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Abstract

Fossil carbon emission substitution and carbon storage effects of wood-based products

Forests and forest products contribute to climate change mitigation by sequestering carbon into forests, storing part of the carbon in harvested wood products (HWPs) and by avoiding fossil-based greenhouse gas (GHG) emissions in substitution for alternative materials and energy. Often, there are trade-offs in sequestering carbon into forests and harvesting trees for substitution, which means that these two strategies cannot be optimized at the same time. Which strategy is the most effective depends on a number of assumptions including the time horizon, metrics to characterize the climate effects, the development of forest carbon stocks, the way harvested wood is processed and used, and the alternative products to be substituted.

Assessing the climate effects of the use of wood, changes in carbon stocks in forests and HWPs, as well as changes in fossil carbon emissions should be considered coherently. To do that, two systems are compared; the one with the studied wood use, and its reference system without the wood use being studied. In this report, the focus was on assessing carbon stock changes in HWPs and fossil emission substitution due to using HWPs and wood-based fuels in place of non-wood materials and fuels. The key knowledge and challenges encountered in the assessment and characterization of carbon storage in harvested wood products, substitution effects and the effect of cascading use of wood on them were summarized and discussed. Finally, some practical guidelines to conduct an assessment on an annual basis at a multiproduct and company level and over the life cycle at the product level were provided.

Keywords: wood products, harvest, material, energy, carbon storage, climate change, substitution, fossil fuel, emissions

Tiivistelmä

Puuperäisten tuotteiden vaikutukset fossiilisten hiilipäästöjen substituutioon ja tuotteiden hiilivarastoon

Metsien ja puun käytön avulla voidaan hillitä ilmastonmuutosta sitomalla hiiltä metsiin ja puutuotteisiin sekä välttämällä fossiilisia kasvihuonekaasupäästöjä korvaamalla uusiutumattomia raaka-aineita puulla. Usein metsien hiilen sidonnan ja puunkäytön välillä on vaihtosuhde, eikä metsien hiilen sidontaa ja substituutiossa vältettäviä fossiilisia päästöjä saada optimoitua samanaikaisesti. Optimaalinen metsien käyttöstrategia ilmastonmuutoksen hillitsemiseksi riippuu useista oletuksista, muun muassa tarkastelujen aikajänteestä, ilmastovaikutusten määrittämisestä, metsien hiilivaraston kehityksestä, puun käyttötavoista ja vaihtoehtoisten tuotteiden korvaamisesta.

Arvioitaessa puunkäytön ilmastovaikutuksia, tulee muutokset metsien hiilivarastossa, puutuotteiden hiilivarastossa ja fossiilisissa päästöissä huomioida johdonmukaisesti. Siihen tarvitaan vertailua, jossa tarkasteltavaa puunkäyttöjärjestelmää verrataan tilanteeseen, jossa sitä ei olisi. Tässä raportissa keskitytään tarkastelemaan puutuotteisiin sitoutuvaa hiiltä ja puun materiaali- ja energiakäytöllä vältettäviä fossiilisia päästöjä. Raportissa käydään läpi puutuotteiden hiilivaraston, puun käytön substituutiovaikutusten ja puun kaskadikäytön määrittämiseen liittyvät keskeiset haasteet. Lopuksi annetaan joitakin käytännön suosituksia siihen, miten näitä tekijöitä voi arvioida vuosittain monituotteisesti ja yritystasolla ja elinkaarisesti tuotetasolla.

Asiasanat: puutuotteet, korjuu, materiaalit, energia, hiilivarasto, ilmastonmuutos, substituutio, fossiiliset polttoaineet, päästöt

Sammandrag

Substitutions- och kollagringseffekter av träbaserade produkter

Användningen av skog och trä kan mildra klimatförändringarna genom kolbindning i skogar och träprodukter, samt genom att ersätta icke-förnybara råvaror med för att undvika fossila växthusgasutsläpp. Det finns ofta ett utbytesförhållande mellan kolbindning i skogar och användningen av trä, varför det inte går att optimera kolbindningen i skogar och de fossila utsläpp som undviks vid substitution samtidigt. Den optimala strategin för skogsanvändning för att mildra klimatförändringarna beror på flera antaganden, bland annat tidsspannet i översynerna, fastställandet av klimatpåverkan, utvecklingen av kollagret i skogarna, sätten att använda trä och ersättning av alternativa produkter.

Vid bedömning av klimatpåverkan som orsakas av träanvändningen, måste man konsekvent beakta förändringarna i skogarnas och träprodukternas kollager och i fossila utsläpp. För detta behövs jämförelser där det granskade systemet för träanvändning jämförs med att det inte skulle finnas. Denna rapport fokuserar på att granska kol som binds i träprodukter samt fossila utsläpp som undviks genom att trä används som material och energi. I rapporten sammanfattas central kunskap och centrala utmaningar när det gäller att fastställa kollagret i träprodukter, substitutionseffekterna hos träanvändning och kaskadanvändningen av trä. Slutligen ges några praktiska rekommendationer om hur dessa faktorer kan bedömas årligen enligt multiproduktbasis på företagsnivå och livscykelbasis på produktnivå.

Nyckelord: träprodukter, drivning, material, energi, kollager, klimatförändringar, substitution, fossilt bränsle, utsläpp

Preface

This report was carried out in collaboration with the Finnish Environment Institute SYKE and Institut für energie- und umweltforschung Heidelberg (IFEU) in (Germany). The work was carried out by Head of Unit Sampo Soimakallio (SYKE), Head of Department Horst Fehrenbach (IFEU), Senior Scientist Susanna Sironen, Senior Scientist Tanja Myllyviita, Senior Scientist Nabil Abdalla (IFEU), and Professor Jyri Seppälä (SYKE). The work was funded by UPM-Kymmene Corporation and SYKE. The report reflects only the views of the authors and not necessarily those of the funders.

March 2022

Authors

Contents

1. Introduction

Forests and forest products contribute to climate change mitigation by sequestering carbon into forests, storing part of the carbon in harvested wood products and by avoiding fossil-based greenhouse gas (GHG) emissions in substitution for alternative materials and energy (Pingoud et al. 2010). Often, there are trade-offs in sequestering carbon into forests and harvesting trees for substitution, which means that these two strategies cannot be optimized at the same time (Soimakallio et al. 2016, 2021, Seppälä et al. 2019, Camia et al. 2021). Which strategy is the most effective depends on the number of assumptions, including the time horizon, metrics to characterize the climate effects, the development of forest carbon stocks, the way harvested wood is processed and used, and alternative products to be substituted (Soimakallio 2014, Soimakallio et al. 2016, Seppälä et al. 2019, Cherubini et al. 2016, Cowie et al. 2021).

Wood demand is expected to increase due to an overall increase in well-being and general consumption and the need to substitute fossil-based raw materials with renewable raw materials (Lauri et al. 2017). Increasing wood demand increases pressure on forests which provides fiber, fuels and food, as well as various other ecosystem services such as sustaining biodiversity and genetic resources, filtering water supplies, controlling floods and erosion, sequestering and storing carbon, cleaning the air, and providing opportunities for recreation, education, and cultural enrichment. Thus, it’s very important to end deforestation globally, restore formerly deforested areas and manage existing forests in a sustainable manner to ensure that wood removals do not exceed growth and no significant harm is caused to any other ecosystem services. The circular use and lengthening of the life-time of wood products can reduce demand for wood and are thus important aspects to be considered.

Intergovernmental Panel on Climate Change (IPCC) distinguishes between the slow domain of the carbon cycle and the fast domain of carbon cycle. For the slow domain the turnover times exceed 10,000 years, and for vegetation and soil belonging to the fast domain (the atmosphere, ocean, vegetation and soil) the turnover times are in the magnitude of 1–100 years and 10–500 years, respectively (IEA Bioenergy, 2018). Thus, the biomass and fossil carbon cycle differ fundamentally from each other.

According to the IPCC guidelines, biomass-based CO_2 emissions from sources and removals by sinks are accounted for through carbon stock changes instead of carbon flows when reporting GHG emissions to the United Framework Convention of Climate Change (UNFCCC). This means that CO_2 emissions from the combustion of biomass are accounted as zero in the energy sector when reporting GHG emissions to the United Framework Convention of Climate Change (UNFCCC). Additionally, this means that the carbon content of forest biomass harvested from forest is accounted as CO_2 emissions in the forest pool and carbon input into the harvested wood product pool is accounted for as carbon removal (negative emissions).

Typically, biomass-based carbon dioxide flows have been excluded from life cycle assessment (LCA) studies (Soimakallio et al. 2015), based on the assumption that as biomass is renewable, new growing biomass sequesters back carbon dioxide released in the combustion or decay process of biomass. While this assumption might be true over the carbon cycle of the biomass in question, there might be a time lag between the carbon release and sequestration (Cherubini et al. 2011, Helin et al. 2013). Due to this time lag, the carbon stock of forests and other land ecosystems have been reduced compared to what they would have been without intensified land-use (Erb et al. 2018). Simultaneously, the carbon stock in harvested wood products has increased (Lauk et al., 2012) and energy and materials made from wood may have been partly used in place of fossil-based energy and materials (Naudts et al. 2016).

GHG balances related to the use of wood can be separated into three different main categories; 1) changes in forest carbon stocks, 2) changes in harvested wood products (HWPs) carbon stocks, and 3) changes in fossil-based GHG emissions. Although these GHG balances are to some extent connected, they are also to some extent separate issues. One important further question is how temporary carbon storage and the timing of emissions should be considered in LCAs.

In this report, the focus is on the carbon storage of harvested wood products and fossil-based GHG emissions avoided in substitution. First, it is clarified how these issues should be handled coherently related to wood harvested from forests. Second, the life cycle modeling approaches that exist are clarified, including which questions they answer and how the results should be understood given the approach chosen and the question studied. Third, there is a summary of how harvested wood product carbon stocks and fossil-based GHG emissions avoided in substitution have been considered in the previous literature. Considering the carbon storage, the aim was to reveal how the carbon storage of HWPs has been assessed in the literature with a special focus to determine how carbon storage accounting could be applied at the international corporate level. The aim was to find what modeling components are needed in carbon storage accounting for HWPs. Fourth, the key issues related to and the importance of the cascading use of wood at the end of the life cycle of products are discussed and clarified. Fifth, the significance of the carbon storage effect, GHG emissions avoided in substitution and the influence of the cascading use of wood on them are analyzed and discussed using selected UPM products as examples. Finally, recommendations and conclusions on methodological choices, data requirements, and interpretation of results are made for practical claims of the carbon storage effects of HWPs and fossil-based GHG emissions avoided in substitution.

2. On the key concepts and methodologies

2.1 Perspectives in carbon accounting and reporting

GHG flows to and from the atmosphere can be studied from two different perspectives; namely, absolute balances or in relation to the predefined reference system (Soimakallio et al. 2015). Both of perspectives may be relevant in different contexts but should be understood coherently to avoid misleading conclusions. In the following, it is explained how these perspectives should be understood and how the carbon in HWPs and avoided fossil-based GHG emissions are connected to them.

The absolute GHG balance describes the physical flows into and from the atmosphere as they exist or are assumed to exist. In principle, such flows can be measured, verified, and monitored. Carbon and other GHGs are removed from the atmosphere by sinks and released into the atmosphere from sources. The state of the environment and its development can be described through absolute GHG balances. Thus, they are applicable for following how well-predefined targets to reduce GHG emissions or maintain carbon sinks are achieved. (Koponen et al. 2018)

The GHG balance in relation to a predefined reference system describes how absolute GHG balances change as a response to certain measure. The absolute GHG balance of a system, for example a certain wood use, is studied in comparison to a reference system, in which the studied measure (e.g. certain wood use) does not exist. Thus, in this context, the GHG balance is determined as the difference between “with the studied measure” (e.g. certain wood utilization) and “without the studied measure” (e.g. without certain wood utilization) (Cowie et al. 2021, Koponen et al. 2018).

In economics, substitution refers to two or more goods that the consumer perceives as similar or comparable, thus as substitutes to each other. Regarding wood use, substitution refers to any alternative material or energy service that is displaced by the use of a material or energy service provided by wood. The GHG emissions avoided refer to those emissions not generated because a service provided by wood takes place, and the alternative service displaced does not take place. The GHG emissions avoided are generated as a difference between a ‘with wood use system’ and its reference system, namely, the ‘without wood use system’. In other words, the emissions avoided are not absolute but they are generated compared to a predefined reference system. Note that only one of the systems may become real; in cases when the studied system becomes real, the reference system never takes place, and vice versa. Due to this feature, the emissions avoided cannot be measured, verified or monitored. Instead, they can only be modeled using assumptions.

2.2 Life cycle modelling principles

The term substitution is at the core of this study. In economics, substitution refers to two or more goods that the consumer perceives as similar or comparable, and thus as substitutes to each other. But what exactly is meant by this in the sense of LCA? In the context of the question posed in this report, substitution is understood as the replacement of an alternative non-wood-based product by—as the title makes clear—the production and provision of a wood-based product. In this sense, the aim here is to highlight the difference between the wood-based product and the functionally equivalent alternative product[1] concerning their respective GHG balance or net saving (or net emission) by the wood-based product. The alternative product system is also termed “reference system” (ISO 14044:2006; 4.4.3.2.2) or “comparator” within the Renewable Energy Directive; (EU, 2018/2001).

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[1] For comparative assertions, the ISO 14040:2006 requires equivalence of one product versus a competing product that performs the same function.

The term substitution is also used in LCA as a method to solve multi-functionality by reducing multi-product systems into single-product systems. In this sense, the substitution means the subtraction of the burdens avoided related to co-products that are not part of the functional unit. (Heijungs et al. 2021)

Substitution can also be understood as a market effect that is assessed to take place. This means that due to changes in the behavior of consumers or policies, market shares of products that are considered substitutes for each other, e.g. fossil fuels and biofuels, are changed so that an increase in one means a decrease in another one. A typical assumption made in most LCA studies is that one energy unit of a biofuel replaces one equivalent energy unit of a fossil fuel (i.e. perfect substitution or 1:1 substitution). However, due to market-effects, such as supply and demand elasticities and changes in prices, the share of fossil fuels substituted by biofuels (the substitution effect) may be significantly lower or in some cases even higher than the theoretical 'one to one' share (Sims et al. 2014, p. 631–632). The lower is the substitution effect compared to the perfect substitution the lower is the GHG emissions saving of biofuels in replacing fossil fuels (Soimakallio 2014). Such market-effects may be included in LCAs if they are considered relevant given the goal and scope of a study.

The different uses of the term need to be emphasized here because they are associated with highly different principles of the LCA method in each case, namely *attributional LCA* and *consequential LCA* (Finnveden et al. 2009). Both differ in their view of a system and subsequently in their general understanding of substitution and their approaches to the end-of-life (EoL) analysis of a product or a product system (JRC 2010). Regardless of their differences, both are subject to ISO 14040, 14044 and ISO 14067. Additionally, there is no clear-cut definition for either in the literature (Detzel et al. 2016). In the following box, the key characteristics and differences of both of these key modeling principles are explained.

Box 1. Differences between Attributional and Consequential LCA.

Attributional LCA (ALCA) is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems. Thus, the function served is assumed not to substitute any alternative way of serving the same function. Instead, the results of two or more stand-alone ALCA studies serving the same function measured in equivalent functional units and defined based on comparable other methodological choices, may be compared with each other. *ALCA* is an applicable method to respond questions such as "what are the GHG balances attributable to energy or material services produced from wood?" If carried out coherently and by applying comparable methodological choices, it is also applicable to make comparative assertions in accordance with ISO 14044 between different functions, such as those provided by wood and those provided by alternative raw materials. In such a case, the modeling principle can be applied to respond questions such as "*What is the difference in GHG emissions attributable to the equivalent energy or material service produced from wood or from fossil-based raw materials?*"

Consequential LCA (CLCA) has been defined by its aim to describe how environmentally relevant flows would change in response to possible decisions regarding the product system. For example, a decision may concern an increase in wood use to serve increased demand, and the consequences of a particular decision are those related to the market effects caused. The GHG emissions avoided in substitution inherently belong to the consequential modeling principle. Note that the market effects causing the consequences do not necessarily mean that each functional unit provided by alternative materials or energy is substituted equivalently by wood. For example, if the increased use of wood is achieved through certain policies, it may result in increased prices of energy or material services, thus there may be a reduced consumption of services within the policy area but an increased consumption of services outside the policy area (Rajagopal & Plevin 2013). The change in consumption of services is not necessarily 1:1 between "*with promoted wood use*" and "*without promoting wood use*". CLCA is applicable to respond questions such as "*What are the consequences on GHG emissions of a decision to increase the use of wood?*".

Concisely, the difference can be summarized as follows: While an ALCA model tries to approximate environmental burdens of a life cycle that is assumed to exist through attribution, the consequential model attempts to assess the consequences and effects of a decision that is assumed to occur (JRC 2010). Both modeling principles can describe historic, existing or future systems (Finnveden et al. 2009).

Proponents of CLCA apply the substitution in the sense of a subtraction of burdens by co-products to avoid allocation, as mentioned in the text. They consider this approach to be the same as the system expansion.[2] This understanding of substitution does not correspond to the sense in which it is understood in this study.

The choice between the ALCA and CLCA modeling principles is not only relevant concerning the term substitution (see also Chapter 4.1) but also when dealing with recycling, cascading, and EoL. Therefore, Chapter 5.1 deals with the related methodological aspects.

2.3 Connection between substitution and wood use

The fossil GHG emissions avoided by using wood-based materials and energy in place of alternative materials and energy are generated compared to a reference system. This means that the substitution credits (avoided GHG emissions) are relative to changes in the production of wood-based materials and energy between "*with the studied wood use system*" and its "*without the studied wood use reference system*".

In case the wood use system being studied is determined to include all the wood products produced within the system (e.g. country, region or company over a certain given time horizon), the coherent reference system then excludes any wood use. With wood harvests, this means that the wood harvest rates relevant to produce the studied amount of wood-based materials and energy are compared to the zero harvest rate (no harvests) in the reference system. It is further assumed that materials and energy alternative to all the wood-based materials and energy within the system being studied are used in place of wood in the reference system, and that those are replaced by wood in the wood use system.

In case the wood use system being studied is determined to include only part of the wood products produced within the system (e.g. intensification of production within a country, region or company over a certain given time horizon), the coherent reference system then excludes the particular wood use. With wood harvests, this means that more and less intensive wood harvest rates are compared. The more intensive wood harvest rate produces the studied amount of wood-based materials and energy, and the less intensive harvest rate (reference system) provides all the other required but not the intensified amount of wood. Materials and energy alternatives to the studied amount of wood-based materials and energy within the system are used in place of wood in the reference system.

Whether the wood use system should be compared with no wood use or less wood use depends on the goal and scope of a study. If the aim is to study the effects that can be attributed to the overall wood use of a system, then the former approach is relevant. If the aim is to study a change (increase) in wood use from a certain reference level, then the latter approach is relevant.

2.4 Conclusions on the key concepts and methodologies

An LCA is a suitable method to be applied to assess GHG balances related to wood use. Two different main modeling principles have been developed in LCAs: namely, attributional and consequential LCA. Both methods are suitable to be applied to study GHG balances related to use of wood and its alternatives but from

[2] https://consequential-lca.org/glossary/#determining-product

different perspectives. Attributional LCA is applied to study GHG balances of product systems while consequential LCA is applied to study GHG balances of decisions related to product systems. In this report, we handled the carbon storage of HWPs and substitution of using wood in place of alternative raw materials mainly from attributional perspective. In such a case, the substitution is generated as a difference in GHG emissions between a wood-based system and its alternative system, both serving equivalent functions, and the market-rebound effects related to the production and consumption of goods and services are ignored.

3. Carbon storage in harvested wood products

3.1 General aspects

The consideration of HWPs as a carbon storage mechanism is relatively new, although there exists a large acceptance that HWPs have the potential to reduce carbon emissions and contribute to climate change mitigation strategies (e.g. Steel 2021). Carbon storage in HWPs remain as long as HWPs are in use or disposed of so that the carbon remains unreleased to the atmosphere. Thus, carbon storage in HWPs contributes to the overall carbon balances of forest biomass use. However, reliable information to quantify the carbon storage function of HWPs is still somewhat lacking (Steel 2021). While various accounting approaches have been proposed, carbon storage in HWPs has been addressed mainly for national purposes and through approaches provided by the Intergovernmental Panel on Climate Change (IPCC) (e.g. IPCC 2019, Jasinevičius et al. 2018).

In addition to global and national level carbon assessments, forest industries are increasingly interested in characterizing the carbon storage of HWPs in use, since one of the most significant aspects of forest industries' overall impact on atmospheric CO_2 levels will occur because of impacts of sequestered carbon in their direct operations as well as along their value chain (WRI/WBCSD 2015). Information on forest industries' effects on HWP carbon storage could be used for strategic planning and for informing stakeholders, who are increasingly concerned about climate change and want forest product companies to address both GHG emissions and the sequestration of carbon in forests and HWPs in use. Additionally, assessing the carbon storage of HWPs may help identify opportunities to improve the company's GHG profile and to create value from reductions created in the value chain by companies themselves or in partnership with raw material providers or customers (WRI/WBCSD 2015). Despite the increasing interest, very little attention has been paid to designing solutions to address forest industries' corporate level carbon storage accounting.

3.2 Carbon storage accounting

Methods and calculation approaches provided by the IPCC

The most frequently applied carbon storage calculation approaches are described in detail by the Intergovernmental Panel on Climate Change (IPCC). Especially, the IPCC's production-based accounting approach has been used by many individual countries in their GHG inventory reporting, e.g. in Finland (IPCC 2019, Jasinevičius et al. 2018). In Finland, HWP is reported as a carbon stock change in production-based HWP stocks originating from wood harvested in Finland and divided into HWP produced and consumed domestically and HWP produced domestically and exported. HWP comprise solid wood products (sawn wood and wood-based panels) and paper products. However, a more detailed, country-specific classification of wood products is used as a basis. (Statistics Finland 2020)

The IPCC outlines the methods, i.e. so-called tiers, for estimating carbon storage in HWPs. The tiers mostly differ by the availability of activity data and the level of aggregation of HWPs. The latest guidance (IPCC 2019) contains the following tiers:

First-order decay (Tier 1) may be applied if sufficient data is available for the three semi-finished HWP categories, i.e. sawn wood, wood-based panels and paper and paperboard. Tier 1 should be applied using a first-order decay approach with default IPCC half-life values and emission factors.

Country-specific data (Tier 2) may be applied if it is available, e.g. based on national surveys, country-specific service and half-life information. The emission factors should be available and the country-specific activity data should be suitable to be used in conjunction with the default method provided and are as accurate as under the Tier 1 method. Estimates of the annual CO_2 emissions and removals arising from HWP are derived using the same equations as provided for the Tier 1 method.

Country-specific methods (Tier 3) may be applied if they could be used to estimate emissions and removals of CO_2 arising from HWP and the methodologies used are at least as detailed and accurate as under the Tier 1 method. Countries should clarify which approach they choose and ensure that methods for HWP and for carbon stock changes in wood-producing land categories (e.g. forests) are designed to avoid non-counting or double-counting of CO_2 emissions or removals.

The IPCC's guidance (2019) includes four main accounting approaches for carbon storage in HWPs, namely, stock-change, production, atmospheric-flow and simple-decay approaches. Generally, these differ in what percentage each country can specify to account for the global annual change in HWP storage or net emissions to the atmosphere. The different approaches may be divided according to the following:

1) The underlying conceptual framework. The first framework concentrates on the estimation of changes in carbon stocks within the defined HWP pools and requires tracking changes in carbon stocks in the HWP pool that occur from one year to another, and then deriving net emissions and removals of CO_2 from HWP from these stock changes. The second framework focuses on identifying and quantifying actual CO_2 fluxes of HWP from and to the atmosphere.

2) The system boundaries applied when calculating carbon storage, which are not necessarily similar to the national boundaries of countries.

The stock-change and production approaches are based on the first conceptual framework and the atmospheric-flow and simple-decay approaches on the second conceptual framework (IPCC 2019). These two approaches differ in how biomass carbon, including forest carbon pools, i.e. the biomass harvested to produce the HWPs as well as fuelwood, for example, are taken into account (Steel 2021). The first two approaches account for stock changes in biomass carbon pools, whereas the latter two account for carbon flows to and from the atmosphere. Furthermore, the stock-change and atmospheric-flow approaches cover the stock changes or CO_2 fluxes associated within a consuming country, whereas the production and simple-decay approaches cover those associated with a producing country regardless of where HWPs are consumed (IPCC 2019) (see Table 1).

Generally, the HWP calculation approaches can be divided according to differences in how they calculate the carbon pools of: (1) forest land, (2) domestically produced and domestically utilized HWPs, (3) exported HWPs utilized in other countries, and (4) imported HWPs from other countries utilized domestically (Sato and Nojiri 2019). The IPCC provides detailed information on the compilation of the activity data for each accounting approach as well as calculation techniques used in the estimation (IPCC 2006, 2014, 2019).

The stock-change approach estimates the net change in carbon stocks in the forest and in the HWP pool within national boundaries. Thus, carbon stock changes in domestic forests are accounted for in the reporting country, whereas stock changes in HWPs are accounted for in the reporting country where the wood products are used, i.e. reported by the consuming country (IPCC 2019). The stock-change approach estimates net changes in carbon stocks in forests and HWP pools through the carbon gain and loss. Carbon transferred from domestic forest carbon pools to the HWP pool is once accounted for as a carbon loss from the forest land pool in the reporting country and subsequently as a carbon gain in the HWP pool in the consuming countries.

The production approach also estimates net changes in carbon stocks in the domestic forest and in the HWP pool, but both are attributed to the producing country. Thus, the HWP pool consists of all wood products that are consumed domestically and those products that are exported and used in other countries. In other words, when applying the production approach, the producing country reports carbon stock changes from HWPs produced by that country regardless of where the HWPs are consumed and used. The production approach takes only domestically produced HWP stocks into account and the impact of imported wood is not evaluated. Carbon transferred from the domestic forest carbon pool to the HWP pool is once accounted for as a carbon loss in the forest land pool in the reporting country and subsequently as a carbon gain in the HWP pool of the producing (and reporting) country (IPCC 2019).

The atmospheric-flow approach estimates the flows of carbon between the biosphere and the atmosphere within national boundaries. The uptake of carbon from forest growth is accounted for in the producing country, while carbon emissions from the oxidation of wood or wood products are accounted for in the consuming country (IPCC 2019). The atmospheric-flow approach allocates the emissions from the oxidation of HWP to the consuming country, i.e. where they occur. Thus, carbon transferred from forest carbon pools to the HWP pool is not accounted for as a carbon loss in forest land pools in the producing country, but as emissions at the time of the end-of-life of the HWP in the consuming country.

Similarly to the atmospheric-flow approach, the simple-decay approach handles actual fluxes of carbon associated with HWPs into the atmosphere. However, the simple-decay approach covers CO_2 emissions arising from wood harvested by the producing country, so that emissions from HWPs and woody biomass used for energy are reported by the producing country (IPCC 2019). Thus, the carbon transfer from forest carbon pools to the HWP pool is not accounted for as a carbon loss in the forest land pools of the producing country, but as emission from the HWP pool at the time of end-of-life of HWPs in the producing country. As in the atmospheric-flow approach activity data is required, including both feedstocks for processing wood for its use as a material and wood biomass burned for energy (IPCC 2019).

Additionally, a stock-change approach of domestic origin (SCAD) has been described in the literature (Sato and Nojiri 2019). However, the IPCC does not consider SCAD as an independent approach (Sanquetta et al. 2019). The SCAD approach estimates net changes in carbon stocks in forest and HWP pools (Sato and Nojiri 2019). The SCAD approach takes only domestically produced stocks consumed within the producing country into account, while the impacts of imported and exported wood are not evaluated. Changes in the forest carbon pool are accounted for in the country in which the wood is grown, i.e. the producing country. The carbon transferred from the forest carbon pool to the HWP pool is once accounted for as a carbon loss in the forest land pool of the producing country and subsequently accounted as a carbon gain in the HWP pool in the producing country, but only for domestically consumed HWPs (Sato and Nojiri 2019).

Table 1. Each HWP carbon storage calculation approach includes the pools in which the carbon stock changes are estimated in light grey and excludes the ones in white (figure adopted from Sato and Nojiri, 2019).

Carbon storage accounting approaches		System boundaries (pools in which carbon stock changes are estimated)			
Pool-based method	Flux-based method				
Production approach	Simple-decay approach	Forest land carbon pools	HWP pool domestically produced and used	HWP pool exported and used in other countries	HWP pool imported and used domestically
Stock-change approach	Atmospheric-flow approach	Forest land carbon pools	HWP pool domestically produced and used	HWP pool exported and used in other countries	HWP pool imported and used domestically

SCAD		Forest land carbon pools	HWP pool domestically produced and used	HWP pool exported and used in other countries	HWP pool imported and used domestically

A simple first-order decay (FOD) equation which is based on the exponential decay distribution may be applied when reliable data for sawn wood, wood-based panels and paper and paperboard are available, but there is no adequate country-specific data for more advanced methods (IPCC 2019). The availability of data for at least the three aggregate semi-finished HWP commodities allows estimating the HWP carbon stocks and the changes in these stocks based on the production or stock-change approaches (IPCC 2019). The method to estimate the amount of carbon stock in the specified HWP pool in use and its changes applying the first-order decay function is as follows:

$$c_i(i+1) = e^{-k} \bullet c_i(i) + \left[\frac{1-e^{-k}}{k}\right] \bullet Inflow_i(i) \quad (1)$$

$$\triangle\, c_i(i) = c_i(i+1) - c_i(i), \quad (2)$$

, where

=year

=the carbon stock of the HWP pool in the beginning of year, Gg C

= decay constant of first-order decay given in units, (, where HL is half-life of the HWP pool in years. A half-life is the number of years it takes to lose one-half of the material currently in the pool.

= the inflow to the HWP pool during year , Gg C

= carbon stock change of the HWP pool during year , Gg C .

Depending on the choice of the approach, which also determines the system boundaries, the annual carbon inflow to the carbon stock of the specified HWP class may be calculated as follows:

$$Inflow_i(i) \coloneqq \begin{cases} Inflow_{SCA_l}(i) & for\ the\ stock-change\ approach \\ Inflow_{PA_l}(i) & for\ the\ production\ approach \end{cases} \quad (3)$$

, where

$Inflow_{SCA_l}(i)$=carbon inflow in HWP from the calculated domestic consumption of the respective HWP commodity class l in the year i, in Mt C y^{-1}

$Inflow_{SPA_l}(i)$=carbon inflow in HWP from the production of the respective HWP commodity class l originating from domestic harvest in the year i, in Mt C y^{-1}.

If only the future carbon stocks and the subsequent carbon stock changes of HWPs in use are considered, the carbon stock in the initial year is set to zero and the accumulation of future inflows to the HWP stock in use is taken into account for a specified time horizon. In order to produce an estimate of the existing carbon stock of HWPs in use and the subsequent changes of this HWP stock, the historical wood use, i.e. the accumulation of the historic inflow to the HWP pool, has to be included (IPCC 2006). However, the availability of activity data series varies and especially long-term historical data may not always be available. Thus, the latest IPCC guidance (2019) suggests applying the average value of over the first 5 years since 1990 (e.g. for the years 1990 to 1994) or later. The approximation of the carbon stock at initial time, i.e. the starting year for which data is available is calculated as:

$$c_t(t_0) = Inflow_{l_{average}} / k\,(t) = \frac{Inflow_{l_{average}}}{k} \tag{4}$$

with

$$Inflow_{l_{average}} = \left(\sum_{i=t_0}^{t_4} Inflow_i\,(i)\right) / 5 \tag{5}$$

, where

k= decay constant of first-order decay for each HWP commodity class l.

When additional data is available, more advanced methods may be applied. These include e.g. additional country-specific sources of activity data, disaggregated half-life estimates and emission factors as well as country-specific models (Steel 2021). The IPCC provides good practices and principles for the application of additional data and more advanced methods (IPCC 2019). For example, the calculation and reporting of CO_2 emissions from wood biomass used for energy depend on the choice of HWP approach in terms of the conceptual framework and the system boundary of each approach. The IPCC guidance clarifies when CO_2 emissions from wood biomass burned for energy are reported in GHG inventories and whether they are estimated by a producing or consuming country.

Wood product models

Specific carbon accounting models or wood product models are also frequent in literature. The wood product models have been developed to estimate carbon stocks in HWPs and changes in these stocks. Wood product models that simulate carbon balance of wood production, HWP in use and at the EoL usually complement forest growth models to evaluate the mitigation potential of the forest sector. Wood product models use the allocation of harvested carbon for different purposes to estimate the carbon inflow into different HWP classes and to evaluate how the inflow evolves during the chosen time horizon. Some wood production models are specifically built for estimating carbon stock changes to report to the UNFCCC using the IPCC guidelines (Brunet-Navarro et al. 2016). These models can be easily applied in any country using data from the FAOSTAT database, for example. Additionally, many country-specific models have been developed for the same purpose, but they require country-specific data. A simple structure may be adequate for wood product models that are constructed to estimate carbon stock in HWPs. However, more complex models are required to analyze climate change mitigation options, for example. Thus, awareness of which model characteristics are relevant for a specific model application is required. Brunet-Navarro et al. (2016) and Jasinevičius et al. (2015) have conducted extensive reviews of the wood product models.

Usually, wood product models allocate carbon from harvested wood to HWPs in use via processes of primary (e.g. sawmills or wood-based panel producers) and secondary (e.g. construction, furniture, or packaging) wood processing industries, paper, and energy industries (Brunet-Navarro et al. 2016). Industrial processes, recycling and disposal define the allocation parameters used in each transformation step. Some models allow these parameters to change over time to account for technical improvements or behavioral changes, for example. Brunet-Navarro et al. (2016) identified two types of models according to the way they present industrial processes. The first type considers industrial production as an input, thus industrial processes are not represented. The second type uses harvested wood as the input and industrial processes are represented by allocation parameters. Information on allocation parameters may be acquired from industry surveys, life cycle inventories, expert knowledge or by applying parameters from previous studies, for example.

Applying specific wood product models should include a sensitivity analysis, calibration, validation and an uncertainty analysis (Brunet-Navarro et al. 2017). The goal of the sensitivity analysis is to gain an overview of which parameters have stronger impacts on the results, while calibration aims to improve the estimates of all parameters. Validation compares how close to reality the results of the models are, while an uncertainty analysis evaluates how certain the user can be regarding the obtained outcome. However, the main problem in applying wood product models is the lack of data concerning relevant and reliable time- and location-specific data regarding industrial processes, use phase of HWPs as well as data concerning lifespan and removal rate of HWPs (Brunet-Navarro et al. 2016).

Regional applications for carbon storage accounting

So-called carbon-offset programs have been developed to respond to a need for the means to monitor the contribution of HWPs to carbon pools and GHG mitigation at sub-national scales. Forest owners may not have the tools to accomplish monitoring goals established at the national level, but they need accessible and practical tools for estimating and monitoring carbon stocks and the flux in HWPs at the agency or company level (Stockmann et al. 2012, Anderson et al. 2013). The programs have emerged as a strategy for climate change mitigation. Offset projects sequestering carbon earn credits that can be traded on the cap-and-trade market to compensate for carbon emissions (Bates et al. 2017). Carbon offset projects are an inventory to show that carbon discharged elsewhere has been offset by carbon storage in forests and HWPs. The cap-and-trade program and other mitigation programs include carbon offsets as an avenue for compensating for emissions, and forest project owners receive credits when they can verify that their actions have resulted in carbon sequestration above and beyond what typical practices would have caused. For example, the California Forest Project Protocol (CFPP) was designed to apply to smaller geographic areas and uses a simpler accounting approach focused on carbon storage for a single harvest year rather than the net annual carbon change due to current year additions to HWP pools and current year emissions from those pools (Anderson et al. 2013). The protocols such as the Forest Project Protocol (FPP) by Climate Action Reserve (Nickerson et al. 2019) and the Compliance Offset Protocol U.S. Forest Projects by the California Air Resources Board (CA ARB) (California ARB 2015) provide requirements and guidance for quantifying the net climate benefits of activities that sequester carbon on forest land. The protocols base the accounting of HWPs on the average amount of carbon sequestered over 100 years. The 100-year average has been chosen since GHG reductions and GHG removal enhancements must be effectively permanent enough, thus sequestered carbon associated with GHG reductions and removals must remain stored for at least 100 years (California ARB 2015).The 100-year average approaches apply mill efficiency factors and decay curves for individual product classes to estimate the average amount of carbon that is likely to remain stored in HWPs in use from a given year’s harvest over 100 years of time (Stockmann et al. 2012, Anderson et al. 2013). The estimation of tree harvesting and determination of carbon stored in HWPs in use require:

- accounting for the CO_2-eq. associated with harvested trees,
- determining the amount of carbon in the trees harvested that is delivered to mills,
- accounting for mill efficiencies,
- estimating the average carbon storage over 100 years in HWPs in use,
- summing the results to determine the total average carbon storage over 100 years (California ARB 2015).

Approaches for company-level carbon storage accounting

Consensus methods have not yet developed for company-level carbon storage accounting, although accounting methods would support forest product companies to consistently and transparently explain their overall carbon pool to stakeholders (WRI/WBCSD 2015). Generally, companies should carefully explain the methods used to maintain transparency. Setting operational boundaries for carbon inventories would help companies transparently report their impacts on stored carbon along their value chain. The reporting should consider which pools are included in the analysis and which are not and the rationale for the selections, for example. The base year data considering carbon storage may need to be averaged over multiple years to accommodate the year-to-year variability. The temporal scale used in carbon storage accounting may often be closely tied to the spatial scale over which the accounting is done (WRI/WBCSD 2015).

In some cases, the calculation approaches used in national inventories can be adapted for the company-level quantification of HWP carbon storage. However, national accounting approaches are focused on current HWP storage, thus they are not particularly useful for examining opportunities for future improvements. The national accounting approaches are heavily influenced by factors that affect the size of the current pool, i.e., the amounts of past production and historical product use patterns, making it difficult to use the results to characterize future performance. Attempts have been made to overcome the retrospective nature of national accounting approaches by allowing companies to consider only the current and future production (Miner, 2006). Additionally, national accounting approaches to the carbon storage of HWPs in use may not be suitable for company-level accounting due to the required technical details, the complexity of the calculations and data requirements that are usually unavailable at the sub-national level. The national accounting approaches require past production and data on product use that cannot be disaggregated to an individual company level (e.g. Miner 2006). For example, the IPCC’s production approach requires data on the stock of HWPs from domestically grown wood, which relies on assumptions as the data may not be directly available. The production approach lacks transparency due to the number of assumptions required. The basic difficulty is to follow the life cycle of harvested wood over country borders. Wood harvested in one country can be transported as roundwood to a second country, where it is processed and transported as a semi-finished product to a third country, where it is finished and used. Additionally, HWPs can be mixtures of wood harvested in several countries. Further, the true end use in a one country of wood harvested in a second country is not known as the first country may import roundwood from many other countries and often of different quality and for different end uses (Pingoud et al. 2003).

Miner (2006) suggested a so-called 100-year method for companies to characterize the carbon pool in HWPs in use. The method is suitable for the corporate sector and value chain carbon storage accounting. In contrast to other carbon storage calculation approaches, the 100-year method is limited to the technosphere, i.e. the amount of wood biomass harvested to produce the HWPs in use is not taken into account. The method uses information on the expected time-in-use of HWPs to estimate the amount of carbon therein that will still be sequestered in HWPs in use for the next 100 years. The method applies a 100-year time horizon, although the selection of the time horizon for the estimations is somewhat arbitrary. The 100-year method applies many of the national carbon storage accounting concepts, but instead of estimating changes in current stocks of carbon in HWPs in use, it estimates future changes in HWP stocks attributable to current production. Current year additions to carbon stocks of HWPs in use are allocated against future losses from current year additions. The result is the amount of carbon in the current year’s production that is expected to remain in use for a defined period. The calculations for each year’s production are independent of past years’ production (Miner 2006).

The 100-year method is both conceptually and mathematically simple, thus it is easy to use by companies, and therefore it will more probably be applied consistently from one assessment to the next than the national carbon accounting approaches. Such considerations are especially important in company-level carbon accounting, where transparency and consistency are important for maintaining stakeholders’ trust. The 100-

year method also provides information that reflects conditions and improvement opportunities that may be applied to current production. The method produces information that could be used in policies intended to promote the use of products that store carbon, especially for long-lived HWPs. The applied time horizon is long enough to focus attention on HWPs that are unlikely to return carbon to the atmosphere over a short to medium timespan, while being short enough to recognize the value of temporary sequestration in mitigating the rise in atmospheric CO_2 (Miner 2006). Applying the 100-year method includes five steps:

- Identify the types and amounts of wood-based products (e.g. sawn wood) that are made in the year of interest and enter final product (e.g. house building).
- Express this annual production in terms of the amount of biogenic carbon per year for each product.
- Divide the products into categories based on their function and allocate the carbon to these functional categories. Some functions for sawn wood could be single-family homes, other construction wood or packaging wood, for example. If it is impossible to identify the end uses, production can be divided into the categories for which required information has been developed for national or international accounting.
- Use decay distribution or other time-in-use information to estimate the fraction of carbon in each functional category expected to remain in use for 100 years.
- Multiply the amount of carbon in the annual production in products in each functional category by the fraction remaining at 100 years. The result is the amount of carbon stored in the products in each functional category attributable to the selected year's production.

Data on production amounts may be obtained from production records or statistics. Then, the carbon content is estimated by multiplying the production amount by its carbon content. Since the wood producing companies may not have information on the specific end uses or final products, semi-finished or intermediate product groups may be applied. Furthermore, companies may calculate alternative scenarios considering their semi-finished production going to different final products. The current production is further divided into such functional categories for which lifespan or half-life estimates are available, since these are required in applying decay distributions. The decay distributions are often represented by mathematical equations that describe the decay pattern of HWPs. A key parameter in these equations is usually the HWP's half-life, i.e., the time over which one-half of the original carbon leaves the pool of HWPs in use (IPCC 2006). Nevertheless, the results of the 100-year method are sensitive to the selection of decay distributions. Existing decay distributions, which have often been created to develop national carbon inventories, should be used in the 100-year method only after their suitability for making long-term projections has been evaluated (Miner 2006).

3.3 Product-level carbon storage

The IPCC's carbon storage approaches have also been applied at the product level, and at the national or large regional scale. Product-level carbon storage estimations have been conducted, e.g. for cork products (Dias and Arroja 2014) and particleboard and fiberboard (Canals et al. 2014) applying the IPCC's stock-change, production, and atmospheric-flow approaches as well as logs applying the production approach (Manley and Evison 2018). However, solely at the product level, an LCA is the more frequently applied methodology to assess the environmental impacts of products that consider their entire life cycle (e.g. Brandão and Levasseur 2011). Since carbon removal from the atmosphere and storage in HWPs may have the potential to help mitigate climate change, there has been increasing debate concerning accounting for temporary carbon storage in the LCA of HWPs. Currently, LCA methodology does not permit benefits to temporarily keep carbon away from the atmosphere. Concerning delayed emissions due to carbon storage, most LCA studies consider that CO_2 is released as a single pulse after a specific storage period in the biosphere or in the anthroposphere (Cherubini et al. 2012). A suitable method is needed to account for the possible benefits of temporary carbon storage for use in the LCA of products. However, there is no consensus on how to account for temporary carbon storage of a product within the context of LCA (e.g. Brandão and Levasseur, 2011). The issues regarding the importance of temporary carbon storage and timing of GHG emissions are discussed in detail in Chapter 6.

3.4 Data requirements for carbon storage accounting

Carbon storage accounting requires information on inflows of HWPs into the carbon pool, carbon conversion factors of the HWPs, the lifespan or half-life values of the HWPs, as well as decay patterns of the products (e.g. Pilli et al. 2017). The latest IPCC guideline (2019) proposes estimating carbon stocks for the three default HWP categories, i.e. paper and paperboard, wood-based panels, and sawn wood, but carbon stocks can also be accounted for in more detailed HWP categories. The IPCC's Tier 1 method suggests using the default HWP categories and activity data provided by the Food and Agriculture Organization (FAO) in addition to the IPCC's default half-life values and carbon conversion factors.

Carbon conversion factors

The carbon conversion factors for the different HWP subcategories are largely dependent on the composition of countries' production amount of each particular subcategory (e.g. particle board). The IPCC default carbon conversion factors for the three default categories are 0.229 Mg C/m^3 for sawn wood (coniferous 0.225 and non-coniferous 0.28), 0.269 Mg C/m^3 for wood-based panels and 0.386 Mg C/Mg for paper and paperboard. Wood-based panels are further divided into more specific subcategories (e.g. medium-density fiberboard (MDF) and oriented strand board (OSB) each with their own carbon conversion factors (IPCC 2019). Additionally, IPCC guideline (2019) includes default carbon conversion factors for imports and exports of woody biomass serving as wood fuel and raw material (pulp) for the subsequent manufacturing of semi-finished HWP to be used with the atmospheric-flow approach. The carbon conversion factors for the selected UPM products were acquired from IPCC (2019), Rüter (2011) and UPM's Environment Product Declarations (UPM 2021) (see Appendix A). Thus, the data was not very comprehensive, especially concerning different fiber products, biochemicals and bio composites (see Appendix A).

Lifespan and half-life values

Forest products have various end uses, and the expected lifespans even within a single product type can vary substantially. Thus, it is important to understand how forest products are used, since information on the lifespans or half-lives of HWPs is typically associated with specific end use functions. The lifespan of an HWP often means the time needed so that majority of the HWP pool has decayed (e.g. 90% or 95%).

Consequently, the average lifetime of HWPs in use is much shorter than their lifetime under ideal conditions, in which the lifetime can be hundreds of years, e.g. wooden frames in buildings (e.g. Pingoud et al., 2003). Considering HWPs from the viewpoint of atmospheric carbon balance, the lifetime of an HWP consists of the time in use and possible recycling for re-use and the time the HWP is out of service e.g. in landfills.

The lifespan values were scarce in the literature since most of the decay distributions apply half-life values. The average lifetime or lifespan values presented in Appendix A were acquired from Pingoud et al. (2003), Marland et al. (2010), Dias et al. (2011), Braun et al. (2016) and Iordan et al. (2018). The values include minimum, maximum and average values based on the literature for the final products. The lifespan values considering sawn wood used for buildings acquired from Pingoud et al. (2003) seemed to be quite extreme, especially as the values concerned single-family and multi-family houses.

The half-life is the time required for carbon stored in HWPs to decrease by half (IPCC 2019). Clearly, half-lives differ for HWPs both for the semifinished HWPs and final products. The IPCC default half-life values for default semi-finished HWP categories are 35 years for sawn wood, 25 years for wood-based panels and 2 years for paper and paperboard. The half-life values for the selected final products produced by UPM were derived from IPCC (2019), Pingoud et al. (2003), Skog (2008), Marland et al. (2010), Pearson et al. (2012), Canals et al. (2014), and Braun et al. (2016). The acquired minimum, maximum and average values for the final products are presented in Appendix A. There was no literature concerning the half-life data for biochemicals and bio composites.

Decay distributions

One key issue in carbon storage accounting is oxidation or decay patterns of HWPs, which are defined by a chosen statistical distribution and by the time after production when a certain percentage of the product remains in use (Brunet-Navarro et al. 2016). In some cases, HWPs remain in use for very long time. Many building materials belong to this category. However, for tissue papers, the use phase is very short. Although statistics on the production and international trade rates of HWPs are compiled, the decay and disposal rates of HWPs are not very well known (Pingoud et al. 2003). There are many approaches to incorporate the dynamic nature of carbon flows in an HWP's life cycle explicitly considering the timing of carbon emissions.

Many distributions have been applied, including uniform, linear, exponential, logistic, normal, chi-squared and gamma distributions. The decay distribution is defined by one or two of the following descriptors defining the years after production: median or 50% carbon left, i.e., half-life, 5% of carbon left, mean, or average life and mode, or time at the maximum rate of carbon loss. Also, other parameters exist, but they are based on the previous. For example, a linear distribution assumes a constant annual oxidation rate, while a normal distribution uses the mean and standard deviation, and a gamma distribution requires the definition of shape and scale parameters. The selection of a distribution function may have a substantial effect on the resulting carbon stock calculations (Brunet-Navarro et al. 2016).

The simplest way is to assume that HWPs will be for an equally long time in use before they are fully discarded at the end of their lifetime. Many LCA studies concerning delayed emissions from HWP carbon storage assume that CO_2 is released as a single pulse after a specific storage period in the biosphere or in the anthroposphere (Cherubini et al. 2012). Another simple way is to assume that a fixed fraction of the initial amount of HWP is discarded or decayed every year, which results in a linear decay over time (Pingoud et al. 2003). The maximum time it takes before all HWP is gone, is the inversion of the discard or decay rate. The decay distribution can be modeled with a uniform distribution (Cherubini et al. 2012).

The exponential distribution is currently the dominant approach to model decaying rate of HWPs as it is recommended by the IPCC guidelines for national GHG inventories (Cherubini et al. 2012, IPCC 2019). The first-order decay (FOD) model assumes that the decaying rate is proportional to the size of the pool meaning that every year a fixed fraction of the current amount of HWP is decayed every year. This fraction is expressed by decay constant of the FOD model for each HWP commodity class by the decay constant, where HL is the half-life of the particular HWP commodity in the HWP pool in years (IPCC 2019). The fraction of carbon (FR) remaining in use in the year can be calculated as the following:

$$\mathrm{FR} = \left(\frac{1}{1+(\ln(2)/\mathrm{HL})}\right)^{i}, \qquad (6)$$

, where HL is half-life in years and is the elapsed time in years (Miner 2006).

The rate of decay over time according to the exponential distribution means that the largest decay occurs in the first years after production, and then gradually decreases over time approaching zero asymptotically (Cherubini et al., 2012). Long-lived HWPs are unlikely to decay or be taken out of service in the first years after production as it is assumed in the exponential decay distribution (Marland et al., 2010). Thus, the exponential decay can be an oversimplification of the real decay rate as the decay peak will more likely occur around the mean life of the HWPs (Iordan et al., 2018). The exponential distribution is useful for products that can be treated as a single pool and in which decay occurs most rapidly just after the production including fuel and other short lived HWPs such as tissues, for example.

There exist more realistic probability distributions centered around the mean half-life of the product, e.g. a gamma distribution or a chi-squared distribution. According to Marland et al. (2010), the gamma distribution is much more qualitatively accurate for modeling the decay of long-lived HWPs. The gamma distribution assumes that the rate of decay would peak around the half-life of HWP. Thus, the gamma distribution has a more realistic decay pattern, but the difficulty in modeling the decay with the gamma distribution is that it requires two parameters to be specified, i.e. the mean half-life and year of expected 95% decay or the year of maximum decay (Bates et al., 2017; Iordan et al., 2018).

The chi-squared and standard gamma distributions are alternative one-parameter simplifications of the gamma distribution that can be used to model the decay of HWPs. The chi-squared distribution is a special case of the gamma distribution as it only requires the mean half-life of the product to shape the bell-like decay curve (Iordan et al., 2018). In the chi-squared distribution, the scale parameter is set to be equal to 2 and the shape parameter is varied, while in the standard gamma distribution the scale parameter is 1 and the shape parameter is varied (Bates et al., 2017). According to Bates et al. (2017), both the chi-squared and standard gamma distributions could model most of the decay of HWP occurring around the time of the product's half-life.

Row and Phelps (1996) applied a piecewise distribution to the decay of HWPs. The piecewise structure of the distribution was conceptually simple to implement. The Row and Phelps approach divides the decay curve into three pieces and the fraction of carbon (FR) remaining in use in year can be calculated as the following:

$$\mathrm{FR} = 1 - \left(0.4191 \frac{i}{\mathrm{HL}}\right), i < \mathrm{HL}/2 \tag{7}$$

$$\mathrm{FR} = 1 - \left(\frac{0.5}{1 + (2\ln(\mathrm{HL}/i))}\right), \quad i > \mathrm{HL}/2 \; and \; i < \mathrm{HL} \tag{8}$$

$$\mathrm{FR} = \left(\frac{0.5}{1 + (2\ln\,(i/\mathrm{HL})}\right), \quad i > \mathrm{HL} \tag{9}$$

, where HL is the half-life in years and is the elapsed time in years (Miner, 2006).

Alternative decay distributions were compared by calculating the fraction of carbon removed (Fig. 1 and 2) and the fraction of carbon remained (Fig. 3 and 4) in the carbon pool of medium-lived sawn wood and short-lived pulpwood. The decay distribution chosen for the analysis was linear, exponential and gamma distributions and Row and Phelps (1996) decay curve. The average half-life values and gamma parameters were acquired from Marland et al. (2010). The average half-life values were 44.2 years for sawn wood and 1.32 years for pulpwood.

The decay rate as a function of time was at its maximum at the beginning year for the linear and exponential distributions and near the half-life for the gamma and Row and Phelps distributions. Thus, the former overestimated the decay at the beginning, especially for sawn wood. The Row and Phelps distribution peaked exactly around the half-life and the gamma distribution slightly before the half-life for sawn wood (Fig. 1). The maximum decay rate was slightly larger for sawn wood when applying the gamma distribution, and vice versa for pulpwood. The difference between the decay rates were markedly larger for sawn wood than for short-lived pulpwood. Except for the linear distribution, all decay distributions asymptotically approached zero as the time increased considering both the fraction of carbon removed and the fraction of carbon remaining. The gamma distribution reached a zero decay rate sooner than the exponential distribution. Additionally, the fraction of carbon remaining was zero earlier for the gamma distribution. The fraction of carbon remaining was larger and did not reach zero close to the other distributions when applying the Row and Phelps distribution.

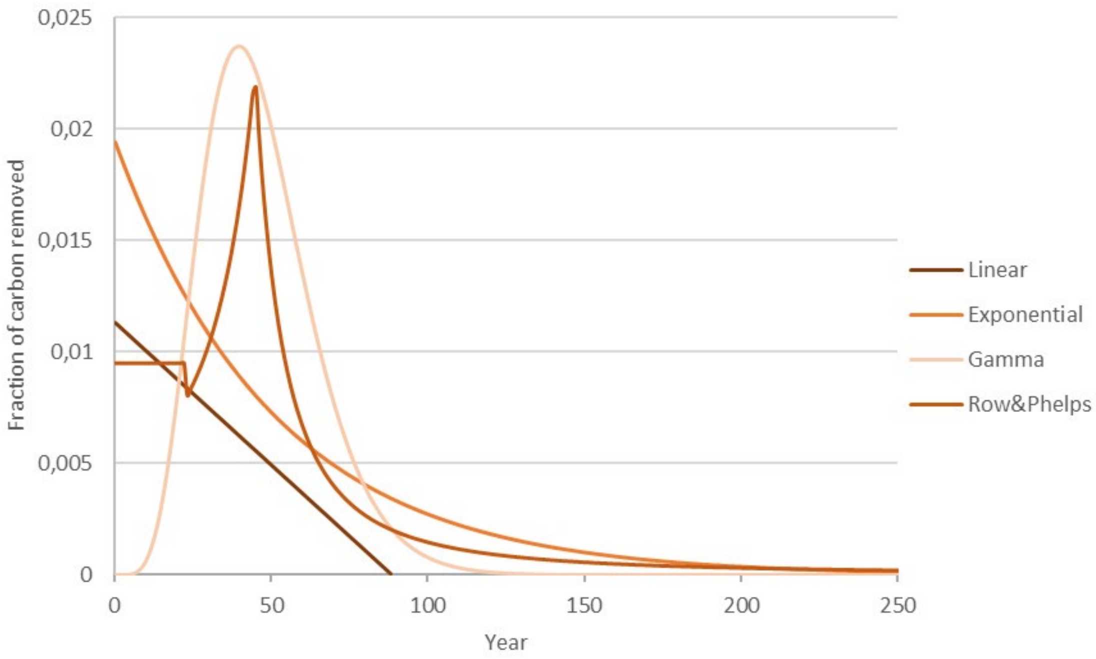

Figure 1. Fraction of carbon removed from the sawn wood carbon stock as a function of time applying the different decay distributions.

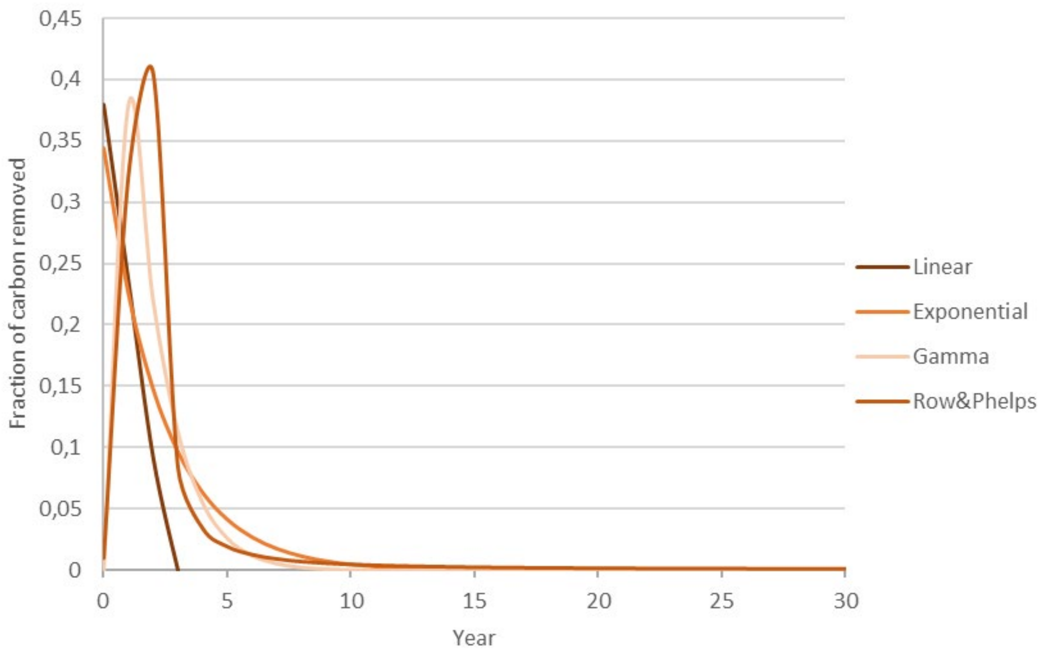

Figure 2. Fraction of carbon removed from the pulpwood carbon stock as a function of time applying the different decay distributions.

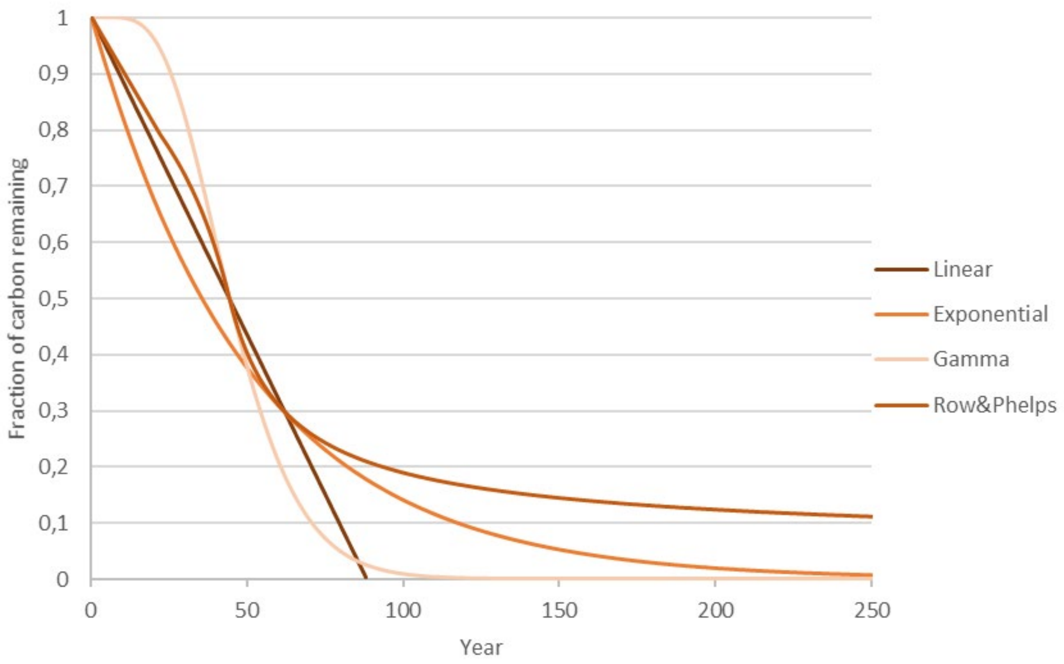

Figure 3. The remaining fraction of carbon for sawn wood as a function of time applying the different decay distributions.

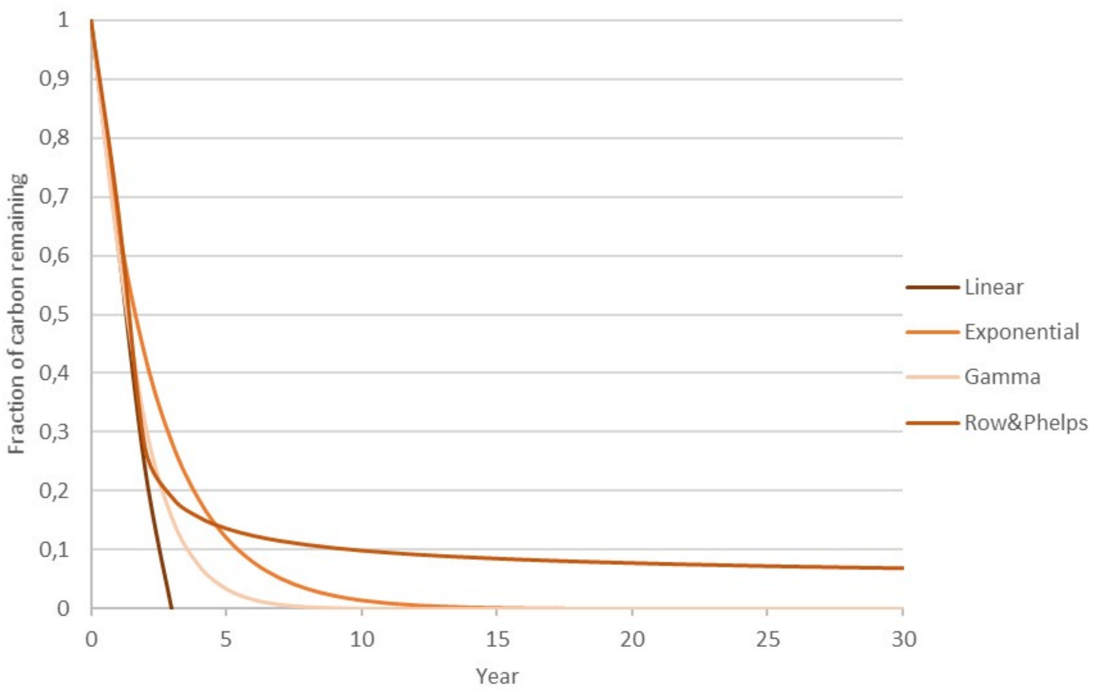

Figure 4. The remaining fraction of carbon for pulpwood as a function of time applying the different decay distributions.

At the end of use, wood products may also be collected and transformed into new products to be recycled. Recycling and cascading use affects the length of time the carbon remains in the use phase. The higher the utilization rate, i.e. the fraction of the product reused to make a new product, the longer the carbon remains in use (Miner and Lucier, 2003).

Scenario analysis of the half-life values

The literature-based carbon conversion factors and half-life values were compared to the IPCC's default carbon conversion factors and half-life values applying two different scenarios. The purpose was to examine how one year's production would decay over time and how much carbon is left in the future. The decay distribution used was the exponential decay as in the IPCC's recommendations. The production amount was based on UPM's one-year production amounts of the different HWPs. Only sawn wood, wood-based panels and paper and paperboard were included, since there are no IPCC's default values for biochemicals and bio composites. The first scenario included more short-lived products and the second one more long-lived HWPs. The applied scenarios are presented in Table 2.

Table 2. The applied scenarios, where scenario 1 contains more short-lived HWPs and scenario 2 more long-lived HWPs.

<table>
<tr><th></th><th>Sawn wood</th><th>Wood-based panels</th><th>Paper and paperboard</th></tr>
<tr><td rowspan="3">Scenario 1</td><td rowspan="3">25% for house construction, 25% for other construction wood materials, 25% for furniture production and 25% for packaging and pallets (average half-life)</td><td rowspan="2">80% of plywood for construction and 20% for molds (average half-life)</td><td>40% of chemical softwood pulp for cardboard, 40% for tissue products and 20% for copy papers (average half-life)</td></tr>
<tr><td>40% of chemical hardwood pulp for cardboard, 40% for specialty papers (packaging material and labels) and 20% for copy papers (average half-life)</td></tr>
<tr><td>100% veneer for parquet flooring (average half-life)</td><td>80% of other paper products having average half-life and 20% of other paper products having maximum half-life</td></tr>
<tr><td rowspan="3">Scenario 2</td><td rowspan="3">60% for house construction, 20% for other construction wood materials and 20% for furniture production (average half-life)</td><td rowspan="2">50% of plywood for construction and 50% for molds (average half-life)</td><td>20% of chemical softwood pulp for cardboard, 20% for tissue products and 60% for copy papers etc. (average half-life)</td></tr>
<tr><td>20% of chemical harwood pulp for cardboard, 20% for specialty papers (packaging material and labels) and 20% for copy papers etc. (average half-life)</td></tr>
<tr><td>100% veneer for parquet flooring (average half-life)</td><td>40% of other paper products having average half-life and 60% of other paper products having maximum half-life</td></tr>
</table>

The initial carbon stock was altogether about 5 642 000 Mg C for scenario 1 and 5 438 000 Mg C for the scenario 2 applying the literature-based values, and 5 778 000 Mg C for both scenarios applying the IPCC default values. After 100 years, the carbon stocks applying the literature-based values were about 57 600 Mg C (1% of carbon remaining) and 119 000 Mg C (2.2% of carbon remaining), respectively. Applying the IPCC default values the carbon stocks after 100 years were about 59 000 Mg C (1% of carbon remaining) for scenario 1 and 73 700 Mg C (1.3% of carbon remaining) for scenario 2. The percentages of carbon remaining after 100 years were about 12% for sawn wood, 8% for wood-based panels and zero for paper and paperboard applying the literature-based values in scenario 1, and 24% for sawn wood, 16% for wood-based panels and 9% for paper and paperboard in the scenario 2. The remaining carbon content values were 14% for sawn wood, 6% for wood-based panels applying the IPCC default values in scenario 1 as well as 14% and 13% in scenario 2, respectively. The remaining carbon stocks of paper and paperboard were zero for both scenarios applying the IPCC default values (Fig. 5 and 6).

The percentages for carbon remaining after 200 years were quite similar for both methods in scenario 1, but there was a marked difference for sawn wood in scenario 2 (Fig. 5 and 6). The figures showed that with lower half-life values, the difference between the literature-based values and IPCC default values was greater, although the decay pattern was similar. The IPCC default values especially underestimate the decay of short-lived sawn wood and wood-based panels at the beginning of the chosen time horizon (Fig 5). The decay rate of wood-based panels was larger for the IPCC method in the early years, but changed after ca. 65 years. The IPCC default values underestimate the decay rate of wood-based panels in the beginning compared to the literature-based values. Considering sawn wood, the decay rate was similar for the beginning, but after 25 years the IPCC default values overestimated the decay rate compared to the literature-based values. Both methods and scenarios gave quite similar decay patterns for paper and paperboard products, since even at maximum half-life values of the short-lived paper products were relatively small. However, a similar pattern was detected as in other semi-finished HWP classes, i.e. scenario 1 overestimated and scenario 2 underestimated the decay or discard of the paper and paperboard products applying the IPCC's default half-lives (Fig. 5 and 6). Thus, the results showed that for some products the difference may be small, but for others, particularly those with a long life expectancy, the difference can be significant.

Uncertainties involved in carbon storage accounting

The IPCC guidelines propose identifying, quantifying, and reducing uncertainties as much as practicable. The uncertainties may be divided into uncertainties associated with applied methods, activity data, and with emission factors and parameters (IPCC 2019). Additionally, Pingoud et al. (2003) have provided an analysis of the uncertainties and problems associated with different HWP calculation approaches and data gathering methods. Acquiring reliable activity data, i.e. the annual production number of different HWPs, usually does not pose a problem at a forest company level, but future production amounts in evolving HWP markets may be uncertain, especially, if the time horizon is long. Furthermore, forest companies have information on their intermediate or customer products, but the companies do not always know the specific final use of the products.

The half-life values are in general the most uncertain part of the carbon storage calculations. The main problem is the lack of data for estimating HWP removals and decay rates, since reliable data regarding HWP time in use is lacking (Brunet-Navarro et al., 2016). The half-life or lifespan estimates of HWPs in use appear to be uncertain and the same applies to their decay patterns. Thus, carbon storage accounting heavily relies on assumptions. Additionally the applied decay distributions differ and the exponential decay distribution especially is not well suited for long-lived HWPs (e.g. Marland et al. 2010). These conditions lead to uncertain estimates of the climate change mitigation effect of HWP usage, and therefore weaken the climate change mitigation claims of the forestry sector. Further uncertainties associated with activity data are caused by conversion factors. In particular, the conversion factors provided for sawn wood reflect averages, which are likely to be under- or overestimated for specific wood products (IPCC 2019). Additionally, lifespan and half-life values for recently introduced wood-based products such as wood-based textiles, biochemicals and bio composites are absent in the literature, although their market share is increasing in the forest sector, and some of them may constitute a relatively long carbon storage.

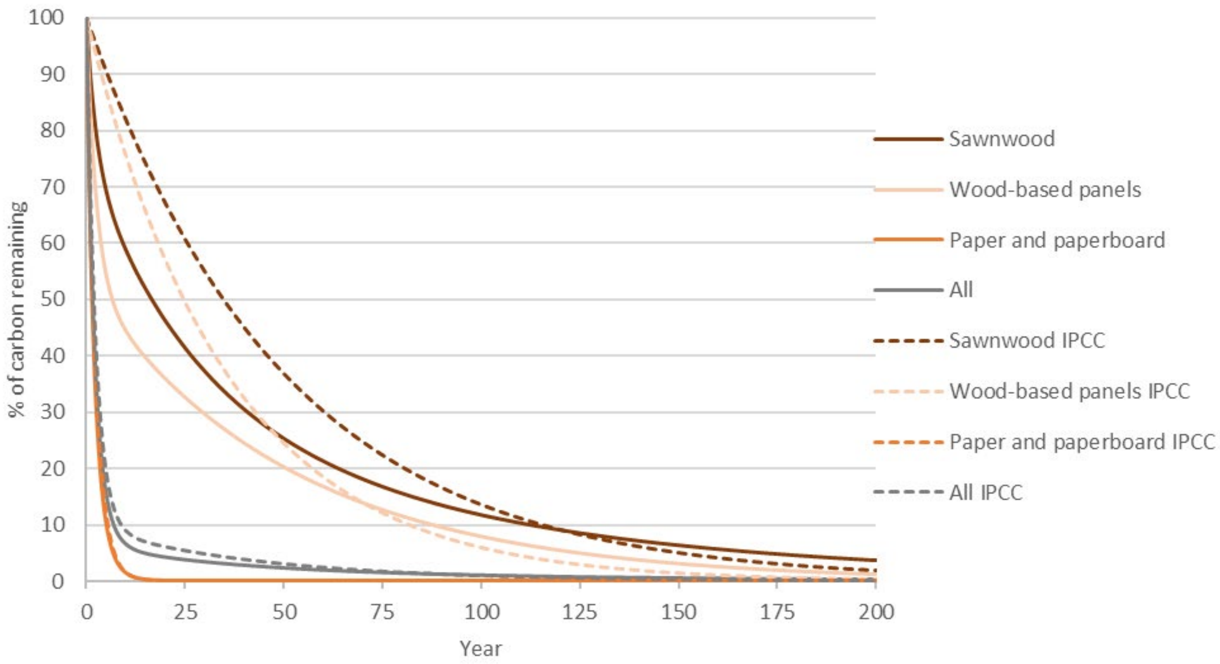

Figure 5. Percentages of carbon remaining after 200 years in the sawn wood, wood-based panels and paper and paperboard carbon stocks applying either the literature-based carbon conversion factors and half-life values or the IPCC's default values in scenario 1.

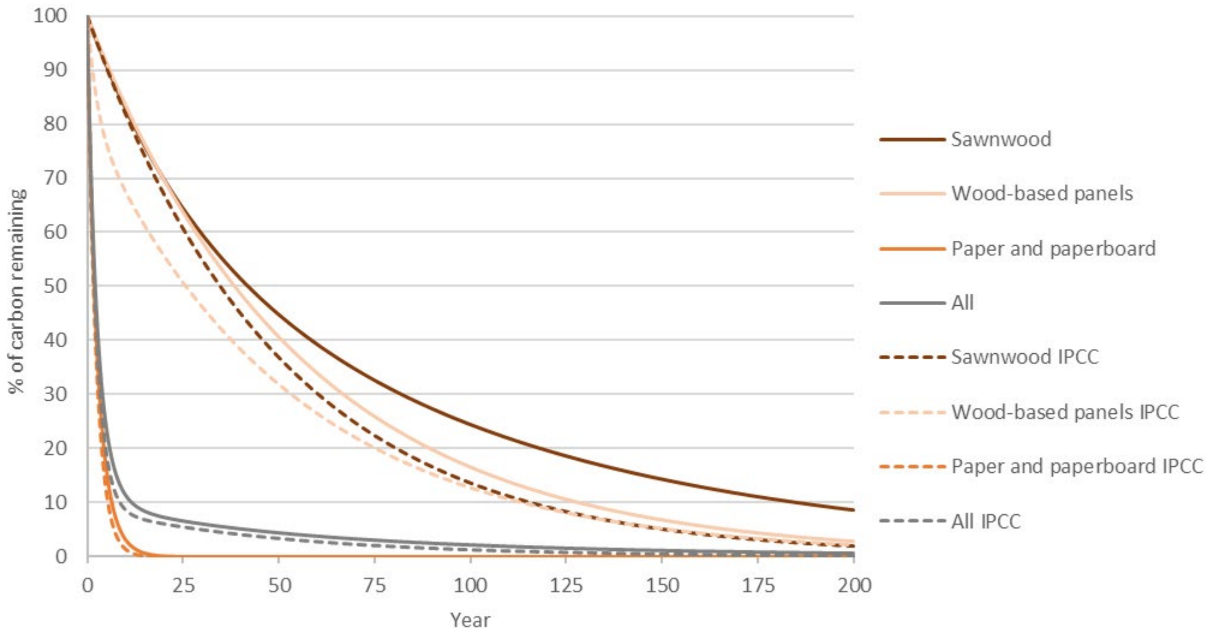

Figure 6. Percentages of carbon remaining after 200 years in the sawn wood, wood-based panels and paper and paperboard carbon stocks applying either the literature-based carbon conversion factors and half-life values or the IPCC's default values in scenario 2.

3.5 Conclusions on the HWP carbon storage accounting

The literature review showed that:

- Lifespan or service life values of HWPs were scarce and old, since the frequently used IPCC approaches apply half-life values as well as most of the other decay distributions. Thus, if they are used e.g. in LCAs in which it is assumed that carbon is stored according to its lifespan and released at the EoL as a single pulse, the appropriate sensitivity analyses concerning the lifespan values is recommended.
- Lifespan and half-life values for the new wood-based products are still absent.
- The exponential decay distribution is widely used and easy to apply, although it is argued not to be the most suitable to express the decay patterns of single HWPs, especially with long lifespans.
- If a short time horizon is considered, it is easier to consider the temporary carbon storage of HWPs permanently over the time horizon being studied and ignore the delayed emissions. Thus, the selection of the time horizon may significantly influence the performance of carbon storage in HWPs. Often, at least a 100-year time-horizon is applied to analyze the fate of carbon in HWPs in use.

The study showed that national level accounting approaches dominate the literature considering carbon storage of HWPs. Both the IPCC's calculation approaches and specific wood product models are frequently applied to assess changes in HWP carbon pools. Minor attention has been placed on developing methodologies and practical solutions for company-level carbon storage accounting. Additionally, especially the IPCC's calculation approaches are designed to estimate the current HWP carbon pool changes at the national level and not the future-oriented estimations at the product or company level.

Considering the product or company-level carbon storage accounting, the applied system boundaries and scope of the calculations should be transparently defined and informed abreast with the methodologies and data used. The following recommendation can be provided:

- Carbon balance accounting may be carried out by following either a stock change approach or carbon flow (to and from the atmosphere) approach, but they should be coherently chosen to cover the forest and HWP carbon pools or to be limited to concern only the fate of carbon of HWPs in use.
- Carbon storage may be calculated for the whole company, the company's production in different countries, or for specified HWPs.
- Calculations may include imports and exports (e.g. intermediate products or materials to and from other countries or other companies) or include only the HWPs within a given country.
- The scope may vary from calculation of the current carbon stock based on the past production amount or future carbon stock based on the assumed production amounts. Furthermore, the scope may vary from calculating the fate of carbon based on a one-year production of specified HWPs (e.g. comparing alternative scenarios for the final products) or calculating the annual carbon stock and subsequent stock changes.

To account HWP carbon at the product or company level, there is a need to:

1) define whether the product or company level will be studied

2) define the time horizon (e.g. annual, or multi-annual, over the whole life cycle) considered

3) define the system boundaries for the HWPs considered and choose the appropriate approach for accounting, e.g. one of those suggested by the IPCC

4) define the appropriate half-life or lifespan values to be used for product carbon (e.g. based on the IPCC default values or other relevant data)

5) define the suitable decay distribution to be used to calculate the fraction of carbon remaining each year (e.g. based on IPCC First Order Decay function), and

6) finally calculate the changes in the carbon stock (or carbon flows) for subsequent years over the time horizon being studied.

4. Fossil emissions avoided through the substitution of alternative products

4.1 Determining a displacement factor

As discussed in Chapter 2.1, the term substitution is used in this study in the sense of an attributional LCA, to refer to functional equivalency between two or more products serving certain materials or energy functions, and not in the sense of a consequential LCA, which uses substitution to solve multi-output processes and may include market-mediated effects such as price elasticities and rebound effects. The selected attributional approach is also typically used in the literature of substitution. The objective here is to define the reference product to be compared to the product at the core of the study (a harvested wood product). The GHG emissions associated with the life cycle of this product are *replaced* by the HWP. In this sense, the term *replacement or displacement* is synonymous with the substitution.

A *displacement factor* (DF, also known as substitution factor or replacement factor) typically describes the efficiency of using wood products and fuels in reducing fossil GHG emissions by quantifying the amount of fossil GHG emission reduction achieved by wood use (Sathre and O'Connor 2010, Myllyviita et al. 2021). DF should describe the actual substitution impact of wood use. This means that a wood product should replace a non-wood product with a similar function. In many cases, it is impossible to assume that a similar amount (in mass or volume) is needed to replace the non-wood alternative. For instance, it is not necessarily realistic to assume that one ton of wood would replace one ton of plastic in packaging industry. In reality, the ratio of wood required to replace non-wood alternatives should be defined case-by-case. Additionally, in the case for example of packaging, it is possible that there are differences in how many times the product can be used. Plastic packaging, for example, can be used several times, whereas alternatives based on the use of cardboard may be for single use only (Koskela et al. 2014).

The major problem related to the definition of a wood-based product and a replaced non-wood product is that there are enormous numbers of products on the market and each product and its substitutes are case-specific. Thus, defining all the non-wood products and their substitutes would require an immense amount of work, and would still be subject to the assumptions made about what is substituted and what the related avoided fossil GHG emissions are. Thus, it is recommended to focus on products and product groups with large production volumes as their substitution impacts are likely to outweigh those of smaller production volumes. Because of numerous substitutes, however, the accuracy in assessing the substitution impacts of wood use include inevitable uncertainties.

The GHG emissions of products compared are often calculated according to the rules of a LCA (ISO 14040) and the GHG emissions avoided caused by wood products used in place of alternative products are obtained from the difference of GHG emissions between the wood and non-wood products. A positive DF implies that the wood products would reduce GHG emissions, whereas a negative value implies the opposite.

According to Sathre and O'Connor (2010), DF can be aggregated as follows:

$$DF_1 = \frac{GHG_{non-wood} - GHG_{wood}}{WU_{wood} - WU_{non-wood}} \qquad (10)$$

$GHG_{non\text{-}wood}$ and GHG_{wood} include aggregated GHG emissions of non-wood and wood products, and WU_{wood} and $WU_{non\text{-}wood}$ describe the amount of wood (in carbon tons) used in the wood and non-wood products. Typically, only biogenic, wood-based carbon is considered in the numerator of Eq. 10. $WU_{non\text{-}wood}$ can also include biogenic carbon in case e.g. of a building with wooden structures (Sathre and O'Connor 2010).

According to Sathre and O'Connor (2010), DFs could be calculated in other units as well, e.g. the emission reduction per ton of wood product per m^3 of wood product. However, the common practice in the scientific literature is to assess the substitution potential per wood (or carbon) content in a product (Myllyiita et al. 2021). Leskinen et al. (2018) note that it is possible that WU includes only the wood contained in the end-use products or all harvested wood used for producing a wood end-product. In the first approach the DF is typically larger than in a case where all the wood entering the process considered is included. If the DF is determined per the carbon content in a final product, it is impossible to detect how much wood has been used to manufacture a final product. In such a case, the DFs may be relatively high for those HWPs which use significant amounts of wood but not much fossil fuels in their life cycle than if determined per the amount of wood used in the process considered (Myllyviita et al. 2021).

In the scientific literature, DF typically includes only the production emissions of alternative products (Myllyviita et al. 2021). Such an assumption is appropriate as long as both a non-wood and wood product have a similar use-phase and the end-of-life treatment of products is considered separately. However, in cases where a non-wood or wood product requires more maintenance, for example, than its counterpart, the use phase typically causes more GHG emissions as well. If only production emissions are included, DF may mistakenly favor a product with higher GHG emissions during its whole lifecycle. For instance, the majority of the GHG emissions associated with the life cycle of textiles are caused by washing at high temperatures not by the production phase (Manda et al. 2015). Leskinen et al. (2018) point out that DF should include four components, i.e. production, use, cascading, and end-of-life to fully describe the emissions reduction during the whole life-cycle of a product. Based on literature review by Myllyviita et al. (2021), this is one topic that remains poorly understood or lacks transparency in the scientific literature.

An illustrative example of aggregating a DF is described in Fig. 7 and determined according to Eq. 10. In this hypothetical example it is assumed that 1 kg wood replaces 0.5 kg of construction steel. The production emissions (including wood harvesting, transportation, and production emission) are 0.1 kg CO_2-eq. The emissions of steel are 0.2 kg CO_2/kg. It is assumed that the two alternative products have a similar use phase concerning maintenance, for example. The end-of-life impacts are excluded. The emissions of wood are subtracted from the emissions of an equivalent amount of construction steel in the numerator. In the denominator, the carbon content of the steel is subtracted from carbon content of wood. In this example, wood use reduces the GHG emissions in the techno system 0.89 t CO_2/t of C.

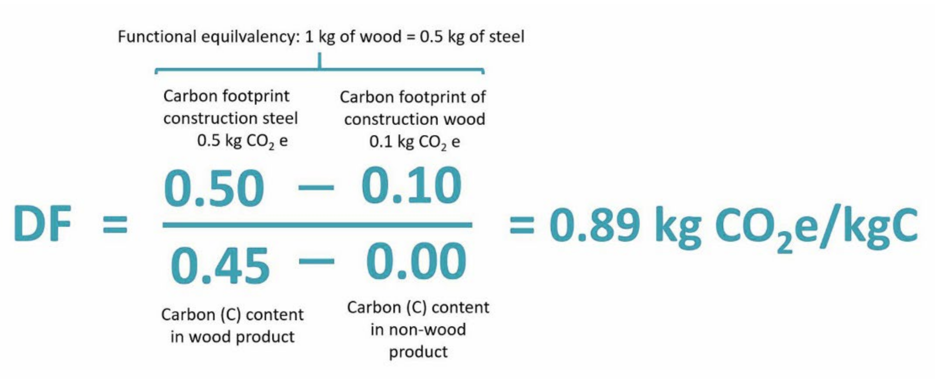

Figure 7. Illustrative example of aggregating a DF.

When applying the DFs available in scientific literature, one must be careful, as including substitution credits to the end-of-life of a product typically generates a substantially larger DF than in a case where energy recovery from a discarded wood product is not considered (e.g. Sathre and O'Connor 2010). When assessing the substitution impact of end-of-life energy recovery, it should be acknowledged that substituted products may have generated end-of-life energy as well. For instance, plastic packages can be incinerated after use, thus, generating an energy recovery function similar (or larger) to wood-based packaging. In both cases the carbon content of the products is released into the atmosphere in energy recovery, thus the substitution credits of the energy recovery depend on the displaced alternative fate of the products (e.g. landfilling or incineration without energy recovery). Generally, energy DFs are likely to change in the future. Thus, the assumptions made concerning the development of GHG intensities of product systems in the future significantly influence the substitution credits available. In several studies (e.g. Brunet-Navarro et al. 2021), it has been addressed that end-of-life substitution credits will probably be substantially lower in the future because of the emission reductions necessary to achieve the Paris Agreement target. However, this is not commonly addressed in the scientific literature focusing on the substitution impacts of wood use.

There are no standards or even well-accepted rules on how to determine a DF for a wood-based product. As such guidance is not available, it is recommended to use the same principles as used in an LCA to assesses the carbon footprint of a product. This is a recommended approach for both wood-based and non-wood products. The carbon contents of the wood-product and non-wood product required in the calculation of the DF according to Eq. 10 are usually readily available. Typically, it is assumed that half of the wood dry mass is carbon, but if more detailed information is available on the carbon content, it is recommended to use such information. As discussed earlier, it is possible to determine the DF for embodied carbon or all wood use entering the process considered (among other options), but this assumption should be clearly stated to avoid misinterpretations.

There are no unique and commonly accepted recommendations regarding how the allocation should be made within LCA. As wood use is typically a complex system with multiple by-products, allocation is a fundamental issue, which is typical in attributional LCA (see also Section 5.2). Separate DFs could be calculated for a main product and for each by-product. The share of GHG emissions caused by the acquisition of raw material should be allocated for each product, and the emissions caused by the production of each product. When applying such separate DFs, the share of wood required for and transferred into each product should be considered appropriately.

In previous reviews and meta-analyses (e.g. Myllyviita et al. 2021, Leskinen et al. 2018) it has been noted that the DFs in the scientific literature are highly variable. Some of the variation is caused by different system boundaries, but major sources of uncertainty remain poorly understood. Holmgren and Kolar (2019) detected that the literature provides various approaches and results for forest products. According to them, key variations between studies result from differences in handling biomass-based carbon, i.e. whether emissions caused in the value chain are included or not, as in some studies also changes in the forest carbon stock and/or HWPs are included in the overall substitution effect. In cases where forest and product carbon stocks are included in the DFs, the DFs are not directly comparable to cases where they are excluded. In cases where carbon stock changes in the forest and HWPs are included in the DFs, it is crucial to avoid double-counting if those DFs are used to assess the overall GHG impact of wood use. However, if the DFs only cover fossil-based GHG emissions, changes in forest and harvested wood product carbon stocks should be considered separately and attached to the analysis to assess the overall GHG balances of forest biomass use. In a review by Myllyviita et al. (2021) it was detected that a vast majority of the DFs in the scientific literature include only fossil-based GHG emissions and forest and HWP carbon stocks were considered separately and applied together with DFs for the overall GHG balances of forest biomass use.

Large variations in the DFs for specific products occur depending on assumptions about which material or energy source it is assumed will be replaced. Most studies include a basket of products and a basket of the materials/energy sources they replace. Substitution effects are presented as ranges across the basket or as weighted averages. Some studies include all substitution effects throughout the lifetime of the product, while some do not. This further reduces the possibilities to compare DFs in different studies as they are not based on the same system boundaries and temporal scope.

4.2 Displacement factors for various product groups in the scientific literature

We identified DFs in the scientific literature and estimated the substitution potential of those product groups without DFs available in scientific literature. Meta-analyses by Leskinen et al. (2018) and a review by Myllyviita et al. (2021) were used as a basis for this review. We estimated a range and average value for DFs identified in the review by Myllyviita et al. (2021). Only DFs with the same system boundaries were included, i.e. studies that excluded forest or product carbon stock or end-of-life energy credit or substitution in a DF. Consequently, the DFs considered here describe the substitution of fossil-based GHG emissions for the primary products. In the determination of DF factors, the following assumptions were applied: the wood density was assumed to be 0.5 t/m^3 and the dry mass was assumed to have a carbon content of 50%. In cases where it was unclear how DFs were determined, they were excluded from an assessment of a range and an average value.

Energy

Wood can be used in various ways to replace fossil energy sources. Ways to use wood as an energy source also vary depending on the country. Based on energy DFs in a review by Myllyviita et al. (2021), a range for energy substitution DFs from close to zero to 2.5 tC/tC with an average value of 0.71 tC/tC was generated. However, for instance Köhl et al. (2020) merely estimated the required DF to achieve carbon neutrality, not the likely substitution effect. In their study, the DFs were 1.9 tC/tC for lignite and 2.5 tC/tC for gas. These two DFs are substantially higher than other DFs. Many studies estimated alternative DFs for various substitution cases. For instance, Ji et al. (2016) estimate 0.96 tC/t C DF for coal, 0.79 tC/tC for oil and 0.56 tC/tC for natural gas. In a study by Cintas et al. (2016) the range for DFs was 0.55–1.27 Mg of fossil C displaced per Mg of the C in the biomass used. Similarly, Soimakallio et al. (2016) applied a range from 0.5 to 1.0 tC/tC. Smyth et al. (2017a) estimated a range for harvest residues with an energy use of 0–2 tC/tC and Smyth et al. (2014) did the same for domestic bioenergy from -0.08 to 0.79 tC/tC. The negative value implies that wood use increases the net emissions in the techno system. Smyth et al. (2014) conclude that several DFs occur because the bioenergy displaced different energy sources in different regions of Canada.

Based on scientific studies, Leskinen et al (2018) determined the climate benefits of using biomass residues from timber harvests, finding that using harvest residues for bioenergy increases DFs by about 0.4 - 0.8 kg C. Stump harvesting can provide an additional substitution benefit of 0.2 - 0.5 kgC/kgC. Some substitution benefits can often be obtained from energy recovery at the end-of life stage: the average of all reported substitution factors for this life cycle stage was 0.4 kg C/kg C in wood products (Leskinen et al., 2018).

Although pulp mills can produce a substantial amount of energy, it has not been given substitution credit as generated energy is used to manufacture pulp products (Seppälä et al., 2019). In a case of excess energy, it is possible to give some substation credit for pulp mill energy.

Table 3. Energy DFs identified in the scientific literature (Myllyviita et al. 2021).

Authors	Country	Description	DF	Unit
Fortin et al. (2012)	France	Domestic energy wood and industrial wood pellets replace electricity and oil	0.076	Mg/m^3 of C-eq.
Fortin et al. (2012)	France	Wood pellets	0.126	Mg/m^3 of C-eq.

Authors	Country	Description	DF	Unit
Böttcher et al. (2012)	Germany	Heating oil by biomass	0.8	Fossil fuel-C /ton biofuel-C harvested
Smyth et al. (2014)	Canada	Domestic bioenergy	−0.08–0.79	Mg C/Mg C
Smyth et al. (2014)	Canada	International bioenergy	0.6	Mg C/Mg C
Knauf et al. (2015, 2016), Knauf (2016)	Germany	Fuel substitution	0.67	t C/tC
Soimakallio et al. (2016)	Finland	Paper products (fossil fuel substitution)	0.8	t C/tC
Soimakallio et al. (2016)	Finland	Paperboard products (plastics, fossil fuel substitution)	1.4	t C/tC
Soimakallio et al. (2016)	Finland	Energy and post-used mechanical wood products (fossil fuel substitution)	0.47–0.89	t C/tC
Han et al. (2016)	South Korea	Sawn wood and industrial roundwood substituting fossil fuels for heating	0.076	Mg/m^3 C-eq.
Han et al. (2016)	South Korea	Wood pellets and industrial roundwood substitute fossil fuels for heating	0.126	Mg/m^3 C-eq.
Matsumoto et al. (2016)	Japan	Logging residues, process residues and waste wood; substitution of residues and waste wood for heavy oil kg	108.9	kg C/m^3
Cintas et al. (2016)	Sweden	Forest-based bioenergy	0.55–1.27	Mg of fossil C is displaced/Mg of C in biomass used
Smyth et al. (2017a)	Canada	Bioenergy from harvest residues	0–2	t C/t C
Härtl et al. (2017)	Germany	Timber used in energy production	0.67	t C_{fossil}/t C_{timber}
Smyth et al. (2017b)	Canada	Bioenergy using an optimized selection of bioenergy facilities, which maximized the emissions avoided from fossil fuels.	0.47–0.89	t C/t C
Ji et al. (2016)	China	Substitute for coal	0.96	t C/tC
Ji et al. (2016)	China	Substitute for oil	0.79	t C/tC
Ji et al. (2016)	China	Substitute for natural Gas	0.56	t C/tC
Baul et al. (2017)	Finland	Energy biomass	0.5	t C/tC
Suter et al. (2017)	Switzerland	Heat replacing light fuel oil	0.55	t CO_2-eq./m^3
Suter et al. (2017)	Switzerland	Heat replacing natural gas	0.32	t CO_2-eq./m^3
Suter et al. (2017)	Switzerland	Electricity mix CH	0.12	t CO_2-eq./m^3

Authors	Country	Description	DF	Unit
Schweinle et al. (2018)	Germany	Displacement of fossil fuel with wood fuel	0.67	t C/t C
Chen et al. (2018)	Canada	Wood used to produce energy for the HWP industry reduced fossil fuel-based emissions	2	t CO_2-eq./t C in wood
Smyth et al. (2018)	Canada	Harvest residues for bioenergy, energy demand and displacement factors two forest management units	0.38, 0.95	t C/tC
Köhl et al. (2020)	Germany	Lignite substitution to achieve carbon neutrality	1.9	t C/tC
Köhl et al. (2020)	Germany	Gas substitution to achieve carbon neutrality	2.5	t C/tC
Hurmekoski et al. (2020)	Finland	Wood use replacing CHP of fossil origin	0.7	t C/tC
Hurmekoski et al. (2020)	Finland	Wood-based transport fuel replacing diesel	0.63	t C/tC
Hurmekoski et al. (2020)	Finland	Wood-based ethanol replacing transport fuel	0.7	t C/tC

Construction

The substitution effects of wood construction are much discussed in the scientific literature. The variation in the substitution effects of construction are substantial. In a meta-analysis with 51 studies conducted by Leskinen et al. (2018) the DF for structural construction (e.g. buildings, internal or external walls, wood frames, beams) was 1.3 kg C / kg wood product with 95% of the values ranging between -0.9 and 5.5 and for non-structural construction 1.6 kg C/ kg wood product with 95% of th values ranging between 0.2 and 4.7 kg C/kg wood product. The DFs in a review by Myllyviita et al. (2021) were used to generate an average value of 1.2 tC/tC (minimum -0.43 and maximum 8.55 tC/tC) for all construction DFs. The Average DF for buildings (with 7 house types considered) was 1.28 tC/tC (minimum 0.60 tC/tC and maximum 2.40 tC/tC). This is a clearly smaller value than the average DF in the highly cited meta-analysis by Sathre and O'Connor (2010). Their study generated DFs for alternative building types from -2.3 tC/tC to 15 tC/tC, with an average value of 2.1 tC/tC. This average value of 2.1 tC/tC is often cited in the scientific literature, although it is based on only a few case studies and should not be considered an average DF for construction. Buchanan and Levine (1999) detected a range from 1.1-15 tC/tC of reduced carbon dioxide emissions due to an increase in the stored carbon. Kayo et al. (2015) estimated that the substitution of wooden buildings for non-wooden buildings would generate an emissions reduction of 60.56 kg C/m^2. Chen et al. (2018) estimated that 3.64 tCO_2-eq. of emissions could be reduced per ton of C for non-residential construction and 9.56 t CO_2-eq. for residential construction. The actual substitution effects are heavily influenced by the house types considered. As no comprehensive assessments of the substitution effects of wood construction exist, it is impossible to state an average DF for a construction. Nevertheless, wood construction is given moderately high DF estimates, especially in the most optimistic calculations.

It is possible to determine a DF for a whole building but also for construction raw materials. In construction standards, it is recommended that in comparative studies the functional unit should be a building, not e.g. construction materials. In the literature, however, DFs for both whole buildings and houses as well as construction materials and components (e.g. plywood, wood panels etc.) are available. As only a few studies include DFs for construction materials, it is impossible to determine a range of DFs. Based on review by

Myllyviita et al. (2021), it can be concluded that construction material DFs are highly variable. For instance, Knauf et al. (2015) estimated a DF of 1.62 tC/tC for wooden window frames vs. PVC or aluminum frames, and Rüter et al. (2016) estimated a DF of 5.53 kg CO_2-eq./ kg for windows. Suter et al. (2017) determined a close to zero DF for insulation materials, whereas Rüter et al. (2016) estimated a DF of -0.40 kg CO_2-eq./ kg of HWP.

Substitution effects can also be assessed per material use. According to Petersen and Solberg (2005), substitution between wood and steel is in the range of 36–530 kg CO_2-eq. per m^3 input of timber (with a 4% discount rate, depending on the waste management assumptions and how carbon fixation on forest land is included) and in the range of 93–1062 kg CO_2-eq. for substitution between wood and concrete, if the wood is not landfilled after use. However, as discussed earlier, it is essential to ensure functional equivalency of wood and non-wood products. When assessing substitution effects based on intermediate products, it is possible that the assumptions behind the calculation are too coarse and lead to misinterpretations of the substitution effect.

Table 4. Construction DFs (Myllyviita et al. 2021).

Authors	Country	Description	DF	Unit
Buchanan and Levine (1999)	New Zealand	Concrete to wood, hostel	1.05	Reduced carbon emissions due to the increase in stored carbon
Buchanan and Levine (1999)	New Zealand	Concrete to wood, office	1.1	Reduced carbon emissions due to the increase in stored carbon
Buchanan and Levine (1999)	New Zealand	Steel to wood, industry	1.6	Reduced carbon emissions due to the increase in stored carbon
Buchanan and Levine (1999)	New Zealand	Concrete, steel to wood, houses	2.1–15	Reduced carbon emissions due to the increase in stored carbon
Fortin et al. (2012)	France	Truss and flooring	0.169	Mg/m^3 of C-eq.
Fortin et al. (2012)	France	Exterior cladding	0.024	Mg/m^3 of C-eq.
Fortin et al. (2012)	France	Interior coverings	0.024	Mg/m^3 of C-eq.
Fortin et al., 2012)	France	Other end-use products	0.024	Mg/m^3 of C-eq.
Böttcher et al. (2012)	Germany	Building construction *(Picea)*	0.16–0.24	t fossil fuel-C substituted/t of wood-C harvested
Chen et al. (2014)	Canada	Wood replacing houses with fossil raw materials (steel, concrete)	2.4	t C/t C
Knauf et al. (2015)	Germany	Roundwood (poles, fences, buildings, also treated) vs. steel, concrete, aluminum	2.4	t C/t C
Knauf et al. (2015)	Germany	Softwood lumber, sawn, wet, for packaging concrete shuttering vs. plastics (foils, 3-D elements)	1.8	t C/t C

Authors	Country	Description	DF	Unit
Knauf et al. (2015)	Germany	Softwood lumber, planned and dried for building Purposes	1.4	t C/t C
Knauf et al. (2015)	Germany	Softwood-based glued timber products (glue-lam, CLT) vs.	1.3	t C/t C
Knauf et al. (2015)	Germany	Plywood, also overlaid vs. aluminum profiles, glass-fiber plastic	1.62	t C/t C
Knauf et al. (2015)	Germany	Wood-based panels like particleboard, MDF, OSB (for walls, ceilings, roofs) vs. gypsum board, plaster, concrete, brick type walls	1.1	t C/t C
Knauf et al. (2015)	Germany	DIY products like lumber, panels, profile boards vs. mineral	1.35	t C/t C
Knauf et al. (2015)	Germany	Wooden flooring (one layer, multi layers), laminate flooring vs. ceramic tiles, plastic flooring, wall to wall carpet	1.35	t C/t C
Knauf et al. (2015)	Germany	Doors (interior, exterior) – only framing/construction vs. steel, aluminum, PVC	1.62	t C/t C
Knauf et al. (2015)	Germany	Wooden window frames vs. PVC, aluminum	1.62	t C/t C
Knauf et al. (2015)	Germany	Wooden furniture (solid wood) vs. glass, plastic, metal	1.62	t C/t C
Knauf et al. (2015)	Germany	Wooden furniture (panel based) vs. glass, plastics, metal	1.42	t C/t C
Knauf et al. (2015)	Germany	Wooden kitchen furniture vs. glass, plastics, metal	1.62	t C/t C
Knauf et al. (2015)	Germany	Wooden transportation products vs. plastic, metal	1.62	t C/t C
Kayo et al. (2015)	Japan	Building construction: substitution of wooden buildings for non-wooden buildings	60.56	kg C/m^2
Kayo et al. (2015)	Japan	Civil engineering: substitution of wooden piles for cement and sand piles	46.77	kg C/m^3
Kayo et al. (2015)	Japan	Civil engineering: substitution of wooden guardrails for metal guardrails	64.48	kg C/m^3
Kayo et al. (2015)	Japan	Furniture: substitution of wooden furniture for metal furniture	43.17	kg C/m^3
Nepal et al. (2016)	United States	Extra wood products used in nonresidential construction buildings	2.03	t CO_2-eq./t CO_2-eq.
Rüter et al. (2016)	Europe	Core and shell 2010	1.58	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Core and shell 2030	1.25	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Insulation 2010	−0.40	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Insulation 2030	−0.32	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Windows 2010	5.53	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Windows 2030	4.42	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Claddings 2010	0.9	kg CO_2-eq./kg HWP

Authors	Country	Description	DF	Unit
Rüter et al. (2016)	Europe	Claddings 2030	0.72	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Laminates 2010	1.52	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Laminates 2030	1.22	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Parquets 2010	−0.0164	kg CO_2-eq./kg HWP
Rüter et al. (2016)	Europe	Parquets 2030	−0.0131	kg CO_2-eq./kg HWP
Matsumoto et al. (2016)	Japan	Sawn wood and plywood; substitution of wooden buildings for non-wooden buildings	301.3	kg C/m^3
Matsumoto et al. (2016)	Japan	Roundwood and sawn wood; substitution of wooden piles for cement and sand piles	46.8	kg C/m^3
Matsumoto et al. (2016)	Japan	Roundwood and sawn wood; substitution of wooden guardrails for metal guardrails	64.5	kg C/m^3
Matsumoto et al. (2016)	Japan	Sawn wood and plywood; substitution of wooden furniture for metal furniture	43.2	kg C/m^3
Geng et al. (2017)	China	Ceramic tile replaced with wood flooring	0.17–0.78	tC/m^3
Härtl et al. (2017)	Germany	Timber as sawn logs used in construction	1.66	t C_{fossil}/t C_{timber}
Xu et al. (2018)	Canada	Sawn wood for single-family home, multi-family home, and multi-use building	2.1	t C/t C
Xu et al. (2018)	Canada	Panels for single-family home, multi-family home, and multi-use building	2.2	t C/t C
Chen et al. (2018)	Canada	Residential construction	9.56	t CO_2-eq. emissions reduced per ton of C
Chen et al. (2018)	Canada	Non-residential construction	3.64	t CO_2-eq. emissions reduced per ton of C
Geng et al. (2019)	China	Furniture sector	1.46	t C/t C
Hurmekoski et al. (2020)	Finland	Sawn wood in construction	1.1	t C/t C
Hurmekoski et al. (2020)	Finland	Plywood in construction	1.1	t C/t C

Furniture

The furniture industry is an assembly industry, employing a mixture of raw materials, including wood, metals, plastics, textiles, leather, glass and many others (FAO 2016). Given that the amount of material substituted by wood products varies with different end-use categories, it is difficult to generalize the potential substitution benefits of wood-based materials for furniture uses.

For furniture, several DFs were determined in scientific articles. Geng et al. (2019) estimated a DF of 1.46 tC/tC for the Chinese furniture sector and Kayo et al. (2015) estimated 43.17 kg C / m^3 for Japanese furniture. Knauf et al. (2015) estimated DFs for various German furniture: 1.42 tC/tC for wooden furniture (panel based) replacing glass and plastics, and 1.62 tC/tC for wooden kitchen furniture replacing glass, plastics and metal. Fortin et al. (2012) estimated DFs for French furniture with a DF of 0.043 Mg m^{-3} of

C-eq., 0.069 Mg m^{-3} of C-eq. for kitchen furniture, 0.043 Mg m^{-3} for home furniture, 0.043 Mg m^{-3} of C-eq. for chairs and 0.043 Mg m^{-3} of C-eq. for beds.

Textiles

Wood-based cellulose fibers for textiles have been on the market for a long time, but their market share has remained modest (Kallio, 2021). The environmental concerns related to cotton and the oil-based textile materials, among other reasons, are now favoring wood-based textiles and their share is estimated to increase in the future (Kallio, 2021). For textiles, the average DF in a meta-analysis by Leskinen et al. (2018) was 2.8 tC/tC. Compared to DFs in other product group DFs, this is one of the largest substitution benefits estimated in the scientific literature. This is because the production of wood-based textiles typically emit fewer GHG emissions than cotton or synthetic fibers (Rüter et al., 2016). However, it is impossible to state what the actual substituted products are (e.g. natural materials such as cotton or wool or synthetic fibers). Shen et al. (2010) estimated the environmental impacts of various wood-based textiles. In their study all artificial cellulose fibers have lower GWP than PET fibers, all artificial cellulose fibers except for Lenzing Viscose Asia has lower GWP than PET, PP, PLA and cotton, Lenzing Modal and Tencel Austria 2012 have nearly zero carbon emissions; and Lenzing Viscose Austria had a negative GWP, which means that it sequestrates more carbon in the product than it emits. The allocation method and assumed pulp mix influences the GWP of man-made cellulose fibers. As the study of Shen et al. (2010) also includes the carbon intake of artificial cellulose fibers, the assumed substitution effects are higher compared to the assumption where the carbon intake is not a part of the DF aggregation.

Published data on advanced wood-based textile processing techniques are not currently available. Thus, the GHG emissions of wood-based textile production remains unclear. Regardless of this, textiles appear to be one promising option to increase the substitution effects of wood use.

Chemicals

It has been estimated that wood-based chemicals will increase their market share in the future (Hurmekoski et al., 2018). In the scientific literature, only one DF was identified for polyol by Rüter et al. (2016) (0.77 kg CO_2-eq./ kg HWP). Although DFs for chemicals are scarce, the substitution impacts of wood-based chemicals have been discussed in the scientific literature. For instance, Cashman et al. (2016) investigated the carbon and energy LCA of pine chemicals derived from crude tall oil. Using their assumptions for the GHG emissions of tall oil chemicals and their substitutes and Eq. 10, the DF equals 1.7 tC/tC. This is based on assumptions that the production GHG emissions in Europe are 0.74 kg CO_2-eq./kg of crude tall oil distillative product and the emissions of non-wood alternatives are 2.08 kg CO_2-eq./kg. In their study, however, the emission data of non-wood alternatives were not updated, thus, it is possible that the emissions of non-wood alternatives in the current situation are significantly lower. This would reduce the substitution effect of pine chemicals as well.

Packaging and paper

The size of the food packaging market is expected to rise from 303.3 to 456.6 USD billion over the period 2019–2027 (GVR 2020). Currently, paper and cardboard contribute a major part of packaging materials (40.9%) followed by plastics (19%) and glass (18.7%) (Eurostat 2020). For packaging products, however, only some DFs are identified in the scientific literature. Hurmekoski et al. (2020) estimated a DF of 1.40 tC/tC for packaging replacing PE and 1.7 tC/tC PET and kraft pulp-based packaging (carton boards, sack paper). Härtl et al. (2017) estimated that paper, cardboard, and chipboard packaging replacing plastic would generate a substitution effect of 1.30 t C fossil/t C timber. Knauf et al. (2015) estimated that the DF for wood-based packaging is 1.35 tC/tC. Thus, the range found from scientific literature for DFs is 1.3-1.7 tC/tC for packaging.

Although packaging DFs were not abundantly available in the literature, GHG emissions of wood-based packaging have been much discussed in the scientific literature. For instance, Abejón et al. (2020) estimated that reusable plastic crates should be used instead of single-single use cardboard boxes. In their study, the scenario based on use of cardboard boxes generated GHG emissions several times larger than in the scenario where plastic crates were used. Thus, it is possible that in cases where a plastic crate is used several times, it is the more preferred alternative from the climatic perspective.

New, innovative wood-based packaging could replace currently used packaging materials such as plastics, glass, and metals. Biodegradable bioplastics have gained increased popularity in the plastics manufacturing industry as a way to improve certain sustainability aspects and reduce plastic pollution (Gerassimidou et al., 2021).

It appears that the DF literature is moderately optimistic on the substitution effects of wood-based packaging as the DFs available in the literature were moderately high. However, only a few DFs are currently available, and based on them it is impossible to estimate the overall substitution effects of wood-based packaging. For instance, Chen et al. (2016) estimated that woody-biomass based PET bottles have 21% less global warming potential and require 22% less fossil fuel than their fossil-based counterparts. Based on the GHG data, the DF for woody biomass-based PET bottles is about 0.67 tC/tC, which is substantially lower than the DFs in the scientific literature.

Pallets can be made of wood, plastics, composites, and metals. According to the market estimates, global demand for pallets surpassed 5 billion pallets in 2017 (Freedonia World Pallets 2014), emphasizing the importance of GHG balance in pallet production and use. Three pallet management strategies dominate the industry: single use, buy/sell, and pooled (Deviatkin et al., 2019). Single use is the simplest strategy in which pallets are discarded after a single use, but standardized pallets are usually designed to last several uses (Deviatkin et al., 2019). Only one DF for pallets was found in a study by Rüter et al. (2016) (0.35 kg CO_2-eq./ kg HWP). In a meta-analysis by Deviatkin et al. (2019) plastic pallets were found to exert a higher impact on climate change compared to wooden pallets. The effect was calculated to range from 22 to 166 kg CO_2-eq. per pallet if virgin plastic was used and from 3.7 to 4.1 kg CO_2-eq. per pallet if waste plastic was used. The use of waste plastic reduced the effect due to the zero-burden approach. Datasets on the climate change effects or substitution potential of composite pallets were not available.

For paper and print products it is typically assumed that no substitution impact exists. Electronic media have substituted the use of paper and print products rather than vice versa, but the opposite might take place in certain conditions. Leskinen et al. (2018) identified one scientific article where it was revealed that the DF can be positive or negative, depending on the number of readers of the tablet version, among other factors (Achachlouei et al. 2015).

Other products

The market share of wood-plastic composites is small but expected to grow sharply in Europe (Sommerhuber et al. 2017). Wood-based composites are one promising option to use wood to replace fossil materials. However, only one DF was identified in the scientific literature by Hurmekoski et al. (2020), which was Finland plastic components for cars (replacing virgin polypropylene), equaling 7.38 tC/tC. This is one of the largest DFs identified in the literature. GHG emissions from the production of composites are typically low, as they are usually based on waste streams, for example from the construction sector. Wood-based composites could be used in several sectors such as packaging, construction (Sommerhuber et al. 2017).

Emerging wood products

Wood can be used as a raw material for new, innovative bioproducts, alongside more conventional forest industry products. Examples of new wood-based products include nanocellulose, formable plywood, wood plastics and bio-composites. Additionally, new wood-based textiles are currently being developed. In the scientific literature, DFs for new emerging wood products are not yet available. It is possible that new wood-based products could have higher DFs than the product DFs described in this report if they replace more emission-intensive materials. The problem with new wood-based product development is that they are still under development and production processes are not yet well-established. This can make production processes more emission intensive than those of non-wood products currently available in the market. Thus, it is difficult to assess the substitution effects of new innovative wood products based on the currently available information.

Moon et al. (2013) estimated that the GHG emissions arising from preparing cellulose nanofibers ranged from 1.2 to 3.7 kg CO_2-eq. kg^{-1}. These results support the idea that nanofibers could contribute to energy saving and GHG emissions reduction. DFs for alternative products, however, cannot be determined based on this as functional equivalency should be ensured. For wood-based textiles some DFs were available in the literature, but little information is available on GHG emissions of emerging wood-based textiles. As textile DFs appear to be one of the most promising wood-based product types to support GHG emissions reduction, new wood textiles are an interesting product group. As there are several ongoing research projects connected to the development of wood-based textiles, it is likely that in the near future there will be more information available on the substitution potential of new wood-based textiles. Furthermore, there are several wood-based products (e.g. pharmaceutics) currently under an active development. Because of their lower production volumes, their substitution impact from the viewpoint of GHG emission reduction is not as relevant as products and product groups with large production volumes.

Wood has several possible uses in the construction sector. Replacing concrete with wood is much discussed in the scientific literature. Not only the possibilities to replace concrete with wood are being considered, but also using wood as an additive to cement has been suggested. Improving the strength or other features of concrete with wood-based additives could lead to a situation where wood use enables the use of a smaller amount of concrete, i.e. this would mean a reduction in GHG emissions. Mejdoub et al. (2017) used nanofibrillated cellulose from eucalyptus pulp as a cement replacement. With an addition of 0.3% of nanofibrillated cellulose, the compressive strength of cement was improved by more than 50%. Peters et al. (2010) tested a combination of nanocellulose and micro cellulose fibers to increase the toughness of reactive powder concrete. Based on their preliminary results, the addition of 3% micro- and nanofibers in combination increased the fracture energy by more than 50% relative to the unreinforced material and required only minor changes to the processing procedure. Such substitution potential of wood, however, has not been considered in the scientific literature focusing on DF. In the most optimistic scenarios, such substitution could generate substantially larger DFs than the ones described in this report.

Lignin is currently used mainly to produce bioenergy but could be used as a raw material for chemicals as well. Lignin is an abundant raw material and it is produced as a side stream of the pulp and paper industry. Lignin-based products were not given much attention in the DF literature. However, a review by Moretti et al. (2021), based on several LCA studies, suggests that often lignin-based products offer better environmental performance than fossil-based products, especially regarding climate change. Still, typical methodological problems related to LCA (e.g. allocation and modelling biogenic carbon flows) as well as technical aspects such as selecting counterparts to lignin products make statements concerning their environmental superiority premature. Lignin has several potential applications in the construction sector as well. Lignin is likely to start replacing typically petroleum-based rigid polyurethane foams in the near future (Jędrzejczak et al. 2021). Lignin can also be used to increase the performance of concrete (see e.g. Bajwa et al. 2019). There are several other new areas for the use of lignin in the future, yet their substitution effects remain unclear.

4.3 Conclusions on the emissions avoided

The fossil emissions avoided through substitution are generated as the difference in fossil emissions between wood use and its alternative system. This may depend strongly on the wood product in question, its alternative to be replaced and many methodological assumptions required in the assessment to define system boundaries and to handle the end-of-life treatment of products and the timing of emissions, among other aspects. The avoided fossil emissions can be described using so called displacement factors (DFs), which indicate the avoided emissions per additional use of wood. DFs found in the literature vary for different products depending on many methodological and case-specific assumptions but have found to be on average 0.55 (ranging from 0.27–1.16) tC/tC at the market level (Hurmekoski et al. 2021). When applying DFs derived from the literature, it is important to understand the underlying assumptions appropriately.

5. Cascading use of wood or bio-based products

5.1 General aspects of biomass recycling and cascading

While biomass-based resources are renewable in their nature compared to mineral or fossil resources, they are not entirely unlimited, mainly due to the limitations of the availability of cultivation land as a finite resource. The tank vs. plate debate regarding biofuels showed that there is a need for a general strategy to maximize the environmental and societal benefits provided by biomass. To improve resource efficiency and availability, the EU adopted the Waste Framework Directive (2008/98/EC) with its latest amendment (EU 2018/851). The directive defines a clear hierarchy as to how to treat material flows related to waste management: while the most important factor is the prevention of any waste in the first place, for waste streams that still exist, the so-called “waste hierarchy” dictates the priority of (preparation for) re-use over recycling over recovery (e.g., energy recovery). Waste disposal is the least preferred option and should only be considered, if no other of the above-mentioned utilization pathways apply to non-hazardous waste. The latter are regulated differently and are thus excluded in the following (EU 2018/851).

One possible solution to mitigate limited land availability and to increase the resource efficiency is the cascading use of biomass-based products as a direct implementation of the waste hierarchy of the Waste Framework Directive. The cascading principle describes sequential utilization of a—in this case biogenic—resource or product within different life cycles (e.g., the recycling of waste paper to paper fibers as inputs for a new paper product) or applications (e.g., waste biomass utilization as the main input for the production of poly lactic acid-based bioplastics) (Fehrenbach et al., 2017). In this way, the share of biomass within our economy increases while other non-sustainable resources are replaced (EU Commission 2021).

To this date, a consistent definition of the term *cascading* does not exist, neither in science nor politics. However, in most cases, biomass cascading follows a certain hierarchy, which comprises (preceding) material utilization of the biomass and subsequent energy recovery/utilization, with a clear desired emphasis on the former. In the currently debated EU Commission’s proposal for an amendment to the Renewable Energy Directive (Directive (EU) 2018/2001), the utilization of biomass resources is addressed. Moreover, a clear-cut prioritization of the use of biomass following the cascading principle is outlined, which comprises/distinguishes the following steps (European Commission 2021):

1. wood-based products
2. extension of their service life
3. re-use
4. recycling
5. bio-energy
6. disposal.

Here, too, the material use of biomass is clearly highlighted compared to the energy use, which is purposely placed at the lower end of the hierarchy. This is further underlined, as energy recovery is only then applicable for woody biomass, where any other form of utilization is either not “economically viable or environmentally appropriate” (EU Commission 2021). Additionally, the proposal also stresses the function of forests as carbon sinks, whereas in contrast to the current practice, the generation of energy from some specific forms of woody biomass[3] should no longer find support. Furthermore, the promotion of the energy utilization of quality roundwood should be avoided except in “well-defined circumstances” (EU Commission

[3] The proposal mentions saw logs, veener logs, stumps and roots.

2021). For other forest biomass sources, the proposed amendment to the RED prohibits the utilization in electricity-only-installations except a subsequent CCS scheme to reduce emissions[4] (EU Commission 2021).

The literature cites the energy recovery of biomass as the de facto end of the cascade. This, however, is increasingly challenged by the emergence and implementation of carbon capture and usage (CCU) technologies as end-of-pipe emission reduction and, additionally, the provision of a feedstock for the chemistry sector or Power-to-X (PtX) fuels, for example (Olsson et al., 2021). Fehrenbach et al. (2017) expand the definition of biomass cascades by adding a differentiation between single-step vs. multi-step cascading. Single-step cascading describes a system with direct energy recovery of the product after serving in a single life cycle. In contrast, the multi-step cascade comprises multiple material life cycles of the biomass resource before energy recovery as shown in the illustration below (Carus et al. 2014, Olsson et al. 2016, Fehrenbach et al. 2017).

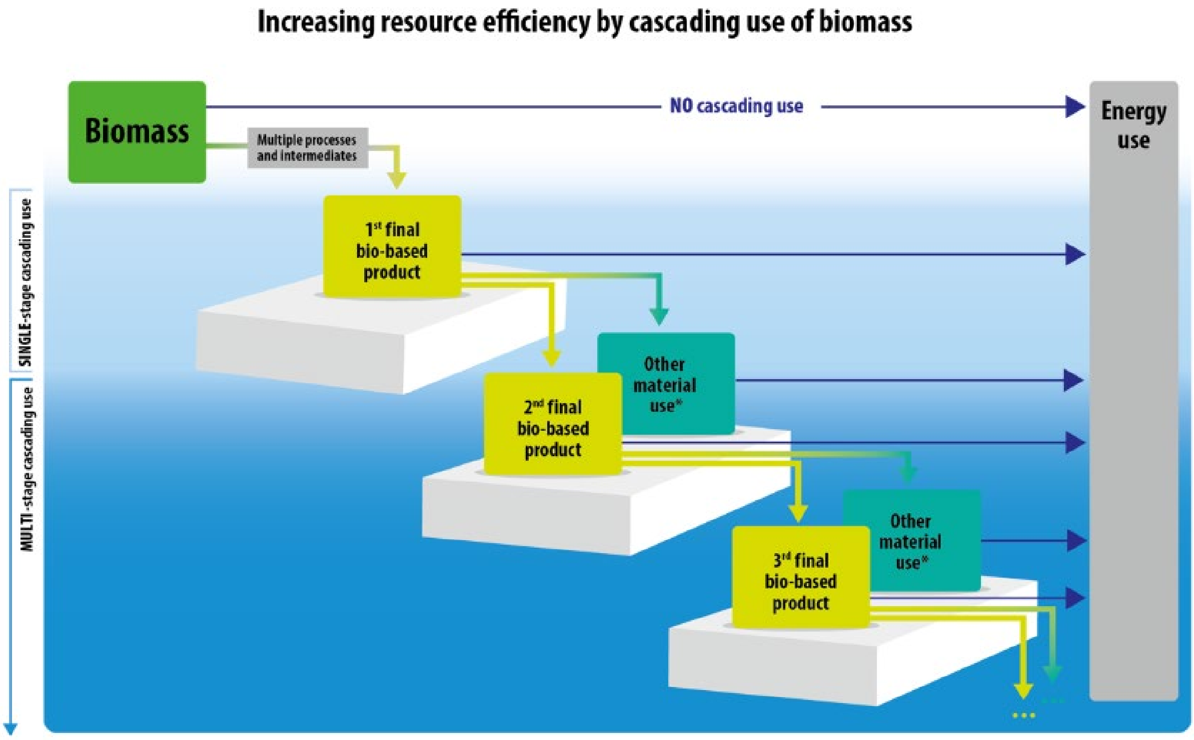

Figure 8. Illustration of cascading use of biomass by single or multiple steps (Fehrenbach et al. 2017).

In either case, the goal of cascading is to increase resource efficiency and value creation of biogenic raw materials used compared to a reference scenario with a single purpose only. Hence, in theory, less biomass cultivation, and thus, less accompanying land use/land use change will occur with the associated benefits of avoided burdens of land use change, fertilizer use, etc. (Fehrenbach et al., 2017). In LCA, the primary/reference process avoided which is substituted/displaced through the utilization of a cascade with corresponding avoided burdens is credited to the cascading system. On the other hand, through the utilization of the cascade, emissions arise and resources are needed. The difference in the burdens associated with the cascade and the burdens avoided through the substituted reference process constitutes the net benefit (or burden) of cascade utilization.

[4] Another exception is the production in regions subject to a territorial just transition plan.

Generally, Fehrenbach et al. (2017) distinguish between direct effects, e.g. fewer GHG emissions, and indirect effects, like less intensive cultivation. To—in theory—maximize the benefits of cascading, multi-step cascading would thus be preferable, compared to a single-step cascade. This, however, is not feasible in each case or product, as cascading/recycling in principle is determined by the physical degradation of materials involved, which could lead to product quality issues (Fehrenbach et al., 2017). Fiber products in the paper sector, for instance, have a distinct lifespan of 3–4 cycles before they are no longer applicable to the production of paper. In contrast, metallic resources such as aluminum can—in theory—be recycled or cascaded ad infinitum, since no or negligible material degradation occurs (European Aluminium 2020).

Additionally, the cascading of biogenic materials can also serve as an artificial additional carbon storage, as the carbon is stored within the material and thus, the techno-sphere in contrast to an immediate return to the ecosphere in the form of CO_2[5] after the product has reached its EoL status. The carbon retention times of this form of temporary carbon storage depend on the specific products and their characteristics, both in terms of their suitability for cascading in general and the product lifespan. Bio-based materials and products vary widely in terms of their (physical and chemical) properties, as well as their respective lifespan for that matter, following their individual use and purpose. This, in turn, also determines their potential for cascading and recycling, in general. A solid wood product without any additional treatment for instance has a higher chance of successful material recycling compared to a similar product which has been highly contaminated with wood preservatives or other additives. Moreover, some contaminants determine the subsequent applicable treatment options. For instance, the often-cited coffee-to-go cup may not be disposed of via conventional paper collection, but only via mixed municipal waste bins (UBA 2021). However, the collection of many small applications such as labels or composite materials is difficult. They are often lost to any form of material recovery since the separation from other waste streams is challenging or not economically viable.[6] Especially in post-consumer waste streams, these so-called dissipative losses of small-scale applications are hard to mitigate due to in large part to product design.

Generally, one has also to distinguish between waste arising during production processes and post-consumer waste. The former is usually easier to re-use or recycle since it is less mixed with other disruptive materials or contaminants and the purity of a waste stream often determines the success of a recycling process. Moreover, the policy framework for the waste management system in place is of utmost significance. Fehrenbach et al. (2017) compared different recycling schemes determined by the policy framework, e.g. highly regulated waste paper collection vs. handling of PLA (polylactic acid). While the former is well established and widely successful, the latter is usually lost to thermal recovery since the market volume is not yet adequate and would not thus justify separate collection. The economics of a potential second life of a product or material need to be taken into consideration, as well. It is easy to see that—in the absence of a policy framework specifically addressing cascading—in most cases, a cheaper primary resource will be the preferred option, compared to more expensive re-used or recycled resources.

To better understand the afore-mentioned dynamics, a general differentiation of bio-based materials and products helps map out the potential for cascading:

1. fiber-based materials, such as paper, cardboard, cartons and the like;
2. solid wood products, e.g., sawn timber, plywood, furniture;
3. biochemicals, such as bio-naphtha, bio-fuels, and lubricants, textiles, plastics, etc.

The following chapters describe the potentials for cascading in the respective product groups outlined above.

[5] Either directly after incineration/energy recovery or through decomposing processes in the short to mid-term
[6] https://eu-recycling.com/Archive/7887

5.2 Principles for modelling

The attributional vs. consequential LCA issue in the context of recycling/cascading

In Chapter 2.1, the two principal approaches of attributional (ALCA) and consequential LCA (CLCA) are described. Furthermore, Chapter 4.2 distinguishes between the understanding of substitution in the sense of this report and the understanding of substitution as a method used in CLCA. These methodological issues are also relevant when considering the second life of a product or recycling, Within the context of LCA, recycling and the second life of products constitutes a special case of multifunctionality. The latter refers to a system with multiple outputs of value, e.g. the energy utilization of a fuel can produce both steam and electricity. This leads to the question of how to fairly distribute the emissions associated with the energy utilization between both products, the steam and electricity in this example. The ISO provides a guideline with a certain hierarchy on which method to use to solve the problem of multifunctionality. However, the proposed hierarchy only works when all options are viable. This is not always a given. The attributional LCA model is more flexible, while the consequential LCA model by its nature is more limited. An attributional analysis allows for either allocation of emissions and system expansion with burdens/benefits. Both approaches will be briefly described in the following.

Allocation means the association of a certain amount of emissions with each product by a certain function or logic.[7] In the case of a wood-based product and potential second-life value creation, the emissions associated with the whole life cycle of both the primary and secondary products can be allocated by the market value of both products. Here, one product constitutes the first primary life cycle of the wood product, while the other product could be a raw material input for another product, for example.

However, allocation procedures do not have to necessarily follow an objective distributive function based on certain attributes, such as the market value or a lower heating value. Depending on the purpose of the LCA, other distribution mechanisms are plausible, such as 100:0 allocation, 50:50 allocation or other subjective allocation formulas. A 100:0 allocation means that the primary process carries all life cycle emissions up until the point of the beginning of the waste treatment or waste collection. In this way, the secondary life cycle is not burdened by the initial raw material acquisition or the emissions associated with the processing and production of the products' first life cycle. Additionally, the primary life cycle is free of any burden of waste treatment or the second life cycle in general. However, full credits or benefits, too, remain within the second product system. This method of 100:0 allocation is known as a cut-off, and is a special case of subjective allocation. Here, the goal is to highlight the benefits of a recycling scheme or a secondary product compared with a primary product (Detzel et al., 2016).

Allocation, regardless of objective or subjective emission distribution, however, constitutes an exception when focusing on the question of second life and EoL analysis, and should be carried out only when substitution/system expansion is not feasible (JRC 2010). On the other hand, accounting for benefits/burdens through system expansion is also a viable option. A system expansion describes a method where the analyzed product system is expanded by one or more systems reflecting the additional value of a multifunctional output. In the context of the EoL treatment assessment of wood-based products, the energy recovery of waste wood produces additional value by delivering electricity and/or heat. This additional value creation displaces or substitutes other products—in this case: electricity/heat. The primary system, the value chain of the now-turned-to-waste-wood product must be expanded by the average electricity mix production system, as it replaces/substitutes a certain amount of electricity, which leads to avoided burdens equivalent to the amount of substituted electricity. These avoided burdens now constitute the gross benefits of the energy recovery of the waste wood product. However, one must not forget the burdens, which go hand in hand with the burning

[7] The allocation can be carried out by market value (economic allocation) or physical attributes, e.g., lower heating value or exergy.

of the waste wood product, subsequent flue gas treatment, transport processes and the like. Considering this and subtracting it from the gross benefit results in the net-benefit of the energy treatment of waste wood product. If in the example of electricity generation, the displaced energy mix is fossil dominated, the net-benefit can be high. However, if mostly other renewables are displaced, the net-benefit could be negative, leading to a burden instead of a benefit for the primary product system.

Against the background of the additional modeling and calculation effort of system expansion, it is easy to see why it is considered a more complex methodology. Nevertheless, when there is no basis for an allocation or EoL treatment, system expansion often poses the best option. This also holds true for ALCA models, where, in terms of a comparison of two or more products/product systems, it is critical to ensure functional equivalency. Functional equivalency means that when a product's second life adds a unique value compared to the other, be it energy or as a raw material, for example, either the other system is negatively expanded with a comparable primary product to thus ensure theoretical functional equivalence, or a unique system is credited with the avoided burden associated with the same primary product life cycle expenditures. Otherwise, a comparison lacks any significance, since one product system generates more value than the other does.

Regarding a CLCA model in contrast, only the system expansion approach is applicable and reflects the inherited idea of depicting the consequences of a decision. Since there does not exist a be all end all solution as to how to account for recycling/multi-functionality of products in general, one must always consider the purpose of the analysis and refer to conventions.

Proposed approach on how to account for recycling, cascading and end-of-life

The key question is how the emissions and savings of recycling or cascading use can be included in the carbon footprint of products (CFP). This question is not methodologically standardized in the literature, despite multiple mentions of the topic (Carus et al. 2014, Höglmeier et al. 2015, Olsson et al. 2016, Rehberger & Hiete 2020, Vis et al. 2016, Fehrenbach et al. 2017). The attributional approach (allocation) mainly considered in this study is in line with the proposal of the ISO 14067:2018 standard within the informative annex "Possible procedures for treating recycling in CFP studies". The mentioned annex in ISO 14067:2018 handles possible procedures for how to treat recycling in CFP studies without precluding alternative procedures, provided those are in line with the ISO standards for LCA (ISO 14040 and ISO 14044).

Coherence with the LCA standards means ensuring that the allocation principles are observed. Furthermore, specific care should be taken when defining the system boundary concerning recovery processes. The standard—and the LCA community as a whole—distinguishes between two basic cases of recycling closed-loop recycling and open-loop recycling.

In the first case, the material of a product is returned to the manufacture of the same product after the use phase, or the material can be produced for the manufacture of another product without changing its original properties. The second case applies to open-loop product systems where the material is recycled into other product systems and the material undergoes a change to its inherent properties (ISO 14044:2006, clause 4.3.4.3.3, b). Among UPM's products, closed-loop recycling is classically applied in the area of paper products.

Apart from that, the question rather arises of how the potential for recycling or cascading of UPM products (as described in detail in the previous chapters) can be included in the carbon footprinting for these products. A basic possibility herewith exists within the framework of the open-loop approach. When a product consists of 100% primary material, then, in the case of open-loop recycling, the GHG emissions related to the raw material acquisition and end-of-life operations can be calculated in accordance with this formula:

$$E_M = E_V + E_{EoL} - R \cdot A \cdot E_V \tag{11}$$

Where:

E_M represents the GHG emissions tied to the raw material acquisition and end-of-life operations;
E_V represents the GHG emissions tied to extracting or producing all the raw material needed for the product from natural resources;
E_{EoL} represents the GHG emissions tied to end-of-life operations (being part of the product system which delivers recycled material);
R is the recycling rate
A is the allocation factor.

Thus, the term R · A · EV represents the recycling credit. This credit means sharing the GHG burden from "shared unit processes" for the open-loop recycling, such as processes for extraction and processing of raw material and the final EoL operations. In other words, the burden of providing raw materials for the first primary product, from which the later recycled product benefits are to be divided between these two products.

Two variables are of particular importance in the formula: 1) the recycling rate R, and 2) the allocation factor A. The recycling rate requires an empirical survey and precise knowledge of the extent to which the material is recycled in one or more subsequent product systems. This is a complex process. Official or industry-related statistics can be used here.

The allocation factor, on the other hand, is unempirical. A well-founded value-based judgment is required here. As explained in Chapter 2.2, this factor can be based on physical or economic parameters. Initially, there are two extreme options: an allocation of 100:0 (all loads go to the primary system) or 0:100 (all loads go to the secondary system, which is hardly justifiable in the application case). Another option is the cut-off option, i.e. the primary and the following secondary systems are sharply separated. Rarely can one of the options mentioned be compellingly justified and certainly not generalized. On the other hand, reliable data for the exact determination of specific allocation factors are also difficult to justify.

A 50:50 approach can therefore serve as a kind of compromise solution. This approach is mentioned, as it were, in the annex to ISO 14067 and is considered, among other things, to be the base case from the viewpoint of the German Federal Environment Agency. Detzel et al. (2016) evaluated numerous other standard procedures[8] and support this approach as a base case. However, there is no consensus among international experts on this issue, which becomes even more complex if instead of one recycling step after one use there is a staged multiple cascade. Various studies show that cascade use can lead to a higher overall environmental performance through more efficient use of primary resources within an interconnected material flow system (Vis et al. 2016, Fehrenbach et al. 2017, Höglmeier et al. 2015).

[8] E.g.: [BP X30-323] Afnor normalization: Affichage environnemental des products grande consommation. BP X30-323; Normalisation française. Numéro du document: N 066, 2011

[GPPS 2011] The Consumer Goods Forum (Hrsg.). Global Protocol on Packaging Sustainability 2.0. Moulineaux. www.theconsumergoodsforum.com

[ILCD 2010] European Commission - Joint Research Centre - Institute for Environment and Sustainability: InternationalReference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union; 201

[PAS 2050]: PAS 2050: Specification for assessing the life cycle greenhouse gas emissions of goods and services, publicly available specification. Defra (Department for Environment, Food and Rural Affairs, UK), DECC (Department of Energy and Climate Change, UK), BIS (Department for Business, Innovation and Skills, UK), 2011

But how should this benefit be distributed among the entirety of the various steps in a cascade? Rehberger and Hiete (2020) have analyzed the above options (100:1, 50:50, cut-off) concerning multi-stage cascades and have also evaluated numerous publications on this. Their findings reflect the diversity of possible approaches and the difficulty of developing a uniform solution. The conclusion to be drawn from this is that, in view of the increase in complexity, the methodological approach should be designed to be as uncomplicated as possible for practical application.

Additionally, there is the following problem: it is usually unknown, which uses are present in subsequent cascades in reality. To gain insight into this, regular market analyses would have to be carried out across many sectors. For this reason, the LCA studies in the literature are almost always based on constructed assumptions about the sequence of use cascades.

The most simplified approach would be a clear *cut-off*. The 100:0 option would also come very close to this, by the way. However, this approach has the disadvantage that the balance for the primary system has no benefit from subsequent cascades. Conversely, these cascades do not receive any burden from the primary system. To justify this approach, one could argue that the secondary systems are the essence of recycling and the primary system does not contribute anything. Such an extreme approach is therefore rather suggested for a sensitivity analysis.

In the case that it is definitely unknown which recycling loops a product takes, one approach would be to leave them out and credit the ultimately unavoidable final fate in the balance of the primary product. This would almost certainly be energy recovery—the last link in a cascade. This proposal would maintain consistency and cascade use would at least be taken into account in the single stage. Thus, this approach is subsequently termed *supposed single stage cascading*. What of all things supports such an approach? The property of consisting of biogenic raw material is based on the production of the primary product, since biomass is primarily used for this. Without this product, the biomass would not be in circulation, not even for later products in a cascade. It is therefore justified to credit the final energy benefit of this biomass to the primary product.

In the case that the recycling steps for the product are known, the consideration is to include them. However, this should be done with a method that is as pragmatic and straightforward as possible. One such method could be the 50:50 method. ISO 14067 describes the setting of such deterministic allocation factors as e.g. 0.5 as arbitrary "such a factor is justified if the criteria for the allocation mentioned in ISO 14044 (e.g. physical properties, economic value, number of subsequent uses) are neither feasible nor applicable."[9] Here, neither physical nor economic criteria offer a solution to this allocation problem. The first life cycle stage of cascade benefits from being "waste-free" but requires full input of raw materials. The second life cycle stage, on the other hand, is free of raw materials but bears the burden of disposal. Considering both as a coupled system, there is no more plausible approach than to share the burdens of raw materials and disposal equally between the two subsystems (50:50). Moreover, this approach is recommended because a comprehensive LCA or carbon accounting for a complex cascade system cannot be carried out using simple guidance. These systems are too complex and the question of how individual products produced in this system should be attributed is equally complex. Such tasks can only be carried out with comprehensive LCA of the entire context of a cascade system in accounting for an individual product.

The following figures illustrate these possible approaches. Figure 9 shows the complete basic basis of a cascade (example construction wood → particle board → energy recovery), including the alternative product system subsuming the substituted process systems throughout the cascade (construction steel; aluminum board; fossil energy).

[9] ISO 14067, informative annex D4, open loop allocation procedure

Figure 10 shows the option of excluding the possible but not exactly known cascades with the inclusion of the final energy use, referred to as *supposed single stage cascading.*

Figure 11 shows the 50:50 approach, which is limited to the first subsequent cascade/recycling step. The formula from ISO 14067 shown above is applied here, while the allocation factor A is set generally as 0.5. The carbon footprint for the construction wood takes only 50% of the raw material load, while the other 50% is allocated to the recycling particle board. Within the substituted alternative systems only the one corresponding to the construction wood system is accounted for. A cut-off approach would mean that any process below the first one (construction in the example) would be ignored, including the final energy use.

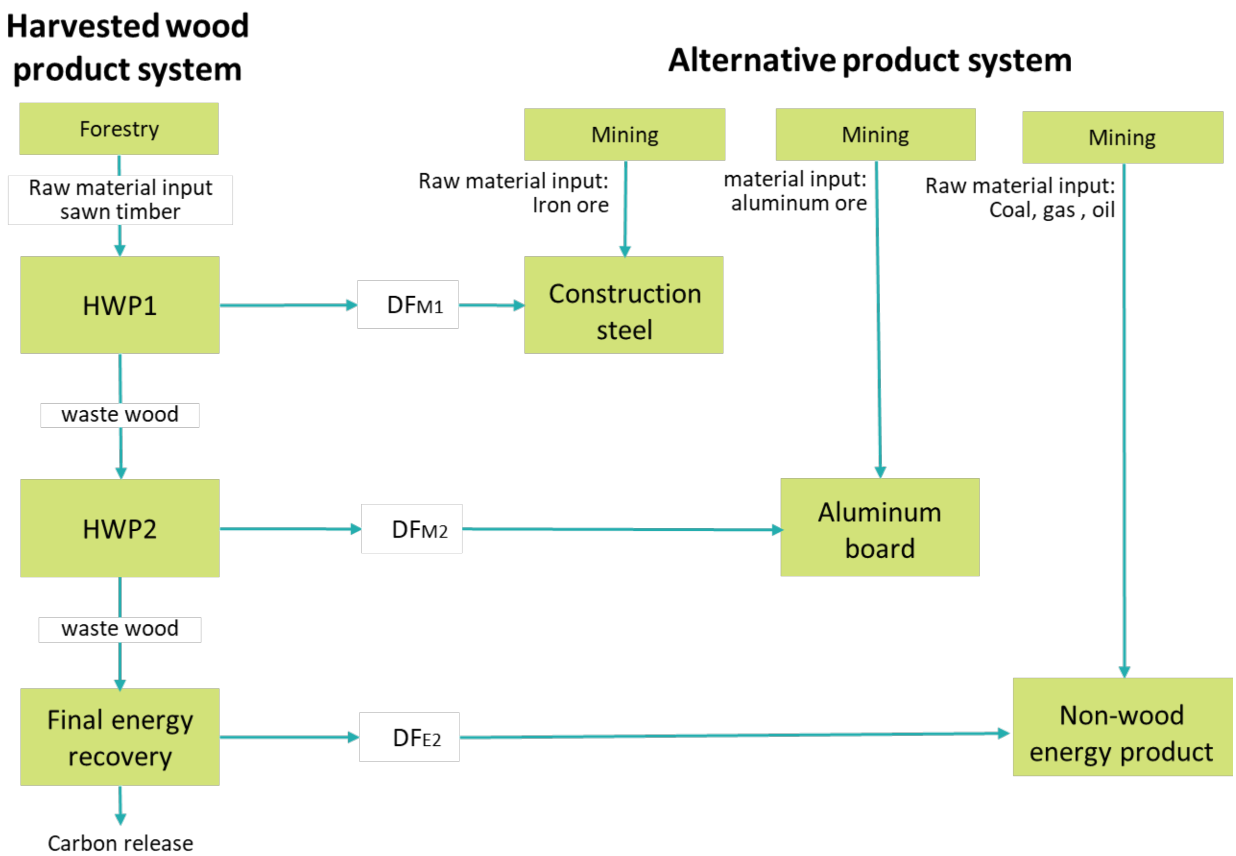

Figure 9. Illustration of cascading use of biomass by multiple steps including the alternative product system. ‘HWP’ refers to harvested wood product, ‘DF’ refers to displacement factor, ‘M’ refers to material and ‘E’ refers to energy.

5.3 Biomass cascading in different product groups

Fiber-based materials (paper sector)

This group comprises bio-based materials in the general paper sector. While the term ‘fiber’ applies to other products, most noticeably textile fibers. Due to the unique characteristics of paper fibers and their significance in the economy of bio-based materials, they constitute their own product group. The paper sector along with the separate collection of waste paper constitutes a well-established, significant, and successful biomass cascade (Fehrenbach et al., 2017). For 2020, the umbrella organization of the Confederation of European Paper Industries (CEPI) calculates a recycling rate of 73.9% for the EU27, Switzerland, Norway,

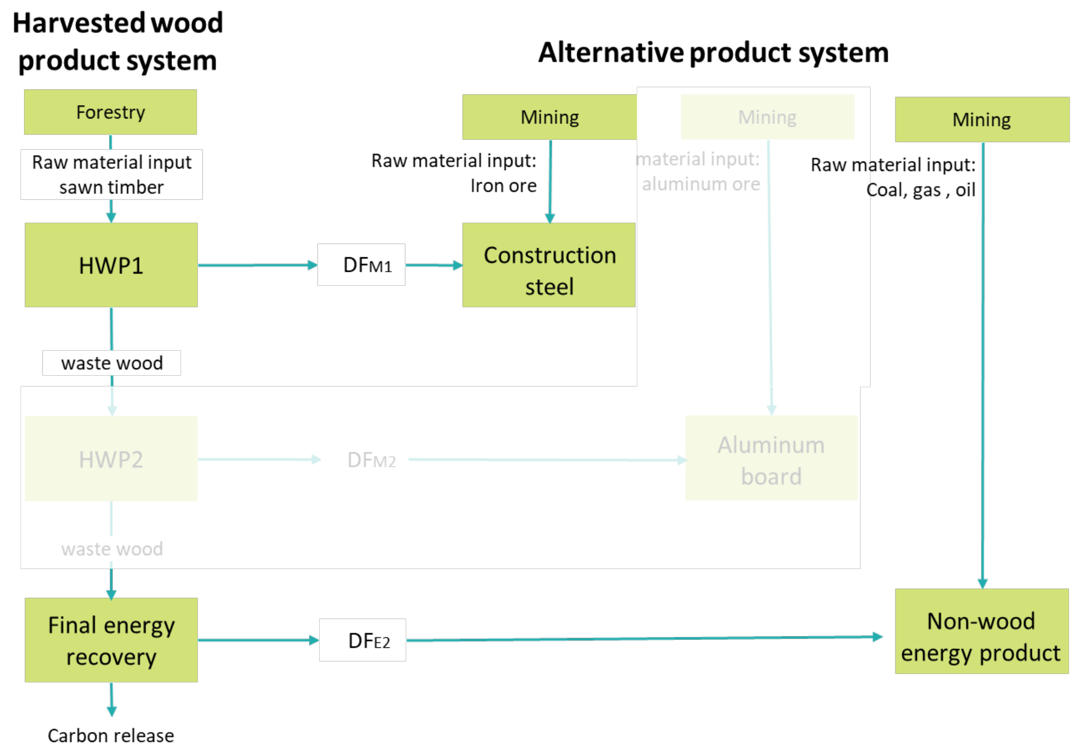

Figure 10. Illustration of cascading use of biomass by multiple steps excluding the intermediate recycling steps but including the final energy use (supposed single stage cascading). 'HWP' refers to harvested wood product, 'DF' refers to displacement factor, 'M' refers to material and 'E' refers to energy.

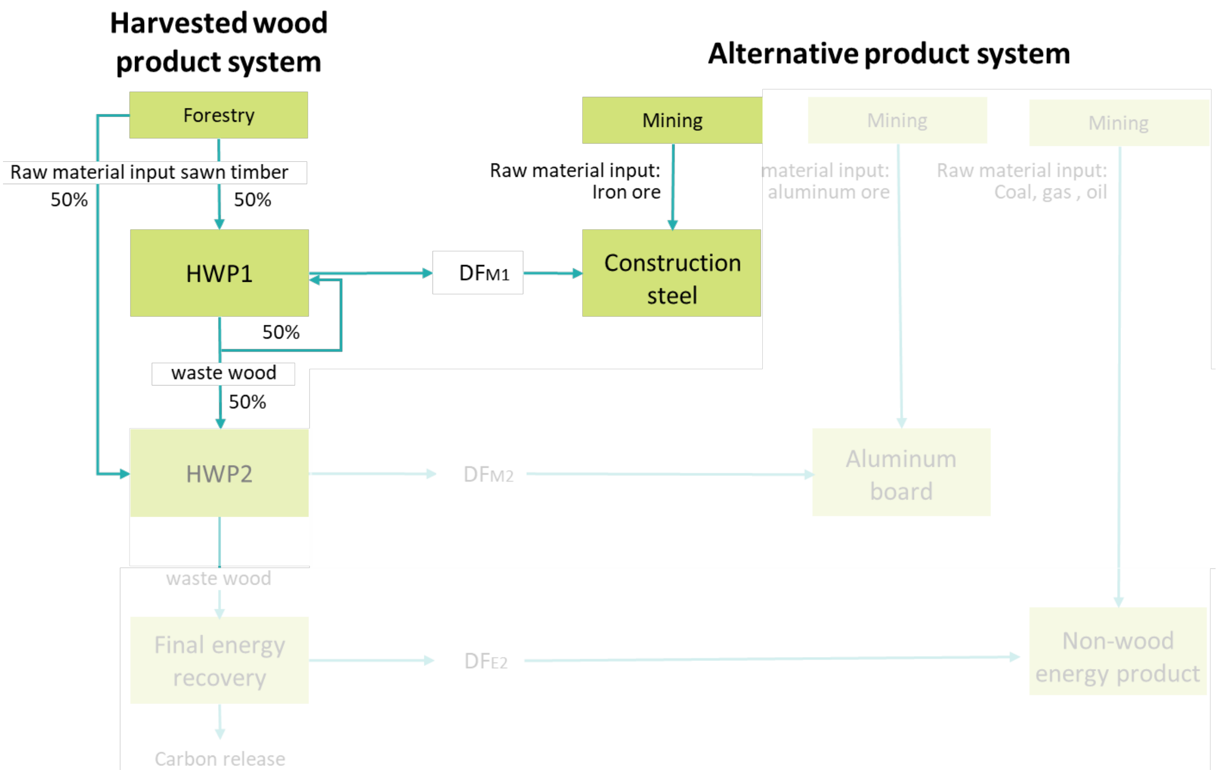

Figure 11. Illustration of cascading use of biomass by multiple steps referring to the so-called 50:50 allocation method. 'HWP' refers to harvested wood product, 'DF' refers to displacement factor, 'M' refers to material and 'E' refers to energy.

and the UK (CEPI 2021a).[10] To date, Europe achieves the highest recycling rate globally, with an average of 58.6% worldwide. The wood-based paper can—in theory—be recycled multiple times[11] (Putz & Schabel, 2018). In Europe, the average lifecycle of paper fibers comprises 3.8 cycles (global average: 2.4 cycles) on average, meaning that a paper fiber is used 3.8 times within the value chain in its intended or similar role (CEPI 2021b). Compared to the base year 1998, a 40% increase in recycling has been achieved within the paper sector for the EU27, Switzerland, Norway, and UK.

Further improvements within the paper sector pertain the design of fiber-based products toward a more design-for-recycling/recycling-friendly approach and a more general further increase in a separate paper collection, which could result from the former. To this end, the utilization of different printing inks and adhesives, the likes of which are certified with the "Blauer Engel" ("Blue Angel)" ecolabel constitute opportunities (UBA 2021).

Generally, UPM fiber products comprise the following sub-categories:

- Tissue paper products (hygiene applications)
- Self-adhesive label stock (food packaging labels, beverage labels, cosmetic products labels,..)
- Graphic paper products (newspaper, magazines,..)
- Packaging material
- Fine paper (printing paper, books).

While the first group—tissue paper—is not suitable for the well-established paper fiber recycling due to its application and subsequent disposal, the cascade ends with energy recovery (single-step cascade). The carbon retention time in these products is therefore limited to the lifetime of the tissue paper and thus constitutes only a short-time carbon sink. With the energy recovery, the carbon content is released again into the ecosphere.

Although the second group, self-adhesive label stock presents a material group which is usually lost due to dissipative losses, in the case of UPM, there exists a recycling scheme geared for material recovery of parts of the waste that arise during the manufacturing process, the so-called liner. Here, a distinction has to be made.

Case a) the liner material waste is collected and treated within the UPM label recycling scheme (UPM RafCycle). Then, a multi-step cascading is possible with a carbon retention time equivalent to the lifecycle time of the label material multiplied by the number of possible cycles, before the material is no longer suitable for its purpose of feedstock for labeling. Here, again a distinction must be made, either, the material will be utilized for other applications in a material sense (case a1), e.g., packaging material, then the lifetime of the bound carbon will extend correspondingly depending on the number of cycles of the packaging material. Another possible utilization route would be energy recovery (case a2). Here ends the cascade, the carbon is released into the ecosphere.

Case b) the liner material waste is not collected and treated within the UPM label recycling scheme. Then, assuming the default waste treatment of the country of application, most likely the energy will be recovered. The cascade ends in a single-step cascade.

[10] The CEPI defines the recycling rate as the ratio of used paper recycling, including the net trade of paper for recycling, on the one hand and the total consumption of paper and board material on the other, meaning that paper fibers destined for recycling in other regions outside Europe are included.

[11] The often-cited figure of up to seven recycling cycles before the paper fiber no longer fulfills quality requirements is disputed. Putz and Schabel (2018) conclude that the theoretical recycling limit of paper fibers is determined by increasing impurities instead of deterioration of the fibers themselves.

In the case of the other product groups, a separate collection of paper products can be assumed. Here, the above-described established paper recycling leads to multiple product cycles with corresponding carbon retention. The overall carbon retention time depends then on the specific product. A book can reach long lifetimes in the context of paper fiber material, whereas a newspaper is more short-lived, with printing paper constituting an intermediate case, depending on the consumer. Another important differentiation must be made with respect to the primary product quality and possible contamination with other materials, e.g. chemicals, plastics or coating that could render a waste unsuitable for paper recycling. Here, the only treatment option would be energy recovery.

Solid wood products

The product group of solid wood products in the context of cascading can mainly be divided into the following categories: construction and demolition wood, particleboard, wood packaging, wood furniture and wood-based bio-refinery concepts (Vis et al. 2016). Compared with the fiber sector (see above), the utilization of solid wood products is more diverse with more application options. This, however, has significant influence on the possibilities of a successful cascade, as quality and availability are key parameters determining the fate of post-consumer solid wood waste.

Each product group described above has specific influencing factors that are primarily determined by the post-consumer sector and vary with the intended use of the products. According to Vis et al. (2016), the main challenges regarding the availability of waste wood for successful multi-stage cascading in wood-based biorefineries can be summarized as follows:

- lack of a source-separate collection of post-consumer wood (e.g. compared to other waste flows such as glass or paper) and/or related legislation
- contamination due to the application of material utilization poses a challenge, coating, wood preservatives and other materials [paint, glue, heavy metals (Pb, As, Cd)]
- contamination/Mixing with other waste streams, such as metals (e.g. in furniture) or minerals (e.g. demolition waste)
- general lack of defined and internationally agreed upon waste wood categories and legislation, inconsistencies in contamination thresholds and End-of-Waste criteria
- lack of a holistic assessment/strategy of biomass application in both energy and material utilization[12]
- economic- and technical limitation of waste wood recycling.

Wood-based bio-refineries valorize wood as a raw material and produce several outputs, such as bio-ethanol, PLA, and other chemicals. While these are either not suitable for a cascading scheme, e.g. bio-ethanol as a bio-fuel in transport, other products, e.g. bio-plastics, will leave the biomass cascade in ways of recycling/recovery in other waste sectors (Vis et al. 2016). However, the bio-refinery concept based on wood can be an opportunity for the material valorization of waste wood as input and thus constitutes a second/subsequent step in the wood cascade rather than a significant first step. Additionally, the transformation of wood within the bio-refinery can pave the way for a successful multi-step recycling/cascade, if the derived product shows corresponding characteristics and a recycling scheme for the new product category is in place (Vis et al. 2016).

In contrast to the paper sector, cascading solid wood products is significantly less well developed and for a successful implementation, there are still numerous challenges to be solved. However, on the other hand, the

[12] This could change with a possible ratification of the RED III draft of the EU Commission

effects of cascading solid wood products could contribute significantly toward a greener future. Especially, the utilization of wood in the construction sector could constitute additionally increasing carbon storage for meaningful time frames of decades or even centuries. With each successful additional cascade of wood in this sector, the high specific carbon retention times add up and could thus in absolute terms surpass other efforts with shorter carbon retention durations, even though the latter might outweigh construction wood.

Similar effects hold true for other, more short-term applications of wood cascades, such as particleboard or wood furniture, albeit to a lesser extent due to shorter product life spans. Generally, solid wood products cascades are thinkable in various configurations and pathways, depending on the condition of input material (waste wood) and market demand. Especially, the development and emergence of the bio-refinery principle opened up additional opportunities, e.g. the production of bio-chemicals to substitute fossil-based materials or biofuels. Fehrenbach et al. (2017) investigated four different possible cascades for solid wood products with a focus to Germany, comprising: 1) increasing the waste wood share in particleboard production, 2) redirection of primary wood resources bound for energy utilization to material use, 3) redirection of primary wood resources bound for energy utilization/waste wood to the chemicals sector (Biomass-to-liquid, BtL), and 4) redirection of all primary wood resources bound for energy utilization to a material use. All options 1) – 4) showed advantages in predominantly all investigated impact categories with especially option 4) showing significant ecologic advantages.

Even though solid wood products constitute a more diverse product group in terms of application, UPM products can be summarized in four sub-groups, following their utilization:

- construction wood in housing/buildings
- construction wood in mobile applications, such as ships, busses, cargo transport
- wood furniture and flooring
- solid wood packaging.

The first product sub-group is usually recovered together with other construction materials or heavily contaminated with either other construction materials or wood preservatives applied to prevent decay or other hazards. For these materials, a dedicated recycling scheme for material recovery is not in place to this date. As the only option, energy recovery remains at the end of a cascade. However, wood-only construction without any added chemicals is thinkable. Here, a multi-step cascade is thinkable with a second life of the wood material in the form of either chipboard panels with different applications such as furniture, for example, or a direct use for construction in a simplified manner, such as cabins or wooden banks. In the case of chipboard panels, a contamination with wood glue and possible other chemicals (e.g. fire protection agents) takes place, leading to an EoL with subsequent energy recovery. A potential second use in construction, the application and circumstances (indoor vs. outdoor, moisture content of the surrounding area etc.) determine the degradation of the untreated wood. In principle, the same options as for the primary product material remain, with the state of decay and remaining quality as the main determinants. Carbon retention times are very high in this sub-group, due to the long—on average—lifetimes of buildings. Following this, the building application of wood constitutes a real increasing carbon storage with a plausible effect on the climate, if there is a sufficiently large-scale application or a special emphasis on building with wood is placed. Moreover, the forest carbon stock can recover better the longer a wood product is in use and no new wood has to be produced as a replacement, leading to an added bonus, when compared with short-lived applications.

For the second and third UPM sub-groups, construction or structural wood in mobile applications and wood furniture and flooring, the general same principles apply if a wood product is treated with chemicals or wood preservatives, material recovery is impossible in most cases. If this is not the case, the products in this sub-group can be part of a dedicated wood recycling scheme with a second material lifetime, which, in most cases consists of utilization in chipboards. Here, the same principles for chipboard recovery apply, with energy recovery as the most likely outcome. The carbon retention time of material used in these sub-groups varies according to the specific application and respective lifetimes. If, for example, the wood product is used for the interior cladding of a ship, the lifetime of the wood product will usually mirror the ship's lifetime, which can span several decades. A second life in the form of a chipboard for use in furniture, may add some additional several years to decade(s). On the other hand, a wooden floor can last as long as the building it is part of, which can even exceed centuries. Here, too, UPM wood products can constitute a de facto carbon sink in the technosphere with the added bonus of forest carbon stock recovery, as stated above. However, the range of carbon retention time is larger and more dependent on the specific usage form.

In case of wooden packaging, the fourth UPM sub-category, re-use of the solid packaging or container in the same or a comparable role can be assumed when the packaging is used in a B2B case. In the sense of cascading, the re-use does not constitute a cascade if the packaging has not reached EoL status. For this group, the same general principles in the context of wood treatment apply. Additionally, the transported goods matter, too. If the transported good contaminates the wood packaging, e.g. in the case of solvents or other (highly) volatile substances, the packaging wood could be contaminated, which leaves few treatment options and, depending on the grade of contamination, even hazardous waste incineration could be appropriate. Compared to the above-mentioned applications, solid wood packaging material has a shorter lifetime due to its utilization and material strain. The carbon retention time will be scaled accordingly. If no contamination, be it due to wood preservation or contamination via packed goods, takes place, a material recovery is plausible, comparable to the cases described above, with the additional second life and subsequent energy recovery.

For all solid wood products that are separately collected, in principle, utilization as a feedstock for bio-refinery concepts is thinkable. However, very few such refineries exist to this date. The suitability and conditions of the solid waste wood is thus unclear. Additionally, the economies and logistics of this path as a competitor to established routes are yet to be determined. If the bio-refinery of the future thus constitutes a real option remains to be seen and is subject to legislature and markets.

Bio-Chemicals

The product group of bio-chemicals comprises various gaseous, liquid and solid materials and is thus even more heterogeneous, compared to solid wood or fiber products. Independent of the aggregate state of the product, a chemo-technical transformation of the wood input occurs, which determines the potential for subsequent cascading. Moreover, the question whether or not a recycling scheme for the derived products is already established, or not, is crucial (Fehrenbach et al., 2017). The latter is influenced by the degree of market penetration of the products. As an example, PET as a high-value plastic is collected separately in some countries, leading to a high degree of material recycling—and thus cascading in terms of bio-PET.

Solid bio-chemicals in general can be subdivided into product groups with either a conventional equivalent (bio-PE, bio-PET) or a new product (group), like PLA. Furthermore, if bio-chemicals are equal in their chemical structure (as is the case in terms of bio-PET and conventional fossil derived PET, another example would be bio-PE and conventional PE), the bio-variants are collected together with their respective conventional equivalent. This mixing with conventional equivalents makes further analysis of the cascade of bio-chemicals impossible. This is not the case, if the bio-chemicals constitute a new product without conventional equivalent. Fehrenbach et al. (2017) discuss PLA as such a new product. However, as outlined above, new products often lack the necessary market volume to economically justify a separate collection infrastructure.[13]

Gaseous products such as biogas or bio-methane (produced from digestate as a by-product) as well as some liquid products, e.g. bio ethanol can be utilized to produce energy, where—in most cases—further cascading

[13] Fehrenbach et al. (2017) cite a threshold of 20.000 t /p.a. PLA material.

is impossible. However, in light of the current debate and the emergence of synthetic fuels for shipping or aviation or the provision of non-fossil chemical feedstock, CCU could provide a solution, which could apply also to the aforementioned energy carriers. In this way, a further cascading of the biogenic carbon can be achieved, though here, too, a dedicated and transparent comprehensibility of the biomass cascade would require biomass specific products in the subsequent downstream process chain and avoidance of mixing with conventional products/waste.

Liquid bio-chemicals, e.g. biodiesel or bioethanol are overwhelmingly used in transport. If there is no downstream carbon capture,[14] further cascading is not possible. In other cases, the same principles as outlined above apply. Other liquid bio-chemicals entail bio-naphtha, which is usually a chemical feedstock to produce plastics. Here, too, the chemical equivalence to conventional products along with a collection and recycling scheme determines the cascade success and traceability. Other applications of liquid bio-chemicals are coating and other (thin film) surface applications. Though theoretically possible to separate the surface application from the bulk material, to this date, there does not exist an established recycling scheme focusing on different vapor pressures for different materials in this sector. The coating/surface materials can thus be considered dissipative losses and even, in some cases, a contamination of the bulk material (e.g. wood coating, see above).

The UPM bio-chemicals can be clustered as follows:

- chemicals for energy purposes
- liquid chemicals
- solid chemicals.

In case of the chemicals for energy purposes, UPM products are utilized in mobile applications, mainly transport. Here, no cascading is possible as no mobile CCS or CCU units exist to this day. The carbon retention time can thus be considered from days to months, at best, depending on the value chain or, in other words: How long will it on average take for an energy carrier to be stored, sold and used.

The other sub-categories, liquids and solid chemicals, are mainly determined by the specific end product, its characteristics and moreover, the existence of recycling scheme of the specific product group or material group. All solid products (PE, PVC, PET, rubber etc.) are chemically indistinguishable from their fossil equivalents and thus, when entering EoL status, subject to the same recycling schemes. Generally, if a product consists of different materials (so called composites), material recovery is usually impossible due to the significant effort required to break down the composite into its source materials. Energy recovery remains the sole treatment option currently. If a product is not composed of different source materials, e.g. a PET bottle or bulk PE application, the success of a material recovery is mainly determined by the recycling scheme of the country where the product reaches EoL or, in the case of waste exports, is treated. In Germany, for instance, a separate collection scheme for PET bottles exists, leading to a separate PET waste stream with subsequent material recovery. The material can, in principle, be recycled multiple times, depending on the contamination and recycling scheme of the material and country. Carbon retention time varies, accordingly. Here, various possibilities must be assumed, depending on the specific use case, and recycling scheme in place. The short-lived composite with bio-chemicals undergoes energy recovery after a couple of days, only, while a PVC product in a building can last for decades. For the purified Kraft lignin/phenolic resins, the application as a (surface) coating renders a material recovery as is impossible. The cascade is determined here by the bulk material/product to which the phenolic resin has been applied.

[14] To this date, a CC stage in mobile application is not subject of discussion

6. The importance of temporary carbon storage and the timing of greenhouse gas emissions

6.1 Characterization of temporary carbon storage and the timing of emissions

When considering GHG balances over the life cycle of HWPs and their alternative products, GHG emissions and carbon sequestration occur at different points of time. This is particularly the case for long-lived products and/or material recycling and end-of-life energy recovery. Temporary carbon storage in HWPs or in its alternative products means that carbon is released later than without such temporary carbon storage (i.e., immediate release). Similarly, GHG emissions generated and/or avoided in recycling and energy recovery of primary products take place at later points of time compared to production and use of primary products.

When accounting for GHG balances on an annual basis, like in the reporting of GHG emissions to the UNFCCC, there is no need to characterize GHG emissions occurring at different points of time. Similarly, carbon dioxide sequestration to and emissions from HWPs may be accounted for on an annual basis based on the assumptions used for carbon inflow and outflow from the HWP stock (see Chapter 3). This information may be relevant for companies when reporting the GHG balances related to their activities. However, in LCA, there is a need to combine GHG balances occurring at different points of time over the life cycle or temporal scope determined to a single indicator. Consequently, in LCA a key question to examine is the climate effect of delayed emissions or delayed avoided emissions and how they should be considered and characterized compared to immediate emissions.

Due to temporary carbon storage, the concentration of CO_2 in the atmosphere is temporarily reduced and some radiative forcing is avoided. In terms of cumulative radiative forcing, storing an amount of carbon dioxide over a certain period is equivalent to delaying an equivalent amount from a carbon dioxide pulse emission for an equivalent period (Levasseur et al., 2010). The longer the delay in emissions or the shorter the time horizon considered, the lower the cumulative radiative forcing of the delayed emissions, thus the higher the related climate benefit of temporary carbon storage (Levasseur et al., 2012; Helin et al., 2016). However, temporary carbon storage only reduces climate change impacts related to the cumulative effect of increased temperature and could even worsen impacts mediated via the instantaneous effect of temperature (e.g. measured using global temperature potentials, GTPs) or the rate of temperature change (Kirschbaum, 2006). This is because the global carbon cycle and the instant carbon dioxide concentration and the related instant radiative forcing of a carbon pulse emission are at their highest at the closest to the time of the emission. Thus, the importance of temporary carbon storage and the timing of emissions on the climate depend on the climate metrics and time horizon chosen (Brandão et al., 2013).

The 100-year GWP as a climate metric is commonly applied to determine the relative contribution of different GHG emissions. The GWP metric takes into account the cumulative radiative forcing of a GHG (or some other climate forcer) over a given time horizon (e.g. 100 years) and neglects the impacts thereafter. The 100-year GWPs compare radiative forcing integrated over 100 years for non-CO_2 GHGs with that of CO_2, thus they ignore radiative forcing beyond 100 years in determining their relative warming impact. A conventional practice in LCA has been to ignore the timing of emissions, thus temporary carbon storage or delayed emissions are not assigned any credits or debits within the time horizon considered (Brandão et al., 2013). In other words, this mean that all GHG emissions occurring at any point of time within a time horizon considered are of equal importance. If the time horizon considered is infinite, temporary carbon storage plays no role and has no effect as carbon sequestered is assumed to be fully released at some point in time. If the time horizon considered is finite, the credits or debits of temporary carbon storage depend on how much carbon is assumed to remain unreleased within the given time horizon. For example, within a 100-year time horizon, the storage time of 20 years is assigned with no credits (full emissions occurring), while a storage time of 101 years is assigned with full credits (no emissions occurring).

Determining the cumulative radiative forcing over a fixed 100-year period (e.g. 2022–2121) for a delayed emission (or avoided emission) occurring in the middle of a particular period (after 50 years, i.e. in 2072), ignores radiative forcing beyond a fixed 100-year period (i.e. after 2121), thus only accounting cumulative radiative forcing over 50 years (i.e. 2072–2121). This is the idea for so called dynamic LCA method (Levasseur et al., 2010) or $GWP_{bio,\ product}$ method (Helin et al., 2016).[15] These concepts would benefit delayed emissions whether of biomass-based or fossil origin. There are also methods developed to handle the atmospheric stay of carbon dioxide related to temporary carbon storage and the timing of emissions. These include the Moura-Costa method (Moura-Costa and Wilson, 2000), the Lashof method (Courchesne et al., 2010), the PAS 2050 method (BSI 2011), and the ILCD handbook method (JRC 2010). Some of these have similarities with the above-mentioned dynamic LCA approach, while some of them are simpler, and some combine properties from different approaches. The pros and cons of the six methods referred above are discussed by Brandão et al. (2013).

The fundamental problem with GWP metrics and dynamic LCA methods with similar properties (Levasseur et al., 2010; Helin et al., 2016) is that they only account for cumulative radiative forcing within the time horizon chosen and exclude the effect beyond that. Consequently, the results and conclusions may depend significantly on the time horizon chosen. In addition, the global warming potential based on cumulative radiative forcing is a midpoint indicator in the life cycle impact assessment, while the global temperature potential (GTP), which describes the global mean surface temperature change at a given future time horizon, can be seen closer to an end-point indicator for climate change. Besides the global warming potential or global temperature potential, there are also climate metrics available such as aSET (Smith et al., 2012), which is an absolute metric that refers to the contribution to a global mean temperature peak without time dimension.

6.2 Conclusions on temporary carbon storage and the timing of emissions

Temporary carbon storage only reduces the climate change impacts related to the cumulative effect of increased temperature and could even worsen the impacts mediated via the instantaneous effect of temperature or the rate of temperature change (Kirschbaum, 2006). Thus, the credits or debits assigned to temporary carbon storage or delayed emissions depend on the climate metrics and time horizon chosen (Brandão et al., 2013). As it is impossible to suggest a unique metric that would be appropriate to tackle different climate impact perspectives, various metrics applied over various time horizons may be required, depending on the goal and scope of a study.

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[15] Helin et al. (2016) presented a cumulative radiative forcing over a fixed time horizon (1-101 years) of a carbon dioxide pulse emission taking place in different points in time due to temporary storage time of 5, 10, 20, 40, 60 or 80 years in comparison to the carbon dioxide pulse emission taking place in the beginning of the time horizon. Such information can be used to weight the temporary carbon storage in terms of cumulative radiative forcing compared with an immediate release of carbon dioxide. According to Helin et al. (2016), for example the temporary carbon storage time of 40 years has roughly 35% lower cumulative radiative forcing over 100 years than immediate release of the same amount of CO_2. Simultaneously, this means that carbon dioxide pulse emission taking place at year 41 results in 65% (100% minus 35%) cumulative radiative forcing between years 1 and 101 compared with carbon dioxide pulse emission taking place at year.

7. Guidance for assessing the climate impacts of HWPs and wood-based energy in the technosphere

7.1 General

Today it is possible to roughly assess the substitution and carbon storage effects of HWPs and wood-based energy caused by production stages at product and company levels. In addition, this concerns the energy recovery of HWPs. Furthermore, the carbon storage change of HWPs can be assessed. Evaluating the climate effects of the use of wood, changes in carbon stocks in forests and HWPs and changes in fossil carbon emissions should be considered coherently. To do that, two systems are compared to each other; namely the one with the wood use being studied and its reference system without the wood use being studied (Figure 12).

Next, we provide a practical example on how to assess GHG effects related to HWPs and wood-based energy including substitution effects. We follow the attributional LCA approach, in which the product system is assessed as it is, and this is compared to the reference system, in which the particular product system would not exist but the alternative product system would exist. The relevant life cycle stages from forest to wood end-use and from alternative raw material extraction to its end-use must be considered and connected to each other coherently. According to the international rules for GHG accounting and reporting, forest-biomass-based CO_2 balances are accounted for and reported through carbon stock changes in forests and HWPs, thus CO_2 emissions from bioenergy combustion are accounted for and reported as zero in the energy sector. However, in this guideline we only focus on assessing carbon stock changes in HWPs and fossil emission substitution due to using HWPs and wood-based fuels in place of non-wood materials and fuels (Figure 12). The assessment requires several assumptions and the quality of input data vary case by case. For this reason, it is important to report the results transparently, showing the used methodology, assumptions and input data.

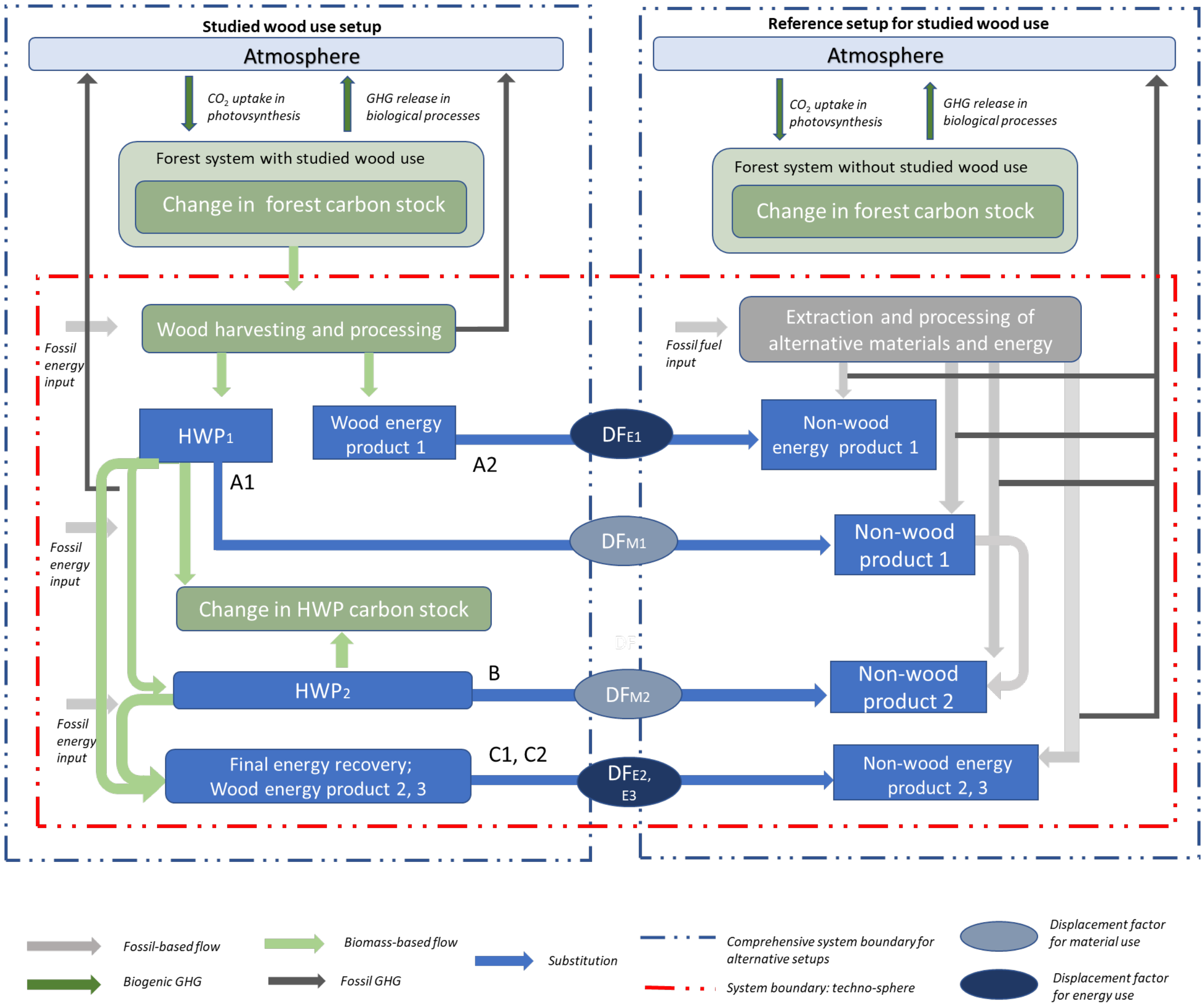

Figure 12. Schematic description of factors affecting the greenhouse gas (GHG) balances when using wood as materials or energy in place of alternative non-wood materials or energy. The red line illustrates the boundary of factors considered in this guideline. The IPCC rules for GHG emission reporting are followed, i.e. CO_2 emissions from biomass combustion are not accounted for in the energy sector, instead carbon stock changes in forests and harvested wood products (HWPs) are accounted. The indexes A1, A2, B, C1, and C2 refer to the substitution effect of HWP1 (A1), wood energy product 1 (A2), HWP2 (B), energy recovery of HWP1 (C1) and energy recovery of HWP2 (C2), and DF_{M1}, DF_{M2}, DF_{E2}, and DF_{E3} refer to the displacement factor of material use 1, 2, and energy use 1, 2.

Fig.12 is a simplification describing the flows of HWPs and wood-based fuels and their replacement of alternative products and fuels. In spite of that, we follow the wood flows of the figure describing each stage to quantify substitution effects of HWPs and wood-based fuels and changes in carbon storage of HWPs.

7.2 Assessment at the product level

Substitution effect of an HWP and wood-based energy over the complete life cycle

This guidance divides the stages of substitution effects as follows:

Stage A1: Substitution effect of a production HWP1 – DF_{M1} – Non-wood product 1
Stage A2: Substitution effect of the production stage: wood energy product 1 – DF_{E1} – non-wood energy product 1
Stage B: Substitution effect of the recycling and cascading of HWP1: HWP2 – DF_{M2} – non-wood product 2
Stage C1: Substitution effect caused by the final energy recovery of HWP1: wood energy product 2 – DF_{E2} – non-wood energy product 2
Stage C2: Substitution effect of the final energy recovery of HWP2: wood energy product 2 – DF_{E2} – non-wood energy 2

Stage A (A1 and A2) represents the primary origin of the product life cycle and is therefore fundamental. Stage A2 may apply in case co-products for energy recovery occur align with stage A1, or independently in case of pure energy production. Stages B and C (C1 and C2) represent different cases of recycling, cascading and EoL of HWPs. An illustrative example on how to calculate the substitution effect for a sawn wood product is provided in Appendix B.

Stage A1) Substitution effect of a production HWP1 – DF_{M1} – Non-wood product 1

Stage A1 is straightforward: the emissions of and carbon storage remaining in the disposal are part of the life cycle of HWP1.

When HWP1 has been manufactured and delivered to the market, it is assumed that HWP1 replaces an alternative non-wood product 1 in the market. Before we can make this assumption, we must ensure that HWP1 and non-wood product 1 have the same functionality, i.e. the products have the same functional unit (ISO 14044). Another requirement is that HWP1 really replace non-wood product 1 in the market. This is more a theoretical requirement as it is difficult to ensure (see the notes at the end of this sub-section).

If the production of HWP1 will cause less fossil GHG emissions than the production of non-wood product 1, the HWP1 avoids the GHG emissions in the market and this is quantified as a substitution effect. The substitution effect (SE) is quantified by multiplying the production amount of HWP1 by its corresponding displacement factor (DF_{M1}). Thus,

$$SE_{HWP1} = DF_{M1} * PA_{A1} \qquad (12)$$

, where DF_{M1} is expressed in avoided fossil carbon emissions per used carbon content in HWP1 (t C/t C) related to the production stage and PA_{A1} is the produced amount of HWP1, expressed in mass of carbon (t C).

DF_{M1} is determined according to the following equation (Sathre and O'Connor 2010):

$$DF_{M1} = \frac{GHG_{HWP1} - GHG_{non-wood\,product\,1}}{WU_{HWP1} - WU_{non-wood\,product\,1}} \quad (13)$$

, where GHG_{HWP1} and $GHG_{non\text{-}wood\,product\,1}$ are the *GHG* emissions resulting from the use of HWP1 and non-wood-product 1, expressed in mass units of carbon (t C). WU_{HWP1} and $WU_{non\text{-}wood\,product\,1}$ are the amounts of wood included in HWP1 and non-wood product 1 expressed in mass units of *C* contained in wood. Note that non-wood product 1 can also contain wood, but wood is not a major material in it.

The calculation of GHG emissions in Eqs 12, 13 is based on the use of GWP (global warming potential) factors for different GHG emissions to express results as CO_2-eq. of the emissions. The GHG emissions represent fossil-based emissions along the life cycles of products and fuels in the techno-sphere. Furthermore, the calculation of the avoided GHG emissions caused by a production stage includes the fossil GHG emissions allocated to an end-product caused by forestry and harvesting practices, transportation of roundwood and manufacturing. Additionally, the corresponding life cycle data for the non-wood-based alternative is required. The calculation guidelines of life-cycle based GHG emissions are described in general in various guidelines of LCA (e.g. JRC 2010), and for this reason, we do not present them in detail here.

The GHG emissions for HWP1 and non-wood product 1 in Eq. 13 can be obtained directly from the life cycle databases such as Ecoinvent. However, before they can be used in the determination of DF_{M1} the system boundaries of both products should be analogical and data applied should be representative (e.g. both data should represent similar geographical areas and years).

The value of DF_{M1}´can be directly found from the literature without using Eq 13. However, the assessment bases for DF_{M1} should be known before using it for assessing substitution effects.

In practice, HWP1 may replace several different non-wood products with different market shares. Additionally, in some cases there might not be any specific non-wood product alternative available. This information can only be obtained from the specific market surveys or assumed case by case. The results are always subject to uncertainties as they are assumed to be estimations of reality. Let's assume that HWP1 replaces non-wood product 1a with a market share of 50%, non-wood product 1b with market share of 10% and 40% of the use of HWP1 is assigned with no non-wood product alternative, i.e. 40% of HWP1 does not replace any non-wood product alternative. In such a case, DF_{M1} can be calculated in the example as follows: $DF_{M1} = 0{,}5 * DF_{HWP1\text{-}NWP1a} + 0{,}1 * DF_{HWP1\text{-}NWP1b}$. Thus, the determination consists of two determinations of the sub-DFs. The sub-DFs should be determined according to Eq. 13 with the requirements presented above.

Stage A2) Substitution effect of the production stage: wood energy product 1 – DF_{E1} – non-wood energy product 1

When wood energy product 1 is manufactured and delivered to the market, it is assumed that it replaces an alternative non-wood energy product 1 in the market. Wood energy product 1 may be a co-product of HWP1 or a main product. The substitution effect of stage A2 can be calculated similarly than in Stage A1, using Equations 14 and 15:

$$SE_{WE1} = DF_{E1} * PA_{A2} \quad (14)$$

, where DF_{E1} is expressed in the fossil carbon emissions avoided per the used carbon content in wood energy product 1 (t C/t C) related to the production stage and PA_{A2} is the produced amount of wood energy product 1, expressed in mass of carbon (t C).

DF_{E1} is determined:

$$DF_{E1} = \frac{GHG_{WE1} - GHG_{non-wood\ energy\ product\ 1}}{WU_{WE1} - WU_{non-wood\ energy\ product\ 1}} \quad (15)$$

, where GHG_{WE1} and $GHG_{non\text{-}wood\ energy\ product\ 1}$ are the *GHG* emissions resulting from the use of wood energy product 1 and non-wood energy product 1, expressed in mass units of carbon (C). WU_{WE1} and $WU_{non\text{-}wood\ energy\ product\ 1}$ are the amounts of wood included in wood energy product 1 and non-wood energy product 1 expressed in mass units of *C* contained in wood. Note that non-wood energy product 1 can also contain wood, but wood is not a major material in it.

The functional unit of wood and non-wood fuels is an equivalent amount of energy (MWh) delivered or service it provides (e.g. km driven by car).

Stage A2 (energy recovery) might be considered close to Stage A1 or independently in case of pure energy production.

Stage B) Substitution effect of the recycling and cascading of HWP1: HWP2 – DF_{M2} – non-wood product 2

In Figure 12, it is assumed that the carbon content of HWP1 is partly recycled after its use phase and a new wood product, HWP2, is produced. The share of amounts of HWP1 used for recycling or cascading should be determined. Recycling and re-manufacturing HWP1 would cause their own fossil-based emissions that should be taken into account. In Fig. 12, it is assumed that HWP2 replaces non-wood product 2 that is also recycled and remanufactured and causes fossil-based GHG emissions. It is assumed that HWP2 and non-wood product 2 have the same functionality. The substitution effect of stage B can be calculated similarly as in Stage A1 and A2, using Equations 16 and 17:

$$SE_{HWP2} = DF_{M2} * PA_B \quad (16)$$

, where DF_{M2} is expressed in fossil carbon emissions avoided per the used carbon content in HWP2 (t C/t C) related to the production stage and PA_B is the produced amount of HWP2, expressed in mass of carbon (t C).

DF_{M2} is determined:

$$DF_{M2} = \frac{GHG_{WP2} - GHG_{non-wood\ product\ 2}}{WU_{HWP2} - WU_{non-wood\ product\ 2}} \quad (17)$$

, where GHG_{HWP2} and $GHG_{non\text{-}wood\ product\ 2}$ are the *GHG* emissions resulting from the use of HWP2 and non-wood product 2, expressed in mass units of carbon (C). WU_{HWP2} and $WU_{non\text{-}wood\ product\ 2}$ are the amounts of wood included in HWP2 and non-wood product 2 expressed in mass units of *C* contained in the wood. Note that non-wood product 2 can also contain wood, but wood is not a major material in it.

When including recycling or cascading in carbon accounting for a specific product (here: HWP1), note that the products of the cascade (HWP2, final energy use) each have their own carbon footprints that need to be taken into account in the assessment. The comparison in LCA (ISO 14044) should be made using analogical methodologies regarding system boundaries, allocation procedure, functional equivalency and other methodological choices. The substitution credits must be taken into account using coherent methodologies. For example, if HWP1 is recycled as material in HWP2 and if the same holds true for non-wood materials 1 and 2, providing functional equivalency with HWP1 and 2, HWP2 does not replace a primary product but recycled non-wood product 1.

The above situation described is only one possibility that can happen after the use of HWP1. When HWP1 has reached its EoL, there are several options regarding the further life-cycle:

a. It may be disposed of without any energy recovery, which means direct end-of-life (EoL).
b. It may be used for its energy (single step cascade), which means a second life as fuel.

c. It may be recycled in closed loop (e.g. particle board to particles, which means decrease of/loss of quality), which means a second life as a secondary material product.

d. It may be recycled in an open loop (multiple cascade), which also means a second life as a secondary material product.

The first step is to identify the whereabouts of the used HWP1. The second step is to define the system boundary (within an LCA, actually, this has to be determined in the very first step of goal and scope definition).

Here, it must be decided whether and how the second life should be included in the accounting of HWP1. It is important to note that the primary product HWP1 and a recycled product HWP2 generated from it are both distinct products in their own right and each have their individual carbon footprint. If emissions and savings from recycling are taken into account for HWP1, they may no longer be considered in the accounting for HWP2. It is a basic rule of LCA that mass consistency is maintained, respectively, that double counting is avoided.

The options are:

- Cut-off: the life cycles of HWP1 and HWP2 are separated from each other. Both get neither benefits nor burdens from the other system.
- Allocation by 50:50: both systems share the loads associated with raw material acquisition. Each system carries 50%.
- The supposed single stage cascading: possible, but undetermined recycling stages of a used HWP1 are disregarded, but not the final energy use.

The cut-off-option could be used for sensitivity analysis. It should not be the basic approach since it completely ignores any connection between HWP1 and subsequent life cycles.

The **50:50 approach** is recommended in cases where the subsequent life cycles are well-known. It is still a strongly simplified and therefore practical approach. It means the attribution of 50% of the primary raw material for HWP1 to the production system of HWP2, however the substitution effect of HWP2 (replacing NWP2) is completely attributed to the product system of HWP2. The formula for the effect of the recycling and cascading via this option is therefore (in line with ISO 14067, Annex D):

$$SE_{RC} = \mathrm{GHG_V} - R \cdot A \cdot \mathrm{GHG_V} \qquad (18)$$

, where:

SE_{RC} is the substitution effect (GHG emissions) tied to raw material acquisition in case of material recycling of HWP1;

$\mathrm{GHG_V}$ is the GHG emissions tied to extracting or producing all the raw material needed for the product from natural resources;

R is the recycling rate;

A is the allocation factor – in this case determined to be 50%.

The approach ***supposes single stage cascading*** and is recommended in case the subsequent life cycles are not apparent or deliberately kept out of consideration (goal and scope definition). It encloses the final energy use (disregarding possible intermediate recycling cascades) as part of the product system of HWP1 and adds a further functional unit (the amount of energy produced, see following paragraphs).

Stage C1) Substitution effect caused by the final energy recovery: wood energy product 2 – DF_{E2} – non-wood energy product 2

In this stage, the carbon content of HWP1 is partly used after its primary use for the energy production purpose through wood energy product 2. The share of amounts of HWP1 used for energy as wood energy product 2 without material recycling or cascading use should be determined. The life cycle emissions of wood energy product 2 from HWP1 in Eq. 13 only cover the fossil-based emissions caused by the transportation and preparation of the fuel. Furthermore, energy production occurs in the future when HWP1 will have reached its end-of-life situation. The time frame strongly depends on the product and its use. The timing for energy recovery should be considered and it can be assessed with the help of market information or the assumed lifetime of HWP1 (see Section 3.4). There is also a need to assess the content of an alternative energy product 2 in the future year and determine its life cycle emissions in order to calculate the substitution effect of Stage C1 using Equations 19 and 20:

$$SE_{WE2} = DF_{E2} * PA_{C1} \tag{19}$$

, where DF_{E2} is expressed in the fossil carbon emissions avoided per the used carbon content in wood energy product 1 (t C/t C) related to the production stage and PA_{C1} is the produced amount of wood energy product 2, expressed in mass of carbon (t C).
DF_{E2} is determined:

$$DF_{E1} = \frac{GHG_{WE2} - GHG_{non-wood\ energy\ product\ 2}}{WU_{WE2} - WU_{non-wood\ energy\ product\ 2}} \tag{20}$$

, where GHG_{WE2} and $GHG_{non\text{-}wood\ energy\ product\ 2}$ are the GHG emissions resulting from the use of wood energy for product 2 and non-wood energy product 2, expressed in mass units of carbon (C). WU_{WE2} and $WU_{non\text{-}wood\ energy\ product\ 2}$ are the amounts of wood included in wood energy product 2 and non-wood energy product 2 expressed in mass units of C contained in the wood. Note that non-wood energy product 2 can also contain wood, but wood is not a major material in it.

Stage C2) Substitution effect of the final energy recovery: HWP2 -wood-based energy – DF_{E3} – non-wood energy product 3

In Fig. 12, it is assumed that the carbon content of HWP2 is used as energy at the EoL and the energy replaces non-wood energy product 2. Furthermore, a functional unit for both energy products is the equivalent amount of energy delivered (MWh) or service it provides (e.g. km driven by car). The preparation of wood-based energy from HWP2 may cause its own fossil-based emissions that should be taken into account. Furthermore, the fossil-based GHG emissions of manufacturing non-wood energy product 2 should be accounted to calculate the substitution effect of Stage C2 using Equations 21 and 22:

$$SE_{WE3} = DF_{E3} * PA_{C2} \tag{21}$$

, where DF_{E3} is expressed in fossil carbon emissions avoided per used carbon content in wood energy product 1 (t C/t C) related to the production stage and PA_{C2} is the produced amount of wood energy product 3, expressed in mass of carbon (t C).

DF_{E3} is determined:

$$DF_{E1} = \frac{GHG_{WE3} - GHG_{non-wood\ energy\ product\ 3}}{WU_{WE3} - WU_{non-wood\ energy\ product\ 3}} \tag{22}$$

, where GHG_{WE3} and $GHG_{non\text{-}wood\ energy\ product\ 3}$ are the GHG emissions resulting from the use of wood energy product 3 and non-wood energy product 3, expressed in mass units of carbon (C). WU_{WE3} and $WU_{non\text{-}wood\ energy\ product\ 3}$ are the amounts of wood included in wood energy product 3 and non-wood energy product 3 expressed in mass units of C contained in wood. Note that non-wood energy product 3 can also contain wood, but wood is not a major material in it.

Notes related to the substitution assessment

One of the most critical choices in the calculation is how the GHG emissions of non-wood products are determined, i.e. what products HWP1 will replace in the market. In practice, HWP1 may replace several non-wood products and their replacement shares may vary in the market. For example, HWP1 can replace non-wood product1a and 1b (NWP1a and NWP1b) at the same time. However, the replacement degree between HWP1 and NWP1a could be, for example only 50% in the market, i.e. only half of the HWP1 amount will replace NWP1a. Furthermore, for example 40% of HWP1 amounts will replace any products in the market by a replacement degree between HWP1 and NWP1b. The market share information can only be obtained from specific market surveys. The results are always estimation of reality. If the appropriate information is available, DF_{M1} can be calculated in the example as follows:

$$DF_{M1} = 0{,}5 * DF_{HWP1\text{-}NWP1a} + 0{,}1 * DF_{HWP1\text{-}NWP1b} \quad (23)$$

Thus, the determination consists of two determinations of the sub-DFs. The sub-DFs should be determined according to Eq. 13 with the requirements presented above.

In the market, the function served by HWPs can consist of several HWPs. For example, this is a case in the context of buildings, where wood-based buildings (where wood is a main material) are typically compared to non-wood material buildings with the same functionalities. In practice, this assessment is carried out by inventing all material components with their life cycle emissions in both building types. Both wood-based and non-wood-based buildings typically consist of wood and non-wood materials but to varying degrees. The life cycle inventory data are obtained for example from available life cycle databases for construction or directly gathered from companies. Finally, all life cycle emissions are summed up in both buildings and the values are used in Eqs. 12 and 13. The same concerns the carbon content in both buildings. However, the data only represents material substitution in the construction stage. In case there are differences in the energy consumption of the buildings or requirements for renovations and maintenance between the wood-based and its alternative building, these factors should be taken into account in the assessment.

Fig. 12 is a simplification. It excludes the possible substitution effects of material use in the end-of-life stage. If the material use for wood waste can be identified, its potential substitution effect can be determined by applying the principles and equations mentioned above.

Considering temporary carbon storage and timing of GHG emissions

The carbon removed from the forest in harvests is accounted for as an emission in the reporting of GHGs of managed forests of countries. For this reason, the increase in carbon storage of HWPs is reported as a negative emission (so called carbon removal). Over a certain time horizon, some products may be burned (releasing CO_2), some may end up as waste (with or without CO_2 release), some may be recycled and some may remain as the original product. In cases where the carbon is released at some point or gradually, the carbon storage is temporary. The key question is, how temporary carbon storage or delayed emissions are considered and characterized. In the following, some options that could be applied to consider and characterize temporary carbon storage or delayed emissions are shown, but none of them are suggested as the default.

The calculation of the climate effect related to temporary carbon storage requires an understanding of the fate of the products over time and selection of the climate metrics to describe the effect based on the goals and scope of a study. As presented in Section 6, there is no unique metric or time horizon to choose to characterize the climate effects of temporary carbon storage and delayed emissions, and the choice may significantly influence the results.

As non-CO_2 GHGs are converted to carbon dioxide equivalents typically using their cumulative radiative forcing over 100 years, the same basis may be justified for temporary carbon storage and delayed emissions in an LCA. According to Helin et al. (2016), temporary carbon storage of 5, 10, 20, 40, 60, or 80-year results in roughly -5%, -9%, -18%, -34%, -52%, and -72% lower cumulative radiative forcing over 100 years compared with the immediate release of the same amount of carbon dioxide (Fig. 13). Such information can be used to characterize the temporary carbon storage in LCA. For example, a 40-year storage time of 100 kg CO_2 is equivalent to an immediate emission of 66 kg CO_2 [100 kg CO_2 * (1–0.34)].

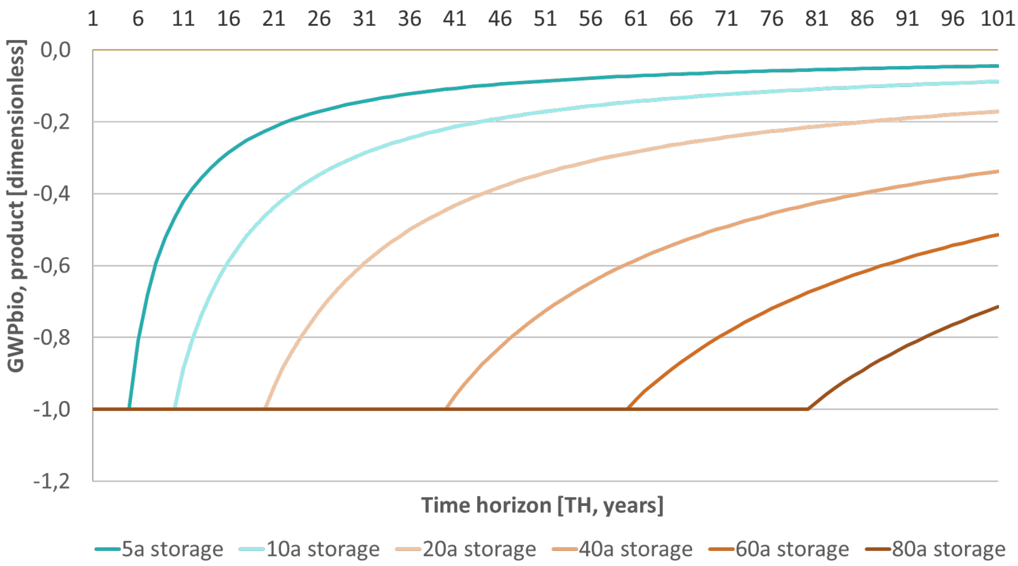

Figure 13. Time dependent weighting factor based on cumulative radiative forcing and expressed as global warming potential (GWPbio) for various (5, 10, 20, 40, 60 and 80 a) temporary storage times of carbon in harvested wood products (HWPs) (redrawn from Helin et al. 2016).

The substitution effects described in stages A–C are expressed over the 100 years because the 100-year GWP (Global Warming Potential) as the climate metric is typically applied in LCA studies to determine the relative contribution of different GHG emissions. The substitution effects of cases A and B take place at the beginning of the time horizon of 100 years. For this reason, they are consistent with the carbon storage change calculation mentioned above. Thus, the results of cases A to B and the carbon storage changes can be summed up.

In the stages of B, C1, and C2, the negative emissions of substitutions occur at different points of time compared to stage A. In Section 6, it was pointed out that these substitution effects occurring at different points in time cannot be directly compared with each other because their climate effect depends on the metrics and time horizon applied similarly than with temporary carbon stock in HWPs. In case it is chosen that 100-year time horizon is applied and cumulative radiative forcing is applied as a metric, avoided emission occurring at some later point should have a lower cumulative radiative forcing than immediately avoided emission. Based on the information provided by Helin et al. (2016), it can be concluded that avoided emissions after 5, 10, 20, 40, 60, or 80-year results in 95%, 91%, 82%, 66%, 48%, or 28% of cumulative radiative forcing over 100-years compared to that of immediately avoided CO_2 emissions. However, the typical practice in LCA is that the fossil-based emissions released in different years should be summed up as the timing of emissions is not considered. This approach provides significantly different results for the emissions avoided for stages B, C1, and C2 in Fig. 12 than the previously mentioned one derived from Helin et al. (2016). Note that the exclusion of the timing of emissions also means that there is no value for the temporary carbon storage of HWPs. Basically, analogical metrics and a time horizon should be applied for both fossil and biomass-based carbon. In an illustrative example provided for sawn wood in Appendix B, the weight of the GHG emissions generated and those avoided were not discounted (i.e. weight of the emissions 100% regardless of the timing of the emissions).

7.3 Assessment on an annual basis at the company level

Combined effects of HWP carbon storage changes and substitution

A starting point here is that the annual information on GHG emissions caused or avoided by production of HWPs and wood energy is relevant for companies producing several products at the same time. When accounting for GHG balances on an annual basis, there is no need to characterize the GHG emissions occurring at different points of time. Similarly, CO_2 sequestration to and emissions from HWPs may be accounted at annual basis based on the assumptions used for carbon inflow and outflow from the HWP stock.

The combined effects on GHG emissions (TCB) of all HWPs and energy produced by a company (or companies) in a certain year *i* in the techno-sphere can be calculated as follows:

$$TCB(i) = TSE(i) + TCCS(i) \qquad (24)$$

, where TSE(i) is the annual substitution effect of all HWPs and wood-based energy produced in a year *i* (t CO_2-eq.) and TCCS (i) is the annual change in carbon stocks of HWPs produced by the forest industry (expressed as t CO_2).

TSE(i) can be assumed to consist of three components as follows

$$TSE(i) = TSE_P(i) + TSE_{RC}(i) + TSE_{EOL}(i) \qquad (25)$$

, where

$TSE_P(i)$ = the annual substitution effect of all HWPs due to their production and primary use

$TSE_{RC}(i)$ = the annual substitution effect of all new HWPs due to the recycling and cascading of the old HWPs

$TSE_{EOL}(i)$ = the annual substitution effect of all HWPs (incl. new ones due to recycling and cascading) in their end-of-life.

The substitution effects caused by the recycling and cascading of the old (earlier produced) HWPs (TSE_{RC}) can be done with the help of the principles mentioned in Section 2.2. Due to its complexity, it is impossible to express according to one equation.

When old and new HWPs produced due to recycling and cascading are utilized in the end-of-life stage, $TSE_{EOL}(i)$ can be calculated according to

$TSE_{EOL}(i) = TSE_{EOL\_Material} + TSE_{EOL\_Energy}(i)$ (26)

, where

$TSE_{EOL\_Material}(i)$ = the annual substitution effect of material use from the end-of-life of HWPs

$TSE_{EOL\_Energy}(i)$ = the annual substitution effect of final energy recovery caused by the end-of-life of HWPs

It is important to notice that $TSE_{EOL\_Material}(i)$ and $TSE_{EOL_}$Energy(i) will occur at different points of time. The same concerns $TSE_{RC}(i)$ which is very complicated to assess. Due to its complexity, it is impossible to be expressed using one equation.

Furthermore, the annual change in carbon stocks of HWPs produced (TCCS (i)) can be divided to

$TCCS(i) = TCCS_{P\_HWPs}(i) + TSCC_{EC\_HWPS}(i)$ (27)

, where

$TCCS_{P\_HWPS}(i)$ = the annual change (increase or decrease) in carbon stock of HWPs caused by their production and primary use

$TCCS_{RC\_HWPS}(i)$ = the annual change (increase or decrease) in carbon stock of new HWPs caused by the recycling and cascading of the old HWPs.

Aggregation of substitution effects

Substitution effect of all HWPs and wood-based energy due to their production and primary use (TSE_P)

The total substitution effects of HWPs and wood-based energy produced in a certain year *i* are calculated by summing up the fossil-based GHG emissions avoided due to the production and use of the particular HWPs and wood-based energy in year *i*. Thus, TSE_P in Eq. 25 in a certain year *i* can be calculated as follows:

$$TSE_P(i) = \sum_1^n (DF_{Pj} * PA_j(i) + \cdots + DF_{Pn} * PA_n(i)) \quad (28)$$

, where DF_{Pj} is the displacement factor of a HWP or wood-based energy *j* (t C/t C) related to the production *P* stage and $PA_j(i)$ is the annual produced amount of HWP or wood-based energy *j,* expressed as the mass of carbon (t C).

Substitution effect of HWPs due to recycling and cascading (TSE_{RC})

This is a complicated area to assess. In practice, the results of recycling and cascading are missing even in the scientific literature. In principle, the assessment can be made following the principles mentioned in Section 7.1. It is important to note that the substitution effects of recycling and cascading in year *i* are caused by HWPs produced in years before *i.*

Substitution effect of wood energy from the end-of-life of HWPs ($TSE_{EOL\_Material}$ and TSE_{EOL_Energy})

TSE_{EOL_Energy}(i) is assessed from the amounts of HWPs that have reached their end-of-life in year *i.* The amounts originate from the HWPs produced years before year i and they can be assessed for year i on the basis of a carbon storage model called "HWP in use" (see the next section). The same concerns

$TSE_{EOL\_Material}$. To assess the substitution effects, the carbon amounts are multiplied by the corresponding DFs.

Changes in carbon stocks of HWPs

Increase or decrease in carbon stock of HWPs caused by their production and primary use ($TCCS_{P\_HWPS}$)

When carbon stocks of HWPs increase, the carbon is accounted as removal (negative emission). In the opposite situation, HWPs are accounted as carbon emissions.

The general method to estimate the magnitude of the defined carbon (*C*) stock in the HWP pool in use and its net changes involve the so-called “HWP in use” method (IPCC 2019, see Section X). The stock is calculated as follows:

$$C_l(i+1) = e^{-k} * C_l(i) + \left[\frac{(1-e^{-k})}{k}\right] * Inflow_l(i) \tag{29}$$

, where I = year, $C_l(i)$ = the carbon stock in the particular HWP commodity class *l* at the beginning of the year *i* (t C), k = the decay constant (= ln(2)/HL, where HL is the half-life of a particular HWP commodity in the HWP pool in years (see Section X, table X)), $Inflow_l(i)$ = the carbon inflow to the particular HWP commodity class *l* during the year *i* (t C yr^{-1}).

To apply Eq. 29 at a factory level for the forest industry, the initial stocks of all HWPs in a year *i*-1 are zero before the beginning year of the factory. The annual inflows of HWPs correspond to the annual production amounts of HWPs. The amounts should be expressed in t C units. Each HWP *j* is classified to its corresponding HWP commodity class *l* to obtain its half-life number.

The carbon stock increase/decrease of HWP is simply calculated as the annual carbon change in the stock of each HWP $_j$:

$$\Delta Cj\ (i) = Cj\ (i+1) - Cj\ (i) \tag{30}$$

, where I = year, $C_l(i)$ = the carbon stock in the particular HWP commodity class *l* at the beginning of the year *i* (t C), k = the decay constant (= ln(2)/HL, where HL is the half-life of a particular HWP commodity in the HWP pool in years), $Inflow_l(i)$ = the carbon inflow to the particular HWP commodity class *l* during the year *i* (t C yr^{-1}).

Finally, the annual change in the carbon stocks of the HWPs in each year *i* (expressed as t CO_2) is calculated as follows:

$$TCCS(i) = \frac{44}{12} * \sum_1^n (\Delta C_j(i) \tag{31}$$

For example, assuming the annual sawn wood production of a factory is 1000 t C/a (with the half-life (HL) of 35 years for HWPs), the increase in HWP carbon stock (positive values) over time will be developed as follows:

Year	C (1000 t CO_2/a)
1	1633
2	1600
3	1568

4	1536
5	1505
…	
10	1359
…	
30	903
…	
50	600

Carbon stock changes of HWPs caused by recycling and cascading ($TSCC_{RC\_HWPS}$)

Recycling HWPs as materials lengthen the lifetime of products, thus also the temporary carbon storage time. However, the effect is case-specific and depends on not only the type and quality of recycled product but also the way it is used. Improved recycling and cascading of HWPs should lengthen the half-lives of carbon, so when it is an adequately common practice, the effect should be taken into account in the half-life values applied. However, it may take time that adequate and reliable data become available and that the recycling and cascading use of wood may have influenced the HWP stocks so that the default half-lives could be revised accordingly.

8. Summary and conclusions

Forests and forest products contribute to climate change mitigation by sequestering carbon into forests, storing part of the carbon in HWPs and by avoiding fossil-based GHG emissions in substitutions for alternative materials and energy. Often, there are trade-offs in sequestering carbon into forests and harvesting trees for substitution, which means that these two strategies cannot be optimized at the same time. Which strategy is the most effective depends on a number of assumptions, including the time horizon, metrics to characterize the climate effects, the development of forest carbon stocks, the way harvested wood is processed and used, and alternative products to be substituted.

Assessing the climate effects of the use of wood, changes in carbon stocks in forests and HWPs and changes in fossil carbon emissions should be considered coherently. To do that, two different systems are compared to each other; namely, the one with the wood use under study and its reference system without the wood use being studied. In this report, the focus was on assessing carbon stock changes in HWPs and fossil emission substitution due to using HWPs and wood-based fuels in place of non-wood materials and fuels. The key knowledge and challenges encountered in the assessment were summarized and discussed, and some practical guidelines to carry out the assessment were provided.

An LCA is a suitable method to assess GHG balances related to wood use. Two main modeling principles have been developed; namely, attributional and consequential LCA. Both methods are suitable to study GHG balances related to use of wood and its alternatives but from different perspectives. Attributional LCA can be applied to respond questions such as "*What is the difference in GHG emissions attributable to equivalent energy or material service produced from wood or from fossil-based raw materials?*" On the other hand, consequential LCA is applicable to respond questions such as "*What are the consequences on GHG emissions of a decision to increase the use of wood?*" This inherently covers market mechanisms such as substitution. In other words, attributional LCA is applied to study GHG balances of product systems while consequential LCA is applied to study GHG balances of decisions related to product systems. In this report, we handled the carbon storage of HWPs and the substitution of using wood in place of alternative raw materials mainly from the attributional perspective. In such a case, the substitution is generated as the difference in GHG emissions between the wood-based system and its alternative system, both serving equivalent functions.

The carbon storage time of HWPs varies significantly depending on the product in question and the way it is used. Additionally, possible recycling as materials at the EoL influences the overall storage time. There are two options concerning how carbon storage times are handled in an assessment: 1) assuming the storage time and release of carbon after that, and 2) assuming an average decay rate of carbon. The first option is more relevant at the single product level, while the latter is more relevant when a number of similar products are assumed to be produced and used at the same time. In particular, for future-oriented assessments, uncertain assumptions must be used for the storage time or decay rate of carbon.

Fossil emissions avoided through substitution are generated as the difference in fossil emissions between wood use and its alternative system. These may depend strongly on the wood product in question, its alternative to be replaced and many methodological assumptions required in the assessment to define the system boundaries, and to handle the end-of-life treatment of products and timing of emissions, among other aspects.

The cascading use of wood as materials and energy may increase the overall substitution benefits of wood in replacing fossil-based alternative materials and energy sources. Multiple products are created from one and the same raw material in temporal sequence, leading to the reduction of raw material intensity and extension of the carbon storage time through repeated material use. On the other hand, recycling of used wood products requires energy and causes emissions, which depending on the case, does not necessarily lead to a reduced

GHG balance compared to the first primary production system or the product system substituted by the recycled product. Moreover, it is important to note that recycling and/or energy recovery is also possible for many alternative products such as plastics. Consequently, the benefits of cascading wood use should be assessed coherently compared to alternative materials and they are case-specific. Concerning the GHG balance for a particular product, a crucial point is that cascading use means expanding the system boundaries to include other separate products. It is therefore critical to credit the effect of recycling on the GHG balance of the initial product. This requires a consistent allocation rule between the product systems within the cascade. Three possible approaches are proposed in this study.

The GHG balances related to carbon storage of HWPs and fossil GHG emissions avoided in material and energy substitution are dynamic and evolve. Once assumptions have been made, they can be assessed on an annual basis in dynamic GHG inventory. Such information may be relevant for certain reporting purposes. However, additional methodological challenges are encountered when aiming to make temporary carbon storage and timing of GHG emissions equivalent to immediate GHG emissions. In LCA the time dimension is typically excluded, which means that temporary carbon storage and delayed emissions are not assigned any credits/debits compared to immediate GHG emissions. However, in practice the timing of emissions matters, but the related climate effects depend on what climate metrics are applied and over which time horizon the effects are assessed. Regardless of the chosen metrics and time horizon, the temporary carbon storage of HWPs and delayed fossil emissions (or delayed avoided emissions) should be considered analogically.

The challenges encountered in assessing the carbon storage of HWPs, cascading use of wood and substitution of wood for alternative raw materials are typical to those found in LCA in general. These are related to finding and determining reliable data on the product lifetime, end-of-life treatment and life cycle emissions, identification of functionally equivalent alternatives to be replaced by wood, finding reliable and analogically determined life cycle data for alternatives, as well as handling the timing of emissions and associated climate effects. These include the definition of spatial system boundaries, choosing allocation procedure and definition of temporal scope and characterization factors (climate metrics). Choosing appropriate methods are fundamentally subject to the goal and scope definitions of a study, and it is impossible to define specific choices, which would be uniquely superior to others. However, there are always more and less coherent choices given the defined goal and scope.

The practical guidelines provided in this report can help conduct a GHG balance assessment of the temporary carbon storage of HWPs, cascading use of wood and substitution of wood for alternatives. However, the practical guidelines are in many parts a simplification and should not be interpreted as a unique set of methodological choices. Finally, the exclusion of forest carbon issues in this report should not be interpreted so that they should be excluded in the GHG balance assessment of wood use. The handling of the carbon storage of wood and substitution effects are tightly connected to the wood harvested from forests. Consequently, the effect the wood harvest being studied has on forest carbon storage should be assessed analogically compared to its reference system, i.e. ‘non-wood system’ or ‘without wood-use system’.

Lexicon

ALCA	attributional life cycle assessment
aSET	absolute metric that refers to the contribution to a global mean temperature peak without time dimension
B2B	Business to business
BtL	Biomass-to-liquid
CA ARB	California Air Resources Board
CCS	carbon capture and storage
CCU	carbon capture and usage
CFP	carbon footprint of products
CFPP	California Forest Project Protocol
CLCA	consequential life cycle assessment
CO_2	carbon dioxide
DF	Displacement factor
FAOSTAT	Food and Agriculture Organization of the United Nations' Statistics
FOD	first-order decay
FPP	Forest Project Protocol
FR	fraction of carbon
EOL	end-of-life
GHG	greenhouse gas
GTP	global temperature potentials
GWP	Global Warming Potential
HL	half-life
HWP	Harvested wood product
IPCC	Intergovernmental Panel on Climate Change
LCA	life cycle assessment
MDF	medium-density fiberboard
Mg	megagram
MWh	megawatt hour
NWP	non-wood product
OSB	oriented strand board
PE	Polyethylene
PET	polyethylene terephthalate
PLA	polylactic acid
PP	polypropylene
PtX	Power-to-X
PVC	Polyvinyl chloride
SCAD	stock-change approach of domestic origin
UNFCCC	United Framework Convention on Climate Change
USD	United States dollar

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Appendices

Appendix A: Certain factors for harvested wood products

Table A1. Literature-based carbon conversion factors, minimum, maximum and average lifespan as well as minimum, maximum and average half-life values for the selected final paper and paperboard products.

					Values for the final products						
					Life span			Half life			Average decay rate
Product	Intermediate product or customer product*	Final products	Carbon conver-sion factor	Unit of C conver-sion factor	Min	Max	Average	Min	Max	Average	k
Chemical softwood pulp	Cardboard	Packaging material	0.3987	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
	Tissue paper	Tissue products (toilet paper, diapers)	0.45	Mg C/Mg	1	1.25	1	1	1	1	0.6931
	Graphic paper	Copy papers, magazines, leaflets, advertising material, books	0.3051	Mg C/Mg	1	20	5.72	1	6	2.78	0.2493
Chemical hardwood pulp	Cardboard	Packaging material	0.459	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
	Specialty papers	Packaging material, labels	0.45	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
	Graphic paper	Copy papers, magazines, leaflets, advertising material, books	0.3051	Mg C/Mg	1	20	2.57	1	6	2.78	0.2493
Deinked pulp (internal use)	Graphic paper	Newspapers, magazines, leaflets, advertising material	0.4105	Mg C/Mg	1	13	4.25	1	4	2	0.3466
Mechanical pulp (internal use)	Graphic paper	Newspapers, magazines, leaflets, advertising material	0.3052	Mg C/Mg	1	13	4.25	1	4	2	0.3466

Paper-based labels	Self-adhesive label stock	Food packaging labels, wine and spirit bottle labels, home and personal care product labels, pharmaceutical product labels, cosmetics product labels, security (brand protection) labels	0.3436	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
Face paper	Labeling	Labels	0.3436	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
Base paper (including release liner)	Labeling, backing for tapes	Labels	0.3436	Mg C/Mg	1	5	2.6	1	4	1.6	0.4332
Pack paper	Food packaging	Bags, wrappers, food packaging material	0.3027	Mg C/Mg	1	1.25	1	1	1	1	0.6931
Barrier paper	Food packaging	Food packaging material, food wrappers	0.3027	Mg C/Mg	1	1.25	1	1	1	1	0.6931
Fine paper, China		Office and home printing papers	0.3068	Mg C/Mg	3	20	9.8	3	6	5.33	0.13
Fine paper, Europe		Office and home printing papers, books	0.318	Mg C/Mg	3	20	9.8	3	6	5.33	0.13
Newsprint		Direct marketing material, advertising material, newspapers	0.4091	Mg C/Mg	1	13	4.25	1	4	2	0.3466
Magazine paper		Direct marketing material, advertising material, magazines	0.318	Mg C/Mg	1	13	4.25	1	4	2	0.3466

Table A2. Literature-based carbon conversion factors, minimum, maximum and average lifespan as well as minimum, maximum and average half-life values for the selected final sawn wood products.

					Values for the final products						
					Life span			Half-life			Average decay rate
Product	Intermediate product or customer product*	Final products	Carbon conver-sion factor	Unit of C conver-sion factor	Min	Max	Average	Min	Max	Average	k
Sawn timber		Sawn wood for construction, packaging, furniture, planing, joinery	0.229	Mg C/m^3	10	80	36.82	10	45	27.63	0.0251
		Construction (houses and buildings)	0.229	Mg C/m^3	30	330	175	34.7	127.5	71.08	0.0098
		Other construction materials	0.229	Mg C/m^3	15	80	50	10	67	28	0.0248
		Packaging and pallets	0.229	Mg C/m^3	2	20	8	1.39	3.43	2.07	0.3349
		Furniture	0.229	Mg C/m^3	11	43	24.8	8.3	38.8	16.32	0.0425
		Joinery	0.229	Mg C/m^3	20	36	28	16	36	23.77	0.0292
Veneer	Veneer	Parquet flooring	0.253	Mg C/m^3	20	36	28	15	30	22.5	0.0308
Coated plywood		Construction, bus floors, truck cargo compartment floors, LNG ship building	0.267	Mg C/m^3	30	99	49.8	30	50	38.75	0.0179
Uncoated plywood		Construction, bus floors, truck cargo compartment floors, LNG ship building	0.267	Mg C/m^3	30	99	49.8	30	50	38.75	0.0179
		Plywood molds	0.267	Mg C/m^3	2	20	8	1.39	3.43	2.1	0.3301

Table A3. Literature-based carbon conversion factors, minimum, maximum and average lifespan as well as minimum, maximum and average half-life values for the selected final products.

					Values for the final products						
					Life span			Half-life			Average decay rate
Product	Intermediate product or customer product*	Final products	Carbon conver-sion factor	Unit of C conver-sion factor	Min	Max	Average	Min	Max	Average	k
Renewable diesel (2nd generation biodiesel)	Diesel for transport*		0.693	Mg C/Mg	1	1	1	1	1	1	0.6931
Renewable naphtha	Naphtha for transport*		0.626	Mg C/Mg	1	1	1	1	1	1	0.6931
Renewable naphtha	Ethlylene	Plastic surface in paper board beverage carton	0.626	Mg C/Mg	1	1.25	1	1	1	1	0.6931
Renewable naphtha	Propylene	Labels e.g Raflatac forest film, Formi EcoAce	0.626	Mg C/Mg	1	20	5.72	1	6	2.78	0.2493
Renewable naphtha	Ethlylene	Floor materials etc.	0.626	Mg C/Mg	20	36	28	15	30	20.8	0.0333
Mono-ethylene glycol (MEG)	PET preforms, granulates	Bottles	0.417	Mg C/Mg	1	1	1	1	1	1	0.6931
Renewable functional filler (RFF)	Rubber*	Hoses and profiles	0.417	Mg C/Mg			15			4	0.1733
		Tires	0.417	Mg C/Mg			6			2	0.3466
Purified kraft lignin	Phenolic resins*	Wood panels e.g. plywood, OSB and phenolic surface films	0.265	Mg C/Mg	2	80	49.8	1	50	38.75	0.0179
Biocomposites	Granulates for injection moulding, ForMi*	Furniture, home and electric appliances (e.g. cutlery), 3D printing	0.078 - 0.232	Mg C/Mg			25			16	0.0433
Biocomposites	ProFi	Decking	0.1514	Mg C/Mg			25			16	0.0433

Appendix B: Calculation examples

Example of a product-level analysis for a sawn wood product (for illustrative purposes only)

Stage A1

Description

The primary product is a sawn wood product used in construction. The amount of sawn wood product used is 2 t C. The sawn wood product is used in buildings in which it replaces a functionally equivalent amount of alternative products (concrete, steel, etc.). Sawn wood is used in both wooden building and concrete building. The displacement factor is determined as a difference between the GHG emissions in the wooden building (0.1 t C) and concrete building (0.45 t C) divided by the difference between the sawn wood used in wooden building (0.5 t C) and concrete building (0.05 t C), equaling -0.78 t C / t C. The avoided emissions in using wood in place of alternative materials are -1.56 t C (a negative value means a positive substitution effect) and they take place at the beginning of the life cycle and they are not discounted (weight 100%).

Calculation

Stage A1		unit
PA_A11	2	t C
GHG_HWP1	0,1	t C
GHG_non-wood_product1	0,45	t C
WU_HWP1	0,5	t C
WU_non-wood_product1	0,05	t C
DF_M1	-0,77778	t C/t C
Weight of SE_HWP1 (based on timing of emissions)	100 %	
SE_HWP1	-1,55556	t C

Stage A2:

Description

Sawmill residues not used as internal processing energy for sawn wood are used to produce heat and power. The amount of sawmill residues available for this purpose are 1.8 t C. This is used in place of fossil energy. The displacement factor is determined as the difference between the fossil GHG emissions associated with wood-based energy (0.1 t C) and the fossil GHG emissions associated with fossil-based energy (0.9 t C) divided by the difference in the wood used for the wood energy product (1 t C) and for the fossil energy product (0 t C), equaling -0.8 t C / t C. The biomass-based emissions from wood combustion are accounted for as zero. The avoided emissions are -1.44 t C (a negative value means a positive substitution effect) and they take place at the beginning of the life cycle and they are not discounted (weight 100%).

Calculation

Stage A2		
PA_A2	1,8	t C
GHG_WE1	0,1	t C
GHG_non-wood_energy_product_1	0,9	t C
WU_WE1	1	t C

WU_non-wood_energy_product_1	0	t C
DF_E1	-0,8	t C/t C
Weight of SE_WE1 (based on timing of emissions)	100 %	
SE_WE1	-1,44	t C

Stage B

Description

50% of the sawn wood used in construction in stage 1 is recycled as a product (HWP_2). The recycled product replaces an alternative non-wood product. Fossil fuels are used to generate the process energy required to produce the products. The displacement factor is determined as the difference between the fossil GHG emissions of processing HWP_2 (0.1 t C) and processing non-wood_product_2 (0.5 t C) divided by the difference between wood content of HWP2 (1 t C) and non-wood_product_2 (0 t C), equaling -0.4 t C / t C. The avoided emissions are -0.4 t C (a negative value means a positive substitution effect) and they take place at the point of decommissioning the building but are not discounted here (weight 100%).

Calculation

Stage B		
PA_B	1	t C
GHG_HWP2	0,1	t C
GHG_non-wood_product_2	0,5	t C
WU_HWP2	1	t C
WU_non-wood_product_2	0	t C
DF_M2	-0,4	t C/t C
Weight of SE_HWP2 (based on timing of emissions)	100 %	
SE_HWP2	-0,4	t C

Stage C1

Description

50% of the sawn wood used in construction is used to process liquid biofuel, which replaces the fossil alternative fuel. 50% (0.5 t C) of the sawn wood coming into the process (1 t C) which is used as process energy. Additionally, fossil fuels are required to process the fuels. The displacement factor is determined as the difference between the fossil GHG emissions of the wood-based energy (0.1 t C) and alternative fossil energy (0.8 t C) divided by the difference between the wood content of wood-based energy (1 t C) and that of the alternative fuel (0 t C), equaling -0.7 t C / t C. The avoided emissions are -0.35 t C (a negative value means a positive substitution effect) and they occur at the point of decommissioning the building but are not discounted here (weight 100%).

Calculation

Stage C1		
PA_C1	0,5	t C
GHG_WE2	0,1	t C
GHG_non-wood_energy_product_2	0,8	t C
WU_WE2	1	t C

WU_non-wood_energy_product_2	0	t C
DF_E2	-0,7	t C/t C
Weight of SE_WE2 (based on timing of emissions)	100 %	
SE_WE2	-0,35	t C

Stage C2

Description

The recycled product HWP2 is used at the end of its life to process liquid biofuel, which replaces fossil alternative fuel. 50% (0.5 t C) of the wood coming into the process (1 t C) is used as process energy. Additionally, fossil fuels are required to process the fuels. The displacement factor is determined as the difference between the fossil GHG emissions for the wood-based energy (0.1 t C) and the alternative fossil energy (0.7 t C) divided by the difference between the wood content of the wood-based energy (1 t C) and that of the alternative fuel (0 t C), equaling -0.6 t C / t C. The avoided emissions are -0.3 t C (a negative value means a positive substitution effect) and they occur at the point of EoL of HWP2 but are not discounted here (weight 100%).

Calculation

Stage C2		
PA_C2	0,5	t C
GHG_WE3	0,1	t C
GHG_non-wood_energy_product_3	0,7	t C
WU_WE3	1	t C
WU_non-wood_energy_product_3	0	t C
DF_E3	-0,6	t C/t C
Weight of SE_WE3 (based on timing of emissions)	100 %	
SE_WE3	-0,3	t C

Total SE (Stages A1, A2, B, C1, C2)

Description

The total GHG emissions avoided in material and energy substitution is -4.04 t C (negative value means positive substitution effect).

Calculation

TOTAL SE (Stages A1, A2, B, C1, C2)	-4,04556	t C

Example of a company-level analysis (for illustrative purposes only)

A company is annually producing 0.5 Mt C of sawn wood, 0.1 Mt C of wood-based panels, 2 Mt C of pulp and 0.2 Mt C in energy products. The half-life of carbon is 35 years for sawn wood, 25 years for wood-based panels and 2 years for pulp. The carbon content of the energy products is released within the production year. The displacement factors are 0.93 for the material use of sawn wood and wood-based panels, 0.25 for the material use of pulp and 0.70 for the energy use of wood. At the EoL of the HWPs they are fully used as energy to substitute fossil fuels with a displacement factor of 0.70. No material recycling of HWPs takes place.

The annual substitution effect of all the HWPs and wood-based energy produced in a year i [TSE(i)] is presented in Fig. B1. The annual change in carbon stocks of HWPs produced annually by the company [TCCS (i)] is presented in Fig B2. The combined effects on GHG emissions (TCB) of all HWPs and energy produced by the company in a certain year i in the techno-sphere is presented in Fig. B3.

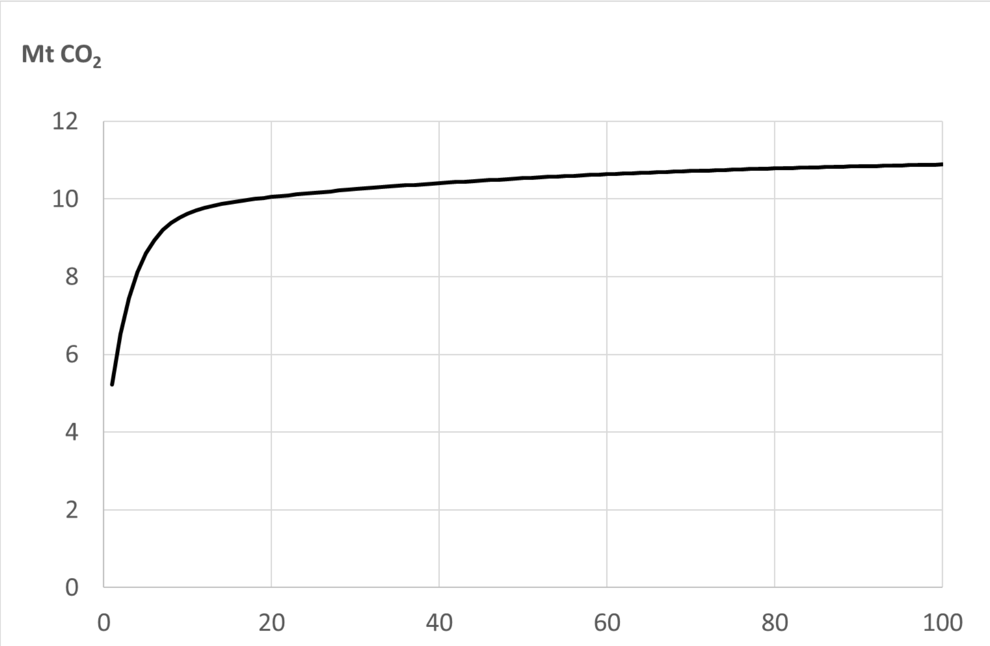

Figure B1. The annual substitution effect of all HWPs and wood-based energy produced in year i [TSE(i)] as a function of time (years).

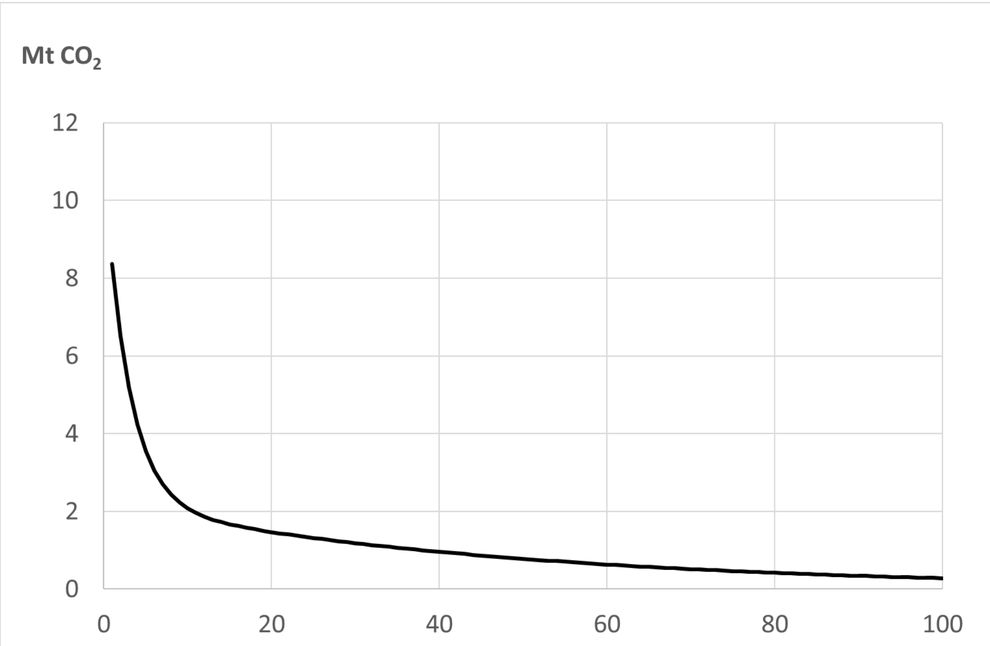

Figure B2. The annual change in carbon stocks of HWPs produced annually by the company [TCCS (i)] as a function of time (years).

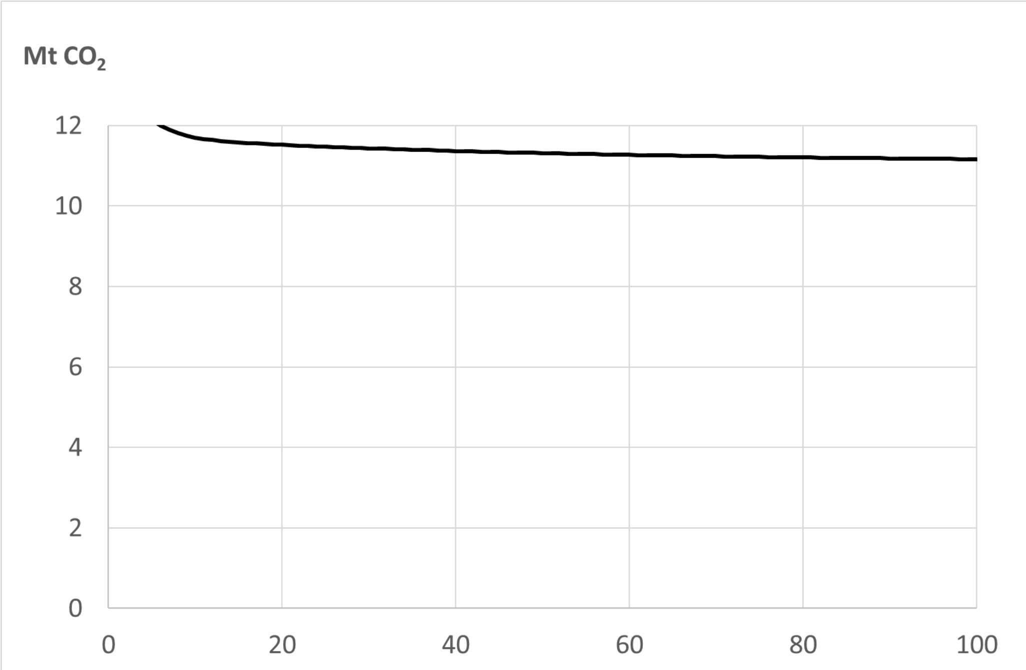

Figure B3. The combined effects on GHG emissions (TCB) of all HWPs and energy produced by a company (or companies) in a certain year i in the techno-sphere as a function of time (years).

Fossil carbon emission substitution and carbon storage effects of wood-based products

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