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Improved environmental assessment in the development of wood-based products

Capturing impacts of forestry and uncertainties of future product systems

GUSTAV SANDIN

Chemical Environmental Science
CHALMERS UNIVERSITY OF TECHNOLOGY
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Chemical Environmental Science
Department of Chemical and Biological Engineering
Chalmers University of Technology
SE-412 96 Gothenburg
Sweden
Telephone + 46 (0)31-772 1000
www.chalmers.se

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Gustav Sandin, Chemical Environmental Science, Department of Chemical and Biological Engineering, Chalmers University of Technology, Sweden

Abstract

The prospect of reducing environmental impacts is a key driver for the development of new wood-based products. But as wood-based products are not necessarily environmentally superior to non-wood alternatives, there is a need to assess the environmental impact of the product and guide the development process. The aim of this research is to improve the methodology of such environmental assessments, to better capture the inherent uncertainties of future, still non-existent product systems and to improve the impact assessment of forestry.

For capturing uncertainties, two approaches for scenario modelling were used in life cycle assessments (LCAs) of wood-based roof constructions and textile fibres. In the first approach, scenarios were set up to explore how different future technologies and methodological approaches (consequential and attributional) influence the assessment of life cycle processes occurring in a distant and uncertain future. In the second approach, scenarios with different geographical locations for the life cycle processes were generated by varying the future demand for textile fibres and the competition for forest land. Both approaches generated results which differed significantly between the scenarios; thus the approaches enabled a more comprehensive assessment than if only one scenario had been set up. The approaches can be recommended particularly for assessments of long-lived products and products with globally distributed supply chains.

For improving the impact assessment of forestry, methods suggested in the literature were used and further developed in an LCA of wood-based textile fibres. The methods captured the land use impact on biodiversity and the water use impact on human health, ecosystem quality and resources. A new inventory approach was developed to better capture the system-scale effects that forestry can have on the hydrological cycle. Besides identifying opportunities for further methodological improvements, the methods generated meaningful results beyond what is offered by established methods for impact assessment. In particular, the consequential inventory approach made it possible to discern that land use can contribute positively to downstream water availability under certain conditions.

Keywords: product development, environmental assessment, sustainability assessment, land use, water use, impact assessment, end-of-life modelling, scenario modelling, wood, forestry

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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. Manuscripts of the papers are appended at the end of the thesis.

I. Sandin G, Peters GM, Svanström M (2013) Life cycle assessment of construction materials: The influence of assumptions in end-of-life modelling. Submitted to the International Journal of Life Cycle Assessment

II. 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. Accepted for publication in Resources, Conservation and Recycling

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

A. Sandin G, Peters GM, Pilgård A, Svanström M, Westin M (2011) Sustainability assessment in product development: Experiences from two projects. Poster contribution to the Alliance for Global Sustainability Annual Meeting 2011, Gothenburg, Sweden

B. 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

C. Sandin G, Peters GM, Svanström M (2011) Location dependency of the sustainability of textile fibres. Poster contribution to the Avancell Conference, Gothenburg, Sweden

D. 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. Oral presentation and full paper in conference proceedings of the 8th International PRA Woodcoatings Congress, Amsterdam, the Netherlands

E. Sandin G, Clancy G, Heimersson S, Peters GM, Svanström M, ten Hoeve M (2012) Clarifying the role of life cycle assessment in technical research and development projects: Recommendations for project planning. Oral presentation at the 18th SETAC LCA Case Study Symposium, Copenhagen, Denmark

Table of contents

1 Introduction

Society faces a wide range of challenges related to the degradation of the Earth's natural capital, including major impacts on climate (IPCC 2007) and biodiversity (Millennium Ecosystem Assessment 2005). These challenges have been summarised by the "planetary boundaries" concept, which suggests nine biophysical boundaries that are intrinsic for the Earth system and important for it to function, of which at least three (rate of biodiversity loss, climate change and interference with nutrient cycles) have already been surpassed due to anthropogenic pressures (Rockström et al. 2009). This 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 provide for our demand from nature. Humankind's ecological footprint is currently estimated to be about 50% larger than the area of the Earth (Global Footprint Network 2012).

The consumption of products is a main driver for this environmental degradation and there is wide international agreement that development of environmentally improved products is important for lowering the degradation (UN 2012). This is, however, a grand challenge, formalised for example in bold targets of decreasing the resource intensity per provided service unit (i.e. the eco-efficiency) in industrial sectors or countries by a factor of 4, 10, 20 or even 50 (Reijnders 1998). The challenge 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). The joint endeavour for reduced environmental impact and improved human well-being has been termed "sustainable development", famously defined as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987), and generally agreed upon to require consideration of environmental, social and economic aspects (UN 2012). Regardless of how much more eco-efficient the average product of tomorrow must be in order to reduce humankind's ecological footprint and enable us to stay safely within the planetary boundaries, and thereby contribute to sustainable development, the message is clear: the environmental impact of products must be considerably reduced.

Without an assessment of the sustainability of products under development, many efforts may, however, be misdirected or even counter-productive. Therefore, there is a need for tools that can be applied in the development process in order to accurately estimate the most relevant sustainability impacts of the developed product and guide the development towards an outcome that leads to improved sustainability. Sustainability assessments carried out early in the development of products are 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 once the product has been commercialised (McAloone and Bey 2009; Yang and Shi 2000; Steen 1999).

The topic of my research is methodological advancements of sustainability assessment applied in early stages of product development. This thesis focuses particularly on the assessment of environmental aspects of sustainability. Hereafter, this is called "environmental assessment" for sake of simplicity, although "environmental sustainability assessment" may be a more correct term, as the assessment attempts not only to measure the environmental impact but also to adopt the long-term perspective emphasised by the sustainability concept (WCED 1987).

The research is based on environmental assessments carried out as part of wider sustainability assessments in two particular development projects: the WoodLife and the CelluNova projects (SP 2013; EcoBuild 2010). The WoodLife project aims at developing new coatings and adhesives for wood-based construction products. The CelluNova project aims at developing a new process for the production of a wood-based textile fibre. An important driver for both projects is the prospect to increase the use of wood-based products in society on the expense of non-wood alternatives, and thereby mitigate environmental impacts. This prospect is based on some environmentally favourable properties that wood has compared to many other feedstocks. For example, wood is biodegradable and, if derived from well-managed forestry, renewable and potentially climate neutral. However, as will be elaborated on in this thesis, wood-based products do not necessarily have a lower environmental impact than non-wood alternatives, which is why environmental assessments are needed.

1.1 Research questions

There are many challenges involved in carrying out environmental assessments in the two projects, many of which are related to the use of life cycle assessment (LCA), the primary assessment tool used in the projects. This thesis addresses some of these challenges, primarily those dealt with in papers I and II (see list of publications, page vii). The aim of the thesis is to answer the following research questions:

1. How can the inherent uncertainties of future, still non-existent product systems be captured in environmental assessments?

Here, the focus has been on better capturing uncertainties of the type of technology that can be expected in end-of-life processes of long-lived products, and uncertainties of the geographical location of production processes, which is expected to significantly influence the environmental impacts of forestry. In doing this, it has also been explored how different methodological approaches (attributional and consequential) influence the modelling of end-of-life processes and hence the LCA results.

2. How can the impact assessment of forestry be improved?

Here, we have moved down the cause-effect chain of land and water use impacts – two potentially important environmental impacts of wood-based products – in an attempt to assess these impacts more accurately, beyond what is offered by today's established methods for impact assessment in LCA. Apart from testing methods suggested by others, a consequential approach for the inventory analysis of the forestry's water use has been developed and tested.

1.2 Overall methodological approach

In addressing these research questions, the overall methodological approach has been to look for methods available in the literature, select appropriate methods, when necessary develop them further, and apply them in the WoodLife and CelluNova projects. Then, there has been reflections on difficulties encountered in the process and how the methods can be developed further to, for instance, make them more useful in projects aimed at developing wood-based products.

1.3 Guide for readers

Chapter 2 gives a comprehensive background of the context and methods of the research, and the environmental assessment aspects of particular importance for the above listed research questions. Chapter 3 includes summarises of papers I and II and a discussion on how the results of the papers contribute to finding answers to the research questions. Chapter 4 summarises how the research has contributed to answering the research questions, and chapter 5 lists future research needs, including the particular direction of my own research.

2 Background and methods

This chapter describes the strength and weaknesses of wood-based products, thereby outlining key drivers for the development, and difficulties in the environmental assessment, of such products. Then the chapter describes the WoodLife and CelluNova projects, reviews methodologies of sustainability assessment and gives an elaborate background of methodological aspects of particular importance for the research questions of this thesis.

2.1 Strengths and weaknesses of wood-based products

As mentioned, the prospect of reducing environmental impacts is a common driver for projects aimed at developing wood-based products. It has been shown that wood-based products in general tend to have favourable environmental performance compared to non-wood alternatives (Werner and Richter 2007), but the use of wood as a feedstock is no guarantee that the end product is environmentally superior to non-wood alternatives.

2.1.1 Renewability

Wood's potential renewability is an often recognised advantage of wood compared to, for example, mineral-based materials, which may be subject to scarcity (see, e.g., Owen et al. (2010) for an introduction to the "peak oil" debate). For wood 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)[1], whereas the stocks in tropical rain forests are decreasing (Forster et al. 2007). However, in some specific temperate and boreal regions, the biomass stocks may be decreasing, and in some specific tropical regions there may be constant or growing stocks of biomass. Thus, whether wood from boreal, temporal or tropical regions can be seen as renewable depends both on the specific geographical location of the forestry and the study's resolution and definition of renewability.

The temporal perspective of the study also matters. For example, the biomass stocks of temperate and boreal forests may not increase in the future, when the

[1] In Swedish forests, the biomass stock doubled in 1926-2008 (SLU 2011).

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 the increase may not continue once the historical biomass stocks have been achieved. 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 in the future result in declining boreal biomass stocks (Kane and Vogel 2009; Kurz et al. 2008). Furthermore, expected increases in demand may considerably increase the withdrawal of forest biomass and 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 energy use mix by 2020 (EU 2007) may be a threat towards the long-term biomass growth of European forests (Mantau et al. 2010; Rettenmeier et al. 2010; Nabuurs et al. 2007).

Whether to define wood as renewable or not also depends on how to view indirect land use change (iLUC), which does 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 the feedstock to a product, competition for land increases compared to if the product would not have been produced, which may result in higher commodity prices and therefore increased 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 (Searchinger et al. 2008). How to view iLUC depends on, e.g., whether consequential or attributional assessment approaches are applied (these concepts are explained in section 2.3.2), where consequential approaches strive to capture market mechanisms such as iLUC. The choice of approach may also influence the spatial and temporal system boundaries that were discussed in previous paragraphs.

To summarise, whether wood can be viewed as renewable or not depends on the location of the forestry, the spatial and temporal scope of the assessment and on other methodological choices of the assessment.

2.1.2 Climate change impact

Perhaps the most emphasised environmental benefit of wood-based products concerns their climate change impact. It is commonly claimed that wood-based and other bio-based products are climate neutral. However, such claims rely on premises that have been questioned (see, e.g., Searchinger et al. 2008). To understand this debate, there is a need to introduce the basics of how to account for the climate change impact of bio-based products.

By far, the most common metric for climate change impact is the global warming potential (GWP), which is used both in LCAs and for national emissions reporting, e.g. under the Kyoto protocol – the international environmental treaty regulating greenhouse gas (GHG) emissions (UNFCCC 2013). GWP is based on how much a GHG emission 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 (Forster et al. 2007). As different GHGs have different atmospheric residence times, the chosen time period influences the relative impact of a GHG. Both in LCA and in national emission reporting, it is common to use a time period of 100 years (indicated by a subscript: GWP_{100}). This means that, counted from the moment of the emission of the GHG, any climate change impact of that emission occurring after more than 100 years is fully disregarded.

When harvesting a tree, the carbon stock in the forest decreases. If the wood is used as a feedstock for a product, the carbon is temporarily stored in the product, and if the product is incinerated at the end of its service life, the carbon (mainly in the form of carbon dioxide) is released to the atmosphere. As the forest regrows, carbon is drawn from the atmosphere and once again sequestered as biomass. If the regrown forest contains just as much carbon as the initial forest did (i.e., if the wood is actually renewed), the initially harvested wood is often seen as being carbon and climate neutral. This is the rationale behind why biogenic carbon dioxide emissions are often disregarded when calculating the climate change impact of wood-based and other bio-based products in LCA, and why bioenergy is considered carbon neutral under the Kyoto protocol (Brandão et al. 2013). It has even been shown that a well-managed forest can function as a carbon sink; thus, it is argued, mitigating climate change impacts (Perez-Garcia et al. 2005; Liski et al. 2003).

The critique of bio-based products' claim of carbon or climate neutrality often concerns the handling of the temporal dimension: (i) the time horizon of the characterisation method, (ii) the time period over which a product system's GHG emissions and removals are taken into account, and (iii) the time period of the product life cycle (Brandão et al. 2013). For example, it has been argued that if a bio-based product is combusted long before the plantation or forest has been fully regrown (which is usually the case for biofuels and other short-lived products), the product temporarily contributes to an increase in radiative forcing and hence climate change. In such a case, the product can be seen as carbon neutral over the forest rotation period, but hardly climate neutral (Helin et al. 2012). On the other hand, one may argue that if the product stores carbon for a long time (which may be the case for the construction products developed in the WoodLife project) and carbon in the meantime is sequestered in the regrowing forest, the product temporarily contributes to a decreased radiative forcing and thus mitigates climate change. Furthermore, the usual way of using GWP in LCA has been criticised for disregarding when an emission takes place, thereby not applying the chosen time horizon consistently (Brandão et al. 2013). Consider an LCA where the GWP_{100} is calculated for a product life cycle where some emissions occur today and others occur in the future (e.g. during the disposal of the product). Then, impact of the emissions occurring today is counted until 100 years from now, but the impact of the future emissions are counted until 100 years from the day they occur. Thus the 100 year time horizon is not used consistently: some emissions that occur after 100 years from today are considered, while others are not. There have been suggestions of using dynamic characterisation factors in LCA, in which emissions occurring closer to the end of the set time horizon are given less weight (Levasseur et al. 2010). The ILCD Handbook also permits the discounting of future, or delayed, emissions, but only in studies which specifically aim at assessing the effects of delayed emissions on the overall results (European Commission 2010). These types of dynamic approaches could particularly change results of assessments of long-lived products, for which emissions in the disposal stage occur in a distant future. In paper I, the climate change impact of a category of long-lived product, roof constructions, was calculated using the standard GWP_{100} metric. In section 3.1, there is a discussion on how the scenario modelling approach used in paper I could facilitate the use of more dynamic approaches for the impact assessment of climate change.

Furthermore, if land transformation (e.g. deforestation) occurs as a result of the studied product system, it may contribute to climate change by reducing the carbon stored by the land (as vegetation, as litter and in the soil) and by altering the land surface's capacity to reflect solar radiation, i.e. its albedo (Forster et al. 2007). Whether wood is considered to contribute to deforestation depends also, just as its renewability, on how iLUC is accounted for, i.e. it depends on the scope of the assessment and on methodological choices. Also, the purpose of the forestry may influence the amount of carbon being stored in the forest. For example, using wood as biofuel implies shorter rotations in forestry, resulting in less carbon storage (Ciais et al. 2008).

Claims of bio-based products being net carbon sinks may also rely on assumptions on their end-of-life treatment: that the wood is incinerated and the generated energy is used to produce electricity and/or heat which substitute alternative (often fossil) sources of energy, thereby resulting in avoided emissions of fossil carbon dioxide. The importance of end-of-life assumptions is further discussed in section 3.1 and in paper I.

To summarise, just as for its renewability, the climate change impact of wood-based products depends on the long-term management of the forest and the methodology used in the assessment. The current standard practices for measuring the climate change impact of wood-based products support claims of climate neutrality, but there is an on-going debate on whether these methods reflect realities. See, e.g., Brandão et al. (2013) and Helin et al. (2012) for more elaborate discussions on how to assess the climate change impact of bio-based products and reviews of alternatives to the GWP metric.

2.1.3 Biodegradability

Another often recognised benefit of wood is its biodegradability, which means that it will normally not accumulate in nature once it has become a waste material, as some other materials will (see Derraik (2002) for a review of plastics debris in the marine environment). In the disposal stage of wood-based products, this is often seen as an environmental benefit, although it may not always be a benefit. When wood waste degrades, for instance in landfills, part of the carbon is emitted to the atmosphere as methane, a potent GHG. Globally, methane emissions from landfills may constitute up to 20% of all anthropogenic methane emissions and 4% of all anthropogenic GHG emissions (Frøiland Jensen and Pipatti 2002).

The biodegradability may also be problematic in the use phase of wood-based products, and they may therefore require more preservatives, surface treatments and maintenance to meet the same service-life performance as non-wood alternatives (which is an issue addressed in the WoodLife project, see section 2.2.1). The biodegradability may even make wood-based materials unsuitable for some products, such as containers for certain foodstuff.

2.1.4 Land use impacts

A potential environmental problem of wood is that it is land use intensive compared to many abiotic materials. Apart from potential problems with renewability and climate change, as discussed above, poor land use management can result in a range of other disturbances to human health, ecosystem quality and resources. Due to looming land scarcity (Lambin and Meyfroidt 2011), environmental consequences of land use will probably increasingly gain attention, also in countries that seemingly have an abundance of land, such as Sweden. Land use impacts are addressed by research question 2 of this thesis and discussed further in section 2.5 and paper II.

2.1.5 Factors not related to the main feedstock

The main feedstock of a product is not the only factor determining its environmental impact. For example, in the production and maintenance of wood-based products, many non-wood materials may be used, sometimes even more (in mass) than used in the production of alternative non-wood products. Besides, as previously discussed, wood often requires chemical treatments to withstand weathering and degradation, which may lead to the exposure of toxic compounds to humans and ecosystems (Werner and Richter 2007). The amount and type of energy used in the life cycle are also key factors determining a product's environmental impact; factors which are normally rather independent of the main feedstock of the product.

To conclude, the fact that a product is wood-based is no guarantee that it is environmentally beneficial compared to non-wood alternatives. Many aspects need to be taken into account if we want to ensure that wood-based products that replace non-wood alternatives indeed contribute to reduced environmental impact. A number of these aspects are further discussed later on in this thesis.

2.2 Projects

2.2.1 WoodLife

The objective of the ongoing WoodLife project is 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 can potentially widen wooden materials' scope of application, for example allowing wood to replace more energy intensive materials or materials of non-renewable origin (e.g. aluminium or PVC in window frames, respectively) and thereby reduce the environmental impact of construction products. Improved coatings are to be achieved by the inclusion of metal oxide nanoparticles (particles with diameters of 1-100 nm) that absorb light with wavelengths in the UV range (250-440 nm), thereby protecting 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 are to be developed by designing silica and clay nanoparticles with surface properties that make them compatible with adhesive binders. Introducing nanoparticles can 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 gluelaminated (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 (US Department of Health and Human Services 2011). Fig. 1 illustrates the project idea.

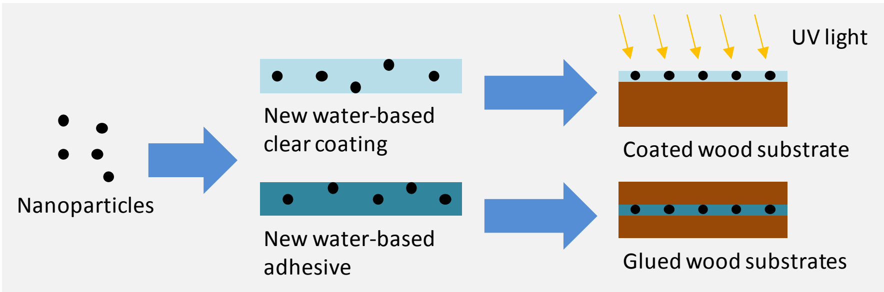

Fig. 1 Visualisation of the WoodLife project idea. The idea is to add nanoparticles and thereby improve the UV-protecting properties of clear coatings, and the strength of adhesives, for wood applications.

The project spans over 3 years, is funded from both private and public (the European Seventh Framework Programme) sources and involves 11 participating organisations, including universities, research institutes and private companies. Project work is divided into nine work packages, covering 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 environmental assessment) of the developed technologies, and technology demonstration, validation and exploitation.

2.2.2 CelluNova

The recently ended CelluNova project aimed at developing a new process for dissolution and spinning of wood pulp into a textile fibre. Such regenerated cellulose fibres already exist on the market (e.g. viscose fibres), but the CelluNova project aimed at developing an environmentally preferable process, producing fibres that can be blended with cotton fibres into a textile material with cotton-like qualities (as illustrated in Fig. 2). This can lessen the textile industry's dependence on cotton – a fibre associated with large use of pesticides, fertilisers and water (Chapagain et al. 2006; WWF 2003). The relatively low environmental impact was to be achieved by integrating the process into a pulp mill, e.g. by using chemicals well-known to the pulp mill operators and utilising energy generated as a by-product in the pulping process.

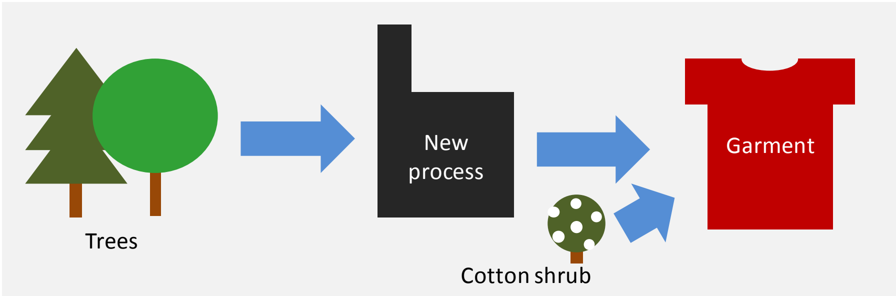

Fig. 2 Visualisation of the CelluNova project idea. The idea was to develop a new process that can turn wood pulp into a textile fibre, which can be blended with cotton fibres into a textile of cotton-like quality.

The project spanned over 3 years, was funded by both private and public (among others, by the Swedish Governmental Agency VINNOVA) sources and involved 14 participating organisations, including universities, research institutes and private companies. Project work was divided into five work packages, focussed on dissolution of cellulose, spinning of fibres, textile manufacturing and testing, full-scale modelling of the process, and sustainability assessment (including environmental assessment) of the fibres. The project is continued in the VINNOVA-funded ForTex project, in which the aim is to prepare for building a pilot plant for the developed process.

2.3 Sustainability assessment

It is increasingly common to conduct sustainability or environmental assessments in product development[2]. In publicly funded development projects, this is often a demand from the project commissioner. For example, projects funded by the European Seventh Framework Programme are often required to include an LCA of the technology under development (Tilche and Galatola 2008). Therefore, there is currently intense use and development of methodologies for sustainability assessment of products.

[2] The inclusion of environmental considerations in product development is often referred to as “ecodesign”, “design for the environment” or “environmental product development”; to some extent, these concepts refer to other phases and contexts of product development than discussed in this thesis.

2.3.1 Sustainability assessment methodology

Life cycle sustainability assessment (LCSA) is a framework for life-cycle based methods for assessing environmental, social and economic aspects of sustainability (Ciroth et al. 2011). The framework advocates the use of LCA for environmental assessments, social LCA (SLCA) for social assessments and life cycle costing (LCC) for economic assessments, and it provides advice on how to make methodological choices to facilitate the integration of these three methods. The framework builds on previous research and initiatives, such as the CALCAS project (Zamagni et al. 2009) and the WE-LCA technique (Poulsen and Jensen 2004).

Another, similar product sustainability assessment approach is PROSA (Grießhammer et al. 2007), which also advocates the use of LCA, SLCA and LCC. In addition, PROSA provides a framework for using a range of other tools for, e.g., stakeholder involvement, brainstorming, decision-making, market and consumer analysis, and for interpreting the results in terms of strategic development. PROSA appears to be intended primarily for sustainability work within a company, and for integration of LCA and related methods with long-term strategic development of product portfolios. As such it can be useful in many product development processes, all the way from generating ideas to commercialisation, but it is less useful in intra-organisational and time-limited projects such as the WoodLife and CelluNova projects.

The chemical company BASF has developed a socio-eco-efficiency analysis tool called SEEbalance (Schmidt et al. 2004), which enables the comparison of product alternatives in terms of six LCA-based environmental impact categories, LCC-based cost metrics and social sustainability parameters on working conditions and job creation. SEEbalance was perhaps the first example of a workable sustainability assessment tool covering environmental, economic and social aspects. However, it was not designed for the prospective assessment of future, still non-existent product systems, and it is not transparent enough to be a viable alternative for studies of non-BASF products or for studies aimed at testing and developing new methods for impact assessment (which is central to the aim of my research; see research questions, section 1.1).

A tool particularly designed for the sustainability assessment of forest product chains is ToSIA (EFORWOOD 2010). The tool is freely available computer software, although it is recommended that a consultant from the EFORWOOD consortium assists in conducting analyses. The software supports the calculation of 26 predefined sustainability indicators (10 environmental, eight economic and eight social ones) and the use of cost-benefit and multi-criteria analysis. Also, the tool supports the building of future what-if scenarios (for more on this concept, see section 2.6), to be used for, e.g., improving forest product chains or testing different designs of future forest products. However, it is not intended for the assessment of other product categories; consequently it does not support the type of comparisons of wood- and non-wood products done in the WoodLife and CelluNova projects. Also, the tool does not support the use or development of other impact categories or impact assessment methods, as the choice of impact categories and impact assessment methodology are predefined.

There are many other examples of ready-made sustainability assessment methods and/or suggestions on how to integrate different methods (often LCA) into product development (e.g. Askham et al. 2012; Clancy 2012; Devanathan et al. 2010; Manmek et al. 2010; Othman et al 2010; Vinodh and Rathod 2010; Colodel et al. 2009; Byggeth 2007; Waage 2007; Ny 2006; Rebitzer 2005; Fleischer et al. 2001). Often, these are screening or simplified methods particularly designed for the assessment of preliminary product designs (see Rebitzer (2005) for a review of screening methods). What most tools have in common is the emphasis of a range of different sustainability criteria and the recognition of the need for some type of multi-criteria decision analysis for handling potential trade-offs between different sustainability dimensions and/or impact categories. However, most tools are primarily intended for assessments carried out in rather specific contexts (e.g. for certain product categories) and focussed on inter-organisational product development (i.e. carried out within a company). Consequently, most of them are of limited use in the WoodLife and CelluNova projects.

In my research, LCSA was chosen as the foundation for the sustainability assessment, for a number of reasons: it is holistic (i.e. supports consideration of the whole life-cycle of a product and numerous impact categories), it is subject to extensive development through a broad and active research community (2010-2020 has even been termed the “decade of life cycle sustainability assessment” in LCA research (Guinée et al. 2011)), it is generic and thus applicable for any type of product category, it is transparent which enhances its credibility and gives the assessment practitioner full control over the methodology, and it does not depend on a specific software that runs the risk of being rarely updated or even abandoned.

2.3.2 Life cycle assessment methodology

As this thesis focuses on the environmental dimension of sustainability, the research is primarily based on the use and development of LCA, the environmental assessment method recommended in the LCSA framework. 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 (Frischknecht et al. 2009). Still, LCA may not always be sufficient for assessing all the relevant environmental impacts of a product, and other assessment tools may also be needed. For example, in the WoodLife project, a toxicological evaluation (including a literature study and ecotoxicological testing) also had to be carried out in order to evaluate the possible toxicological risks of the nanoparticles (as reported in publication D; see list of publications, page vii).

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

I. *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 comparisons of 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.

II. *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 or 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).

III. *Life cycle impact assessment (LCIA)*: By means of characterisation models, the LCI data is translated into potential environmental interventions, classified into impact categories. Traditionally, the focus has been on environmental interventions from emissions and on global and regional effects, such as climate change, stratospheric ozone depletion and eutrophication. Sometimes, LCA covers more geographically dependent impacts as well, such as ecotoxicity and human toxicity, although there are large uncertainties in the modelling of such impacts as they are highly dependent on local characteristics (e.g. local flora and fauna, soil structure and presence of other chemicals) which are difficult to consider in an LCA. This thesis addresses the challenge of assessing two impact categories which are not driven by emissions and which are highly dependent on local characteristics: land and water use impacts (see section 2.5 and paper II).

Impact categories can be measured by midpoint or endpoint indicators. Midpoint indicators reflect links in the cause-effect chain of environmental impacts, whereas endpoint indicators are metrics of the actual end impact. 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 be 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.

The LCIA can also include normalisation and weighting, in which results for several impact categories are aggregated on a single yardstick (e.g. by using value-based judgements based on the opinions of a panel of experts). Aggregation of several midpoint impact categories is also provided by end-point characterisation models (see, e.g., the ReCiPe framework (Goedkoop et al. 2012)).

IV. *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) or contribution analysis (in which the contribution of different life cycle processes are analysed).

2.3.3 Methodological frontiers of life cycle assessment

Three issues in LCA methodology of particular interest for this thesis deserve a further introduction: whether to use attributional or consequential LCA approaches, what type of LCI data to use and how to handle multifunctional processes.

The consequential-attributional controversy is a current topic of debate in the LCA research community (Earles and Halog 2011). Traditionally, LCA has relied on an attributional approach, which means that the LCA only considers emissions and resource use that take place at the locations of the life cycle processes, i.e. it only accounts for impacts physically connected to the studied product system. This can also be described as the average impact of the studied production system per provided functional unit. A consequential (also called change-oriented) approach, on the other hand, seeks to map the consequences of a decision. This can also be described as the consequences of a change in production output, i.e. what is the environmental consequence if more, or less, functional units are provided. A consequential approach entails inclusion of system-scale effects not necessarily physically connected to the product system, but occurring due to, e.g., market mechanisms (Earles and Halog 2011). Section 2.1 described one such market mechanism: iLUC. The choice between an attributional and a consequential approach determines, for example, what LCI data to use and how to handle multifunctional processes (see the following paragraphs). Later in this thesis, there are several examples of how consequential and attributional approaches lead to different LCA results. See Zamagni et al. (2012) for a 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, i.e. emission data on the marginal source for electricity. 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, i.e. the technology that is expected to respond to a change in demand (Lund et al. 2010).

Typically, average data is used for attributional studies, and marginal data for consequential studies. To use 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 may be 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 LCA results. It can, however, be difficult to determine the marginal technology. 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). Also, some authors have suggested that in markets constrained by regulation, the marginal technology should be defined as the planned or predicted technology rather than the uninstalled technology with the lowest marginal cost (Schmidt et al. 2011). Thus, the choice between, and the selection of, average or marginal LCI data is a much discussed LCA controversy.

Another question which can significantly influence the results of an LCA is how to handle processes with several functions. For example, many waste incineration processes are multifunctional: they treat the waste of a product and produce heat and/or electricity. Another example is container ships, which transport many different products in one shipping. This raises the question of how to allocate the environmental impact of such processes between the many functions. The allocation can be based on some physical property (the mass, the volume or the energy content) or the monetary value of the produced functions (ISO 2006b). In the example of the container ship, the emissions of the ship could be allocated to the shipped products based on their mass (this is particularly reasonable if mass is the limiting factor for how much the ship can carry). The allocation can also be avoided by system expansion, sometimes called substitution (ISO 2006b). For example, presume that we are studying a product that is incinerated at the end of its service life, and that the incineration process generates heat to a district heating system. System expansion implies that the marginal technology for heat generation in the district heating system is identified, and that this technology is assumed to be substituted as a consequence of the studied product system, thereby resulting in avoided emissions. These emissions are then subtracted from the LCI data of the studied system, i.e. the system is given credit for the avoided emissions. With this procedure, allocation of the environmental impact of the multifunctional waste incineration process is avoided. Typically, some type of allocation is applied in attributional studies and allocation avoidance by system expansion is applied in consequential studies.

2.4 End-of-life modelling of construction products

As previously mentioned, the WoodLife project aims at developing 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). Due to technological change, the nature of such processes is highly uncertain (Frischknecht et al. 2009). This time-dependent uncertainty has previously been acknowledged as a challenge typical for LCAs in the construction industry (Singh et al. 2011; Verbeeck and Hens 2007). Nevertheless, this uncertainty is often neglected in LCAs of construction products, and end-of-life practices of today are assumed without any explicit explanation, even when the aim is to support decisions concerning contemporary or future construction products that are expected to stand for a long time (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) set up scenarios to account for different possible future means of end-of-life treatment of a bridge.

To consider end-of-life uncertainties is especially important when end-of-life practices may significantly influence the environmental impact. For buildings, efficient recycling at the disposal stage may save energy that corresponds to 29% of the energy use in manufacturing and transportation of the construction materials (Blengini 2009). Moreover, energy savings from efficient recycling may correspond to 15% of the total energy use of a building's life cycle (Thormark 2002). Although a building's use phase is often said to contribute 80-90% of its environmental impact (Cuéllar-Franca and Azapagic 2012; Ortiz et al. 2010), the relative importance of end-of-life processes is now rising due to increasingly energy-efficient buildings (Dixit et al. 2012); it has even been argued that poorly defined functional units often lead to exaggerated data on energy usage in the use phase (Frijia et al 2011). The environmental impact of the waste handling of construction materials is also considered significant simply because of the sheer amount of such materials existing in society (Bribián et al. 2011; Singh et al. 2011; Blengini 2009).

So there are strong reasons to improve the modelling of end-of-life processes in environmental assessments of construction products. This can contribute to more robust decision-making in the development of the construction products of tomorrow, e.g. in contexts such as the WoodLife project.

How to improve the end-of-life modelling of construction products was addressed in paper I, where we tested how assumptions of the modelling of end-of-life processes influence LCA comparisons of alternative internal roof constructions (glulam beams and steel frames) for an industrial hall, and thereby tested the robustness of the result from the comparative assessment. Tested assumptions relate to the technology used for end-of-life processes and the use of attributional or consequential end-of-life modelling approaches. Section 3.3 includes a summary of paper I and a discussion on its implications for research question 1 of this thesis.

2.5 Land and water use impact assessment

As shown in section 2.1, some of the environmental impacts of wood-based products are difficult to address in LCAs with the currently established methods for impact assessment. In particular, impacts related to the land and water use of forestry are seldom addressed in a satisfactory manner, neither at an LCI or LCIA level, and there is a need for improved methods for assessing such impacts. To contribute to this research, the CelluNova project was used as a case study in an attempt to apply and develop impact assessment methods suggested in the LCA research community, as is reported on in paper II. Sections 2.5.1 and 2.5.2 provide a review of methods for land and water use impact assessment and brief descriptions of the used methods (see paper II for detailed descriptions).

2.5.1 Land use impact assessment

There are many proposed methods for land use impact assessment, as reviewed by, e.g., Curran et al. (2011). The development of new methods are to some extent driven by increased availability of databases on land use and land cover, such as the CORINE database on European land cover (EEA 2013) and the GlobCover database on global land cover (ESA 2013).

Among biodiversity indicators, species richness of a certain species is a commonly used indicator. Primarily, the focus has been on the species richness of vascular plants (Schmidt 2008; Goedkoop and Spriensma 2000; Köllner 2000; Lindeijer 2000), but there have been proposals to consider other species, alone or in combination, such as the richness, abundance and evenness of vertebrate species (Geyer et al. 2010), the species richness of vascular plants, molluscs, mosses and threatened species (Köllner and Schulz 2008) and the species richness of vascular plants, birds, mammals and butterflies (Mattsson et al. 2000). De Baan et al. (2012) proposed a method using relative species richness as indicator, which allows the use of data on any group or groups of species that species richness data exist for.

Other methods exist for land use impact assessment which measure other facets of biodiversity or other ecosystem attributes (which indirectly may influence biodiversity), sometimes in combination with species richness indicators. Examples include the measurement of net primary productivity (Weidema and Lindeijer 2001; Lindeijer 2000), biotic production potential (Brandão and Milà i Canals 2012), naturalness as defined by 11 qualitatively described classes (Brentrup et al. 2002), soil quality (Milà i Canals et al. 2007; Mattsson et al. 2000), number of red-listed species in combination with several biotope-specific key features (Kylärkorpi et al. 2005), or 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 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 one index; for example, Muys and Quijano (2002) proposed a metric of the exergy of ecosystems in which 18 indicators are combined.

In the study reported in paper II, the method by Schmidt (2008) was used for assessing the land use impact on biodiversity. Biodiversity was chosen to be studied as biodiversity loss is a particularly urgent environmental problem according to the planetary boundaries concept (Rockström et al. 2009) and land use is one of the main drivers of biodiversity loss (Millennium Ecosystem Assessment 2005). The method uses the species richness of vascular plants as a proxy for biodiversity and distinguishes between impact due to transformational and occupational land use. Transformational impact occurs when the land is changed from type of land to another and occupational impact occurs when the land is in use. The method enables the calculation of characterisation factors that depend on the geographical location of land use. This calculation is based on the altitude and latitude of the location of land use and the intensity of land use in the surrounding region – factors which can be expected to influence the vulnerability of ecosystems towards environmental interventions such as occupation and transformation of land.

The possibility to calculate characterisation factors that depend on geographical factors was desirable in the CelluNova project, as we wanted to identify whether or not the geographical location of operations would be an important factor in determining the environmental impact of a future textile fibre in relation to cotton fibres (land use in Sweden, Russia, China and Indonesia were compared). Most of the other proposed methods have been developed for the assessment of land use in specific regions, and, due to a lack of data, they are most often not yet applicable in assessments of globally distributed supply chains. 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. See paper II for a further discussion of the chosen method.

2.5.2 Water use impact assessment

Just as for land use impact assessment, the development of methods for the impact assessment of water use is in a phase of intense development and many methods are being proposed, as showed in the reviews by Kounina et al. (2012) and Berger and Finkbeiner (2010). The simplest of impact assessment methods do not characterise the impact of water use further than at the inventory level (e.g. Goedkoop et al. 2012), which can be a suitable in studies aimed at comparing the water demand of generic product systems. However, case studies show that an inventory metric may not correlate well with measures of water use impact (Ridoutt, 2011). Therefore, in studies such as the one reported in paper II – which aims at exploring the influence of the location of operations – there is a need to assess water use impacts further down the cause-effect chain. In doing this, two main difficulties arise: what volume of water to consider in the LCI 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, e.g. by irrigation systems, and to disregard naturally supplied water, e.g. 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 water availability downstream and thus have environmental consequences (e.g. Hoekstra and Chapagain 2007). However, LCI methods for water use generally disregard system-scale effects of land use, e.g. catchment-scale 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, or when land is transformed from one use to another. The consequential LCI approach suggested and tested in paper II is an attempt to capture such factors. 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 how forestry operations in total influence downstream water availability. Changes in runoff has previously been included in LCIs of water use for static agricultural systems (Peters et al. 2010), but never (to our knowledge) for systems with forestry or transformation of land – as was done in our study. The consequential approach was compared to a more traditional, attributional LCI approach, based on the evapotranspirational losses of the harvested trees. Fig. 3 illustrates the difference between the attributional and consequential approaches. The approaches are further described in paper II.

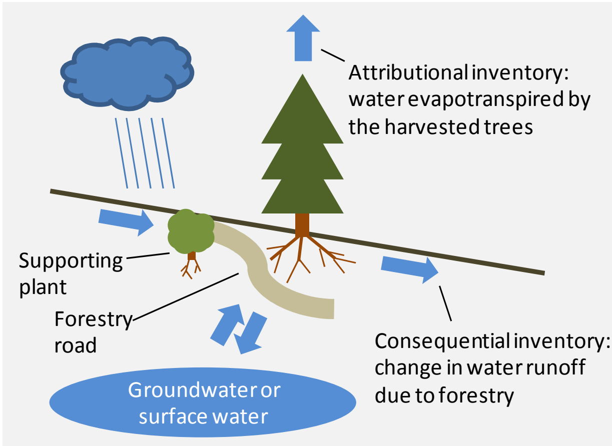

Fig. 3 Schematic view of the water flows in the forestry system. With an attributional LCI approach, the forestry's water use is estimated by the evapotranspirational losses of the trees during their growth. With the consequential LCI approach, water use instead refers to the change in runoff that occurs 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. Bösch et al. (2007) proposed an exergy indicator for resource consumption, including the use of water. Milá i Canals et al. (2008) discussed a number of impact pathways of how freshwater use, and the change of land, 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 of agricultural water scarcity, and of human health impacts from 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 proposed a method for aggregating different types of environmental impacts related to water (e.g. impacts of water use and impacts of water pollution) for which it calculates the water volume necessary to dilute emissions to freshwater to such an extent that the water quality adheres to water quality standards (Hoekstra et al. 2011). Recently, another such single score approach has been suggested, drawing on the latest developments in the LCIA modelling of water related impacts (Ridoutt and Pfister 2013). Most suggestions for LCIA 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 resources (Kounina et al. 2012).

For the study reported in paper II, the method by Pfister et al. (2009) was used as it was deemed the most promising and comprehensive LCIA method available. For example, it captures all the impact pathways recognised by Kounina et al. (2012), as it uses four approaches for characterising the impacts of water use: a midpoint indicator on water deprivation and three endpoint indicators on human health, ecosystem quality and resources. Also, it offers the possibility for end-point characterisation by the Eco-indicator 99 method. Moreover, it was possible to combine with the consequential LCI approach and it could account for regional parameters (e.g. water stress) influencing the impact. Consequently, the method was applicable in the context of our study, in which we wanted to identify the influence of the location of operations. 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 project, for use in the subsequent supply chain design. The method is further described in paper II and by Pfister et al. (2009).

In the study, we set up five possible scenarios of the future product system, and for comparison, two scenarios of cotton product systems with different production sites – as we hypothesised that the location of operations is a key factor influencing these types of impacts. This was an attempt to capture the uncertainty of the location of the future production sites. The management of the uncertainties of future product systems is further discussed in the next section. Section 3.4 includes a summary of paper II and a discussion on its implications for the research questions of this thesis.

2.6 Scenario modelling

In development projects such as the WoodLife and CelluNova projects, little may be known about the future full-scale product system, and the potential environmental impacts of the system may depend largely on factors in the surrounding world which are inherently uncertain (Frischknecht et al. 2009). For example, as was discussed in section 2.1, new political policies may considerably alter the demand for wood, which has implications for many environmental parameters. Therefore, there is a need to construct scenarios of the future surrounding world, and assess how different scenarios influence the product system, in order to develop a product system that is able to contribute to reductions in environmental impact regardless of future world development. The need for scenario modelling in prospective assessments of product systems under development has been recognised before (Hospido et al. 2010; Spielmann et al. 2005). In particular, scenario modelling has been recognised as important for consequential LCAs (Zamagni et al. 2012).

There are various systems for classifying scenarios in LCA. Börjesson et al. (2005) distinguished between predictive (what will happen?), explorative (what can happen?) and normative (how can a specific target be reached?) scenarios. 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 in order to increase the understanding of the studied system. The scenarios in papers I and II can be seen as explorative or cornerstone 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.

There have been previous suggestions for how to generate scenarios in LCA. For example, Spielmann et al. (2005) proposed a method for generating a set of "possible, consistent and diverse cornerstone scenarios representing future developments of an entire LCI product system" (Spielmann et al. 2005, p. 326). The method outlines a series of steps for selecting socio-economic and technological factors that can be expected to influence each process in the studied product system, for analysing how each process can be influenced in LCI terms, and for integrating the influence on each unit process into cornerstone scenarios for the entire product system. Mathiesen et al. (2009) proposed a method for scenario modelling in consequential studies of energy systems, in which a number of different marginal technologies are assumed in order to generate a set of fundamentally different future scenarios.

In paper I, the end-of-life scenarios in the assessment of construction products were generated by assuming different technologies (for disposal practices and for the production of substituted products) to be representative for the future average technologies Additionally, the scenarios tested the influence of using either an 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 I. 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 with technologies with about the same environmental impact as today's technologies, and (ii) a future with technologies with considerably lower environmental impact than today's technologies. If the developed product performs better in all scenarios, it is likely to be a long-term environmentally and commercially attractive alternative. If, on the other hand, there turns out to be small or no 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 (or careful planning of the supply chain design) may be appropriate both for environmental and commercial reasons. As the approach accounts for technological factors that influence the mapping of the product system, including the consequential modelling of end-of-life processes, it has similarities with the methods by Spielmann et al. (2005) and Mathiesen et al. (2009) described above.

Likewise, in paper II, scenarios of the fibre product system were set up. We identified mechanisms of the surrounding world that are expected be important for the geographical location of life cycle processes and the type of land used on the margin – factors of importance for the assessment of land and water use impacts. To generate scenarios, two such 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 was discussed in section 2.5, impact assessment methods were chosen that would allow for this type of location-dependent analysis. The scenarios are further described in paper II. The outcome of this type of scenario modelling can, for example, show whether 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. As this approach explores the future by accounting for market mechanisms, it can be seen as a consequential approach. Also this approach has similarities with the method proposed by Spielmann et al. (2005), as it accounts for how socio-economic factors influence the product system.

In section 3.3, there is a discussion of how the scenario modelling approaches of papers I and II contribute to the research questions of this thesis.

3 Summary of appended papers and discussion of research findings

This chapter contains a discussion on how the presented research contributes to answering the research questions of this thesis, but first there are summaries of the procedures and results of papers I and II (for more details, see appended papers).

3.1 Summary of paper I

Paper I explored how end-of-life assumptions influence the LCA comparisons of two alternative roof constructions: glulam beams 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.

A scenario modelling approach was developed and used to test assumptions regarding the future technologies of end-of-life processes and the use of attributional or consequential approaches in end-of-life modelling. The following elements of the end-of-life processes were tested: energy source in demolition, fuel type used for transportation to the disposal site, means of disposal, and method for handling the allocation problems of the end-of-life modelling. Two assumptions regarding technology development were tested: no development from today's state or that today's low impact technologies have become representative for the average future technologies. For allocating environmental impacts of the waste handling to by-products (heat or recycled materials), an attributional cut-off approach was compared to a consequential substitution approach. A scenario excluding all end-of-life processes was also considered.

In all comparable scenarios, glulam beams showed clear environmental benefits compared to steel frames, except for in a scenario in which steel frames are recycled and today's average steel production is substituted, for which impacts were similar. No particular approach (attributional, consequential or fully disregarding end-of-life processes) was thus beneficial for a certain roof construction alternative. In absolute terms, four factors were shown to be critical for the results: whether end-of-life phases are considered at all, whether recycling or incineration is assumed in the disposal of glulam beams, whether a consequential or attributional approach is used for end-of-life modelling, and whether today's average technology or a low impact technology is assumed for the substituted technology.

3.2 Summary of paper II

Paper II reported on an LCA of land and water use impacts of bio-based textile fibres. LCIA methods suggested in the literature were used. The LCIA method for land use assessed the impact on biodiversity, and the LCIA method for water use considered water deprivation at the midpoint level and the impact on human health, ecosystem quality and resources at the endpoint level. An innovative consequential water use LCI approach was developed and used; this was compared to a more traditional attributional approach.

Five wood-based fibre production scenarios were set up in order to account for uncertainties in the future location of operations and the possible occurrence of land transformation. For comparison, two cotton production scenarios were set up.

The results showed that biodiversity impacts from transformation of natural land were much higher than impacts from occupation of land. If transformation of land takes place, 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 use showed that the location of operations is critical for the results, as water extracted from relatively water-stressed environments leads to higher impacts. Furthermore, for some scenarios, the result differed considerably between the attributional and consequential LCI approaches. Moreover, it was shown that the consequential approach adds the possibility of recognising increased runoff as a potential benefit for certain types of land use.

3.3 Capturing uncertain product system parameters

In papers I and II, we used two different approaches for capturing uncertain product system parameters. The approaches involved setting up scenarios in order to explore possible future states. The approach of paper I was designed particularly for the assessment of long-lived products, such as construction products with expected life lengths of 50-100 years. A temporal dimension was introduced in the mapping of the product system to identify processes occurring in a distant and uncertain future (i.e. end-of-life processes), and then, an uncertain parameter of these future processes (the type of technology utilised) and a key end-of-life modelling parameter (the choice between an attributional and a consequential approach) were varied to generate different possible future states. The approach of paper II instead tested the influence of the geographical location of life cycle processes – an uncertain factor for the type of future textile fibre product system developed in the CelluNova project – as it was hypothesised that this would considerably influence the assessed land and water use impacts. Possible locations were identified by studying how the location and type of marginal land use are expected to depend on the future demand of the fibre and future competition for land.

Although it is likely that the actual impact of the product systems will fall within the range of possible futures represented by the scenarios, it is of course not yet possible to tell whether the created scenarios indeed provide a credible indication of possible future states. For example, in the case of paper I, it will take 50-100 years until we know whether the assumed technologies were reasonable choices, and whether the potential benefits of the wooden roof construction were actually realised. That the outcome of the scenario modelling approaches cannot be empirically endorsed or rejected is of course a limitation of this analysis. Still, it is possible to discuss whether the approaches helped us capture important parameters, by comparing the results of the different scenarios.

In paper I (on construction products), the scenario modelling did significantly influence the LCA results in absolute terms, thus the modelling approach did succeed in capturing parameters of importance for the outcome of the LCA. The approach made it possible to (within an uncertainty range) quantify the potential benefit of a successful WoodLife project, and show under what conditions the environmental benefits of a wooden roof construction appear to be the greatest.

In paper II (on textile fibres), the LCA results also differed between the set up scenarios, which made it possible to draw conclusions about the studied system that would not have been possible to draw without some type of scenario modelling, such as: (i) the geographical location of land use does not seem to influence the impact on biodiversity very much, however, it significantly influences water use impacts, and (ii) the transformation of land from a high to a low biodiversity state seems to be a greater driver for biodiversity loss than the occupation of land.

Furthermore, the scenario modelling of paper I enabled us to see that the selected methodological approach (attributional or consequential), and assumptions concerning the modelling of multifunctional end-of-life processes, is of uttermost importance for the result of the LCA. It appears to be particularly important to set up several distinctly different scenarios in consequential end-of-life modelling with substitution, as the outcome of the assessment largely depends on the type of technology assumed to be substituted; this has been recognised before (Mathiesen et al. 2009). It has even been argued that consequential end-of-life modelling should not be done *without* setting up several scenarios with different assumptions of the substituted technology, as the identification of the substituted technology is highly speculative (Heijungs and Guinée 2007). Until there is consensus in the LCA community on when to use attributional or consequential approaches, there is a need to use both approaches simultaneously to facilitate robust LCA-based decision making. There is also a need to generate several distinctly different scenarios for each approach, e.g. regarding the type of substituted technology for consequential end-of-life modelling. To use 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 use of a temporal dimension in the mapping of the product system, as was done in paper I, could potentially offer additional benefits if combined with new LCIA methods. As discussed in section 2.1.2, there are many proposals for, and potential benefits of, more elaborate LCIA methods for climate change impact. For example, dynamic metrics proposed by Levasseur et al. (2010) give less weight to emissions the further into the future they occur. Such time dependent LCIA methods require the introduction of a temporal dimension in the mapping of the product system, so if they gain widespread use, LCA software will probably support temporally dynamic process flowcharts, and thus facilitate the scenario modelling approach of paper I.

Each scenario modelling approach was developed for the specific needs in each case study. It would have made less sense to introduce a temporal dimension in the LCA of textile fibres, as a typical garment is disposed of after a rather short time period[3] (apart from the fact that the study excluded end-of-life processes, as they were expected to be of limited importance for land and water use impacts). Likewise, it would not have been as relevant to test the influence of the geographical location of the life cycles processes of the roof constructions, as supply chains of heavy construction materials are typically not as geographically distributed and flexible with regard to market mechanisms as supply chains for apparel.

3.4 Assessment of forestry impacts

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 for land and water use impact assessment applied in paper II represent improvements

[3] In UK, the average service life of apparel is 2.2 years (WRAP 2012).

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 water use offers the possibility to calculate system-scale hydrological effects of land use at a level not possible in previously proposed LCI approaches. In the case study of paper II, the consequential LCI approach generated results significantly different from results generated by a traditional, attributional approach, which indicates that the consequential approach did add something not captured by attributional approaches. 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 picture of the volume of water used by the product system, 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 or similar. This would be a considerable development of current practices in the impact assessment of water use – which solely focuses on water deficiency as a potential issue – into an impact category perhaps better called “water cycle disturbance”. It may already be possible to combine the consequential LCI approach with more recent suggestions for LCIA methods, such as the water footprint method suggested by Ridoutt and Pfister (2013).

The consequential LCI approach for water use can potentially also improve the calculation of water impact offsets, for example where a change in land use practices causes a hydrological benefit. An offset may be thought of as an environmental impact avoided when a system performs some secondary function, and is a concept already used in natural resource management (the concept is further explained in paper II). Offset calculation does not make sense in an attributional inventory approach that focuses 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 hydraulic consequences of the forestry operation estimated. Thus, offset calculation could be facilitated by the proposed consequential LCI approach. Note that offset calculation is similar to the consequential approach for handling by-products of multifunctional processes (see section 2.3.3).

The LCIA method for water use applied in this study (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 methods in LCA, which simply give the volume of water used, sometimes distinguishing between surface water and groundwater, but not acknowledging the potential consequences further down the cause-effect chain. In the context of the CelluNova project, the method made it possible to quantify the potential benefits of carefully siting the fibre production plant and sourcing the wood.

The method for land use impact assessment (developed by Schmidt 2008) also enabled us to identify product system parameters of importance beyond what would have been possible by using established methods only. 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. Also, the results showed that, in the case study, the location of land use mattered less, as geographical differences in 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 both LCIA methods appears to be the use of regionalised characterisation factors, which is a necessity for improving the assessment of geographically dependent impacts such as land and water use impacts. The land use impact assessment method cannot, however, assess differences between closely related activities, due to limitations in data availability. This is a drawback recognised previously for other methods for characterising land use impact on biodiversity (Antón et al. 2007). Therefore, the applied method is, in its current form, most useful for assessments supporting strategic macro level decision-making (e.g. whether to transform natural land or not, or which regions to source wood from), but less useful for supporting micro level decision-making (e.g. what specific forest to source land 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 enable comparisons between uncertified land and land managed according to certain certification principles, such as the Forest Stewardship Council (FSC 2013) or the Programme for the Endorsement of Forest Certification (PEFC 2013). The method for water use impact assessment can also be used with more refined data. In the case study reported in paper II, 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 actually been published online for over 11,000 watersheds in a format compatible with Google Earth (ETH 2011). The consequential LCI approach for water use could also be improved by more refined data, such as runoff data for specific land management practices.

Another critical methodological aspect that was identified in paper II was how to allocate the transformational impact between the first harvest after transformation and subsequent harvests. This proved very important in the comparison of a wood-based fibre from forestry with a rotation time of 62.5 years and a cotton fibre from a cotton plantation with a rotation time of 0.5 years. How to solve this allocation problem deserves further research, as it can be expected to be a recurring dilemma of significance for many comparisons of products derived from crops with different rotational times, i.e. in comparisons of wood-based and other bio-based products.

Furthermore, the method for land use impact assessment could be improved by complementary indicators, on other groups of species or other facets of biodiversity and/or ecosystem quality. The need for multiple impact factors has been emphasised also by Curran et al. (2011). As was concluded in the review of methods in section 2.5.1, many 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 route in this direction. It opens up for using several groups of species, where the choice of species groups can be based on the data available for a certain region. However, their method does not yet support transformational impact assessment, which, as was shown in paper II, needs to be included in a robust method for land use impact assessment.

To conclude, by using emerging methods of land and water use impact assessment in the CelluNova project case study, it was possible to identify their benefits and drawbacks, and pinpoint where further research is needed. Also, by using a consequential LCI approach for water use, 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 use. For the context of the CelluNova project, the applied methods led to findings that would have been missed if only established methods had been used.

4 Conclusions

This chapter summarises how the research presented in this thesis contributes to answering the research questions. The questions are handled one at a time.

1. How can the inherent uncertainties of future, still non-existent product systems be captured in environmental assessments?

In paper I, a two-step approach was developed and used in an LCA on construction materials in order to deal with the uncertainties of future life cycle processes. First, a temporal dimension was introduced in the mapping of the product system in order to separate life cycle processes occurring in the near future and in a distant and more uncertain future. Then, scenarios were set up to explore how different future technologies (with low or high environmental impact) and different methodological approaches (consequential and attributional) influence the assessment of processes occurring in a distant future. The LCA results differed significantly between the scenarios.

In paper II, another approach for scenario modelling was developed and used in an LCA on textile fibres. Two factors of the product system which are expected to influence the assessed environmental impacts were identified: the geographical location and the type of marginal land use. Then, two parameters that are expected to influence these factors were varied – the market demand for fibres and the competition for land – to generate a range of scenarios of the product systems. Again, the results differed significantly between the set up scenarios.

Both approaches provided a more comprehensive assessment than would have been generated with only one scenario (e.g. the most probable one); therefore, the methods can be recommended for further use in assessments of future and uncertain product systems. Their use should, however, fit the context of the study. The approach of paper I particularly suits assessments of long-lived products and the approach of paper II particularly suits assessments of products with globally distributed supply chains.

2. How can the impact assessment of forestry be improved?

In paper II, impact assessment methods suggested in the literature were used to capture textile fibres' land use impact on biodiversity and water use impact on human health, ecosystem quality and resources. A consequential LCI approach was developed and used for the water use impact assessment, to better capture the effects that forestry can have on the hydrological cycle.

The methods made it possible to generate results which depended on the geographical location of land use and whether or not land was transformed. This is beyond what is offered by currently established methods for impact assessment of land and water use. It was also concluded that the results did reflect realities in a meaningful way; for example, transformation of land from a high to a low biodiversity state significantly increased the biodiversity impact score, water use in water-stressed areas generated a higher impact score than water use in areas without water stress, and the innovative LCI approach made it possible to, for the first time in LCA (to our knowledge), generate impact scores reflecting that certain types of land use can positively contribute to downstream water availability. Although some methodological controversies remain to be solved, the methods successfully contributed towards an improved impact assessment of forestry. With additional development and/or more refined data, the methods could offer further improvements to the impact assessment of forestry.

5 Future research

This chapter summarises research needs identified in my work to date, including a brief description of the future direction of my own research.

Apart from research needs directly connected to the research questions of this thesis, there is a need for research on the particular challenges encountered when conducting sustainability assessments in the context of technical development projects. My colleagues and I will particularly address how to identify and decide on the appropriate roles of environmental assessments in the type of intra-organisational projects that the case study projects represent, and how to plan projects accordingly. Preliminary results were shown in conference presentation D (see list of publications, page vii).

There is a need for further research on scenario modelling in LCA, for example on how to validate the quality of generated scenarios, or on what types of scenario modelling that are suitable in different product development 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 end-of-life modelling approaches have managed to capture the actual end-of-life practices of the future.

Furthermore, there is a need for more research into attributional and consequential modelling, particularly for end-of-life processes of long-lived products. It is desirable to build a consensus within the LCA community on under what circumstances attributional and consequential approaches are suitable, and to further develop LCA guidelines (such as the ISO 14040/14044 standard and the ILCD Handbook) that provide clear and consistent guidance for when and how to use either approach. More research is also warranted for exploring what end-of-life assumptions that are of importance 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 methods for land and water use impact assessment suggested in the literature. In particular, there is a need for case studies that compare different methods, e.g. with regard to the reliability of the methods and their applicability in various contexts. Concerning methods for land use impact assessment, 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 work by de Baan et al. (2012), in which they studied the correlation between their proposed 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 only available for specific regions or product categories. Also, it is essential to further discuss how to allocate transformational land use impact between the first harvest after transformation and subsequent harvests.

As paper II presented the first forestry case study for the consequential LCI approach for water use, other researchers need to be involved in providing further case studies where the approach is tried out. Moreover, further developments of the approach could utilise some of the potential benefits of the approach discussed in section 3.4, such as enabling offset calculation or the capturing of potential problems of too much water in certain regions.

My colleagues and I will continue to contribute with case studies for improving the impact assessment of forestry and other land uses. Within a research programme aimed at making the Swedish fashion industry more sustainable (Mistra Future Fashion 2013), we will expand the LCA in paper II to include more types of textile fibres and entire garment life-cycles. This will give us opportunities to further test methods of land and water use impact assessment as well as further explore potential improvements of other impact categories of relevance for wood-based products.

It is apparent that many aspects of sustainability assessments in the development of wood-based products deserve further research. Hopefully, research will continue to improve the quality and reliability of assessments, and thereby contribute to making the products of tomorrow more sustainable than the products of today.

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