time as climate indicators of forest biomass use. Mitigation and Adaptation Strategies for Global Change, 17, 369–383.

Plevin RJ, Delucchi MA, Creutzig F (2013) Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. Journal of Industrial Ecology (in press) DOI: 10.1111/jiec.12074

Raulund-Rasmussen K, De Jong J, Humphrey JW et al (2011) Papers on impacts of forest management on environmental services. EFI Technical Report 57:2011.

Ridoutt BG, Page G, Opie K, Huang J, Bellotti W (2013) Carbon, water and land use footprints of beef cattle production systems in southern Australia. J Clean Prod. doi:10.1016/j.jclepro.2013.08.012

Rockström J, Steffen W, Noone K et al. (2009) Planetary Boundaries: Exploring the Safe Operating Space for Humanity. Ecology and Society 14(2): 32 [online] URL: http://www.ecologyandsociety.org/vol14/iss2/art32/

Saad R, Koellner T, Margni M (2013) Land use impacts on freshwater regulation, erosion regulation and water purification: a spatial approach for a global scale. Int J Life Cycle Assess 18:1253–1264.

Salminen H, Lehtonen M, Hynynen J (2005) Reusing legacy FORTRAN in the MOTTI growth and yield simulator. Computers and Electronics in Agriculture 49, 103–113.

Sedjo RA (2011) Carbon Neutrality and Bioenergy. A Zero-Sum Game? RFF DP 11–15. Resources for the Future, Washington DC.

Schlamadinger B, Apps M, Bohlin F et al. (1997) Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. Biomass and Bioenergy, 6, 359–375.

Schmidt JH (2008) Development of LCIA characterisation factors for land use impacts on biodiversity. J Clean Prod 16:1929–1942.

Schulze ED, Körner C, Law BE, Haberl H, Luyssaert S (2012) Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. Global Change Biology Bioenergy, 4: 611–616. doi: 10.1111/j.1757-1707.2012.01169.x

Schweinle et al. (ed.) (2002) The assessment of environmental impacts caused by land use in the life cycle assessment of forestry and forest products. Final report working group “land use” of COST Action E9

Searchinger T, Heimlich R, Houghton RA et al., (2008) Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land Use Change. Science 319, 1238–1240.

Searchinger T, Hamburg SP, Melillo J et al. (2009) Fixing a Critical Climate Accounting Error. Science 326, 527–528.

Sedjo RA (2011) Carbon Neutrality and Bioenergy. A Zero-Sum Game ? RFF DP 11–15. Resources for the Future, Washington DC.

Spracklen DV, Bonn B, Carslaw KS (2008) Boreal forests, aerosols and the impacts on clouds and climate. Philosophical Transactions of the Royal Society A, 366, 4613–4626.

Stanners D, Bosch P, Dom A, Gabrielsen P, Gee D, Martin J, Rickard L, Weber JL (2007) Frameworks for environmental assessment and indicators at the EEA. In: Hàk T, Moldan B, Dahl AL (eds) Sustainability indicators: a scientific assessment. Island, Washington, pp 127–144.

Steen-Olsen K, Weinzettel J, Cranston G, Ercin AE, Hertwich EG (2012) Carbon, land, and water footprint accounts for the European Union: consumption, production, and displacements through international trade. Environ Sci Technol 46:10883–10891

Strauss W (2011) How Manomet got it Backwards: Challenging the “Debt-Then-Dividend” Axiom. Futuremetrics. Available at: http://www.futuremetrics.net/papers/Manomet%20Got%20it%20Backwards.pdf.

Suh S, Yang Y (2014) On the uncanny capabilities of consequential LCA. Int J Life Cycle Assess (in press) DOI 10.1007/s11367-014-0739-9.

Tillman A-M (2000) Significance of decision-making for LCA methodology. Environmental Impact Assessment Review 20: 113–123.

Udo de Haes H (2006) How to approach land use in LCIA or, how to avoid the Cinderella effect? Int J Life Cycle Assess 11:219–221.

Udo de Haes H, Heijungs R (2007) Life-cycle assessment for energy analysis and management. Applied Energy, 84, 817-827.

UNCSD United Nations Conference on Sustainable Development Rio+20 (2012). Future we want – Outcome document. The UNSCD general assembly. URL: http://sustainabledevelopment.un.org/futurewewant.html

UNFCCC United Nations Framework Convention on Climate Change (1992) United Nations Framework Convention on Climate Change.

UNFCCC United Nations Framework Convention on Climate Change (1997) Kyoto Protocol to the United Nations Framework Convention on Climate Change.

UNFCCC United Nations Framework Convention on Climate Change (2009) Copenhagen Accord. United Nations. Available at: http://www.unfccc.org

Wackernage lM, Schulz NB, Deumling D, Linares AC, Jenkins M, Kapos V, Monfreda C, Loh J, Myers N (2002) Tracking the ecological overshoot of the human economy. PNAS 99:9266–9271

Walker T, Cardellichio P, Colnes A et al., (2010) Biomass Sustainability and Carbon Policy Study. Natural Capital Initiative at Manomet Report. Manomet Center for Conservation Studies, MA, USA.

WCED Wold Commission on Environment and Development (1987) Our Common Future. Oxford University Press, Oxford.

Weiss M, Haufe J, Carus M, Brandão M, Bringezu S, Hermann B, Patel MK (2012) A Review of the Environmental Impacts of Biobased Materials. Journal of Industrial Ecology 16: S169–181.

Wessman H, Alvarado F, Backlund B, Berg S, Hohenthal C, Kaila S, Lindholm E-L (2003) Land use in ecobalance and LCA of forest products. Scan Forsk Report 746.

WRI: World Resources Institute (2006) The Greenhouse Gas Protocol. The Land Use, Land-Use Change, and Forestry Guidance for GHG Project Accounting. World Resources Institute, Washington.

Zamagni A, Guinee J, Heijungs R, Masoni P, Raggi A (2012) Lights and shadows in consequential LCA. The International Journal of Life Cycle Assessment, 17, 904–918.

Zanchi G, Pena N, Bird DN (2010) The Upfront Carbon Debt of Bioenergy. Joanneum Research Report.

ARTICLE I

Land use indicators in life cycle assessment

A case study on beer production

The International Journal of Life Cycle Assessment
17: 277–286

ARTICLE II

Approaches for inclusion of forest carbon cycle in life cycle assessment

A review

Global Change Biology Bioenergy 5: 475–486.

GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2013) **5**, 475–486, doi: 10.1111/gcbb.12016

REVIEW

Approaches for inclusion of forest carbon cycle in life cycle assessment – a review

TUOMAS HELIN*, LAURA SOKKA†, SAMPO SOIMAKALLIO†, KIM PINGOUD† and TIINA PAJULA*

**VTT Technical Research Centre of Finland, Sustainability Assessment, Tekniikantie 2, Espoo, FI-02044 VTT, Finland,*
†VTT Technical Research Centre of Finland, Climate Change, Tekniikantie 2, Espoo, FI-02044 VTT, Finland

Abstract

Forests are a significant pool of terrestrial carbon. A key feature related to forest biomass harvesting and use is the typical time difference between carbon release into and sequestration from the atmosphere. Traditionally, the use of sustainably grown biomass has been considered as carbon neutral in life cycle assessment (LCA) studies. However, various approaches to account for greenhouse gas (GHG) emissions and sinks of forest biomass acquisition and use have also been developed and applied, resulting in different conclusions on climate impacts of forest products. The aim of this study is to summarize, clarify, and assess the suitability of these approaches for LCA. A literature review is carried out, and the results are analyzed through an assessment framework. The different approaches are reviewed through their approach to the definition of reference land-use situation, consideration of time frame and timing of carbon emissions and sequestration, substitution credits, and indicators applied to measure climate impacts. On the basis of the review, it is concluded that, to account for GHG emissions and the related climate impacts objectively, biomass carbon stored in the products and the timing of sinks and emissions should be taken into account in LCA. The reference situation for forest land use has to be defined appropriately, describing the development in the absence of the studied system. We suggest the use of some climate impact indicator that takes the timing of the emissions and sinks into consideration and enables the use of different time frames. If substitution credits are considered, they need to be transparently presented in the results. Instead of carbon stock values taken from the literature, the use of dynamic forest models is recommended.

Keywords: boreal, climate change, climate indicators, forest carbon cycle, life cycle assessment

Received 21 January 2012 and accepted 23 July 2012

Introduction

Biomass plays a significant role in the global carbon cycle. Between 2000 and 2008, terrestrial biomass (including soil) sequestered annually altogether approximately 20–25% of the carbon emitted from fossil fuel combustion and cement production, that is, functioning as a carbon sink of similar magnitude to the oceans (IGBP/GCP, 2010). However, land-use changes, mainly due to deforestation of tropical rain forests and forest degradation, emitted approximately half of the carbon that was sequestered in terrestrial biomass back to the atmosphere (IPCC, 2007a). As the carbon emissions to the atmosphere and the sinks from the atmosphere are not in balance, the atmospheric carbon dioxide concentrations are growing.

Correspondence: Tuomas Helin, tel. + 358 40 827 8355, e-mail: tuomas.helin@vtt.fi

Forests (including soils) are a significant pool of terrestrial carbon. They have been recognized as gaining carbon in many areas (DeFries *et al*., 2002; Pan *et al*., 2011), with even an increased growth rate in some regions, for example, in Finland (Kauppi *et al*. 2010). For example, in Europe, the growth of forests was significantly higher compared with removals between 1990 and 2010, resulting in an annual average net carbon sink in the standing biomass (Köhl *et al*., 2011). The annual sink varies from year to year, mainly due to the intensity of removals. Thus, a change in land management (with no change in land-use class) contributes to a change in terrestrial C stocks. Transition toward bioeconomy and ambitious targets to increase the use of renewable energy sources (e.g., Directive 2009/28/EC) may significantly increase the removals of wood from boreal forests, thus decreasing their carbon sink compared with the current trends (Böttcher *et al*., 2012).

The target of the mitigation fundamentally influences the effectiveness of the different actions on mitigating climate change. The lower the desired limit of global temperature increase, the lower the stabilization level of greenhouse gas concentrations in the atmosphere, and the more rapidly the greenhouse gas emissions need to be reduced. The European Union (EU) (EC: Commission of the European Communities, 1996, 2007) and, later, all the countries that have ratified the United Nations Framework Convention on Climate Change (UN 1992) have recognized 'the scientific view that the increase in global temperature should be below 2 °C' (UNFCCC, 2009). According to the IPCC, global greenhouse gas (GHG) emissions should peak no later than 2015 and should be reduced by at least 50–85% by 2050, and maybe by even more than 100% prior to the end of the century, from their levels in 2000, to keep the probability of limiting the global mean surface temperature increase to under 2 °C compared with the preindustrial level reasonable (IPCC, 2007b). Continuously increasing global GHG emissions, together with the ambitious climate policy targets, requires the introduction of effective measures within an increasingly short time.

Life cycle assessment (LCA) is a methodological framework for estimating and assessing the environmental impacts related to the life cycle of a product system (Rebitzer *et al.*, 2004; Finnveden *et al.*, 2009). LCA is initiated by a definition of the goal and scope; this is followed by life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation of the results (ISO 14040, 2006; ISO 14044, 2006). The development of LCA has led to a definition of different LCA modes or types (e.g., Ekvall & Weidema, 2004; Curran *et al.*, 2005; Finnveden *et al.*, 2009; Zamagni *et al.*, 2012). Attributional LCA (ALCA) has been defined as a method 'to describe the environmentally relevant physical flows of a past, current, or potential future product system' (Curran *et al.*, 2005). In contrast, consequential LCA (CLCA) can be defined as a method that aims to describe how environmentally relevant physical flows would have been or would be changed in response to possible decisions that would have been or would be made (Curran *et al.*, 2005). The ALCA reflects the system 'as it is', whereas the CLCA attempts to respond to the question: 'What if'? In ALCA, average data depicting the actual physical flows are used, whereas in CLCA, marginal data are used when relevant for the purpose of assessing the consequences (Ekvall & Weidema, 2004; Finnveden *et al.*, 2009).

Handling long-rotation forest biomass carbon emissions and sequestration remains as one of the controversial issues in LCA (e.g., Guinée *et al.*, 2009). Traditionally, the use of sustainably grown forest biomass has been considered carbon neutral in LCA, based on the assumption that the carbon that is released during combustion or decay of biomass is sequestered back into the growing biomass. In such a case, the biomass system is carbon neutral over the rotation period. The timing difference between the release and sequestration of forest biomass carbon leads to a situation where part of the carbon remains in the atmosphere until it is fully sequestered back into the growing forest. This results in a warming impact if sequestration lags emission. Thus, carbon neutrality over a forest rotation period is not equal to climate neutrality (see e.g., Cherubini *et al.*, 2011).

Many biomass GHG accounting methods have been developed (e.g., IPCC, 2006; WRI: World Resources Institute, 2006; Eriksson *et al.*, 2010; Pena *et al.*, 2011; Cherubini *et al.*, 2011; Lippke *et al.*, 2011) resulting in notably different conclusions. The aim of this study is to discuss and identify which characteristics of the different approaches are most suitable for the treatment of biomass carbon flows within LCA. The crucial methodological choices include selection of the reference situation and setting of the spatial and temporal system boundary. In addition, consideration of timing of emissions and sinks, and different forest carbon stocks, as well as the selection of climate indicators used, are critical issues for modeling. Furthermore, the aim of this study is to summarize, clarify, and assess how the different approaches take these critical issues into account. As this study focuses on the inclusion of biogenic carbon flows in LCA, the climatic impacts of direct changes in surface albedo and indirect changes in cloud albedo caused by forest land use and management have been left outside the scope of this study, even though some recent studies indicate that their impact may be considerable (e.g., Kulmala *et al.*, 2004; Spracklen *et al.*, 2008; Bright *et al.*, 2011, 2012).

Literature review

Materials and methods

A relevant set of documents that address the question of biomass carbon stock changes in forests related to climate change was identified and selected for the review on the basis of literature searches and previous experience of the publications in the field. These included peer-review articles, legally binding documents, voluntary standards, and gray reports (see Table S1).

The selected documents were reviewed through a set of five questions. The questions (below) were chosen based on previous knowledge of the critical factors in the assessment of the forest biomass carbon flows.

1. Does the approach apply some reference situation for land use? If it does, specify what.
2. Does the approach consider timing of emissions and sinks, and on what time horizons?
3. What indicators are used for measuring the GHG emissions or their warming impact?

4. Forest modeling: Does the approach consider the whole carbon stock of the forest or only part of it? Is the approach based on the use of specific forest models or literature values?
5. Forest product use: Does the approach consider biomass carbon stored in a product? What about product substitution impacts?

It is important to note that the climate implications of forest products' life cycles can be studied from three separate perspectives:

- The impact of forest biomass acquisition activity on the biomass carbon stocks in forests;
- The impact of biomass carbon storage in long-lived products;
- The impact of the potential substitution of other products (e.g., construction or packaging materials, electronic media, or fossil fuels) with forest products.

The third perspective, the substitution of other products, is often considered to be part of the CLCA modeling, whereas its applicability in ALCA remains disputed.

Reference situation for forest

When the environmental impacts of any activity are assessed with LCA (following the standards ISO 14040, 2006; ISO 14044, 2006), it is essential to ensure that the environmental inputs and outputs included in the assessment reflect the studied activity; In other words, to ensure that the environmental impacts attributed to the studied system would not have occurred without the existence of the activity. Consistent with this basic principle, the ILCD Handbook (JRC-IES: Joint Research Centre – Institute for Environment & Sustainability, 2010) gives guidance for modeling agro and forestry systems. According to Chapter 7.4.4.1 of the ILCD Handbook:

'only the net interventions related to human land management activities shall be inventoried in LCI. Interventions that would occur also if the site was unused shall not be inventoried.'

This leads to the need to define a 'no use' reference situation to represent the agro or forestry system in which the studied human activity did not occur. JRC-IES: Joint Research Centre – Institute for Environment & Sustainability (2010) continues, stating that:

'The "no use" reference system shall be the independent behaviour of the site, starting from the status of the land at that moment when the area of the analyzed system is prepared for the modelled system' under ALCA modeling. The determination of the reference system in LCA is discussed in the following paragraphs.

The UNEP-SETAC Life cycle initiative[1] has released a framework of LCIA for land use (Milà i Canals *et al.*, 2007). It includes a detailed discussion and suggestions on the modeling of the 'no use' reference situation. In ALCA, the 'no use' reference situation is the natural relaxation of the land area, whereas in CLCA, an alternative land-use situation becomes the reference. Alternative land use can be either natural vegetation or some human land use, depending on the situation in that specific area, and it can be derived, for example, from statistical time series or economic models (Milà i Canals *et al.*, 2007). Both natural relaxation and alternative land use can be considered as 'the independent behaviour of the site', as stated in JRC-IES: Joint Research Centre – Institute for Environment & Sustainability (2010), and the choice depends on the modeling approach selected (attributional or consequential). An example of carbon sequestration over time in different forest management scenarios is presented in Fig. 1. In terms of the approaches suggested by Milà i Canals *et al.* (2007), 'not harvested' corresponds to natural relaxation and is the reference situation in ALCA. In CLCA, 'once intensified harvested' shall be compared with the alternative land use derived from statistical time series or economic models, such as 'once harvested' or 'not harvested' in the example in Fig. 1.

Gustavsson *et al.* (2000) discussed principles for project-based greenhouse gas accounting with a focus on baselines and additionality. They concluded that a primary consideration in the selection of baselines must be their efficacy in helping to achieve the ultimate objective of the UNFCCC, that is, the stabilization of greenhouse gas concentrations in the atmosphere. Furthermore, they concluded that a baseline should provide an accurate description of the path of net emissions in the absence of a purposeful intervention. These project-based scenario analyses have many features in common with CLCA, thus providing relevant aspects for the definition of the reference situation.

The majority of the reviewed publications have applied some reference situation, that is, they have presented the environmental impacts of the studied activity relative to a reference scenario. The natural evolution of the site is applied as reference scenario only in Perez-Garcia *et al.* (2005), Müller-Wenk & Brandão (2010), Lippke *et al.* (2011), and in Pingoud *et al.* (2012) for stemwood. In Eriksson *et al.* (2010), the reference is the carbon sequestration rate (not carbon stock) in the natural situation some decades in the future, and the seques-

[1] In 2002, the United Nations Environment Programme (UNEP) and the Society for Environmental Toxicology and Chemistry (SETAC) launched an International Life Cycle Partnership, known as the Life Cycle Initiative, to enable users around the world to put life cycle thinking into effective practice. For more information: http://lcinitiative.unep.fr/

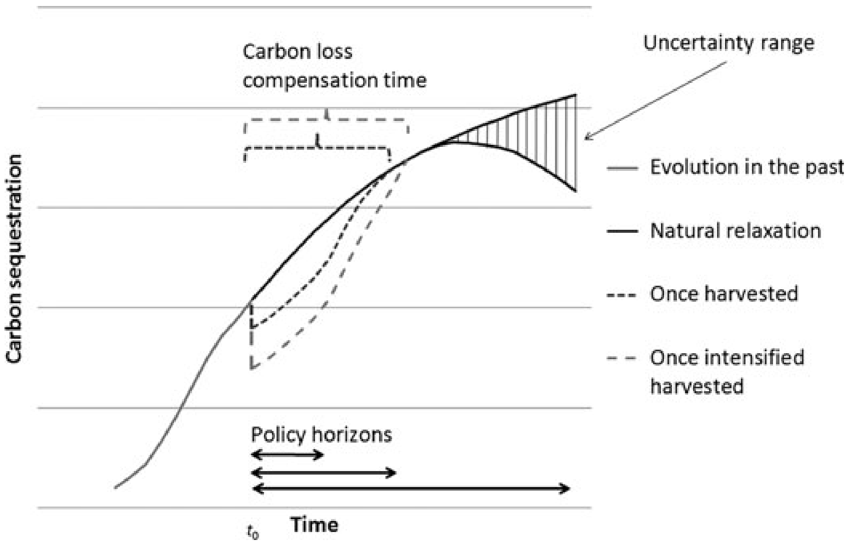

Fig. 1 Carbon sequestration in a forest stand over time in various forest management options, including no harvesting, one harvesting, and one intensified harvesting. Carbon loss compensation time may vary between management options and policy horizon to consider climate impacts may vary from carbon loss compensation time. The curves are for illustrative purposes only and do not represent results of any actual forest modeling run.

tration rate of that time is assumed to equal a net C sequestration rate of near zero.

Most of the publications (Schlamadinger *et al.*, 1997; WRI: World Resources Institute, 2006; Kirkinen *et al.*, 2008; Zanchi *et al.*, 2010; Kujanpää *et al.*, 2010; Repo *et al.*, 2011; Holtsmark, 2012; McKechnie *et al.*, 2011; Pingoud *et al.*, 2012 for forest residues) have applied alternative land use as the reference situation. However, in some of them (e.g., Kirkinen *et al.*, 2008; Zanchi *et al.*, 2010; McKechnie *et al.*, 2011; Repo *et al.*, 2011; Pingoud *et al.*, 2012), forest management without forest residue harvesting is used as the reference. Holtsmark (2012) studies the impact of a 30% increase in stemwood harvests relative to current harvesting. Kujanpää *et al.* (2010) use a similar step change approach for the forest residue case, but study the marginal impacts of harvesting pulpwood with a so-called pulse approach. In this approach, a small change is applied to the annual baseline harvests in the first year(s) of harvests, and the impact of this marginal change is studied over time relative to the current use baseline.

A few of the reviewed studies consider constant terrestrial carbon stock level as the reference or do not present any reference situation.[2] Cherubini *et al.* (2011, 2012) and Bright *et al.* (2012) define GWP_{bio} factors for bioenergy use without a reference situation. Kilpeläinen *et al.* (2011), on the other hand, focus on one stand rotation from stand establishment to final felling, and no reference is applied. Miner (2010) identifies no reference situation in the analysis that focuses on the different spatial system boundaries. This 'zero level' reference approach can be considered to be consistent with the principles of ALCA modeling that aims to reflect the system'as it is', although the environmental relevance of such an approach can be disputed. It is noteworthy that the most well-known directives, voluntary standards, and footprinting guidelines (ISO/DIS 14067; EC: Commission of the European Communities, 2011; PAS 2050, 2011; Directive 2009/28/EC) do not provide guidance on the use of a reference situation for land-use activities if no direct land-use change (from one land-use class to another) occurs.[3] For land-use change, the reference in these is the land use in a defined year (e.g., 2008).

The question of whether to assess the forest carbon flows at stand, local, regional, or country level has raised dispute (see e.g., Miner, 2010; Sedjo, 2011). This dispute may stem partly from the inconsistent application and identification of reference situations in different climate impact assessment studies (e.g., Eriksson *et al.*, 2010; Cherubini *et al.*, 2011; Kilpeläinen *et al.*, 2011). According to some (e.g., Eriksson *et al.*, 2010; Miner, 2010; Sedjo, 2011), the assessment of the biomass carbon cycle at stand level leads to a flawed picture of

[2]Both (with or without intent) lead to the use of a zero level, i.e., a situation with constant terrestrial carbon stock, as the reference.

[3]Neither do these publications give instructions on the calculation of carbon flows from land use in cases where no land-use change occurs.

the impact of wood harvest on the forest carbon flows. It is argued that if new wood is grown on the harvested land, then across the whole wood-producing region, there is a net zero carbon emission or net sink impact because the new growth on the other plots of the region will offset the carbon emissions from the harvested plot. This argument has been criticized for not taking into account the fact that, if there had been no biomass harvest, the forest would likely have continued to stock carbon (e.g., Walker *et al.*, 2010; Pingoud *et al.*, 2012). This dispute finds its roots in the exclusion of the independent evolution of the forest land as the reference situation in the climate impact assessment: for example, Miner (2010) and Sedjo (2011) do not consider that the forests that are not affected by the studied harvest either in the case studied or in the reference situation have no influence on the forest carbon stock changes regardless of the definition of the spatial system boundary. Careful definition of the reference situation for land use is therefore very important.

Timing of emissions and sinks and time horizon

Emissions and sinks of biomass carbon usually occur at different points in time, particularly in the case of forest biomass use (Fig. 1). The timing of emissions and sinks has an impact on the overall cumulative climatic impact of the activity over a certain time frame (see e.g., Levasseur *et al.*, 2010, 2012). The carbon emission into the atmosphere has a warming impact (radiative forcing), whereas the sequestration has a cooling impact. The carbon debt between emission and sequestration (carbon loss compensation time in Fig. 1) results in a warming effect if sequestration lags emission. As a consequence, the result of the climate impact assessment is dependent on the time horizon (referred as policy horizon in Fig. 1) of the assessment (Cherubini *et al.*, 2011).

The reviewed studies have considered the timing of emissions and sinks in different ways. As LCA is typically applied as a static tool, in which the emissions are assumed to take place at the same time, it is not necessarily well suited to assessing the complexity of forest carbon dynamics (Bright *et al.*, 2011; McKechnie *et al.*, 2011). Therefore, Levasseur *et al.* (2010) have proposed a dynamic LCA approach for climate impact assessment, in which the temporal profile of emissions is included in the LCI results and time-dependent characterization factors are applied in the LCIA phase. Most of the reviewed studies (e.g., Kirkinen *et al.*, 2008; Kujanpää *et al.*, 2010; Zanchi *et al.*, 2010; Kilpeläinen *et al.*, 2011; Repo *et al.*, 2011; Pingoud *et al.*, 2012) have applied different forest models to assess the dynamic impacts of forest harvest (see section Modeling of forest carbon stocks). On the other hand, some of the reviewed documents (e.g., Eriksson *et al.*, 2010; Miner, 2010) have adopted a static approach. These approaches consider the annual average net change in the C stock of the surrounding forest system and ignore the timing of the sinks and emissions within the specific product system relative to a reference situation. Such static approaches are analogous to the guidelines for annual national greenhouse gas inventories in the agriculture, forestry, and other land use sector (IPCC: Intergovernmental Panel on Climate Change, 2003; IPCC, 2006), but have limited applicability in LCA.

ISO 14040, 2006 and ISO 14044, 2006 do not address the timing issue, but the draft of ISO/DIS 14067; on the product carbon footprints (PCF) of products includes some guidelines. According to the draft, all greenhouse gas emissions arising from fossil and biogenic (biomass) sources shall be included in PCF calculations. However, biogenic carbon is reported separately from fossil carbon, showing transparently both removals and emissions, and shall be treated as if released and sequestered instantaneously in the first year of the assessment period. This loss of a temporal dimension in biomass carbon emissions and sequestration leads to a neutral climate impact in cases of no land-use (class) change.

The Directive 2009/28/EC; PAS 2050, 2011; guidelines, and the EU PEF guidance (EC: Commission of the European Communities, 2011) under preparation consider direct land-use change (i.e., a situation where the land-use class changes as a result of the action within the defined system boundary). In all of these documents, the emissions from land-use change are to be annualized by dividing the total emissions by 20. This emission term disappears after 20 years of cultivation. However, if forest land remains as forest land after final fellings, it is not typically considered as a land-use (class) change. These modeling principles might be reasonable for bioenergy from agricultural biomass, but their relevance for long-rotation biomass (i.e., boreal forests) can be questioned.

Within the other reviewed studies and documents, there are different perspectives on the time frame of the study. Most of the reviewed documents with a life-cycle scope suggest dynamic approaches with the inclusion of a longer time frame than the current year. Some of the publications (e.g., Kendall *et al.*, 2009; Directive 2009/28/EC) considered only 20–30 years, whereas others took into account a longer period of time, even 500 years (e.g., Müller-Wenk & Brandão, 2010; Cherubini *et al.*, 2011). Pingoud *et al.* (2010) propose a method in which equilibrium strategies of forest management and wood use with the best climate benefits in the very long run are sought. However, most of the reviewed approaches could be used to assess some other time frame as well (e.g., Schlamadinger *et al.*, 1997; Kirkinen *et al.*, 2008; Kendall *et al.*, 2009; Müller-Wenk & Brandão, 2010; Cherubini *et al.*, 2011; Holtsmark, 2012; Kilpeläinen *et al.*, 2011; McKechnie *et al.*, 2011; Repo *et al.*, 2011). Some studies have made calculations over different rotation periods or ages (e.g., Walker *et al.*, 2010; Cherubini *et al.*, 2011).

In many of the reviewed studies, 100 years has been chosen as the studied time frame (e.g., Perez-Garcia *et al.*, 2005; Kujanpää *et al.*, 2010; Walker *et al.*, 2010 and Repo *et al.*, 2011). McKechnie *et al.* (2011) justify the 100 years time frame through its relevance for long-term forest management planning. For the same reason, Kilpeläinen *et al.* (2011) have chosen 80 years. Many studies emphasize that there is no real scientific answer to what the studied time frame should be, but that it depends on the (political) aims of the assessment (Schlamadinger *et al.*, 1997; Kirkinen *et al.*, 2008; Walker *et al.*, 2010; Zanchi *et al.*, 2010). It is possible that shorter time frames than 100 years will be emphasized in climate policy in the future, due to rapid emission reduction requirements to stabilize atmospheric greenhouse gas concentrations at a low level, for example, in accordance with the 2 °C target, or to lower the temperature increase rate.

Most of the reviewed studies were forward-looking, focusing on the development of the forest carbon stocks from the present situation onwards (e.g., Kirkinen *et al.*, 2008; Repo *et al.*, 2011; Pingoud *et al.*, 2012; Holtsmark, 2012). Some others, however, argue that the forward-looking view ignores the fact that forest investment decisions in recent decades were usually based on anticipation of wood use in the future (Strauss, 2011). For example, Sedjo (2011) argues that past forest management, such as the planting of trees, was based on expectations of future use. Thus, from a broad forest-system perspective, burning biomass today would not release new carbon into the atmosphere, but would only emit carbon that was sequestered in the past in anticipation of this future use. Both perspectives can be appreciated, but the forward-looking approach seems to answer the most relevant questions from the 2 °C climate mitigation perspective.

Climate indicators

In LCA, the impacts on climate change are commonly measured with carbon dioxide equivalents (CO_2-eq.) based on global warming potential (GWP) coefficients (IPCC, 2007a). The GWP coefficients of the individual greenhouse gases are different for different time frames because the atmospheric lifetimes and radiative efficiencies of distinct forcing agents differ from each other (IPCC, 2007a; Table 2.14). The time frame for GWP coefficients is agreed to be 100 years (GWP-100) under the UNFCCC: United Nations Framework Convention on Climate Change (2009) and the Kyoto Protocol. This decision is political, without an unambiguous scientific basis. It has become a general practice within the LCA community to apply this time frame in LCIA. In addition, product carbon footprint standards and guidance documents, such as Directive 2009/28/EC; JRC-IES: Joint Research Centre – Institute for Environment & Sustainability (2010), PAS 2050, 2011; ISO/DIS 14067, and EC: Commission of the European Communities, 2011 require the use of GWP-100 in the climate impact assessment for fossil CO_2 and other greenhouse gases. GWP-100 is also applied, for example, in WRI: World Resources Institute (2006), Eriksson *et al.* (2010), Walker *et al.* (2010), and McKechnie *et al.* (2011).

The loss of the temporal dimension, which holds true when using the GWP coefficients (IPCC, 2007a) with fixed time frames, is particularly problematic for CO_2 originating from the changes in terrestrial carbon stocks. The reason is that the average stay of CO_2 from the terrestrial C cycle released to the atmosphere in sustainable land management activities is shorter than the average stay of CO_2 released from fossil reserves. A ton of CO_2 added to the atmosphere will be absorbed by the top layers of the sea and by terrestrial carbon stocks, according to the Bern 2.5 CC model (IPCC, 2007a, p. 213). However, for a ton of CO_2 released into the atmosphere from sustainably managed biomass resources, additionally a similar amount of CO_2 can be considered to be circulated back into the specific terrestrial C stocks (within the rotation period of the biomass, due to photosynthesis).

To get around this issue, some of the reviewed publications have applied other types of indicators for climate impact assessment of forest biomass use. Like the GWP-100 coefficients (IPCC, 2007a), these indicators find a basis in the cumulative radiative forcing (W m^{-2}) measure (e.g., Kirkinen *et al.*, 2008; Kendall *et al.*, 2009; Cherubini *et al.*, 2011; Repo *et al.*, 2011; Pingoud *et al.*, 2012). The use of radiative forcing as a function of time (e.g., Kirkinen *et al.*, 2008; Repo *et al.*, 2011) as an indicator to measure climate impacts is well justified. However, a drawback of this indicator is that it is somewhat unfamiliar to the general public and even to the LCA community, and cannot be applied with widely used GWP-100 factors.

Another application of the cumulative radiative forcing measure is the GWP_{bio} factor introduced in Cherubini *et al.* (2011). The GWP_{bio} factor is derived by approximating the atmospheric decay of carbon from long-rotation biomass with a simplified forest growth equation. These GWP_{bio} factors are calculated for situations in which carbon in stemwood (with a rotation period of 1–100 yr) is released into the atmosphere within a year after harvest.[4] This indicator makes the comparison with fossil CO_2 (and other greenhouse gas) emissions easy, but is currently only applicable to energy use. Pingoud *et al.* (2012) define the GWP_{bio} factor somewhat differently from Cherubini *et al.* (2011) by considering the temporary C debt due to forest biomass harvest with respect to the no-use baseline. In addition, Pingoud *et al.* (2012) present a GWP_{biouse} factor describing the net climate impacts of the biomass use cycle compared with a functionally equivalent fossil fuel-based cycle, including the impacts of temporary carbon sequestration into long-lived wood products. The displaced greenhouse gas emissions of the product could also be incorporated in the factor especially in CLCA. The advantage of the GWP_{bio} approach is that it provides a kind of physically based discounting factor by which the biomass emissions with deviating timing can be transformed into a permanent fossil carbon emission whose cumulative warming impact within a given time horizon is the same.

Müller-Wenk & Brandão (2010) and Kujanpää *et al.* (2010) have introduced other indicators that can be compared with a ton of fossil CO_2 emissions and that have been developed for all forest biomass uses. The fossil-combustion equivalent indicator (Müller-Wenk & Brandão, 2010) first defines the average fraction of fossil CO_2 remaining in the atmosphere within a finite time frame (for example the average fraction remaining is 0.475 within 0–100 year) and expresses this value in terms of years (47.5 years for the preceding example). They then calculate the average stay in the atmosphere of terrestrial CO_2 released by land-use activities and present the result relative to the stay of fossil CO_2. The approach in Kujanpää *et al.* (2010) is similar, but the result is presented as equivalent kilograms of fossil CO_2, not as relative time units. The drawback of these indicators is that they are only close approximations to the real warming impact because the fossil-combustion equivalent indicator (Müller-Wenk & Brandão, 2010) is based on average (Tier 1[5]) data and the approach by Kujanpää *et al.* (2010) does not take the impact of CO_2 absorption by the top sea layer and other terrestrial carbon stocks into consideration.

[4]For example, for burning stemwood from a forest with an 80 years rotation period, the respective GWP_{bio} for 20, 100, and 500 years time horizons would be 0.94, 0.34, and 0.06, respectively.

[5]For various categories of land-use greenhouse gas emission sources, there are several ways of calculating the emissions, described as tiers (e.g., Tier 1, Tier 2, Tier 3), and each tier has an associated increasing level of detail and accuracy (IPCC: Intergovernmental Panel on Climate Change, 2003).

Kendall *et al.* (2009) introduce a time correction factor (TCF) for accounting of timing of the land-use change induced greenhouse gas emissions. The TCF is calculated based on the relative climate change effect of an emission (measured as cumulative radiative forcing) occurring at the outset of biofuel feedstock cultivation. The TCF is analogous to the GWP coefficients in that it estimates the relative climate change effect of a gas based on its cumulative radiative forcing over time. However, whereas the GWP coefficients give equivalency factors for different gases in relation to CO_2, the TCF provides an equivalency factor for the relative impact of the emissions occurring at different times. The amortized (annualized) emissions are then multiplied by the TCF.

Many complementary indicators have been developed that aim to quantify the time needed to compensate for the initial release of terrestrial carbon caused by the harvesting of biomass for energy, for example, in relation to the anticipated fossil fuel substitution benefits. It needs to be noted that substitution impacts are often considered to be a part of the consequential LCA modeling approach, whereas their applicability in attributional LCA remains disputed. The carbon neutrality factor in Schlamadinger *et al.* (1995), Zanchi *et al.* (2010), and Holtsmark (2012) presents the time needed for assumed substitution impacts to compensate for the carbon released at the beginning of the activity, also known as the carbon debt. This indicator does not, however, take into account the fact that more greenhouse gases have been warming the atmosphere during this time before carbon neutrality was reached. In practice, climate neutrality will be reached later in time than carbon neutrality in such a case (for more details, see e.g., Cherubini *et al.*, 2011). The carbon debt used by Walker *et al.* (2010) refers to the initially higher CO_2 emissions from burning biomass-based fuels compared with fossil fuels.[6] The size of the carbon debt depends on the type of alternative fuel (e.g., coal, natural gas, other biomass) being replaced by the biofuel. Walker *et al.* (2010) also present a further indicator called 'carbon dividend', which is defined as the fraction of the equivalent fossil fuel emissions that are offset by forest growth at some specified point in time. The transition from debt to dividend occurs at the point where the atmospheric carbon level resulting from the life-cycle biomass emissions just equals the level resulting from the reference life-cycle fossil carbon emissions. Pingoud *et al.* (2012)

[6]The direct CO_2 emissions from fuel combustion are higher per net calorific value of the fuel for forest biomass than for fossil fuels (Statistics Finland, 2011).

present the warming payback time indicator, which describes the time frame after which forest biomass use is superior to its functionally equivalent fossil fuel-based alternative, considering cumulative radiative forcing. The GWP_{biouse} factor in Pingoud *et al.* (2012) is best suited for CLCA when the displaced greenhouse gas emissions of the product are incorporated in the factor.

Modeling of forest carbon stocks

In the assessment of the greenhouse gas impacts of forest biomass harvesting,[7] it is important to acknowledge that there are multiple carbon stocks and exchanges. In boreal forests, more than half of the carbon stock may be contained in the soil and litter (e.g., Liski *et al.*, 2006), which implies that it is also important to take possible changes in the soil carbon into account in the climate impact assessment.

Within the reviewed publications, a variety of approaches has been adopted. Some of the reviewed publications (e.g., Kendall *et al.*, 2009; Directive 2009/28/EC) consider only the C stock changes in conjunction with land-use change, from one land-use class to another, and ignore the changes in C stocks within a land-use class. On the other hand, voluntary life-cycle carbon accounting standards ISO/DIS 14067 (draft) and PAS 2050, 2011; as well as the method applied by Cherubini *et al.* (2011), have only focused on the amount of carbon in stemwood. This approach is simple and less time consuming, but it overlooks the possible impacts on the remaining C stocks in the forest.

Many of the reviewed studies have included all the aboveground C stocks, but have excluded the soil carbon stocks in their approaches (Perez-Garcia *et al.*, 2005; Eriksson *et al.*, 2010; Miner, 2010; and Walker *et al.*, 2010). This is potentially problematic, especially if forest residues are collected from the forest, as the activity has been identified to have an impact on the soil carbon stocks (Kirkinen *et al.*, 2008; Repo *et al.*, 2011). On the other hand, Lippke *et al.* (2011) concluded in their review of soil carbon-related literature that the accumulation of carbon in forest soils is largely driven by soil moisture, the amount of nitrogen (and other nutrients) in the soil, and climatic conditions. The amount of wood left on site, the length of the rotation period, and specific treatment options had an insignificant impact on the forest soil carbon contents over time. It was stated, however, that the rate of soil carbon accumulation slows down with forest residue collection activities. On the other hand, Helmisaari *et al.* (2011) have concluded that the amount of wood (especially harvesting residues) left on site has a significant impact on the nitrogen balance of the forest soil, and thus has an impact on the soil carbon pools. In addition, Johnson & Curtis (2001) conclude that whole-tree harvesting activities lead to decreases in soil C stocks and sawlog harvesting (with no forestry residue collection) leads to increases in soil C stocks.

Most of the reviewed publications take all the terrestrial carbon stocks in the forest into account, including soil (e.g., WRI: World Resources Institute, 2006; Kujanpää *et al.*, 2010; Müller-Wenk & Brandão, 2010; Zanchi *et al.*, 2010; Holtsmark, 2012; Kilpeläinen *et al.*, 2011; McKechnie *et al.*, 2011; Repo *et al.*, 2011). This approach seems the most appropriate, as it identifies and quantifies all the carbon exchanges within the forest ecosystem. Furthermore, the probability of changes in the soil carbon stocks due to the activity studied can be tested with the semiempirical soil process models.

A further issue related to the modeling of the life-cycle climate impacts is the background data used in the approach. Some approaches rely on terrestrial carbon data from the literature, whereas others find their basis in dynamic forest models. The majority of the standards included in the review (Directive 2009/28/EC; EC: Commission of the European Communities, 2011 draft; PAS 2050, 2011) use default (average) global terrestrial carbon stock values from the IPCC: Intergovernmental Panel on Climate Change (2003). The use of this so-called Tier 1 data is easy, but it includes high uncertainties, as there are large variations in local terrestrial C stocks compared with the global default values (see discussion in e.g., Müller-Wenk & Brandão, 2010). Another drawback is that these default values cannot show the differences inside one land-use category, for example for different forest management options. Therefore, some of the approaches reviewed (Eriksson *et al.*, 2010; Miner, 2010; Müller-Wenk & Brandão, 2010) have adopted the default values at country or region level, as used in national forest inventories. This is in accordance with the so-called Tier 2 approach by the IPCC: Intergovernmental Panel on Climate Change (2003) and reduces the uncertainties to some extent. However, the problem with the use of literature values as background data is that such an approach can never reflect the detailed, actual impacts of the activity.

The majority of the reviewed publications with a life-cycle perspective (e.g., Kirkinen *et al.*, 2008; Kujanpää *et al.*, 2010; Walker *et al.*, 2010; Zanchi *et al.*, 2010; Kilpeläinen *et al.*, 2011; Lippke *et al.*, 2011; McKechnie *et al.*, 2011; Repo *et al.*, 2011; Holtsmark, 2012) base the assessment of the climatic impacts of forestry on dynamic

[7]Please note the difference between forest biomass harvesting (described here) and use (described in section Climate implications of forest product use).

forest models. This approach is consistent with the Tier 3 approach by the IPCC: Intergovernmental Panel on Climate Change (2003), which is the most detailed level for assessment of carbon stock changes. Some of the models have been applied at site level (e.g., SIMA in Kilpeläinen *et al*., 2011 and GORCAM in Zanchi *et al*., 2010), based on one forest rotation cycle. However, most of the models applied in the reviewed publications are based on national forest inventory data and have been used at country or regional level to identify the evolution of carbon stocks and exchanges over time, under different forest management scenarios (e.g., Yasso07 in Repo *et al*., 2011; Pingoud *et al*., 2012; EFISCEN in Kujanpää *et al*., 2010; MOTTI in Pingoud *et al*., 2012; FORCARB2 in McKechnie *et al*., 2011; U.S. Forest Service Forest Vegetation Simulator in Walker *et al*., 2010; and the anonymous model in Holtsmark, 2012). A common feature of all these forest models is that they include all the carbon stocks and exchanges in the forest, with the exception of NE-TWIGS in Walker *et al.* (2010), which excludes soil carbon.

Climate implications of forest product use

The climate implications of forest product use can be studied from two perspectives: on the one hand, carbon can be stored in long-lived products, and on the other hand, forest products can substitute other products (e.g., fossil fuels).

Most of the reviewed studies and methods have been developed for bioenergy and therefore do not consider product carbon (e.g., Schlamadinger *et al*., 1995; Kendall *et al*., 2009; Miner, 2010; Walker *et al*., 2010; Zanchi *et al*., 2010; Cherubini *et al*., 2011; Holtsmark, 2012; Repo *et al*., 2011). The question of carbon storage is not in the scope of ISO 14040, 2006; nor Directive 2009/28/EC; PAS 2050, 2011; guidelines and the EC: Commission of the European Communities, 2011 guidance, on the other hand, take carbon in products into account. PAS 2050, 2011 states that if some or all of the carbon is stored in the product (e.g., wood fiber in a table) after 100 year, the portion of carbon not emitted to the atmosphere during that period shall be treated as stored carbon. However, possible potential additional carbon storage resulting from forest management activities is not included within the scope of PAS 2050, 2011;. In the draft of ISO/DIS 14067 the inclusion of carbon stored in products is stated to be optional. If it is included, it shall be treated separately from the product footprint.

Kilpeläinen *et al.* (2011) divide products into different groups based on their life span, and calculated carbon emissions from products no longer in use by applying an equation based on Karjalainen *et al.* (1994). Pingoud *et al.* (2012) introduce a negative GWP_{biouse} factor by which the climate impact of the temporary carbon sequestration into biomass products can be related to a permanent fossil C emission. Cherubini *et al.* (2012) explore the impact of delayed biomass carbon release due to product use and put a special focus on the (probability distributed) uncertainty of the exact timing of the emission at the end of the life of the product. They propose an approach in which the climate impacts of biomass carbon storage (i.e., delayed release) in biomass products is assessed by applying probability distributions of decay of biomass carbon, based on general life spans of generalized product groups. Both Cherubini *et al.* (2012) and Pingoud *et al.* (2012) propose an LCIA method that already includes data (i.e., assumptions on product life span) traditionally included independently of the LCIA in the LCI modeling phase of LCA. This restricts the flexible application of these approaches in LCA in practice, as it can be questioned if, for example, the probability distributions of lifetimes of product groups (as proposed by Cherubini *et al*., 2012) should override the possibility of an LCA practitioner selecting the most suitable inventory data in the LCI phase.

Net credits from the substitution of competing products[8] depend on the efficiency of the substitution: the fewer GHG emissions generated in the biomass system and the more GHG emissions generated in the competing product system, the higher the substitution credits. An indicator of the substitution benefits is the displacement factor (Schlamadinger & Marland, 1996; Marland & Schlamadinger, 1997; Sathre & O'Connor, 2010) describing the units of fossil C that can be displaced by 1 unit of biomass C. A meta-analysis in Sathre & O'Connor, 2010 results in an average displacement factor value of 2.1 (most applications in the range 1.0–3.0) for wood products. Different biomass resources and applications, as well as market mechanisms, provide different substitution credits. For example, Perez-Garcia *et al.* (2005) included three carbon pools in the study: the forest, forest products, and fossil fuels displaced by forest products in the end-use markets. Wood products were further divided into long-term (lumber) and short-term (wood chips, sawdust, bark, shavings) products. While in use, the long-term products were assumed to replace other building materials, such as concrete. According to the results, the shorter the rotation period was, the more products there were available on the markets to replace fossil-fuel intensive products. Thus, the lower amount of carbon stored in the forests was offset by the higher substitution credits achieved with short rotation periods. The review study by Lippke *et al.* (2011) includes a similar approach and ends up with the same conclusion.

[8]Here, substitution refers to the greenhouse gas benefits of the main product, i.e., the functional unit of the study, in comparison to other, competing products that fulfill the same function (e.g., substitution of fossil fuels with forest bioenergy).

On the other hand, Pingoud *et al.* (2010) come up with the opposite conclusion in their steady-state case study on normal forests. The saw log yield per hectare and year increased when prolonging the rotation period compared with the existing silvicultural guidelines in Finland. As saw log used in construction was assumed to have higher substitution credits than pulpwood and energy wood, the overall climate benefits were higher irrespective of the lower total biomass yield in some cases. The carbon neutrality factor presented by Schlamadinger *et al.* (1995) and Zanchi *et al.* (2010), and the carbon debt indicator presented by Holtsmark (2012), also take into account the credits from substituted reference energy system.

Conclusions

The aim of this study was to discuss and identify which characteristics of the many approaches presented in the literature for biomass carbon accounting are most suitable for the treatment of biomass carbon flows within LCA. The results of the review are summarized in Table S1. The literature cited mostly focuses on boreal forests, but the conclusions presented below can be considered valid for conventional forestry in any climatic region.

To capture the dynamic nature of forest carbon stocks, a reference situation for forest land use has to be defined appropriately, consistent with the goal and scope of the study. On the basis of the review it can be concluded that the reference situation should be natural relaxation in attributional LCA and alternative land use in consequential LCA. It can be appreciated that under the schemes requiring emission verification (e.g., Directive 2009/28/EC), the application of a virtual reference situation describing something that did not take place, is to some extent controversial. On the other hand, ignoring the 'no use' reference land-use situation in ALCA might result in conclusions that do not reflect the environmental impacts of the system studied.

Regarding the modeling of the evolution of carbon stocks in the studied forest system and the 'no use' reference situation, the use of dynamic forest models is recommended. The forest models enable a more detailed analysis compared to the use of literature values, although more resources are needed from the LCA practitioner. Changes in all the different forest C stocks, such as stemwood, branches, roots, litter, and soil, need to be considered. Special attention should be paid to the consideration and transparent reporting of the uncertainties related to the modeling of future development of the biomass stocks.

To assess GHG emissions and related climate impacts objectively, biomass carbon stored in products should be taken into account in LCA. The climate impact LCIA methods should allow flexible selection of case-specific inventory data on product use in the LCI phase, and not include predefined generalized assumptions about the life cycle of the product. Similarly, when aiming to assess the overall consequences of forest use on GHG emissions, the market effects through product substitution should be considered in consequential LCA. The application of substitution effects in an attributional LCA context remains controversial. It can be recommended that transparent documentation is used when reporting LCI or LCA results to make the results more useful and the conclusions more understandable.

Forests are a renewable resource and, provided the land area stays as forest, within some time frame the resource will be renewed. Although a sustainably managed biomass system usually is carbon neutral or even accumulates carbon over time, the timing difference between emission and sequestration results in a warming effect if sequestration lags behind emission. Therefore, the conclusions of a study strongly depend on the time frame chosen for the assessment. As there is no scientifically correct time frame, it can be recommended that different time frames should be considered. To be able to conduct the analysis under varying time frames, a method that considers the timing of emissions and sinks, and that can be used within different time spans, seems most suitable. The indicator should take cumulative radiative forcing into account, but it is advisable that this is communicated relative to a pulse emission of one mass unit of fossil CO_2. In this way, dynamics in carbon stock changes will be included and the result can be communicated in a unit familiar to a broad audience (fossil CO_2 equivalent).

The climate indicator should not include any predefined assumptions regarding the substitution impacts of biomass use because such an approach always includes uncertain assumptions on what products will be substituted with biomass. Such impacts can be analyzed independently and separately with the selected climate impact indicator to ensure transparency. Moreover, to improve understanding and usability of the results, it is crucial that the climate impacts of (i) forest biomass production and harvesting activities (ii) biomass carbon storage in long-lived products, and (iii) product or energy substitution are considered independently and reported separately.

Acknowledgements

This study was carried out in the Finnish Forest Cluster project EffFibre, funded by the Finnish Funding Agency for Technology and Innovation (TEKES) and several private companies. We thank the financiers of the study, as well as the three anonymous reviewers whose valuable comments helped to improve the quality of this study.

References

Böttcher H, Verkerk PJ, Gusti M, Havlík P, Grassi G (2012) Projection of the future EU forest CO_2 sink as affected by recent bioenergy policies using two advanced forest management models. *Global Change Biology Bioenergy*, **4**, 773–783.

Bright RM, Strømman AH, Peters GP (2011) Radiative forcing impacts of boreal forest biofuels: a scenario study for Norway in light of albedo. *Environmental Science & Technology*, **45**, 7570–7580.

Bright RM, Cherubini F, Strømman AH (2012) Climate impacts of bioenergy: inclusion of carbon cycle and albedo dynamics in life cycle assessment. *Environmental Impact Assessment Review*, **37**, 2–11.

Cherubini F, Peters GP, Berntsen T, Strømman AH, Hertwich E (2011) CO_2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. *Global Change Biology Bioenergy*, **3**, 413–426.

Cherubini F, Guest G, Strømman AH (2012) Application of probability distributions to the modelling of biogenic CO_2 fluxes in life cycle assessment. *Global Change Biology Bioenergy*, **4**, 784–798.

Curran MA, Mann M, Norris G (2005) The international workshop on electricity data for life cycle inventories. *Journal of Cleaner Production*, **8**, 853–862.

DeFries R, Houghton RA, Hansen M *et al*. (2002) Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s. *Proceedings of the National Academy of Sciences of the United States of America (PNAS)*, **99**, 14256–14261.

Directive 2009/28/EC. Directive of the European Parliament and of the Council on the promotion of the use of energy from renewable sources. The Official Journal of the European Union. Available at: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=Oj:L:2009:140:0016:0062:en:PDF (accessed 20 July 2012).

EC: Commission of the European Communities (1996) *Communication on Community Strategy on Climate Change Council Conclusions*. European Council, Brussels.

EC: Commission of the European Communities (2007) Communication from the Commission to the Council and the European Parliament – Renewable Energy Road Map; Renewable energies in the 21st century: building a more sustainable future. COM(2006) 848 final. Brussels.

EC: Commission of the European Communities(2011) DRAFT Product Environmental Footprint – General Guide. Joint Research Centre. Available at: http://ec.europa.eu/environment/eussd/product_footprint.htm (accessed 20 July 2012).

Ekvall T, Weidema BP (2004) System boundaries and input data in consequential life cycle inventory analysis. *The International Journal of Life Cycle Assessment*, **9**, 161–171.

Eriksson E, Karlsson P-E, Hallberg L, Jelse K (2010) Carbon footprint of cartons in Europe – Carbon Footprint methodology and biogenic carbon sequestration. IVL Report B1924.

Finnveden G, Hauschild MZ, Ekvall T *et al*. (2009) Recent developments in life cycle assessment. *Journal of Environmental Management*, **1**, 1–21.

Guinée JB, Heijungs R, van der Voet E (2009) A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. *The International Journal of Life Cycle Assessment*, **14**, 328–339.

Gustavsson L, Karjalainen T, Marland G, Savolainen I, Schlamadinger B, Apps M (2000) Project-based greenhouse-gas accounting: guiding principles with a focus on baselines and additionality. *Energy Policy*, **28**, 935–946.

Helmisaari H-S, Hanssen KH, Jacobson S *et al*. (2011) Logging residue removal after thinning in nordic boreal forests: long-term impact on tree growth. *Forest Ecology and Management*, **261**, 1919–1927.

Holtsmark B (2012) Harvesting in boreal forests and the biofuel carbon debt. *Climatic Change*, **112**, 415–428.

IGBP/GCP (2010) Global carbon dioxide budget. The International Geosphere-Biosphere Programme, The Global Carbon Project. Available at: http://www.globalcarbonproject.org/carbonbudget/index.htm (accessed 20 July 2012).

IPCC (2006) Agriculture, Forestry and Other Land Use. Intergovernmental Panel on Climate Change, IPCC Guidelines for National Greenhouse Gas Inventories. Availble at: http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (accessed 20 July 2012).

IPCC (2007a) Fourth Assessment Report (AR 4). Working Group I Report 'The Physical Science Basis'. Available at: http://www.ipcc.ch/ipccreports/ar4-wg1.htm (accessed 20 July 2012)

IPCC (2007b) Fourth Assessment Report (AR 4). Working Group III Report "Mitigation of Climate Change". Available at: http://www.ipcc.ch/publications_and_data/publications_and_data_reports.shtml (accessed 20 July 2012).

IPCC: Intergovernmental Panel on Climate Change (2003) Good Practice Guidance for Land Use, Land-Use Change and Forestry. Available at: http://www.ipcc-nggip.iges.or.jp/public/gpglulucf/gpglulucf.html (accessed 20 July 2012).

ISO 14040 (2006) *Environmental Management. Life cycle Assessment. Principles and Framework*. International Organization for Standardization, Geneva, Switzerland.

ISO 14044 (2006) *Environmental Management. Life cycle Assessment. Requirements and Guidelines*. International Organization for Standardization, Geneva, Switzerland.

ISO/DIS 14067 (2012) (Draft international standard). *Carbon Footprint of Products – Requirements and Guidelines for Quantification and Communication*. International Organization for Standardization, Geneva, Switzerland.

Johnson DW, Curtis PS (2001) Effects of forest management on soil C and N storage: meta analysis. *Forest Ecology and Management*, **140**, 227–238.

JRC-IES: Joint Research Centre – Institute for Environment and Sustainability (2010) *International Reference Life Cycle Data System (ILCD) handbook*. JRC-IES, Ispra.

Karjalainen T, Kellomäki S, Pussinen A (1994) Role of wood-based products in absorbing atmospheric carbon. *Silva Fennica*, **28**, 67–80.

Kauppi P, Rautiainen A, Korhonen K *et al*. (2010) Changing stock of biomass carbon in a boreal forest over 93 years. *Forest Ecology and Management*, **259**, 1239–1244.

Kendall A, Chang B, Sharpe B (2009) Accounting for time-dependent effects in biofuel life cycle greenhouse gas emission calculations. *Environmental Science & Technology*, **43**, 7142–7147.

Kilpeläinen A, Alam A, Strandman H, Kellomäki S (2011) Life cycle assessment tool for estimating net CO_2 exchange of forest production. *Global Change Biology Bioenergy*, **3**, 461–471.

Kirkinen J, Palosuo T, Holmgren K, Savolainen I (2008) Impact due to the use of combustible fuels: life cycle viewpoint and relative radiative forcing commitment. *Environmental Management*, **42**, 458–469.

Köhl M, Bastup-Birk A, Marchetti M *et al*. (2011) Criterion 3: maintenance and encouragement of productive functions of forests (wood and non-wood). In: *State of Europe's Forests 2011. Status and Trends in Sustainable Forest Management in Europe* (eds FOREST EUROPE, UNECE and FAO). pp. 51–64, Ministerial Conference on the Protection of Forests in Europe, Oslo.

Kujanpää M, Eggers J, Verkerk H, Helin T, Lindner M, Wessman H (2010) Carbon balance of forest residue collection and combustion in Southern-Finland. *Proceedings of the 18th European Biomass Conference and Exhibition*, Lyon.

Kulmala M, Suni T, Lehtinen KEJ *et al*. (2004) A new feedback mechanism linking forests, aerosols, and climate. *Atmospheric Chemistry and Physics*, **4**, 557–562.

Levasseur A, Lesage P, Margni M, Deschenes L, Samson R (2010) Considering time in LCA: dynamic LCA and its application to global warming impact assessments. *Environmental Science & Technology*, **44**, 3169–3174.

Levasseur A, Lesage P, Margni M, Brandão M, Samson R (2012) Assessing temporary carbon sequestration and storage projects through land use, land-use change and forestry: comparison of dynamic life cycle assessment with ton-year approaches. *Climatic Change*, **115**, 759–776.

Lippke B, Oneil E, Harrison R, Skog K, Gustavsson L, Sahre R (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. *Carbon Management*, **2**, 303–333.

Liski J, Lehtonen A, Palosuo T, Peltoniemi M, Eggers T, Muukkonen P, Mäkipää R (2006) Carbon accumulation in Finland's forests 1922-2004 - an estimate obtained by combination of forest inventory data with modelling of biomass, litter and soil. *Annals of Forest Science*, **63**, 687–697.

Marland G, Schlamadinger B (1997) Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. *Biomass and Bioenergy*, **13**, 389–397.

McKechnie J, Colombo S, Chen J, Mabee W, MacLean HL (2011) Forest bioenergy or forest carbon? assessing trade-offs in greenhouse gas mitigation with wood-based fuels. *Environmental Science & Technology*, **45**, 789–795.

Milà i Canals L, Bauer C, Depestele J *et al*. (2007) Key elements in a framework for land use impact assessment within LCA. *The International Journal of Life Cycle Assessment*, **12**, 5–15.

Miner R (2010) Biomass Carbon Neutrality. NCASI Discussion paper. Available at: http://nafoalliance.org/wp-content/uploads/NCASI-Biomass-carbon-neutrality.pdf (accessed 20 July 2012).

Müller-Wenk R, Brandão M (2010) Climatic impact of land use in LCA – carbon transfers between vegetation/soil and air. *The International Journal of Life Cycle Assessment*, **15**, 172–182.

Pan Y, Birdsley RA, Fang Y *et al*. (2011) A large and persistent carbon sink in the World's forests. *Science*, **333**, 988–993.

PAS 2050 (2011) *Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services*. BSI Group, London.

Pena N, Bird DN, Zanchi G, (2011) *Improved methods for carbon accounting for bioenergy*. Occasional paper 64. CIFOR, Bogor, Indonesia.

Perez-Garcia J, Lippke B, Comnick J, Manriquez C (2005) An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. *Wood and Fiber Science*, **37**, 140–148.

Pingoud K, Pohjola J, Valsta L (2010) Assessing the integrated climatic impacts of forestry and wood products. *Silva Fennica*, **44**, 155–175.

Pingoud K, Ekholm T, Savolainen I (2012) Global warming potential (GWP) factors and warming payback time as climate indicators of forest biomass use. *Mitigation and Adaptation Strategies for Global Change*, **17**, 369–383.

Rebitzer G, Ekvall T, Frischknecht R *et al*. (2004) Life cycle assessment part 1: framework, goal and scope definition, inventory analysis, and applications. *Environment International*, **30**, 701–720.

Repo A, Tuomi M, Liski J (2011) Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. *Global Change Biology Bioenergy*, **3**, 107–115.

Sathre R, O'Connor J (2010) Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy*, **13**, 104–114.

Schlamadinger B, Marland G (1996) The role of forest and bioenergy strategies in the global carbon cycle. *Biomass and Bioenergy*, **10**, 275–300.

Schlamadinger B, Spitzer J, Kohlmaier GH, Lüdeke M (1995) Carbon balance of bioenergy from logging residues. *Biomass and Bioenergy*, **8**, 221–234.

Schlamadinger B, Apps M, Bohlin F *et al*. (1997) Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. *Biomass and Bioenergy*, **6**, 359–375.

Sedjo RA (2011) *Carbon Neutrality and Bioenergy. A Zero-Sum Game ?* RFF DP 11-15. Resources for the Future, Washington DC.

Spracklen DV, Bonn B, Carslaw KS (2008) Boreal forests, aerosols and the impacts on clouds and climate. *Philosophical Transactions of the Royal Society A*, **366**, 4613–4626.

Statistics Finland (2011) Fuel classification 2011. Available at: http://www.stat.fi/tup/khkinv/khkaasut_polttoaineluokitus.html (accessed 20 July 2012).

Strauss W (2011) How Manomet got it Backwards: Challenging the "Debt-Then-Dividend" Axiom. Futuremetrics. Available at: http://www.futuremetrics.net/papers/Manomet%20Got%20it%20Backwards.pdf (accessed 20 July 2012).

UN: United Nations (1992). United Nations. Available at: http://unfccc.int/resource/docs/convkp/conveng.pdf (accessed 20 July 2012).

UNFCCC: United Nations Framework Convention on Climate Change (2009) Copenhagen Accord. United Nations. Available at: http://www.unfccc.org (accessed 20 July 2012).

Walker T, Cardellichio P, Colnes A *et al*. (2010) *Biomass Sustainability and Carbon Policy Study. Natural Capital Initiative at Manomet Report*. Manomet Center for Conservation Studies, MA, USA.

WRI: World Resources Institute (2006) *The Greenhouse Gas Protocol. The Land Use, Land-Use Change, and Forestry Guidance for GHG Project Accounting*. World Resources Institute, Washington.

Zamagni A, Guinee J, Heijungs R, Masoni P, Raggi A (2012) Lights and shadows in consequential LCA. *The International Journal of Life Cycle Assessment*, **17**, 904–918.

Zanchi G, Pena N, Bird DN (2010) The Upfront Carbon Debt of Bioenergy. Joanneum Research Report.

Supporting Information

Additional Supporting Information may be found in the online version of this article:

Table S1. Summary of the reviewed documents.

ARTICLE III

Is land use impact assessment in LCA applicable for forest biomass value chains?

Findings from comparison of use of Scandinavian wood, agro-biomass and peat for energy

The International Journal of Life Cycle Assessment
19: 770–785.

ARTICLE IV

Global warming potentials of stemwood used for energy and materials in Southern Finland

Differentiation of impacts based on type of harvest and product lifetime

Submitted manuscript.

Global warming potentials of stemwood used for energy and materials in Southern Finland: Differentiation of impacts based on type of harvest and product lifetime.

Tuomas Helin[1]*, Hannu Salminen[2], Jari Hynynen[2], Sampo Soimakallio[1], Saija Huuskonen[2], Kim Pingoud[1]

[1]VTT Technical Research Centre of Finland, Tekniikantie 2, Espoo, FI-02044 VTT, Finland

[2] Finnish Forest Research Institute METLA, P.O. Box. 18, FI-01301 Vantaa, Finland

* Corresponding author: Tuomas Helin, tel. +358 40 827 8355, current email: tuhelin@gmail.com

Abstract

Wood harvesting in boreal forests typically consists of sequential harvesting operations within a rotation: a few thinnings and a final felling. The aim of this paper is to model differentiated relative global warming potential (GWP) coefficients for stemwood use from different thinnings and final fellings, and correction factors for long-lived wood products, potentially applicable in life cycle assessment studies. All thinnings and final fellings influence the development of forest carbon stocks. The climate impact of a single harvesting operation is generated in comparison to no-harvesting, thus encountering a methodological problem on how to handle the subsequent operations. The dynamic forest stand simulator MOTTI was applied in the modelling of evolution of forest carbon stocks at landscape level in Southern Finland. The landscape-level approach for climate impact assessment gave results similar to some stand-level approaches presented in previous literature that included the same forest C pools and also studied the impacts relative to the no-harvest situation. The climate impacts of stemwood use decreased over time. For energy use the impacts were higher or similar in the short term and 0–50% lower in the mid-term in comparison with an identical amount of fossil CO2. The impacts were to some extent (approximately 20–40%) lower for wood from intermediate thinnings than for wood from final fellings or first thinnings. However, the study reveals that product lifetime has higher relative influence on the climate impacts of wood-based value chains than whether the stemwood originates from thinnings or final fellings. Although the evolution of future C stocks in unmanaged boreal forests is uncertain, a sensitivity analysis suggests that landscape-level model results for climate impacts would not be sensitive to the assumptions made on the future evolution of C stocks in unmanaged forest. Energy use of boreal stemwood seems to be far from climate neutral.

Keywords: forest bioenergy, boreal forest, climate impacts, forest growth model, GWP, landscape, long-lived wood products, stemwood, thinning wood.

Introduction

Within recent years a vivid discussion on the climate impacts of harvesting forests for material and energy has taken place in the scientific literature (e.g. Lippke *et al.*, 2011; Cherubini *et al.*, 2011; Holtsmark, 2012; Haberl *et al.*, 2012; Schulze *et al.*, 2012; Bright *et al.*, 2012; Lamers & Junginger, 2013), in other reports (JRC, 2013; Matthews *et al.*, 2014) and in the public media (BirdLife, 2010; Miner, 2010; Sedjo, 2011; Mainville, 2011; Cowie *et al.*, 2013). Assessment of the related impacts requires a comparison between the harvesting and no-harvesting scenarios studied (IPCC, 2014, Chapter 11, p. 88). The general methodological framework of life cycle assessment (LCA) is appropriate for the assessment procedure (e.g. Ness *et al.*, 2007; Finnveden *et al.*, 2009). A no-harvesting reference situation may reflect the most likely alternative option (consequential LCA) or natural regeneration (attributional LCA), depending on the goal and scope of the study (Milà i Canals *et al.*, 2007; JRC-IES, 2010; Helin *et al.*, 2013; Koellner *et al.*, 2013).

Scandinavian managed forests are long-rotation biomass production systems which, on the landscape level, enable continuous harvests of stemwood while sustaining constant or growing standing stocks (carbon pools) in the forests. From a narrow, small-owner or stand-level viewpoint, one 60–100 year stand rotation cycle consists typically of regeneration, early cleaning (optional) and precommercial thinning, first commercial thinning, one or two intermediate thinnings and final cutting, so there is a multi-decadal time-lag between regeneration operations and different harvests, and over time the carbon stock of individual stands fluctuates accordingly. Irrespective of the spatial viewpoint, a harvesting decision today will influence the evolution of forest carbon stocks for decades, or even centuries, into the future (e.g. Holtsmark, 2013). Nonetheless, the early interference of any future harvesting operation in a managed forest on carbon stock evolution (the next year at landscape level, within the next few decades at stand level), provides a limited opportunity to observe the full impacts on forest carbon stocks of an individual harvesting operation. If the aim of a study is to assess the impacts of a single wood-harvesting operation, which is typically the case in product LCA, then impacts related to future harvesting operations should be ignored from the system boundary (Helin *et al.*, 2013). This can be done by creating appropriate harvesting and no-harvesting scenarios for comparison. A number of previous studies have considered the development of forest carbon stocks either in various management scenarios (e.g. Hudiburg *et al.,* 2011; Lippke *et al.,* 2011; Routa *et al.,* 2011; Repo *et al.,* 2012; Mitchell *et al.,* 2012; Kallio *et al.,* 2013), in a simplified single clear-cut without thinnings (Cherubini *et al.,* 2011) or in wood fellings compared to a no-harvesting reference (Pingoud *et al.,* 2012; Holtsmark, 2013; Soimakallio, 2014). However, according to the authors' knowledge, there is no study available that focuses on the impacts of a single wood-harvesting operation as a part of continuous forest management.

The dynamics of forest carbon emissions and sinks influence the climate impacts of forest biomass use. Global warming potential (GWP) is a climate metric that has been introduced initially for climate policy purposes, and is widely adopted in LCA for the comparison of climate impacts of different greenhouse gases over multiple timeframes (IPCC, 2007. Table 2.14). The term GWP has originally been defined as 'the time-integrated radiative forcing due to a pulse emission of a given component, relative to a pulse emission of an equal mass of CO_2' (IPCC, 2013, Chapter 8.7.1.2). In recent years, the concept of GWP has been extended to capture the temporal pattern of emissions and sinks of biogenic CO_2. A method for determining the GWP coefficients for energy use of biomass, including long-rotation forest stemwood biomass, termed GWP_{bio}, was originally proposed in Cherubini *et al.* (2011). The GWP_{bio} factor was derived in this initial approach by approximating the atmospheric decay of carbon from long-rotation biomass using a simplified forest growth equation. Several studies have since proposed additional modifications to the GWP_{bio} indicator by considering the climate impact of forest bioenergy in comparison to a no-harvesting reference situation (Pingoud *et al.*, 2012), by the inclusion of the carbon dynamics of harvest residues (Guest *et al.*, 2013), by considering the no-harvesting reference situation and carbon dynamics of all carbon stocks in the forest (Holtsmark, 2013) and by considering impacts of delayed release from long-lived products and/or product substitution (Cherubini *et al.*, 2012; Pingoud *et al.*, 2012). An advantage of GWP_{bio} coefficients is that the climate impacts can be communicated in a unit familiar to LCA practitioners and the broad audience: fossil CO_2 equivalents.

The primary aim of this study is to model separated relative GWP factors (hereafter termed GWP_{bio} in line with previous literature) resulting from changes in forest and wood product carbon pools for harvested stemwood biomass originating from final fellings and different thinning operations in Southern Finland. The core research question is "What are the direct forest carbon stock related climate impacts of the use of wood originating from different harvest types in comparison to a no-harvest reference situation?" Another aim from the product perspective is to show how the delayed release of carbon (storage in products) influences climate impacts by compiling $GWP_{bio,product}$ correction factors potentially applicable for LCA. The influence of uncertainties in future evolution of forest C stocks on the model results is also studied. The system is examined in isolation from impacts exogenous to the studied system (that is, excluding market-mediated impacts such as product substitution and rebound effects), thus in a context that could probably be considered as attributional LCA. Regarding LCA taxonomies (attributional and consequential), bioenergy, and support for political decision-making, see a recent discussion in e.g. Plevin *et al.*, 2014; Brandão *et al.*, 2014; Suh & Yang, 2014.

Materials and methods

/ Forest growth and carbon stock modelling /

The direct climate impacts of stemwood use are modelled in this study in relation to a no-use reference situation. How do we apply this to forest biomass? Let us assume that one is producing energy today from forest biomass sourced from intermediate thinnings (harvesting activity occurring in e.g. the 40th year of a rotation cycle). This action causes a known change in the forest carbon stocks, and the future evolution of carbon stocks can be modelled for both harvesting and no-harvesting scenarios. This difference in the evolution of future carbon stocks can be allocated to the wood products obtained in that specific harvest operation, in line with the methodological suggestions given in Milà i Canals *et al.* (2007), JRC-IES (2010), Helin *et al.* (2013) and Koellner *et al.* (2013). However, subsequent harvesting operations will increase the product pool (change the function of the studied product system) and interfere with the evolution of carbon stocks in the next year in the landscape-level approach, and within the next few decades at stand level. Thus observations are limited to very short timeframes compared to the need to be able to apply data on the evolution of forest carbon stocks, for example for the next 100 years in the formulation of GWP_{bio}-100 coefficients. To be able to isolate the impact of wood harvested today from the impacts caused by past and future forest management activities, forest growth modelling is needed. Forest modelling can be applied for both stand and landscape level. The landscape-level approach (cf. Lamers & Junginger, 2013) has been selected in this study to enable information on the impacts of harvests within a given year in a given region.

The MOTTI forest stand simulator, designed to simulate stand development under alternative management regimes in Finnish growth conditions (Hynynen *et al.*, 2002; Matala *et al.*, 2003; Salminen *et al.*, 2005), was applied in the modelling of the evolution of forest carbon stocks. MOTTI produces stand projections under various management schedules built up on user-defined parameters. The core of MOTTI comprises stand-level models for stand dynamics from regeneration to sapling stage, and distance-independent individual-tree-level models for predicting development of the further stages from saplings to mature trees (Hynynen *et al.*, 2002, 2014; Siipilehto *et al.*, 2014). Distribution models provide the link from stand level to individual-based models (Siipilehto *et al.*, 2014). MOTTI's basic operational unit is a stand, and its models are usually driven in five-year time-steps. If necessary, interpolation for higher time-resolution is also applied. The performance of the MOTTI simulator has been assessed in young Scots pine stands (Huuskonen & Ahtikoski, 2005; Huuskonen, 2008), in mixed stands (Hynynen *et al.,* 2002) and in intensively managed Scots pine stands (Mäkinen *et al.*, 2005).

Growth and yield models of the MOTTI system are designed to provide predictions for assessing the impacts of alternative forest management practices in forest stands as they have been applied over recent decades (Hynynen *et al.,* 2002). The version of MOTTI used in this study also produces a description of carbon dynamics related to stocking under different management regimes. The description covers carbon sequestrated in growing stock, and carbon released from dead wood, including natural mortality and logging residues (see Hynynen et al., 2014, pp. 50–51). The amount of carbon is based on predicted living and dead biomasses; carbon mass was obtained by multiplying biomass by a constant equal to 0.5. The biomass of trees comprises foliage, stem, branches, coarse and fine roots, and stump. Along each simulation step, growth increases biomass, whereas cuttings and natural mortality turn a part of living biomass into dead wood (for details, see Hynynen et al., 2014, Appendix 1). Decomposition of above-ground dead wood was estimated using the output of MOTTI and the model by Mäkinen *et al.* (2006). Decomposition of below ground dead wood, i.e. stumps and coarse roots, was predicted with modified model of Mäkinen *et al.* (2006) based on empirical results reported by Shorohova *et al.* (2008), Melin et al. (2009) and Palviainen et al. (2010) (see Hynynen et al., 2014, pp. 52–53). Regarding soils, complete soil organic carbon stocks are not taken into account, only the changes in the carbon pools on forest floor after time t0 due to input, accumulation and decay of dead wood.

The forest modelling is initiated from the concept of a normal forest that flows directly from the principle of sustained yield (see Leslie, 1966). In this study, a normal forest consists of equally sized stands (forest compartments) – each representing one regeneration year – while the number of stands equals the length of rotation in years. Each stand is intended to be fully stocked and evenly distributed. In normal forest, one stand is regenerated and one stand final-felled in every year, and all stands are treated according to management recommendations based on geographical location, site type and tree species (Rantala, 2011). In order to reflect commercial forests on mineral soils in Southern Finland, a combination of normal forests was constructed by including the most common types of forests, each of them being represented according to its share of the total forest area. Mesic heath (*Myrtillus* type according to Cajander 1909, 1926, and Cajander & Ilvessalo 1921) is the most common site type in Southern Finland, populating 37% of the area of commercial forests. About half of this is spruce-dominated, and the other half pine-dominated. Spruce-dominated herb-rich heath (*Oxalis-Myrtillus* site type) is an economically important site type due to both its high share of the area and relatively high productivity. Two other significant site types are dryish (*Vaccinium-Myrtillus* site type) and dry heath (*Calluna* type), both typically dominated by pine. These five forest classes (site type × main species) represent about 74% of the total area of commercial forests in Southern Finland (10.9 million hectares). The dynamics of each of these forest classes was represented by one normal forest, and when composing final results the forest classes were scaled up according to their relative area; spruce-dominated herb-rich heath covered 27%, spruce-dominated mesic heath 25%, pine-dominated mesic heath 25%, pine-dominated dryish heath 20%, and pine-dominated dry heath 3% of the total area of these forest classes.

The management recommendations lead to rotation lengths of 58, 62, 68, 78, and 89 years in spruce-dominated herb-rich and mesic heath, and pine-dominated mesic, dryish and dry heath, respectively. There are three commercial thinnings during a rotation in the most fertile site while the others are thinned twice. According to these management principles, a combined normal forest in a steady state result in a total of 355 stands. Each year, management practices are carried out on 6.5% of the total forest area. First thinnings are carried out on 1.5%, intermediate thinnings on 3.5%, and final fellings on 1.5% of the total area.

The steady-state normal forest described above is used as a starting point in the analysis. Two landscape-level scenarios are constructed and compared. In the harvesting scenario, the maximum level of annual harvests that still maintain the standing stock (and carbon stock) of the forest is carried out in the 1st year of the model run, in line with the principles of sustainable forest management. No further harvests are included in years 2–100. In the no-harvest reference scenario, no harvests occur over the 100-year modelling run. Comparison of the difference in these two scenarios enables isolation of the impact of harvested wood products of an individual year from impacts caused by wood products obtained in past and future forest management activities. Another approach would be to study a harvesting scenario in which the annual sustainable levels of harvests are carried out in every year over the modelling period, and compare this to a no-use, natural regeneration scenario. From the perspective of impact assessment (LCA) of a product obtained today, allocating the impacts of potential future product systems to a product manufactured and used today does not seem relevant for the decision-maker. Thus we propose and apply a method that aims for the isolation of impacts caused by the actual harvesting actions carried out for today's product systems, and at the same time focuses on the forest landscape level, recommended by e.g. (Sedjo, 2011; Lamers & Junginger, 2013; Jonker *et al.,* 2013).

/ From forest carbon stocks to climate impacts /

The difference in the total carbon stocks of the studied forest area in harvesting and no-harvesting scenarios over time is allocated to the harvested wood products originating from the harvests in the 1st model step (year). The carbon stock difference between two modelled scenarios is considered as the impact that the initial harvesting operation would have had on the future evolution of forest carbon stocks. This difference in the C stocks is first separated into impacts originating from distinct harvesting types, namely first commercial thinnings, intermediate thinnings and final fellings, then converted into CO_2, and considered as annual pulses of net emissions or sinks of CO_2 (see an example of net impact of all harvests in Table S1). Finally the change in forest C stocks is reviewed relative to the C content of harvested stemwood (Table S2). The influence of each individual harvesting type on the future evolution of the forest carbon stocks is thus differentiated.

Modelling of GWP_{bio} is initiated by the modelling of the time integral of radiative forcing (RF), also known as absolute global warming potential (AGWP), that is caused by the difference in forest carbon stocks due to the initial biomass harvesting (Pingoud *et al.,* 2012). $AGWP_{bio}$ is here defined as

$$AGWP_{bio} = \int_0^T RF\big(S_{bio}(t)\big)dt, \qquad \text{(Eq.1)}$$

where $S_{bio}(t)$ is the change in atmospheric CO_2 concentrations due to the difference in biomass carbon stocks in the harvesting and no-harvesting scenarios (cf. Table S2). To estimate the RF in time, an impulse response model, REFUGE-3 (Pingoud *et al.,* 2012), based on the Bern Carbon Cycle Model 2.5CC (IPCC, 2007, p. 213), is applied in this study.

AGWP for the reference gas, CO_2 of fossil origin, is formulated as

$$AGWP_{fos} = \int_0^T RF\left(S_{fos}(t)\right)dt, \quad \text{(Eq.2)}$$

where $S_{fos}(t)$ is the atmospheric CO_2 concentration due to the unit pulse of fossil CO_2 and RF is estimated with the same REFUGE-3 model (Pingoud *et al.,* 2012).

Finally, the GWP_{bio} is defined as a ratio of AGWP for forest biomass over AGWP of an identical amount of reference gas, fossil CO_2 (Cherubini *et al.,* 2011).

$$GWP_{bio}(T) = \frac{AGWP_{bio}(T)}{AGWP_{fos}(T)}. \quad \text{(Eq.3)}$$

Differentiated GWP_{bio}(T) factors for stemwood are modelled separately for all distinct harvesting types.

/ Dealing with uncertainty in evolution of future C stocks /

There remains substantial uncertainty in the future evolution of carbon stocks in boreal forests if these are left unmanaged due to the unpredictability of the timing, occurrence and extent of natural disturbances and self-thinning (Kneeshaw & Gauthier, 2003; Luyssaert et al., 2008; Ter-Mikaelian et al., 2013). The MOTTI forest growth model was originally designed for growth prediction of managed forests, and the predictions do not account for the possible impact of the increasing occurrence of natural disturbances in old stands. Thus, the unadjusted output of the MOTTI model for forest carbon stocks is most probably an overestimate of the actual evolution of future carbon stocks in the no-harvesting scenario. To deal with this uncertainty in future carbon stocks in old-growth forests, a sensitivity analysis was carried out to test how the final results will change as a response to estimated minimum and maximum levels of potential forest C losses due to the occurrence of natural disturbances. The MOTTI model output was applied as the maximum estimate, and a linearly increasing deduction from 0% to 60% over the 100-year modelling run as a minimum estimate. This leads to future forest carbon stock estimates between 90 to 220 tC ha-1 (cf. Fig. 2a), which is in line with data in Liu et al. (2014) on mature and old growth boreal forests.

/ Correction factors for delayed release in long-lived products /

The GWP_{bio} indicators compiled in this study include an inherent assumption that the carbon content in the harvested forest biomass is released to the atmosphere within the 1st year after harvest. In many harvested wood product value chains this assumption is counter-factual, and correction factors are needed in order to take account of the impact of delayed release in long-lived products. Some previous estimates are presented in the scientific literature, but do not fully satisfy the need. Cherubini et al. (2012) includes GWP_{bio} factors that aggregate both the results of their forest model and the delayed release in the product system into one value, and Pingoud et al. (2012) include both substitution impacts and delayed emission in their GWP_{biouse} coefficients. Here new $GWP_{bio,product}$ correction factors are formulated that allow transparent separation of the impact of the product storage stage (delayed release) from the impacts of forest harvesting and/or product substitution.

Modifying the approach presented in Pingoud *et al.* (2012) by the exclusion of substitution impacts, the $GWP_{bio,product}$ correction factor is defined here as

$$GWP_{bio,product}(T) = \frac{AGWP_{bio,product}(T)}{AGWP_{fos}(T)} = \frac{\int_0^T RF\left(S_{seq}(t)\right)dt,}{\int_0^T RF\left(S_{fos}(t)\right)dt,} \quad \text{(Eq.4)}$$

where $S_{seq}(t)$ is the reduced CO_2 concentration due to delayed release (or permanent storage) of C in biomass products. $S_{seq}(t)$ is given a value -1 over the product lifetime τ (t: 0 → τ) and instant release to the atmosphere is assumed at the end of product lifetime for simplicity. For forest biomass use with instant release to the atmosphere in t = 0, such as bioenergy, $S_{seq}(0 \rightarrow 100)$ equals 0, thus $GWP_{bio,product}$ = 0.

Net climate impact of the forest biomass value chain, including impact of harvests and delayed emission by C storage in long-lived product, is formulated as follows:

$$GWP_{netbio}(T) = GWP_{bio}(T) + GWP_{bio,product}(T) \quad \text{(Eq.5)}$$

Results

/ Future forest C stocks, product output and uncertainty /

The difference in the total carbon stocks of the studied forest area in harvesting and no-harvesting scenarios over time is presented in Fig. 1a (solid lines), and the marginal carbon stock difference between two modelled scenarios, separated into distinct harvests, is presented in Fig. 1b. The MOTTI model result for the amount of C in harvested stemwood is presented in Table 1.

Table 1. Amount of harvested stemwood (expressed as C content) acquired from different harvesting modes in the 1st modelling step (year) in the studied normal forest. Note that stemwood output is presented relative to the area of the whole normal forest area, not relative to single forest stands.

Harvesting mode	Harvested stemwood [MgC ha-1], [(%)]
First thinnings	0.15 (9.7%)
Intermediate thinnings	0.44 (28.4%)
Final fellings	0.96 (61.9%)
All harvests, net	1.55 (100%)

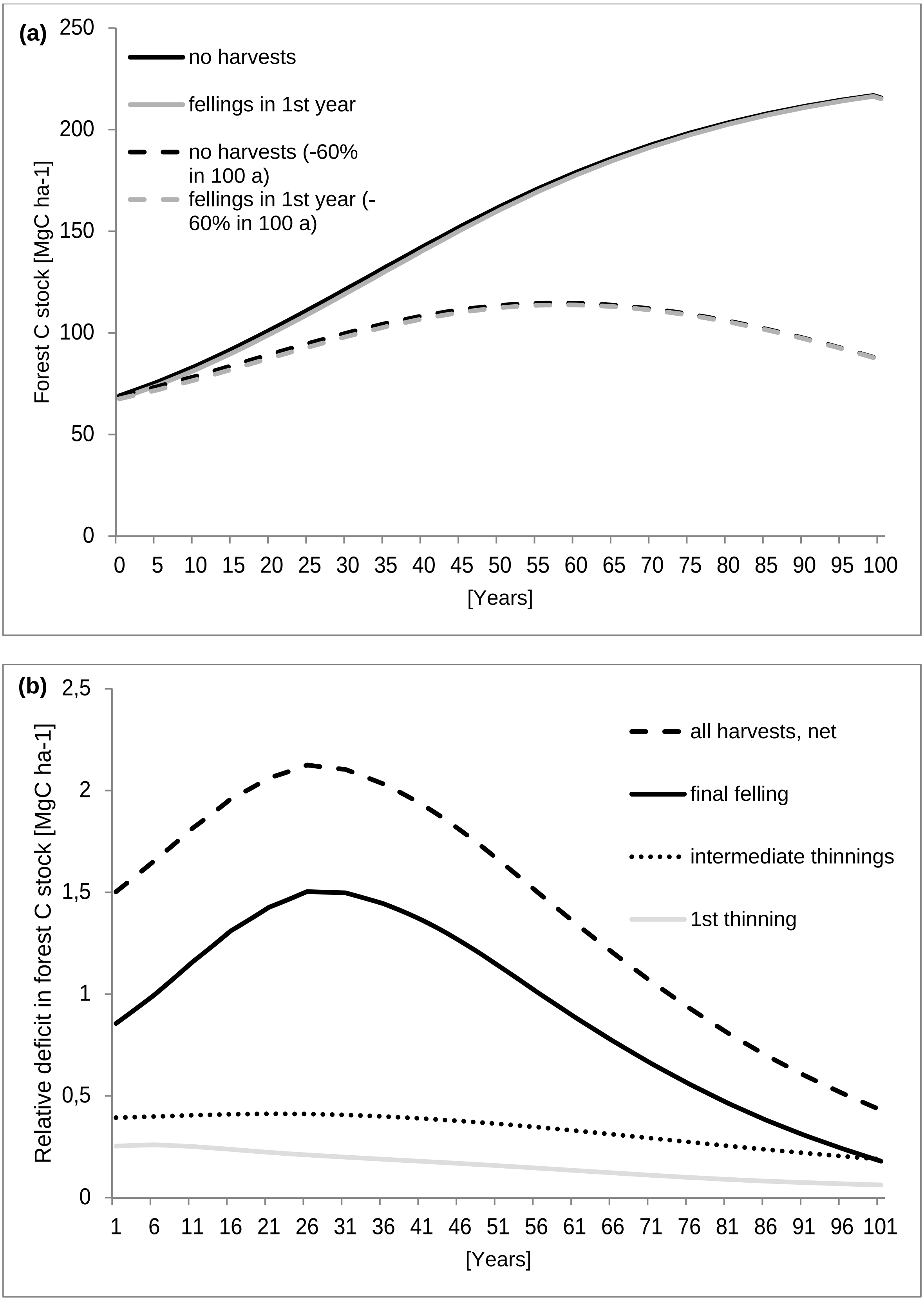

Figure 1. (a) Evolution of forest carbon stocks in scenarios of no fellings (black lines) and fellings in the 1st year (grey lines) with minimum (dashed lines) and maximum (solid lines) model estimates of future evolution forest C stocks. (b) Deficit in forest carbon stocks in the four harvesting scenarios relative to a no-harvest reference situation.

Climate impacts of stemwood harvesting and immediate combustion are indicated with the relative GWP_{bio} coefficients. The climate impacts for stemwood from final fellings and first thinnings are higher per unit of biogenic CO_2 released than for an identical amount of fossil CO_2 for the majority of the modelled 100-year time horizon (Fig. 2). Stemwood from intermediate thinnings is the only biomass fraction studied for which the GWP indicator results are smaller than for fossil CO_2 for the whole 100-year modelling horizon. The underlying reasons for such a result are that the C accumulation (growth) is slower for the following decades in a final-felled or thinned site than if the studied site is not harvested, thus leading to an increasing *relative* carbon deficit in the first decades after the studied harvest. Differentiated results for distinct harvesting types (Fig. 2) suggest that the relative climate impact of harvesting and energy use of stemwood are probably lower for wood originating from intermediate thinnings than for wood from first thinnings of final fellings, especially in the short term (a few decades). No significant difference in model result was observed for the medium to long term (60 … 100 years).

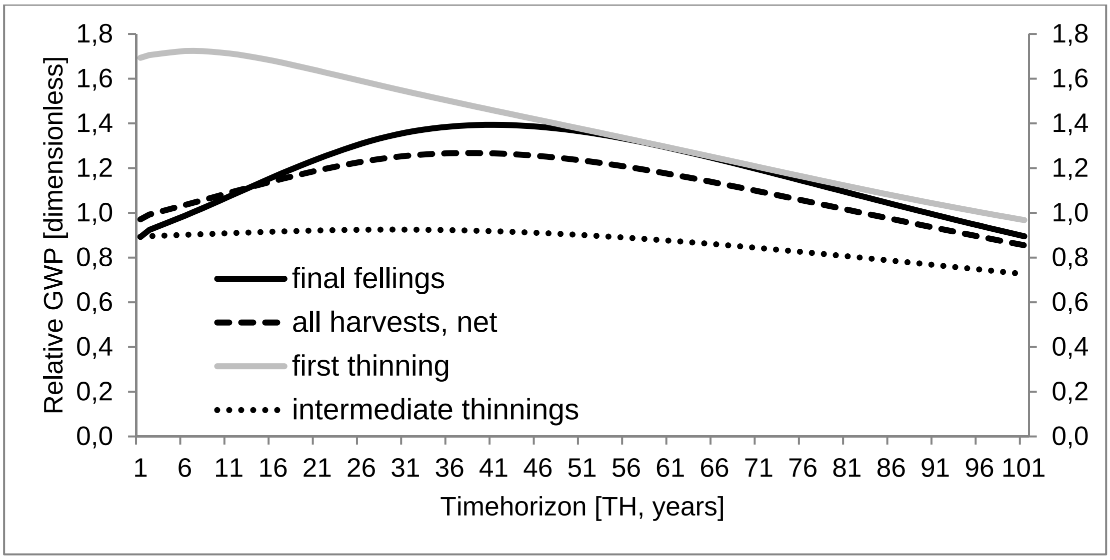

Figure 2. Relative GWP (GWP_{bio}) factors for stemwood from different harvest types for timehorizons 1…100 a.

Regardless of the significant uncertainty in the future evolution of C stocks in forests if left unmanaged, the sensitivity analysis for both differentiated and aggregate (net) harvests confirms that the results are not very sensitive to the assumptions made on the estimates of potential high and low ends of future forest C accumulation (Fig. 3). The differentiated and aggregated results for stemwood harvests overlap to a large degree in the medium to long term (GWP_{bio}-100 values in the range of 0.5…1), and the range of model results for stemwood from first thinnings and final fellings is higher than for stemwood from intermediate thinning operations in the first few decades after harvest in Fig. 3. The sensitivity analysis suggests that, irrespective of the assumptions made on the uncertain accumulation rate of C in an unmanaged forest, the climate impact of stemwood harvesting (irrespective of harvesting type) and immediate release to the atmosphere is similar, or higher, than the impact of the reference gas, fossil CO_2, when the forest is studied at a landscape level.

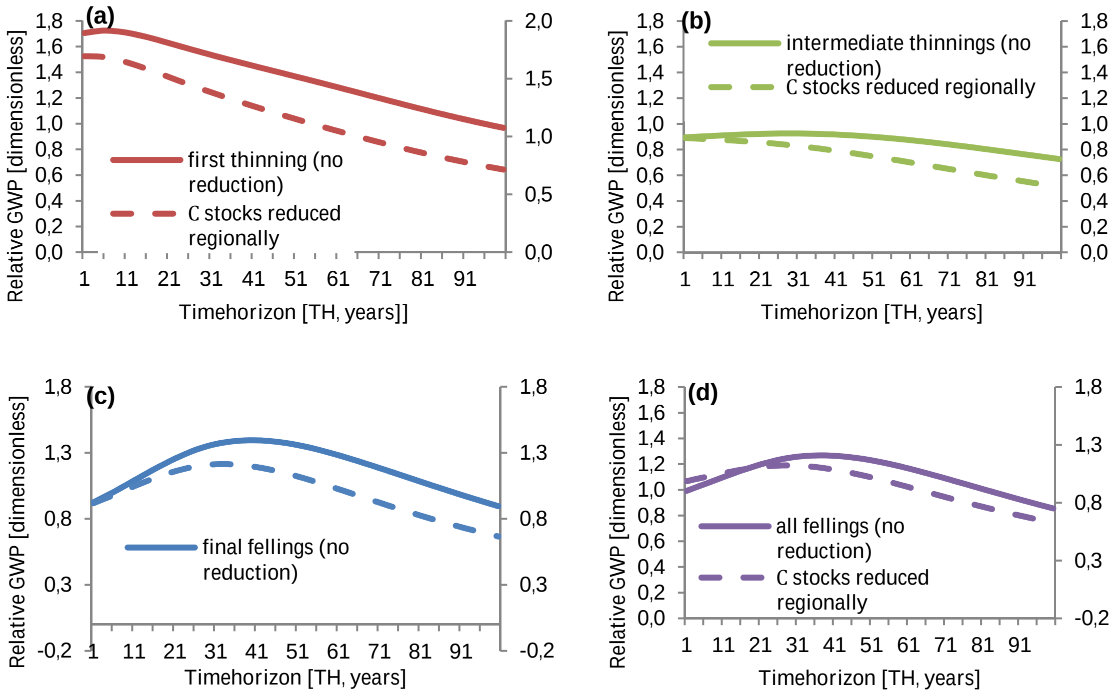

Figure 3. Uncertainty ranges of relative GWP (GWP_{bio}) factors for stemwood from (a) first thinning (b) intermediate thinnigs (c) final felling and (d) net for all harvests for timehorizons 1…100 a.

/ Impact of delayed carbon release (storage in products) /

The climate impact results presented above (Figs. 2 and 3) are representative of a forest value chain in which the C content in harvested stemwood is released to the atmosphere immediately (within the next year) after the harvest (e.g. for bioenergy). However, stemwood biomass from final fellings is most often used in the mechanical forest industry, and some fraction of the stemwood biomass may be stored in wood-based products for decades. The impact of delayed release was taken into consideration in modelling of the climate impacts of long-lived products with the $GWP_{bio,product}$ correction factors, modelled for different storage periods in this study (Fig. 4, Table 2). These correction factors can be applied flexibly in LCA studies for the fraction of stemwood biomass that is expected to be stored in wood-based products.

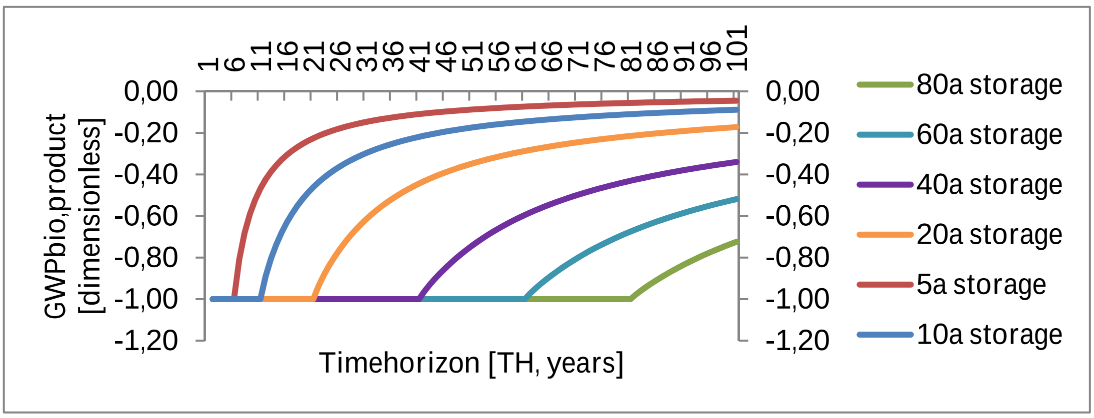

Figure 4. $GWP_{bio,product}$ correction factors for delayed release of carbon (storage in long-lived products) for different storage times over timehorizons 1…100 a.

Some examples of the net climate impacts, GWP_{netbio}, of exemplified stemwood-based forest value chains with different product lifetimes are presented in Fig. 5 and Table 2. The model results suggest that the forest value chain may reach relative climate neutrality in 100 years if the majority of the harvested biomass can be stored for 100 years. This is more likely to be reached in the case where C accumulation in old-growth forests is according to the low-end estimate of this study (Fig. 5b, light grey dashed line). Results confirm that the product lifetime has much higher relative influence on the climate impacts of the wood-based value chain than whether the stemwood raw-material originates from first thinnings, intermediate thinnings or final fellings. However, it needs to be noted that only a fraction of harvested stemwood is stored in the final wooden product (e.g. newspaper, book, furniture or construction panel) and some share of the wooden raw material is always used for process energy or for other co-products, such as wood chip co-product from mechanical wood processing or tall oil from pulp mills. The $GWP_{bio,product}$ correction factors should be applied only to the harvested wood fraction that is actually anticipated to be stored in the long-lived product pool, not to the whole stemwood biomass input to the processing plant.

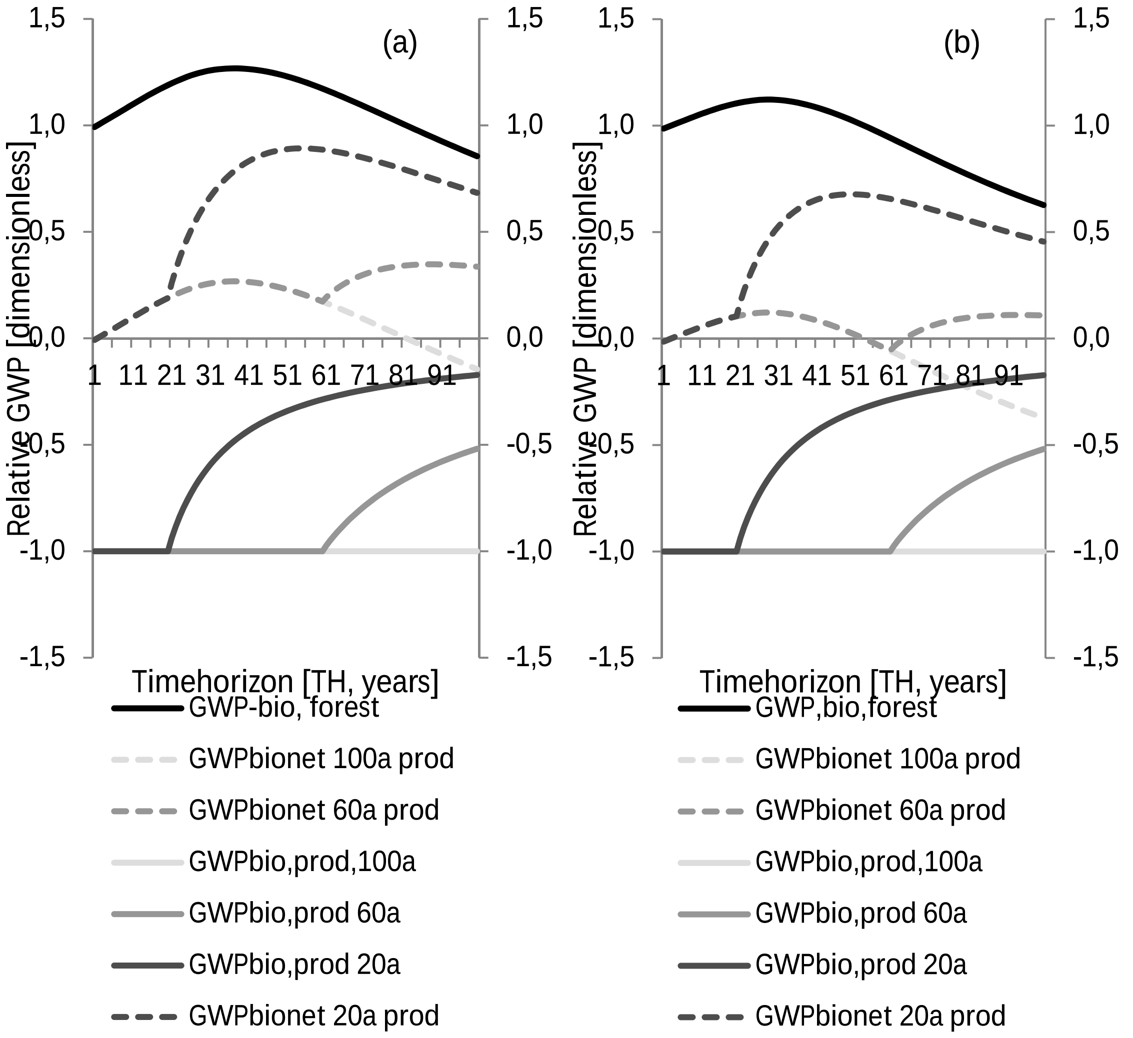

Figure 5. Relative GWP factors for timehorizons 1…100 for all harvested stemwood (solid black line above x-axis), correction factors for selected carbon storage timeframes (solid lines below x-axis) and net impacts (for sum of harvest and delayed release) for different product storage times (dashed lines). (a) High estimate and (b) low estimate for evolution of future forest C stocks.

Table 2. Relative GWP factors for timehorizons (TH) 20 a, 50 a and 100 a for (a) harvested stemwood from different harvest types, (b) $GWP_{bio,product}$ correction factors for selected carbon storage times and (c) Examples of net relative GWP_{netbio} factors for simplified cases. Min refers to minimum estimate and max to maximum result from the sensitivity analysis.

	TH = 20 a		TH = 50 a		TH = 100 a	
	min	max	min	max	min	max
(a) $GWP_{bio,forest}$						
First thinning	1.53	1.64	1.16	1.37	0.71	0.97
Intermediate thinnings	0.86	0.92	0.75	0.90	0.51	0.73
Final fellings	1.15	1.24	1.13	1.36	0.67	0.90
All harvests	1.10	1.20	1.03	1.23	0.63	0.86
(b) $GWP_{bio,product}$						
Storage 20a	-1.00		-0.35		-0.17	
Storage 60a	-1.00		-1.00		-0.52	
Storage 100a	-1.00		-1.00		-1.00	
(c) GWP_{netbio}						
First thinning + instant combustion	1.53	1.64	1.16	1.37	0.71	0.97
Intermediate thinning + 20a storage	-0.14	-0.08	0.40	0.55	0.34	0.56
Final felling + 100a storage	0.15	0.24	0.03	0.23	-0.33	-0.10

Discussion

The approach and the results presented in this study aim to answer the question regarding the direct climate impacts related to land use of the use of boreal stemwood originating from different harvest types, relative to a no-harvest reference situation. The direct impacts of the activity were studied in isolation from other product systems; that is, excluding market-mediated impacts such as potential competing uses of forested land area and forest biomass. The GWP_{bio} factors for stemwood presented in this study can only describe the climate impacts of continued, ongoing and unchanged forest value chains, and do not represent the impacts of changes in forest management or demand for wood-based products. A potential application area is in (comparative) attributional LCA studies for current or future forest value chains which do not imply changes in the current forest management regime. A potential decision-support context is, for example, micro-level decision on the selection of raw material or energy carrier in any existing or planned product value chain. The authors do not suggest the application of the results of this study in supporting macro-level political decision-making. Such change-oriented questions require identification and modelling of changes caused on exogenous systems and value chains, within so-called consequential LCA context (cf. Plevin et al., 2013), interactions which are outside the scope of the research questions in this study.

The results obtained (see Table 2a for details) show that there are significant direct climate impacts from the use of stemwood originating from all the harvest types studied. The underlying reason for such a result is that the living wood would continue growing (sequestering carbon) at a faster rate in the counterfactual baseline than occurs on the site after thinning or final-felling activity, thus contributing to a relative C deficit in the forest. The climate impacts of an instant release of biogenic CO_2 in the energy use of boreal stemwood biomass are lower (GWP_{bio}-100 = 0.5…0.95) than the climate impact of the release of a similar amount of the reference gas, fossil CO_2 (GWP-100 = 1) over a 100-year timeframe. However, in shorter timeframes, the climate impacts of energy-use of stemwood from first thinnings and final fellings are higher (GWP_{bio}-20 = 1.1…1.6 and GWP_{bio}-50 = 1.0…1.4) than for the reference gas, fossil CO_2. Stemwood from intermediate thinnings resulted in the lowest climate impacts over the whole 100-year modelling time horizon, and stemwood from first thinnings the highest. It can be anticipated, however, that in long, multi-centurial timescales the release of 1 unit of biogenic CO_2 in the energy-use of stemwood from boreal forests will result in smaller climate impact than in the release of 1 unit of fossil CO_2 in the combustion of fossil fuels (cf. Bright *et al.,* 2012; Holtsmark, 2013). The climate impacts of forest biomass use have been modelled only up to 100 years into the future, as the reliability of forest growth models, and thereby the quantitative estimates of forest C stocks, decreases when the models are applied outside their intended rotation times.

The results of this study allow better identification and differentiation of the climate impacts of the raw material acquisition phase for stemwood biomass originating from different thinnings or final fellings. Previously published estimates of GWP_{bio} for energy use of stemwood from boreal forest have focused on the wood obtained from final fellings. It needs to be stressed that stemwood from final fellings is *not* commonly applied for bioenergy in Finland (Finnish Forest Research Institute,

2013. Table 8.0), mainly due to its inherent properties and high economic value as wood-based material. However, to allow comparison, the GWP_{bio} results for stemwood from final fellings in this and previous studies is presented in Table 3. The comparison reveals that the results for the 100-year time horizon in this study are similar (but lower) than the results in Holtsmark (2013) and higher than in other studies considered.

The similarities with results in Holtsmark (2013) can be anticipated as Holtsmark's (2013) model definition bears the closest similarity to this study, that is, in including the evolution of C pool of natural deadwood and soil and presenting the results relative to a no-harvest reference. The reasons for higher GWP_{bio} results in this study than in Cherubini *et al.* (2011), Pingoud *et al.* (2012), Guest *et al.* (2013) and Soimakallio (2014) very probably stem from differences in modelling system boundaries. Pingoud *et al.* (2012), Cherubini *et al.* (2011), Guest *et al.* (2013) and Soimakallio (2014) do not include the C pool in natural deadwood, which was included in the present study. This C pool in natural deadwood is higher in a no-harvest scenario than in managed forests (cf. Liu *et al.*, 2014), thus the inclusion of the C pool in deadwood leads to higher C stock difference between a harvesting and no-harvesting scenario (thus higher climate impact). Additionally, Cherubini *et al.* (2011) and Guest *et al.* (2013) did not consider no-harvest reference situations (and respectively increasing C stocks), which leads to lower climate impact indicator results.

Table 3. A comparison of estimates of GWP_{bio} for stemwood from final fellings in the previous literature and in this study. TH = time horizon. Note that only the values from previous literature that do not include the collection of forest residues are presented here to enable inter-model comparison.

	TH = 20 a	TH = 50 a	TH = 100 a
Cherubini *et al.,* 2011	1	n.a.	0.4
Pingoud *et al.,* 2012	1	0.9	0.6
Guest *et al.,* 2013	1.3	n.a.	0.6
Holtsmark, 2013	1.6…1.9	>2	1.1…1.5
Soimakallio 2014	0,9…1.0	n.a.	0.2…0.7
Final fellings, this study	1.1…1.3	1.1…1.4	0.7…0.9

It is common practice to utilise stemwood from first and intermediate thinnings in direct energy use. Thus it is interesting to compare the results obtained in this study to those published previously for other wood fractions widely used for energy in Finland: harvest residues and stumps. Pingoud *et al.* (2012) and Soimakallio (2014) have estimated that the GWP_{bio} for branches would be circa 0.6 and 0.3 for 20- and 100-year timeframes, respectively, results that are significantly lower than the model estimates for thinning wood obtained in the current study. Figure 5 in Repo *et al.* (2012) would imply that the relative GWP_{bio}-100 coefficient for stumps and young stand thinning wood[1] would be circa 0.6, thus similar to stemwood from intermediate thinnings and lower than for stemwood from first thinnings over a 100-year timeframe.

The approach of this study is future oriented. At the same time, it is known that the reason for rapid C accumulation in managed forest, especially in the case of no future harvests, is caused by past forest management actions and strategies that have been implemented to optimise and increase the growth rate of quality stemwood. As the past and current management regime has led to the enhancement of forest biomass growth, and as the Faustmann rule of optimal net present value of wood harvests leads to the final felling in managed forests occurring before the growth curve has culminated (Faustmann, 1849; Holtsmark, 2013), and as thinnings and final fellings occur long after the slow C accumulation phase in the first steps of forest succession, all these factors lead to rapid C accumulation in the no-use scenario. This rapid forest C accumulation rate in a no-harvesting reference situation is one of the main reasons for the high relative climate impacts attributed in this study to forest biomass use, not the long rotation period of boreal stemwood suggested in previous literature that does not consider the C stock evolution in the absence of harvesting decision (e.g. Cherubini *et al.,* 2011, Table 3). The length of rotation period, *per se*, is not the key variable in climate impacts of biomass use when the no-use reference scenario is considered in the assessment. From the viewpoint of burden sharing between different sectors of society in climate mitigation efforts, although one can only influence the future, one should give at least some consideration and potential credit to active past management decisions made in the forest sector that have led to the rapid C accumulation and thus C sequestration rate in managed forestlands in Finland.

In summary, the climate impact of energy use of boreal stemwood was found to be higher in the short term and lower in the long term in comparison with fossil fuels that emit an identical amount of CO_2 in combustion, due to changes implied in forest C stocks. The climate impacts of energy use of boreal stemwood were found to be higher than previous estimates suggest regarding forest residues and stumps. The product lifetime was found to have a much higher influence on the climate impacts of wood-based value chains than the origin of stemwood either from thinnings or final fellings. Climate neutrality seems to be likely only in the case where almost all the carbon of harvested wood is stored in long-lived wooden products.

Acknowledgements

This study was carried out in the ECOSUS project funded by the Academy of Finland. We thank the financiers of the study and the three anonymous reviewers whose comments helped in improving the study.

---

[1] Repo *et al.,* (2012) refer to young stand thinning (equal to first commercial thinnings) residuals left for forest flood to decompose. The current study models the climate implications of this thinning operation relative to no thinnings occurring at all.

References

BirdLife (2010) Bioenergy. A carbon accounting time bomb. Birdlife International. Available at: www.birdlife.org/eu/pdfs/carbon_bomb_21_06_2010.pdf (Accessed on 20.9.2013)

Brandão M, Clift R, Cowie A, Greenhalgh S (2014) The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on “Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation …”. Journal of Industrial Ecology 18, 461-463.

Bright RM, Cherubini F, Astrup R et al., (2012) A comment to “Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral”: important insights beyond greenhouse gas accounting. Global Change Biology Bioenergy, 4, 617–619.

Cajander AK (1909) Über Waldtypen [About forest types]. Fennia 28 (2): 1–-175 (in German).

Cajander AK, Ilvessalo Y (1921) Über Waldtypen, II [About forest types, II]. Acta Forestalia Fennica 20: 1–-77 (in German).

Cherubini F, Peters GP, Berntsen T, Strømman AH, Hertwich E (2011) CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. Global Change Biology Bioenergy, 3, 413–426.

Cherubini F, Guest G, Strømman AH (2012) Application of probability distributions to the modelling of biogenic CO2 fluxes in life cycle assessment. Global Change Biology Bioenergy, 4, 784–798.

Cowie A, Berndes G, Smith T (2013) On the timing of greenhouse gas mitigation benefits of forest-based bioenergy. IEA Bioenergy Executive Committee statement 2013:4. Available at: http://www.ieabioenergy.com/LibItem.aspx?id=7787 (Accessed on 20.9.2013)

Faustmann M (1849) Berechnung des Werthes, weichen Waldboden sowie nach nicht haubare Holzbestande f€ur de Weltwirtschaft besitzen. Allgemeine Forst und Jagd Zeitung, 25, 441.

Finnish Forest Research Institute METLA (2013) Finnish Statistical Yearbook of Forestry. Chapter 8 Wood Consumption. Metsäntutkimuslaitos METLA, Vantaa. ISBN 978-951-40-2450-4.

Finnveden G, Hauschild MZ, Ekvall T et al., (2009) Recent developments in life cycle assessment. Journal of Environmental Management, 1, 1–21.

Guest G, Cherubini F, Strømman AH (2013) The role of forest residues in the accounting for the global warming potential of bioenergy. Global Change Biology Bioenergy, 5, 459–466.

Haberl H, Sprinz D, Bonazountas M et al., (2012) Correcting a fundamental error in greenhouse gas accounting related to bioenergy. Energy Policy 45, 18-23.

Helin T, Sokka L, Soimakallio S, Pingoud K, Pajula T (2013) Approaches for inclusion of forest carbon cycle in life cycle assessment – a review. Global Change Biology Bioenergy, 5, 475–486, doi: 10.1111/gcbb.12016

Holtsmark B (2012) Harvesting in boreal forests and the biofuel carbon debt. Climatic Change, 112, 415–428.

Holtsmark B (2013) Quantifying the global warming potential of CO2 emissions from wood fuels. Global Change Biology Bioenergy (in press) doi: 10.1111/gcbb.12110

Hudiburg, TW, Law BE, Wirth C, Luyssaert S (2011) Regional carbon dioxide implications of forest bioenergy production. Nature Climate Change 1:419-423.

Huuskonen S, Ahtikoski A (2005) Ensiharvennuksen ajoituksen ja voimakkuuden vaikutus kuivahkon kankaan männiköiden tuotokseen ja tuottoon. Metsätieteen aikakauskirja 99-115.

Huuskonen S (2008) The development of young Scots pine stands - precommercial and first commercial thinning [In Finnish]. Dissertationes Forestales 62. 61 p.

Hynynen J, Ojansuu R, Hökkä H, Siipilehto J, Salminen H, Haapaka P, (2002) Models for predicting stand development in MELA system. The Finnish Forest Research Institute. Research Papers 835, ISBN 951-40-1815-X, ISSN 0358–4283

Hynynen J, Salminen H, Huuskonen S et al. (2014) Impact of alternative management and utilization of Finnish forest resources on the raw material supply for forest and energy industry. Working papers of Finnish Forest Research Institute 302. 106 pp.

IPCC (2007) Fourth Assessment Report (AR 4). Working Group I Report ‘The Physical Science Basis’.

IPCC (2013) Working group I contribution to the IPCC fifth assessment report (AR5), Climate change 2013: the physical science basis.

IPCC (2014) Working group III contribution to the IPCC fifth assessment report (AR5), Climate change 2014: Mitigation of Climate Change.

Jonker JGG, Junginger M, Faaij A (2013) Carbon payback period and carbon offset parity point of wood pellet production in the South-eastern United States. Global Change Biology Bioenergy (in press) doi: 10.1111/gcbb.12056

JRC-IES (2010) International Reference Life Cycle Data System (ILCD) handbook. Joint Research Centre – Institute for Environment and Sustainability. JRC-IES, Ispra.

Kallio AMI, Salminen O, Sievänen R (2013). Sequester or substitute—Consequences of increased production of wood based energy on the carbon balance in Finland. Journal of Forest Economics, 19(4), 402-415.

Kneeshaw D, Gauthier S (2003) Old growth in the boreal forest: A dynamic perspective at the stand and landscape level. Environmental Reviews 11(S1): S99-S114. DOI 10.1139/a03-010

Koellner T, de Baan L, Beck T, Brandão M, Civit B, Margni M, Milà i Canals L, Saad R, de Souza DM, Müller-Wenk R (2013) UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment (in press). DOI 10.1007/s11367-013-0579-z

Lamers P, Junginger M (2013) The 'debt' is in the detail: A synthesis of recent temporal forest carbon analyses on woody biomass for energy. Biofuels, Bioproducts and Biorefining 7 (4), 373-385.

Leslie AJ (1966) A review of the concept of the normal forest. Australian Forestry, 30, 139-147.

Lippke B, Oneil E, Harrison R, Skog K, Gustavsson L, Sahre R (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. Carbon Management, 2, 303–333.

Liu Y, Yu G, Wang Q, Zhang Y (2014) How temperature, precipitation and stand age control the biomass carbon density of global mature forests. Global Ecology and Biogeography 23, 323-333.

Luyssaert S, Schulze ED, Börner A, Knohl A, Hessenmöller D, Law BE, Ciais P, Grace J (2008) Old-growth forests as global carbon sinks. Nature 455:213-215

Mainville N (2011) Fuelling a BioMess. Why burning trees for energy will harm people, the climate and forests. Greenpeace Canada. Available at: http://www.greenpeace.org/canada/Global/canada/report/2011/10/ForestBiomess_Eng.pdf (Accessed on 20.9.2013)

Mäkinen H, Hynynen J, Isomäki A (2005) Intensive management of Scots pine stands in southern Finland: First empirical results and simulated further development. Forest Ecology and Management, 215, 37-50.

Mäkinen H, Hynynen J, Siitonen J, Sievänen R (2006) Predicting the decomposition of Scots pine, Norway spruce, and birch stems in Finland. Ecological Applications, 16, 1865-1879.

Matala J, Hynynen J, Miina J et al. (2003) Comparison of a physiological model and a statistical model for prediction of growth and yield in boreal forests. Ecological Modelling 161, 95-116.

Melin Y, Petersson H, Nordfjell T (2009) Decomposition of stump and root systems of Norway spruce in Sweden—A modelling approach. Forest Ecology and Management, 257, 1445-1451.

Milà i Canals L, Bauer C, Depestele J et al., (2007) Key elements in a framework for land use impact assessment within LCA. The International Journal of Life Cycle Assessment, 12 (1), 5-15.

Miner R (2010) Biomass Carbon Neutrality. NCASI Discussion paper. Available at: http://nafoalliance.org/wp-content/uploads/NCASI-Biomass-carbon-neutrality.pdf (accessed 20 September 2013).

Mitchell SR, Harmon ME, O'Connell KEB (2012) Carbon debt and carbon sequestration parity in forest bioenergy production. Global Change Biology Bioenergy, 4, 818-827.

Ness B, Urbel-Piirsalu E, Anderberg S, Olsson L (2007) Categorising tools for sustainability assessment. Ecological economics 60, 498-508.

Palviainen M, Finér L, Laiho R, Shorohova E, Kapitsa E, Vanha-Majamaa I (2010) Carbon and nitrogen release from decomposing Scots pine, Norway spruce and silver birch stumps. Forest Ecology and Management, 259, 390-398.

Pingoud K, Ekholm T, Savolainen I (2012) Global warming potential (GWP) factors and warming payback time as climate indicators of forest biomass use. Mitigation and Adaptation Strategies for Global Change, 17, 369–383.

Plevin RJ, Delucchi MA, Creutzig F (2014) Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation Benefits Misleads Policy Makers. Journal of Industrial Ecology 18, 73-83.

Rantala S (Ed.) (2011) Finnish forestry practice and management. Metsäkustannus, Helsinki.

Repo A, Känkänen R, Tuovinen J-P, Antikainen R, Tuomi M, Vanhala P, Liski J (2012) Forest bioenergy climate impact can be improved by allocating forest residue removal. Global Change Biology Bioenergy 4, 202-212.

Routa J, Kellomäki S, Kilpeläinen A, Peltola H, Strandman H (2011) Effects of forest management on the carbon dioxide emissions of wood energy in integrated production of timber and energy biomass. Global Change Biology Bioenergy 3, 483-497.

Salminen H, Lehtonen M, Hynynen J (2005) Reusing legacy FORTRAN in the MOTTI growth and yield simulator. Computers and Electronics in Agriculture 49, 103–113.

Schulze E.-D, Körner C, Law BE, Haberl H, Luyssaert S (2012), Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral. Global Change Biology Bioenergy, 4: 611–616. doi: 10.1111/j.1757-1707.2012.01169.x

Sedjo RA (2011) Carbon Neutrality and Bioenergy. A Zero-Sum Game ? RFF DP 11-15. Resources for the Future, Washington DC. 9 p + Appx

Siipilehto J, Ojansuu R, Miina J, Hynynen J, Valkonen S, Saksa T (2014) Metsikön varhaiskehityksen kuvaus MOTTI-ohjelmistossa [In Finnish]. Metlan Työraportti / Metla Working Paper 286, 1-43.

Shorohova E, Kapitsa E, Vanha-Majamaa I (2008) Decomposition of stumps 10 years after partial and complete harvesting in a southern boreal forest in Finland. Canadian Journal of Forest Research, 38, 2414-2421.

Soimakallio S (2014) Toward a More Comprehensive Greenhouse Gas Emissions Assessment of Biofuels: The Case of Forest-Based Fischer−Tropsch Diesel Production in Finland. Environmental Science & Technology 48, 3031-3038.

Suh S, Yang Y (2014) On the uncanny capabilities of consequential LCA. The International Journal of Life Cycle Assessment 19, 1179-1184.

Ter-Mikaelian MT, Colombo SJ, Chen J (2013) Effects of harvesting on spatial and temporal diversity of carbon stocks in a boreal forest landscape. Ecology and Evolution 3(11): 3738–3750.

Supporting information legends

Electronic supplementary 1. Data for annual pulses of net emissions and sinks of CO_2 in tabulated form. Tables S1 and S2.

Table S1. Annual pulses of net emissions and sinks of CO_2 derived from the difference in forest carbon stocks in several scenarios, relative to a no-harvest scenario. Studied cases include only first thinnings in 1st modelling year (FT scenario), only intermediate thinnings in 1st modelling year (IT scenario), only final fellings in 1st modelling year (FF scenario) and sum of all harvests in 1st modelling year (AH scenario). Carbon in stemwood harvested in 1st modelling year is presented in red.

Table S2. Annual pulses of net emissions and sinks of biomass C relative to C content of harvested stemwood.

Series title and number	VTT Science 75

Title	**Evaluating land-use related environmental impacts of biomass value chains for decision-support** Comparison and testing of methodologies proposed for environmental life cycle impact assessment
Author(s)	Tuomas Helin
Abstract	Life cycle assessment (LCA) is one of the most established quantitative tools for environmental impact assessment of products. To be able to provide support to environmentally-aware decision makers on environmental impacts of biomass value-chains, the scope of LCA methodology needs to be augmented to cover land-use related environmental impacts. This dissertation focuses on analysing and discussing potential impact assessment methods, conceptual models and environmental indicators that have been proposed to be implemented into the LCA framework for impacts of land use. The applicability of proposed indicators and impact assessment frameworks is tested from practitioners' perspective, especially focusing on forest biomass value chains. The impacts of land use on biodiversity, resource depletion, climate change and other ecosystem services is analysed and discussed and the interplay in between value choices in LCA modelling and the decision-making situations to be supported is critically discussed. It was found out that land use impact indicators are necessary in LCA in highlighting differences in impacts from distinct land use classes. However, many open questions remain on certainty of highlighting actual impacts of land use, especially regarding impacts of managed forest land use on biodiversity and ecosystem services such as water regulation and purification. The climate impact of energy use of boreal stemwood was found to be higher in the short term and lower in the long-term in comparison with fossil fuels that emit identical amount of CO2 in combustion, due to changes implied to forest C stocks. The climate impacts of energy use of boreal stemwood were found to be higher than the previous estimates suggest on forest residues and stumps. The product lifetime was found to have much higher influence on the climate impacts of wood-based value chains than the origin of stemwood either from thinnings or final fellings. Climate neutrality seems to be likely only in the case when almost all the carbon of harvested wood is stored in long-lived wooden products. In the current form, the land use impacts cannot be modelled with a high degree of certainty nor communicated with adequate level of clarity to decision makers. The academia needs to keep on improving the modelling framework, and more importantly, clearly communicate to decision-makers the limited certainty on whether land-use intensive activities can help in meeting the strict mitigation targets we are globally facing.
ISBN, ISSN	ISBN 978-951-38-8208-2 (Soft back ed.) ISBN 978-951-38-8209-9 (URL: http://www.vtt.fi/publications/index.jsp) ISSN-L 2242-119X ISSN 2242-119X (Print) ISSN 2242-1203 (Online)
Date	January 2015
Language	English, Finnish abstract
Pages	63 p. + app. 65 p.
Name of the project	
Commissioned by	
Keywords	Life cycle assessment, environmental impacts of land use, biodiversity, resource depletion, ecosystem services, climate regulation, decision-support
Publisher	VTT Technical Research Centre of Finland P.O. Box 1000, FI-02044 VTT, Finland, Tel. 020 722 111

	Julkaisun sarja ja numero VTT Science 75
Nimeke	**Biomassa-arvoketjujen maankäyttöön liittyvien ympäristövaikutusten arviointi päätöksenteon tueksi** Vaikutusarviointimenetelmien vertailu ja testaus elinkaariarvioinnin viitekehyksessä
Tekijä(t)	Tuomas Helin
Tiivistelmä	Elinkaariarviointi (LCA) on yksi vakiintuneimmista tuotteiden ympäristövaikutusten arvioinnin työkaluista. LCA-menetelmää tulisi laajentaa kattamaan myös maankäytön ympäristövaikutukset, jotta sillä voitaisiin antaa tukea ympäristönäkökohdat huomioivaan päätöksentekoon myös biomassa-arvoketjujen osalta. Tässä väitöskirjassa analysoidaan vaikutusarviointimenetelmiä, konsepteja ja ympäristövaikutusindikaattoreita, joita on ehdotettu käytettäväksi maankäytön vaikutusten arviointiin LCA:ssa. Ehdotettujen indikaattorien ja vaikutusarvioinnin lähestymistapojen soveltuvuutta testataan LCA-mallintajan näkökulmasta, keskittyen erityisesti metsäbiomassa-arvoketjuihin. Analyysi ja pohdinta keskittyvät maankäytön vaikutuksiin luonnon monimuotoisuudelle, resurssien ehtymiselle, ilmastonmuutoksen hillinnälle ja muille ekosysteemipalveluille. LCA-mallinnuksessa tehtävien arvovalintojen ja tuettavien päätöksentekotilanteiden riippuvuuksia pohditaan myös kriittisesti. Maankäytön vaikutusindikaattorien käytön voidaan todeta olevan välttämätöntä, jotta eri maankäyttöluokkien vaikutusten erilaisuus voidaan huomioida. Useita avoimia kysymyksiä kuitenkin liittyy siihen, onnistutaanko menetelmällä osoittamaan maankäytön ympäristövaikutuksia todenmukaisesti, erityisesti talousmetsämaankäytön ympäristövaikutusten osalta. Epäselvyyksiä liittyy erityisesti vaikutuksiin luonnon monimuotoisuuteen ja luonnon ekosysteemipalveluihin, esimerkkeinä veden puhdistuskyky ja virtaaman hallinta. Boreaalisen metsän runkopuun energiakäytön ilmastovaikutukset todettiin lyhyellä aikavälillä suuremmiksi ja pitkällä aikavälillä pienemmiksi kuin niiden fossiilisten polttoaineiden, joiden poltossa vapautuu vastaava määrä hiilidioksidia. Runkopuun energiakäytön vaikutus syntyy metsän hiilivarastoille aiheutetuista muutoksista. Runkopuun energiakäytön ilmastovaikutus todettiin suuremmaksi kuin aiemmat arviot ovat hakkuutähteiden ja kantojen energiakäytölle. Puuntuotteista havaittiin, että tuotteen eliniällä on raaka-aineen hakkuutyyppiä (harvennus tai päätehakkuu) suurempi vaikutus puutuotearvoketjun kokonaisilmastovaikutuksiin. Ilmastoneutraalius vaikuttaa todennäköiseltä ainoastaan silloin, jos lähes kaikki hakkuussa korjattu puubiomassan hiili saadaan varastoitua pitkäikäisiin puutuotteisiin. Maankäytön ympäristövaikutuksia ei pystytä nykymuodossaan mallintamaan riittävän luotettavasti, eikä niitä pystytä viestimään riittävän yksiselitteisesti päätöksentekijöille. Tiedeyhteisön on jatkettava mallinnuskehikon parantamista. Päätöksentekijöille on viestittävä erityisen selkeästi, ettei tiedetä riittävän luotettavasti, voivatko paljon maa-alaa vaativat toiminnot auttaa saavuttamaan tiukat globaalin ympäristönmuutoksen hillinnän tavoitteet.
ISBN, ISSN	ISBN 978-951-38-8208-2 (nid.) ISBN 978-951-38-8209-9 (URL: http://www.vtt.fi/publications/index.jsp) ISSN-L 2242-119X ISSN 2242-119X (Painettu) ISSN 2242-1203 (Verkkojulkaisu)
Julkaisuaika	tammikuu 2015
Kieli	Englanti, suomenkielinen tiivistelmä
Sivumäärä	63 s. + liitt. 65 s.
Projektin nimi	
Rahoittajat	
Avainsanat	Life cycle assessment, environmental impacts of land use, biodiversity, resource depletion, ecosystem services, climate regulation, decision-support
Julkaisija	VTT PL 1000, 02044 VTT, puh. 020 722 111

Evaluating land-use related environmental impacts of biomass value chains for decision-support

Comparison and testing of methodologies proposed for environmental life cycle assessment

This thesis focuses on analysing and discussing potential impact assessment methods, conceptual models and environmental indicators that have been proposed to be implemented into the LCA framework for impacts of land use. The applicability of proposed indicators and impact assessment frameworks is tested from practitioners' perspective, especially focusing on forest biomass value chains. The impacts of land use on biodiversity, resource depletion, climate change and other ecosystem services is analysed and discussed and the interplay in between value choices in LCA modelling and the decision-making situations to be supported is critically discussed.

The climate impacts of energy use of boreal stemwood were found to be higher than the previous estimates suggest on forest residues and stumps. The product lifetime was found to have much higher influence on the climate impacts of wood-based value chains than the origin of stemwood either from thinnings or final fellings. Climate neutrality is likely only if almost all the carbon of harvested wood is stored in long-lived wooden products.

Many open questions remain on certainty of highlighting actual impacts of land use, especially regarding impacts of managed forest land use on biodiversity and ecosystem services such as water regulation and purification. The academia needs to keep on improving the modelling framework, and more importantly, clearly communicate to decision-makers the limited certainty on whether land-use intensive activities can help in meeting the strict mitigation targets we are globally facing.

ISBN 978-951-38-8208-2 (Soft back ed.)
ISBN 978-951-38-8209-9 (URL: http://www.vtt.fi/publications/index.jsp)
ISSN-L 2242-119X
ISSN 2242-119X (Print)
ISSN 2242-1203 (Online)