REPORT

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THE METHOD'S INFLUENCE ON CLIMATE IMPACT ASSESSMENT OF BIOFUELS AND OTHER USES OF FOREST BIOMASS

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Photo: Hans Holmberg

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THE SWEDISH KNOWLEDGE CENTRE
FOR RENEWABLE TRANSPORTATION FUELS

PREFACE

This report is the result of a collaborative project within the Swedish Knowledge Centre for Renewable Transportation Fuels (f3). f3 is a networking organization, which focuses on development of environmentally, economically and socially sustainable renewable fuels, and

- Provides a broad, scientifically based and trustworthy source of knowledge for industry, governments and public authorities,
- Carries through system oriented research related to the entire renewable fuels value chain,
- Acts as national platform stimulating interaction nationally and internationally.

f3 partners include Sweden's most active universities and research institutes within the field, as well as a broad range of industry companies with high relevance. f3 has no political agenda and does not conduct lobbying activities for specific fuels or systems, nor for the f3 partners' respective areas of interest.

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EXECUTIVE SUMMARY

TOWARDS A BIO-ECONOMY: THE ROLE OF THE FOREST

Biomass has an increasingly important role in replacing fossil and mineral resources, and it is central in environmental impact-reduction strategies in companies and governments, locally, nationally and internationally. The European Union (EU) has recently taken action to strengthen the bio-economy, defined as "*…the sustainable production and conversion of biomass into a range of food, health, fibre and industrial products and energy*".

Two thirds of the land area in Sweden is covered by forests, and forestry has been an important industry for centuries. Increased and/or more efficient use of forest biomass thus has a great potential for replacing the use of fossil and mineral resources in Sweden.

There are two main reasons for why forest- and other bio-based products are seen as environmentally beneficial. Biomass is (most often) a renewable resource, in contrast to finite fossil and mineral resources, and there is often a balance between CO_2 captured when the biomass grows, and CO_2 released when the bio-based product is incinerated.

THE CHALLENGE: CALCULATE CARBON FOOTPRINTS

Moving towards a bio-economy means replacing non-renewable fuels and materials with bio-based fuels and materials. This is a transition on many levels: technology, business models, infrastructure, political priorities, etc. To guide such a grand transition, there is a need to understand the environmental implications of new bio-based products. This includes assessing their climate impact, so-called carbon footprinting.

Carbon footprinting of forest products is not as simple as saying that forest products are carbon and climate neutral by definition. Fossil energy used for producing and transporting the products has a carbon footprint. Also, the carbon balance can differ between forest products, which can influence their carbon footprint. For example, carbon stored in products, while CO_2 is captured in the re-growing forest, can mitigate climate change. The modelling of the carbon balance is influenced by the study's geographical system boundaries – national, regional, landscape and single-stand perspectives often yield different results. Forestry can also lead to positive or negative changes in the levels of carbon stored in the soil, the levels of aerosols emitted by the trees (influencing cloud formation), and the albedo (surface reflectivity) of the forest land. An indirect effect of forestry can be increased competition for land, with expanding or intensified land use elsewhere, with positive or negative climate effects. All these factors are potentially important when calculating carbon footprints.

There is limited knowledge about how and to which extent the aforementioned factors influence the carbon footprint of forest products. Also, there is a lack of methods for assessing some of these factors. In light of this, can the carbon footprints of today be trusted? And can we ensure that they provide relevant and robust decision support?

OUR APPROACH: TESTING THREE DIFFERENT CARBON FOOTPRINT METHODS IN FIVE CASE STUDIES

In this study, we have:

1. Identified different carbon footprint methods.
2. Used the identified methods to calculate the carbon footprint of different forest products and non-forest benchmarks (using life cycle assessment, LCA).
3. Compared the results to find out how and why they differ.

We identified three main categories of carbon footprint methods: (i) the common practice in LCA, (ii) recommendations in standards and directives (we tested the EU sustainability criteria for biofuels and bioliquids and the Product Environmental Footprint (PEF) guide), and (iii) more advanced methods proposed in the scientific literature (we tested dynamic LCA). For dynamic LCA, we tested different time horizons (20 and 100 years) and different geographical system boundaries, based on (a) the national level, assuming a net annual growth of biomass (which is the case in Sweden); (b) the landscape level, assuming a balance between the annual harvesting and growth (the level at which forests are often managed); and (c) the stand level, assuming regrowth during a time period of 80 years (a stand is the part of a landscape that is harvested in one year, a level often used by researchers developing new methods for modelling the dynamics of forest carbon flows).

These methods were applied to five forest products: two automotive fuels (a lignin-based fuel produced from black liquor and butanol), a textile fibre (viscose), a timber structure building, and a chemical (methanol, used for different end products).

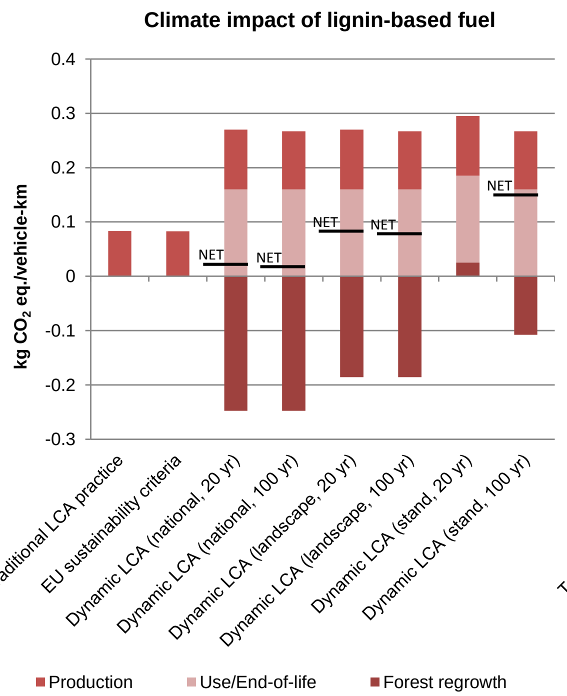
OUR FINDINGS

We found that different carbon footprint methods can give different results, as shown for the biofuel case studies in Figure A. The common practice is close to the recommendation in the EU sustainability criteria and the PEF guide. Results from dynamic LCA differ considerably, as it accounts for the timing of (fossil and biogenic) greenhouse gas (GHG) emissions and CO_2 capture, which is ignored by the other methods. The results of dynamic LCA depend primarily on the geographical system boundaries, but also on the time horizon.

When applying dynamic LCA with a stand perspective, we assumed that the CO_2 uptake occurs *after* harvest. Alternatively, one could assume that the CO_2 uptake occurs *before* harvest, which would give different (lower) results.

When comparing the carbon footprints of the forest products with products they could be expected to replace, we see that the results for the forest products could range from being definitely favourable to worse (see Figure B).

More results can be found in the full report. Results were produced to answer the research questions of this study, and should not be used out of context.



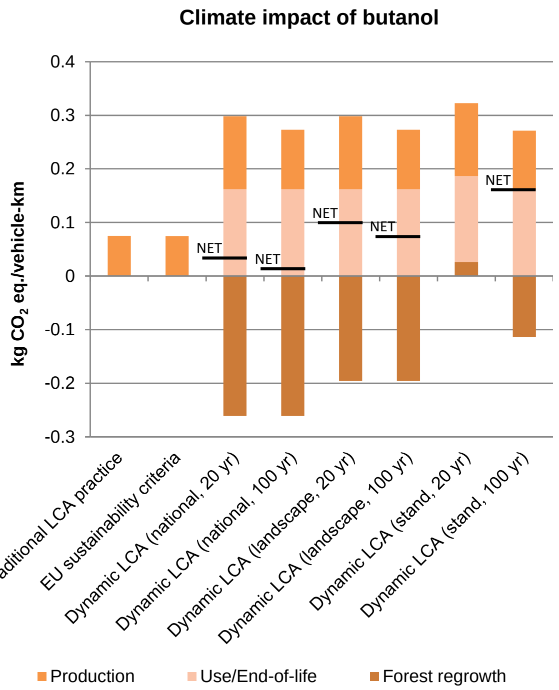


Figure A. Climate impact of the biofuels for different carbon footprint methods.

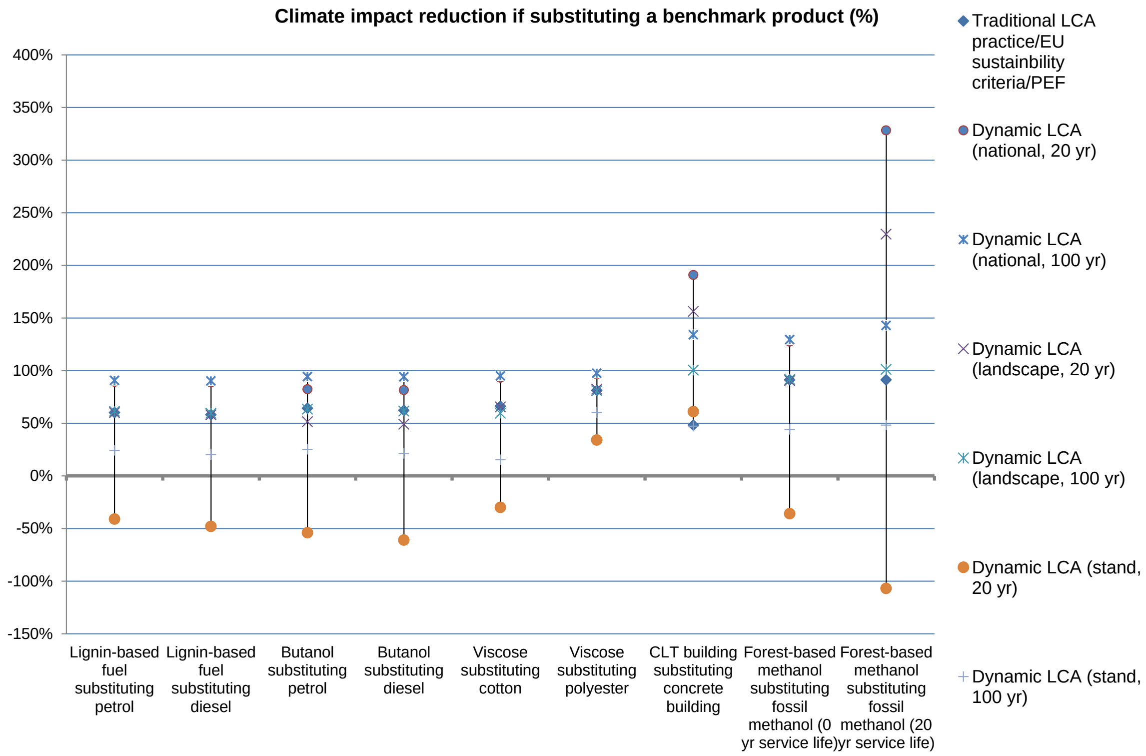


Figure B. Climate impact reductions, if each forest product is assumed to substitute its benchmark product (values >0% mean that substituting the benchmark reduces impact; values >100% mean that more than all the impact of the benchmark is offset).

CONCLUSIONS AND RECOMMENDATIONS

Because there is (still) limited knowledge about how forest products influence the climate, and as carbon footprints will always depend on value-based assumptions (e.g. regarding geographical system boundaries), it is not possible to recommend one specific method which is suitable regardless of context. As different carbon footprint methods can give very different results, our key message is that we need to increase **consciousness** on these matters. It is important to be aware of the assumptions made in the study, the effects of those assumptions on results, and how results can and cannot be used for decision support in a certain context. More specific recommendations for decision makers are listed below. Further details and results can be found in the main report, along with recommendations for LCA practitioners and researchers.

- Decision makers must be aware that the main methodological choices influencing carbon footprints of Swedish forest products are the choice of geographical system boundaries (e.g. national-, landscape- or stand-level system boundaries) and whether the timing of CO_2 capture and GHG emissions is accounted for. This is because Swedish forests are, in general, slow growing.

- If the aim of the decision is to obtain short-term climate impact reduction – for example, the urgent reduction that is possibly needed for preventing the world average temperature to rise with more than 2°C – the timing of CO_2 capture and GHG emissions should be taken into account. Decision makers must be aware that a particular method for capturing timing (such as dynamic LCA) can be combined with different system boundaries, which can yield different results.

- When conclusions from existing LCA studies are synthesized for decision support, the decision maker must be aware that most existing studies do not account for the timing of CO_2 capture and GHG emissions. This is particularly important when the decision concerns the prioritization of forest products with different service lives (e.g., fuels versus buildings).

- When timing is considered, decision makers must be aware that there are different views on when the CO_2 capture occurs, which will influence the carbon footprint. One could either consider the CO_2 captured before the harvest (i.e., the capture of the carbon that goes into the product system), or the CO_2 captured after the harvest (i.e., the consequence of the harvest operation). In this study, we tested the second alternative when we applied dynamic LCA with a stand perspective – this does not mean we advocate the use of the second alternative over the first alternative.

- Decision makers must be aware that the location and management practices of the forestry influence the climate impact of a forest product. For example, growth rates, changes in soil carbon storages and fertilisers (a source of GHGs) differ between locations.

- Based on our results, we cannot say that the carbon footprints of some product categories are more robust than for others, i.e. less influenced by choice of methodology. However, the more forest biomass use in the product system, the higher the influence of the choice of method.

- As many interactions between the forest and the climate are still not fully understood, it is important to be open to new knowledge gained in methodology development work.

- Regarding how to use Swedish forests for the most efficient climate impact reduction, it is impossible to draw a general conclusion on the basis of our results. Factors that influence the "optimal" use are:
  - Which fraction of forest biomass that is used. Various products use different fractions (as was the case in our case studies) and do not necessarily compete for the same biomass. However, a production system may be more or less optimised for a specific output. So there may be situations of competition also when feedstocks are not directly interchangeable.
  - Which non-forest product that is assumed to be replaced by the forest product (if any). The carbon footprint of the non-forest product matters, but also how large the substitution effect is (i.e., does the forest product actually replace the non-forest alternative, or merely add products to the market, and what are the rebound effects from increased production?).
  - If all other factors are identical: the longer the service life of the forest product the better, due to the climate benefit of storing carbon and thereby delaying CO_2 emissions. This effect is particularly strong if the aim is to obtain short-term climate impact reduction. Moreover, the effect supports so-called cascade use of forest biomass, e.g. first using wood in a building structure, then reusing the wood in a commodity, and at end-of-life, as late as possible, recovering the energy content of the wood for heat or fuel production.
- Traditional LCA practice and methods required by the EU sustainability criteria and PEF have limitations in the support they can provide for the transition to a bio-economy, as they cannot capture the variations of different forest products in terms of rotation periods and service lives. Thus, decision makers need to consider studies using more advanced methods to be able to distinguish better or worse uses of forest biomass. We have tested one such advanced method (dynamic LCA), that proved applicable in combination with several different geographical perspectives, but also other methods exist (e.g. GWP_{bio}).
- Climate change is not the only environmental impact category which is relevant in decision making concerned with how to use forests. Other environmental issues, such as loss of biodiversity and ecosystem services, are also important. There are also non-environmental sustainability issues of potential importance, e.g. related to indigenous rights and job creation.

SAMMANFATTNING (SUMMARY IN SWEDISH)

OM VÄGEN MOT EN BIOEKONOMI: SKOGENS ROLL

Biomassa spelar en allt viktigare roll i att ersätta ändliga resurser och är därför en central resurs i olika strategier för att minska miljöpåverkan, hos företag och myndigheter, lokalt, nationellt och internationellt. Till exempel har EU tagit viktiga steg mot en mer biobaserad ekonomi, bland annat genom politiska mål och styrningen av medel till forskning och utveckling.

Två tredjedelar av Sverige är täckt av skogar och skogsnäringen är en viktig svensk industri sedan århundraden. Ökad och/eller mer effektiv användning av skogsbiomassa har därför en stor potential att ersätta användningen av icke-förnyelsebara resurser i Sverige.

Det finns två huvudanledningar till att skogsprodukter och andra biobaserade produkter ses som miljömässigt fördelaktiga. Biomassa är (oftast) en förnyelsebar resurs, till skillnad mot ändliga fossila resurser och mineraler, och det det är ofta en balans mellan CO_2 som binds när biomassa växer till och CO_2 som släpps ut när biobaserade produkter förbränns.

UTMANINGEN: BERÄKNA KOLFOTAVTRYCK

Vägen mot en bioekonomi innebär att biobaserade bränslen och material ersätter icke-förnyelsebara bränslen och material. Denna omställning sker på flera nivåer: teknologi, affärsmodeller, infrastruktur, politiska prioriteringar, m fl. Guidning av en sådan omställning fordrar förståelse av de miljömässiga konsekvenserna av nya biobaserade produkter. Detta innefattar bland annat beräkning av klimatpåverkan, så kallat kolfotavtryck (*carbon footprint*, på engelska).

Att beräkna skogsprodukters kolfotavtryck är inte så enkelt som att säga att de per definition är kol- och klimatneutrala. Fossil energi används i produktion och transporter av skogsprodukter, vilket ger ett kolfotavtryck. Dessutom kan kolbalansen se olika ut för olika skogsprodukter, vilket påverkar deras kolfotavtryck. Till exempel kan den kol som lagras i skogsprodukter – samtidigt som CO_2 fångas in i den återväxande skogen – bidra till minskad klimatpåverkan. Modellering av kolbalansen beror på studiens geografiska systemgränser – nationellt, regionalt, landskaps- eller bestånds-perspektiv kan ge olika slutsatser. Skogsbruk kan också leda till positiva och negativa förändringar i den mängd kol som lagras i mark, i hur mycket aerosoler som träd avger (som påverkar molnbildning) och skogens albedo (ytreflektivitet). En indirekt effekt av skogsbruk kan vara ökad markkonkurrens, som kan leda till ökad eller intensifierad markanvändning i andra delar av världen, med positiva eller negativa klimateffekter. Alla dessa faktorer är potentiellt viktiga vid beräkning av kolfotavtryck.

Det finns begränsad kunskap om hur, och hur mycket, flera av de ovan beskrivna faktorerna bidrar till skogsprodukters kolfotavtryck. Därför är befintliga beräkningsmetoder otillräckliga för att fånga alla potentiellt relevanta faktorer. Med detta i åtanke, går det att lita på dagens beräkningar av kolfotavtryck? Och kan vi säkerställa att kolfotavtryck bidrar till relevanta och robusta beslutsunderlag?

TILLVÄGAGÅNGSSÄTT: TESTA TRE OLIKA KOLFOTAVTRYCKSMETODER I FEM OLIKA FALLSTUDIER

I denna studie har vi:

1. Identifierat olika metoder för att beräkna kolfotavtryck.
2. Använt dessa metoder i livscykelanalyser (LCA) på fem olika skogsprodukter och jämförbara referensprodukter tillverkade från andra resurser.
3. Jämfört resultaten för att se hur och varför de skiljer sig.

Vi fann tre huvudkategorier av kolfotavtrycksmetoder: (i) det tillvägagångssätt som LCA-utövare normalt använder, (ii) rekommendationer i standarder och direktiv (vi testade EU:s hållbarhetskriterier för biodrivmedel och flytande biobränslen samt guiden för Product Environmental Footprints, PEF), samt (iii) avancerade metoder som föreslås i den vetenskapliga litteraturen (vi testade dynamic LCA). För dynamic LCA testade vi olika tidshorisonter (20 och 100 år) och olika geografiska systemgränser, baserat på (a) nationell nivå (årlig nettotillväxt av biomassa, vilket är fallet i Sverige); (b) landskapsnivå (balans mellan årlig avverkning och tillväxt, ofta nivån på vilket skogar sköts); och (c) beståndsnivå (den del av landskapet som avverkas under ett år, där återväxt sker under 80 år; en nivå som ofta används av forskare som tar fram nya metoder för modellering av skogens kolflöden).

Dessa metoder användes på fem olika skogsprodukter: två drivmedel (ett lignin-baserat drivmedel producerat från svartlut samt butanol), en textilfiber (viskos), en byggnad med timmerkonstruktion, och en kemikalie (metanol, använd för olika slutprodukter).

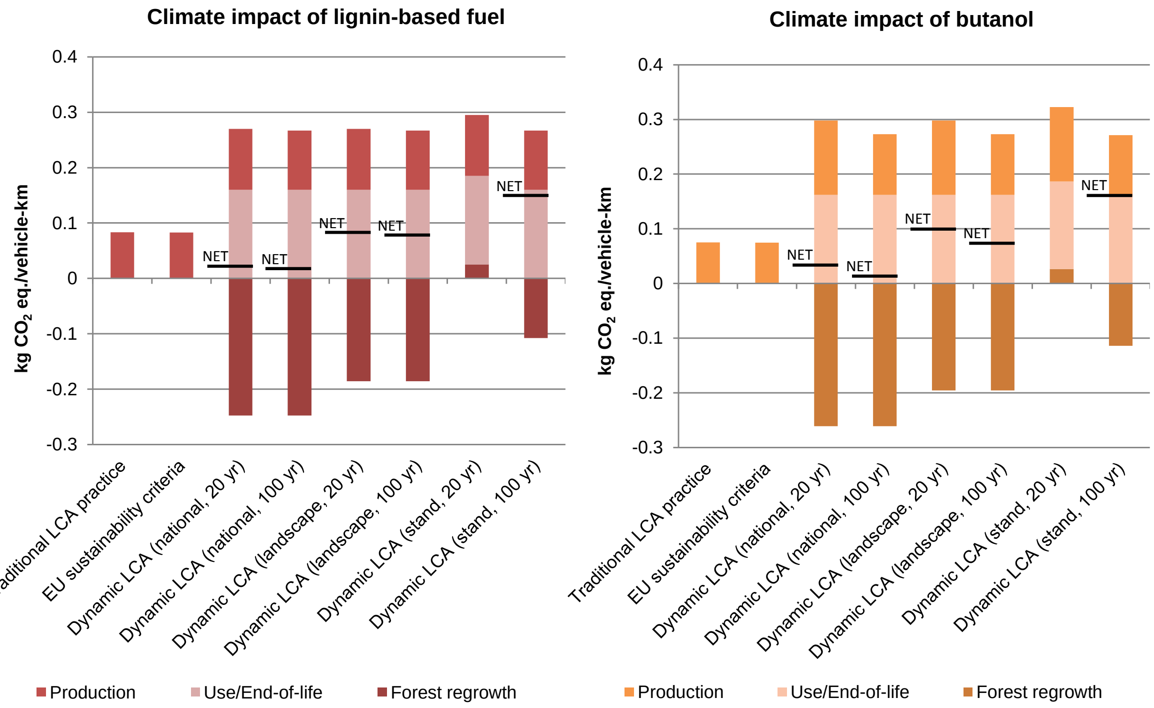
RESULTAT

Vi fann att olika metoder för att beräkna kolfotavtryck kan ge olika resultat, vilket visas för de studerade biodrivmedlen i Figur A. Den vanliga LCA-metoden är snarlik de metoder som rekommenderas i EU:s hållbarhetskriterier och i PEF-guiden. Däremot är resultat från dynamic LCA helt annorlunda, då metoden beaktar när (biogena och fossila) utsläpp av växthusgaser och CO_2-upptag sker, till skillnad från övriga metoder som bortser från detta samt utelämnar CO_2-upptag och biogena CO_2-utsläpp. Vidare beror resultatet för dynamic LCA primärt på geografiska systemgränser men även på tidshorisont.

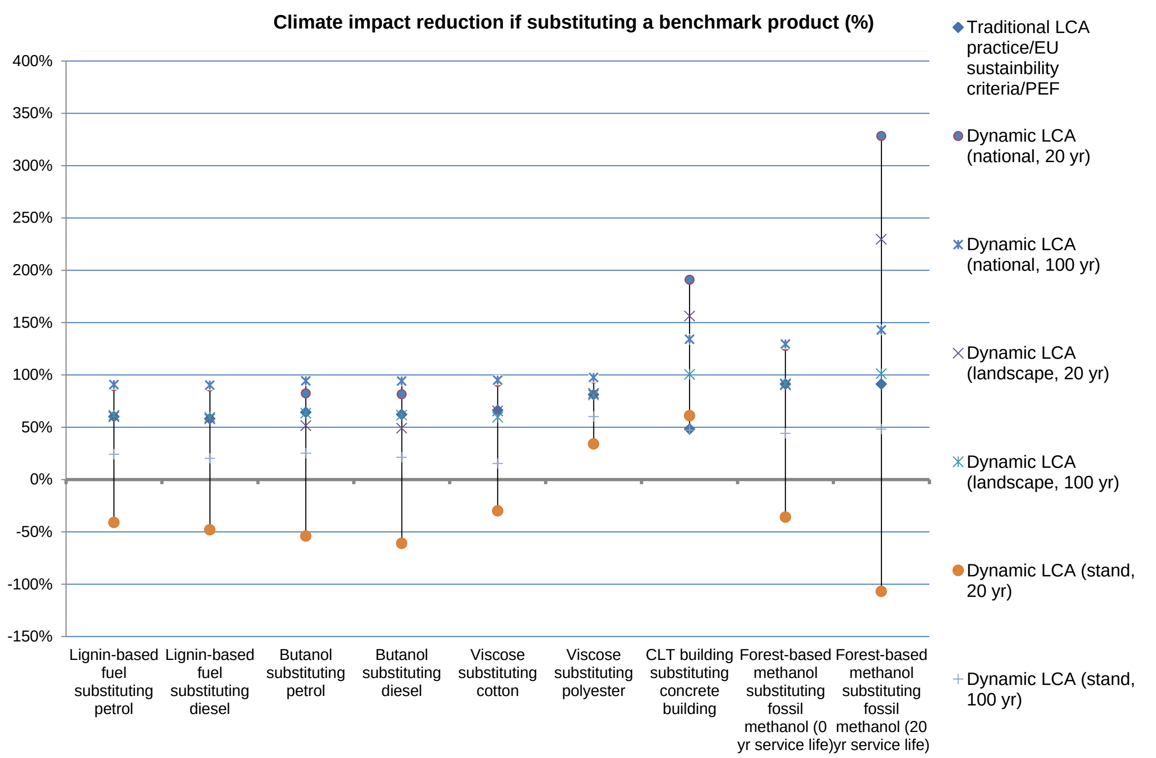
När vi använde dynamic LCA med beståndsbaserade systemgränser så antog vi att CO_2-upptag sker *efter* avverkning. Alternativt kan man anta CO_2-upptag *innan* avverkning, vilket skulle ge olika (lägre) resultat.

Vid jämförelse av kolfotavtryck från skogsprodukterna och de referensprodukter som de kan antas ersätta, så kan skogsprodukterna vara antingen klimatmässigt betydligt bättre eller sämre (Figur B).

Ytterligare resultat finns i projektrapporten. Det ska understrykas att resultaten är framtagna för att svara på studiens forskningsfrågor och inte är avsedda att användas i andra sammanhang.



Figur A. Klimatpåverkan från två olika biodrivmedel för olika metoder för att beräkna kolfotavtryck.



Figur B. Minskad klimatpåverkan vid antagandet att varje skogsprodukt ersätter sin referensprodukt (värden högre än 0 % innebär att ersättandet av referensprodukten minskar klimatpåverkan; värden högre än 100 % innebär att mer än hela referensproduktens klimatpåverkan undviks).

SLUTSATSER OCH REKOMMENDATIONER

Eftersom det (ännu) finns begränsad förståelse för hur skogsprodukter påverkar klimatet, och eftersom kolfotavtryck alltid kommer att bero på värdebaserade antaganden (t ex angående geografiska systemgränser), så är det inte möjligt att rekommendera en specifik metod som är lämplig oberoende av sammanhang. Eftersom olika metoder för att beräkna kolfotavtryck kan ge väldigt olika resultat, så är vårt huvudbuskap att vara **medveten** och **uppmärksam**. Det är viktigt att vara medveten och uppmärksam på de antaganden som gjorts i en studie, de effekter dessa antaganden har på resultatet, och huruvida det är lämpligt att använda resultat som beslutsunderlag i ett visst sammanhang. Mer specifika rekommendationer för beslutsfattare listas nedan. För ytterligare detaljer och resultat hänvisar vi till projektrapporten, där det även finns rekommendationer riktade till LCA-utövare och forskare.

- Beslutsfattare bör vara medvetna om att de metodval som har störst påverkan på beräkningar av kolfotavtryck av svenska skogsprodukter är valet av geografiska systemgränser (t ex landskaps- eller beståndsbaserade gränser) och huruvida metoden beaktar när upptag och utsläpp av växthusgaser sker. Detta kommer sig av att svenska skogsbestånd har relativt långsam återväxt.

- Om studien ska utgöra underlag till beslut som syftar till att åstadkomma kortsiktiga klimatvinster – till exempel den snabba utsläppsminskning som kan krävas för att uppfylla tvågradersmålet – så är det viktigt att beakta när upptag och utsläpp av växthusgaser sker. Beslutsfattare bör vara medvetna om att en metod som kan beakta detta (såsom dynamic LCA) kan kombineras med olika systemgränser, vilket kan ge olika resultat.

- När slutsatser från tidigare LCA-studier ska användas som beslutsunderlag så måste beslutsfattare vara medveten om att de flesta tidigare studier inte beaktar när upptag och utsläpp av växthusgaser sker. Detta är särskilt viktigt när beslutet rör prioriteringar av olika skogsprodukter med olika livslängd (t ex drivmedel jämfört med byggnader).

- Om tidpunkten för upptag och utsläpp av växthusgaser beaktas så måste beslutsfattare vara medvetna om att det finns olika syn på när CO_2-upptaget sker, vilket påverkar kolfotavtrycket. Man kan beakta upptaget innan avverkning (d v s upptaget av det kol som sedan återfinns i produkten), eller upptaget efter avverkning (d v s konsekvensen av avverkningen). I vår studie har vi testat det senare alternativet när vi använde dynamic LCA med beståndsperspektiv – detta innebär inte att vi förordar detta alternativ.

- Beslutsfattare bör vara medvetna om var skogen finns och hur skogsbruk sker påverkar skogsprodukters kolfotavtryck. Till exempel påverkar skogens återväxttakt. Dessutom är förändringar i markkol och gödsling (en källa till växthusgaser) vanligare i vissa delar av landet än i andra.

- Baserat på våra resultat går det inte att säga att kolfotavtryck på vissa typer av produkter är mer robusta än andra, det vill säga mindre influerade av metodval. Dock gäller att ju mer skogsbiomassa som används i produktsystemet, desto högre påverkan från metodval.

- Eftersom det fortfarande finns kunskapsluckor gällande samspelet mellan skog och klimat så är det viktigt att vara mottaglig för ny kunskap som genereras av att metoder förbättras.

- Baserat på våra resultat går det inte att dra generella slutsatser angående hur vi bör använda svensk skog för att mest effektivt minska klimatpåverkan. Faktorer som påverkar "optimal" användning är:

    - Vilken fraktion av skogsbiomassa som används. Olika produkter använder olika fraktioner (vilket var fallet i vår studie) och konkurrerar därför inte om samma biomassa. Dock kan ett produktionssystem vara mer eller mindre optimerat för en viss tillverkning. Det kan således uppstå konkurrenssituationer även när råmaterial inte är direkt utbytbara.

    - Vilken (icke skogslig) produkt som skogsprodukten antas ersätta (om någon). Det spelar roll både hur stor denna produkts kolfotavtryck är och hur stor substitutionseffekten är (det vill säga, till vilken grad ersätts denna produkt, till vilken grad är skogsprodukten snarare ännu en produkt på en ökande marknad, och vad är reboundeffekten av ökad produktion?).

    - Om övriga faktorer är lika: ju längre livslängd som en skogsprodukt har, desto lägre kolfotavtryck. Detta beror på klimatvinsten av att fördröja CO_2-utsläpp från förbränningen genom att lagra kolet i produkten. Denna vinst är särskilt stor om syftet är att åstadkomma kortsiktiga klimatvinster. Fördelen med lång livslängd stödjer kaskadanvändning av skogsbiomassa, till exempel att trä först används som byggnadsmaterial, sedan återvinns exempelvis i möbler, för att vid sluthanteringen, så sent som möjligt, energiåtervinnas till värme- eller drivmedelsproduktion.

- Angående förmågan att ge beslutsunderlag för omställningen till en bioekonomi, så visar vår studie brister med kolfotavtryck som beräknas med vanliga LCA-metoder eller metoder som rekommenderas av EU:s hållbarhetskriterier eller PEF-guiden. Dessa metoder är dåliga på att fånga upp nyanserna i olika sorters skogsprodukter, till exempel olikheter gällande skogens rotationsperiod eller produktens livslängd. Därför bör beslutsfattare beakta studier baserade på mer avancerade metoder, om de vill kunna särskilja bättre från sämre användning av skogsbiomassa. I vår studie har vi testat en sådan metod (dynamic LCA), som visades sig vara applicerbar i kombination med olika geografiska systemgränser, men det finns även andra metoder (t ex GWP_{bio}).

- Klimatfrågan inte är det enda miljöområdet som är relevant att beakta vid beslut om hur vi ska använda våra skogar, även förlust av biologisk mångfald och ekosystemtjänster är viktiga. Dessutom kan andra hållbarhetsaspekter vara viktiga, till exempel att skydda ursprungsbefolkningar och att skapa arbetstillfällen.

CONTENTS

1 INTRODUCTION

1.1 BACKGROUND

As a response to climate change and our dependency on finite resources, the idea of the bio-based society has emerged, in which the use of biotic resources increasingly replase the use of abiotic resources. In Sweden, an important biotic resource is forest biomass, which, among others, can be used for producing biofuels, wood-based building materials, bio-based chemicals, and regenerated cellulose fibres for textile applications.

To guide the transition from a fossil- and mineral-based society to a bio-based society, there is a need to assess and compare the climate impact of different bio-based products in relation to non-renewable alternatives. One of the most used tools for such assessments is life cycle assessments (LCAs), in which environmental impacts of products or services are studied in a life-cycle perspective, from raw material extraction, via production, transportation and use, to waste handling. Among others, LCAs can support decision-making regarding investments in new technologies, the formulation of calls for research and development projects, and the design of product-related policies – which all are important components in a transition to a bio-based society. In Sweden, LCA-based decision-making can help in the development and diffusion of forest products that contribute to achieving the Swedish environmental objective regarding reduced climate impact (Swedish Environmental Protection Agency 2015) and the objective of a fossil-independent vehicle fleet in Sweden by 2030 (Regeringen 2012). Similarly, on the European level, LCA can be used to support the goals formulated in, for example, the renewable energy directive (RED) regarding increased percentage of renewable energy use (European Commission (EC) 2009).

A common misconception regarding the climate impact of forest produc is that renewability equals climate neutrality. This misconception has been increasingly challenged in recent years (see, e.g., Ter-Mikaelian et al. 2015; Agostini et al. 2013; Johnson 2009). A more accurate climate impact assessment of forest products depends on the specific characteristics of the studied system, including the forest from which the biomass is derived (species, rotation period, forestry practices, etc.), on assumptions regarding substituted products (e.g. fossil gasoline, in the case of biofuels), and on methodological choices and delimitations of the study (Matthews et al. 2014; Lamers & Junginger 2013). A particularly crucial methodological choice concerns how the carbon flows throughout the products' life cycle are modelled and quantified, which in turn depends on the spatial and temporal perspectives chosen in the study and the choice of baseline for separating the product system from the natural system (e.g., how does the forestry influence the forest carbon flows). Methodological choices can be influenced by traditions among LCA practitioners, by non-established methods proposed in the scientific literature, or by requirements in standards, directives or other consensus documents (e.g. the EU sustainability criteria for biofuels and bioliquids, or product category rules (PCRs) for product environmental footprints (PEFs)). Methodological choices may significantly influence the outcome of assessments of the climate impact of forest (Røyne et al. 2016; Garcia & Freire 2014; Guest & Strømman 2014; Matthews et al. 2014; Zanchi et al. 2012; Sjølie & Solberg 2011).

1.2 AIMS

The aims of this report are to:

(a) Contribute to more robust decision making concerning how to use Swedish forest biomass for reducing climate impact, with a focus on decision making within the biofuels sector.

(b) Contribute to the process of improving the methods and practices of climate impact assessment in LCAs of forest products.

These aims are addressed by evaluating outcomes of applying different ways of assessing the climate impact of a selected set of forest products, listed in Table 1. Hereafter, we refer to assessments of the climate impact as "carbon footprints". This terminology was chosen as it encompasses both the inventory and impact assessment phases of LCA, which are both influenced by the choice of methodology. The selected products represent different categories for which there are high expectations of increased production, and which are thus subject to intensive technical research and development (R&D) in Sweden and elsewhere. The studied forest products are compared with conventional non-forest products, so-called benchmark products. The tested carbon footprint methods include normal practices in LCAs (as identified by Røyne et al. (2016)), methods required by the EU sustainability criteria for biofuels and bioliquids and PEF, and more advanced, non-established methods proposed in the scientific literature (dynamic LCA), which attempt to capture aspects of the climate impact which are not captured by established methods and practices.

Table 1. Forest products selected for the evaluation of carbon foorprint methods and the non-forest products these are compared to. The functional units represent the base of comparison.

Forest biomass product		Functional unit	Benchmark product
Automotive fuel	Lignin-based fuel	Passenger car driven 1 km	Gasoline
	Butanol		Diesel
Building component	Cross-laminated timber	1 load-bearing structure/m^2 living area	Concrete
Textile fibres	Viscose	1 kg fibers	Cotton
			Polyester
Chemical	Methanol	1 tonne methanol	Natural gas based methanol

By highlighting advantages and disadvantages of established carbon footprint practices, in relation to more advanced methods suggested in the peer-reviewed literature, the results can also potentially contribute to the long-term development of carbon footprint standards and practices, e.g. regarding how to account for biogenic CO_2 emissions in frameworks such as RED.

1.3 DELIMITATIONS

Important delimitations of the study include:

- The study focuses on forest biomass in Sweden. Other biomass types and regions can have other challenges. For example, indirect land use change (ILUC) is more prominent when it comes to agricultural products, and rainforests have other growth rates and conditions than boreal forests.
- The study focuses on climate change. Other environmental impacts, such as biodiversity loss, the impact on ecosystem services and water scarcity, may also be relevant for biomass products and should be considered in decision making.
- The different carbon footprint methods and practices tested in the study capture, in different ways, some of the key mechanisms in how forest products cause climate impact, but not all. Future development of carbon footprint methodology may make it possible to capture additional mechanisms that may significantly influence the outcome of studies of forest products.
- The study does not provide a final answer on how LCA practitioners ought to assess the climate impact of forest products. Carbon footprint methodology is under continuous development, and the study contributes to this development.
- The study does not provide a final answer on the "optimal" use of Swedish forests. The study contributes to the discussion on how to best use the forest, but decision making must consider many aspects, including but not limited to climate impact, as well as uses of forests and forest biomass not accounted for in this study. Also, decisions concerned with specific forests must consider site-specific characteristics not captured in the carbon footprint methods applied in this study.

1.4 INTENDED AUDIENCE AND APPLICATION

The intended audiences of aim (a) – contribute to more robust decision making – are the stakeholders of the forest product sector in Sweden, particularly the stakeholders of the biofuels sector, such as producers of biofuels, forest companies, potential investors and policy makers. Here, the present study can increase the understanding in several ways, for example regarding:

- Uncertainties of carbon footprints of forest products.
- What is important to consider in different decision making contexts.

The intended audience of aim (b) – improve methods and practices – is primarily the international community of LCA researchers and practitioners, which can use the study to increase their understanding of the consequences of the choice of carbon footprint methodology in studies of forest products. This can contribute to improved practices among LCA practitioners and help in directing the future development of carbon footprint methodology (e.g. in terms of what aspects of the climate impact that are important for future methods to capture). This can result in more robust and context-aligned LCA practices that provide a better decision support for the intended audience, which ultimately will be better guided by LCA-informed decision-making.

1.5 SCOPE IN RELATION TO OTHER RESEARCH PROJECTS

The study is carried out within f3, the Swedish Knowledge Centre for Renewable Transportation Fuels, which has financed several related projects before, such as "Alternative sources for products competing with forest based biofuel, a pre-study" (Staffas et al. 2013), "Biofuel and land use in Sweden – an overview of land-use change effects" (Höglund et al. 2013), "Biorefineries and LCA-methodology" (Ahlgren et al. 2013), "GHG Calculations", "Kristianstad biorefinery LCA" (Ekman et al. 2013) och "Beyond LCI" (fore more information, see www.f3centre.se). In Sweden, related research is also carried out within the Swedish Energy Agency's research program "Bränsleprogrammet hållbarhet" (In English: The Fuel Programme for Sustainability), which, among others, aims at producing data for calculating carbon footprints of biofuels from Swedish forest biomass (Energimyndigheten 2013; SLU 2013a). The present study differs from aforementioned research, as it (i) considers a more varied set of different uses of Swedish forest biomass (ranging from short-lived fuels to long-lived building components); (ii) focusses on uses that are expected to increase in importance for the Swedish forest industry; (iii) considers several different carbon footprint methods – not only methods required/established today, but also those that can be expected to become required/established as well as practically applicable in LCAs in a not-to-distant future (i.e. supported by LCA software and commercial databases for inventory data).

1.6 GUIDE TO READERS

Following the introduction, Chapter 2 describes LCA methodology, the challenges of assessing the climate impact of forest products, and the carbon footprint methods selected and tested in the present study. Chapter 3 describes the modelling of the five studied product systems and the corresponding benchmark products. Chapter 4 presents the results of the five case studies and a discussion of the results. The main conclusions are summarised in Chapter 5. Supplementary materials are included in the Appendix.

This report is intended to be comprehensive and detailed in terms of the content of the work carried out. For some of the intended audiences, this may make the report difficult to read and comprehend (e.g. for those unused to LCA methodology). If this is the case, we recommend the reader to primarily focus on those chapters deemed interesting, or the executive summary (p.ii) which has been designed to accessible for the target audience of aim (a): the stakeholders of the Swedish forest product sector.

2 THEORY AND METHOD

In this section, we describe LCA methodology and the challenges of assessing the climate impact of forest products in LCAs. Then we describe the carbon footprint methods selected for application in this study and how these relate to the challenges.

2.1 LCA

LCA is the most widely used tool for assessing the environmental impact of products and services. The tool adopts a system perspective in the sense that it allows the LCA practitioner to study the full life cycle of products – from production of raw materials to the product's end-of-life – and a wide range of environmental impacts, although only a subset of life cycle processes and impacts may be studied depending on the goal and scope of the study. LCA consists of a number of steps, according to ISO 14040 and 14044 (ISO 2006a, ISO 2006b):

In the *goal and scope definition*, the LCA practitioner defines the aim of the study, the intended audience, the functional unit, and the studied product system, including descriptions of the processes of the product system and a specification of the temporal and geographical scope of the study. The product system typically includes processes of raw material extraction, production, transportation, product use and waste handling. The functional unit is a quantitative unit reflecting the function of the product, which enables comparisons of different products with identical functions, as is done in each of the case studies of the present study. Examples of functional units can be found in section 1.2, Table 1.

In the *life cycle inventory analysis (LCI)*, the LCA practitioner maps the relevant material and energy flows between processes in the product system, and between the product system and other product systems and the environment. In case studies of the present study, relevant flows *from* the environment include fossil resources (e.g. oil, coal and natural gas) and carbon dioxide (CO_2) captured from the atmosphere by the growing forest, and relevant flows *to* the environment include CO_2 and other greenhouse gases (GHGs) emitted to the atmosphere.

In the *life cycle impact assessment (LCIA)*, the LCA practitioner uses characterisation methods (also called LCIA methods) to translate the LCI data into potential environmental effects per functional unit. The environmental effects are sorted into impact categories. LCIA methodology for the impact category of climate change is in focus in the present study. The choice of LCIA methodology affects also how the LCI is done.

In the *interpretation*, the LCA practitioner interprets the LCIA results in relation to the goal and scope of the study and recommendations are made to the intended audience. The interpretation can, for example, include sensitivity and uncertainty analyses.

2.1.1 Attributional vs. consequential modelling

An important methodological choice in LCA that influences the carbon footprint of forest products is the choice between attributional and consequential modelling. Therefore, these modelling approaches are briefly introduced in below paragraphs, and section 2.2 pinpoints connections between aspects of carbon footprints and the choice of attributional or consequential modelling approaches.

Traditionally, LCA relies on attributional modelling, which (most often) means that only emissions and resource use that are physically connected to the product (e.g. at the production site) are included in the modelling of the product system. Attributional modelling most often attempts to capture the *average* impact of the product system per functional unit.

In contrast, consequential modelling attempts to capture the change of emissions and resource use that occurs as a consequence of a decision (Zamagni et al. 2012; Earles & Halog 2011; Ekvall & Weidema 2004). Alternatively, this can be described as the consequence of increased or decreased production output (i.e., an increase or decrease in the number of functional units provided by the product system), i.e. the *marginal* impact of the product system. Consequential modelling (most often) requires the consideration of effects not necessarily occurring at the site of the life cycle processes, but occurring because of market effects (Earles & Halog 2011; Ekvall & Weidema 2004).

Whether attributional or consequential modelling is suitable depends on the goal of the study. The choice in turn influences several other critical methodological choices, such as the setting of system boundaries, e.g. in terms of whether to account for indirect land use change (see section 2.2.1) and the choice of baseline (see section 2.2.3).

2.2 CHALLENGES OF CARBON FOOTPRINTS OF FOREST PRODUCTS

Challenges of assessing the climate impact of forest products relates to:

(i) limitations in the understanding of how forests and the climate interact,

(ii) limitations in the understanding of how this interaction is influenced by the extraction of forest biomass,

(iii) limitations in the ability to model this interaction, and

(iv) value-based modelling choices, e.g. in terms of the setting of spatial and temporal system boundaries.

Below, we provide detailed descriptions of some of these challenges, with a focus on those challenges that in recent years have been extensively discussed in the LCA literature. This means that the focus is on methodological challenges in LCA rather than on the limited understanding of interactions between forests and climate. The reason for describing these challenges in detail is for the reader to understand the limitations of current LCA practices, the challenges facing LCA practitioners assessing the climate impact of forest products, and the ongoing development of improved carbon footprint methodology. Understanding these challenges also helps the reader to grasp the differences between the LCI and LCIA methods applied in the present study.

First, we describe the challenges of the spatial and temporal modelling of the carbon flows. Then, the role of the choice of baseline for the modelling is described, which influences both the spatial and temporal modelling. The final subsection describes non-carbon climate aspects of importance for forest products: non-carbon GHGs emitted from forests and the influence of forestry on the formation of aerosols and the albedo effect; aspects which are also influenced by the spatial and temporal modelling and the choice of baseline. The non-carbon aspects are described in less detail than the aspects related to the modelling of the carbon flows. This is because of the focus on different approaches of modelling the carbon flows among the selected carbon footprint methods (which are further described in section 2.3), which in turn is because of a lack of methods for capturing several of the non-carbon aspects.

It should be noted that some of the described challenges of carbon footprinting belong to the goal and scope definition of an LCA, some rather belong to the LCI, and some to the LCIA (hence the use of the term "carbon footprint" instead of "impact assessment"). Related to this, the choice of LCIA method influences how the LCI is carried out, and the opportunities to apply certain LCI methods can be limited by the availability of LCI data.

2.2.1 Spatial aspects of modelling the carbon flows

The choice of spatial system boundaries can influence the view on the carbon balance in the forest (spatial system boundaries also influences other climate aspects of forests, e.g. the albedo effect discussed in section 2.2.4). For example, the system boundaries influence whether it is reasonable to assume that the extraction of forest biomass is carbon neutral, a carbon sink or a source of CO_2 emissions. LCA practitioners can, for example, choose between global, regional, national, landscape-level, and stand-level system boundaries. As global boreal biomass stocks increase (Liski et al. 2003) while tropical stocks decrease (IPCC 2013), *global system boundaries* makes it reasonable to assume that extraction of forest biomass from boreal forests is a carbon neutral activity or even a carbon sink, while extraction of tropical biomass contributes to increased atmospheric CO_2 concentration. However, assumptions regarding the carbon balance can be different with *regional/national system boundaries*, for example as biomass stocks in certain boreal regions/countries can *decrease* while biomass stocks in certain tropical regions can *increase*. Furthermore, with *landscape*- or *stand-level system boundaries* the view on the carbon balance can be different still. It is often these two levels of spatial system boundaries that are discussed in the LCA literature (Cherubini et al. 2013) and in studies of the "carbon payback time" or "carbon debt" of biobased energy sources (Jonker et al 2014; Lamers & Junginger 2013). A well-managed forest is often said to be (at least) carbon neutral at the landscape level. Landscape level means considering an area that is managed systematically in such a way that each year as much (at least) biomass grows as is harvested (thus one could interpret the landscape level as being, by definition, at least carbon neutral); i.e., "a forest estate with equal areas of each age class" (Berndes et al. 2013, p. 292). In contrast, on a stand level, forests are not carbon neutral, at least with a short time perspective: a stand harvested in a boreal forest can take nearly 100 years to regrow to attain carbon neutrality. For a long-term strategy regarding how we should manage forests globally, nationally or regionally, a landscape perspective has been argued to be a reasonable perspective (e.g., see Berndes et al. 2013), and it is from this perspective that the usual assumption of the carbon neutral forest biomass has emerged. Apart from the above described system boundaries, one could also base system boundaries on ownership. For example, if a company owns forests at several locations (even at several continents), and those forests are managed sustainably in terms of the carbon balance, this could be the basis for assuming carbon neutral forest biomass.

Another important dimension of the spatial system boundaries is whether one considers climate impact occurring at the sites of the product system (in particular, at the site of the forestry operations) or also climate impact occurring elsewhere as a consequence of the product system, so-called indirect land use and land use change (see the earlier discussion on attributional and consequential LCA in section 2.1.1). If, for example, the forest biomass is used to produce a certain product, the demand of forest biomass increases and so does its market price, which may cause more forest biomass to be extracted elsewhere. Thus the harvest of forest biomass may cause more intensive and/or extensive forestry elsewhere with associated climate impacts. Such indirect market-driven effects can be important for carbon footprints of biofuels (Berndes et al. 2013; Kløverpris & Mueller 2013; Hertel et al. 2010; Plevin et al. 2010; Searchinger et al. 2008). Discussions regarding indirect effects most often focus on biofuels made from agricultural biomass rather than forest biomass (Ahlgren et al. 2013), due to the higher competition for agricultural biomass and the shortage of agricultural land compared to forest land. Indirect effects can, however, be increasingly important also for the climate impact of forest products due to increasing demand for forest biomass (as is further discussed in the next subsection) and because consequential studies are becoming increasingly common (indirect effects are more often captured in consequential studies, as described in section 2.1.1). It should be noted that higher demand for forest biomass and the subsequent increase of market prices for forest biomass also can have positive climate effects, as it can (i) prevent the transformation of forest land to agricultural land, and (ii) result in forestry practices that store more carbon than would otherwise have been the case (Miner et al. 2014).

It should be acknowledged that the spatial (and temporal) modelling of the carbon flows does not only concern the carbon that is captured and emitted above ground, but also fluxes of below-ground carbon, so-called soil organic carbon (SOC), which is not always accounted for but can be of considerable importance in the carbon footprints of forest products (Brandão et al. 2011; Repo et al. 2011; Stephenson et al. 2010).

To conclude, the spatial system boundaries influence the carbon footprints of forest products. The spatial resolution of system boundaries is important (e.g. the choice between a landscape and a stand perspective) as is the choice to include or exclude indirect land use and land use change. Figure 1 illustrates these spatial aspects.

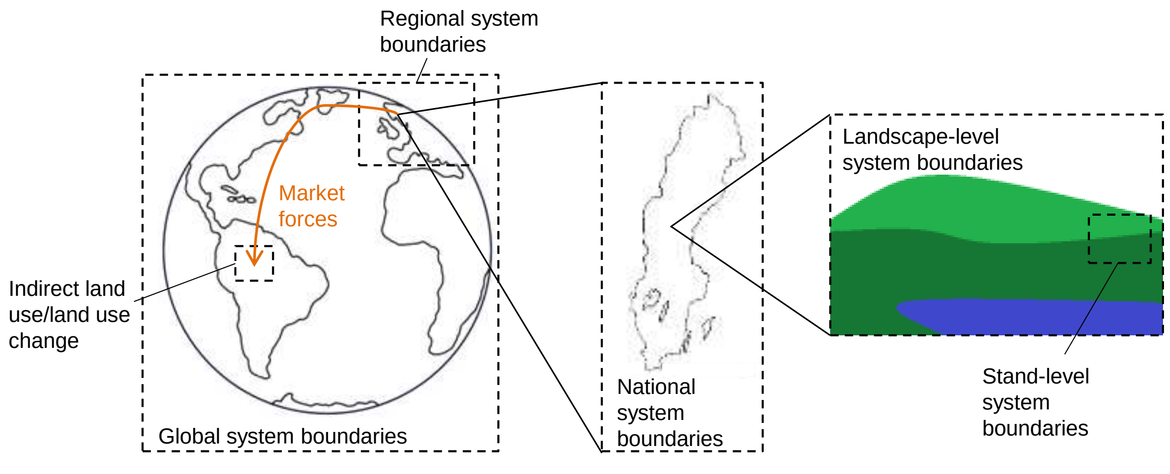


Figure 1. Visualisation of spatial aspects influencing carbon footprints of forest products.

2.2.2 Temporal aspects of modelling the carbon flows

As mentioned, the spatial and temporal system boundaries are interlinked: the carbon balance with certain spatial system boundaries depends on the temporal system boundaries, as is further described below.

An important aspect of the temporal system boundaries is whether the harvest is seen as a consequence of previous biomass growth (i.e. CO_2 has been captured in the forest in the past, and the harvest merely restores the forest to a previous state) or whether the biomass growth is seen as a consequence of the harvest (i.e. CO_2 is captured as a consequence of the harvest). In the first case, the product system started after the last previous harvest, i.e. one rotation period ago (the time period from harvest to harvest, often 50–100 years for boreal forests). By considering the CO_2 captured during this period one considers the capture of the actual carbon in the studied product. In the second case, the temporal dimension of the product system starts when the biomass used in the product is harvested.

Furthermore, it is important to emphasise that past and current net growth of forest biomass in certain areas (such as Europe) is (at least to some extent) a consequence of a recovery from previous forest management practices (Kauppi et al. 2010). This net growth will not necessarily continue once previous levels of forest biomass stocks have been attained. Indeed, in Europe there are signs of declining net growth of forest biomass (Nabuurs et al. 2013). Moreover, disturbances due to climate change, such as increased frequency of forest fires, can eventually cause decreased forest biomass in areas that at present experience an increase (e.g. boreal regions), even if the fertilising effect from a higher atmospheric CO_2 concentration is accounted for (Kane & Vogel 2009; Kurz et al. 2008). Increased demand of forest biomass, with a subsequent increase of biomass extraction, may also threaten the capacity of boreal and/or European forests to function as carbon sinks (Mantau et al. 2010; Nabuurs et al. 2007). Thus, assumptions that forest biomass from a certain region is carbon neutral may be valid assuming today's situation of forest biomass growth, but not necessarily valid in a future situation.

The *time perspective of the study* is another dimension of the temporal system boundaries that influences the carbon neutrality assumption of forest biomass. If the study aims at supporting decisions that concern short-term reduction of climate impact (e.g. based on the perspective that there is an urgent risk of irreversible climate change and that society therefore must optimise product systems for short-term climate impact reduction), it can be problematic to use forest biomass in products with a short service life that are incinerated at end-of-life. This is due to a temporal shift between the time of the incineration (and the resulting CO_2 emissions) and the time at which an equal amount of CO_2 once again has been captured and stored in the regrowing forest. Such products can, for example, be biofuels or other products with service lives considerably shorter than the forest's rotation period. This may be particularly problematic with a stand perspective; with a perspective based on a landscape (or higher) level, one can argue that as long as the carbon is in balance at a landscape (or higher) level, also forest products with short service lives are carbon neutral (considering the biogenic carbon flows; of course, there may still also be fossil GHG emissions). On the other hand, it could still be preferable to use the biomass for products with long service lives; as such products then store carbon, which contributes to short-term mitigation of climate change.

Another dimension of the study's temporal system boundaries is the *time horizon of the characterisation factors* (CFs), i.e. the time period for which one considers the cumulative change in the radiative forcing (most often expressed in terms of the GWP, which is based on the Bern carbon flux model; IPCC 2013). In LCA, a time period of 100 years is most common (Røyne et al. 2016). When comparing the contribution of different GHGs, one thus considers the cumulative climate impact occurring within 100 years. If one instead uses time horizons of 20 or 500 years, the relative importance of different GHGs changes.

An additional dimension of the study's temporal system boundaries is *the timing* of CO_2 capture and GHG emissions (fossil as well as non-fossil). Most often, the timing is disregarded and the climate impact from CO_2 capture and GHG emissions are calculated in the same manner regardless of when they occur, which may not provide an accurate representation of actual climate impacts due to two reasons: (i) the time horizon of the CFs (e.g. 100 years) is not applied consistently: for emissions occurring today, the factor accounts for impact occurring within the time horizon counting from today (e.g. within 100 years from today); but for emissions occurring X years from now, the factor accounts for impact occurring within the time period counting from the moment these emissions occur (e.g. within X+100 years from now); and (ii) risks of transgressing self-reinforcing tipping points in the climate system imply that climate impact may have to be reduced rapidly, which in turn implies that the the timing of emissions matters (Jørgensen et al. 2014; Helin et al. 2013; Levasseur et al. 2010). Another reason for why the timing matters is that a delay between the emission of a certain amount of biogenic CO_2 and the point in time when an equal amount of CO_2 has been captured in the regrowing forest contributes to a temporary increase of the radiative forcing and thus also an increase in the cumulative change of radiative forcing within a given time period (Helin et al. 2013). Thus, *carbon* neutrality does not automatically imply *climate* neutrality. A delay may also occur in the other direction: that is, the carbon is stored in a product while CO_2 is captured in the regrowing forest, which would give a beneficial climate impact compared to a reference situation in which the forest was not harvested.

To conclude, the temporal system boundaries influence the carbon footprints of forest products. This includes: (i) whether the capturing of the CO_2 is considered to occur before or after the harvest, (ii) whether the study aims for short- or long-term climate impact reduction, and (iii) whether the timing of CO_2 uptake and GHG emissions is accounted for. Figure 2 illustrates some of these temporal aspects.

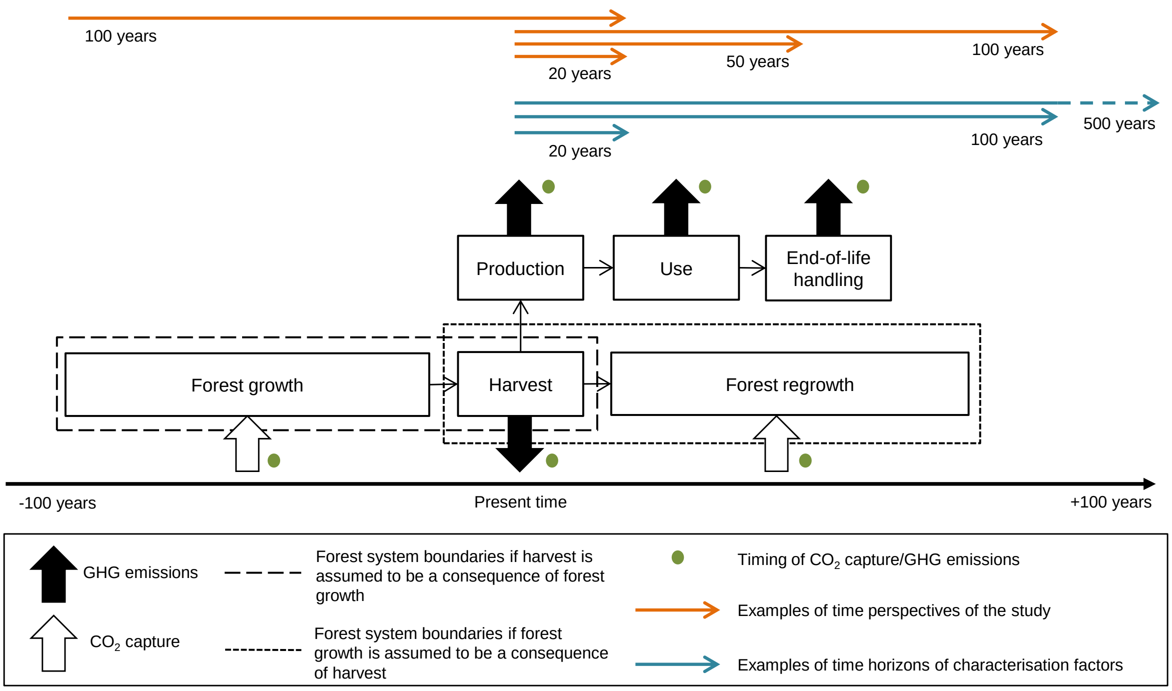


Figure 2. Visualisation of temporal aspects influencing carbon footprints of forest products.

2.2.3 The choice of baseline in modelling the carbon flows

In modelling carbon flows of forestry, another key aspect is the assumption on what happens with the forest land in the absence of the harvest associated with the studied product system, i.e. the so-called baseline or reference/counter-factual situation. This is essential for defining the environmentally relevant flows in the product system, i.e. for quantifying the inventory of the studied product system. In other words, defining a baseline is necessary to be able to separate the technosphere from natural processes (Soimakallio et al. 2015).

Some examples of baselines are illustrated in Figure 3. In the upper part of the figure, possible outcomes with stand-level system boundaries are shown (here we assume that there are net emissions of CO_2 for some years after harvest before CO_2 capture starts to dominate; as is the case in the forest carbon model applied in our case studies, see section 3.1). In the lower part of the figure, possible outcomes with a higher resolution than stand-level are shown (here we assume an annual growth of forest biomass within the defined area, see section 3.1).

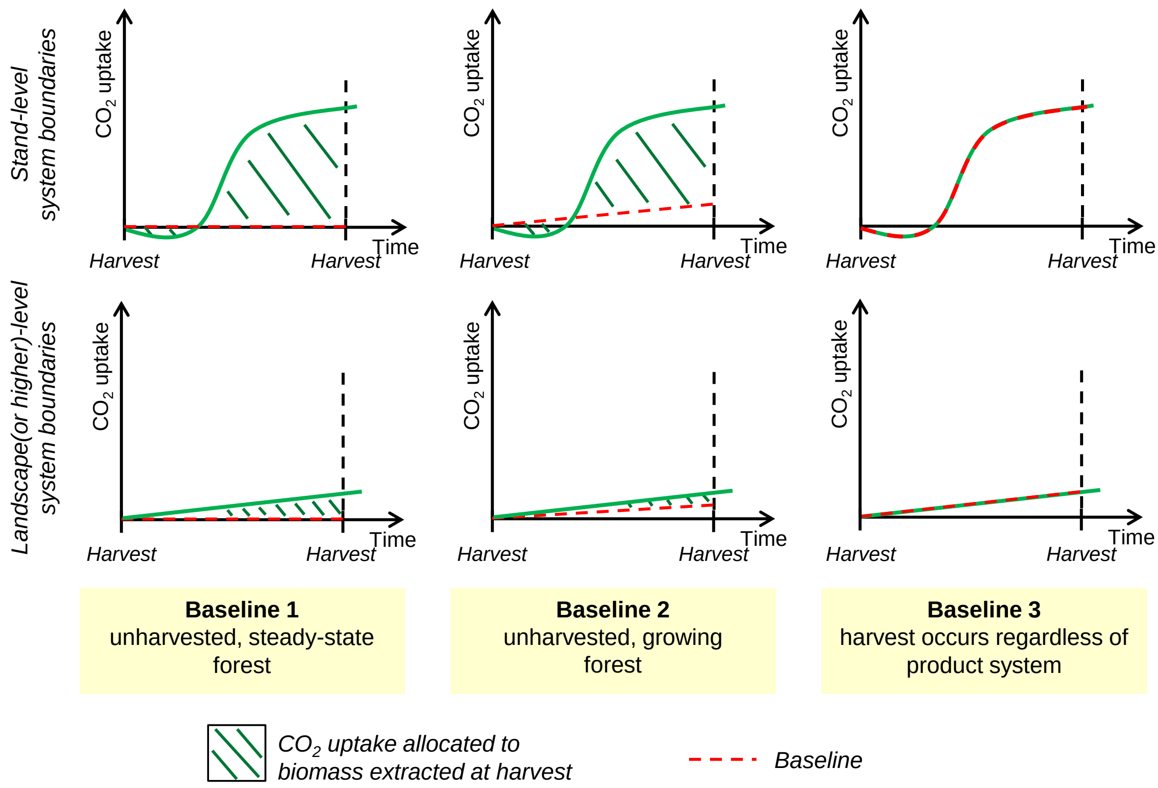


Figure 3. Visualising possible baselines in modelling the forest for carbon footprint calculations of forest products, both for stand- and higher-level system boundaries. The curves have been produced to illustrate the baseline concept, and do not reflect real data.

In Figure 3, baseline 1 means that all CO_2 captured in the forest is allocated to the harvested biomass. This could, for example, be based on the assumption that no net carbon flows occur in the absence of harvest (e.g. because CO_2 uptake in a mature, unharvested forest is balanced by the GHG emissions from the breakdown of dead wood, i.e. the forest has reach a steady-state). Baseline 2 means that one assumes that only the difference between the CO_2 captured in the forest after harvest and the CO_2 that would have been captured in an unharvested but still growing forest, is allocated to the harvested biomass. This can also be described as "using natural regeneration as the land use baseline" (Koponen and Soimakallio 2015) or as the "no-harvest baseline" (Ter-Mikaelian et al. 2015). Baseline 3 means that one assumes that harvest occurs regardless of the product system. In other words, that the harvesting of biomass for the studied product system does not influence the total amount of biomass harvested within the defined geographical system boundaries. As is seen in Figure 3, the choice of baseline can significantly influence the CO_2 uptake allocated to the studied product system, and thus it is an important factor in the inventory analysis. It should be emphasised that Figure 3 shows the role of the baseline for modelling the carbon flows, but that the choice of baseline influences also the modelling of other climate effects of forestry (e.g. those described in 2.2.4).

The choice of baseline can depend on whether the study is attributional or consequential (see section 2.1.1). Hypothetical what-if scenarios are usually not dealt with in attributional studies, and it has been argued that such studies usually do not account for baselines; however, implicitly a baseline is always assumed (Soimakallio et al. 2015). For example, ignoring the baseline is in many cases equivalent to applying baseline 1. Furthermore, it has been argued that in attributional studies, the most coherent baseline would be to assume baseline 2 (Soimakallio et al. 2015). In a consequential study, on the other hand, LCA practitioners are probably more likely to consciously apply a certain baseline, most likely baseline 1 or 2. Baseline 3 is problematic to apply in LCA, because if one assumes that the forest would be used for biomass extraction also in absence of the product system, one could argue that also other natural resources used in the product system would have been used in the absence of the product system; for example fossil resources, which would mean that GHG emissions of fossil origin should not be accounted for.

2.2.4 Non-carbon aspects of carbon footprint methodology

Nitrogen (N) fertilisation of forests causes emissions of nitrous oxide (N_2O, a potent GHG) as a by-product in the nitrification of ammonium (NH_4^+) to nitrate (NO_3^-), and as an intermediate in the denitrification of nitrate to nitrogen gas. These emissions can be direct, from the soil to which N is applied, and indirect, through the volatilisation of N as ammonia (NH_3) and oxides of N (NO_x), the deposition of these gases and their products (NH_4^+ and NO_3^-) onto soils and water bodies, and the subsequent formation of N_2O (De Klein et al. 2006; Wrage et al. 2005; Cai et al. 2001). For forest products, the climate impact of such N_2O emissions can be of the same order of magnitude as the climate impact of fossil GHG emissions from forestry operations, as shown when comparing the climate impact of N_2O emissions in Skogsstyrelsen (2015) with those from forestry operations in Berg and Lindholm (2005).

Another potentially important aspect of the climate impact of forest products relates to changes in the albedo due to land use and land use change. The albedo is the capacity of the Earth surface to reflect incoming solar radiation. In case clear-cutting is applied, or in case of deforestation, the albedo can increase and cause a beneficial climate effect. This effect is enhanced by snow and is therefore particularly strong at northern latitudes (Cherubini et al. 2012; Schwaiger & Bird 2010). Whether to consider albedo effects can also depend on the spatial and temporal system boundaries and the choice of baseline. For example, the choice between stand and landscape perspectives matter: at the stand level, a harvest may cause a significant change of the albedo, whereas no change is seen at the landscape level.

Reflection of solar radiation can also be influenced by the forest's ability to form organic vapours, a process which can be influenced by land use and land use change. These vapours can form aerosols that reflect sunlight. Also, the aerosols can aggregate into particles catalysing the formation of clouds, which in turn reflect sunlight. This effect can be significant in relation to other climatic effects of forests (Spracklen et al. 2008). Once again, whether to account for this effect can depend on the spatial and temporal system boundaries and the choice of baseline.

Figure 4 illustrates the other aspects influencing carbon footprints of forest products.

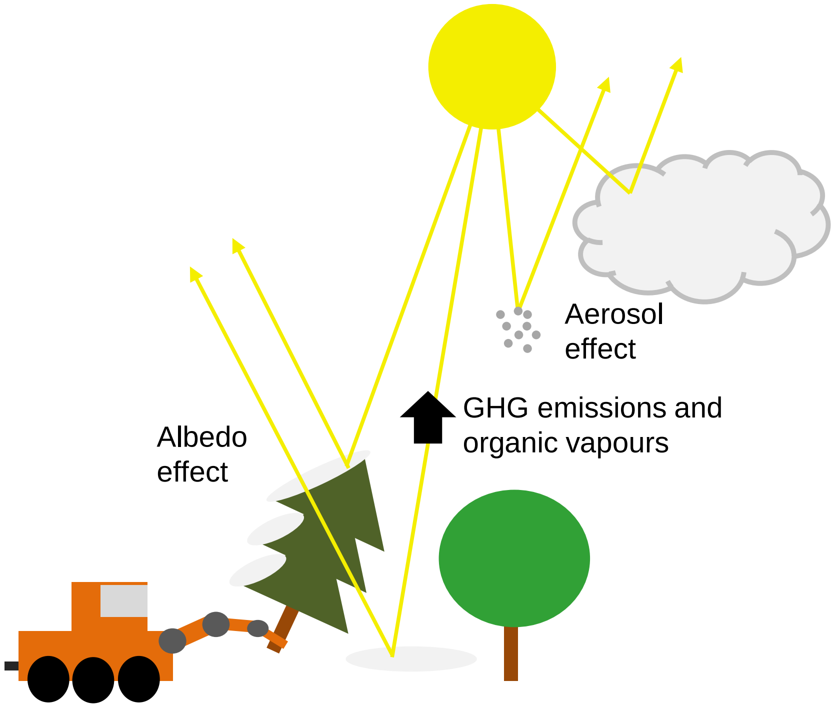


Figure 4. Visualisation of three non-carbon aspects influencing the carbon footprints of forest products: changes in the albedo, the formation of aerosols that directly or indirectly (through cloud formation) reflects solar radiation, and non-carbon GHG (e.g. CH_4 and N_2O) emissions.

2.3 SELECTED CLIMATE IMPACT ASSESSMENT METHODS

There is not one method for assessing the climate impact of forest products that everyone uses. As described in the previous section, this is both because of limitations in the understanding of, and ability to model, the climate impact of forest products, and because of difficulties to agree on value-based choices, which for example relate to what we believe fall under our responsibility. The lack of established methods places the LCA practitioner in a situation where he or she can and must choose between various methods and various ways of applying the chosen method (i.e., the practice). The choice can be influenced or guided by: (i) approaches traditionally used in LCA, (ii) requirements in standards, directives or other consensus documents (hereafter termed "requirements"), or (iii) new methods proposed in the scientific literature. As requirements are the outcome of often long consensus processes, they seldom reflect the state of-the-art knowledge in the field, and methods reflecting new knowledge found in scientific literature could eventually be included in requirements. In this study, we selected one method for each of the three above rationales for method selection (i-iii), in order to explore and clarify the similarities and differences between the rationales.

Below we describe how the methods were identified and selected. Some of the methods are part of wider methodological frameworks, not only influencing the carbon footprinting, but also the general modelling of the product system (e.g. in terms of allocation methods). In such cases, we have only considered what the methods prescribe in terms of carbon footprinting, and disregarded prescriptions in terms of the general system modelling. This was done in order to test and compare different means for carbon footprinting, rather than different means for system modelling. Considering the scope of this report, it makes sense to keep variables not related to carbon footprinting constant.

2.3.1 Traditional LCA practice

The most common way to assess the climate impact of forest products, the "traditional LCA practice", was identified through a literature study of peer-reviewed LCAs of forest products, published 1997-2013. A range of product categories were represented among the LCAs: fuel and transportantion fuels, construction materials, plastics, chemicals and other commodities. The literature study is presented in Røyne et al. (2016). From the literature review it was apparent that the choice of carbon footprint methodology is seldom adapted to the specific product category (forest products).

The characteristics of the carbon footprint methodology used in traditional LCA practice are summarised in Table 2, and further described in below paragraphs. The characteristics are structured according to the aspects of carbon footprints described in section 2.2. Noteworthy is that some aspects are strongly connected; for example, the choice of spatial system boundaries determines whether the forestry operations are carbon neutral or nor within the spatial system boundaries.

Table 2. Characteristics of the traditional LCA practice for carbon footprints of forest products. The characteristics are structured according to the aspects of carbon footprints described in section 2.2.

	Carbon footprint aspects	How the aspects are treated in the traditional LCA practice
Spatial system modelling	Geographical system boundaries	Unclear. Biogenic CO_2 from the combustion of forest biomass is assumed to be climate neutral and thus given a GWP-value of zero. This indicates a landscape perspective (or biomass growth in the past).
	Indirect land use change	Disregarded
Temporal system modelling	Forest regrowth in the past/future	Unclear. The climate neutrality assumption indicates that it is assumed that the planting of forest was done in the past with the purpose of harvest (or landscape perspective), or that future CO_2 uptake is assumed to occur within a sufficiently short time period to be considered part of the product system.
	Based on an aim for short or long-term impact mitigation	Unclear. But long term is implied by the time horizon of the CFs and the climate neutrality of biogenic CO_2 emissions.
	Time horizon of CFs	100 years
	Timing of climate impact	Disregarded
Baseline	Steady-state forest, growing forest, or harvest occurs regardless of produc system	Not applicable, as climate impact from changes in the carbon balance are ignored (yields the same result as baseline 3 in Figure 3).
Non-carbon aspects	N_2O emissions from land use (fertilisation)	Depends on LCI data, but normally disregarded
	Albedo effects	Disregarded
	Aerosol effects	Disregarded

The traditional LCA practice assumes that biogenic CO_2 from the combustion of forest biomass is climate neutral and thus given a GWP-value of zero. This is based on an assumption of carbon neutrality, which either represents a landscape perspective (i.e., at a landscape level the forest biomass is carbon neutral) or a long-time perspective (i.e., in the long run, the forest biomass is carbon neutral). The second perspective is based on one of two viewpoints: (i) that the planting of the forest (in boreal forests: 50-100 years ago) was done with the purpose of harvesting, and is thus part of the product system; or (ii) that climate change is a long-term issue and thus future CO_2 uptake (in boreal forests: 50-100 years into the future) occurs within a sufficiently short time period to be considered part of the product system. In either case, the traditional LCA practice assumes that at some time and/or at some place, an amount of CO_2 is captured that equals the biogenic CO_2 emissions released at end-of-life, and that the possible temporal shift between these activities does not cause a climate impact.

Moreover, the traditional approach does not account for most of the other aspects of the climate impact of forest products described in section 2.2, such as the timing of emissions, indirect land use change, the forests' influence on the formation of aerosols and changes in the albedo.

For other GHGs, the traditional LCA practice uses the GWP_{100} CFs. In the present study, we use the GWP_{100} CFs available in the Gabi Professional LCA software (Thinkstep 2015), as using the CFs available in a software reflects the most common LCA practice; these are, for methane (CH_4) 25 kg CO_2 eq./kg and for N_2O 298 kg CO_2 eq./kg. These numbers differ from the latest numbers from IPCC, which are 28 and 265 kg CO_2 eq./kg for CH_4 and N_2O, respectively (Myhre et al. 2013).

2.3.2 Method from requirements in standards and directives

Before selecting methods from requirements outlined in standards or directives, several methods were examined, including general and product-specific standards and directives. Among general requirements, we examined the international reference life cycle data system (ILCD) handbook (EC 2010a), the publicly available specification (PAS 2050) for the assessment of the life cycle GHG emissions of goods and services (BSI, 2011), the PEF (EC 2013) and the GHG protocol (Greenhouse gas protocol 2012). Among product-specific requirements, we examined the EU sustainability criteria for biofuels and bioliuqids (EC 2009), which is used under the renewable energy directive (RED), the PCRs for the environmental product declaration (EPD) of construction products and services (EPD 2012), and the core rules for construction product EPDs (CEN 2012), and the PCRs basic module for pulp, paper and paper products (EPD 2011).

We decided to apply different requirements for the fuel and non-fuel case studies. For fuels, we selected the EU sustainability criteria for biofuels, because this is the requirement used for calculating the climate performance of biofuels within RED. For non-fuel products, we selected the PEF requirements, developed by the EC in order to unify LCA practice. This was chosen since it is based on a range of previous consensus processes (the ILCD handbook, the ecological footprint standard, the GHG protocol, and PAS 2050), and since it is relatively recently developed (2013). Below, these requirements are further described.

EU sustainability criteria for biofuels

The characteristics of the carbon footprinting methodology required by the EU sustainability criteria for biofuels are summarised in Table 3, and further described in below paragraphs.

Table 3. Characteristics of the carbon footprinting methodology required by the EU sustainability criteria for biofuels. The characteristics are here structured according to the aspects of carbon footprints described in section 2.2.

	Carbon footprint aspects	**How the aspects are treated in the EU sustainability criteria for biofuels**
Spatial system modelling	Geographical system boundaries	Not clearly stated
	Indirect land use change	Disregarded
Temporal system modelling	Forest regrowth in the past/future	Not clearly stated
	Based on an aim for short or long-term impact mitigation	Not clearly stated
	Time horizon of CFs	100 years
	Timing of climate impact	Disregarded
Baseline	Steady-state forest, growing forest, or harvest occurs regardless of produc system	Not applicable, as climate impact from changes in the carbon balance are ignored (unless land use change occurs)
Non-carbon aspects	N_2O emissions from land use (fertilisation)	Considered (but only in the case of fertilizing, which we assume not to occur in the product systems of this study)
	Albedo effects	Disregarded
	Aerosol effects	Disregarded

The rules in the EU sustainability criteria for biofuels for calculating the climate impact of biofuels are outlined in Annex V, part C, of the RED (EC 2009). The calculating procedure is as follows:

$$E = e_{ec} + e_l + e_p + e_{td} + e_u - e_{sca} - e_{ccs} - e_{ccr} - e_{ee} \qquad \textit{Eq. 1}$$

where

E = total emissions from the use of the fuel;

e_{ec} = emissions from the extraction or cultivation of raw materials;

e_l = annualised emissions from carbon stock changes caused by land-use change;

e_p = emissions from processing;

e_{td} = emissions from transport and distribution;

e_u = emissions from the fuel in use;

e_{sca} = emission saving from soil carbon accumulation via improved agricultural management;

e_{ccs} = emission saving from carbon capture and geological storage;

e_{ccr} = emission saving from carbon capture and replacement; and

e_{ee} = emission saving from excess electricity from cogeneration.

Based on the above described rules, a number of default values have been produced for some general fuel types (see EC 2009), expressed both in terms of g CO_2 eq./MJ biofuel and as emission savings in relation to fossil references (in %). As the fuels of our case studies (lignin-based fuel and butanol) do not belong to any of these predefined fuel types, we instead calculated numbers using the above procedure. To proceed, some details of the procedure need to be further elaborated on.

In the LCI data we use, e_{ec}, e_p, e_{td}, and e_u are accounted for (see chapter 3). According to the rules, e_u should be assumed to be zero for biogenic CO_2 emissions (EC 2009). Furthermore, land-use change refers to changes in land cover between the following six land categories: forest land, grassland, cropland (distinguishing between regular cropland and cropland for cultivating perennial crops such as short-rotation coppice and oil palm), wetlands, settlements, and other land (EC 2010b). This means that e_l can be disregarded for our product systems. Also e_{sca}, e_{ccs}, e_{ccr} and e_{ee} can be disregarded due to the nature of our product sytems. Furthermore, EC (2009) prescribes that CFs for CO_2, CH_4 and N_2O should be 1, 23 and 296 kg CO2 eq./kg, respectively (which reflects an old version of the IPCC's GWP_{100} CFs, see section 2.3.1).

In Alberici and Hamelinck (2010) – which provides a guide for how to implement the above calculation procedure – it is clear that one should account for N_2O field emissions from N fertilisers. Then, the question arises whether or not we should assume that fertilisation occurs, and thus include field emissions of N_2O (they were not included in Berg and Lindholm (2005), one of the datasets used to model the GHG emissions from forestry operations, see section 3.1). We have decided to assume that there is no fertilisation in our case studies, because of two reasons. First, fertilisers have been applied to only about 10% of forest land in Sweden, thus most harvested forest biomass is from non-fertilised land (Skogsstyrelsen 2015). Second, fertilisation also has other effects that influence the carbon footprint, which then would have to be considered as well: fertilisers increase the productivity of land (with higher sequestration of SOC) and reduce the rotation period (to 35-55 years in Sweden), which can more than offset the climate impact from the emissions of N_2O (Skogsstyrelsen 2015). These factors would then have to be accounted for also when applying the other carbon footprint methods, in order to yield comparative scenarios, an exercise that is deemed to be outside the scope of this report. Finally, it should be emphasised that for site-specific LCAs concerned with areas where fertilisers are applied, field emissions of N_2O should be accounted for – this can, for example, become more common if short-rotation forestry expands in Sweden (Larsson et al. 2009).

In summary, for the case studies of this report, the values of the GWP_{100} CFs are the only difference between the carbon footprint methodology prescribed by the EU sustainability criteria for biofuels and the traditional LCA practice for carbon footprinting.

Product Environmental Footprint – PEF

PEF is a guide developed by the EC's environment directorate and the EC's Joint Research Centre (JRC IES) aiming to harmonize the methodology for calculating the environmental footprint of products (EC 2013). The scope of the guide includes all kinds of products and impact categories, and although PCRs for specific types of products are planned, they are still in the development phase. Consequently, the recommendations for dealing with some of the aspects of carbon footprint studied in the present report are broadly discussed. The guide refers to the LCI data as a "resource use and emissions profile", which is defined as all material and energy inputs, outputs and emissions for the product's supply chain. Then, it states that classification and characterization of the flows identified in this profile are compulsory. The characteristics of the carbon footprint methodology of recommended by the PEF are summarised in Table 4, and further described in below paragraphs.

Table 4. Characteristics of the PEF guide for carbon footprints of forest products. The characteristics are structured according to the aspects of carbon footprints described in section 2.2.

	Carbon footprint aspects	How the aspects are treated in the PEF guide
Spatial system modelling	Geographical system boundaries	Not clearly stated
	Indirect land use change	Disregarded
Temporal system modelling	Forest regrowth in the past/future	Not clearly stated
	Based on an aim for short or long-term impact mitigation	Not clearly stated
	Time horizon of CFs	100 years
	Timing of climate impact	Disregarded
Baseline	Static system, or growing unharvested forest, or harvest occurs anyway	Not applicable, as climate impact from changes in the carbon balance are ignored (unless land use change occurs)
Non-carbon aspects	N_2O emissions from land use (fertilisation)	Disregarded
	Albedo effects	Disregarded
	Aerosol effects	Disregarded

The PEF guide assumes that biogenic CO_2 emissions from the combustion of forest biomass are climate neutral, and the emissions are thus given a GWP-value of zero. Biogenic CO_2 emission and removals should be reported, but not included in the impact assessment. Timing of emissions is not mentioned or dealt with in the PEF guide. The guide prescribes that the time horizon of CFs should correspond to the IPCC factors for GWP_{100} from 2007.

The guide defines land use change as the climate impacts resulting from changes in land carbon stocks transformation from one land use type to the other. However, the product systems we studied extract raw materials only from land which has been used for forestry for years, meaning that there is no transformation to the other types of land use established in the guide. Therefore, we interpret it as there is no land use change in our studied systems if the PEF guide is followed. When it comes to ILUC, the guide specifies that it shall not be considered unless explicitly required by the PEF category rules (PEFCRs). As the first version of the PEFCR is currently under development, we disregarded ILUC.

Changes in soil carbon stocks are only mentioned in the guide in connection to land use change. As discussed above, there is no land transformation in any of our studied systems, meaning that SOC should not be accounted for according to the PEF.

2.3.3 Advanced method

We define "Advanced methods" as methods available in the peer-reviewed scientific literature, but not yet included in any requirements in standards, directives or some other consensus documents. Therefore, advanced methods do not offer complete guides but rather suggestions related to specific methodological aspects. New such methods to capture the aspects described in section 2.2 are constantly being developed.

We examined three suggested methods for capturing the temporal dynamics of forest products, which we believe to be perphaps the most important dimension of carbon footprinting that is ignored by established methods and practices. The first method, developed by Cherubini et al. (2011), assigns CFs to biogenic CO_2 emissions (so-called GWP_{bio} CFs) based on the rotation period of the cultivation from which the carbon originates and the time period that the carbon is stored before it is incinerated. In the second method, developed by Costa and Wilson (2000), an equivalence factor between avoided emissions of CO_2 and sequestration is given. This offers a way to quantify the temporal value of storing carbon. The third method, dynamic LCA, developed by Levasseur et al. (2010), defines different GWPs for each GHG emission pulse (positive or negative, i.e. GHG emissions or GHG sinks) occurring in a product system within a given time horizon.

For this study, the dynamic LCA method was selected, since it can account for several of the carbon footprint aspects described in section 2.2, and because it can be combined with existing methods for modelling the dynamic carbon flows of forests (which thus provides LCI data that fits the LCIA method prescribed by the carbon footprint method). The characteristics of the dynamic LCA method are summarised in Table 5, and further described in below paragraphs.

Table 5. Characteristics of the dynamic LCA method for carbon footprint of forest products. The characteristics are structured according to the aspects of carbon footprints described in section 2.2.

	Carbon footprint aspects	How the aspects are treated in the dynamic LCA method
Spatial system modelling	Geographical system boundaries	Depends on the forest carbon model (in this study we apply three different models corresponding to three different geographical system boundaries)
	Indirect land use change	Not in the scope of dynamic LCA, but possible to combine with methods for accounting for indirect effects
Temporal system modelling	Forest regrowth in the past/future	Either scenario can be chosen (in this study, we assumed future regrowth)
	Based on an aim for short or long-term impact mitigation	Can be adapted to either aims (as any time horizon can be chosen)
	Time horizon of CFs	Any time horizon possible (in this study, we compare two time horizons: 20 and 100 years.
	Timing of climate impact	Considered
Baseline	Steady-sate forest, growing forest, or harvest occurs regardless of produc system	Depends on the forest carbon model (in this study, we used forest data derived using a steady-state baseline, i.e. baseline 1 in Figure 3).
Non-carbon aspects	N_2O emissions from land use (fertilisation)	Can be considered (but we assume it not to occur in the type of product systems treated in this study since we assume no fertilizing)
	Albedo effects	Not in the scope of dynamic LCA, but possible to combine with methods for accounting for albedo effects
	Aerosol effects	Not in the scope of dynamic LCA, but possible to combine with methods for accounting for aerosol effects

Dynamic LCA assigns GWP factors to all emission pulses in a product system (regardless of whether they are of fossil or biogenic origin) depending on their contribution to the increased radiative forcing within a given time period. For example, if a time horizon of 100 years is chosen, an emission pulse in year 0 has the same CF as with the traditional GWP_{100} method, but an emission pulse in year 0+n (n>0) will have a lower CF. And if n>100, the CF will be zero, as such an emission pulse does not contribute to any warming within the 100 year time horizon. Likewise, a carbon sink in year 0 (e.g. due to CO_2 capture in a forest) is given a larger negative CF than if the same carbon sinks occur in year 0+n. In this way, the method captures the timing of CO_2 capture and GHG emissions (fossil and non-fossil), the potential climate impact (both positive and negative) of biogenic CO_2 emissions (if there is a time shift between the biogenic CO_2 emission and the CO_2 uptake in the forest) and the potential climate benefit from carbon stored in products (the later a CO_2 emission occurs, the smaller the GWP value assigned to it). Also, the method enables LCA practitioners to be consistent in terms of the chosen time horizon, i.e. only climate impact occurring within the chosen time horizon is accounted for regardless when the emission pulse takes place. Finally, dynamic LCA allows the LCA practitioner to align the temporal system boundaries of the study with the time horizon of the characterisation factors. The other methods mentioned in the previous paragraphs were not chosen, as they do not offer all these advantages.

In this study, we combined dynamic LCA with three different spatial system boundaries, based on national, landscape and stand perspectives (as further described in section 3.1.2).

2.3.4 Summary of selected LCIA methods

Table 6 summarises Tables 2-5.

Table 6. Characteristics of the carbon footprint methods used in the present study.

	Carbon footprint aspects	**Traditional LCA practice**	**EU sustainability criteria for biofuels**	**PEF**	**Dynamic LCA**
Spatial system modelling	*Geographical system boundaries*	Unclear	Not clearly stated	Not clearly stated	Depends on forest model
	Indirect land use change	Disregarded	Disregarded	Disregarded	Not in the scope
Temporal system modelling	*Forest regrowth in the past/future*	Unclear	Not clearly stated	Not clearly stated	Depends on forest model
	Based on an aim for short or long-term impact mitigation	Long term	Long term	Long term	Can be adapted to either aim
	Time horizon of CFs	100 years	100 years	100 years	Any time horizon possible
	Timing of climate impact	Disregarded	Disregarded	Disregarded	Considered
Baseline	*Steady-sate forest, growing forest, or harvest occurs regardless of produc system*	Not applicable	Not applicable	Not applicable	Depends on forest model
Non-carbon aspects	*N_2O emissions from land use (fertilisation)*	Unclear	Case-specific	Case-specific	Depends on forest model
	Albedo effects	Disregarded	Disregarded	Disregarded	Not in the scope
	Aerosol effects	Disregarded	Disregarded	Disregarded	Not in the scope

3 MODELLING OF PRODUCT SYSTEMS

This chapter first describes some general modelling assumptions, then the forest modelling (the life cycle phase all products systems have in common), and then the modelling of the five product systems.

For all product systems, economic allocation was used in the modelling of the foreground processes. This was done as multi-functional processes in the product system often concern both material and immaterial products, which made it difficult to apply allocation based on physical attributes. As allocation is not in focus in the present study, we did not test the influence of different allocation procedures. It should be emphasised, however, that allocation is an important issue that can considerable influence the results in LCAs of forest products (Sandin et al. 2015).

For the modelling of the forest product systems, the CO_2 captured in the forestry was balanced to match the biogenic CO_2 emissions throughout the product life cycles (except for the scenario where dynamic LCA was combined with national-level system boundaries). Related to this, all biogenic CO_2 emissions in the forest product life cycles were assumed to originate from Swedish forest biomass. This is a simplication, as some of the biogenic CO_2 emissions may, for example, originate from some non-forest biogenic feedstock grown elsewhere (e.g. a fraction of the biogenic CO_2 emission from transportation within each product system may be from ethanol produced from agriculture crops abroad). This simplification was deemed reasonable considering the goal and scope of our study, and that only a small fraction of biogenic CO_2 emissions can be expected to be of non-forest origin.

GHG emissions other than CO_2, CH_4 and N_2O are not shown in the LCI tables in below subsections, as their contribution in terms of CO_2 equivalents were several order of magnitudes lower, and thus insignificant for the goal and scope of this study. Also, several of the product systems represent emerging technologies, and the LCI data is therefore highly uncertain. Thus, the data should not be assumed to be representative for the future full-scale implementation of the technologies, and not be used outside the scope of this report.

3.1 FOREST MODELLING

Below, the forest and forestry modelling is decribed; first the data for forestry operations, needed for all the tested carbon foorptint methods, and then the dynamic model needed for dynamic LCA. For all cases, we did not distinguish between biomass types. That is, we assumed that emissions from forestry are independent of the harvested biomass type, and that they scale with mass. This is a simplified way of avoiding allocation by dividing into subprocesses, the preferred method for handling multi-functionality in LCA (ISO 2006b; it should be noted that this is, in this case, equivalent to mass-based allocation). Also, we assumed that all biomass used in our product systems are from the final harvest; in reality, some fractions are more likely to come from, for example, thinnings compared to other fractions. This was a simplification made in order to enable us to use the available dynamic forest models, which was deemed acceptable considering the aim of our study.

3.1.1 Forestry

The LCI data for the forestry is from Berg and Lindholm (2005). This data represent forestry operations in Sweden in 1996-1997, including seedling production, silviculture, logging and transportation to forest industries. This was used due to a lack of available more recent data. Today, GHG emissions from forestry operations can be expected to be lower due to more efficient machinery and transportation, so the used dataset should be consider a worst case scenario.The data is shown for each product system in sections 3.2-3.6.

3.1.2 Dynamic forest model

Dynamic LCA requires detailed LCI data of the timing of GHG emissions and CO_2 capture in the studied product system. In the product systems studied in the present study, some of the GHG emissions and CO_2 capture take place during forest growth. In our study, we have tested three ways of modelling forest growth: with national-, landscape-, and stand-level system boundaries. In all cases, we use a steady-state baseline (baseline 1 in Figure 3), i.e. we assume that all CO_2 captured in the forest is allocated to the harvested biomass.

For modelling the LCI data with the national- and landscape-level system boundaries, we assume that the CO_2 uptake occurring during one year within the geographical system boundaries is allocated to the biomass harvested during the same year (for a further explanation of these system boundaries, see section 2.2.1). Thus the CO_2 uptake is simply negative emission pulses in year 0, and then the method of dynamic LCA assigns credits to the product system the further the harvested carbon is stored before released to the atmosphere. For the national-level system boundaries, we used data from Skogsstyrelsen (2014, p.156) which says that the ratio between annual growth and gross fellings is about 120:90. Thus, with the national-level system boundaries, we assume that the CO_2 capture in year 0 is 33% higher than the biogenic CO_2 emission in the forest product life cycles. For the landscape-level system boundaries, we assume an annual carbon balance at the landscape level; i.e. in the year of harvest, the amount of CO_2 captured corresponds to the amount of carbon embedded in the harvested biomass.

For modelling the LCI data with stand-level system boundaries, one has to use data from dynamic models on the carbon flows in forests as LCI data, as done by Levasseur et al. (2012) and Fouquet et al. (2015). Possible models to use are Helin et al. (2015), who used an unharvested, growing forest as a baseline (i.e., baseline 2 in Figure 3), or Kilpeläinen et al. (2011), who allocated all CO_2 captured in the forest during the rotation period to the forest product (i.e., baseline 1 in Figure 3). In the present study, the dynamic carbon footprinter software tool (DynCO2) from CIRAIG (2013) was used to calculate the inventory data necessary for applying dynamic LCA, using data from the forest carbon model developed by Kilpeläinen et al. (2011) as input. Thus baseline 1 in Figure 3 is used. Moreover, forest growth is assumed to occur after harvest. The used model corresponds to a forest stand in southern Finland, where growth conditions and harvesting practices are comparable to those in Sweden. The assumed harvesting period in this model is 80 years, and the data used corresponds to traditional timber (Figure 3a in Kilpeläinen et al. 2011) and bioenergy production (Figure 3c in Kilpeläinen et al. 2011) regimes. A yearly net carbon sequestration per forest hectare was obtained from this data as the difference between the total CO_2 capture and the losses to decomposition, while the forest area required to produce the biomass input for each of the studied products was calculated using above-ground productivity (m^3/ha*year) values for each region where the forestry takes place in each product system (SLU 2013b). Also, Kilpeläinen et al. (2011) includes data for soil carbon changes.

3.2 BUTANOL (AUTOMOTIVE FUEL)

3.2.1 Introduction

The butanol case study was developed in the project Skogskemi (in English: "Forest chemistry"; Vinnova 2014), financed by Vinnova, a Swedish government agency that administers state funding for R&D. The purpose of the project was to explore the possibility of exchanging fossil raw materials for bio-based raw materials and thereby strengthening the long-term sustainable and competitive production for two of Sweden's primary industries: the forest industry and the chemical industry (Joelsson 2014).

Butanol is generally used as an industrial solvent in products such as lacquers and enamels, but it can also be used as a vehicle fuel. There is no commercial butanol fuel production today. Butanol as a fuel has several benefits, such as a higher energy content than ethanol. 85% butanol/gasoline blends can be used in unmodified petrol engines vehicles (in the present study, we assume 100% butanol). The type of butanol explored in the project is n-butanol (normal butanol).

3.2.2 Functional unit

The chosen functional unit is a *European average passenger car driven 1 km*, and the reference flow is thus the fuel needed to supply this functional unit.

3.2.3 System description

The production pathway pursued in the project was to produce butanol via ethanol made of pulp wood, with subsequent processing to butanol via acetaldehyde. Data for the ethanol and acetaldehyde production was supplied by SEKAB, whereas the data for the subsequent processing was supplied by Perstorp. Further description of the production pathway can be found in Joelsson (2014). LCI data (input and output types, but not volumes) for the ethanol productionare listed in Røyne et al. (2015). Three ethanol production pathways are explored, but the one selected for this study is the SEKAB production in Örnsköldsvik.

Conversion of fossil based ethanol to acetaldehyde is already performed in a large scale production by the company SEKAB, which is the European market leader for acetaldehyde. Inventory data was collected from SEKABs existing production facility in Örnsköldsvik. The acetaldehyde was assumed to be sold and transported to Stenungsund for further valorisation towards n-butanol. Commercial production of crotonaldehyde exists in Europe, but further processing into n-butanol is not done today. Crotonaldehyde was assumed to be produced at the Stenungsund site. LCI data for the processes is confidential.

The Ecoinvent dataset on distribution from the refinery to the petrol station for the gasoline benchmark (see section 3.2.5) was assumed also for the butanol. Based on the heating value of butanol compared to gasoline (80%), we assume that the fuel consumption is 1.25 times that of the EURO5 car: 0.0675 kg/km. Combusting 1 kg of n-butanol results in 2.38 kg CO_2, which is therefore assumed as the level of exhaust emissions.The production route of the case study is illustrated in Figure 5. Further description of the production pathway can be found in Joelsson (2014).

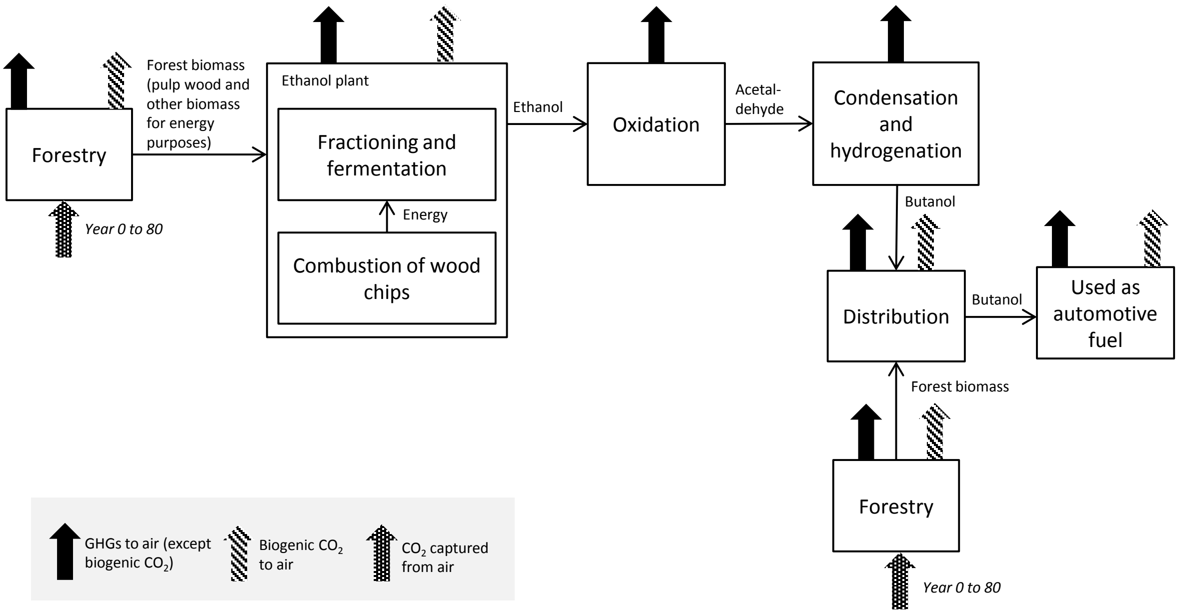


Figure 5. Process flowchart for the butanol product system, including the relevant flows for the carbon footprint.

3.2.4 LCI table

Table 7 shows the LCI data of the butanol production system. The CO_2 uptake during forest growth accounts for both the carbon embedded in the materials and the biomass used for energy purposes in production.

Table 7. LCI data for the n-butanol, per functional unit (1 km driven). The production phase includes emissions in forestry and distribution of the fuel.

LCI data / **Process**	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Forest growth	0.20	0	0	0	0
Production	0	0.061	0.034	0.00055	0.0000021
Use/end-of-life	0	0	0.16	0	0

3.2.5 Benchmark products

Gasoline and diesel were chosen as benchmarks. These product systems were modelled to enable comparisons with the butanol and lignin-based fuels, thus we included the production of the fuel and distribution (i.e. well-to-tank) and exhaust emissions (i.e. tank-to-wheel).

For the gasoline benchmark, we selected an Ecoinvent dataset on European average low-sulphur gasoline, including distribution (as described in section 3.3.3), combined with the operation of a Swiss average EURO5 gasoline passenger car (fuel consumption: 0.054 kg/km). See Spielmann et al. (2007) for further details.

For the diesel benchmark, we used an Ecoinvent dataset on European average low-sulphur diesel, including distribution, combined with the operation of a Swiss average EURO5 diesel passenger car (fuel consumption: 0.053 kg/km). See Spielmann et al. (2007) for further details.

Table 8 shows the LCI data of the gasoline and diesel product systems. Biogenic CO_2 emissions were disregarded as they are insignificant in relation to the fossil CO_2 emissions.

Table 8. LCI data for the gasoline and diesel, benchmarks in the case study of the lignin-based automotive fuel, per functional unit (1 km with a European average car).

LCI data / **Process**	*Fossil CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Gasoline (well-to-tank)	0.036	0.0000055	0.0000013
Gasoline (tank-to-wheel)	0.17	0.00011	0.00000057
Diesel (well-to-tank)	0.025	0.0000027	0.00000056
Diesel (tank-to-wheel)	0.17	0.000095	0.00000047

3.3 LIGNIN-BASED FUEL (AUTOMOTIVE FUEL)

3.3.1 Introduction

The case study of the lignin-based fuel is based on LCA work carried out in the GreenGasoline project (Mistra Innovation 2012). The project lasted from 2012 to 2014 and concerned the development of a process, GreenGasoline, for recovering lignin from the black liquor stream of a sulphate pulp mill and purifying it into a gasoline-type automotive fuel (of confidentiality reasons, we can not five further details on the nature of the fuel). The commissioner and financer of the project was the Swedish Foundation for Strategic Environmental Research, Mistra, with co-financing from project partners. After the end of the project, funding was secured for further development of the process, which is currently ongoing. Thus the product represents a novel way of utilising a side stream in the Swedish biomass industry, for which there are expectations on future full-scale production in Sweden, and therefore it was deemed a relevant case study for the goal and scope of the present study. Fossil gasoline and diesel were selected as benchmark products.

3.3.2 Functional unit

As for the butanol case, the functional unit is a *European average passenger car driven 1 km*, and the reference flow is thus the fuel needed to supply this functional unit.

3.3.3 System description

The GreenGasoline process is a process for recovering lignin from the black liquor stream of a sulphate pulp mill and purifying it into a precursor (depolymerised lignin) that can be further processed (e.g. in a conventional petroleum refinery) into an automotive fuel. The LCA model covers all production processes until the final automotive fuel, and the subsequent distribution and use of that fuel. The process was designed for Swedish conditions, for integration of the process into an existing, Swedish, sulphate pulp mill, and the LCA model is thus also modelled for Swedish conditions. The final fuel can be blended with gasoline and run in ordinary automotive vehicles (although, for calculation reasons, in the present study we assume 100% lignin-based fuel; i.e. we study the substitution effect of replacing a fossil fraction of conventional fuels with a bio-based fraction). Figure 6 shows the process flowchart, including the relevant flows for the carbon footprint. GaBi 6 was used for modelling the product system (Thinkstep 2015). As the process is still under development, it is subject to secrecy agreements; therefore, the production processes and the LCA model could not be described in more detail than presented below. Also, the early development phase means that the LCA model and the resulting carbon footprint should not be considered to be representative for any fuel that eventually reaches the market.

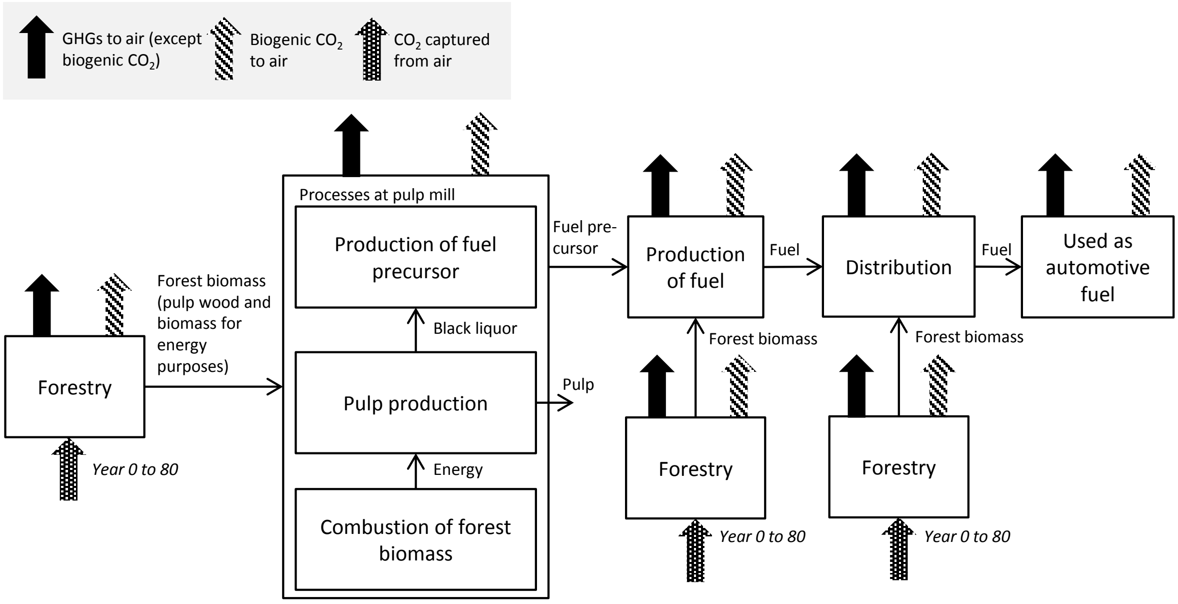


Figure 6. Process flowchart for the lignin-fuel product system, including the relevant flows for the carbon footprint.

The processes at the pulp mill are modeled based on a traditional sulphate pulp mill and data on additional emissions due to the conversion of this traditional sulphate pulp mill into a biorefinery also producing the fuel precursor, which were estimates calculated in the GreenGasoline project. The fuel precursor production includes lignin recovery through membrane separation, lignin purification, lignin blending and fluid catalytic cracking. The lignin is, in a conventional pulp mill, used to produce energy for the pulping process. With production of the fuel precursor, this energy needs to be produced by increased combustion of some other fraction of forest biomass. This other means will most probably be some forest residue, such as bark or GROT (branches and tops). In this study we assumed wood chips, due to a lack of data for bark or GROT (this does not influence the result much, as the upstream processes are rather similar); this is included in the aggregated data for the pulp mill processes. For allocating LCI data between the fuel precursor and the pulp, economic allocation was used (based on an market price estimates calculated in the GreenGasoline project). Data on emissions from the refinery where the fuel precursor is further processed into the final fuel are also estimates calculated in the GreenGasoline project.

The Ecoinvent dataset on distribution from the refinery to the petrol station for the gasoline benchmark (see section 3.2.5) was assumed also for the lignin-based fuel, a dataset based on an estimation of European average distribution of low-sulphur gasoline, including transportation of product from the production to the end user, operation of storage tanks and petrol stations and emissions from evaporation and treatment of effluents.

The fuel consumption was in the GreenGasoline project estimated to be the same as for the gasoline benchmark (0.054 kg/km, see section 3.3.5 for further details), and the exhaust emissions of CO_2 were estimated to also be the same as for the gasoline benchmark, but of biogenic instead of fossil origin (i.e. the automotive fuel is assumed to be to 100% produced from the lignin-based precursor).

3.3.4 LCI table

Table 9 shows the LCI data of the lignin-based fuel product system. The CO_2 uptake during forest growth accounts for both the carbon embedded in the materials and the biomass used for energy purposes in production.

Table 9. LCI data for the lignin-based fuel, per functional unit (1 km with a European average car). The production phase includes emissions in forestry and distribution of the fuel.

LCI data / **Process**	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Forest growth	0.19	0	0	0	0
Production	0	0.080	0.024	0.000067	0.0000044
Use/end-of-life	0	0	0.16	0	0

3.3.5 Benchmark products

The benchmark products are assumed to be the same as for the butanol (see section 3.2.5).

3.4 CROSS-LAMINATED TIMBER (BUILDING COMPONENT)

3.4.1 Introduction

Cross-laminated timber (CLT) is a building material where sawn, small dried timber panels are bonded together with durable adhesives resulting in a bigger and stronger element. CLT panels can be used to prefabricate load-bearing and non-load-bearing elements in timber buildings, especially for floors, walls and roofing; some of its advantages are ease to manage, dimensional stability, high load capacity and a high degree of prefabrication. The product is already established on the Swedish market, where suppliers of certified products can be found. CLT is very similar to glue-laminated timber (glulam), with the difference that the timber panels are oriented in crossed directions when glued together. Besides this difference, the manufacturing processes are very much alike.

"Brf Viva" is a multi-family housing project by Riksbyggen in Gothenburg, Sweden, where different actors are using the complex to carry out research for sustainable housing and city planning in a full-scale living lab (Riksbyggen 2015). Within Brf Viva, a project financed by the Swedish Energy Agency was carried out with the purpose of supporting Riksbyggen in developing a building design with the lowest possible climate impact. One work package in this project compares different design alternatives using LCA, and identifies opportunities to reduce the climate impact of the building. Two main alternative designs for the same building were proposed in the project, one using a CLT structure and one using a concrete alternative. Riksbyggen provided the architectural design of the building including areas for each element and total living area, while the industrial partners in the project provided the technical specifications of the designs from which the material amounts were estimated. More information about the project and the details of the design can be found in Norén et al. (2015).

3.4.2 Functional unit

The functional unit used is the *building structure per m^2 of living area for 50 years*. This means that apart from CLT, also other building materials are considered. Nevertheless, CLT accounts for most of the mass content of the structure (the same goes for concrete in the benchmark product, see section 3.4.5). Most of the other materials in the CLT building are sawn timber, plywood and glulam, which are also forest-based and affected by the choice of carbon footprint methodology.

3.4.3 System description

The flowchart of the CLT building is presented in Figure 7. After forestry, roundwood is transported to the supplier's sawmill where it is sawn and debarked. The planks are then dried in a kiln to reduce their moisture content and quality-tested, so the ones suitable are used in CLT manufacturing. The dry timber planks are then bonded together using Melamine-Urea-Formaldehyde (MUF) resin, a thermosetting polymer which is highly resistant to moisture and weathering and relatively low formaldehyde emissions during use. This is how the CLT is manufactured, but the production of the remaining forest materials is similar until the kiln drying of the sawn timber. What varies is the technology used to bond together the timber to produce different materials such as glulam or plywood, while for regular sawn timber only a treatment process is required depending on the intended use of the plank (exterior or interior). The LCI data used to model the product system of CLT and other timber products used in the building structure was based on data from Ecoinvent 2.0 and modified to more accurately reflect Swedish practices. These modifications are based on information obtained from industrial partners in the projects and general knowledge. Among these modifications are the transport distances of raw materials, electricity from the Swedish electricity mix, heat from bioenergy for drying and the amount and type of adhesive used for lamination. Furthermore, for the construction activities only the energy used by building machines was taken into account using literature values, assuming that the machinery is diesel-powered. All the transport processes were modelled using Ecoinvent transport datasets and estimated distances. Since the building is intended to be energy positive, it was assumed that the energy demand for the operation of the building is supplied from wind power (electricity) and heat pumps (heat), for which Ecoinvent data was used as well. No maintenance was accounted for, while for the demolition activities only the energy for the machinery was included and assumed to be diesel, using literature values similarly to the construction activities. Finally, for the end-of-life scenario it was assumed that 70% of the non-forest-based materials are recycled and the remaining waste is incinerated. For the forest-based materials, it was assumed that 90% of the demolition waste is incinerated and the rest is recycled. More detailed information about the process modelling for the life cycle stages different than manufacturing can be found in Norén et al. (2015).

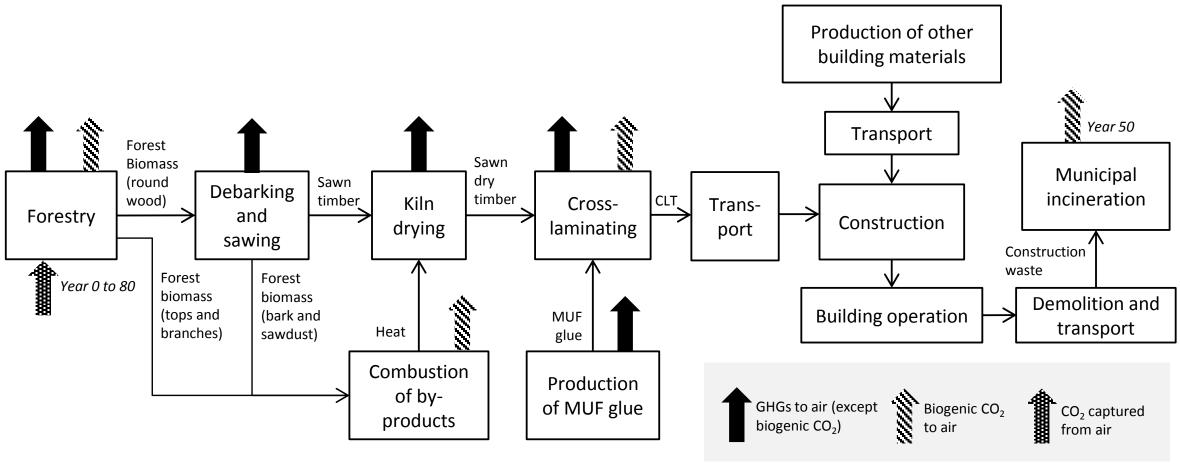


Figure 7. Process flowchart for the CLT product system, including the relevant flows for the carbon footprint.

Along the product system of the CLT and the timber products used in construction there are several processes with multiple outputs which require allocation. All of these outputs are biomass by-products from the process, including bark from debarking, sawdust from sawing. In Sweden it is a common practice to use the total of the output of these by-products in the same product system, to generate the heat that is required for the kiln. The climate impacts from these multi-functional processes have been allocated using factors based on the economic value of each of the by-products.

3.4.4 LCI table

The LCI data for the CLT building is presented in Table 10. The CO_2 uptake during forest growth accounts for both the carbon embedded in the materials and the biomass used for energy purposes in production.

Table 10. LCI data for the CLT building design, per functional unit (1 m^2 of living area for 50 years). The production phase includes emissions in forestry.

LCI data / **Process**	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions(kg)*	*N_2O emissions (kg)*
Forest growth	460	0	0	0	0
Production	0	150	35	0.18	0.010
Use (building operation)	0	48	0	0.20	0.0040
End-of-life	0	28	430	0.030	0.0010

3.4.5 Benchmark products

A building with a concrete structure was chosen as benchmark, as it is a widely-used non-forest alternative for construction. It is also possible to build structures in steel, but there was no steel alternative design available in the Brf Viva project. Similarly to the timber, the LCI data used for the concrete structure was based on Ecoinvent data but modified to represent Swedish conditions. The recipe for concrete is the same as in Ecoinvent, but the data for cement was obtained from an environmental product declaration for Cemex cement in Scandinavia. Meanwhile, European datasets were used for the remaining components of the concrete such as gravel and sand; and

Swedish data from Ecoinvent was used for water. The data for the energy inputs for the manufacturing of concrete reflects the Swedish electricity mix and Swedish district heating. The remaining life cycle stages were modelled similarly to the CLT building. More details can be found in Norén et al. (2015).

Table 11. LCI data for the benchmark product to the CLT building, a concrete building, per functional unit (1 m^2 of living area for 50 years).

LCI data / Process	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Production	2.3	410	2.3	0.81	0.0049
Use (building operation)	0	48	0	0.20	0.0040
End-of-life	0	32	0.030	0.033	0.0012

3.5 VISCOSE (TEXTILE FIBRES)

3.5.1 Introduction

Viscose, also known as rayon, is an example of a regenerated cellulose fibre, a category of textile fibres produced from forest cellulosics. In the production of regenerated cellulose fibres, dissolving pulp is produced in a chemical pulp mill and then dissolved and spun into fibres. This production route differs from the other two dominant classes of textile fibres on the market: synthetic fibres produced from crude oil (e.g. polyester) and natural fibres directly harvested and spun into fibres (e.g. cotton and wool). Regenerated cellulose fibres are thus produced from biotic resources but via synthetic processes, and therefore they are alternately labelled as man-made/synthetic, semi-synthetic or natural fibres.

Today, regenerated cellulose fibres constitute about 5% of the world market of textile fibres (Oerlikon 2010). Different types of regenerated cellulose fibres are characterised by different chemistry in the dissolving and spinning steps. For example, viscose – the most common type of regenerated cellulose fibres – is dissolved and spun in aqueous sodium hydroxide and carbon disulfide. Other types of regenerated cellulose fibres are Modal®, produced via a modified viscose process, and Tencel®, produced via the lyocell process, which uses N-Methylmorpholine N-oxide (NMMO) as a solvent (Shen et al. 2010). In recent years, there have been several initiatives in Nordic countries to develop new processes for the production of regenerated cellulose fibres, such as the CelluNova process developed in the CelluNova and ForTex projects (Vinnova 2014) and the Ioncell-F process (Michud et al. 2015). At present, no commercial-scale production of regenerated cellulose fibres for clothing exists in Sweden, but, as the aforementioned initiatives suggest, there are expectations on future production (recently, a project was granted within the BioInnovation platform to explore the opportunities of Swedish viscose production; BioInnovation 2015). Future production is likely also in light of the Swedish paper and pulp industry's challenge to stay internationally competitive, which calls for novel high-value uses of pulp, and the textile industry's challenge of finding new feedstocks, as manifested by the "peak cotton" discussion.

In the present study, viscose production was chosen to represent future production of regenerated fibres in Sweden because the viscose process is, globally, an established production route with sufficient LCI data availability. In modelling the viscose process, we used data on global average conditions, and adapted this data to Swedish conditions, as further described below. We selected cotton and polyester as benchmark products, due to their dominant position on the global market for textile fibres. It should be noted that viscose is not, for most applications, a direct substitute to either cotton or polyester. However, there are hopes to develop more cotton-like regenerated cellulose fibres. Also, what is considered a substitute product may change (e.g. because of price changes resulting from deficits of certain fibres) and does not only depend on functional properties of the fibres but also on market mechanisms. For example, polyester fibres are currently meeting nearly all of the increasing demand of textile fibres globally, whereas cotton production is stable – this is not because a lack of demand of cotton fibres, but because of difficulties in increasing the production of cotton.

3.5.2 Functional unit

The chosen functional unit is *1 kg staple textile fibres*. It should be noted that 1 kg of viscose is not necessarily functionally equivalent to 1 kg of cotton or 1 kg of polyester. However, the amounts of a certain fibre type needed for a specific function (e.g. a specific piece of clothing) will be highly dependent on the specific function. The functional unit was chosen to avoid such dependencies. Similar mass-based functional units have been used in LCAs of textile fibres elsewhere (Shen et al. 2010).

3.5.3 System description

Within the system boundaries, we included all processes to the gate of the textile fibre production and the end-of-life handling of the textile product, for which we assumed municipal incineration, the most common means of handling textile waste in Sweden (Östlund et al. 2015). Excluded life cycle phases are fabric and garment production, distribution and retail, consumer transportation and laundry. These were excluded as they depend more on the garment type than the fibre type, so including them would add more noise than value for the scope of the study. For further information on the relative environmental importance of different life cycle phases for clothing, see Roos et al. (2015). Figure 8 shows the process flowchart, including the relevant flows for the carbon footprint. GaBi 6 was used for modelling the product system (Thinkstep 2015). Below, each unit process is described in detail.

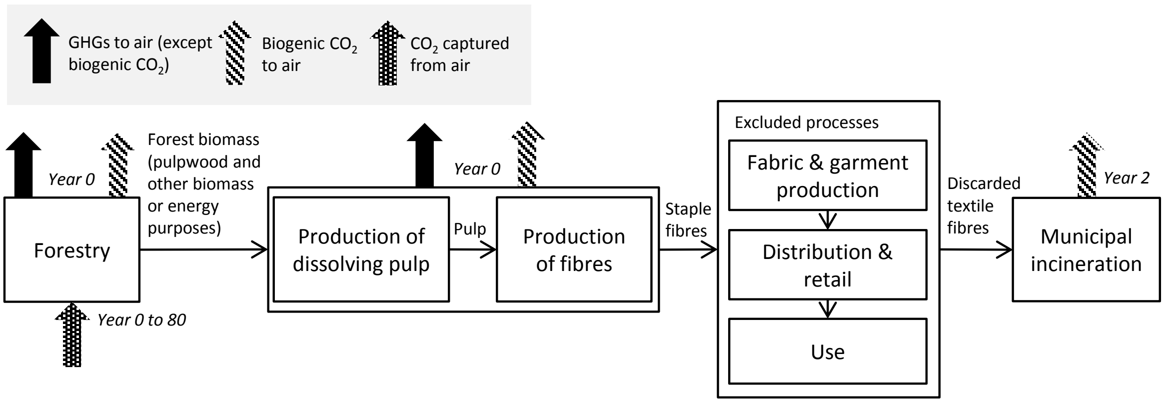


Figure 8. Process flowchart for the viscose product system, including the relevant flows for the carbon footprint.

Dissolving pulp is a bleached and relatively pure chemical pulp quality with high cellulose content (>90%). In the present study, LCI data for the production of dissolving pulp is estimated by data for 2011 on the Södra Mörrum pulp mill in southern Sweden, which produces several qualities of pulp, including dissolving pulp (which is currently exported). The dataset is based on public data (Södra 2012) and non-public data on the constituents specified in Södra (2012) as "chemicals, oils, etc." and "other raw materials" (accessed through personal communication in the CelluNova project). Because of the non-public status of this data, the modelling cannot be disclosed in detail (e.g., quantities of specific flows and names of assumed LCI dataset); instead we show aggregated LCI data on the relevant flows (in our case: GHGs) for the process as a whole. However, a few things can be said regarding the modelling. Datasets from the LCI database Ecoinvent were used when available, but in a few cases datasets from the LCI database GaBi Professional 2013 were used (Thinkstep 2015). We selected datasets reflecting Swedish conditions when available, otherwise datasets reflecting European average conditions, and as a last resort datasets on global average conditions. Furthermore, by-products were off-set against inputs as far as possible (e.g. if electricity was produced as a by-product, the input of electricity was assumed to be reduced by the same amount). Allocation between pulp and remaining by-products (heating and electricity) was based on economic allocation, using pulp and heating prices from Sandin et al. (2015) and electricity prices from Statista (2015). With this allocation, pulp was allocated 97% of the environmental burden of the pulp mill.

Textile fibre production was modelled using the Ecoinvent dataset on global average viscose production in 2001 (accessed at www.ecoinvent.org, but modelled in GaBi), adapted to Swedish conditions. The dataset is based on an Austrian company (presumably Lenzing) having production sites in several countries. For the full report of the dataset, see Althaus et al. (2007). The adaptation to Swedish conditions means that we foremost used LCI datasets on Swedish conditions, datasets on European average as a second choice, and datasets on global average conditions as a final resort. The Ecoinvent dataset reflects viscose production that also delivers the co-products sodium sulphate and sulphuric acid; the allocation between the co-products is based on economic value. We considered to instead use more up-to-date data from Shen et al. (2010), also on viscose production at the sites of Lenzing, but a lack of transparency made it impossible to combine this data with the carbon footprint methodology tested in the present study. See the Appendix for further details on the modelling of viscose production.

It should be noted that there are large differences between the environmental performance of viscose production at different sites, due to differences in the efficiency of chemical and energy use, and how the inputs (e.g. electricity) are produced in different parts of the world (Shen et al. 2010). Because of this, and because the data is rather old, the current best available viscose production can be expected to be considerably better in environmental terms than what the data used in the present study indicates. Thus the LCI data and the generated results should be seen as rough indications of the environmental performance of a possible future Swedish production of viscose, which is likely an overestimate, but sufficient for the goal and scope of this study.

The textile fibres are assumed to be discarded after about 2 years of use, based on the average service life of apparel in the UK (The Waste and Resources Action Programme 2012). In the incineration, we assume the only GHGs released are from the combustion of the biogenic carbon embedded in the fibre, which corresponds to 1.5 kg CO_2 per kg textile fibres (Shen and Patel 2010).

3.5.4 LCI table

Table 12 shows the LCI data of the viscose product system. The CO_2 uptake during forest growth accounts for both the carbon embedded in the materials and the biomass used for energy purposes in production.

Table 12. LCI data for viscose fibres, per functional unit (1 kg staple fibres). The production phase includes emissions in forestry.

LCI data / Process	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Forest growth	5.6	0	0	0	0
Production	0	1.9	4.1	0.0050	0.00014
End-of-life incineration	0	0	1.5	0	0

3.5.5 Benchmark products

Cotton and polyester textile fibres were chosen as benchmarks. These product systems were modelled to enable comparisons with the viscose fibres, thus we included cradle-to-gate production of staple fibres and end-of-life incineration within the system boundaries.

For the production of cotton staple fibres, we used a cradle-to-gate dataset including the cultivation, ginning and baling of cotton fibres, from Cotton Incorporated (2012), as available in the GaBi Professional database. The dataset represents global average production. The cotton is, just as the viscose fibres, assumed to be incinerated at end-of-life. For end-of-life CO_2 emissions, we used data from Shen and Patel (2010). The CO_2 captured in the cultivation is assumed to correspond to the sum of the biogenic CO_2 emissions from the production and the end-of-life incineration.

Polyester fibres are most often produced from dimethyl terephthalate (DMT) and ethylene glycol (EG). DMT is often produced from fossil petroleum whereas EG can be produced from biobased materials. In the present study we assumed that the polyester is of 100% fossil origin and that recycled materials are not used. Production consists of the production of polyester polymers and the subsequent melt spinning into fibres. For the polymer production, we assumed the Ecoinvent dataset "market for polyethylene terephthalate, granulate, amorphous", as integrated in the GaBi Professional database, which reflects global average production. For the subsequent melt spinning of the polymers into fibres, we used a dataset from Roos et al. (2015), with the difference that we assumed electricity supplied by the Chinese medium voltage electricity mix (the dominant producer of polyester fibres globally (Oerlikon 2010)) and the global market Ecoinvent datasets for the other background datasets. For details on the modelling of the melt spinning, see the Appendix. For end-of-life CO_2 emissions we used data from Shen and Patel (2008) on the CO_2 embedded in polyester pellets.

For both the cotton and polyester systems, CO_2 uptake and GHG emissions in production are assumed to occur at year 0 (a normal rotation period for cotton is 0.5 years (Althaus et al. 2007)), but the end-of-life emissions are assumed to occur year 2 (as was assumed for the viscose fibres). The cotton and polyester product systems give rise to small amounts of biogenic CO_2 emissions. These are assumed to origin from short-rotation crops (such as corn or sugarcane, a common feedstock for, e.g., ethanol production).

Table 13 shows the LCI data of the cotton and polyester product systems.

Table 13. LCI data for the benchmark products to the viscose case study, cotton and polyester, per functional unit (1 kg staple fibres).

LCI data / **Process**	*CO_2 capture (kg)*	*Fossil CO_2 emissions (kg)*	*Biogenic CO_2 emissions (kg)*	*CH_4 emissions (kg)*	*N_2O emissions (kg)*
Cotton production	1.8	2.5	0.074	0.012	0.012
Cotton incineration	0	0	1.7	0	0
Polyester production	0.10	7.3	0.10	0.046	0.00034
Polyester incineration	0	2.3	0	0	0

3.6 METHANOL (INDUSTRIAL CHEMICAL)

3.6.1 Introduction

The methanol case study was developed from an initial patent in the Skogskemi project (Vinnova, 2014), which is the same project from which the butanol case originates. The technique was developed by Processum Biorefinery AB and Metso Power AB, and patented as PuriMeth. The methanol is produced via purification of stripper off gases (SOGs) generated in the pulp production in sulphate pulp mills. The technology does not exist today other than on lab-scale. The potential production in Sweden is 50 000 tonnes per year. Today, the SOGs are burned in the sulphate pulp mills, either for destruction purposes or as an energy source, if the mill is not self-sufficient with energy.

3.6.2 Functional unit

The functional unit is *1 tonne of methanol.*

3.6.3 System description

The product system of the forest-based methanol is shown in Figure 9. The purification technique has been patented as PuriMeth, with the purpose of removing ammonia, turpentine and organosulphur components from the methanol to reduce smell and facilitate handling. The pre-purified methanol is further upgraded in a final purification process which is a central unit that collects pre-purified methanol from a number of pulp mills. Based on distances between mills in Sweden, it was estimated that the transport distance from a mill to the purification plant would be 160 km. Truck is the most likely transport option. Data on resources used for and emissions from the transport were collected from the Network for Transport and Environment, NTM (2013) and the PlasticsEurope database (2013). Data for the sulphate pulp mill were collected from an existing facility in Sweden, but are confidential. More information about the product system can be found in Joelsson (2014) and Røyne et al. (2015).

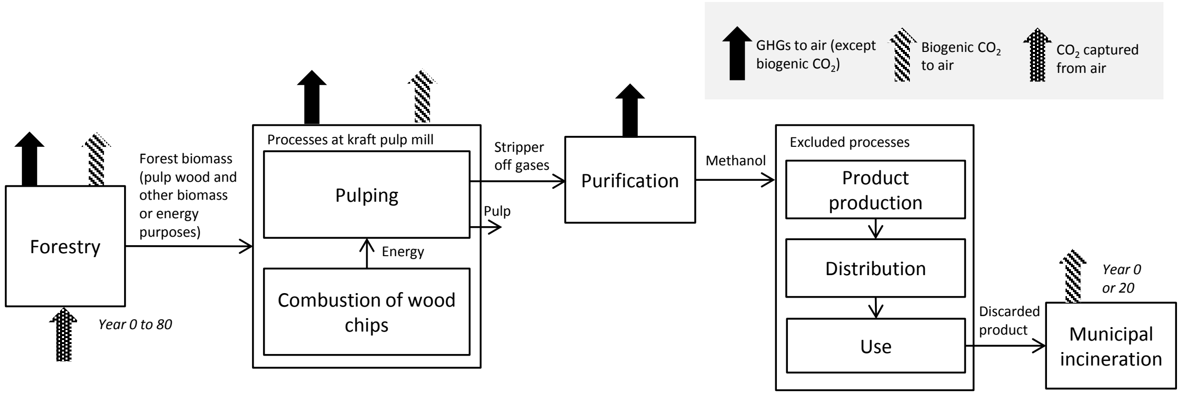


Figure 9. Process flowchart for the forest-based methanol product system, including the relevant flows for the carbon footprint.

Within the time limit of the Skogskemi project, the project group did not manage to produce methanol with a purity that meets the requirements of the International Methanol Producers & Consumers Association (IMPCA) methanol reference specifications (IMPCA 2014). Turpentine levels are at 1.5 g/kg methanol. The methanol can however still be used in a range of processes, from short lived packaging plastics and fuels, to long lived insulation (Methanol Institute 2011). As the service lives of the end product vary, we assumed two scenarios for when end-of-life occurss: 0 and 20 years.

3.6.4 LCI table

Table 14 shows the LCI data of the forest-based methanol product system. The CO_2 uptake during forest growth accounts for both the carbon embedded in the materials and the biomass used for energy purposes in production.

Table 14. LCI data for the methanol, per functional unit (1 tonne of methanol). The production phase includes emissions in forestry.

LCI data / Process	CO_2 capture (kg)	Fossil CO_2 emissions (kg)	Biogenic CO_2 emissions (kg)	CH_4 emissions (kg)	N_2O emissions (kg)
Forest growth	3260	0	0	0	0
Production	0	220	1400	0.75	0.0054
End-of-life	0	0	1800	0	0

3.6.5 Benchmark products

The fossil methanol used as a reference is methanol produced from natural gas, since this is the most common feedstock for methanol production today (Methanol Institute 2011). Thinkstep (2015) reports methanol production from natural gas for four different countries: The Netherlands, Germany, Great Britain and Italy. The electricity mix and energy used in the methanol production are the factors that contribute the most to the variation in the results. The carbon footprint of the processes range from 0.88 to 1.6 tonnes CO_2 equivalents per tonne methanol. The production of methanol in the Netherlands, with a carbon footprint of 1.03, was chosen for the comparison. The data are representative for production in 2011. Data on methanol production is also available in the Ecoinvent database (Swiss Centre for Life Cycle Inventories 2014), this data are however from 1994, and according to the documentation it contains significant uncertainties. This is why the Thinkstep (2015) data was chosen for the comparison.

Table 15 shows the LCI data of the fossil methanol product system.

Table 15. LCI data for the benchmark product for methanol, fossil methanol, per functional unit (1 tonne of methanol).

LCI data / **Process**	CO_2 sequestration (kg)	Fossil CO_2 emissions (kg)	Biogenic CO_2 emissions (kg)	CH_4 emissions (kg)	N_2O emissions (kg)
Production	0	1000	0	0.94	0.014
End-of-life	0	1800	0	0	0

4 RESULTS AND DISCUSSION

This chapter shows the results of the five case studies and a discussion of the results. The chapter is divided into three sections, which in different ways show the influence of the choice of carbon footprint methodology. Section 4.1 first summarises general observations from the results, then shows results for the forest products with information on the contribution of different life cycle phases, and finally shows results of the forest products in relation to the benchmark products. Section 4.2 shows the substitution effect if the forest products substitute the benchmark products. In Section 4.3, we discuss the representativity of the methods and practical considerations of applying them.

4.1 CLIMATE IMPACT OF FOREST PRODUCTS AND BENCHMARKS

4.1.1 General observations across product groups

For all the products studied, there were only marginal differences between the climate impact calculated using traditional LCA practice and the impact calculated using the methods required by EU sustainability criteria or the PEF guide (see Figure 10, Figure 12 and Figure 14). The small differences were due to different GWP_{100} CFs used for the two different approaches. Different CFs were used because new versions of these factors are released regularly based on improved understanding of the relative effect on the radiative forcing of different GHGs, and the version available in the used LCA software (which is the basis for traditional LCA practice) differs from the version recommended by the EU sustainability criteria and the PEF guide. Thus, the small differences were not due to fundamental methodological discrepancies between traditional LCA practice and the EU sustainability criteria/PEF. It should be recognized that the EU sustainability criteria and the PEF guide requires inclusion of aspects not accounted for with traditional practice – which were not relevant in our case studies – such as N_2O emissions from fertilisers and CO_2 emissions from land use change (if land is transformed between the states specified in the guiding documents). Thus, in other case studies, the results of using traditional LCA practice and the methods required by the EU sustainability criteria and the PEF guide may differ more.

In our case studies, a considerable difference is observed when dynamic LCA is used, as this method accounts for the carbon flows in the forest and the timing of GHG uptake and emissions, which, for example, yields credits for delayed emissions due to carbon storage in the product. Accounting for there aspects affects also the comparison between the forest products and their benchmarks, i.e. the climate benefit from substituting fossil or mineral products with forest-based alternatives (see Figure 11, Figure 13 and Figure 15).

4.1.2 Short-lived products: fuels and textile fibres

Figure 10 shows the climate impact calculated with the selected methods for the fuels and the textile fibres, which all have short service lives. These results show a marginal difference between traditional LCA practice and the methods required by the EU sustainability criteria and the PEF guide, while there is a larger gap between these and dynamic LCA with national-level och stand-level system boundaries. Moreover, the results also show that the choice of time horizon for dynamic LCA affects the results considerably. Thus the outcome of dynamic LCA depends strongly on the definition of both the spatial and the temporal system boundaries.

The biofuels and textile fibres have short service lives, i.e. there is a short time period from the harvest of the biomass until the biogenic carbon embedded in the products is emitted to the atmosphere. It takes years for the regrowing forest (using a stand perspective) to once again capture an equal amount of CO_2, a time period depending on the rotation period of the studied forest, which in our case studies was 80 years. Since the forest is a net emitter in the first years of regrowth (with the forest carbon model used in our case studies), using dynamic LCA with a stand perspective and a short time horizon accounts for these emissions but disregards out most of the CO_2 capture during regrowth. This explains why dynamic LCA with a stand perspective and a short time horizon results in the highest calculated climate impact for short-lived products. When using a landscape perspective, we have assumed that the same amount of biogenic CO_2 which is emitted during the life cycle is captured by the forest at year 0 (i.e., the system is carbon). In such a case, the climate impact of short-lived products depend solely on the the amount of fossil CO_2 emissions and other GHG emissions. A national perspective has the same assumption as the landscape perspective, except that it considers that CO_2 captured each year surpasses biogenic CO_2 emissions by 33% (because on a national level, there is a net annual increase of the biomass in Swedish forests; i.e., the system is a carbon sink). Therefore, the carbon footprint is lower with a national perspective compared to a landscape perspective. Moreover, when all biogenic CO_2 emissions occur in year 0 (as for the biofuels) there are no benefits from carbon storage. For the viscose fibres, emissions from end-of-life occur in year 2, a short carbon storage period which renders a very low storage benefit.

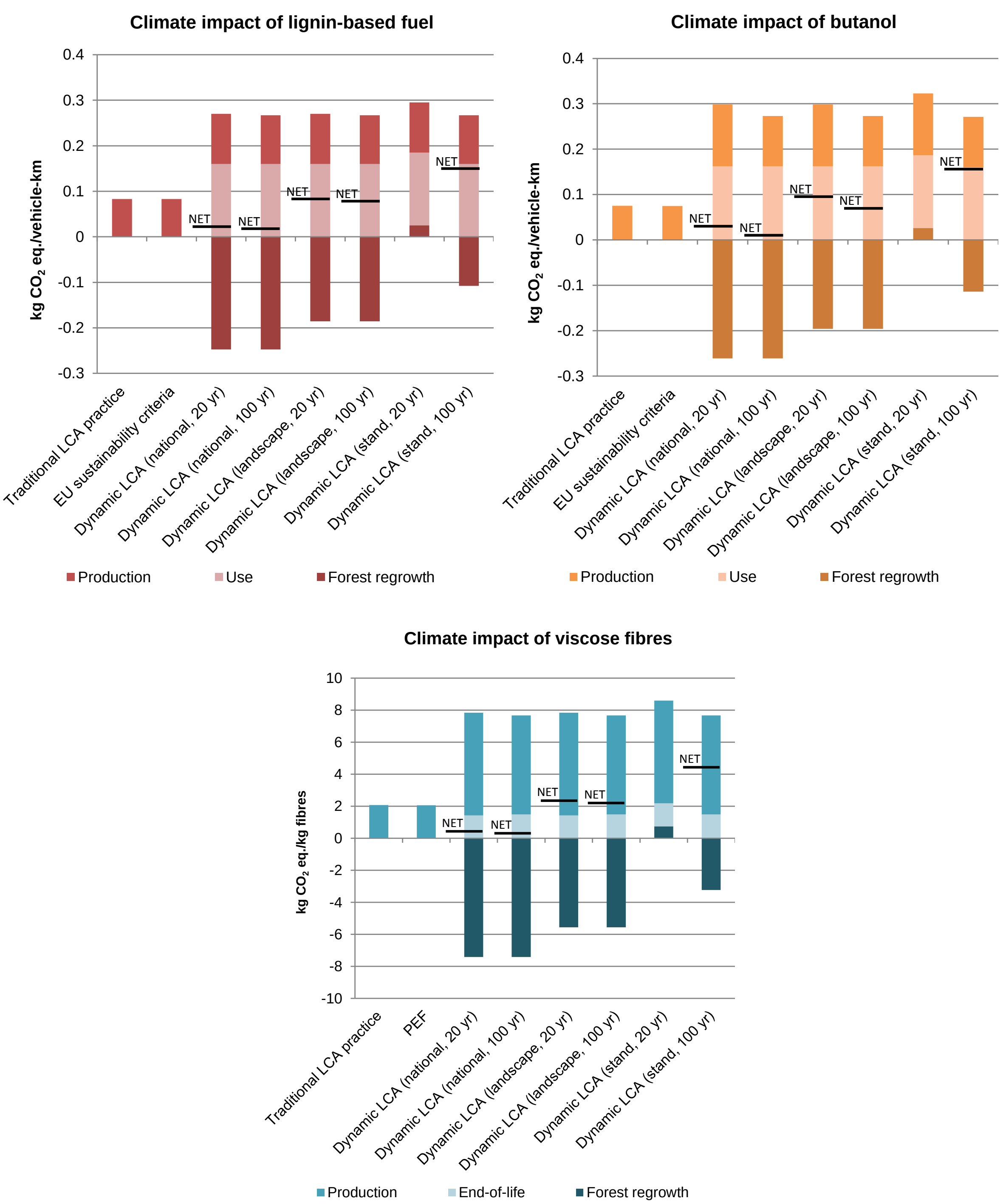


Figure 10. Climate impact of short-lived products (fuels and textile fibres) for different carbon footprint methods. For biofuels, the use and end-of-life phases are one and the same (combustion of the fuel).

Figure 11 shows the consequences of the choice of carbon footprint method on the comparison between the short-lived products and their benchmarks. The results for the biofuel benchmarks were not affected by the choice, even if dynamic LCA was used. This is because the emissions of using the fossil fuels occur in the same year as production, so accounting for the timing does not influence the results. However, as the results for the biofuels were considerably different with dynamic LCA, the comparison between the biofuels and their benchmarks is notably affected by going from traditional LCA practice or the methods required by the EU sustainability criteria and the PEF guide, to dynamic LCA, which accounts for timing. Furthermore, for dynamic LCA, the comparison depends strongly on the choice of spatial system boundaries, where the national-level system boundaries are the most favourable to the biofuels, and the stand-level system boundaries are the least favourable. This depencendy is similar for both the biofuels studied. Also, for dynamic LCA with a stand perspective, the comparison depends strongly on the choice of time horizon, as a short time horizon favours the fossil fuels while a longer time horizon favours the biofuel.

The case is slightly different for the comparison of viscose to its benchmarks. Cotton, one of the benchmarks, is also biobased and thus has embedded biogenic carbon. This means that biogenic CO_2 emissions and CO_2 capture take place also in the cotton product system, but with a very different harvesting period (2 times per year, as opposed to every 80 years as in the case of the boreal forest). Therefore, the cotton case is barely affected by method choice. As the polyester textile fibres are of fossil origin, it could be expected that the results, like the petrol and diesel cases, would barely be affected by method choice. The polyester case is, however, affected, as seen when using dynamic LCA with a time horizon of 20 years. The reason for this is that there are considerable amounts of CH_4 emissions in the product system. The impact of CH_4 compared to other GHGs increases considerably when a short time horizon is applied.

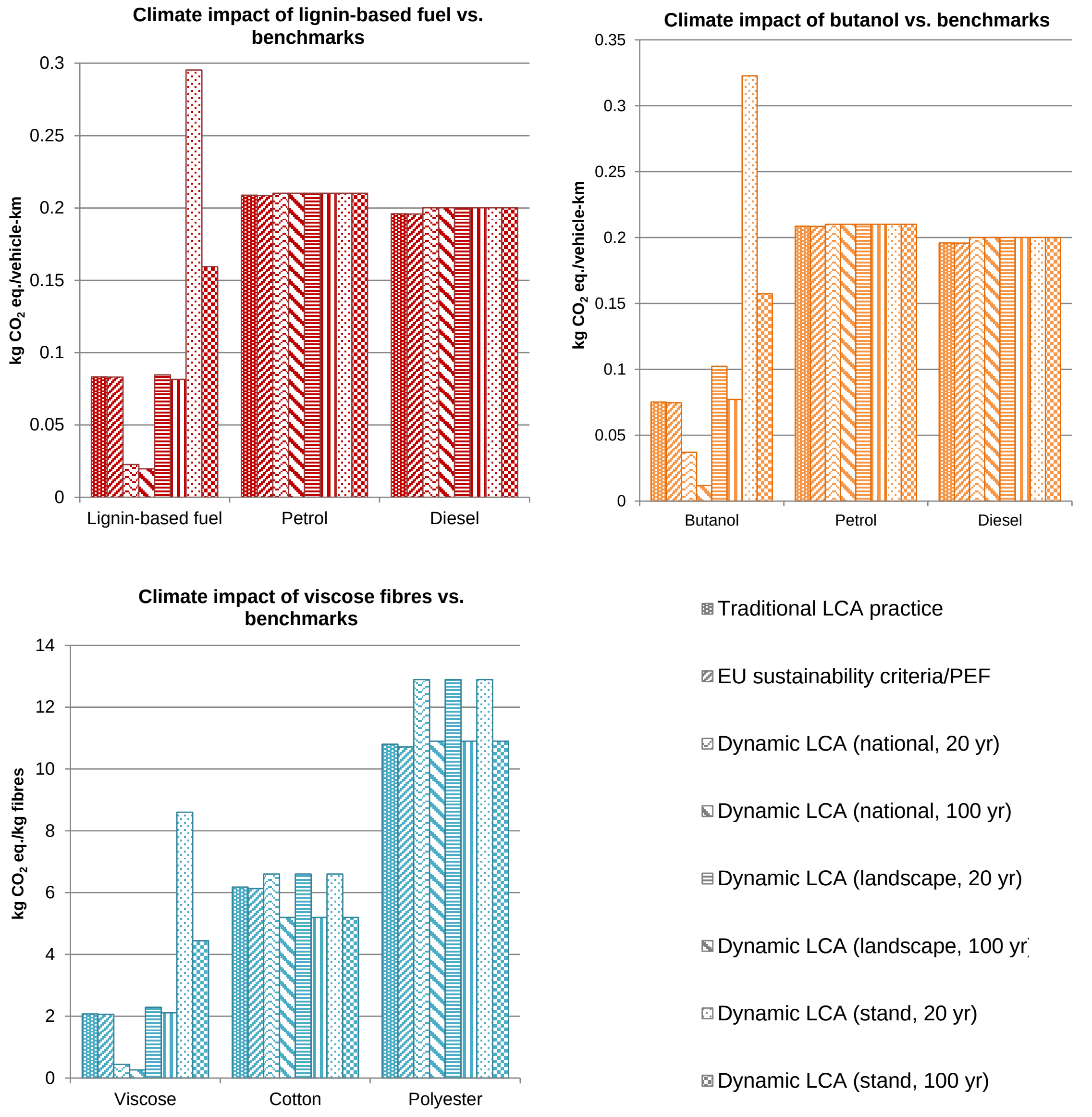


Figure 11. Climate impact of short-lived products (fuels and textile fibres) vs. benchmarks, for different carbon footprint methods (net results).

4.1.3 Long-lived product: building

Figure 12 shows the effects of the choice of carbon footprint practice for the CLT building case study, indicating that, with stand-level system boundaries, the differences between traditional LCA practice, PEF and dynamic LCA are smaller than for the short-lived products.

With national-level and landscape-level system boundaries, the climate benefit of storing carbon is, within both the chosen time horizons (20 and 100 years), larger than or similar as the climate impact from GHG emissions. Thus, considering the effect from carbon storage (an indirect effect from considering the timing of GHG emissions), renders very different results compared to using traditional LCA practice and PEF.

With stand-level system boundaries, the CO_2 capture is spread (although not evenly) along the duration of the rotation period (80 years in this case). Because of this, a time horizon of 20 years excludes the climate impact both from most of the CO_2 capture and the biogenic CO_2 emissions (occurring in year 50). This means that dynamic LCA of long-lived products produced from slow-grown biomass, with stand-level system boundaries and a short time horizon, excludes both all biogenic CO_2 emissions from the end-of-life incineration and a large part of the CO_2 capture, and therefore yields similar results as traditional LCA practice and PEF. However, the results change if the time horizon is extended to 100 years, as the CO_2 capture in the full regrowth period and the biogenic CO_2 emissions at end-of-life are then included. That dynamic LCA considers the time shift between CO_2 capture and emission (i.e., the temporary storage of carbon while regrowth takes place), reduces slightly the climate impact of the CLT building. The reason for why the results are so low with the national and landscape perspectives is that end-of-life emissions occur in year 50. With a 20-year time horizon, these emissions are excluded, and with a 100-year time horizon, there is a 50-year credit for having stored the carbon.

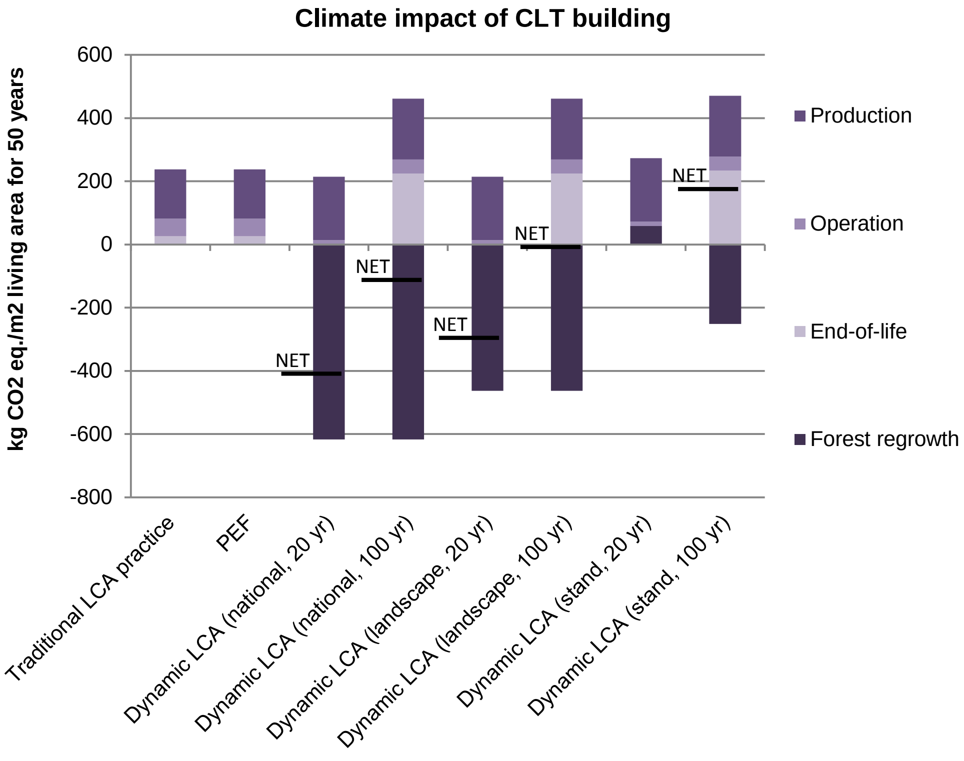


Figure 12. Climate impact of CLT building for different carbon footprint methods.

The choice of carbon footprint method did not considerably influence the result for the concrete building used as a benchmark (see Figure 13). The slight differences between the methods are because emissions occur at different points in time. The comparison between the CLT building and its benchmark was thus not heavily affected by the choice between the traditional LCA practice, PEF and dynamic LCA with a stand perspective. However, the comparison is dramatically affected when dynamic LCA with national- or landscape-level system boundaries is applied, as the storing of carbon until year 50 gives a credit that surpasses the climate impact. Such credits from storage are usually ignored in LCAs of buildings. For example, in Heeren et al. (2015), the carbon footprints of a wooden and a concrete bulding were compared, but the timing of emissions and carbon capture were neither included nor discussed – the only remark about possible carbon capture was made for the concrete building and the carbonization process.

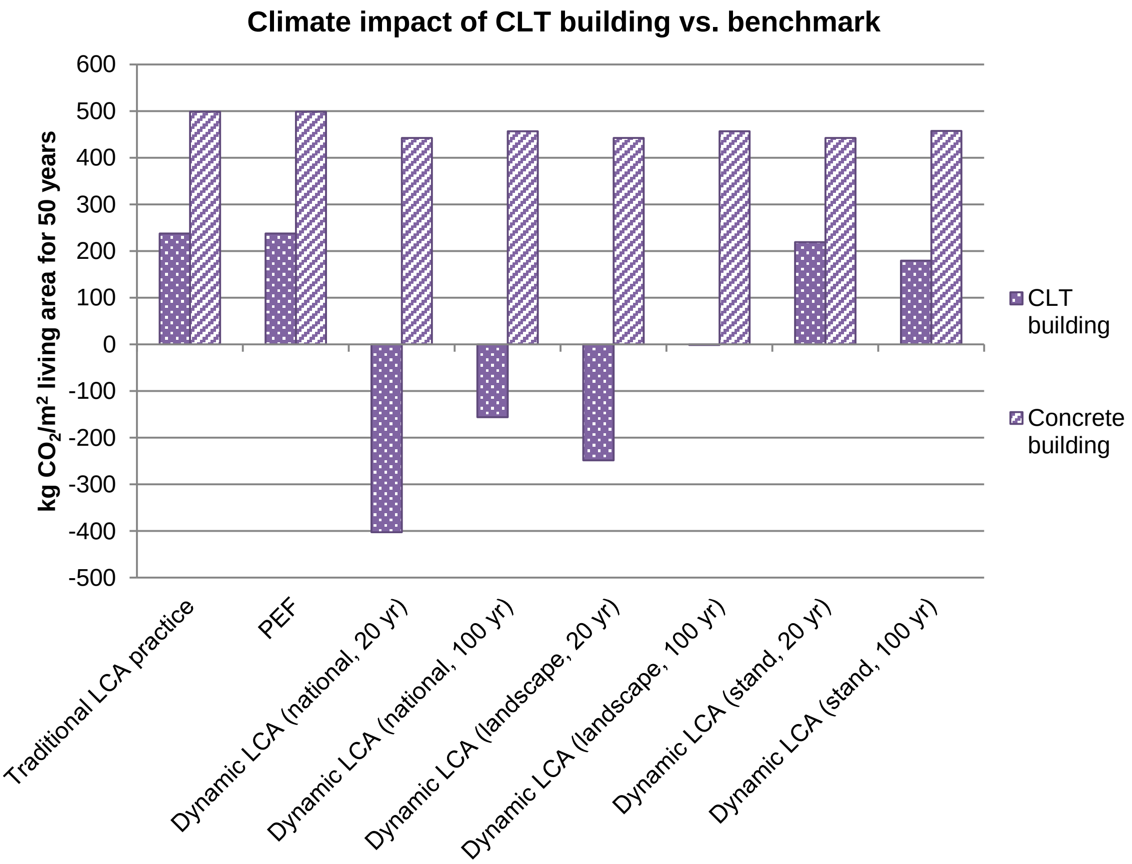


Figure 13. Climate impact of CLT building vs. the benchmark building, for different carbon footprint methods (net results). Results for the CLT building using dynamic LCA (landscape, 100 yr) is not missing in the figure, but is very close to zero.

4.1.4 Product with varying service lives: methanol

The result of the methanol case, shown in Figure 14, shows a substantial difference between the results of the traditional LCA practice/PEF and dynamic LCA, regardless of spatial system boundaries and time horizon. The main reason for this is that there are large biogenic CO_2 emissions from the production, and low fossil CO_2 emissions, which makes the results highly sensitive to how biogenic CO_2 emissions are handled.

The case of the methanol illustrates the relevance of the service life of the product in relation to the time horizon for dynamic LCA. Given that methanol can be used as a precursor for many different products with potentially different service lives, two different service lives were tested. In the case where the service life is 20 years, dynamic LCA with a 20-year time horizon excludes all the impact from the end-of-life emissions – the emissions are released within the time horizon, but at the very end, and therefore they have no impact during the time horizon. On the other hand, when dynamic LCA with a time horizon of 100 years is applied, the difference in results between methanol with different service lives is smaller because the end-of-life emissions have more similar impact in the two product systems (as they contribute to climate impact during 100 respectively 80 years).

Figure 14 also shows that the definition of the spatial system also has a considerable influence on results. National system boundaries render negative (i.e. beneficial) results, regardless of time horizon. Landscape-level system boundaries render lower or similar results compared to traditional

LCA practice and the method required by the PEF guide. Stand-level system boundaries render considerably higher results compared to the other methods, as was also the case for the short-lived forest products discussed in the previous section – this is because the delay between the emission of biogenic CO_2 at the end-of-life (in year 0 or 20) and the capture of an equivalent amount of CO_2 in the regrowing forest stand (which is complete in year 80).

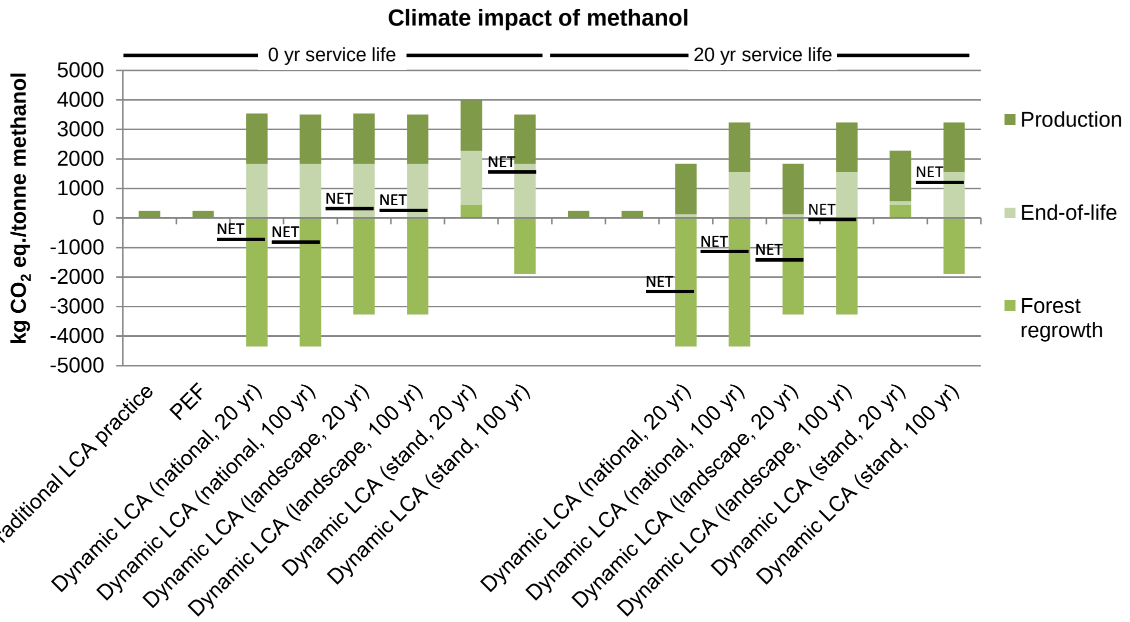


Figure 14. Climate impact of forest-based methanol for different carbon footprint methods.

The difference between methanol and its benchmark is considerably affected by the choice of carbon footprint method. As Figure 15 demonstrates, the traditional LCA practice, PEF and dynamic LCA with national- or landscape-level system boundaries yield overwhelmingly favourable result for the bio-based methanol. With dynamic LCA with a stand perspective and a time horizon of 100 years, the climate impact of the bio-based methanol is substantially higher, but still about half the climate impact of the fossil methanol. With dynamic LCA with a stand perspective and a time horizon of 20 years, however, the bio-based methanol performs worse than the fossil methanol, both for service lives of 0 and 20 years. This is because the aggregated levels of biogenic and fossil GHG emissions are higher in the bio-based methanol product system than in the fossil methanol product system (compare LCI data in Table 14 and Table 15). So when the chosen carbon footprint methodology and system boundaries disregards whether emissions are of biogenic or fossil origin, and give little credit for CO_2 capture, the bio-based product system performs badly (in this case).

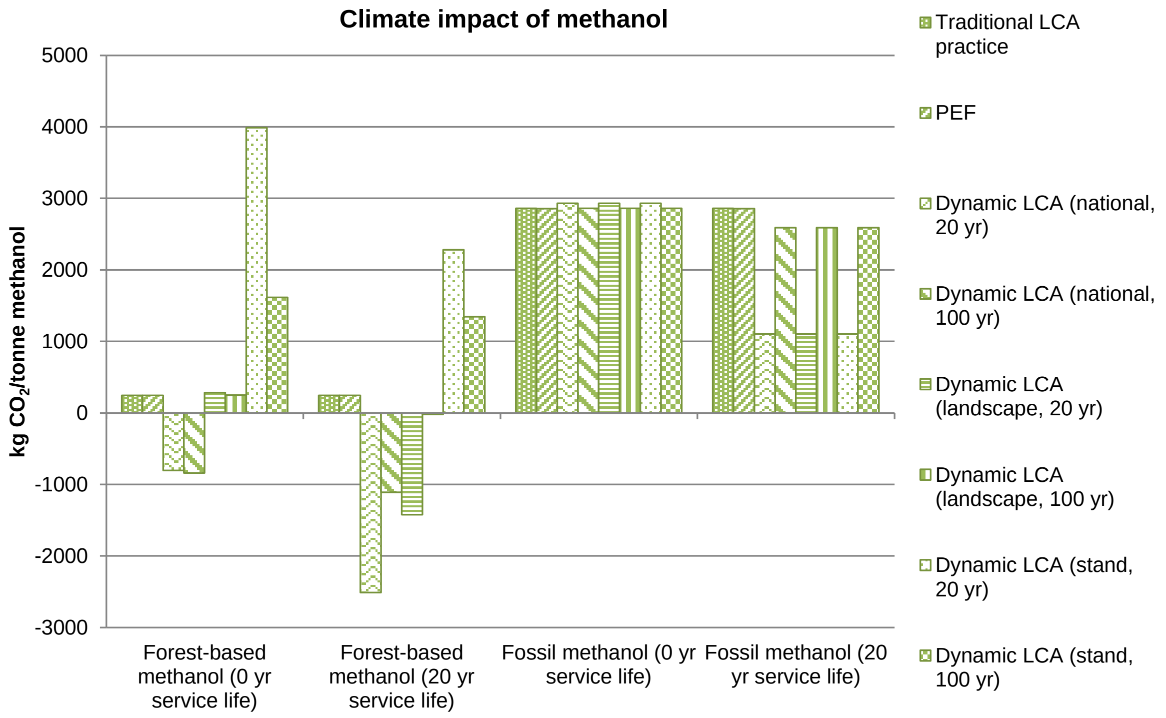


Figure 15. Climate impact of forest-based methanol vs. fossil methanol, for different carbon footprint methods (net results).

4.2 CLIMATE BENEFITS OF SUBSTITUTING BENCHMARKS

Figure 16 shows a comparison between the climate impact reduction obtained if each of the studied forest products is to substitute their respective benchmark product, and how this depends on the choice of carbon footprint method. The sameresults are provided in Table 16. The figure shows no clear trend in the span of results of short-lived versus long-lived products. The figure also shows that the approach contributing most to the span of the results, and negatively for the forest-based product, is dynamic LCA with a stand approach and a 20-year time horizon. The long-lived CLT building substituting the concrete structure has a result span similar to most of the short-lived products. The case where the short-lived viscose substitutes polyester has the smallest span, while the case where bio-based methanol with a 20-year service life substitutes fossil methanol has the most extreme span – the reason for the extreme span is the large amount of biogenic CO_2 emissions in the production of the bio-based methanol, which makes it sensitive to choice of carbon footprint method.

The large potential variations in results shown by our study by applying different carbon footprint methods, time horizons and spatial system boundaries, shows that studies aiming to make recommendations about future use of forest resources should have greater emphasis on the choice of method and system boundaries, to a larger extent align these choices with the specific goal and scope of the study, and, if deemed suitable, apply and explore a range of methods and system boundaries. For example, in a recent publication assessing carbon footprints of the use of wood for energy and materials (Wilnhammer et al. 2015), the temporal dynamics in the product systems were ignored and different spatial and temporal perspectives were not considered. This limits the usefulness of the study.

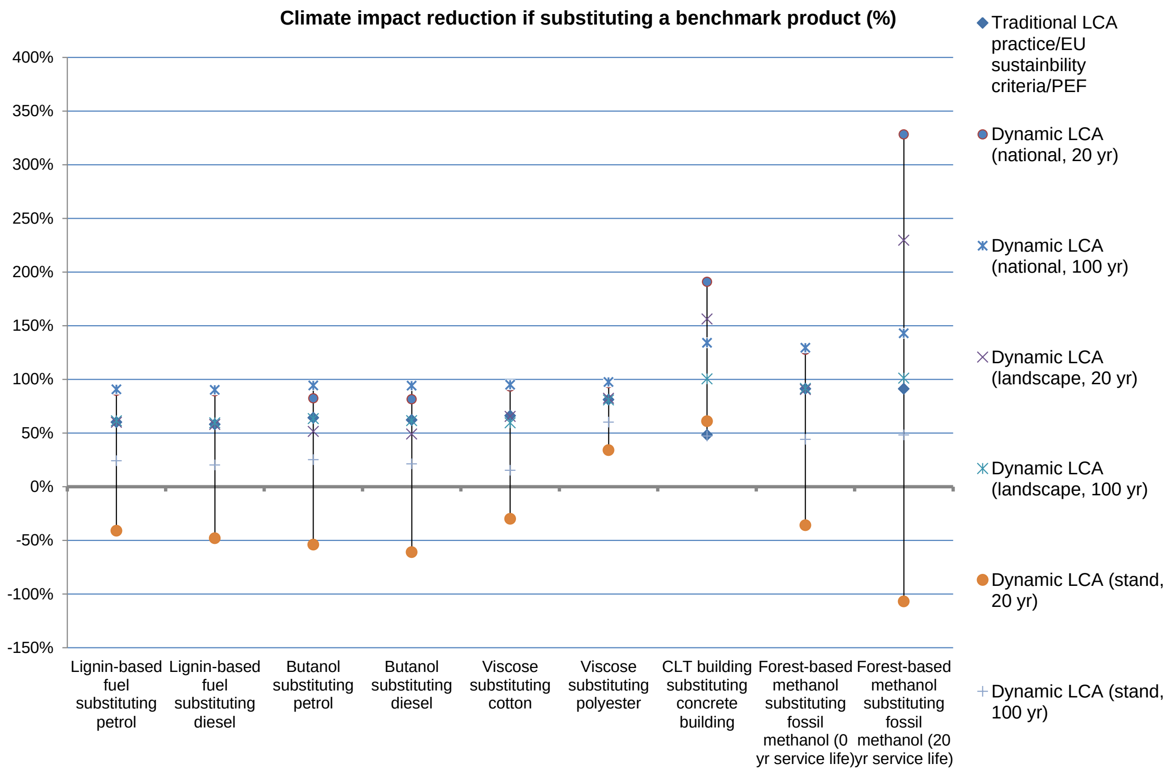


Figure 16. Climate impact reduction, if each forest product is assumed to substitute its benchmark product.

Table 16. Climate impact reduction, in percentage, if each forest product is assumed to substitute its benchmark product.

	Climate impact reduction potential if substituting benchmark product (%)							
	Traditional LCA practice	*EU sustainability criteria/PEF*	*Dynamic LCA*					
			National 20 yr	*National 100 yr*	*Landscape 20 yr*	*Landscape 100 yr*	*Stand 20 yr*	*Stand 100 yr*
Lignin-based fuel substituting petrol	60%	60%	89%	91%	60%	61%	-41%	24%
Lignin-based fuel substituting diesel	58%	58%	89%	90%	58%	59%	-48%	20%
Butanol substituting petrol	64%	64%	82%	94%	51%	63%	-54%	25%
Butanol substituting diesel	62%	62%	82%	94%	49%	61%	-61%	21%
Viscose substituting cotton	66%	66%	93%	95%	65%	59%	-30%	15%
Viscose substituting polyester	81%	81%	97%	98%	82%	81%	34%	60%
CLT building substituting concrete building	48%	48%	191%	134%	156%	100%	61%	47%
Methanol substituting fossil methanol (0 yr service life)	91%	92%	128%	129%	90%	91%	-36%	44%
Methanol substituting fossil methanol (20 yr service life)	91%	92%	328%	143%	229%	101%	-107%	48%

4.3 REPRESENTATIVITY AND PRACTICAL CONSIDERATIONS

Neither traditional LCA practice nor the EU sustainability criteria for biofuels or PEF account for differences in growth rate because of forest type and location. Neither do these methods account for timing of emissions. In contrast, dynamic LCA can account for these temporal dynamics. One can argue whether a landscape or stand perspective is the most appropriate in a certain decision-making context (which defines whether it makes sense to account for growth rates or not), but delaying emissions by storing carbon results in a climate benefit regardless of the spatial and temporal system boundaries that the LCA practitioner deemes suitable. Thus, one can argue whether capturing the stand-level time lag between harvest and CO_2 capture is a better reflection of reality, but it is difficult to argue against the real climate benefit of carbon storage. Thus, it can be claimed that dynamic LCA (and other similar methods) enables LCA practitioners to capture more dimensions of reality than those enabled by established carbon footprinting practices.

The level of detail offered by dynamic LCA comes at a cost, as dynamic LCI data is less available than traditional, static LCI data. For our study, for example, we used Finnish forest data as a proxy for Swedish conditions, which adds to the data uncertainty of the study.

Neither the traditional LCA practice, nor the methods required by the EU sustainability criteria and the PEF guide, account for indirect effects such as ILUC or the effect from albedo or aerosols. Neither does dynamic LCA offer a framework for capturing such effects – as that is not the focus of the method – but it can be combined with methods/data reflecting also those effects. The limits of current methods and practices in capturing such effects make carbon footprints of forest products more uncertain, in general, than carbon footprints of non-forest products. For example, for a fossil product system such as petrol, uncertainties are mainly due to the representativity of LCI data – uncertainties that exist also for forest products – but the temporal and spatial modelling complexities and the unknowns of non GHG-related climate effects (such as albedo changes) can most often be ignored. The uncertainties related to these complexities cannot be overcome by the LCA practitioner, but must be solved by further research.

5 CONCLUSIONS AND RECOMMENDATIONS

In this chapter, we (i) summarise the main observations discussed in chapter 4, (ii) list recommendations for decision makers and for LCA practitioners and researchers, and (iii) list the most important future research needs. Important for formulating this chapter has been the input collected at an open seminar with important stakeholders of the Swedish forest sector; see more in section A.3 in the Appendix.

5.1 MAIN OBSERVATIONS

- For the case studies of this report – reflecting various uses of Swedish forest biomass – there is only a marginal difference in the results of applying the traditional LCA practice for carbon footprinting and the methods required in the EU sustainability criteria and PEF. The latter documents do, however, allow practitioners to capture some climate aspects not covered by traditional LCA practice: land use change (if land is transformed between the states specified in the documents) and N_2O emissions from fertilization, which could influence results in systems where such aspects are prominent.

- The advanced method explored in this study, dynamic LCA, renders remarkably different result compared to other practices. This is because dynamic LCA takes timing of CO_2 capture and GHG emissions into account (a time lag between CO_2 captured in the forest and the biogenic CO_2 emission at end-of-life translates into a temporary increase of the atmospheric radiative forcing).

    - When using dynamic LCA with a stand perspective, the choice of time horizon of the CFs has a large influence on results. This is because only CO_2 capture and GHG emissions occurring within the chosen time horizon are included, and their relative importance depend on when they occur (e.g. an emission occurring today has a large impact, whereas an emission occurring in the end of the chosen time horizon has a small impact).

        - With a short time horizon, end-of-life emissions are accounted for in studies of short-lived products, but not in studies of long-lived products. Thus a short time horizon generally favours long-lived products.

    - When using dynamic LCA with national and landscape perspectives, the choice of time horizon of the CFs has no or small influence on results in studies of short-lived products. For long-lived products the choice has a larger influence, as the impact of carbon storage is taken into account. Just as for dynamic LCA with a stand perspective, a short time horizon generally favours long-lived products.

- In the comparison with fossil benchmarks, the choice of carbon footprint method can make the forest-product perform both better and worse. With dynamic LCA with a short time horizon, most of the short-lived products perform worse than their fossil counterpart. For the studied product with the longest service life (the building), the forest case performs better with all methods, and dynamic LCA with a national perspective renders the largest climate benefit.

- So, using temporally more advanced carbon footprint methodology influences different forest products in different ways: some products benefit in relation to a non-forest counterpart, whereas other products do not benefit. Important determining the influence are the product's service life, the amount of biogenic CO_2 emitted in the production processes compared to the end-of-life incineration, and the time horizon adopted in the impact assessment.

5.2 RECOMMENDATIONS

The main aims of the work presented in this report were to:

(a) Contribute to more robust decision making concerning how to use Swedish forest biomass for reducing climate impact, with a focus on decision making within the biofuels sector.

(b) Contribute to the process of improving the methods and practices of carbon footprints of forest products.

There are thus two groups of actors to whom we intend to provide recommendations: the ones who commission and use the conclusions of LCAs, and the ones who conduct LCAs. Our main message is that the decision maker must be clear about which question he or she wants an answer to, and the LCA practitioner must be clear on what the study includes and excludes, and which implications this has for the results. The decision maker must thus take the responsibility as the expert of the decision at hand and the LCA practitioner must take the responsibility as the expert of the model of the studied system.

5.2.1 Recommendations for decision makers

- As different carbon footprint methods can give very different results, our key message to decision makers is to increase their **consciousness** on these matters. It is important to be aware of the assumptions made in the study, the effects of those assumptions on results, and how results can and cannot be used for decision support in a certain context.

- Decision makers must be aware that the main methodological choices influencing carbon footprints of Swedish forest products are the choice of geographical system boundaries (e.g. national-, landscape- or stand-level system boundaries) and whether the timing of CO_2 capture and GHG emissions is accounted for. This is because Swedish forests are, in general, slow growing.

- If the aim of the decision is to obtain short-term climate impact reduction – for example, the urgent reduction that is possibly needed for preventing the world average temperature to rise with more than 2°C – the timing of CO_2 capture and GHG emissions should be taken into account. Decision makers must be aware that a particular method for capturing timing (such as dynamic LCA) can be combined with different system boundaries, which can yield different results.

- When conclusions from existing LCA studies are synthesized for decision support, the decision maker must be aware that most existing studies do not account for the timing of CO_2 capture and GHG emissions. This is particularly important when the decision concerns the prioritization of forest products with different service lives (e.g., fuels versus buildings).

- When timing is considered, decision makers must be aware that there are different views on when the CO_2 capture occurs, which will influence the carbon footprint. One could either consider the CO_2 captured before the harvest (i.e., the capture of the carbon that goes into the product system), or the CO_2 captured after the harvest (i.e., the consequence of the harvest operation). In this study, we tested the second alternative when we applied dynamic LCA with a stand perspective – this does not mean we advocate the use of the second alternative over the first alternative.

- Decision makers must be aware that the location and management practices of the forestry influence the climate impact of a forest product. For example, growth rates, changes in soil carbon storages and fertilisers (a source of GHGs) differ between locations.
- Based on our results, we cannot say that the carbon footprints of some product categories are more robust than for others, i.e. less influenced by choice of methodology. However, the more forest biomass use in the product system, the higher the influence of the choice of method.
- As many interactions between the forest and the climate are still not fully understood, it is important to be open to new knowledge gained in climate science and in carbon footprint methodology development work.
- Regarding how to use Swedish forests for the most efficient climate impact reduction, it is impossible to draw a general conclusion on the basis of our results. Factors that influence the "optimal" use are:
    - Which fraction of forest biomass that is used. Various products use different fractions (as was the case in our case studies) and do not necessarily compete for the same biomass. However, a production system may be more or less optimised for a specific output. So there may be situations of competition also when feedstocks are not directly interchangeable.
    - Which non-forest product that is assumed to be replaced by the forest product (if any). The carbon footprint of the non-forest product matters, but also how large the substitution effect is (i.e., does the forest product actually replace the non-forest alternative, or merely add products to the market, and what are the rebound effects from increased production?).
    - If all other factors are identical: the longer the service life of the forest product the better, due to the climate benefit of storing carbon and thereby delaying CO_2 emissions. This effect is particularly strong if the aim is to obtain short-term climate impact reduction. Also, the effect supports so-called cascade use of forest biomass, e.g. first using wood in a building structure, then reusing the wood in a commodity, and at end-of-life, as late as possible, recovering the energy content of the wood for heat or fuel production.
- Traditional LCA practice and methods required by the EU sustainability criteria and PEF have limitations in the support they can provide for the transition to a bio-economy, as they cannot capture the variations of different forest products in terms of, for example, rotation periods and service lives. Thus, decision makers need to consider studies using more advanced methods to be able to distinguish better or worse uses of forest biomass. We have tested one such advanced method (dynamic LCA), that proved applicable in combination with several different geographical perspectives, but also other methods exist (e.g. GWP_{bio}).

- Climate change is not the only environmental impact category which is relevant in decision making concerned with how to use forests. Other environmental issues, such as loss of biodiversity and ecosystem services, are also important. There are also non-environmental sustainability issues of potential importance, e.g. related to indigenous rights and job creation.

5.2.2 Recommendations for LCA practitioners and researchers

- A main message to LCA practitioners and researchers is the complexity of carbon footprints of forest products. As seen in this study, a more advanced view on temporal dynamics of the forest product system influences the results considerably compared to using established practices. Moreover, there are many other factors that are potentially important and that are not captured well or not captured at all by established practices (as described in chapter 2). Awareness about this is important for improving the methods and practices of carbon footprinting, and for producing better LCAs.

- Because there is (still) limited knowledge about how forest products influence the climate, and as carbon footprints will always depend on value-based assumptions (e.g. regarding geographical system boundaries), it is not possible, based on our results, to recommend one specific method which is suitable regardless of context.

- LCA practitioners should communicate the shortcomings of the applied carbon footprint practices when communicating results.

- Because of the shortcomings of traditional LCA practices, and because the practice is supposed to depend on the goal and scope of the study, LCA practitioners should guide the audience on how to interpret the results. Here, one should consider not only the intended audience and the intended use, but also consider other potential audiences and uses (e.g. by inserting disclaimers on how the results should *not* be interpreted).

- LCA practitioners should be aware that new, more advanced carbon footprint methods are becoming increasingly available and applicable, which makes it more possible to align methodological choices to the goal of the study – a crucial factor for improving the quality of LCAs.

    - Specifically, this study has shown that methods are available for considering timing of CO_2 capture and GHG emissions, for being more consistent in terms of the time horizons used for calculating CFs for emission pulses occurring at different points in time, and for selecting time horizons that are aligned with the urgency of climate impact mitigation reflected in the decision context.

    - Other aspects of carbon footprints, such as changes of the albedo and creation of aerosols, are more difficult to assess with the currently available methods and LCI data. However, it is possible to make rought estimates based on results available in the literature (e.g., as done by Røyne et al. (2016)).

    - Unfortunately, applying non-established methods is time consuming and requires relatively extensive knowledge about carbon footprinting. Thus, it is important that LCA researchers, climate scientists and LCA software developers collaborate to improve methods and make them more practical to use (as also recommended by Bright 2015).

- Several of the non-traditional carbon footprint aspects discussed and tested in this study depend extensively on the location of the forestry. LCA practitioners must be aware of the strong site-dependency of data and methods and make sure that specific, general or average forest data is not inappropriately used.

- As the choice of substituted product may strongly influence the quantified climate benefit of a certain forest product (see Figure 16), LCA practitioners must be careful when assuming substitution effects (i.e., that a certain forest product is assumed to substitute a certain non-forest product). There are large uncertainties regarding which product is substituted, both directly and as a consequence of the end-of-life handling (especially for long-lived products), and to what extent that product is substituted (e.g., increased production of a forest product may to some extent substitute an alternative product, but may also cause lower market prices, resulting in increased demand, which offsets some of the substitution effect – an example of the rebound effect (Hertwich 2005)). Besides, although a specific product is substituted today, this might not be the case in 10 years, when the decision the LCA is intended to support is starting to have an effect. This uncertainty can be captured quantitavely by scenario analysis (see e.g. Sandin et al. (2014) for an approach for capturing uncertainties in the end-of-life handling of long-lived forest products).

5.3 FUTURE RESEARCH

The results of this project show that there is a need for improved carbon footprint methodology and data and and increased consciousness on these matters among LCA practitioners and decision makers. Future research should develop methodology covering all potentially important aspects of the climate impact of forest products – and other products, to enable fair comparisons – and corresponding LCI data covering a wider range of locations. For example, in applying dynamic LCA with stand-level system boundaries, we used LCI data from a forest carbon model reflecting conditions in Finland – for more accurate assessments of products produced from Swedish forest biomass, it would be valuable with data reflecting Swedish conditions, ideally distinguishing between different regions in Sweden.

Until improved methods and data are available, it is important to enhance the credibility of existing climate footprint practice. This can be achieved by conducting further case studies in which different climate aspects and system boundaries are explored. Such case studies will contribute to an increased understanding among LCA practitioners and decision makers regarding implications of various methodological choices. In our project the focus was on one type of forest, five types of products, and a few specific methods and choices of system boundaries. Future studies should examine:

- Additional forest products and benchmark products.

- The influence of the choice of baseline for modelling the carbon flows in the forest. Different choices of baseselines were discussed in section 2.2.3, but not explored further.

- The influence of assuming forest growth *before* instead of *after* harvest when applying stand-level system boundaries. In this study, we only modelled with forest growth after harvest (when we used dynamic LCA and stand-level system boundaries).

- The potential influence of including non-carbon climate aspects. Some such aspects were discussed in section 2.2.4, but not explored further.

- How different forest types in different locations are affected by the choice of carbon footprint method. In this study, the scope was limited to Swedish boreal forests with a long rotation period.

LCA is designed to assess product systems, which makes it challenging to allocate physical flows to specific products while at the same time not lose sight of the greater system. It is important that research on improved LCA methods and practices does not lose sight of the ultimate question, namely how to mitigate climate change on a global scale. There is thus a need for research on how methods in LCA can be developed to support global aims. This concerns questions of how we should – nationally, regionally and internationally – use limited forest resources to mitigate climate change in the most efficient way. Future research should explore data and methods that are necessary for answering such questions, and explore the role of LCA in such a context.

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APPENDIX

A.1 ABBREVIATIONS

CF = characterisation factor

CH = Switzerland

CH_4 = methane

CLT = cross-laminated timber

CN = China

CO_2 = carbon dioxide

DMT = dimethyl terephthalate

EC = European Commission

EG = ethylene glycol

EU = European Union

EPD = environmental product declaration

GHG = greenhouse gas

GLO = global

Glulam = glue-laminated timber

GROT = branches and tops

GWP = global warming potential

ILCD = international reference life cycle data system

ILUC = indirect land use change

IMPCA = International Methanol Producers & Consumers Association

IPCC = Intergovernmental Panel on Climate Change

ISO = International Organisation for Standardisation

JRC IES = Joint Research Centre for Institute for Environment and Sustainability

LCA = life cycle assessment

LCI = life cycle inventory analysis

LCIA = life cycle impact assessment

N = nitrogen

NH_3 = ammonia

NH_4^+ = ammonium

NMMO = N-Methylmorpholine N-oxide

N_2O = nitrous oxide

NOx = aitrogen oxides

NO_3^- = nitrate

NTM = Network for Transport and Environment

PAS = publicly available specification

PCR = product category rule

PEF = product environmental footprint

PEFCR = product environmental footprint category rules

RED = renewable energy directive

RER = Europe

SE = Sweden

SOC = soil organic carbon

SOG = stripper off gases

A.2 LCI TABLES

Table 17. Modelling of viscose fibre production, based on the Ecoinvent dataset on viscose production, but adapted to Swedish conditions (e.g. by the selection of LCI datasets). Data is given per 1000 kg of viscose fibres.

Flow	LCI dataset (from Ecoinvent 3.1)	Amount
Inputs		
Sulphate pulp	(from the dissolving pulp production process)	1019 kg
Sodium hydroxide, without water, in 50% solution state	GLO: market for sodium hydroxide, without water, in 50% solution state	731.8 kg
Sulfur dioxide, liquid	RER: market for sulfur dioxide, liquid	206.5 kg
Sodium hypochlorite, without water, in 15% solution state	RER: sodium hypochlorite production, product in 15% solution state	156.1 kg
Sodium chloride, powder	RER: sodium chloride production, powder	123.8 kg
Carbon disulife	GLO: market for carbon disulfide	91.15 kg
Sulfuric acid		69.98 kg
Nitrogen, liquid	RER: market for nitrogen, liquid	46.89 kg
Oxygen, liquid		18.61 kg
Chemical, inorganic	GLO: market for chemical, organic	15.54 kg
Zinc monosulfate	RER: zinc monosulfate production	12.48 kg
Heat, district or industrial, other than natural gas	SE: heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014	13 560 MJ
Heat, district or industrial, natural gas	SE: heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014	5 034 MJ
Electricity, low voltage	SE: market for electricity, low voltage	3680 MJ
Electricity, medium voltage	SE: market for electricity, medium voltage	22.45 MJ
Water, river		204.2 m^3
Pulp factory	RER: pulp factory construction	5E-08 pcs.
Outputs		
Viscose fibres	(to further life cycle phases)	1000 kg
Municipal solid waste	SE: treatment of municipal solid waste, incineration ecoinvent	7.43 kg
Hazardous waste, for incineration	CH: treatment of hazardous waste, hazardous waste incineration	0.01782 kg
Water, in air		162.4 m^3
Water, river		41.87 m^3
Wastewater	CH: treatment of wastewater, average, capacity 1E9l/year	0.1021 m^3

Table 18. Modelling of polyester fibre producton, based on Roos et al. (2015). Data is given per 1000 kg of polyester fibres.

Flow	LCI dataset (from Ecoinvent 3.1)	Amount
Inputs		
Polyethylene terephthalate	GLO: market for polyethylene terephthalate, granulate, amorphous	1000 kg
Lubricating oil	GLO: market for lubricating oil	10 kg
Manganese	GLO: market for manganese	0.1 kg
Phosphoric acid	GLO: market for phosphoric acid, industrial grade, without water, in 85% solution state	0.1 kg
Antimony	GLO: market for antimony	0.05 kg
Cobalt	GLO: market for cobalt	0.05 kg
Electricity	CN: market for electricity, medium voltage	17 640 MJ
Outputs		
Polyester	(to further life cycle phases)	1000 kg
Dimethyl terephthalate, to indoor air		0.0001 kg

A.3 SEMINAR

An important part of this project was to organise a seminar, which was held on October 15th 2015 in Gothenburg with video link to Stockholm. The aim of the seminar was to inform about our preliminary results and collect input to ensure that the final report is aligned with the needs of relevant stakeholders. The seminar attracted representatives from industry (Volvo, SCA, Tetrapak and AkzoNobel), universities (Chalmers, KTH and UC-Davis), research institutes (IVL, Swerea IVF and SP), EPD International AB and CIT/f3. In addition to presentations about our own project, SCA held a presentation about their experiences with carbon footprinting. The seminar agenda can be found in the end of this section. In the end of the seminar, a discussion was held concerning the following points:

- Are we addressing climate impacts in a good way in carbon footprinting? For which types of products and studies are the gaps largest?
- Is there a need for further research and development? What kind of work?
- Is there a need for changed practices and standards? In what way?
- Who are important actors in terms of work that remains and changes that need to be made?

The response we received can be divided into two categories: (i) methodological choices made in the project, and (ii) communication to decision makers.

Concerning the first category, many comments concerned the choice and execution of dynamic LCA. Some doubted this was the best representative for "advanced methods" as there are other methods, such as GWP_{bio} (developed by Cherubini et al. 2011), which are also recent and takes timing into account. We agreed that we must be thorough and clear when arguing for our method choice in the report, and that we should show a broader spectrum of potential results by not only combining dynamic LCA with stand-level system boundaries but also landscape-level system boundaries (in addition to this, we also added scenarios with national-level system boundaries).

Another comment concerned the presentation of the various aspects influencing carbon footprints of forest products, and their grouping in either "inventory" or "impact assessment". In several cases it is not possible to put only one of these labels on an aspect. However, when we present how different methods cover the different aspects we must be clear on whether an aspect is excluded based on the disability of the method itself or a lack of inventory data. For example, the reason we could not include albedo in dynamic LCA is not because of the method itself, but because the available forest models do not account for changes of the albedo. Another comment about definitions concerned defining the EU sustainability criteria for biofuels and the PEF as "standards". These documents are not normally called standards, and we therefore changed terminology, and in the final version of this report we instead call them either by their names or "requirements" (or similar).

The challenge of data verification was also highlighted. LCA practitioners and method developers might forget that for industry to be allowed to publish assessments involving site specific data, they depend on someone to verify the data. If a company produces forest products with biomass origin from several locations, this can become both challenging and costly, particularly if the forests are located in numerous countries.

When it comes to the communication to decision makers, all industry representatives agreed that this is a major challenge, and that popular versions of the results and conclusions are needed. We were therefore grateful that the industry participants at the seminar volunteered to review the executive summary, one of the outcomes of the project (see page ii). We also got the comment that existing guidance documents and standards on carbon footprinting lack detailed guidance on many of the climate aspects. It was recommended that we avoid similar confusion by being very clear on all our choices on methods, inventory and assessment procedures. Finally, an important point made was that the quantified results are not what primarily should be communicated to decision makers, but the conclusions relating to the aims of the study. Decision makers need clear and understandable recommendations on how to act.

Agenda for open seminar October 15, 2015: The method's influence on the carbon footprint of biofuels and other forest products

12:00
Registration and lunch

13:00
Introduction (Gustav Sandin Albertsson, SP)
• Challenges related to carbon footprinting of forest products

13:30
How do the carbon footprints of biofuels and other forest products depend on method? (Frida Røyne, SP)
• Established standards and directives vs.
• current practices in life cycle assessment vs.
• state-of-the-art in scientific literature

14:00
Coffee

14:30
SCA's experiences with carbon footprinting (Ellen Riise, SCA)

15:00

Discussion: How can we improve our practices and standards for carbon footprinting? (Magdalena Svanström, Chalmers)

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