THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

# Life cycle assessment in the development of forest products

## Contributions to improved methods and practices

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**Life cycle assessment in the development of forest products**

Contributions to improved methods and practices

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**Abstract**

The prospect of reducing environmental impacts is a key driver for the research and development (R&D) of new forest products. Life cycle assessment (LCA) is often used for assessing the environmental impact of such products, e.g. for the purpose of guiding R&D. The aim of this thesis is to improve the methods and practices of LCA work carried out in the R&D of forest products. Six research questions were formulated from research needs identified in LCA work in five technical inter-organisational R&D projects. These projects also provided contexts for the case studies that were used to address the research questions. The main contributions of the research are as follows:

Regarding the planning of LCA work in inter-organisational R&D projects, the research identified four characteristics that appear to be important to consider when selecting the roles of LCAs in such projects: (i) the project's potential influence on environmental impacts, (ii) the degrees of freedom available for the technical direction of the project, (iii) the project's potential to provide required input to the LCA, and (iv) access to relevant audiences for the LCA results.

Regarding the modelling of future forest product systems, it was found that (i) it is important to capture uncertainties related to the technologies of end-of-life processes, the location of processes and the occurrence of land use change; and (ii) the choice of method for handling multi-functionality can strongly influence results in LCAs of forest products, particularly in consequential studies and in studies of relatively small co-product flows.

Regarding the assessment of environmental impacts of particular relevance for forest products, it was found that using established climate impact assessment practices can cause LCA practitioners to miss environmental hot-spots and make erroneous conclusions about the performance of forest products vis-à-vis non-forest alternatives, particularly in studies aimed at short-term impact mitigation. Also, a new approach for inventorying water cycle alterations was developed, which made it possible to capture catchment-scale effects of forestry never captured before.

To connect the LCA results to global challenges, a procedure was proposed for translating the planetary boundaries into absolute product-scale targets for impact reduction, e.g. to be used for evaluating interventions for product improvements or for managing trade-offs between impact categories.

*Keywords: R&D, LCA, wood, forestry, impact assessment, scenario modelling, end-of-life modelling, allocation, multi-functional, planetary boundaries*

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I would also like to gratefully acknowledge the financial support from the EU FP7 grant 246434 WoodLife, the Mistra Future Fashion research programme, the VINNOVA-funded ForTex project, the Mistra-funded GreenGasoline project, and VINNOVA, KK-stiftelsen, SSF and RISE through their financing of the EcoBuild Institute Excellence Centre and the CelluNova project.

Finally, I would like to express my gratitude to friends and family for believing in me, and, above all, Anna-Maj, for always being there.

**List of publications**

This thesis is based on the following papers, which are referred to in the text by their roman numerals. The papers are appended at the end of the thesis.

I. Sandin G, Clancy G, Heimersson S, Peters GM, Svanström M, ten Hoeve M, 2014. Making the most of LCA in technical inter-organisational R&D projects. *Journal of Cleaner Production* 70, 97–104.

II. Sandin G, Peters GM, Svanström M, 2014. Life cycle assessment of construction materials: the influence of assumptions in end-of-life modelling. *International Journal of Life Cycle Assessment* 19, 723–731.

III. Sandin G, Røyne F, Berlin J, Peters GM, Svanström M, 2015. Allocation in LCAs of biorefinery products: implications for results and decision making. *Journal of Cleaner Production,* doi:10.1016/j.jclepro.2015.01.013.

IV. Røyne F, Peñaloza D, Sandin G, Berlin J, Svanström M, 2015. Climate impact assessment in LCAs of forest products: implications of method choice for results and decision-making. Submitted to the *Journal of Cleaner Production*.

V. Sandin G, Peters GM, Svanström M, 2013. Moving down the cause-effect chain of water and land use impacts: an LCA case study of textile fibres. *Resources, Conservation and Recycling* 73, 104–113.

VI. Sandin G, Peters GM, Svanström M, 2015. Translating the planetary boundaries into impact reduction targets in LCA. Submitted to the *International Journal of Life Cycle Assessment*.

Work related to the thesis has also been presented in the following publications.

A. Sandin G, Peters GM, Pilgård A, Svanström M, Westin M, 2011. Integrating sustainability consideration into product development: a practical tool for identifying critical social sustainability indicators and experiences from real case application. In: Finkbeiner M (ed.). Towards life cycle sustainability management. Springer, Dordrecht, the Netherlands, pp 3–14.

B. Sandin G, Pilgård A, Peters GM, Svanström M, Ahniyaz A, Fornara A, Johansson Salazar-Sandoval E, Xu Y, 2012. Environmental evaluation of a clear coating for wood: toxicological testing and life cycle assessment. Conference proceedings of the 8th International PRA Woodcoatings Congress, Amsterdam, the Netherlands.

C. Peters GM, Svanström M, Roos S, Sandin G, Zamani B, 2015. Carbon footprints in the textiles industry. In: Muthu SS (ed.). Handbook of LCA of textiles and clothing. Elsevier (in press).

## Contribution report

The author of this thesis has made the following contributions to the papers.

I. Main author. Main contributor to formulating the research questions, collecting data, analysing data and results, and discussing the results.

II. Main author. Main contributor to formulating the research questions and carrying out the life cycle assessment (system modelling, collecting inventory data, characterising inventory data and interpreting results).

III. Main author. Main contributor to formulating the research questions and carrying out the life cycle assessment (system modelling, collecting inventory data, characterising inventory data and interpreting results).

IV. Co-author. Main contributor to summarising the background section (Figure 1) and to structuring the decision contexts (Table 2). Active in formulating the research questions, analysing the literature review, formulating the case studies, and analysing and discussing the results.

V. Main author. Main contributor to formulating the research questions and carrying out the life cycle assessment (system modelling, collecting inventory data, characterising inventory data and interpreting results).

VI. Main author. Main contributor to formulating the research questions, collecting data, and calculating, analysing and discussing the results.

**Abbreviations**

EC = European Commission

EU = European Union

FSC = Forest Stewardship Council

LCA = life cycle assessment

LCI = life cycle inventory analysis

LCIA = life cycle impact assessment

GHG = greenhouse gas

Glulam = glue-laminated

GWP = global warming potential

ILCD = international reference life cycle data system

IPCC = Intergovernmental Panel on Climate Change

ISO = International Organisation for Standardisation

MA = millennium ecosystem assessment

MCDA = multi-criteria decision analysis

PEF = product environmental footprint

PEFC = Programme for the Endorsement of Forest Certification

PVC = polyvinyl chloride

R&D = research and development

SETAC = Society of Environmental Toxicology and Chemistry

TEOW = terrestrial ecoregions of the world

UN = United Nations

UNEP = United Nations Environment Programme

UV-Vis = ultraviolet-visible

WFN = Water Footprint Network

WULCA = water use in life cycle assessment

**Content**


1 Introduction ... 1
1.1 The promise of forest products ... 2
1.2 Environmental assessment of future forest products ... 3
1.3 Research questions ... 4
1.4 Overall methodological approach ... 7
1.5 Guide for readers ... 7
2 Contexts to the case studies ... 9
2.1 WoodLife ... 9
2.2 CelluNova and ForTex ... 10
2.3 GreenGasoline ... 11
2.4 Mistra Future Fashion ... 12
3 Theory and methods ... 13
3.1 Strengths and weaknesses of forest products ... 13
3.1.1 Renewability ... 13
3.1.2 Climate change ... 15
3.1.3 Biodiversity loss and water cycle disturbances ... 15
3.1.4 Biodegradability ... 16
3.1.5 Other aspects of the environmental impact of forest products ... 16
3.1.6 Concluding remarks ... 17
3.2 LCA methodology ... 17
3.2.1 Integrating LCA work in R&D processes ... 21
3.3 Theory and methods of specific importance for the research questions ... 23
3.3.1 LCA work in technical inter-organisational R&D projects ... 23
3.3.2 Scenario modelling and sensitivity analysis ... 24
3.3.3 End-of-life modelling ... 29
3.3.4 Handling multi-functionality ... 30
3.3.5 Assessing environmental impacts ... 32
3.3.6 Connecting LCAs to global challenges ... 41
3.4 Positioning the thesis within systems science and systems analysis ... 43
4 Summary of Papers I–VI ... 49
4.1 Paper I ... 49
4.2 Paper II ... 49
4.3 Paper III ... 50
4.4 Paper IV ... 51
4.5 Paper V ... 52
4.6 Paper VI ... 53
5 Discussion of research findings ... 55
5.1 Planning LCA work in technical inter-organisational R&D projects ... 55
5.2 Modelling future forest product systems ... 56

5.3 Assessing environmental impacts of forest products ............................... 59
5.4 Connecting LCAs to global challenges ................................................... 67
6 Conclusions ............................................................................................. 71
7 Future research needs ............................................................................. 75
8 References ............................................................................................... 79

# 1 Introduction

Humankind has entered a new geological epoch, the Anthropocene, in which we are transforming the geology and ecology of the Earth system at a global scale (Steffen et al. 2007). This transformation has been particularly profound following "the great acceleration" after the Second World War – a time period characterised by rapid expansion of the global population, economy, material use and energy use (Steffen et al. 2015a) – with immense consequences for climate (Intergovernmental Panel on Climate Change (IPCC) 2013) and ecosystems (Cardinale et al. 2012; Millennium Ecosystem Assessment (MA) 2005; Chapin et al. 2000). The environmental pressures on the Earth system have been summarised by the "planetary boundaries" concept, which suggests nine biophysical boundaries that are intrinsic for the Earth system and important not to transgress to avoid risks of abrupt, non-linear, irreversible functional collapses in ecosystems and disastrous consequences for humanity (Steffen et al. 2015b; Rockström et al. 2009). Out of the nine planetary boundaries, at least four are considered to have been transgressed due to anthropogenic pressures: changes in biosphere integrity, climate change, land-system change and changes in biogeochemical flows (Steffen et al. 2015b). Others have also pointed out the risks of transgressing biophysical thresholds and thereby causing a "state shift in the Earth's biosphere" (Barnosky et al. 2012) or a global "regime shift" in social-ecological systems (Crépin et al. 2012). The global environmental crisis is also shown by "ecological footprint" calculations, which quantify humankind's pressure on the Earth system by accounting for the water and land area needed to meet our demand from nature and assimilate the generated waste. Humankind's ecological footprint is currently estimated to be about 50% larger than what the Earth can provide for (Global Footprint Network 2014). Another reason for concern is society's dependency on scarce, finite and/or non-renewable resources, for example highlighted in the discussions on "peak oil" (Owen 2010; Sorrell et al. 2009), "peak phosphorus" (Reijnders 2014; Beardsley 2011; Sverdrup & Ragnarsdóttir 2011), "peak rare earth metals" (Ragnarsdóttir et al. 2012) and "peak farmland" (Ausubel et al. 2013).

The production and use of products are major causes behind the environmental degradation and the dependency on finite resources, and there is widespread international agreement that development of environmentally improved products is important for addressing these challenges (United Nations (UN) 2012). The importance of environmentally improved products is also apparent in the next version of the International Organisation for Standardisation’s (ISO) standard 14001 – a widely used international standard for environmental management systems in industry – which is currently under revision (Lewandowska & Matuscak-Flejszman 2014). According to proposals, the standard will expand its scope from organisation-oriented to product life cycle-oriented, will require continuous improvements of the output of organisations (i.e. products and services) and will require the integration of environmental consideration into product design and development processes (Lewandowska & Matuscak-Flejszman 2014).

The venture of developing environmentally improved products is, however, a grand one, as expressed for example by bold targets of reducing the resource intensity per provided service unit (sometimes termed eco-efficiency) in industrial sectors or countries by a factor of 4, 10, 20 or even 50 (Reijnders 1998). The venture is particularly grand if humankind simultaneously intends to reach the UN Millennium Development Goals and increase the standard of living for the world’s poor (UN Millennium Project 2005) – which will most probably require increased resource use in the lives of hundreds of millions of people – on a planet expected to be home to more than 9 billion of us by 2050 (UN 2011). Regardless of how much more environmentally efficient the products of tomorrow must be in order for us to stay safely within the planetary boundaries, avoid a state shift in the Earth system, manage finite yet essential resources, support an increasing population and allow development for the less privileged, the message is clear: the environmental impact and resource intensity of products must be considerably reduced.

## 1.1 The promise of forest products

Increased production of products derived from forest biomass – henceforth denoted “forest products” – at the expense of non-forest products, is often seen as a means for tackling the aforementioned environmental challenges (note that a material derived from forest biomass and used in a product is, for sake of simplicity, termed a forest product in this thesis). This prospect is based on the abundant availability of forest biomass in many parts of the world and some environmentally favourable properties of forest biomass compared with many other feedstocks. For example, forest biomass is biodegradable and, if derived from well-managed forests, renewable and potentially carbon and climate neutral. The promise of forest products has led to many initiatives for more efficient and multifaceted use of forest biomass (e.g. in so-called biorefineries) and an increased interest in the research and development (R&D) of new forest products. For example, many European public funding bodies support R&D of new forest products, sometimes as part of wider R&D programmes focussing on biotechnologies and the bioeconomy (see, e.g., BioInnovation 2014; European Commission (EC) 2014a, 2014b, 2014c; WoodWisdom-net 2014; VINNOVA 2013; Formas 2012). Forest products are, however, not necessarily environmentally preferable compared to non-forest alternatives. For example, forestry and the transformation of non-managed to managed forests can cause biodiversity loss, which in turn can undermine many ecosystem services that are essential for human livelihood (MA 2005). Also, the subsequent production processes in the forest product value-chain can be demanding both in terms of non-renewable energy and non-forest materials, which can more than offset the benefits of using forest biomass as the main feedstock.

### 1.2 Environmental assessment of future forest products

To be able to utilise the full environmental promise of forest products, there is a need to assess the potential environmental implications – advantageous as well as disadvantageous – of new forest products. The results of such environmental assessments can be used to guide the development of new forest products and the design of new forest product value-chains, for example in terms of the sourcing of forest biomass, the management of forests, and the development, optimisation and siting of production processes and subsequent processes in the product life cycle (e.g. waste handling). Besides, the results of environmental assessments can be used for guiding the allocation of public and private funding to future R&D of forest products and for guiding purchases made by consumers or public procurers. Overall, there are many ways in which environmental assessments can contribute to ensuring that future forest products make sense in environmental terms and in supporting their market diffusion.

Performing environmental assessments early in the R&D of forest products is particularly useful as the opportunities for influencing the properties of a product (such as its environmental performance) are greatest in early stages of development and more difficult and expensive later on in the development or once the product has been commercialised (McAloone & Bey 2009; Yang & Shi 2000; Steen 1999; Verganti 1997). As a consequence, to attract public funding for R&D projects aimed at product development, it is sometimes even a requirement to assess the environmental performance of the product under development (Tilche & Galatola 2008).

The topic of this thesis is advancements in the methods and practices[1] of environmental assessments applied in early stages in the R&D of forest products. The thesis focusses on one of the most widely used tools for the environmental assessment of products: life cycle assessment (LCA), which is recognised as the best available method for transparent and reliable assessments of environmental performance in industry (Baitz et al. 2013).

## 1.3 Research questions

The thesis addresses some fundamental challenges of using LCA in the R&D of forest products, primarily those dealt with in Papers I–VI (see list of publications, page vii). The aim of the thesis is formulated in six research questions, as listed below. Before each research question, there is a paragraph introducing the challenge addressed by the research question. These challenges are further described in Chapter 3. The questions are sorted in four categories according to the order in which an LCA practitioner would encounter them in an R&D project aimed at developing a forest product.

<u>Planning LCA work in technical inter-organisational R&D projects</u>

The R&D context in focus in the thesis is publicly-funded, technical, inter-organisational, inter-disciplinary R&D projects concerned with early stages of product development. As mentioned above, in Europe this is a common setting both for the development of forest products and for carrying out environmental assessments of products. Such projects often involve firms, universities and research institutes from different countries and areas of expertise, with varying reasons for joining the project and various expectations of the project outcomes. This creates a high degree of organisational and cultural complexity, which can make it challenging to agree on the roles of LCA in the project, plan LCA work in accordance with the selected roles, influence decision-making in the project (e.g. in terms of the technical direction of the development work), and adapt the project work to unexpected LCA results. These challenges can make it difficult to utilise the full potential of LCA for assessing environmental impacts and influencing product development in an environmentally preferable direction. Because of these complexities, it is of the uttermost importance – already in the pre-project planning – to select appropriate roles for the LCA in the project. How the selection of roles can be improved is the topic of research question 1.

[1] Throughout the thesis, “method” refers to a procedure, often to some extent systematic or formalised, whereas “practice” is the use of a method in a context.

1. Which project characteristics determine the availability of roles for LCAs in technical inter-organisational R&D projects?

Modelling future forest product systems

In early stages of R&D, many aspects of the future product system are inherently uncertain. For example, the end-of-life processes of forest products such as buildings and other constructions are expected to occur in a distant future (for buildings, often 50–100 years after manufacturing (Frijia et al. 2011)) and are thus associated with considerable uncertainties connected to technological change. Such technological uncertainties of end-of-life processes can relate to how constructions are demolished (e.g. in terms of the type of energy used), how demolished materials are transported from the demolition site to further reprocessing, what the demolished materials are used for (e.g. reuse, recycling or energy recovery), and what reused or recycled materials, or recovered energy, will eventually replace (e.g. primary or recycled materials, non-renewable or renewable energy). Another typical uncertainty of future forest products is the geographical location of production processes and whether or not land use change will occur. These uncertainties may be particularly important for forest products, as some environmental impacts of forestry are strongly dependent on the location of forestry and the occurrence of land use change. For example, biodiversity impacts may depend on the local richness of biodiversity and soil structure (Curran et al. 2011) and climate impacts may depend on site-specific land management practices and regional weather patterns (Müller-Wenk & Brandão 2010). The uncertainties of future forest product systems must be captured in LCAs to provide meaningful results and robust decision support. Research question 2 addresses which uncertainties to consider in LCAs of future forest products.

2. Which inherent uncertainties of future forest product systems should be considered in LCAs?

Another typical feature of forest product systems is multi-functional processes (also called multi-output processes). For example: forestry often provides timber, pulpwood and fuelwood; the subsequent production often yields several products (e.g. in the case of biorefineries: paper pulp, fuels, heat and chemicals); and the waste handling may provide recyclable materials, reusable materials, heat and/or electricity. In LCAs, multi-functionality becomes a problem when it is not feasible to split a multi-functional process into sub-processes connected to specific functions. The LCA practitioner needs to find a rationale for allocating the environmental load of the multi-functional process between its functions. As biorefineries become more common and more integrated, and more products are produced at each biorefinery, multi-functionality problems become an increasingly common challenge in LCAs of forest products. The handling of multi-functionality in LCAs of forest products is the topic of research question 3.

3. What are the consequences of the choice of method for handling multi-functionality in LCAs of forest products?

<u>Assessing environmental impacts of forest products</u>

Established methods for assessing environmental impacts in LCAs may fail to sufficiently address some impacts of particular relevance for forestry and forest products. These impacts include climate change (Müller-Wenk & Brandão 2010), biodiversity loss (Curran et al. 2011) and disturbances to the water cycle (Bruijnzeel 2004; Swank et al. 2001). As mentioned above, location-dependencies are one reason for why these impacts may not be sufficiently captured by established methods. To improve the decision support provided by LCAs of forest products, there is a need to develop impact assessment methods that can capture aspects of the environmental impact of forest products in a more relevant way than is done by established methods, and, until better methods are available, find ways of handling the shortcomings of established methods. These challenges are addressed by research questions 4 and 5.

4. What are the potential shortcomings of current methods and practices for climate impact assessment in LCAs, in decision contexts relevant for forest products?
5. How can we improve the assessment of biodiversity loss and water cycle disturbances of forestry?

<u>Connecting LCAs to global challenges</u>

An interesting question is to what extent the potential environmental benefits offered by future forest products are sufficient in the perspective of global environmental challenges. For instance, to what extent can forest products help in the endeavour towards respecting the planetary boundaries as defined by Steffen et al. (2015b)? Understanding this can help those involved in the R&D of forest products to evaluate whether the environmental benefits offered by a product represent a substantial contribution towards reducing environmental impacts, or merely a modest step that needs to be accompanied by other more drastic measures for impact reduction. This understanding can also help to prioritise and manage trade-offs between environmental impacts, to guide the direction of future R&D projects, to guide strategic work in firms or industrial sectors, and to guide the design of public calls for R&D funding that in turn influence the direction of future R&D projects. How this understanding can be achieved is addressed by research question 6.

6. Can we use the planetary boundaries for setting product-scale targets for impact reduction in LCAs?

### 1.4 Overall methodological approach

The above-described research questions were formulated based on the demand for improvements of LCA methods and practices identified through LCA work carried out as part of five specific R&D projects. The five projects are publicly funded, technical, inter-organisational, inter-disciplinary R&D projects, each concerning the development or evaluation of new forest products (the projects are further described in Chapter 2).

The research questions were then primarily addressed by using results and experiences from the LCA work in the five projects. Through the collaboration with other researchers – particularly in the work with Papers I, III and IV – it has been possible to use also the experiences and results from other R&D projects (as further described in the respective papers).

In the research, the overall methodological approach has been to look for methods available in the literature, select appropriate methods, when necessary develop them further and/or come up with new methods, and apply them in the R&D projects. In addition, the papers describe reflections on goal achievements, difficulties encountered in the process and opportunities for the further development of methods, in particular in relation to applications in the development of new forest products.

### 1.5 Guide for readers

Chapter 2 gives further background and context for the thesis by describing the five projects that provided context for the case studies that addressed the research questions. Chapter 3 gives a comprehensive account of the theory and methods used to address the research questions. Chapter 4 summarises the content and findings of Papers I–VI. Chapter 5 contains a discussion on how the papers contribute to addressing the research questions. Chapter 6 summarises the main conclusions from this discussion and Chapter 7 the identified future research needs.

# 2 Contexts to the case studies

This chapter presents the five projects that provided contexts for the case studies that were used to address the research questions. The projects are typical examples of the R&D context in focus in the thesis: publicly-funded, technical, inter-organisational, inter-disciplinary R&D projects.

## 2.1 WoodLife

The WoodLife project lasted from 2010 to 2013. The objective of the project was to improve the UV-protection properties of water-based clear coatings, and the strength of water-based adhesives, intended for wood-based construction products. This could potentially widen the scope of application for wooden materials, for example allowing wood to replace more energy-intensive materials or materials of non-renewable origin (e.g. aluminium or polyvinyl chloride (PVC) in window frames, respectively) and thereby reduce the environmental impact of construction products. Improved coatings were to be created by the inclusion of metal oxide nanoparticles (particles with diameters of 1–100 nm) that absorb light with wavelengths in the ultraviolet-visible (UV-Vis) range (more specifically, 250–440 nm) and thereby protect the coated wood surface from UV degradation (which is mainly due to degradation of lignin at the surface; lignin constitutes 30% of the mass of wood). Improved adhesives were to be developed by designing silica and clay nanoparticles with surface properties that make them compatible with adhesive binders. Introducing nanoparticles could potentially improve the heat- and moisture-resistance of wood-adhesive joints of water-based adhesives, thereby making them more competitive in comparison with formaldehyde-based adhesives, for load-bearing applications such as glue-laminated (glulam) wooden beams. Formaldehyde-based adhesives are significant sources of emissions of formaldehyde, a toxic and volatile compound known to be a human health concern (United States Department of Health and Human Services 2014). Fig. 1 illustrates the project idea.

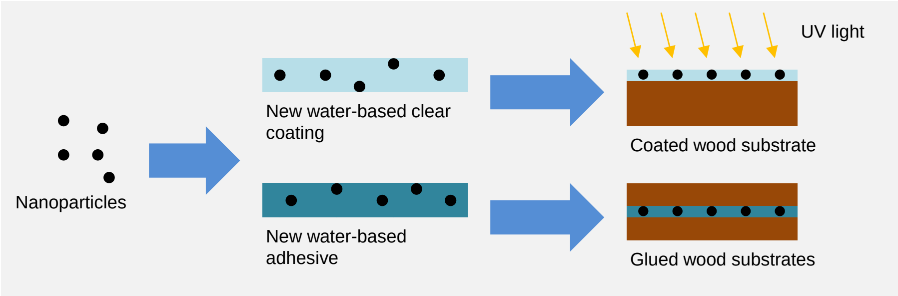


**Fig. 1** Visualisation of the WoodLife project idea. The idea was to add nanoparticles and thereby improve the UV-protecting properties of clear coatings, and the strength of adhesives, for wood applications.

The project was funded from both private and public (the European Seventh Framework Programme) sources and involved 11 participating organisations, including universities, research institutes and private companies. Project work covered the development of metal oxide and clay nanoparticle dispersions; the development of hybrid binders with nanoparticles; the development of coating and adhesive formulations; testing of nanoparticles, clear coatings and adhesives (e.g. characterisation of the physical properties of the particles and natural exposure field tests of coatings and adhesives); sustainability assessment (including LCA work) of the developed technologies; and technology demonstration, validation and exploitation. Work carried out in WoodLife has primarily provided input to Papers I and II and research questions 1 and 2.

### 2.2 CelluNova and ForTex

The CelluNova project lasted from 2009 to 2012, and the follow-up project, ForTex, lasted from 2012 to 2014. The projects aimed at developing a new process for dissolving and spinning wood pulp into textile fibres. Such regenerated cellulose fibres already exist on the market (e.g. viscose fibres), but the CelluNova and ForTex projects aimed at developing an environmentally superior process, producing fibres that can be blended with cotton fibres into a textile material with cotton-like qualities (as illustrated in Fig. 2). Such a fibre could reduce the textile industry's dependence on cotton – a fibre associated with substantial use of pesticides, fertilisers and water (Dai & Dong 2014; Shen & Patel 2008; Tariq et al. 2007; Chapagain et al. 2006). The relatively low environmental impact was to be achieved by integrating the process into a pulp mill, for example by using chemicals well-known to the pulp mill operators and utilising energy generated as a by-product in the pulping process.

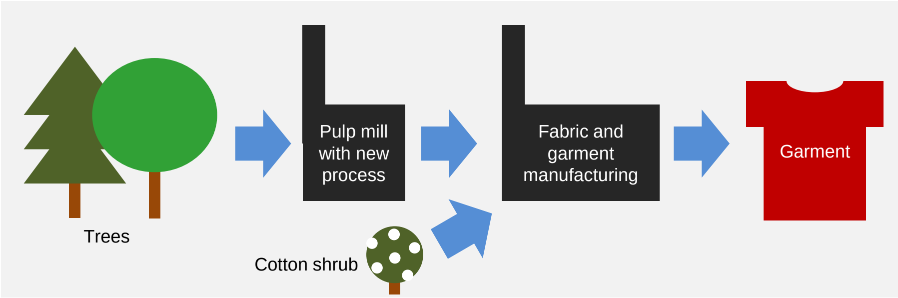


**Fig. 2** Visualisation of the project idea of the CelluNova and ForTex projects. The idea was to develop a new process which can turn wood pulp into a textile fibre, which can be blended with cotton fibres into a textile of cotton-like quality.

The projects were funded by both private and public (among others, the Swedish Governmental Agency VINNOVA) sources and involved 14 participating organisations, including universities, research institutes and private companies. Project work focussed on dissolution of cellulose, spinning of fibres, textile manufacturing and testing, full-scale modelling of the process, sustainability assessment (including LCA) of the developed fibre, and preparation for building a pilot plant. Work carried out in the CelluNova and ForTex projects has primarily provided input to Papers I and V and research questions 1, 2 and 5.

### 2.3 GreenGasoline

The GreenGasoline project lasted from 2012 to 2014. The project aimed at developing a new process for recovering lignin from the black liquor stream of a pulp mill and purifying it into a precursor for an automotive fuel (as illustrated in Fig. 3). Such a fuel could potentially replace fossil fuels and thereby reduce the climate impact and fossil resource dependency of automotive transportation.

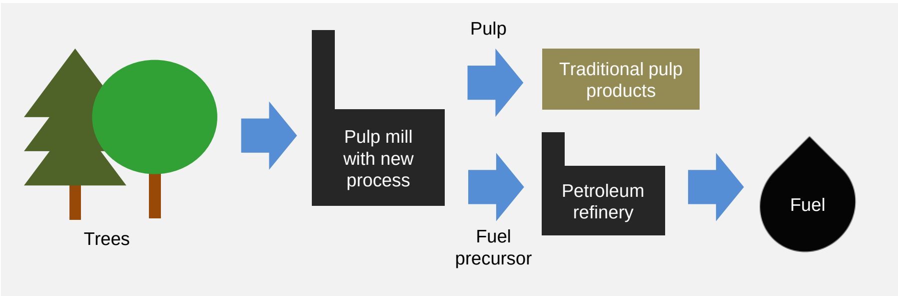


**Fig. 3** Visualisation of the GreenGasoline project idea. The idea was to develop a new process, integrated into a pulp mill, which can turn the lignin in the black liquor into a fuel precursor.

The project was funded by both private and public (the Swedish Foundation for Strategic Environmental Research, Mistra) sources and involved five participating organisations, including a university, a research institute and private companies. Project work covered the technical development of the process (lignin recovery through membrane separation, lignin purification, lignin blending and fluid catalytic cracking); computer modelling of the process; economic and environmental assessments (LCA) of the developed fuel; and the establishment of a plan for pilot testing. Work carried out in GreenGasoline has primarily provided input to Paper III and research question 3.

### 2.4 Mistra Future Fashion

Mistra Future Fashion is an ongoing (the first phase: 2011–2015) research programme aimed at new insights and solutions that can increase the sustainability of the Swedish fashion industry and strengthen its global competitiveness. The programme is publicly funded (by the Swedish Foundation for Strategic Environmental Research, Mistra) and involves 21 participating organisations, including universities, research institutes and private companies. The programme consists of eight subprojects, each focussing on a specific dimension of the fashion industry that influences sustainability: business models, fashion design, development of demonstrators from new bio-based fibres, technical development of reuse and recycling processes, fashion in the public sector, consumption and consumer behaviour, policy instruments, and sustainability assessment of the fashion industry (including LCA work). Thus, the Mistra Future Fashion research programme has a much broader scope than the other projects that the research in this thesis is based upon. The research programme connects to the technological focus of the thesis, since increased use of forest products (i.e. regenerated cellulose fibres from forest biomass) is emphasised as a measure that could potentially increase the sustainability of the fashion industry. Thus, such fibres are considered in the subproject on developing demonstrators from new bio-based fibres, and evaluated in the sustainability assessment subproject. Work carried out in the Mistra Future Fashion research programme has primarily provided input to Paper VI and research question 6.

# 3 Theory and methods

Section 3.1 describes strengths and weaknesses of forest products, thereby outlining key drivers for the development of forest products and key reasons for why environmental assessments are needed to ensure that forest products are environmentally superior. Section 3.2 presents LCA methodology in more detail. Section 3.3 elaborates on the theoretical and methodological aspects focussed on in the research questions and how these were addressed in Papers I–VI. Section 3.4 positions the thesis in the scientific disciplines of systems science and systems analysis.

## 3.1 Strengths and weaknesses of forest products

As described in Chapter 1, and as apparent from the projects presented in Chapter 2, the prospect of reducing environmental impacts is a common driver for projects aimed at developing forest products. It has been argued that forest products in general tend to have favourable environmental performance compared to non-forest alternatives (Miner et al. 2014; Buyle et al. 2013; Taylor 2013; Werner & Richter 2007), but the use of forest biomass as a feedstock is no guarantee that the end product is environmentally superior to non-forest alternatives.

### 3.1.1 Renewability

The potential renewability of forest biomass is an often recognised advantage compared to, for example, abiotic resources subject to scarcity. For forest biomass to be renewable, it must originate from forests which have a constant or growing stock of biomass. Whether this can be claimed depends on a number of factors.

In the world as a whole, biomass stocks in boreal and temperate forests are growing (Liski et al. 2003)[2], whereas the stocks in tropical rain forests are decreasing (IPCC 2013). The biomass stocks may however be decreasing in certain temperate and boreal regions, and there may be constant or growing stocks of biomass in certain tropical regions. Whether or not forest biomass from boreal, temperate or tropical regions can be seen as renewable thus depends on the geographical location of the forestry and the study's geographical resolution. The geographical resolution is sometimes described as a choice between a single-stand

[2] Swedish forest biomass stocks doubled in 1926–2008 (Swedish University of Agricultural Sciences 2011).

and a landscape approach. A single-stand approach means accounting for the re-growth of biomass on the same stand as the harvested biomass, while a landscape approach means considering the forest in a larger area – including different age classes – as a unit and accounting for net biomass increase or decrease of this unit (Cherubini et al. 2013).

The temporal perspective of the study also matters. For example, despite a historical and present increase in stocks, the biomass stocks of temperate and boreal forests may not increase in the future when the products under development today will be produced. The recent increase in boreal biomass stocks is partly a result of long-term recovery from forest degradation in earlier centuries – as noted by Kauppi et al. (2010) for forests in Finland – and this increase may not continue once the historical biomass stocks have been re-established. Indeed, there are signs of a saturation of forest re-growth in Europe (Nabuurs et al. 2013). Moreover, although a higher atmospheric carbon dioxide concentration may induce more biomass growth, disturbances induced by climate change (e.g. increased frequency of forest fires) may eventually result in declining boreal biomass stocks (Kane & Vogel 2009; Kurz et al. 2008). Furthermore, if forest biomass is to replace a substantial share of non-forest (e.g. fossil) resource use, the harvesting of forest biomass will have to increase considerably (Narodoslawsky et al. 2008), which may lead to a net decrease also of the biomass stocks in temperate and boreal forests. In some regions, such an increased demand is probable in view of current energy policies. For example, the European Union (EU) target of achieving a 20% share of renewable energy in the European energy consumption by 2020 (EU 2007) may cause demand for European forest biomass to exceed the potential supply (Mantau et al. 2010) and threaten the European forests' capability to function as a carbon sink (Nabuurs et al. 2007).

Whether or not wood should be defined as renewable can also depend on whether indirect land use and land use change are taken into account. Indirect land use and land use change do not occur at the site of the studied system, but at some other location as a consequence of the activities in the studied system. For example, if land is used for producing a certain product, competition for land increases, which may result in higher commodity prices and therefore more intensive or extensive land use and land use change at some other location. Such indirect market-driven effects have been shown to be significant in environmental assessments of biomass feedstocks for biofuels (Berndes et al. 2013; Kløverpris & Mueller 2013; Hertel et al. 2010; Plevin et al. 2010; Searchinger et al. 2008). There has been a greater focus on such indirect effects in studies of bio-based products derived from agricultural feedstocks than in studies of forest products (Ahlgren et al. 2013). However, considering the potentially increasing competition for forest land (as discussed above), there could be significant indirect effects also in future forest product systems. Whether or not indirect effects should be accounted for depends, for example, on whether consequential or attributional assessment approaches are applied (these concepts are further explained in Section 3.2), where consequential approaches more often strive to capture market mechanisms such as indirect land use and land use change. The exclusion of indirect effects may also depend on methodological shortcomings, since the mechanisms behind indirect effects are complex, interlinked, dynamic and uncertain (or, in one word, “wicked”, a term further described in Section 3.4), and thus difficult to quantify (Ahlgren et al. 2013; Berndes et al. 2013). It should be noted that the choice of approach may in turn influence the spatial and temporal system boundaries that were discussed in previous paragraphs.

To summarise, whether or not forest biomass can be viewed as renewable depends on the location of the forestry, the spatial and temporal scope of the study and on other methodological choices. Whether or not the forest biomass is renewable in turn influences the assessment of forest products’ climate and biodiversity impacts, as is further described below.

### 3.1.2 Climate change

Perhaps the most emphasised environmental benefit of forest products concerns their potential role in mitigating climate change. It is commonly claimed that forest products and other bio-based products are carbon neutral, and as a consequence (it is assumed) climate neutral. Such claims may however rely on questionable premises (Agostini et al. 2013; Searchinger et al. 2008). For example, claims of carbon and climate neutrality often presume renewable biomass, which may not always be the case (as discussed in the previous subsection). Furthermore, there are mechanisms by which the climate system and forest product systems interact that are not captured by the commonly used methods and practices for climate impact assessment – mechanisms that can contribute both positively and negatively to the climate impact of forest products. These mechanisms and other difficulties of climate impact assessment are the subject of research question 4 and further described in Section 3.3.5 and Paper IV.

### 3.1.3 Biodiversity loss and water cycle disturbances

A potential environmental problem of forest biomass is that it is land- and water-intensive compared with many abiotic resources. Apart from potential problems with renewability and climate impact, as discussed above, poor land and water management can result in a range of other disturbances, including biodiversity loss and water cycle disturbances with subsequent impacts to human health, ecosystem quality and resources. Due to looming scarcity of land (Lambin & Meyfroidt 2011) and water (Rockström et al. 2012), such impacts will probably increasingly gain attention, including in countries that seemingly have an abundance of land and water, such as Sweden. The assessment of biodiversity loss and water cycle disturbances is the topic of research question 5 of this thesis and discussed further in Section 3.3.5 and Paper V. It should be noted that the impact category referred to as "water cycle disturbances" in the thesis, traditionally is referred to as "water use" or "water use impact"; the choice of terminology is further discussed in Section 5.3.

### 3.1.4 Biodegradability

Another often recognised benefit of forest biomass is its biodegradability, which means that it will normally not accumulate in nature once it has become a waste material, as some other materials often do, such as plastics (Derraik 2002). In the disposal stage of forest products, this is sometimes seen as an environmental benefit, although it may not always be a benefit. When forest biomass waste degrades, for instance in landfills, part of the carbon is emitted to the atmosphere as methane, a potent greenhouse gas (GHG; Lou & Nair 2009). Globally, methane emissions from landfills may constitute up to 20% of all anthropogenic methane emissions and 4% of all anthropogenic GHG emissions (Frøiland Jensen & Pipatti 2002). The biodegradability may also be problematic in the use phase of forest products and they may therefore require more preservatives, surface treatments and maintenance to meet the same service-life performance as non-forest alternatives (an issue addressed in the WoodLife project, see Section 2.1). The biodegradability may even make forest biomass unsuitable for some products, such as containers for certain foodstuff.

### 3.1.5 Other aspects of the environmental impact of forest products

As previously discussed, forest products often require chemical treatment to withstand weathering and degradation, which may lead to exposure of humans and ecosystems to toxic compounds (Werner & Richter 2007). Furthermore, the availability of forest biomass is highly distributed and seasonally variable compared to the availability of many other biotic and abiotic resources, and the energy content of forest biomass is low compared to fossil energy carriers. This can make forest product systems more transport-intensive compared to non-forest product systems and, as a consequence, considerably influence the environmental performance of forest products (as transportation can be an important contributor to the environmental impact of forest products; Handler et al. 2014). It should, however, be noted that decentralised production in some situations can reduce transportation and the associated environmental impacts. Moreover, the main feedstock of a product is not the only factor determining its environmental impact. For example, in the production and maintenance of forest products, many non-forest materials may be used, sometimes even more (in mass) than used in the production of alternative non-forest products. The amount and type of energy used in the life cycle are also key factors determining a product's environmental impact – factors which can be rather independent of the main feedstock of the product.

#### 3.1.6 Concluding remarks

To conclude, the fact that the main feedstock of a product is forest biomass is no guarantee that it is environmentally superior compared to non-forest alternatives. Many aspects need to be taken into account if one wants to ensure that forest products that replace non-forest alternatives contribute to reduced environmental impact. A number of these aspects are further discussed later on in this thesis.

### 3.2 LCA methodology

LCA is an internationally accepted and widely used method (Baitz et al. 2013; Guinée et al. 2011; Peters 2009) capable of assessing a wide range of environmental impacts over the full life cycle of a product, and it has been recognised as an appropriate tool for assessing future technologies (Hetherington et al. 2014; Frischknecht et al. 2009a).

The LCA procedure consists of four steps, usually carried out iteratively to allow for adjustments following new insights (ISO 2006a, 2006b):

1. *Goal and scope definition*: The aim of the assessment, the functional unit and the product life cycle are defined, including boundaries to other product systems and the environment. The functional unit is a quantitative unit reflecting the function of the product, which enables the LCA practitioner to compare different products with identical functions. The product life cycle typically includes processes related to raw material extraction, manufacturing, use, end-of-life treatment and transportation.
2. *Life cycle inventory analysis (LCI)*: All environmentally relevant material and energy flows between processes within the defined product system, and between the system and the environment and other product systems, are quantified and expressed per functional unit. Flows between the defined system and the environment consist of emissions and the use of natural resources

(including the use of land). These flows are often termed environmental loads, interventions or stressors.

3. *Life cycle impact assessment (LCIA)*: By means of characterisation methods, the LCI data is translated into potential environmental effects, so-called impact categories. Traditionally, the focus has been on environmental stressors from emissions and on global and regional environmental effects, such as climate change, stratospheric ozone depletion and eutrophication. Sometimes, LCA covers more location-dependent impacts as well, such as eco-toxicity and human toxicity, although there are large uncertainties in the modelling of such impacts because they are highly dependent on local or regional characteristics (e.g. local flora and fauna, soil structure or presence of other substances) that are difficult to account for in LCAs.

   Impact categories can be expressed as midpoint or endpoint indicators. Midpoint indicators reflect links in the cause-effect chain from activities causing environmental stressors to environmental effects, whereas endpoint indicators are metrics of actual end effects. For example, GWP is a midpoint indicator for climate change, as it is based on how much an emission influences the radiative forcing. Endpoint indicators for climate change are instead based on how much an emission contributes to possible consequences of a changed radiative forcing, such as sea level rise, increased frequency of extreme weather events or human health consequences of rising temperatures. Endpoint indicators are sometimes grouped into areas of protection: human health, ecosystem quality, resource availability or (more rarely) man-made environment (Goedkoop et al. 2013; Jolliet et al. 2004).

   The LCIA can also include normalisation and weighting. Normalisation can provide understanding of the importance of impacts compared to a reference, by comparing the impact per functional unit to, for example, per capita or aggregated impact in a given area (e.g. global, regional or national) in a certain year (ISO 2006b). Weighting instead compares, and enables the aggregation of, different impact categories on a single yardstick (ISO 2006b). Weighting can be based on, for example, environmental taxes and fees (Finnveden et al. 2006); distances to political goals (Stranddorf et al. 2005); revealed, stated, imputed or political willingness-to-pay for damages (Ahlroth et al. 2011); or end-point models of human, resource and ecosystem damages combined with models of different cultural perspectives (Goedkoop et al. 2013).

4. *Interpretation*: The result of the LCIA is interpreted, taking into account the goal and scope definition (e.g. the system boundaries) and the LCI (e.g. data

gaps and data uncertainties), and recommendations are made to the intended audience. The interpretation can include sensitivity and uncertainty analyses (in which the influence of critical or uncertain system parameters are tested), dominance analysis (in which the contribution of different life cycle processes are analysed), or contribution analysis (in which the contribution of different environmental stressors are analysed).

The above described procedure can be used to assess a wide range of environmental concerns. Still, LCA may sometimes fail to assess all relevant environmental impacts. The present thesis is part of the ongoing research to improve LCA methodology and its practice in various contexts to enable assessments of a wider range of environmental impacts. Nevertheless, it may be necessary to use other assessment tools in certain cases. For example, in the WoodLife project, a toxicological evaluation (including a literature study and eco-toxicological testing) had to be carried out in addition to the LCA in order to evaluate the toxicological risks of nanoparticles (as reported in Publication B; see list of publications, page vii).

Section 3.3 gives a comprehensive background to the theoretical and methodological aspects focussed on in the research questions. To understand these aspects, two aspects of LCA methodology need to be elaborated on in more detail: the choice between attributional and consequential modelling approaches and the choice of LCI data.

The consequential-attributional controversy is a topic of discussion in the LCA research community (Plevin et al. 2014; Suh & Yang 2014; Earles & Halog 2011; Ekvall & Weidema 2004; Tillman 2000). Traditionally, LCA has relied on attributional (also called descriptive or accounting) approaches, which (most often) means that the LCA considers the immediate physical flows (emissions and resource use) occurring at the location of the life cycle processes. Attributional approaches typically imply that the LCA maps the average impact of the studied product system per delivered functional unit. A consequential (also called change-oriented) approach, on the other hand, seeks to map the change of physical flows occurring as a consequence of a decision (Zamagni et al. 2012; Earles & Halog 2011; Ekvall & Weidema 2004). This can also be described as the consequences of a change in production output, i.e. what the environmental consequence would be if more or less functional units were provided. A consequential approach entails inclusion of effects not necessarily physically connected to the product system, but occurring due to, for example, market mechanisms (Earles & Halog 2011; Ekvall & Weidema 2004). Section 3.1 described one such market mechanism: indirect land use and land use change. The choice between an attributional and a consequential approach determines, for example, which processes to include within the system boundaries, which LCI data to use (see the next paragraph) and how to handle multi-functional processes (see Section 3.3.4). Later in this thesis, there are several examples of consequential and attributional approaches that lead to different LCIA results. See Zamagni et al. (2012) for a further review of consequential and attributional LCA methodology.

Concerning the type of LCI data to use, one important question is whether to use average or marginal data. For example, when the studied product requires electricity for its production, it is common to use average LCI data, i.e. data on the annual average emission per unit of electricity produced in the country or region of the production site. However, marginal LCI data can also be used, which are emission data on the marginal source for electricity, i.e. the technology that is expected to respond to a change in demand. The marginal technology is most often considered to be the utilised technology with the highest operating cost (also called marginal cost) or the unutilised technology with the lowest operating cost (Lund et al. 2010). However, some authors have proposed that in markets constrained by regulation, the planned or predicted technology should rather be considered the marginal one (Schmidt et al. 2011).

Typically, average data are used for attributional studies, and marginal data for consequential studies (Ekvall & Weidema 2004). The use of marginal data is based on the consequential logic that if the product is not produced, the marginal technology will not be utilised. In many countries, the marginal technology for electricity generation is coal power, which only contributes to the electricity mix when demand is particularly high. As emissions from coal power can be much higher than emissions from the average electricity generation (which may be dominated by, e.g., hydro or nuclear power), the choice between average and marginal LCI data can significantly influence LCIA results. It can, however, be difficult to determine the marginal technology (Mathiesen et al. 2009). For example, the short-term marginal technology (e.g. at a particular time of the day, or a particular time of the year) may be different from the long-term marginal technology (e.g. annually). Thus, the choice between, and the selection of, average or marginal LCI data is a much discussed aspect of LCA methodology.

### 3.2.1 Integrating LCA work in R&D processes

There are numerous suggestions on how to integrate different methods for environmental assessment (often LCA) into R&D processes[3] (e.g. Chang et al. 2014; Clancy 2014; European Forest Institute 2014; Fazeni et al. 2014; Tambouratzis et al. 2014; Collado-Ruiz & Ostad-Ahmad-Ghorabi 2013; Askham et al. 2012; Devanathan et al. 2010; Manmek et al. 2010; Othman et al. 2010; Vinodh & Rathod 2010; Colodel et al. 2009; Kunnari et al. 2009; McAloone & Bey 2009; Ny 2009; Byggeth et al. 2007; Waage 2007; Rebitzer 2005; Nielsen & Wenzel 2002; Fleischer et al. 2001). These are often screening or simplified methods particularly designed for the assessment of preliminary product or process designs – see Rebitzer (2005) for a review of such methods.

What many of the ready-made methods and procedures have in common is their emphasis on a range of different sustainability criteria in addition to environmental ones (e.g. economic and social criteria) and the recognition of the need for some type of multi-criteria decision analysis (MCDA) for handling potential trade-offs between different sustainability dimensions or impact categories. Also, many methods and procedures are primarily intended for assessments carried out in rather specific contexts – for instance in studies of certain product categories as noted by Hetherington et al. (2014) – sometimes with predefined impact assessment methodology adopted to that specific context. As a consequence, most of them are of limited use for the R&D context focussed on in the present thesis, where there is often a wish to compare the forest product under development with some conventional (often non-forest) product to ensure that the forest product can potentially contribute to reduced environmental impact (by replacing the conventional product), and where the important environmental (or sustainability) criteria should be defined on a project-to-project basis (as emphasised by, e.g., Clancy (2014)). Furthermore, the literature on environmental consideration in R&D most often focuses on intra-organisational R&D contexts (i.e. R&D carried out within a firm), which do not face the same degree of organisational complexity as the inter-organisational R&D context in focus in the present thesis. Thus, many of the suggested methods and procedures do not address the specific challenges dealt with in this thesis. Also, in inter-organisational contexts it may not be possible or even desirable to align and/or integrate the LCA work with the strategic long-term management of the product portfolio of a particular organisation as is often

[3] The inclusion of environmental considerations in product development is sometimes referred to as "ecodesign", "sustainable product design", "design for the environment", "design for life cycle", "environmental product development" or similar terms. The definitions of, and the distinctions between, these terms are not further elaborated on in the present thesis.

desirable in intra-organisational LCA work, and is thus a key element in many of the methods and procedures suggested in the aforementioned literature.

Because of the above-described typical characteristics of the literature on integrating environmental assessments in R&D – i.e. the context-specificity and the focus on intra-organisational contexts – ready-made methods and procedures proposed in the literature were not applied in the case studies used to address the research questions of the present thesis (although elements of such methods and procedures were sometimes used, as referred to throughout the thesis and the appended papers). Awareness about the existence of ready-made methods and procedures for integrating LCA into R&D can, however, be a great asset for LCA practitioners in any R&D context, and for some contexts they may be directly applicable.

Although many solutions offered by the literature on environmental consideration in R&D address case-specific challenges, they also address some recurring general challenges. Hetherington et al. (2014) draw on experiences from case studies in diverse sectors (nanotechnology, bioenergy and food) to identify four such general challenges: (i) comparability, for example the issue of comparing emerging technologies with existing commercial technologies in cases of incomparable functions and/or system boundaries; (ii) scaling issues, for example estimating the material and energy use of commercial scale production when the processes exist only at laboratory scale; (iii) data, for example issues of getting inventory data in time to be able to influence decision-making in the R&D process; and (iv) uncertainty, for example uncertain characterisation methods for the emissions of novel materials (e.g. regarding the toxicity of emissions of nanoparticles) or the inherent uncertainties of future product systems existing in a constantly changing world. These challenges are common for LCAs in general, but are particularly prominent for LCAs of future technologies. The present thesis contributes towards better handling of some of these challenges; particularly it addresses issues related to the inherent uncertainties of future product systems. Although the focus of the thesis is on inter-organisational R&D projects and on forest products, many findings and recommendations of the thesis can certainly contribute also to the growing body of knowledge concerned with environmental consideration in R&D in general.

## 3.3 Theory and methods of specific importance for the research questions

### 3.3.1 LCA work in technical inter-organisational R&D projects

As described in Section 1.3, research question 1 addresses how to improve pre-project planning of LCAs in publicly-funded, technical, inter-organisational, inter-disciplinary R&D projects focussed on early stages of product development. Some characteristics of such projects were outlined in Section 3.2.1. Here, characteristics that influence pre-project planning are further elaborated on to provide a thorough background for research question 1.

As previously described, inter-organisational R&D projects often involve a mix of firms, universities and research institutes from different countries and disciplines. The participants often have different reasons for participating and diverse expectations of the project outcomes. Complexity is further enhanced as multiple activities are carried out in parallel to solve different technical problems. In this setting, it can be difficult to comprehend how different activities will interact and contribute towards the aim of the project. The projects are also characterised by a focus on certain technical ideas or solutions – as it is often required to present a well-developed technical idea or solution to attract the funding – and once the funding has been secured, the application text (e.g. specifying tasks, milestones, deliverables and distribution of budget) and the competences of the project team can set limitations on what can be done in the project. Furthermore, although reduced environmental impact is often one of the stated driving forces of the project, the project work most often focusses on technical R&D, and therefore LCA work is allocated a relatively small share of the budget. LCA work may be included not because of the wishes of the involved organisations but rather because of requirements from the commissioner or the funding agency. Some of the organisations and/or individuals involved in the project may therefore not see the value of, or not be particularly interested in, the LCA work and the LCA results. In such situations, LCA work may become an add-on that does not receive sufficient attention.

The above-described organisational complexities, including limited flexibility of the technical direction of the project and limited interest in the LCA work, can confine the possible roles of LCA and increase the importance of thorough and conscious selection of LCA roles. However, based on experiences in the CelluNova and WoodLife projects (see Chapter 2), and experiences of the co-authors of Paper I in many similar projects, the role selection is often done in an arbitrary manner. This leads to unclear LCA roles, which contributes further to differing and even contradicting expectations of the outcome of the LCA work among project participants, causing situations in which the full potential of LCA to assess environmental impacts and influence the technology development is not utilised.

Research question 1 addresses how project commissioners, project managers and LCA practitioners can select appropriate context-adapted roles for LCAs in the described R&D context. The research question was addressed in Paper 1 by means of an iterative procedure of parallel data collection and qualitative analysis, including a literature review on possible roles of LCA in R&D processes and analyses of experiences from five specific inter-organisational R&D projects (including the CelluNova and WoodLife projects described in Chapter 2) in identifying project characteristics that determine the availability of LCA roles. The procedure can be described as a constant comparative approach (Silverman 2005; Glaser 1965) with elements of triangulation (Peters et al. 2013; Denzin 1978). In Paper I, we also used one of the R&D projects as a case study for discussing how the identified project characteristics can be evaluated to help in the pre-project planning. Section 4.1 includes a summary of Paper I and Section 5.1 includes a discussion of how it contributes to addressing research question 1.

### 3.3.2 Scenario modelling and sensitivity analysis

Little may be known about the future commercial-scale product system under development in R&D projects, and the potential environmental impacts of the system may depend largely on factors in the surrounding world that are inherently uncertain (Frischknecht et al. 2009a). For example, as discussed in Section 3.1, new political policies may considerably alter the demand for forest biomass, with potential implications for many parameters that influence the environmental impact of forest products. Methodological choices may also influence the outcome of the LCA and thus pose a significant uncertainty. It is important to capture these types of uncertainties in LCAs of products under development if one aims to develop a product system that contributes to reduced environmental impact regardless of future world development and whose perceived environmental benefits are not dependent on arbitrary methodological choices. To capture these types of uncertainties in LCAs, scenario modelling and sensitivity analysis are commonly used. The need for sensitivity analysis and scenario modelling in prospective assessments of product systems under development has been recognised before (Hospido et al. 2010; Kunnari et al. 2009; Spielmann et al. 2005). In the present thesis, different approaches for scenario modelling and sensitivity analysis are used for addressing research questions 2–5.

A scenario in LCA has been defined as “a description of a possible future situation relevant for specific LCA applications, based on specific assumptions about the future, and (when relevant) also including the presentation of the development from the present to the future" (Pesonen et al. 2000, p. 23). There are various systems for classifying scenarios in LCA. Börjeson et al. (2005, 2006) distinguished between predictive (what will happen?), explorative (what can happen?) and normative (how can a specific target be reached?) scenarios. For each category, Börjeson and colleagues made a further breakdown: predictive scenarios can be forecasts or what-if scenarios, explorative scenarios can be external or strategic, and normative scenarios can be preserving or transforming. Pesonen et al. (2000) distinguished between what-if and cornerstone scenarios, where what-if scenarios are used to compare the environmental consequences of choosing between well-defined options in a well-known and simple situation, while cornerstone scenarios are used to compare options in a more unknown and complex situation to increase the understanding of the studied system (e.g. in the context of product development, as recognised by Pesonen and colleagues). Scenario analysis has been pointed out to be particularly useful in LCAs dealing with strategic decisions (Guinée et al. 2002) with non-marginal implications for technological systems (Heijungs et al. 2009). Non-marginal implications in this case refer to situations when "technologies can no longer be characterized with constant coefficients; capacities of technologies is [sic] no longer constant; the change in economic structure will affect prices and preferences, and hence induce change in life styles; background concentrations of pollutants will change" (Heijungs et al. 2009, p. 61). Furthermore, Heijungs and colleagues emphasised that scenarios of such changes can be set up endogenously, as the outcome of models, or exogenously, as the outcome of "creative or explorative thinking" (p. 62), which resembles the explorative scenario classification by Börjeson et al. (2005). See Börjeson et al. (2006) for a more extensive review of scenario classification.

With the above terminology, the scenarios of Papers II and V (which are further described below) can be seen as explorative, cornerstone and exogenous scenarios, as they seek to explore how uncertain factors in the surrounding world could influence the environmental impact of the studied product systems, thereby increasing the understanding of the studied systems and making it possible to provide guidance for their development. The scenarios are not predictive as they do not seek to find the most possible states, but rather possible *and* distinctly dissimilar states that combined generate a holistic view of possible futures. The above terminology is not applicable for classifying the scenarios of different decision contexts in Papers III and IV (which are further described below), as these scenarios do not attempt to capture uncertainties of technological systems but rather aim to structure an analysis of the interplay between different methodological and contextual elements of LCAs (the studied methodological element is in Paper III the choice of method for handling multi-functionality, and in Paper IV the choice of impact assessment method).

The literature provides a rich plethora of examples for how to model scenarios or carry out sensitivity analysis in LCAs in order to capture uncertainties of different kinds, for example uncertainties related to the surrounding world, the scope of the study, technological assumptions, LCI data, methods for handling multi-functionality or impact assessment methods (see, e.g., Grant et al. 2014; Peters et al. 2013; Bhattacharyya et al. 2013; Cellura et al. 2011; Cherubini et al. 2011; Mathiesen et al. 2009; Ardente et al. 2008; Spielmann et al. 2005). Some of these papers refer to the different setups tested in a sensitivity analysis as “scenarios”, thus scenario modelling and sensitivity analysis are not necessarily distinctly different concepts. Others have also acknowledged that scenario analysis can encompass sensitivity testing (Börjeson et al. 2006). In addressing research questions 2–5, Papers II–V contributes with further examples of how different uncertainties can be captured in prospective LCAs, as described in the following paragraphs.

In Paper II, we generated end-of-life scenarios in the assessment of construction products by making different assumptions about the nature of future technologies for transportation, demolition of constructions and waste handling (as further described in Section 3.3.3). Additionally, the scenarios tested the influence of using either attributional or consequential modelling approaches, as this was hypothesised to be crucial for the modelling of end-of-life processes. The details of these scenarios are described in Paper II. The scenarios enabled us to study how the product system under development (in this case a glulam beam) would perform compared to an alternative product system (in this case a steel frame) with consequential and attributional modelling approaches in (i) a future in which technologies have roughly the same environmental impact as today’s technologies, and (ii) a future in which technologies have considerably lower environmental impact than today’s technologies. If the developed product performs better in all scenarios, it is probably a long-term environmentally and commercially attractive alternative in many types of decision contexts. On the other hand, if there turns out to be small or non-existent environmental benefits of the developed product in a future dominated by low- or high-impact technologies, or in a study based on a certain modelling approach, further development of the product (and/or careful planning of the supply chain design) is probably appropriate both for environmental and commercial reasons. As the approach evaluates how the product system can be influenced by technological changes, it has similarities with the approach used by Mathiesen et al. (2009), which accounted for the uncertainties of the marginal technology, and the approach used by Spielmann et al. (2005), which used cornerstone scenarios to represent a range of possible future states in terms of technological and socio-economic developments.

In Paper III, we tested the implications of the choice of method for handling multi-functionality in LCAs of biorefinery products. In such production chains, there are multi-functionality problems (which are further introduced in Section 3.3.4) to be handled both due to the multi-functionality of forestry (with outputs such as timber, pulpwood, energy wood, tops and branches, which are used for different end products) and the multi-functionality of biorefineries (with outputs such as paper pulp, industrial chemicals, biofuels and heat). We evaluated different methods for handling multi-functionality in relation to four LCA questions (i.e. the question the LCA is intended to address) representing four different decision contexts relevant for forest products produced at biorefineries. The decision contexts were generated by varying two contextual dimensions deemed to be defining in terms of LCA methodology: contexts relating to *specific* or *general* product systems, and contexts relating to *minor* or *major* modifications of product systems. Minor modifications can for instance be incremental changes of an existing product system or changes induced by choices made by individual consumers. Major modifications can for instance be changes caused by the development of a new product system or by policy-making. The methods for handling multi-functionality and generating decision contexts are further described in Paper III. The aim of the paper was not to test the influence of the choice of method in a particular study, but to contribute to more conscious and context-adapted handling of multi-functionality in LCAs of biorefinery products in general. Since the range of the outcomes of using different methods for handling multi-functionality can – as long as the use of each method is reasonably justified – be seen as a methodological uncertainty of the assessment, the paper serves as an example of how a structured generation of a set of decision contexts can be used to evaluate the possible influence of methodological uncertainty in LCAs.

In Paper IV, we explored the uncertainties imposed by possible climate impact assessment practices in LCAs of forest products (Section 3.3.5 further describes the issues associated with climate impact assessment of forest products). The uncertainties were explored by varying two parameters: the type of product and the impact assessment practices. Product type was tested by studying two forest products with very different life lengths: an automotive fuel and a building. A range of possible climate impact assessment practices were tested, and in the paper we showed the outcome of the currently common practices and the practices yielding the lowest and highest LCIA results, respectively (which can be classified as cornerstone scenarios). A set of distinctly different decision contexts were generated in order to explore potential implications of the choice of climate impact assessment practice for decision-making. As in Paper III, this was done by varying two contextual dimensions that are considered important for methodological choices (Finnveden et al. 2009; Tillman 2000): whether the LCA is performed on a specific or a general product, and whether or not the LCA compares products. The methodological and contextual scenarios are further described in Paper IV. The aim of the paper was not to test the influence of impact assessment practices in a particular study, but to contribute to an increased understanding of current and future impact assessment practices and more conscious and context-aligned choices of impact assessment practices in LCAs of forest products in general. By showing the span of possible LCIA results if LCA practitioners were to use other impact assessment practices than commonly used today, the paper also serves as an example of how uncertainties arising from the choice of impact assessment method can be captured and showed in LCAs.

In Paper V, we handled uncertainties of a future product system by setting up scenarios of the regenerated cellulose fibre product system developed in the CelluNova and ForTex projects (see Section 2.2). We identified mechanisms of the surrounding world that are expected to be important for the geographical location of life cycle processes and the occurrence of land use change – important factors in assessing impacts on the climate, the water cycle and biodiversity. To generate scenarios, two mechanisms were varied: the market demand for fibres (affecting the scale of the product system) and the expected competition for land (affecting the need to turn to previously unused land). As is further described in Section 3.3.5, impact assessment methods were chosen that would allow for this type of location-dependent analysis. The scenarios are further described in Paper V. The outcome of this type of scenario modelling can for example show to what extent the sourcing of the main feedstock of the fibre – where it comes from and how the extraction has been managed – is crucial for the product to be an environmentally preferable alternative. This information can, for example, help to design the supply chain of the new regenerated cellulose fibre. As the scenario modelling approach explores possible futures by accounting for market mechanisms, it can be seen as a consequential approach. Since the approach accounts for socio-economic factors influencing the product system, it has similarities with the method used by Spielmann et al. (2005).

Chapter 5 contains a discussion of how the above-described scenario modelling and sensitivity analysis approaches of Papers II–V contribute to addressing research questions 2–5.

### 3.3.3 End-of-life modelling

As described in Section 2.1, the aim of the WoodLife project was to develop coatings and adhesives for construction products such as window frames and glulam beams. For construction products under development, the manufacturing will take place in perhaps 10–20 years (allowing for further technology development and market diffusion). The end-of-life processes (e.g. demolition and disposal) of such products will take place in an even more distant future, often 50–100 years after manufacturing (Frijia et al. 2011). The nature of such processes is highly uncertain because of technological changes (Du et al. 2014; Frischknecht et al. 2009a). This time-dependent technological uncertainty has been acknowledged as being a common challenge for LCAs in the construction industry (Singh et al. 2011; Verbeeck & Hens 2007). Still, it is an often neglected uncertainty in LCAs of construction products. The end-of-life technologies of today are often assumed without any explicit explanation, also when the aim is to support decisions concerning contemporary or future constructions with end-of-life processes occurring in a distant future (e.g. Habert et al. 2012; Bribián et al. 2011; Persson et al. 2006; Lundie et al. 2004). There are exceptions, for example Bouhaya et al. (2009) used scenarios to account for different possible end-of-life technologies in an LCA of a bridge, Du et al. (2014) used sensitivity analysis to test different steel recycling rates for the end-of-life treatment in an LCA of five bridge designs, and Garcia & Freire (2014) tested the influence from assuming either incineration or landfill as the end-of-life option in carbon footprint calculations of wood-based panels.

Accounting for end-of-life uncertainties is particularly important when end-of-life processes can be expected to strongly influence a product’s environmental impact. Efficient recycling in the disposal of buildings may save energy corresponding to 29% of the energy use in manufacturing and transportation of the construction materials (Blengini 2009) or 15% of the total energy use of a building’s life cycle (Thormark 2002). Although the use phase has been said to contribute 60–90% of a building’s environmental impact (Buyle et al. 2013; Cuéllar-Franca & Azapagic 2012; Ortiz et al. 2010), the relative contribution from end-of-life processes is growing because of increasingly energy-efficient buildings

(Dixit et al. 2012). Furthermore, it has been argued that inadequately defined functional units have led to overstated energy usage in the use phase (Frijia et al. 2011), which implies that the relative contribution from end-of-life processes has been understated. The sheer amount of construction materials existing in society has also been used as an argument for why the end-of-life handling of such materials is environmentally important (Bribián et al. 2011; Singh et al. 2011; Blengini 2009).

To conclude, there are many reasons for improving the modelling of end-of-life processes in LCAs of construction products. This can contribute to more robust decision-making in the development of the construction products of tomorrow, for example in contexts such as the WoodLife project. Paper II contributed towards such improvements by testing how assumptions in the modelling of end-of-life processes influence LCA comparisons of alternative internal roof structures (glulam beams and steel frames) for an industrial hall. Tested assumptions relate to the technologies of the end-of-life processes, the use of attributional or consequential modelling approaches, and (when a consequential modelling approach is used) the technologies of the processes assumed to be substituted. Section 4.2 contains a summary of Paper II and Section 5.2 contains a discussion on how it contributes to addressing research question 2.

#### 3.3.4 Handling multi-functionality

Another issue particularly relevant to LCAs of forest products is how to split the environmental burden of multi-functional processes between the functions. To exemplify, waste handling of forest products is often multi-functional: it can dispose of the waste, produce reusable and/or recyclable materials and produce heat and/or electricity. Production of forest products is also often done by multi-functional processes: a forestry operation can produce timber, pulpwood and fuelwood, and a biorefinery can produce, for example, fuels, industrial chemicals, building materials, heat, paper pulp and textile pulp. Multi-functional processes are very common in product systems and will probably become even more common as principles of industrial ecology and eco-efficiency become more important for the design of product systems (Heijungs & Guinée 2007). Multi-functionality becomes a problem when it is not feasible to split a multi-functional process into sub-processes connected to specific functions. For example, at a pulp mill producing pulp and heat, the use and recovery of chemicals and energy are often too integrated for it to be possible to connect each sub-process (and thereby allocate each emission and resource use) to either pulp or heat.

If the multi-functionality cannot be handled by splitting the multi-functional process into sub-processes, it can be handled by two principally different categories of methods, as acknowledged by ISO 14044 (ISO 2006b). The first category is system expansion, where the functional unit is redefined to account for the functions of all co-products, or where a main product is selected among the co-products and given credit for the avoided environmental burdens from products assumed to be substituted by the other co-products. The second version of system expansion is commonly used although not acknowledged by ISO 14044, and has been criticised for introducing too much speculation in the system modelling (Heijungs & Guinée 2007). The second category of methods for handling multi-functionality is partitioning, whereby the environmental burden of the multi-functional process is divided between the co-products based on an attribute of theirs, for example a physical attribute such as mass, volume or energy content, or a monetary attribute such as production cost, market value or profitability.

The choice of method for handling multi-functionality can be based on recommendations in standards, guidelines or directives, such as ISO 14044 (ISO 2006b), the International reference Life Cycle Data system (ILCD) handbook (EC 2010) or the Fuel Quality Directive (EC 2009a). The choice of method can also depend on whether the study is attributional or consequential – partitioning methods are more common in attributional studies and system expansion is more common in consequential studies. Moreover, the choice of method can depend on preferences of the LCA practitioner or others influencing methodological choices (e.g. the commissioner of the LCA study), which in turn may be influenced by some stakeholder's interest in "proving" the environmental benefits of a certain product or by strong beliefs regarding how multi-functionality ought to be handled (e.g. that a certain co-product should always be considered free of environmental burden). Preferences can also be expressed in terms of natural science approaches (allocation based on some physical attribute) or social/economic science approaches (allocation based on some economic attribute) (Pelletier et al. 2015). Ultimately, the choice of method can considerably influence the results of LCAs of forest products (Karlsson et al. 2014; Cherubini et al. 2011; Luo et al. 2009) and, as a consequence, decision-making based on those results. There has been critique that similar multi-functionality problems are often handled in different ways and that the choice of method often is poorly justified in LCA reports (Pelletier et al. 2015). Inconsistencies of practices may arise because of, for example, unclear or ambiguous recommendations in ISO 14044, or differences between recommendations in ISO 14044 and other standards (Weidema 2014).

The implications of the choice of method for handling multi-functionality in LCAs of forest products is addressed by research question 3. The question was addressed in Paper III by testing how the LCIA results in an LCA of biorefinery products is influenced by the choice of method for handling multi-functionality and by evaluating the potential implications of this choice in four distinctly different decision contexts relevant for studies of biorefinery products. In total, six methods were tested: (i) allocation of all environmental burdens to the main product, (ii) substitution, (iii) exergy-based allocation, (iv) economic allocation, (v) allocation based on the environmental impact of a reference system with identical functions (a method proposed by Cherubini et al. (2011) for LCAs of biorefineries, which they called a "hybrid method"), and (vi) allocation based on the co-products' potential for reducing environmental burdens if a reference system with identical functions is substituted (which we developed as an alternative version of the hybrid method). The methods are further described in Paper III. In Section 5.2, there is a discussion on how the findings of Paper III contribute to addressing research question 3.

#### 3.3.5 Assessing environmental impacts

As discussed in Section 3.1, some of the environmental impacts of forest products may not always be satisfactorily captured in LCAs with established impact assessment methods and practices. Therefore, research questions 4 and 5 are concerned with improving the assessment of the climate, water cycle and biodiversity impacts of forest products. The subsections below introduce the challenges of assessing these impacts and summarise how Papers IV and V address these challenges.

<u>Climate change</u>

Difficulties in assessing the climate impact of forest products mainly relate to temporal dynamics of carbon flows in the forest between soil, vegetation and air from sowing to harvest and regrowth, temporary storage of carbon in the product life cycle, and non-carbon climate effects such as changes in the albedo. To understand these difficulties, there is a need to introduce how the climate impact of forest products is usually assessed in LCAs.

The currently most common metric to assess the climate impact of products is the global warming potential (GWP). GWP is an indicator of how much a GHG emitted to the atmosphere influences the radiative forcing under a set time period. Radiative forcing is a measure of the balance between the incoming solar radiation and the energy radiated back to space (IPCC 2013). Since different GHGs have different atmospheric residence times, the chosen time period influences the relative impact of different types of GHGs. In LCA, it is usual to use a time period of 100 years (indicated by a subscript: $GWP_{100}$). This means that in comparing the contribution of different GHGs, one considers climate impact occurring within 100 years counting from the moment of the emission – an arbitrary choice (Reap et al. 2008) often made without a clear motivation (which is showed in Paper IV). Furthermore, biogenic carbon dioxide emissions are most often considered to have a GWP of zero, i.e. they are assumed to be climate neutral (as is showed in Paper IV). The climate neutrality of biogenic carbon dioxide is based on the assumption that forest products (and likewise other bio-based products) are carbon neutral, i.e. there is a balance between carbon sequestrated in the forest and carbon emitted to the atmosphere once the forest biomass is incinerated (which may be in the product's end-of-life phase, or in case of material recovery: in the end-of-life phase of some subsequent product). As concluded in the discussion on the renewability of forest biomass in Section 3.1.1, the carbon neutrality assumption depends on the spatial and temporal scope of the study and other methodological choices (e.g. whether or not the study accounts for indirect land use and land use change). Even in cases where it is valid to assume *carbon* neutrality, assuming *climate* neutrality may not be valid, as a temporal shift between emitted and sequestered carbon may temporarily contribute to a changed radiative forcing (Helin et al. 2013). How to consider this temporal shift depends, for example, on whether one assumes that the carbon sequestration that is attributable to the studied product occurred *before* the harvest (e.g. based on the argument that the trees were planted with the purpose of harvesting) or *after* the harvest (e.g. based on the argument that replantation is a consequence of harvesting, as this is often required by legislation or forestry certification schemes). It has been shown that results of LCAs of forest products can depend strongly on whether or not biogenic carbon dioxide is considered climate neutral (Garcia & Freire 2014; Guest & Strømman 2014; Zanchi et al. 2012; Sjølie & Solberg 2011; also showed in Paper IV), and assuming climate neutrality by default has therefore been questioned (Ter-Mikaelian et al. 2015; Agostini et al. 2013; Gunn et al. 2012; Schulze et al. 2012; Johnson 2009; Searchinger et al. 2008). It should be noted that abandoning the climate neutrality assumption does not necessarily imply an increase in the calculated climate impact of forest products, as a well-managed forest can function as a net carbon sink and thus result in a negative (i.e. beneficial) climate impact (Guest & Strømman 2014; Perez-Garcia et al. 2005; Liski et al. 2003).

Another temporal aspect of climate impact ignored by established impact assessment methods and practices is the timing of GHG emissions and carbon sequestration. Typically, the climate impacts of emission pulses and carbon sequestration are calculated in the same manner irrespective of when they occur.

This is potentially problematic for two reasons: (i) if using $GWP_{100}$, the 100 year time period is not applied consistently: for emissions occurring today, the metric includes impact within 100 years from today; but for future emissions, it includes impact within 100 years from the moment these emissions occur; and (ii) the risk of passing critical tipping points in the climate system implies that urgent impact mitigation is necessary and that it matters greatly whether emissions occur now or in, for example, 50 years (Jørgensen et al. 2014; Helin et al. 2013; Levasseur et al. 2010). Ignoring the timing of GHG emissions and carbon sequestration is particularly problematic for forest products because GHG emissions (if forest biomass is used for long-lived products such as constructions) and carbon sequestration (as forests are relatively slow-growing) often occur over a long time period. This was emphasised in a recent study on how accounting for the timing of emissions influences the GWP scores of 4,034 LCI datasets in the ecoinvent database, which found that GWP scores of datasets associated with the wood sector are particularly sensitive to assumptions related to the timing of emissions (Pinsonnault et al. 2014).

Another often ignored aspect of climate impact, which is related to the timing of emissions and carbon sequestration, is the temporary storage of carbon in forest products. Assuming that the harvesting of this carbon (in the form of forest biomass) allows more carbon dioxide to be sequestered in the forest than would otherwise have been the case, and assuming that the storage prevents the carbon from being emitted to the atmosphere for some time, it has been argued that carbon storage causes a temporary reduction in radiative forcing which should be accounted for in the climate impact assessment of forest products (Vogtländer et al. 2014; Brandão et al. 2013; Levasseur et al. 2010; Costa & Wilson 2000). On the other hand, research has also suggested that temporary carbon storage may not reduce climate impact because it lowers the carbon dioxide gradient between the atmosphere and potential carbon reservoirs, such as the oceans, and thus reduces carbon dioxide removal from the atmosphere. Once the temporarily stored carbon is released again, the atmospheric carbon dioxide concentration is, the argument goes, higher than would be the case without temporary storage (Kirschbaum 2006).

Other potentially important aspects of climate impact, which are seldom captured in LCAs of forest products, are the climate impact due to disturbances of soil organic carbon (Brandão et al. 2011; Repo et al. 2011; Stephenson et al. 2010) and other disturbances of soil biogeochemistry causing non-carbon dioxide emissions, such as nitrous dioxide and methane emissions (Repo et al. 2011; Wrage et al. 2005; Cai et al. 2001). There are some further seldom-captured aspects of the climate impact of land use and land use change unrelated to GHGs emissions that are potentially important in LCAs of forest products. For instance, the albedo of the Earth's surface – i.e. its capacity to reflect sunlight back into space – increases when the surface is bright (e.g. snow-covered) and smooth (e.g. a clear-cut forest), factors which may be influenced by land use and land use change (Cherubini et al. 2012; Schwaiger & Bird 2010). Changes in the reflection of solar radiation may also be induced by land use and land use changes that impact the forest's ability to form organic vapours. These vapours can create aerosols that reflect sunlight or form particles which catalyse the formation of clouds, which in turn reflect sunlight (Spracklen et al. 2008).

As indicated above, there are many potentially important aspects of the climate impact of forest products that are seldom accounted for by established impact assessment methods and practices. Research question 4 addresses the implications of the shortcomings of current methods and practices. The question was addressed in Paper IV by a three-step approach: (i) we identified current methods and practices for assessing climate impact in LCAs of forest products by means of a literature review of 90 peer-reviewed published LCAs of forest products, (ii) we mapped potential differences between LCIA results derived from using currently common versus alternative impact assessment practices by applying different practices in two LCA case studies (studies on a fuel and a building from forest and non-forest feedstocks), and (iii) we evaluated the implications of differences between results from current and alternative practices in relation to four specific decision contexts relevant for forest products (the generation of these contexts was described in Section 3.3.2). The methodology for this three-step approach is further described in Paper IV. Section 4.4 summarises the findings of Paper IV and Section 5.3 contains a discussion on how these findings contribute to addressing research question 4.

Biodiversity loss

The development of improved methods for assessing biodiversity loss is most often part of a broader development of impact assessment methods for land use impacts, which is therefore the starting-point for the discussion in this section. Traditionally in LCA, simple proxy indicators have been used to account for the environmental impact of land activities – for land use: the land area used and the duration of the use (e.g. in $m^2$*years*), and for land use change: the land area transformed (e.g. in $m^2$) from one state (e.g. forest) to another (e.g. agriculture). However, research has concluded that these are meaningless indicators, as land use and land use change can cause both positive and negative impacts, depending on the specific location of the land activities (Michelsen et al. 2012). As a consequence, there have been many proposals for new impact assessment methods, with indicators reflecting a more sophisticated characterisation of the environmental impact of land use and land use change. The development of new methods is facilitated by an increasing understanding of the ecosystem services provided by functioning land ecosystems and the increasingly harmonised classification of such services (Haines-Young & Potschin 2013; Fisher et al. 2009; MA 2005). The development is also facilitated by the increased availability of databases on land use and land cover, such as the GlobCover database on global land cover (European Space Agency 2011), the Terrestrial Ecoregions of the World (TEOW) classification (Olson et al. 2000) and the Corine database on European land cover (European Environment Agency 1995). Moreover, key for the development is the work by the UN Environment Programme (UNEP)-Society of Environmental Toxicology and Chemistry (SETAC) life cycle initiative (Koellner & Geyer 2013), which leads a consensus process for agreeing on principles for inventorying and characterising environmental impacts (primarily biodiversity impact) of land use and land use change anywhere on Earth (Koellner et al. 2013a, 2013b).

Proposed impact assessment methods often focus on biodiversity impact, as reviewed by Curran et al. (2011). This makes sense since biodiversity loss is a major reason for weakened ecosystem services (Gamfeldt et al. 2013; Hooper et al. 2012; Thompson 2011; MA 2005; Chapin et al. 2000). Among biodiversity indicators, species richness of a certain group of species has been a commonly used indicator. Primarily, the focus has been on the species richness of vascular plants (Schmidt 2008; Goedkoop & Spriensma 2000; Koellner 2000; Lindeijer 2000), but there have been proposals to consider other species, alone or in combination. Proposed indicators include the richness, abundance and evenness of vertebrate species (Geyer et al. 2010), the combined species richness of vascular plants, molluscs, mosses and threatened species (Koellner & Schulz 2008) and the combined species richness of vascular plants, birds, mammals and butterflies (Mattsson et al. 2000). De Baan et al. (2012) proposed a method using the relative species richness as an indicator, which allows the use of data on any group or groups of species for which species richness data exist.

Other methods for assessing impacts of land use and land use change measure other facets of biodiversity or other ecosystem attributes (which may indirectly influence biodiversity), sometimes in combination with species richness indicators. Examples include the measurement of functional diversity (species richness data combined with functional traits[4] data; De Souza et al. 2013), global potential for endemic species extinction (de Baan et al. 2013), net primary productivity (Weidema & Lindeijer 2001; Lindeijer 2000), biotic production potential (Brandão & Milà i Canals 2013), naturalness as defined by 11 qualitatively described land classes (Brentrup et al. 2002), soil quality (Saad et al. 2011; Milà i Canals et al. 2007; Mattsson et al. 2000), soil erosion (Núñez et al. 2013), number of red-listed species in combination with several biotope-specific key features (Kylärkorpi et al. 2005), and conditions for maintained biodiversity (based on the amount of decaying wood, the area set aside and the introduction of alien species; Michelsen 2008). Many of these methods utilise several indicators. For example, the biotic production potential method by Brandão & Milà i Canals (2013) is based on several indicators of soil quality. Moreover, several of the mentioned publications advocate the use of other complementary methods for achieving a holistic view of the impact on ecosystem quality. There have also been attempts to combine indicators of different ecosystem attributes into a single index. Such attempts include a procedure for combining 18 indicators to measure the exergy of ecosystems (i.e. the energy available for work in an ecosystem, which (it is argued) reflects the ecosystem quality; Muys & Quijano 2002), and a methodological framework for combining several biodiversity-related parameters at the level of regional ecosystems into a regional biodiversity potential metric (Lindner et al. 2014).

Research question 5 of the present thesis addresses how the impact assessment of biodiversity loss due to forestry can be improved. The question was addressed in Paper V by testing the method proposed by Schmidt (2008) in an LCA case study that compared regenerated cellulosic textile fibres from forest biomass (such as the fibres developed in the CelluNova and ForTex projects, see Section 2.2) with cotton fibres. The method uses the species richness of vascular plants as a proxy for biodiversity and distinguishes between impact of land use (termed occupational impact) and impact of land use change (termed transformational impact). The method enables the calculation of characterisation factors that depend on the geographical location of the land, accounting for its altitude and latitude and the intensity of land use in the surrounding region – factors that influence the vulnerability of ecosystems to interventions such as land use and land use change. In the context of the CelluNova and ForTex projects, the possibility to calculate location-dependent characterisation factors was desirable as it enabled a study of whether the location of operations would be an important factor for the

[4] The functional traits of a species are “morpho-physiophenological traits which impact fitness indirectly via their effects on growth, reproduction and survival” (Violle et al. 2007, p. 1).

environmental impact of the future regenerated cellulose fibre in relation to cotton fibres (land use in Sweden, Russia, China and Indonesia were compared; the process for generating the different geographic scenarios was described in Section 3.3.2). Most other proposed methods for assessing the biodiversity impact of land use and land use change were developed for the assessment of land activities in specific regions, and, due to a lack of data, they are not typically applicable in assessments of globally distributed supply chains – which was an important factor when selecting a method for Paper V. The method by de Baan et al. (2012) is also applicable on a global scale; however, it does not offer the possibility to calculate characterisation factors that depend on regional factors and does not, presently, support the assessment of transformational impact. The principles for a globally applicable method outlined by Koellner et al. (2013a, 2013b) were not available when Paper V was published; however, the method applied in Paper V is largely consistent with these principles (as further discussed in Section 5.3). Paper V explains the chosen method in more detail along with its application in the case study. Section 4.4 summarises the findings of the paper and Section 5.3 contains a discussion on how these findings contribute to addressing research question 5.

<u>Water cycle disturbances</u>

The traditional inventory-based proxy for assessing water cycle disturbances (or water use impact, as it is traditionally referred to; the terminology is further discussed in Section 5.3) – i.e. the volume of freshwater used – has been criticised for not correlating with actual impacts further down the cause-effect chain (Ridoutt et al. 2012; Ridoutt 2011). This has led to intense development of more elaborate methods, as reviewed by Berger & Finkbeiner (2010), Kounina et al. (2013) and Boulay et al. (2015). These reviews were done within the water use in LCA (WULCA) working group in the UNEP-SETAC life cycle initiative, which has initiated a process for finding a consensus on the assessment of water cycle disturbances, similarly to the initiative for finding a consensus on land use impact assessment, as mentioned above (WULCA 2014). In parallel, there is another consensus process that has recently resulted in an international standard for water footprinting, ISO 14046 (ISO 2014). In the development of more elaborate methods for assessing water cycle disturbances, two main difficulties arise: what volume of water to consider in the LCI (i.e. quantifying alterations to the water cycle) and how to interpret this volume in terms of environmental impact in the LCIA.

In LCIs of bio-based products, apart from including process water, it has been common to include engineered water supplied to the crop, for example by irrigation systems, and to disregard naturally supplied water, for example from precipitation

(Peters et al. 2010). More elaborate LCI approaches have been developed, which consider the water use of the metabolism of the crop by attributing evapotranspirational losses to the studied product – approaches which also account for the use of naturally supplied water, as such water use may influence the water cycle in terms of water availability downstream and thus have environmental consequences (see, e.g., Hoekstra & Chapagain 2007). Attributing evapotranspirational losses to forestry and forest products has, however, been questioned as such losses occur also in unmanaged forests (Launiainen et al. 2014). Moreover, LCI methods for water use generally disregard catchment-scale effects of land use and land use change, such as effects on water runoff due to factors such as the interception of rainfall by vegetation, forestry road construction or changes in soil structure (Bruijnzeel 2004; Swank et al. 2001). These factors may be irrelevant when land use is considered a static system dominated by monocultures, but they certainly are relevant for more complex land use systems, such as forestry, and in cases of land use change. The consequential LCI approach suggested and tested in Paper V is an attempt to capture such factors and avoid relying on evapotranspirational losses as the basis for quantifying water cycle alterations. The approach accounts for the change in water runoff that occurs as a result of forestry during harvesting and the subsequent regrowth of trees. This captures not only the water demand by the harvested trees, but also the total influence on downstream water availability by forestry operations. Changes in runoff have previously been suggested as an important impact pathway for water cycle disturbances (Milà i Canals et al. 2009; Heuvelmans et al. 2005) and have been included in LCIs of water use for static agricultural systems (Peters et al. 2010). However, Paper V provides the first example of including changes in runoff in LCIs for systems with forestry or land use change (recently, however, changes in runoff have been accounted for in an LCA of a forestry system without land use change (Quinteiro et al. 2015)). The consequential approach was compared to a more traditional attributional LCI approach, based on the evapotranspirational losses of the harvested trees. Fig. 4 illustrates the difference between the attributional and consequential approaches. The approaches are further described in Paper V (which also includes a discussion of how the water flows inventoried in the different approaches relate to the "green water" and "blue water" terminology, in which we concluded that the distinction between these terms can be unclear and thus proposed a new term: "cyan water").

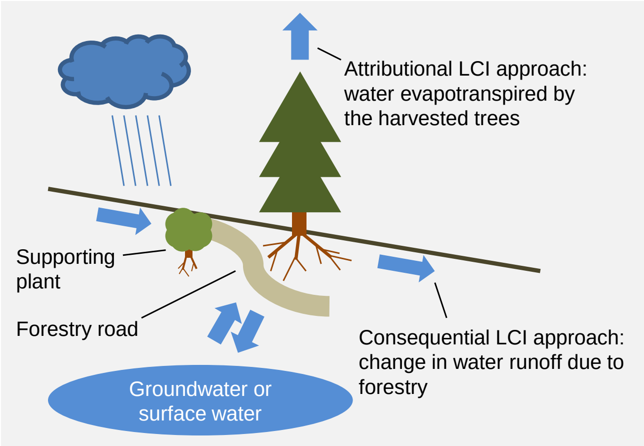


**Fig. 4** Schematic view of water flows in the forestry system. With an attributional LCI approach, the forestry's alteration of the water cycle is estimated by the evapotranspirational losses of the trees during their growth. With the consequential LCI approach, the alteration of the water cycle instead refers to the change in runoff that occurs as a result of forestry during harvesting and the subsequent regrowth of trees, which captures the influence of factors such as the construction of forestry roads and the planting of supporting vegetation.

There are many suggestions for how to characterise the environmental impact of the water volume quantified in the LCI, i.e. the impact caused by the water cycle alteration. Bösch et al. (2007) proposed an exergy indicator for resource consumption, including the use of water. Frischknecht et al. (2009b) proposed a method for characterising water use within the ecological scarcity LCIA framework, in which the water volume withdrawn from a region (or consumed) is characterised based on the ratio between the total water volume withdrawn from the region and the critical water use volume for the region (set at 20% of the renewable water available in the region). Milà i Canals et al. (2009) discussed a number of impact pathways of how freshwater use and land use change may lead to freshwater stress and subsequent impacts on human health and ecosystem quality, and how the use of fossil and aquifer groundwater may reduce freshwater availability for future generations. In an assessment of "freshwater deprivation for human uses", Bayart et al. (2009) distinguished between the quality (low or high) and the type of water (surface water or groundwater) entering and exiting the studied system. Motoshita et al. (2009, 2008) proposed methods for the assessment of undernourishment-related human health damages due to agricultural water scarcity, and of human health impacts due to infectious diseases originating from domestic water use. Van Zelm et al. (2009) proposed a method for the assessment of ecosystem quality impact of groundwater extraction, specific for Dutch conditions. The Water

Footprint Network (WFN) proposed a method for aggregating different types of environmental impacts related to water, for example impacts of water use and impacts of water pollution (Hoekstra et al. 2011). As a proxy for the pollution impacts, the WFN method uses the water volume necessary to dilute emissions to freshwater to such an extent that the water quality adheres to water quality standards. Other such single score approaches have also been suggested, drawing on the latest developments in the LCIA modelling of water-related environmental impacts (Bayart et al. 2014; Ridoutt & Pfister 2013). Most suggestions for characterisation methods in some way relate impacts of water use to water scarcity, water functionality, water ecological value or water renewability, and subsequent impacts on human health, ecosystem quality and/or resource availability (Kounina et al. 2013).

In the study reported in Paper V, the method proposed by Pfister et al. (2009) was used in the LCA case study on textile fibres, since it was deemed the most promising and comprehensive characterisation method available at the time. Others have also deemed the method by Pfister and colleagues as promising for LCAs of bio-based products (Pawelzik et al. 2013). In its favour, the method captures all the impact pathways recognised by Kounina et al. (2013), as it uses four approaches for characterising the water cycle alteration: a midpoint indicator on water deprivation and three endpoint indicators on human health, ecosystem quality and resources. Also, the method offers the possibility for end-point characterisation by the eco-indicator 99 method (Goedkoop & Spriensma 2000). Moreover, in our study it was possible to combine the method with the above-described consequential LCI approach and the method could account for regional parameters (e.g. water stress) that influence the water cycle disturbances. Consequently, the method was applicable in the context of our study, in which we wanted to identify the influence of the location of operations (as was described in Section 3.3.2). Also, the method offers the possibility to define characterisation factors at an even finer resolution than done so far, therefore the study could be updated later on, after the CelluNova and ForTex projects (the projects providing the context to the case study of Paper V), for use in the subsequent supply chain design. The method is further described in Paper V. Section 4.5 summarises the findings of the paper and Section 5.3 contains a discussion on how these findings contribute to addressing research question 5.

### 3.3.6 Connecting LCAs to global challenges

A common challenge in LCAs is how to make sense of LCIA results. In the context that is in focus in the present thesis – LCAs in the R&D of forest products – this challenge can be broken down into questions such as: (i) What is small and what is big in terms of the product's environmental impact? (ii) How important are different impact categories? (iii) What is the potential of the product under development in relation to regional or global environmental challenges? (iv) What are the shortcomings of the product under development in relation to regional or global environmental challenges? Traditional methods for weighting and normalisation can to some degree address these questions, as they can benchmark LCIA results in relation to regional and global scale impacts and policies (see Section 3.2). Questions i and ii above concern the management of potential trade-offs between impact categories; for example, in deciding between two technical options in a development project, one alternative may result in lower climate impact but higher toxic impact, resulting in a trade-off between two impact categories. MCDA can be used for managing trade-offs; see Rowley et al. (2012) for a theoretical discussion of using MCDA in LCAs, and Cinelli et al. (2014) for a review of MCDA methods. Also, in R&D projects, team learning processes have been suggested as a means for managing trade-offs (Clancy et al. 2013). Regardless of method, it is commonly acknowledged that managing trade-offs between impact categories are inherently value-based activities (Finnveden et al. 2009; Hertwich & Hammitt 2001). However, the rapidly growing scientific understanding of the anthropogenic pressures on the Earth system and the risks of transgressing thresholds of biophysical processes – as manifested by the planetary boundaries concept (Steffen et al. 2015b) – suggests that certain environmental impacts are, from a scientific viewpoint, more urgent to mitigate than others. This knowledge could potentially offer a new approach for managing trade-offs between impact categories. Indeed, there has been one attempt to derive a weighting method for LCAs from the planetary boundaries (Tuomisto et al. 2012). However, one of the main opportunities presented by the planetary boundaries concept has not yet been utilised for the benefit of LCAs: that it can be used for quantifying absolute targets for impact reduction. Utilising this is potentially useful for addressing questions i–iv above.

To contribute to the operationalisation of the planetary boundaries concept into LCA practices, research question 6 addresses the question of whether or not we can use the planetary boundaries concept for setting product-scale targets for impact reduction in LCAs. If it is possible, it is hypothesised that such targets can help LCA practitioners to make sense of LCIA results, by aiding in evaluating the potential benefits of the product under development in different impact categories in relation to the impact reduction necessary at a global scale. Such targets can also help LCA practitioners to prioritise impact categories and manage trade-offs between them. Product-scale targets can also be a basis for formulating firm- or sector-level targets for impact reduction, which can enhance the role of LCA in strategic work for mitigating environmental impacts. Furthermore, it is hypothesised that an alignment between global challenges and impact reduction targets at the product scale would help empower those involved in technical R&D projects to make relevant environmental interventions in product development, which is a prerequisite for an effective environmental management system (ISO 2004) to meet global environmental challenges. That product-scale mitigation work is increasingly important for firms and organisations is also stressed by recent proposals on expanding the scope of ISO 14001 from organisation-oriented to product life cycle-oriented (as discussed in Chapter 1).

Research question 6 was addressed in Paper VI by proposing a procedure for translating the global impact-reduction targets suggested by the planetary boundaries into impact-reduction targets at the scale of products or functional units, and by testing out the procedure in a specific LCA context: Swedish clothing consumption (the context of the Mistra Future Fashion research programme, see Section 2.4). The procedure and its application in the LCA context are further described in Paper VI. Section 4.6 shortly describes the procedure and the findings of Paper VI and Section 5.4 contains a discussion on how these findings contribute to addressing research question 6.

### 3.4 Positioning the thesis within systems science and systems analysis

A system has been defined as a "set of objects together with relationships between the objects and between their attributes" (Young 1964). Systems science and systems analysis have emerged as disciplines dealing with problems associated with behaviours of systems and/or interactions between systems, as a response to the fact that such problems are sometimes not satisfactorily dealt with by reductionist nor stochastic research methods. For example, reductionist approaches are insufficient when systems have characteristics or behaviours which occur at a system level but which cannot be deducted from its constituents (Klir 1991). The problems dealt with by systems science and systems analysis are also characterised by: (i) a high degree of complexity and the need for inter-disciplinarity problem-solving (Ingelstam 2002); (ii) the need for holistic systems thinking to avoid a narrow focus on efficiency and sub-optimisations (Churchman 1968); (iii) their "wickedness", i.e. they are unique, difficult to formulate, ever-changing, lack an optimal solution and concern competing value systems or objectives (Coyne 2005; Norton 2005; Rittel & Webber 1973); and (iv) the need for flexible use and design of methods (Midgley et al. 1998), often combining concepts and methods from different scientific disciplines (Sandén & Hillman 2011; Assefa & Frostell 2006; Simon 1962) that must be adapted to the specific context and, as a consequence, the solution is only valid for that specific problem (Sterman 1991; Simon 1962). As these types of problems are common in many scientific disciplines, there has emerged an extensive scientific literature dealing with the study of systems in general (e.g. Ingelstam 2002; Klir 1991; Boulding 1985; Bertalanffy 1968; Churchman 1968; Forrester 1968; Simon 1962).

A sub-discipline within systems science and systems analysis is environmental systems analysis (Finnveden & Moberg 2005), which deals with the sustainability challenges of the interactions between socio-technical systems (e.g. products, services, organisations, companies, projects, regions or nations) and environmental systems at different scales (e.g. habitats, biomes, ecosystems, oceans, the climate or the entire Earth system) – challenges that have been characterised as wicked (Seager et al. 2012). LCA can be considered a sub-discipline within environmental systems analysis, and the research in the present thesis can be described in terms of systems and the behaviours and interactions of systems, while the research questions and methods exhibit typical systems science and systems analysis traits.

At an overall level, the context of the thesis can, using the above definition of systems by Young (1964), be described in terms of four systems (Fig. 5): the environmental system[5], the technical product system, the methodological system of LCA, and the contextual system of the R&D project. In the LCA, the technical product system is defined in the goal and scope definition (as described in Section 3.2), which clearly parallels the general systems theory's emphasis on defining "the system's general goals", "the system's surroundings" and "the system's components, [and] their activities" (Churchman 1968, p. 35). Also, the fundamental idea behind LCA – to avoid sub-optimisation and problem shifting by studying the entire life cycle of a product and a wide (and as complete as possible) range of impact categories – is, as described above, typical for systems science and systems analysis. Further examples of traits from systems science and systems analysis are present in relation to research question 1 – concerning how to better align the use of LCA (system C in Fig. 5) with the R&D context (system D) – since the question is

[5] The environment could also be referred to as a social-ecological system (Folke 2006) as "there are neither natural or pristine systems without people nor social systems without nature. Social and ecological systems are not just linked but truly interconnected and co-evolving across spatial and temporal scales" (Stockholm Resilience Centre 2007). The interconnection of ecological and social systems is also present in LCIA methodology, where the areas of protection in endpoint-modelling typically cover both ecological and social values (see Section 3.2).

dealt with by methods not traditionally used in the LCA discipline (qualitative research methods, as described in Section 3.3.1), and since the proposed solution emphasises the need for contextual adaptation of LCA practices. As described in the previous paragraph, both the combination of methods from different disciplines and the need for contextual adaptation are typical for systems science and systems analysis. These systems science and systems analysis traits are present also in the approaches for dealing with research questions 4 and 5, which concern improved understanding of the interactions between the product system (system B) and the environment (system A) and how the assessment of these interactions (by means of LCA, system C) depends on the R&D context (system D). In addressing question 4, the thesis emphasises contextual dependencies of impact assessment practices, and in addressing question 5, knowledge from one discipline – ecology of a local ecosystem scale, in this case how biodiversity loss and water cycle disturbances depend on local scale parameters – was used to improve the methodology in another discipline (in this case LCA, traditionally dealing with regional- and global-scale environmental impacts). Another example of the cross-fertilisation of disciplines is provided by the approach for dealing with research question 6, which draws on knowledge from the Earth sciences (in terms of the planetary boundaries concept) as well as policy research and applied philosophy (in terms of different ethical rationales for allocating emission budgets). Besides, taking different ethics into consideration has been identified as being important when dealing with the typical wicked problems of systems science and systems analysis (Seager et al. 2012). A final example of the systems science and systems analysis traits of the present thesis is provided by research question 2, which focuses on improved modelling of future uncertain product systems (system C). The modelling approaches used to address the question can be classified as foresight activities, which is a key branch of systems science (Ingelstam 2002).

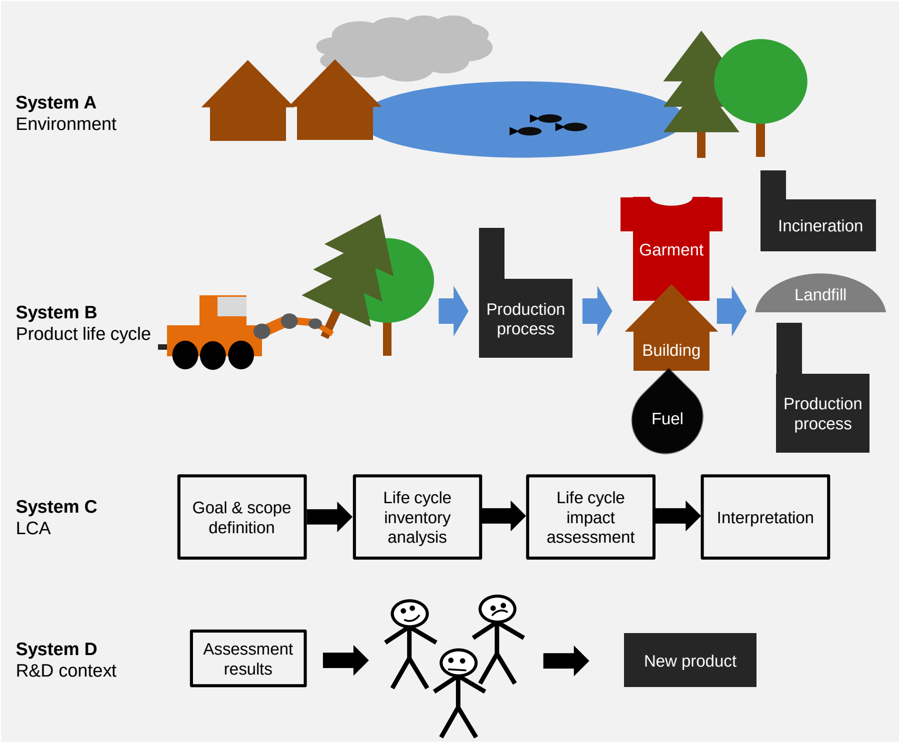


**Fig. 5** A representation of the research in the present thesis in terms of four systems (A–D). The research questions in the thesis deal with different behaviours of, and interactions between, these systems.

The aforementioned need of being flexible and creative with regard to the choices and uses of methods when dealing with typical systems analysis problems highlights a key limitation of the present thesis: its focus on one particular environmental assessment methodology, LCA. For some of the addressed research questions, other methods could potentially be more appropriate or complementary to the use of LCA. The need for using several complementary methods in environmental assessments has been emphasised before; for example, by Chico et al. (2013) for assessing water cycle disturbances and by Buonocore et al. (2014) for environmental assessments of forestry operations. In addition, many of the methods for environmental consideration in R&D mentioned in Section 3.2.1 promote the use of other, often complementary, methods. It is thus important to stress that, just as the use of LCA must be adapted to the specific context of the study – in terms of the choice of method for handling multi-functionality, the choice of impact assessment methods and the choice of scenario modelling approach – the choice between LCA and other methods must also be made in light of the specific context.

In particular, alternative environmental assessment or risk assessment methodology may be needed until certain dimensions of the LCA methodology have been further developed, such as sufficiently location-specific impact assessment methodology. One such example was reported in Publication B, in which we needed to – because of insufficient LCA methodology – use complementary toxicological testing to assess the environmental risks of nanoparticles (see list of publications, page vii).

# 4 Summary of Papers I–VI

This chapter summarises the content and findings of Papers I–VI. For further details, see the appended papers.

## 4.1 Paper I

In Paper I, we explored how project commissioners, project managers and LCA practitioners can select an appropriate role for LCAs in inter-organisational R&D projects, and how the project can be planned in accordance with the selected role. This was performed in an iterative procedure including reviewing literature on possible roles of LCA in R&D processes, identifying project characteristics that determine the LCA role based on analyses of experiences from five specific inter-organisational R&D projects (including the CelluNova and WoodLife projects described in Chapter 2), discussing how the identified project characteristics can be evaluated to help the pre-project planning, and testing the universality of the findings by gathering input from LCA practitioners outside Europe.

The following potential roles of LCA in R&D processes were found: (i) guide the technical R&D, (ii) develop life-cycle thinking in the project team, (iii) support scale-up of the developed technology, (iv) direct future R&D activities, (v) market the developed technology, (vi) demonstrate inclusion of environmental concerns, (vii) contribute to LCA knowledge, and (viii) fulfil a requirement of the commissioner or funding agency that an LCA has to be performed in the project. The identified project characteristics that determine the LCA role are: (i) the project's potential influence on environmental impacts, (ii) the degrees of freedom available for the technical direction of the project, (iii) the project's potential to provide required input to the LCA, and (iv) access to relevant audiences for the LCA results.

## 4.2 Paper II

In Paper II, we evaluated how end-of-life assumptions influence the LCA comparisons of two alternative roof constructions: glulam beams (a potential area of application for the adhesive developed in the WoodLife project, see Section 2.1) and steel frames. The study covered impact categories often assessed in the construction industry: total and non-renewable primary energy demand, water depletion, global warming, eutrophication and photo-chemical oxidant creation. We developed a scenario modelling approach to test two parameters of the end-of-life modelling: assumptions regarding the future technologies of the end-of-life processes, and the choice between attributional and consequential modelling approaches. Technical elements that were varied include: energy source used in the demolition of the construction, fuel type used for transportation to the disposal site, means of disposal, and method for handling multi-functionality in the end-of-life modelling. We tested two alternative assumptions regarding technology development: (i) in the future, there is no development from today's state of technology; and (ii) in the future, today's low-impact technologies are representative of the average technology. For allocating environmental impacts of the waste handling to by-products (heat or recycled materials), we compared an attributional cut-off approach to a consequential substitution approach, and also considered a scenario excluding all end-of-life processes.

In all comparable scenarios, glulam beams showed clear environmental benefits compared to steel frames, except in a scenario where steel frames are recycled and replace today's average steel production, for which impacts were similar to comparable glulam beams scenarios. So, no particular approach (attributional, consequential or fully disregarding end-of-life processes) favoured one roof construction alternative over another. On the other hand, four factors were revealed to be critical for the results in *absolute* terms: whether end-of-life phases are considered at all, whether recycling or incineration is assumed in the disposal of glulam beams, whether consequential or attributional end-of-life modelling approaches are used, and whether today's average technology or a low-impact technology is assumed for the substituted technology.

## 4.3 Paper III

Paper III reported on a study of the consequences of the choice of method for handling multi-functionality in LCAs of biorefinery products from forest biomass, including an evaluation of which decision contexts that are most affected by the choice of method. We tested six methods in a case study of a biorefinery using pulpwood as feedstock. Tested methods included: (i) main product bears all environmental burden; (ii) substitution; (iii and iv) traditional partitioning methods based on economic value or exergy; (v) a hybrid method combining elements of substitution and partitioning (developed by Cherubini et al. 2011); and (vi) an alternative (or inverted) hybrid method (developed by us), which allocates a lesser environmental burden to co-products with a higher potential to mitigate environmental burdens. The methods were tested in relation to four distinctly different decision contexts relevant for biorefineries, which were identified through a structured combination of contextual dimensions.

The results varied considerably between the tested methods, particularly between the method where the main product bears all burden, more traditional partitioning methods and methods accounting for reference products (substitution or hybrid partitioning). The results imply that the choice of method for handling multi-functionality is a critical methodological choice in LCAs of biorefinery products, especially in consequential studies and in studies of physically non-dominant co-products.

Additional findings relate to the investigated hybrid methods, which are potentially useful in consequential policy-related contexts which concern the mitigation of a certain environmental impact, where there are both material and immaterial co-products, where it is not feasible or desirable to choose a main product, and where it is desirable for results to be independent of fluctuating economic values. There are, however, some concerns and limitations of the hybrid methods. The method proposed by Cherubini et al. (2011) is potentially problematic because it favours products with a low potential to mitigate environmental burdens, and the results of the alternative hybrid method (which instead favours products with a high mitigation potential) were shown to be strongly influenced by the relative physical scale of the co-product flows.

### 4.4 Paper IV

In Paper IV, we evaluated how LCA practitioners assess the climate impact of forest products and the implications of the choice of impact assessment methods and practices for LCIA results and decision-making. First, we reviewed 90 published LCAs of forest products to reveal the current methods and practices for climate impact assessment. Next, we compared LCIA results from using the current methods and practices with results using non-established methods and practices, by means of case studies comparing the climate impact of a fuel and a building from forest and non-forest feedstock, respectively. Finally, we studied the implications of different methods and practices in relation to four distinctly different decision contexts identified through a structured combination of contextual dimensions. In short, these contexts were: (i) guiding a development project aimed at incremental improvements of an existing forest product, (ii) guiding public procurement, (iii) determining the priorities of a national research programme, and (iv) guiding policy-making aimed at supporting low-carbon technologies for maximum climate impact mitigation.

Results indicated that the current methods and practices exclude most dynamic features of carbon uptake and storage as well as climate impact of indirect land use change, aerosols and changed albedo. The case studies and decision context analysis demonstrated that including these aspects can strongly influence LCIA results, both positively and negatively, with potentially important implications for decision-making. For example, the elements of the product life cycle that have the greatest climate-impact reduction potential might not be identified, product comparisons might favour the product with highest climate impact and policy instruments might support the development and use of inefficient climate-mitigation strategies.

### 4.5 Paper V

Paper V reported on an LCA focussing on the biodiversity loss and water cycle disturbances of bio-based textile fibres, in which we used impact assessment methods proposed in the literature. The method for water cycle disturbances considered water deprivation at the midpoint level and the impact on human health, ecosystem quality and resources at the endpoint level. For inventorying water cycle alterations, we developed and applied an innovative consequential LCI approach and compared this to a more traditional attributional approach. We set up five production scenarios of regenerated cellulose fibres (such as the fibres developed in the CelluNova and ForTex projects, see Section 2.2) to account for uncertainties in the future location of operations and the possible occurrence of land use change (also referred to as transformation of land). For comparison, we set up two cotton production scenarios.

The results showed that the biodiversity loss due to land transformation was much higher than biodiversity loss due to land occupation. If land transformation occurs, and all impact is allocated to the first harvest, cotton production appeared to have a particularly high impact. However, if the transformational impact is allocated over several subsequent harvests, the impact of cotton and wood-based fibres appeared to be similar.

The impact assessment of water cycle disturbances showed that the location of operations is critical for the results, as water extracted from relatively water-stressed environments cause higher impacts. Furthermore, for some scenarios, the result differed considerably between the novel consequential LCI approach and a traditional attributional approach, and it was shown that the consequential approach adds the possibility of recognising increased runoff as a potential benefit for certain types of land use.

## 4.6 Paper VI

In Paper VI, we proposed a procedure for translating the planetary boundaries into impact-reduction targets at the scale of products or functional units. The procedure consists of four steps: (i) identifying the planetary boundaries quantified in literature that correspond to an impact category which is studied in the LCA in question, (ii) interpreting what the identified planetary boundaries imply in terms of reduction of the current global environmental impact, (iii) allocating the allowed global impact to the particular global market segment to which the studied product belongs, and (iv) allocating the outcome of step iii to the functional unit of the studied product. We tested the procedure by applying it in a specific LCA context: a study of Swedish clothing consumption (the context of the Mistra Future Fashion research programme, see Section 2.4). In doing this, we tested (a) three different allocation rationales for step iii based on different ethical principles, (b) four different allocation rationales for step iv based on different ethical principles for allocating emissions rights, and (c) the procedure's sensitivity towards the geographical context by changing the geographical scope from Sweden to Nigeria. In the paper, we used the planetary boundaries as defined by Steffen et al. (2015b).

The case study showed that the procedure is feasible for use by LCA practitioners when setting product-scale targets for impact reduction. The outcome of the procedure implied that all or most of the impact of Swedish clothing consumption should be eliminated to respect the planetary boundaries. The outcome depended strongly on impact category, which suggests that the procedure can help prioritise impact categories and manage trade-offs between them. The outcome also depended strongly on the geographical context of the study and the chosen rationale for allocating the globally allowed impact between market segments and residents of different regions.

# 5 Discussion of research findings

This chapter contains a discussion of how the presented research contributes to addressing the research questions of the thesis. The chapter is divided into four sections corresponding to the grouping of research questions in Section 1.3. The research questions are handled one at a time.

## 5.1 Planning LCA work in technical inter-organisational R&D projects

1. Which project characteristics determine the availability of roles for LCAs in technical inter-organisational R&D projects?

In Paper I, we identified four project characteristics that seem to determine the availability of LCA roles in technical inter-organisational R&D contexts. If project commissioners, project managers and LCA practitioners would consider the full range of possible roles (eight such roles were identified in Paper I), and evaluate how the four project characteristics influence role selection in their specific project, it is probable that the role selection would be better aligned with the project context compared to what is typically the situation today. This, however, remains to be tested in practice. Since the experiences analysed in Paper I relate to projects in which LCAs have had just a few different roles, it would be particularly interesting to test the theory in contexts where a wider range of LCA roles are selected. This would also help in developing the theory further – for example in terms of modifying the proposed project characteristics or adding other ones – and expanding its applicability.

The project characteristics should be evaluated as early as possible in the planning of the project, as a basis for role selection, and project planning should then be done in a way that supports the selected roles. Important aspects of planning can be the timing of LCA activities in relation to other activities in the project (including delivery of data to the LCA work) and the communication plan for disseminating LCA results, with clearly defined audiences, internally and/or externally to the project. After evaluation of the project characteristics, it may turn out that the desired roles are not available. In such situations, the project manager must remove barriers to the desirable roles, or convince those that request roles that the desired roles are unavailable, a process termed “expectation management” in management and organisation theory (Bosch-Sijtsema 2007).

The input from non-European LCA practitioners gathered in Paper I suggests that the roles of LCA in R&D contexts can be confined not only by the project characteristics proposed above, but also by the traditions among funding agencies and LCA practitioners. This suggests that more wide-spread knowledge of the possible roles of LCA in R&D projects could expand the use of LCA in funding contexts where it is not traditionally used. This would also expand the applicability of the theory generated in Paper I.

As recognised both in Section 3.3.1 and in Paper I, the research community has paid little attention to LCA applied in technical inter-organisational R&D projects – although some of the above-discussed findings probably represent tacit knowledge amongst LCA practitioners. Paper I thus opens up a new field of research. Hopefully, other LCA practitioners will begin to share their experiences from similar contexts in the form of formalised research – this would be important for improving the use of LCA in technical inter-organisational R&D projects, which can be expected to be an increasingly common context for LCA work, at least in Europe. Improving the use of LCA in a certain context is a means for ensuring that the LCA has as much impact as possible in that context, which can be seen as improving the “intervention” capacity of the study, in the terminology by Sandén and Harvey (2008). Sandén and Harvey recognised (as is consistent with my own experiences) that intervention activities are not often seen as an integral part of systems studies. They suggest that intervention activities warrant greater effort, and therefore suggest that intervention should be seen as one of four “core activities” of systems studies, along with problem selection, system design and assessment (Sandén & Harvey 2008). Increased focus on intervention activities, in the context of product development, has also been suggested by Clancy (2014). The research of Paper I on improved role selection can be seen as contributing to improved intervention for a certain type of systems studies: LCAs carried out in the context of technical inter-organisational R&D projects.

## 5.2 Modelling future forest product systems

2. Which inherent uncertainties of future forest product systems should be considered in LCAs?

As mentioned in Section 3.2.1, previous research has recognised some inherent uncertainties of future product systems that prospective LCAs should consider, for example uncertainties in the LCI data for commercial scale processes that today only exist at laboratory scale (Hetherington et al. 2014). The research presented in this thesis has identified some additional uncertainties that appear to be typical for future *forest* product systems, and thus are important to consider in LCAs of such systems.

In Paper II, we identified some uncertainties of the product system which strongly influence LCIA results, namely the type of technology assumed in the disposal of the studied product (e.g. recycling or incineration), and the technology assumed for the substituted processes when substitution is used for handling multi-functionality (e.g. whether today's average technology or a low-impact technology is assumed). That these factors can be important in LCAs of forest products is supported by the results of Paper III (on the handling of multi-functionality) and Paper IV (on climate impact assessment).

Other key uncertainties of future forest product systems are the location of processes (in particular forestry operations) and the occurrence of land use change. These factors can be important in assessing climate impacts (Paper IV), biodiversity loss (Paper V) and water cycle disturbances (Paper V), each a highly relevant impact category for LCAs of forest products.

The further discussion in this section focusses on the limitations and opportunities of different approaches to modelling the aforementioned uncertainties in LCAs. In Papers II and V, we used two different approaches for modelling scenarios of possible future states, in order to evaluate the influence of uncertainties in the product systems (as described in Section 3.3.2). The scenario modelling approach of Paper II made it possible to show that some modelling choices are particularly important, namely the choice to include or exclude end-of-life processes and the choice between consequential and attributional modelling approaches. The results showed that when the LCA practitioner has chosen to use consequential end-of-life modelling with substitution, it appears to be particularly important to set up several distinctly different scenarios, because the outcome of the assessment largely depends on speculative assumptions regarding the type of substituted technology, as has been recognised before (Pelletier et al. 2015; Zamagni et al. 2012; Mathiesen et al. 2009; Heijungs & Guinée 2007). Until there is consensus in the LCA community on when to use attributional and consequential approaches, there is a general need to use both approaches simultaneously – as long as each approach is reasonably justified in the context of the study – to facilitate robust LCA-based decision making. There is also a need to generate several distinctly different scenarios for each approach, for example regarding the type of substituted technology for consequential end-of-life modelling. Use of both approaches, and several scenarios in each approach, within a single case study, can improve our understanding of the approaches and under what circumstances the selection of approach matters most.

The scenario modelling approach of Paper II introduced a temporal dimension in the modelling of the product system to distinguish between processes occurring in the near versus distant future. This approach could potentially offer additional benefits if combined with impact assessment methods that can account for the temporal dynamics of environmental impacts. As discussed in Section 3.3.5, there are many proposals for (and potential benefits of) temporally dynamic methods for assessing climate impacts; for example, the dynamic metrics proposed by Levasseur et al. (2010), which give less weight to emissions the further into the future they occur. Temporally dynamic methods have been suggested also for the impact assessment of water cycle disturbances (Núñez et al. 2015). Such methods require the introduction of a temporal dimension in the product system model. Consequently, if they gain widespread use, LCA software will probably support temporally dynamic process flowcharts, and thus will facilitate the scenario modelling approach of Paper II.

It should be noted that the scenario modelling approaches of Papers II and V were developed for the specific needs in each case study. For example, it would have made less sense to introduce a temporal dimension in the product system model of the LCA of textile fibres in Paper V, as a typical garment is disposed of after a rather short time period[6] (apart from the fact that the study excluded end-of-life processes, as they were expected to be of limited importance for the studied impact categories). The appropriate scenario modelling approach for a specific case study will always depend on the unique context of that study: the approach must generate some understanding that supports the LCA practitioner in reaching the goal of the study, ultimately improving decision-making based on the study. It is thus important that any scenario modelling approach is developed for, or adapted to, the specific context of each study – as stressed by Sterman (1991) for system modelling in general.

3. What are the consequences of the choice of method for handling multi-functionality in LCAs of forest products?

The testing of six methods for handling multifunctionality in Paper III – including an alternative hybrid method we developed – showed that the choice of method can strongly influence the LCIA results and decision-making based on those results.

[6] In UK, the average service life of apparel is 2.2 years (The Waste and Resources Action Programme 2012).

The choice of method was shown to be particularly important in consequential studies and in studies focussing on multi-functional processes with co-product flows of small physical scale. Results also showed that differences in the physical scale of co-product flows can determine the feasibility of applying certain methods; in our case: the alternative hybrid method, for which it may be appropriate with a cut-off rule regarding the relative physical scale of co-product flows. That the relative physical scale of co-product flows can influence the choice of method for handling multi-functionality and the need for comparing different methods is not recognised in LCA standards and guidelines such as ISO 14044 (ISO 2006b), the ILCD handbook (EC 2010) and the Product Environmental Footprint (PEF) recommendations (EC 2013). Recognising the role of the relative physical scale of co-product flows would improve standards' guidance on handling multi-functionality. This recommendation adds to a growing body of recommendations for improving existing standards' guidance on handling multi-functionality (see, e.g., Pelletier et al. 2015; Weidema 2014).

Paper III includes a further analysis and discussion of how the choice of method for handling multi-functionality influences the LCIA results and decision-making in specific decision contexts. Each of these decision contexts (and the associated LCA questions) are of potential relevance for the R&D context in focus in the present thesis, for which there are many potentially relevant roles of LCA (as listed in Section 4.1 and Paper I) and likewise many potentially relevant LCA questions.

### 5.3 Assessing environmental impacts of forest products

4. What are the potential shortcomings of current methods and practices for climate impact assessment in LCAs, in decision contexts relevant for forest products?

In Paper IV, we identified the currently most common climate impact assessment methods and practices in LCAs of forest products, and tested different climate impact assessment methods and practices in LCA case studies comparing buildings and fuels from forest and non-forest feedstocks. The LCIA results depended considerably on the choice of methods and practices, as is consistent with previous studies (see Section 3.3.5). By discussing the results in relation to four decision contexts relevant for LCAs of forest products (as described in Section 4.4), we identified several potential shortcomings of the current methods and practices. In studies concerned with identifying the environmental hot-spots in a product system (contexts 1 and 3 in Section 4.4), results imply that the current methods and practices could potentially miss important hot-spots found by more elaborate methods and practices – for example because of the simplified climate neutrality assumption for biogenic carbon dioxide emissions or the omission of soil carbon disturbances and changed albedo – and could thus provide insufficient decision support. In studies concerned with comparing forest and non-forest products (contexts 2 and 4 in Section 4.4), the results implied that the current impact assessment methods and practices may often show that forest products are preferable, but that this could change with more elaborate, non-established climate impact assessment methods and practices. For LCAs of product systems with undefined geographical locations (contexts 3 and 4 in Section 4.4), a limitation of non-established methods was identified: the difficulty of finding average data for the location-specific data needed to account for many of the location-dependent aspects of climate impact (e.g. information on forestry practices and soil conditions, or information on regional weather patterns influencing the albedo effect).

A contextual dimension that was not analysed by means of the four decision contexts of Paper IV was whether the study is done to support short- or long-term climate impact mitigation. The influence of this dimension was, nevertheless, discussed in the paper. It was concluded that in contexts dealing with short-term climate impact mitigation, it is preferable to use climate impact metrics based on a shorter time perspective (e.g. $GWP_{20}$), or to use dynamic metrics or to discount future emissions – options which are not among the established methods and practices. In such contexts, if the harvested forest is slow-growing, it could also be suitable to account for temporary carbon storage (e.g. in long-lived products) and the climate impact of the temporary contribution of biogenic carbon dioxide emissions to radiative forcing. Moreover, short-term mitigation implies that albedo effects and soil disturbances could be relevant to account for as long as these are short-term effects. End-of-life substitution credits are probably less important to consider, particularly in studies of long-lived products. Also in contexts dealing with long-term climate impact mitigation, dynamic metrics could be suitable. However, in contrast to contexts dealing with short-term mitigation, metrics based on a longer time perspective (e.g. $GWP_{500}$) are preferable. Also, it could be justifiable to disregard biogenic carbon dioxide emissions (if it can be reasonably assumed that the harvested biomass is carbon-neutral in the long term), soil disturbances (if these can be assumed to be reversed over time) and the albedo effect (unless harvesting results in permanent deforestation). It thus appears that the current impact assessments methods and practices, which do not consider most of the aforementioned climate impact aspects, are more suitable for contexts dealing with long-term impact mitigation, than for contexts dealing with short-term impact mitigation.

The aforementioned potential shortcomings warrant the development of more sophisticated methods and practices for climate impact assessment that consider more of the climate impact aspects of forest product life cycles, and the inclusion and consideration of such methods and practices in important standards, guidelines and directives, such as ISO 14040/14044 (ISO 2006a, 2006b), the ILCD handbook (EC 2010), the Renewable Energy Directive (EC 2009b), and the PEF recommendations (EC 2013). Omitting climate impact aspects known to be important from impact assessment modelling is simply not acceptable – or, in the words by Sterman (1991, p. 12) on system modelling in general: "ignoring a relationship implies that it has a value of zero – probably the only value known to be wrong".

It was also apparent from results in Paper IV that methods and practices are seldom clearly stated and/or motivated in published studies, and often do not seem to have been chosen based on the specific context of the study. This calls for improved practices of LCA practitioners both in terms of more context-adapted methodological choices and in terms of being transparent about the choices made.

Improved methods and practices can ensure more accurate assessments of the climate impact of forest products – as has previously been acknowledged as being important (Ter-Mikaelian et al. 2015) – and ensure the avoidance of over-simplified and untrue statements regarding the benefits of forest products vis-à-vis non-forest products. This will be essential if we are to use forest and non-forest resources efficiently for mitigating climate change.

5. How can we improve the assessment of biodiversity loss and water cycle disturbances of forestry?

A new method for impact assessment does not necessarily imply an improvement of existing methods (Baitz et al. 2013). This section discusses whether the methods in Paper V for assessing biodiversity loss and water cycle disturbances represent improvements beyond established methods, and if so, what type of improvements they can offer today and how they can be improved further.

The consequential LCI approach for quantifying water cycle alterations offers the possibility to calculate catchment-scale effects of land use and land use change not possible with previously proposed LCI approaches. In the case study of Paper V, the consequential LCI approach generated considerably different results compared to a traditional attributional approach, which indicates that the inclusion of such effects can be important for LCAs of forest products. Furthermore, the consequential approach made it possible to recognise increased runoff as a potential benefit of certain types of land use; a development of the impact assessment which reflects realities in a meaningful way.

Apart from potentially providing a more accurate quantification of water cycle alterations, the consequential LCI approach could offer further opportunities if combined with more elaborate LCIA methods. For example, a smaller time step in the estimation of runoff change could theoretically make it possible to account for the fact that increased water runoff may be a potential disadvantage in some regions, by expanding the LCIA modelling with an index on the sensitivity to flooding, erosion, soil salinity or similar. It has even been argued that the beneficial impacts of increased runoff due to land use change should never be accounted for unless damaging effects are also considered (Berger & Finkbeiner 2012). Apart from the aforementioned damaging effects due to increased runoff, Berger and Finkbeiner acknowledged other potentially damaging effects from land use change, such as decreased precipitation in other catchments. Including effects such as flooding would be a considerable development of established methods for assessing water cycle disturbances – which solely focus on water deficiency as a potential issue. The desirability of such advancement is the primary reason for why the impact category is, in the present thesis, referred to as “water cycle disturbances” rather than the more traditional term “water use impact” or some term referring exclusively to water deficiency (the terminology is further discussed later on in this section).

The consequential LCI approach for quantifying water cycle alterations can potentially also improve the calculation of water impact offsets in natural resource management; for example, where a change in land use practices causes a hydrological benefit. An offset may be thought of as an environmental benefit (including, but not limited to, avoided impacts) when a system performs some secondary function, and is a concept already used in natural resource management (the offset concept is further explained in Paper V). Offset calculation does not make sense in an attributional LCI approach that focusses on the evapotranspirational losses of crops, where one misses catchment-scale factors such as the interception of rainfall by vegetation or forestry road construction (Bruijnzeel 2004). If offset calculation is to be feasible, all of these factors need to be taken into account at the catchment scale, and the actual hydrologic consequences of the forestry operation estimated. Offset calculation could thus be facilitated by the proposed consequential LCI approach. Note that offset calculation is similar to the consequential approach for handling multi-functionality (see Section 3.3.4).

When applying the consequential LCI approach for quantifying water cycle alterations in Paper V, some rough estimates of runoff changes were made based on the scientific literature. These estimates are very uncertain; for example, estimates made for Swedish forestry resulted in an interval spanning from a negative to a positive change in runoff. Since Paper V was accepted for publication, others have argued that Scandinavian forestry (which includes Swedish forestry) has only a marginal influence on runoff (Launiainen et al. 2014). Thus, to increase the usefulness of the consequential LCI approach, there is a need for more accurate and abundant data on runoff changes for different land use types and management regimes. To achieve this, there is a need to discuss the spatial and temporal representativeness of available data, and collect new data when necessary. Without high-quality data, there is a risk that increasing sophistication will add more noise than information (Hertwich & Hammitt 2001), and will fail to contribute to improved decision support.

The LCIA method for water cycle disturbances applied in Paper V (developed by Pfister et al. 2009) resulted in higher impact scores for life cycle processes located in water-stressed areas. This is an obvious advancement compared to the most commonly used impact assessment methods in LCA, which simply give the volume of water used, sometimes distinguishing between surface water and groundwater, but disregarding whether this represents an actual alteration of the water cycle and not acknowledging the potential consequences further down the cause-effect chain. In the context of the CelluNova and ForTex projects, the LCIA method made it possible to quantify the potential benefits of carefully siting the fibre production plant and sourcing the forest biomass.

The method for assessing the biodiversity loss of forestry applied in Paper V (developed by Schmidt 2008) also enabled us to identify important product system parameters beyond what would have been possible by using less sophisticated methods (such as the simple indicator on land use described in Section 3.3.5: the area of land used and the duration of that use). For example, transformation of land from a high- to a low-biodiversity state was shown to contribute much more to the biodiversity impact than occupational land use. Moreover, the location of land use was shown to be of low importance in the case study, as geographical differences influencing the time from planting to harvest, the annual yield per land area, the renaturalisation time and the ecosystem vulnerability appeared to roughly offset each other.

Overall, the key advancement of the tested LCIA methods for biodiversity loss and water cycle disturbances appears to be the use of regionalised characterisation factors, which is a necessity for improving the assessment of such location-dependent impacts. The biodiversity impact assessment method cannot, however, assess differences between closely related land uses (such as different practices in forestry management), due to limitations in data availability. This is a recognised drawback also with other methods for characterising the biodiversity impact of land use and land use change (Antón et al. 2007). Therefore, the applied method is, in its current form, most useful for assessments supporting strategic macro-scale decision-making (e.g. whether to transform natural land or not, or which regions to source forest biomass from), but less useful for supporting micro-scale decision-making (e.g. what specific forest to source biomass from, or what land management practices to use). This drawback can, however, be overcome with more refined data, such as species richness data for more specific land management practices, which could facilitate comparisons between uncertified land and land managed according to certain certification principles, such as the Forest Stewardship Council (FSC 2014) or the Programme for the Endorsement of Forest Certification (PEFC 2014). The assessment method for water cycle disturbances can also be used with more refined data. In the case study of Paper V, a more refined assessment was primarily confined by how specific the product system could be defined in geographical terms, rather than the availability of data – characterisation factors for the method have been published online for over 11,000 watersheds in a format compatible with Google Earth (ETH Zürich 2015). The consequential LCI approach for quantifying water cycle alterations could also be improved by more refined data, such as runoff data for specific land management practices.

Another critical methodological aspect identified in Paper V was the allocation of transformational impacts between the first harvest after transformation and subsequent harvests. This proved to be very important in the comparison of regenerated cellulose fibres made from biomass from forests with a rotation time of 62.5 years and cotton fibres from cotton plantations with a rotation time of 0.5 years. This allocation problem can be expected to be a recurring dilemma of importance for many comparisons of products derived from crops with different rotational times, for example in comparisons of forest products and other bio-based products. Indeed, the problem was recognised by Koellner et al. (2013b), where it was recommended that, as a base case in the absence of a scientifically robust alternative, the transformational impact should be allocated over 20 years regardless of crop (this recommendation was not available when Paper V was published). They also recommended testing the influence of other amortisation periods in a sensitivity analysis. Furthermore, Schmidt et al. (2015) recently proposed an approach for avoiding the allocation of transformational impacts; the proposed approach is, however, only applicable for the assessment of climate impact.

A further finding was that the assessment method for biodiversity loss could be improved and/or complemented by other indicators, on other groups of species or other facets of biodiversity and/or ecosystem quality. The need for multiple impact factors for the assessment of land use activities has been emphasised before (Koellner et al. 2013b; Curran et al. 2011). As found in the review of methods in Section 3.3.5, some of the proposed methods are based on multiple indicators. The method suggested by de Baan et al. (2012), using a relative species index, is perhaps the most promising development in this direction. Their method permits using several groups of species, where the choice of species groups can be based on the data available for a certain region. However, it does not yet support transformational impact assessment, which, as showed in Paper V, needs to be included in a robust method for assessing the environmental impacts of land use and land use change.

Since Paper V was published, the UNEP-SETAC life cycle initiative has come up with a set of general principles for LCI and LCIA methods for environmental impacts of land use and land use change, with a focus on impacts on biodiversity and ecosystem services (Koellner et al. 2013a, 2013b). These principles could prove to be an important step towards finding a consensus in the LCA community. The proposed principles are largely consistent with the method used in Paper V; they emphasise the need to distinguish between transformational and occupational impacts, recommend the use of reference states to identify changes in ecosystem quality, acknowledge loss of species diversity as an important link in the cause-effect chain between land activities and the areas of protection, recommend the use of absolute metrics for biodiversity loss (in contrast to relative metrics, such as the percentage of species lost within a given area), emphasise the need for geographical differentiation of characterisation factors, and acknowledge the importance of regeneration time and its geographical dependence.

To conclude, by using emerging methods for assessing biodiversity loss and water cycle disturbances in the case study of the CelluNova and ForTex projects, it was possible to identify the benefits and drawbacks of the methods and pinpoint where further research is needed. Also, by using a consequential LCI approach for quantifying water cycle alterations, it was possible to capture effects never captured before in impact assessments of forestry, and generate new ideas for applications and further developments of the impact assessment of water cycle disturbances. For the CelluNova/ForTex context, the applied methods led to findings that would have been missed if relying upon established methods. The conclusion that more elaborate methods for assessing biodiversity loss and water cycle disturbances generate findings that would be missed by established methods can be expected to be true also for studies of other forest products (or any other land-intensive product category).

Finally, there is a need to discuss the terminology surrounding the environmental impacts referred to in the present thesis as "biodiversity loss" and "water cycle disturbances". In the LCA community, these impact categories are traditionally referred to with the less specific terms "land use" or "land use impact" and "water use" or "water use impact", respectively. The terminology in the present thesis was chosen to avoid the potential confusion surrounding the traditional terminology. This confusion arises mostly because of ambiguities regarding where water and land use impacts are placed on the cause-effect chain. Although water and land use impacts are often referred to and presented as mid-point impact categories (i.e. located in the middle of the cause-effect chain from stressor to environmental effects), the traditional proxies used to describe them are actually inventory-level indicators (i.e. stressors), while water and land use in themselves are activities in the product system (i.e. causes of the stressors). Furthermore, the cause-effect chains of water and land use impacts and other impact categories are interconnected and overlapping. For example, land use (and land use change) contributes to water cycle disturbances (as acknowledged in this thesis and by e.g. Koellner et al. (2013b)) and climate change (as discussed in relation to research question 4). Our suggestion to rename the mid-point impact category traditionally termed water use or water use impact to water cycle disturbances is an attempt to clarify its meaning and reduce confusion. Such specification of water-related impact categories has been proposed before, for example Quinteiro et al. (2015) suggested the terms "blue water availability due to land use" and "green water deprivation", and Heuvelmans et al. (2005) suggested the term "regional water balance". A similar solution for the mid-point impact category traditionally termed land use or land use impact is to be more precise in terms of the type of impact concerned, by referring to it as, for example, "physical habitat disturbances", "soil disturbances" or similar, depending on what the impact assessment method actually models. Further down the cause-effect chain, there is a further need to be careful with the terminology. For example, the impact category of "biodiversity loss" most often relates to the biodiversity loss caused by land use and land use change via, for example, physical habitat disturbances (as is also the case in the present thesis), although many other mid-point impact categories (e.g. eutrophication, acidification and climate change) also contribute to biodiversity loss (MA 2005). This confusion can be avoided by either specifying the intended type of biodiversity impact in more detail (e.g. “biodiversity loss due to physical habitat disturbances”) or harmonising the end-point LCIA modelling of different mid-point impacts (e.g. so the impact category termed “biodiversity loss” in fact encompasses all the major drivers behind biodiversity loss). In any case, for effective and accurate communication of LCIA results, it is important to clearly distinguish between impact categories occurring at different places along the cause-effect chain (e.g. by presenting mid-point and end-point impact categories in separate figures) and to ensure that environmental impacts are not double-counted. To conclude, considering the ongoing, extensive progress of more elaborate impact assessment methods for a wide range of environmental impacts, the LCA community must increasingly resolve confusions related to terminology and eventually build a consensus on the terminology surrounding emerging methods. Clarity in terminology will improve the clarity of discussions aimed at managing conceptual and modelling challenges in LCA.

## 5.4 Connecting LCAs to global challenges

6. Can we use the planetary boundaries for setting product-scale targets for impact reduction in LCAs?

The procedure proposed in Paper VI for translating the planetary boundaries into product-scale targets was shown to be applicable in the tested LCA context (a study of Swedish clothing consumption) for five out of the nine boundaries. We did not attempt to use the procedure for the remaining four boundaries. The boundary for the introduction of novel entities was excluded as it has not yet been quantified (Steffen et al. 2015b). The boundary for ocean acidification was excluded as it does not correspond to any impact category usually studied in LCA. Atmospheric aerosol loading and stratospheric ozone depletion were excluded due to difficulties in translating these boundaries into global targets for impact reduction (the latter was also deemed to be of low relevance for the case study). For the five planetary boundaries that we applied the procedure to, it was concluded that sufficient data are available to support the application of the procedure in other contexts concerned with other types of products and geographical scopes.

The outcome of the procedure was shown to depend strongly on value-based choices of how to allocate a limited resource (in this case: the impact allowed within the limits set by the planetary boundaries) between market segments and individuals. However, these value-based choices do not depend on subjective opinions on the urgency of different environmental impacts, as is often the case with traditional procedures for comparing different impact categories, such as weighting. The value-based choices of the proposed procedure instead relate to more fundamental values of what is considered a fair distribution of a limited resource. The proposed procedure therefore allows the LCA practitioner to base the interpretation of LCIA results on more fundamental values than is normally the case in LCAs and avoid biases related to subjective opinions on the importance of different environmental concerns. Moreover, by comparing the outcome of different ethical rationales (as in Paper VI) the procedure allows the LCA practitioner to transparently show the consequences of different ethics.

Apart from helping in prioritising impact categories and managing trade-offs between them, a main opportunity of using the proposed procedure is the formulation of *absolute* targets of impact reduction. Such absolute targets can be used to evaluate interventions for impact reductions at the product scale (e.g. in terms of technology development or policies; Paper VI provides an example of such an evaluation) and as a basis for formulating strategies and goals for impact reduction (at the level of products, firms or industrial sectors). Thus, the research presented in Paper VI represents a novel use of a concept that can potentially expand the use of LCA as a strategic tool in environmental management, which is consistent with recent proposals for the development of ISO 14001 (as was acknowledged in Section 3.3.6).

There are some important areas that need to be researched further to improve the applicability of the proposed procedure. In particular, there is a need to research the matching of planetary boundaries and impact categories (a topic not dealt with extensively in Paper VI) and the interpretation of the planetary boundaries (particularly the four boundaries not dealt with in Paper VI) in terms of global targets for impact reduction.

Further developments of the planetary boundaries concept will probably support a further operationalisation of the concept in LCAs. For example, the recent update of the concept connects environmental concerns at regional and global scales to a larger extent, and provides maps of the geographical distribution of environmental stressors for four of the planetary boundaries (Steffen et al. 2015b). This, and future developments in this direction, could pave the way for a more regionalised interpretation of what the planetary boundaries imply in terms of product-scale targets for impact reduction, and could expand the use of the proposed procedure. Such a development would benefit from more regionalised impact assessment methods in LCAs, such as those discussed in this thesis in relation to climate change, biodiversity loss and water cycle disturbances.

# 6 Conclusions

This chapter summarises the main conclusions from the discussion in Chapter 5; first a general conclusion and then conclusions for each of the research questions.

A key message of the thesis is the importance of context-aligned methods and procedures; for example, in terms of selecting LCA roles in R&D projects, handling multi-functionality, modelling scenarios and selecting LCIA methods. This relates to the fact that the main value of LCA lies in the decision support it can provide in a specific context. In other words, "LCA can be defended as a rational tool only through its use in decision making, and not as a scientific measuring device" (Hertwich & Hammitt 2001, p. 265). This suggests there are limits to the potential for prescriptive standards for carrying out LCAs of forest products.

1. Which project characteristics determine the availability of roles for LCAs in technical inter-organisational R&D projects?

In Paper I, we identified four project characteristics that appear to influence the appropriateness of different roles:

i. The project's potential influence on environmental impacts.
ii. The degrees of freedom available for the technical direction of the project.
iii. The project's potential to provide required input to the LCA.
iv. Access to relevant audiences for the LCA results.

2. Which inherent uncertainties of future forest product systems should be considered in LCAs?

The research in Papers II–V identified some uncertainties of future forest product systems that are important to consider:

i. The technology of the disposal of the studied product (particularly for long-lived products).
ii. The technology of substituted processes (when substitution is used for handling multi-functionality).
iii. The location of processes (in particular forestry operations).
iv. The occurrence of land use change.

3. What are the consequences of the choice of method for handling multi-functionality in LCAs of forest products?

In Paper III, we showed that the choice of method can strongly influence absolute LCIA results, particularly in consequential studies and in studies focussing on physically non-dominant co-products, although the choice of method did not strongly influence the ranking of the studied alternatives.

A recently proposed hybrid method for handling multi-functionality was shown to be logically counterintuitive. We thus developed an alternative version, which was logically more appealing. The applicability of the alternative version was shown to be strongly influenced by the relative physical scale of co-product flows.

4. What are the potential shortcomings of current methods and practices for climate impact assessment in LCAs, in decision contexts relevant to forest products?

In Paper IV, we revealed three main potential shortcomings:

i. In studies concerned with identifying hot-spots, current methods and practices could potentially miss important hot-spots found by more elaborate non-established methods and practices.
ii. In comparisons of forest and non-forest products, current methods and practices may show that forest products are preferable, although more elaborate non-established methods and practices may show that they are not.
iii. Current methods and practices appear to be more suitable for contexts concerned with long-term climate impact mitigation, than for contexts concerned with short-term mitigation.

5. How can we improve the assessment of biodiversity loss and water cycle disturbances of forestry?

The impact assessment methods tested in Paper V made it possible to generate results that, in a meaningful way, depended on the geographical location of the forestry and the presence of land use change, which is beyond what is offered by many of the established methods.

An LCI approach we developed made it possible – for the first time in LCA – to generate impact scores reflecting that forestry and land use change can positively contribute to downstream water availability.

With additional development and/or increasingly refined data, the methods could offer further improvements to the impact assessment of forestry. For example, the new LCI approach could enable the assessment of disadvantages of increased water runoff, such as flooding, erosion and soil salinity.

6. Can we use the planetary boundaries for setting product-scale targets for impact reduction in LCAs?

A procedure proposed in Paper VI made it possible to translate five out of the nine planetary boundaries into product-scale targets for impact reduction. For these five boundaries, data are available to support application of the procedure in a wide range of LCA contexts.

The outcome of the procedure was shown to depend strongly on value-based choices, and challenges remain with regard to associating the planetary boundaries with appropriate impact categories and interpreting some of the planetary boundaries in terms of global targets for impact reduction.

# 7 Future research needs

This chapter summarises future research needs related to the research of this thesis.

For further developing the theory of improved role selection in technical inter-organisational R&D projects, it will be important that LCA practitioners share their experiences from similar contexts. This goes against the norm of publishing only successful outcomes. Also, there is a need to test in practice whether an evaluation of the four project characteristics identified in Paper I actually leads to more context-aligned role selection and improved use of the LCA work in R&D projects. In particular, testing the theory in contexts where other LCA roles are selected could help in developing the theory further – for example in terms of modifying the proposed project characteristics or adding other ones – and expanding its applicability.

In relation to improved modelling of future product systems, there is a need for further research on scenario modelling in LCA, for example on how to validate the usefulness of generated scenarios, or on what types of scenario modelling that are suitable in different R&D contexts. As a suggestion, such research could review early attempts of scenario modelling published in the literature and compare the generated scenarios with the actual outcome. For example, by studying LCAs carried out in the past (e.g. 20 years ago), it would be possible to study how well different approaches for end-of-life modelling have managed to capture actual end-of-life practices.

Furthermore, there is a need for more research on attributional and consequential modelling in LCA, for example in relation to the modelling of end-of-life processes of long-lived products or the handling of multi-functionality. It is desirable to build a consensus within the LCA community on the circumstances under which attributional or consequential approaches (or hybrid approaches including elements of both), as well as specific methods for handling multi-functionality, are preferable. This can help LCA practitioners to make conscious and context-aligned choices of modelling approaches, which is not typically the case today (Zamagni et al. 2012). To facilitate this, there is a need to further evaluate the consequences of different methodological choices in different contexts, for example by building cornerstone scenarios for decision contexts as in Papers III and IV, or by using ready-made decision-context classifications, such as those provided by the ILCD handbook (EC 2010). More research is also warranted for exploring which end-of-life assumptions that are important in assessments of other wood-based construction materials (other than glulam beams), and in comparisons with other non-wood alternatives (other than steel frames).

There is a need for more case studies applying emerging LCIA methods for climate change, biodiversity loss, water cycle disturbances and other location-dependent impacts of forestry. In particular, there is a need for case studies that compare different methods with regard to their applicability in various contexts and how well their outcomes reflect realities, for example as was performed in Paper IV for climate impact assessment. Such research can contribute to concrete recommendations for increasingly context-aligned practices, for example in standards and guidelines. Concerning the proposed consequential LCI approach for quantifying water cycle alterations, there is a need to test its applicability in further case studies and explore its potential advantages in further detail, such as the possibilities of accounting for offsets and the capturing of environmental disturbances due to increased runoff. Concerning the assessment of biodiversity loss, there is a need to use case studies to compare different species richness indicators and how well they reflect realities, and explore how to combine such indicators with other indicators for biodiversity or ecosystem quality. A good example in this direction is the study by de Baan et al. (2012), on the correlation between an indicator on relative species richness and other biodiversity indicators. However, to facilitate comparisons, many of the most promising methods proposed in the literature need to be made less dependent on data that are only available for specific regions or product categories, either by adapting the methods to suit available data or by collecting data for a wider range of regions and product categories. Also, there is a need to further discuss how to allocate transformational impacts of forestry and agriculture between the first harvest after transformation and subsequent harvests, and eventually reach a consensus on a standard amortisation period, for example 20 years, as proposed by Koellner et al. (2013b).

The LCA community should work towards resolving confusion related to the terminology of emerging LCIA methods for biodiversity loss and water cycle disturbances. Clarity in terminology will improve the clarity of discussions on the conceptual and modelling challenges of assessing these impacts. Eventually, the LCA community should try to build a consensus on both the terminology and the methods for assessing these impacts. In the case of water cycle disturbances, this would require the harmonisation of on-going consensus processes for water footprint methodology (more or less intended for use in LCAs), including the ISO

14046 standard on water footprinting (ISO 2014), and the work by the WFN (Hoekstra et al. 2011), the UNEP-SETAC life cycle initiative (WULCA 2014) and the World Business Council for Sustainable Development (WBCSD 2015).

There is a need for other researchers to critically evaluate the proposed procedure in Paper VI for translating the planetary boundaries to product-scale targets for impact reduction. As a suggestion, there is a need to (i) test the procedure in other contexts, preferably where the potential of concrete interventions for impact reduction is evaluated; (ii) study the matching of planetary boundaries and impact categories, and the interpretation of the boundaries in terms of global targets for impact reduction; and (iii) explore the implications of other ethical rationales than those tested in Paper VI (and how assumptions, e.g. regarding temporal dynamics, influence the outcome of different ethical rationales). In such research, it will be important to follow the scientific development of the planetary boundaries concept.

To conclude, it is apparent that many aspects of LCA work in the development of forest products deserve further research. Meanwhile, it is important to acknowledge that LCA is not the only tool for dealing with environmental concerns in the R&D of forest products. A good reminder of this is the words of Maslow (1966, p. 15): "I suppose it is tempting, if the only tool you have is a hammer, to treat everything as if it were a nail”. Relatedly, it is important to remember that there are other dimensions of sustainability than those covered by environmental assessment tools. For example, human rights, working conditions, equality and other social sustainability concerns are better dealt with by tools such as Social LCA (Benoit & Mazijn 2009).

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