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ORIGINAL RESEARCH

Time dynamic climate impacts of a eucalyptus pulp product: Life cycle assessment including biogenic carbon and substitution effects

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Abstract

The forest sector can play a pivotal role in mitigating climate warming by decreasing emissions to the atmosphere and increasing carbon removals. In an expanding bioeconomy, the pulp and paper industry provides opportunities for various low-carbon wood products with promising substitution effects. However, assessing climate effects of wood product systems is complex and requires a holistic approach. The objective of this study was to advance time dynamic climate impact assessment of a bioeconomically promising wood product from a system perspective. For this purpose, a time dynamic life cycle assessment was conducted on a pulp-based beverage carton. The assessment included fossil value chain emissions from cradle to grave, effects from biogenic carbon in a eucalyptus plantation, and credits from substitution. A polyethylene terephthalate (PET) bottle was considered for material substitution (MS) and differing marginal electricity and heat mixes for energy substitution. The results revealed dominating climate warming from value chain emissions and slight offsetting by biogenic carbon from standing biomass and soil organic carbon, and short-term carbon storage in the beverage carton. MS and displacing marginal energy mixes transformed the climate warming into a substantial total cooling effect. However, substitution effects varied strongly in terms of substitution factors and temperature change with varying replacement rate of the beverage carton and different marginal energy mixes. A climate cooling range of $-0.8 \cdot 10^{-15}$ to $-1.8 \cdot 10^{-15}$ K per unit of beverage carton by 2050 was found, highlighting potential relevance for climate policy making. Thus, production and use of wood-based beverage cartons over PET bottles can have climate cooling effects. Further assessments on alternative forestry systems (e.g., Nordic forests) are needed to identify the role of biogenic carbon in holistic climate assessments, with dynamic substitution effects included to increase the validity.

KEYWORDS
beverage carton, biogenic carbon, climate impact, LCA, pulp, substitution effect, temperature change, time dynamic, wood product

1 | INTRODUCTION

The urgently needed mitigation of global greenhouse gas (GHG) emissions requires efforts to reduce emissions to the atmosphere and increase carbon removals (IPCC, 2014). In efforts to limit climate warming below 1.5°C (UNFCCC, 2015), harvested wood products (HWPs) from the forest sector can play a pivotal role within the framework of climate-smart forestry (Nabuurs et al., 2018).

Forests sequester carbon dioxide (CO_2) through photosynthesis and store it as biogenic carbon in biomass and soil organic carbon (SOC). This biogenic carbon from biomass is retained in HWPs, capturing CO_2 from the atmosphere. In addition, production of HWPs generally releases less GHG emissions than processing functionally equivalent non-wood products, especially when the wood is sourced from sustainable forestry (Geng et al., 2017). Thus, wood products can have substitution effects through avoided GHG emissions by displacing non-wood materials (Leskinen et al., 2018).

The European Union (EU) strives for climate neutrality by 2050 (EC, 2020a) and acknowledges the climate mitigation role of the forest sector and HWPs in its bioeconomy strategy, where abating climate change is linked to "[…] a renewed bio-based industrial base reducing energy demand and lowering emissions" (EC, 2018). In this context, "[one] pioneer in making the EU low-carbon bioeconomy an industrial reality" is stated to be the pulp and paper industry (EC, 2020b). In addition, the raw material pulpwood can be the source of a variety of new HWPs with promising market potential in an expanding bioeconomy (Hurmekoski et al., 2018).

However, European demand for pulp is increasingly being met by imports from South American eucalyptus pulpwood production (FAO, 2019; González-Goméz, 2019; Judl et al., 2011), which supplied 72% of the EU's pulp imports in 2011 (Indufor, 2013). This trend can bear the danger of increased GHG emissions, for example, due to related land use change (LUC), if native forest is converted to eucalyptus plantations (Bernstad Saraiva et al., 2017). Consequently, a shift in emissions, that is, "leakage effects" could occur, contravening the EU's ambition for reducing energy demand and GHG emissions in a bioeconomy (Harmon, 2019; Leskinen et al., 2018). Due to their large volumes, pulp products in form of packaging, textile (e.g., viscose), or chemical applications are considered to have a substantial GHG mitigation effect, via displacement of emission-intensive materials, such as plastic packaging, or synthetic fiber used for clothing (Leskinen et al., 2018). However, information about actual GHG substitution effects is lacking, especially for packaging products such as pulp, because of the great variety of alternative materials (Leskinen et al., 2018).

One pulp product with large production outputs and a potentially promising substitution effect is the beverage carton. In a meta-study of 20 life cycle assessments (LCAs), Falkenstein et al. (2010) found that the beverage carton was usually attributed the lowest climate impact among functionally equivalent products such as glass or polyethylene terephthalate (PET) bottles. This has been confirmed by a later review on beverage cartons by O'Sullivan et al. (2016) and a comparative LCA on beverage carton usage set in Northern Europe (Markwardt et al., 2017).

Life cycle assessment (ISO, 2006a, 2006b) is an established methodology for analyzing climate effects of wood utilization. As reviewed by Markwardt et al. (2017) and Falkenstein et al. (2010), various LCAs have investigated the climate effects of the forest sector (Klein et al., 2015), wood utilization for energy (Wolf et al., 2016), material application (Sahoo et al., 2019) or Eucalyptus cultivation in form of a time-dependent approach (Porsö et al., 2016). However, these LCAs often leave a research gap by omitting either system-holistic or time dynamic climate impact assessment, for which the term "system perspective" is used in the following. Filling this research gap is essential to understand the wider implications of HWP use in policy making on the climate (Suter et al., 2017).

A system perspective can integrate five major aspects, by: (i) including climate effects of biogenic carbon which are still mostly considered neutral (Røyne et al., 2016), that is, carbon sequestration via photosynthesis equals the eventual carbon emission along the life cycle (Head et al., 2019); (ii) accounting for substitution (energy substitution [ES], material substitution [MS]) effects of the wood use (Garcia et al., 2020), which is associated with great uncertainties, especially for emerging HWPs (Leskinen et al., 2018); (iii) cascading use of wood, which occurs when "wood is processed into a product and this product is used at least once more either for material or energy purpose" (EC, 2016a), making cascading a potential means to improve the climate performance of a HWP system

(Thonemann & Schumann, 2018); (iv) including sufficient sensitivity analysis for the HWP system assessed, for example, in terms of changing substituted future marginal energy mixes (Hammar & Levihn, 2020); and (v) applying climate metrics appropriate for accounting for time dynamic effects of GHG emissions and sequestrations (Helin et al., 2013; Levasseur et al., 2010), to compensate for the shortcomings of commonly used static climate metrics such as global warming potential (GWP) in terms of time-dependent accounting (Breton et al., 2018). Examples of such metrics are the GWP_{bio} (Cherubini et al., 2011), the time-dependent radiative forcing (RF) (Sathre & Gustavsson, 2012), or the absolute global temperature change potential (AGTP) (Myhre et al., 2013).

The intention of the present study was to apply a system perspective in assessment of the climate effects of wood material application, closing the existing research gap. A system perspective was applied in a case study of a UHT milk beverage carton (hereafter "beverage carton") on the Northern European market produced from South American eucalyptus pulpwood. Therewith, the objective of the study was to conduct a time dynamic climate impact assessment of a bioeconomically promising HWP including biogenic carbon stocks and fluxes, and substitution effects from energy and material displacement, to advance the understanding of climate effects from wood product systems.

2 | MATERIALS AND METHODS

2.1 | Scope

The system boundary of the beverage carton life cycle assessed in this study contained the following three components:

Biogenic carbon stocks and fluxes, including standing aboveground and belowground biomass, and also SOC and carbon storage in the HWP (pulpwood, pulp, beverage carton). A theoretical landscape perspective was modeled (Cintas et al., 2016; Eliasson et al., 2013) for the eucalyptus plantation and grassland was considered as a land-use reference system (Koponen et al., 2018) to account for potential climate impact mitigation from replacing a non-plantation benchmark (Peñaloza et al., 2019).

Fossil value chain emissions of the beverage carton, which were accounted for from cradle to grave and geographically divided into two parts: Uruguay, where emissions from the eucalyptus plantation and pulpwood processing were assumed to occur, and Sweden, where emissions from finishing pulp to beverage carton and the end-of-life stage (incineration) were set. In between both parts, emissions from shipping were included.

Substitution effects of material and energy displacement. A full-barrier PET bottle, including its life cycle from cradle to grave and its potential ES, was considered, since plastic products have the second greatest market share among food packaging materials, after paper and board (Muncke, 2020). Recycling of the PET bottle and a corresponding replacement rate (R) was also accounted for, based on Hammar et al. (2020). ES was matched to the geographical region where the energy was generated, that is, electricity produced from the Uruguayan pulp mill substituted marginal Uruguayan electricity mixes, while energy produced from waste incineration in Sweden displaced marginal Swedish energy mixes. Thus, cascading use was considered by using the materials at the end of life for energy recovery. No use phase was accounted for, because it was assumed that differences in usage, and thus differences in emissions, were negligible.

A dynamic life cycle inventory including annual inputs and outputs was applied for all flows and processes since it improves accuracy of life cycle impact results and thus the entire LCA outcome (Lueddeckens et al., 2020). Two functional units and time horizons were set. The first functional unit covered a time horizon of 100 years of beverage carton production and was based on one hectare of eucalyptus plantation, to enable comparisons with the climate metric GWP_{100} and thus other studies. The second functional unit covered a time horizon of 50 years of beverage carton production and was based on one unit of beverage carton. This was to highlight potential relevance for climate policy making. Climate impact allocation between the products and by-products was avoided, and system expansion including biogenic carbon and substitution effects was applied. On the plantation, harvest residues (leaves, branches, stump, roots, bark) were assumed to be left in the field, and thus to act as input to SOC. The multi-output process in the pulp mill resulted in the product, pulp, and various by-products. The burdens from these by-products were included in form of ES via exported electricity replacing marginal mixes in Uruguay.

Sensitivity analyses were carried out to test the effect of assumptions regarding the substitution effects, one on differing material replacement rates and one on changing the displaced marginal energy mixes considering the functional unit of one beverage carton.

2.2 | Biogenic carbon

Biogenic carbon stocks and fluxes occurred along the entire life cycle (Figure 1). The biogenic carbon from standing biomass and SOC was calculated from a theoretical landscape perspective (Cintas et al., 2016; Eliasson et al., 2013). In the plantations, all standing aboveground

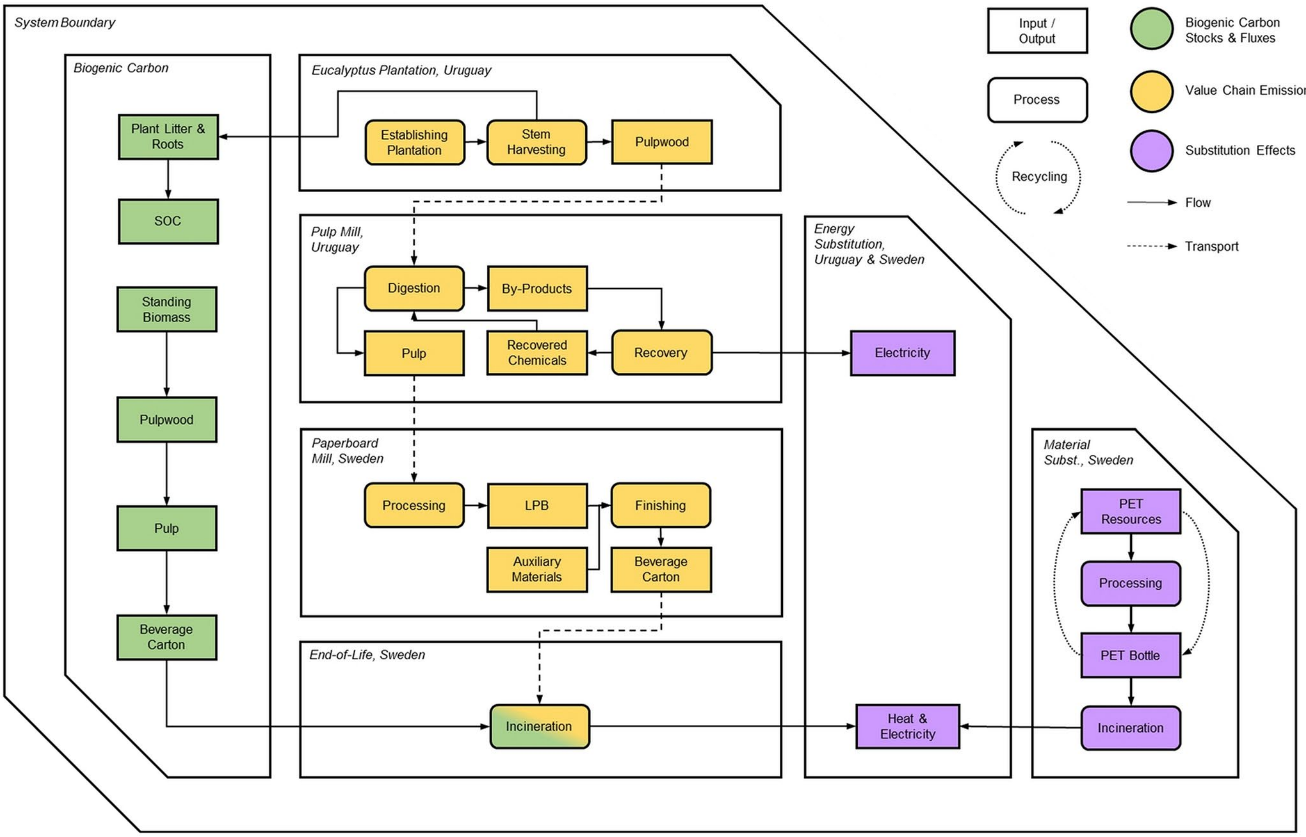

FIGURE 1 System boundary of beverage carton sourced from South American eucalyptus pulp, encompassing biogenic carbon stocks and fluxes, value chain emissions, and substitution effects. LPB, liquid packaging board; PET, polyethylene terephthalate; SOC, soil organic carbon

biomass except for the harvested stems was assumed to be left on the ground to decompose and to function as input to SOC. The resulting annual SOC fluxes were calculated using the dynamic soil carbon model Yasso 15 (Hammar et al., 2020; Järvenpää et al., 2018). An SOC value of 44.7 Mg C ha^{-1} for the former land use category grassland was applied, based on FAO and CMCC (2017), and acted as the reference case. The rotation period of each plantation was set to 9 years and average harvest yield to 76.8 Mg pulpwood ha^{-1}, with a biogenic carbon content of 38.4 Mg ha^{-1} and assuming a moisture content of 50% (Giraldo & Hyman, 1996). Biogenic carbon storage and decay within the HWPs (pulp and beverage carton) were modeled based on Rüter et al. (2019), considering a half-life of 1.9 years.

2.3 | Value chain emissions

2.3.1 | Eucalyptus plantation

Value chain emissions included operations on the eucalyptus plantation (Figure 1), and were modeled according to Gabrielle et al. (2013), with modifications, and based on site-specific data for a Uruguayan cultivation system. Field operations occurred at two different points in time in the rotation. In the first year, they included soil preparation in the form of plowing, mechanical planting of seedlings, fertilizer production, and fertilizer application. In the last year of the rotation cycle, harvesting was performed with a one-grip harvester, followed by ditch cleaning and, finally, field transport of the pulpwood logs to the road. All processes were assumed to be fueled by diesel, with emissions from diesel production and consumption based on Giuntoli et al. (2015). Fertilizer application was set to 110 kg N ha^{-1}, 33 kg P ha^{-1}, and 96 kg K ha^{-1} per rotation period, adapted from Timander (2011). For all fertilizers and forms of biomass litter, direct and indirect N_2O emissions due to, for example, nitrate leaching were included, based on IPCC (2019).

Subsequent lorry transportation (and associated emissions) was assumed to be with a EURO 5 truck. Average transportation distance was set at 180 km from the plantation to the pulp mill, based on site-specific location information. Capacity of the truck (45 Mg) was based on Simons (2019) and tare weight (20.5 Mg) was based on Trzciński et al. (2018). Emissions from diesel use were calculated based on Giuntoli et al. (2015).

2.3.2 | Pulp mill

Eucalyptus pulpwood was assumed to be processed into chemical pulp within modern pulp mill facilities (Figure 1). Like the majority of pulp making today, chemical pulping via the sulfate (or Kraft) process is the dominant practice in Uruguay (Kuparinen et al., 2019). All processes occurring in the pulp mill were divided into two parts: the digestion (pulp making) process and the by-product recovery process including production of surplus energy (Kuparinen et al., 2019). Both processes were modeled based on Corcelli et al. (2018) and primary data from the mill operator, while background data on auxiliary materials and energy-related emissions were taken from the ecoinvent database (Wernet et al., 2016). All by-products from pulp making (components of "black liquor") were assumed to be used for energy recovery applying a LHV of 12 MJ kg^{-1} (ECN, 2021) as it is common practice in modern pulp mills in Uruguay (Kuparinen et al., 2019; Montes del Plata SA, 2021). This surplus energy was redirected to the national grid in form of electricity where it was considered to cause substitution of marginal mixes, whose modeling was based on Hagberg et al. (2017) and MIEM (2019; Supporting Information S1). For the recovery boiler, an efficiency of 67% was assumed, based on Zhao et al. (2019).

The pulp produced was assumed to be shipped to Sweden for finishing into liquid packaging board, which was used in the final product, the beverage carton. Transport distances were modeled based on data from NTMCalc 4.0 (NTM, 2020), while GHG emissions were based on the ecoinvent database 3.6 (Wernet et al., 2016), in accordance with McKinnon and Piecyk (2010) and NTMCalc 4.0 (NTM, 2020). Transport distance amounted to 12,600 km from the pulp mill in Uruguay to the port in Sweden. Transport emissions were modeled using a well-to-wheel approach, that is, emissions from infrastructure were neglected and only those from diesel combustion were included. In Sweden, the pulp was transported by lorry from the port to the packaging board mill, located 260 km away. This transport was modeled similar to the lorry transport within Uruguay.

2.3.3 | Paperboard mill

At the packaging board mill, pulp was processed into liquid packaging board, from which the beverage carton was created. Energy requirements and auxiliary materials (i.e., aluminum, low-density polyethylene [LDPE], polypropylene [PP]) were adapted from Corcelli et al. (2018) and based on first-hand data from the mill operator. Finishing liquid packaging board for the beverage carton required adding LDPE and aluminum for the coating, and PP for the cap. Emissions data on these additives were taken from the ecoinvent database 3.6 (Wernet et al., 2016). A final transport stage of 100 km to the end-of-life stage was assumed, based on the average national transport distance for Sweden (Eurostat, 2020). For this, similar conditions as in previous lorry transportation were applied.

2.3.4 | End of life

The end-of-life stage represented incineration under average Swedish conditions. Processes such as collecting, sorting, and storing were omitted as they were considered to be similar for both beverage carton and PET bottle. A combined heat and power plant with heat efficiency of 45% and electricity efficiency of 30% was assumed for combusting the beverage carton and PET bottle, based on EC (2011). The energy content of the products and emissions from incineration was calculated based on the relative share of the component materials. Emissions data were taken from the ecoinvent database 3.6 (Wernet et al., 2016). The lower heating value (LHV) was set to 18.23 MJ kg^{-1} for the beverage carton and 25.1 MJ kg^{-1} for the PET bottle. The energy content of the components in the products (Swedish Standard Institute, 2004) was set to 12.6 MJ kg^{-1} for liquid packaging board, 22.0 MJ kg^{-1} for PET, 44.0 MJ kg^{-1} PP for the caps, 43.0 MJ kg^{-1} LDPE for the labels or coatings, and 27.0 MJ kg^{-1} for carbon black, which was used as a pigment for the PET bottle. The LHV for aluminum was excluded, since it was assumed not to deliver energy from its combustion.

2.4 | Substitution effects

2.4.1 | Energy and material substitution

Material substitution of the PET bottle was modeled to be representative for a European setting and comprised (Doka, 2013; EFBW, 2020; Fröhlich, 2017): resource sourcing for PET production, the PET production process and PET bottle making, recycling, and disposal by incineration. Between each stage, similar lorry transportation conditions as for the beverage carton were assumed.

Substitution of energy within the system boundary comprised three parts: (i) surplus energy from the Uruguayan pulp mill, which replaced marginal electricity mixes, whose modeling was based on Hagberg et al. (2017) and MIEM (2019); (ii) released heat and electricity from incineration of the beverage carton, which substituted a marginal Nordic heat and electricity mix with composition modeled based on Hagberg et al. (2017);

and (iii) forgone heat and electricity replacement from avoided end-of-life incineration of the substituted PET bottle, also based on the marginal Nordic heat and electricity mix. In addition, a sensitivity analysis was performed using different Uruguayan marginal electricity mixes and Swedish marginal heat and electricity mixes for the years 2020, 2030, and 2040, modeled based on Hagberg et al. (2017) and MIEM (2019). Detailed information on these marginal energy mixes can be found in Supporting Information S1.

2.4.2 | Substitution factors

Substitution factors (SFs) for replacing the PET bottle with the beverage carton were calculated based on Sathre and O'Connor (2010):

$$\mathrm{SF} = \frac{\mathrm{GHG}_{\mathrm{non\text{-}wood}} \cdot R - \mathrm{GHG}_{\mathrm{wood}}}{\mathrm{WU}_{\mathrm{wood}} - \mathrm{WU}_{\mathrm{non\text{-}wood}} \cdot R}, \quad (1)$$

where the SF is given in Mg C_{fossil} Mg^{-1} $C_{biogenic}$; $GHG_{non\text{-}wood}$ and GHG_{wood} denote the GHG emissions from production and incineration of the non-wood and wood product, respectively, expressed in mass units of carbon corresponding to the CO_2 equivalents (CO_2-eq) of the emissions; WU_{wood} and $WU_{non\text{-}wood}$ represent the amount of wood used in the wood and non-wood product, respectively, also given in mass units of carbon; and R is replacement rate (R), based on Hammar et al. (2020).

The terms $GHG_{non\text{-}wood} \cdot R$ and $WU_{non\text{-}wood} \cdot R$ in Equation (1) account for recycling the PET bottle into another one that fulfills the same function (Figure 1). As the baseline case, the European PET bottle recycling rate of 52% (EUNOMIA, 2020) was applied, and thus, $R = 0.48$. A sensitivity analysis was performed by applying the Swedish PET bottle recycling rate of 83% (SCB, 2019; $R = 0.17$) and by assuming no recycling of the PET bottle, meaning that one beverage carton replaced one PET bottle ($R = 1$). Overall, recycling and associated emissions were only assumed for the PET bottle because recycling of PET bottles into the same product is legitimate and common practice. Recycling was not assumed for the beverage carton, as virgin pulp fiber is mainly used in production to guarantee inert and safe food packaging conditions in accordance with EC (2016b).

To account for the entire value chain, the SF of the MS effect was calculated as:

$$\mathrm{SF}_{\mathrm{total}} = \mathrm{SF}_{\mathrm{P}} + \mathrm{SF}_{\mathrm{EoL}}, \quad (2)$$

where the total substitution factor SF_{total} is given in Mg C_{fossil} Mg^{-1} $C_{biogenic}$, SF_P is the substitution factor from the material production stage, and SF_{EoL} is the substitution factor comprising emissions from the end-of-life stage and ES. It follows that the larger the SF_{total}, the greater the assumed climate mitigation effect via replacement.

2.5 | Climate impact metrics

Climate effects can be calculated at different steps along the cause–effect chain from GHG emission to actual climate change and its consequences (Myhre et al., 2013). In this study, two different metrics, GWP and AGTP, were applied to assess the effect on the climate. GWP represents the cumulative RF of a GHG relative to the cumulative RF of CO_2 for a determined time frame (Joos et al., 2013). The time frame is often set for 100 years and the corresponding GWP_{100} is useful for comparisons with other studies. However, GWP does not consider the time dynamics of GHG emissions or accounts for varying points in time along the life cycle when GHG fluxes occur. Apart from CO_2, which stays airborne until it is partly taken up via plants or the oceans, the GWP_{100} includes methane (CH_4) and nitrous oxide (N_2O) emissions. For fossil CH_4, the GWP_{100} is 30-fold stronger than that of CO_2 (28-fold for biogenic carbon) with a perturbation lifetime of 12.4 years, while for N_2O, it is 265-fold stronger with a perturbation lifetime of 121 years (Myhre et al., 2013). In this study, no climate-carbon feedbacks were considered.

Absolute global temperature change potential, defined by Myhre et al. (2013), is synonymous with time dynamic temperature change or temperature change in this study. Compared with GWP, it is a climate metric one step further down the cause–effect chain from emissions to climate change and impacts. It therefore has greater policy relevance, but greater uncertainties are associated with its results. The AGTP value represents the response in global mean surface temperature at a given point in time induced by a change in RF due to a pulse emission of a GHG and is expressed in degrees Kelvin (K). The differing radiative efficiencies of the GHGs, which alter the balance of incoming, short-wave solar radiation, and outgoing, long-wave terrestrial radiation to varying degrees, and the differing perturbation lifetimes of the GHGs in the atmosphere are considered. In this study, AGTP was calculated based on GHG fluxes from CO_2, CH_4, and N_2O over a time horizon of 100 years and 50 years. The perturbation lifetime of CO_2 was modeled based on the Bern carbon cycle model (Joos et al., 2001, 2013), in which the molecule stays airborne until it is taken up by oceans or the biosphere. For CH_4 and N_2O, average perturbation lifetime was 12.4 and 121 years, respectively (Myhre et al., 2013). Indirect effects of ozone and water vapor on the radiative forcing of CH_4 were included in the climate model. The AGTP from each GHG considered is described by:

$$\mathrm{AGTP}_x(H) = \int_0^H \mathrm{RF}_x(t)\, R_T(H-t)\, dt, \quad (3)$$

where RF and the climate response function (R_T) form a convolution over the assessed time horizon (H) by a change in the RF from a pulse emission of a GHG x. Thus, AGTP accounts for the timing of GHG emissions and their perturbation lifetimes, enabling assessment of time-dependent climate effects.

3 | RESULTS

3.1 | Biogenic carbon dynamics of the eucalyptus plantation

Figure 2 shows the modeled time-dependent carbon stocks of the eucalyptus plantation from a stand perspective, divided into SOC, stems and bark, stumps, branches, leaves, and roots over two rotation cycles. The modeled biogenic carbon stocks of standing biomass accumulated throughout each rotation period, by annually decreasing incremental growth. Stems accounted for the largest share of biogenic carbon from standing biomass, followed by stumps, roots, branches, and leaves. At the start of each rotation, total biogenic carbon from standing biomass amounted to 11 Mg C ha^{-1}, while in the final year of the rotation cycle (year 9), it had increased to a total of 56 Mg C ha^{-1}. At the end of year 9, eucalyptus stems (excluding bark) were harvested, which amounted to 38 Mg C ha^{-1}, and the next rotation cycle began. Stocks of SOC showed an opposing pattern. During harvest years, stocks increased since fallen litter (e.g., leaves & roots) and harvest residues (bark and stumps) were added to the soil. Thereafter, the SOC stock decreased slightly from year to year until the rotation cycle reached the next harvest.

3.2 | Global warming potential

Figure 3 shows the GWP_{100} per unit of beverage carton from a landscape perspective. In total, one beverage carton amounted to 66.0 g CO_2-eq per package when excluding substitution (Figure 3a). Value chain emissions and incineration clearly dominated the climate impact, with most GHGs emitted by operations and energy requirements within the Uruguayan pulp mill (25%) and the Swedish packaging board mill (44%), and by combustion of LDPE and PP from the beverage carton (29%). By contrast, value chain emissions from the plantation and transport were minor. Moreover, biogenic carbon from standing biomass, HWPs, and SOC was marginal in the overall CO_2-eq balance of the beverage carton and barely offset the emissions. In total, the sequestration effect from biogenic carbon amounted to −2.0 g CO_2-eq per unit of beverage carton. When substitution effects were included in the GWP_{100} (Figure 3b), one beverage carton amounted to −83.7 g CO_2-eq. When forgone substitution credits from avoided PET bottle combustion and emissions of beverage carton incineration were subtracted from the ES credit, the offset still showed a moderate CO_2-eq saving. In addition, MS contributed substantially to offsetting fossil value chain

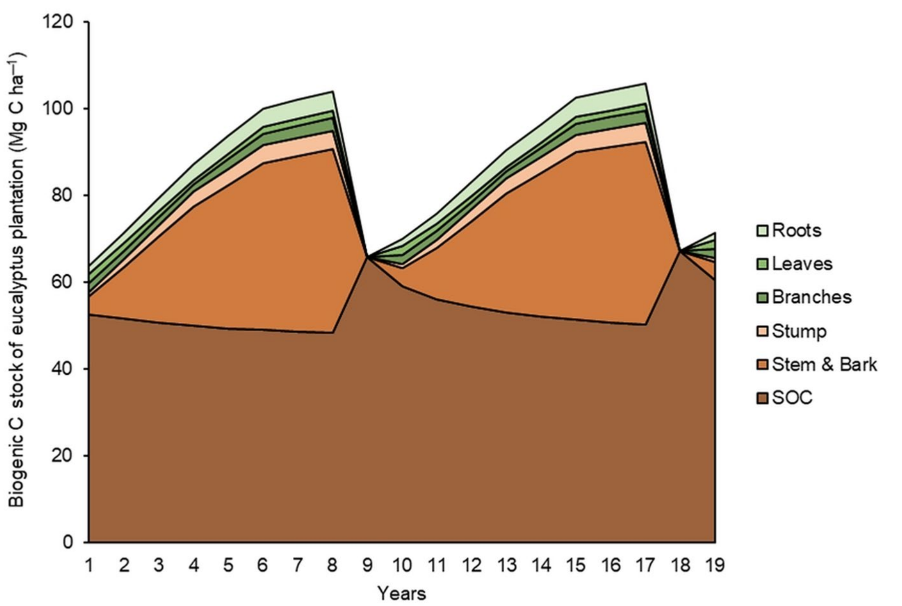

FIGURE 2 Stand perspective of the biogenic carbon stocks during the first two 9-year rotation periods in the modeled eucalyptus plantation. *Note*: Values represent state at the end of the year. SOC, soil organic carbon

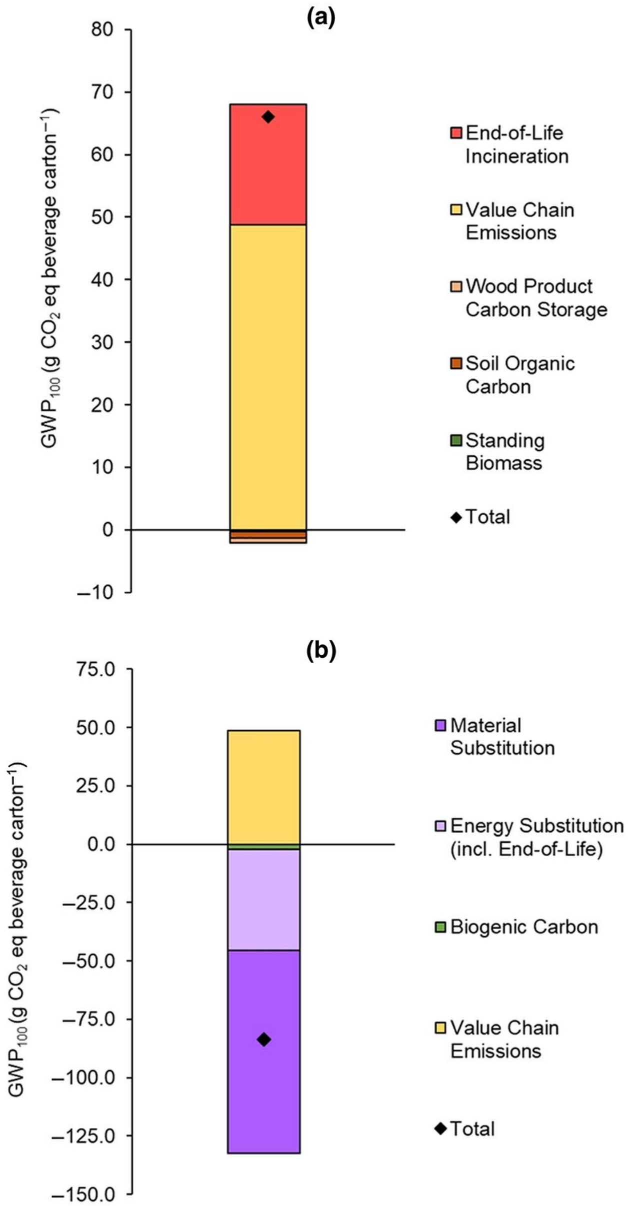

FIGURE 3 (a) GWP_{100} of a beverage carton excluding substitution effects, including value chain emissions, end-of-life emissions from incineration, and biogenic carbon fluxes from different sources. (b) GWP_{100} of a beverage carton including substitution effects, value chain emissions, and biogenic carbon fluxes. Substitution effects assume moderate PET bottle replacement ($R = 0.48$) and a fossil-intense marginal Uruguayan electricity and Swedish energy mix displacement (mix in 2020), according to Hagberg et al. (2017) and adapted from MIEM (2019). *Note*: End-of-life emissions from beverage carton incineration are included in energy substitution, which also considers forgone energy credits from avoided PET combustion. Figures show the results from a landscape perspective. GWP, global warming potential; PET, polyethylene terephthalate

emissions. Biogenic carbon effects continued to make only a minor contribution to total GWP_{100}.

TABLE 1 Substitution factors (SFs) for varying replacement rate (R) of a beverage carton sourced from South American eucalyptus pulpwood and produced and disposed of in Sweden. Total substitution factor (SF_{total}) is divided into the substitution factors caused by production (SF_P) and by the end-of-life stage including energy recovery (SF_{EoL}). Negative values indicate fossil carbon savings

	Substitution Factor (g C_{fossil} g^{-1} $C_{biogenic}$)		
Replacement rate (R)	**SF_P**	**SF_{EoL}**	**SF_{total}**
0.48	−0.1	−2.1	−2.2
1	−1.2	−1.2	−2.4
0.17	0.6	−2.7	−2.1

3.3 | Substitution factors including replacement rates

The total substitution factors (SF_{total}) of the beverage carton are presented in Table 1, where negative values indicate net fossil carbon removal. Moreover, the sensitivity of the SFs to varying replacement rates (R) of the PET bottle is shown. The baseline case, a moderate replacement rate ($R = 0.48$), resulted in a total substitution factor of −2.2 g C_{fossil} g^{-1} $C_{biogenic}$. This saving was almost entirely composed of SF_{EoL}, while SF_P made a minor contribution. The highest SF_{total} was found for one beverage carton replacing one PET bottle ($R = 1$) and amounted to −2.4 g C_{fossil} g^{-1} $C_{biogenic}$. The lowest SF_{total}, −2.1 g C_{fossil} g^{-1} $C_{biogenic}$, was obtained with low replacement ($R = 0.17$). In this case, SF_P was positive, 0.6 g C_{fossil} g^{-1} $C_{biogenic}$, but was outweighed by SF_{EoL} equaling −2.7 g C_{fossil} g^{-1} $C_{biogenic}$.

3.4 | Time dynamic temperature change

3.4.1 | Temperature change from biogenic carbon fluxes

Biogenic carbon fluxes decreased the atmospheric concentration of CO_2 in a landscape perspective over a time horizon of 100 years. This induced a negative temperature change, that is, climate cooling (Figure 4). The effect was the strongest during the first 30 years and leveled off in subsequent decades. In total, the climate cooling effect from SOC was five- to six-fold stronger than the effect from biogenic carbon in standing biomass. Moreover, temperature change caused by biogenic carbon fluxes from HWPs initially followed a similar trend as SOC, while later it had a moderately weaker cooling effect.

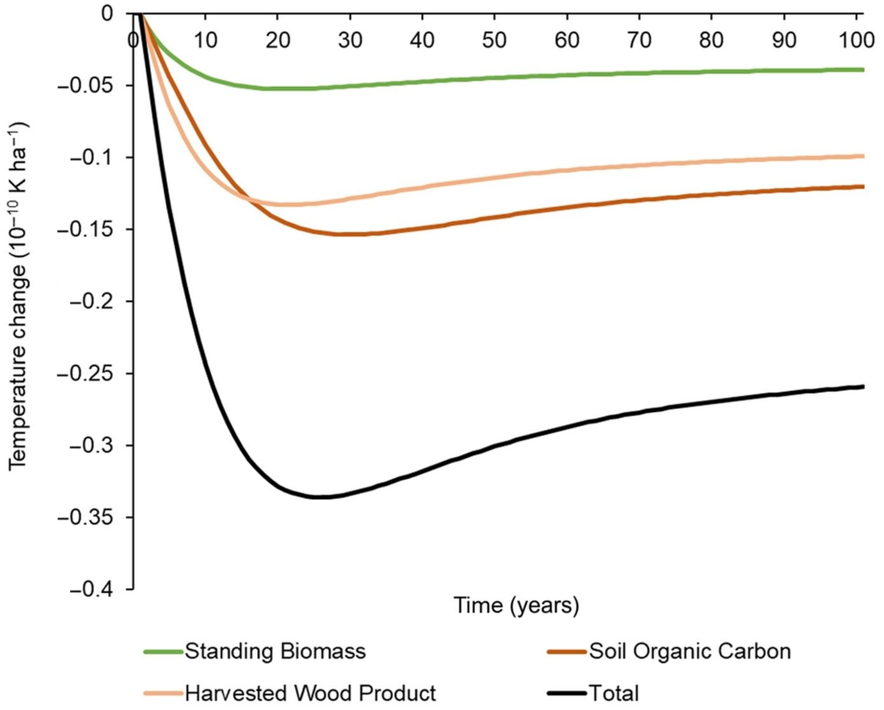

FIGURE 4 Temperature change from biogenic carbon fluxes of the modeled eucalyptus plantation's standing biomass, soil organic carbon, and harvested wood products, given per ha from a landscape perspective

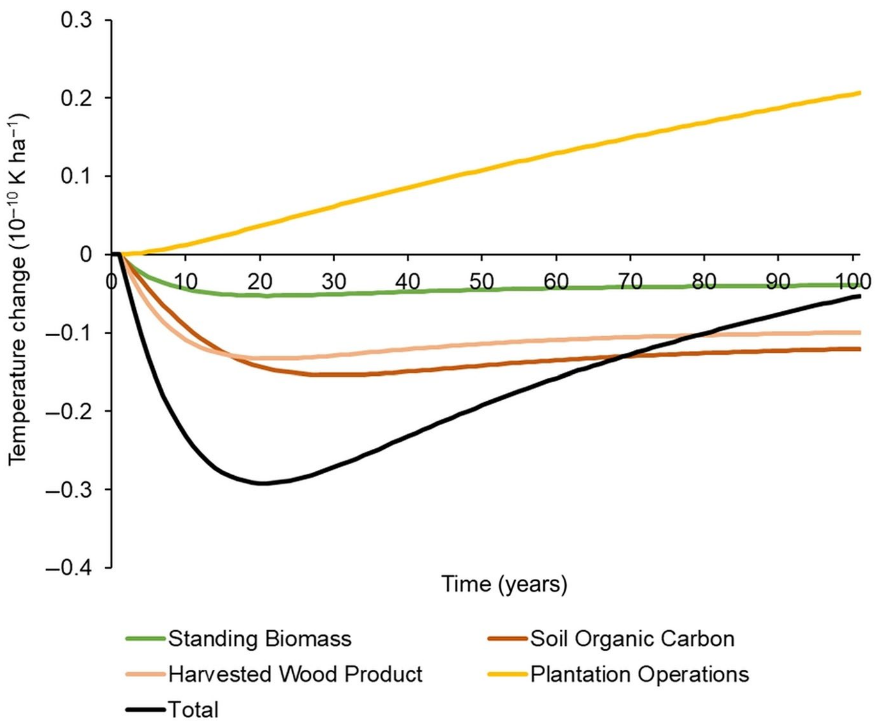

FIGURE 5 Temperature change from all forms of biogenic carbon fluxes and fossil emissions from operations on the plantation, given per ha from a landscape perspective

3.4.2 | Temperature change from biogenic carbon and value chain emissions

Figure 5 illustrates the temperature change from plantation operations and biogenic carbon per ha over a time horizon of 100 years. Irrespective of the continuous climate warming from the fossil emissions of the operations, the sum of biogenic carbon fluxes clearly outweighed this effect and transformed the total into climate cooling. This climate cooling was strongest in the first three decades and was then steadily offset by fossil emissions, but without becoming overall climate warming over the time horizon assessed.

Figure 6 shows the temperature change per unit of beverage carton over a time horizon of 50 years including emissions from the entire value chain, end-of-life incineration, and biogenic carbon. In general, the trend was similar to that obtained for GWP_{100} (see Figure 3a). The

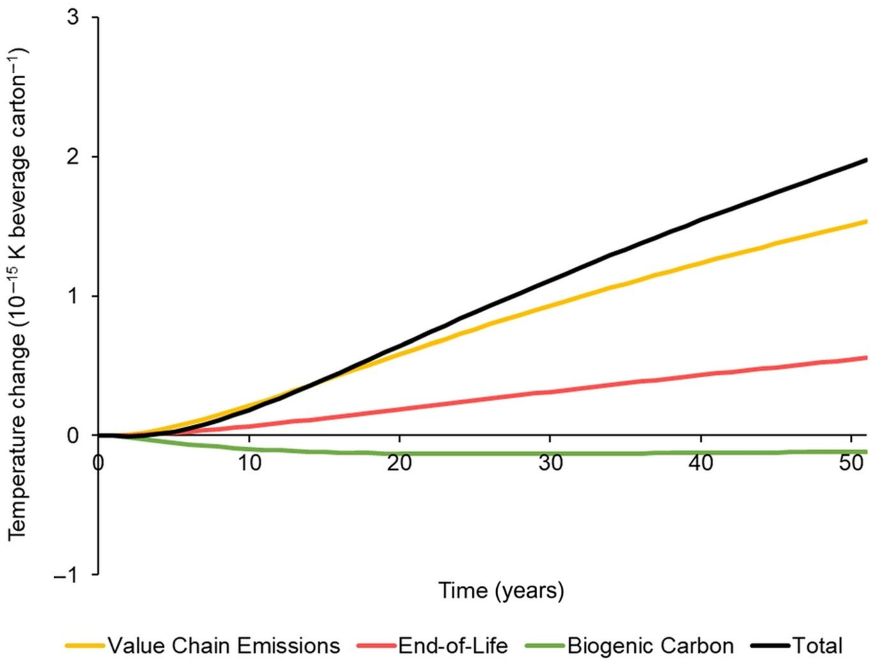

FIGURE 6 Temperature change from value chain emissions, end-of-life incineration, and all biogenic carbon fluxes, given per unit of beverage carton from a landscape perspective

climate cooling effect from biogenic carbon fluxes was negligible compared with the climate warming from fossil emissions from the value chain and end of life. However, in contrast to GWP_{100}, the temperature change showed steadily increasing climate warming over time. In the very first years of the time horizon assessed, climate warming was outweighed by the climate cooling from biogenic carbon fluxes. Overall, end-of-life emissions contributed around one-third, and residual value chain emissions two-thirds, to the total temperature change.

3.4.3 | Temperature change from a system perspective

The system perspective of one unit of beverage carton temperature change (Figure 7) included biogenic carbon, fossil value chain emissions, and energy and MS over a time horizon of 50 years. It showed a similar trend to GWP_{100} (see Figure 3b). From the start, climate cooling from energy and MS strongly offset climate warming from value chain emissions. Simultaneously, biogenic carbon fluxes only slightly influenced the overall climate effect. In total, a climate cooling effect was induced and continued to increase over time, mainly influenced by fossil value chain emissions and MS effects.

3.4.4 | Sensitivity analysis

Figures 8 and 9 present the results of sensitivity analyses for the product system of the beverage carton. Both consider a time horizon of 50 years from 2020 until 2070 and include the EU's climate neutrality target for 2050.

Figure 8 shows temperature changes of different MS, that is, three different replacement rates (R) of the beverage carton and its effects on the entire product system (Total), which is defined as in Figure 7. Irrespective of replacement rate considered, the climate cooling of the entire product system constantly increased over time. However, the magnitude of the cooling effect differed substantially depending on the replacement rate. Strong climate cooling was obtained with a high replacement rate ($R = 1$), whereas low replacement rate ($R = 0.17$) resulted in minor climate cooling over time. By 2050, a total temperature change of approximately $-1.8 \cdot 10^{-15}$ K beverage carton^{-1} was obtained for high replacement ($R = 1$), and $-1.3 \cdot 10^{-15}$ K beverage carton^{-1} for low replacement ($R = 0.17$). Thus, a temperature change difference of approximately $0.5 \cdot 10^{-15}$ K was obtained for the year 2050 by changing replacement rates per unit of beverage carton.

Figure 9 presents the temperature change of the entire product system (Total) for three different cases of ES based on marginal Uruguayan electricity and Swedish energy mixes. The climate cooling effect from ES clearly decreased from a fossil-intense mix (marginal mix for 2020) to a moderately fossil-based mix (marginal mix for 2030). The climate cooling effect became minor when a low fossil-based, that is, renewable-intense ES mix (marginal mix for 2040) was considered. From a system perspective, this meant that fossil-intense ES led to a contribution of approximately $-1.5 \cdot 10^{-15}$ K beverage carton^{-1} to the EU's climate neutrality target for the year 2050. In contrast,

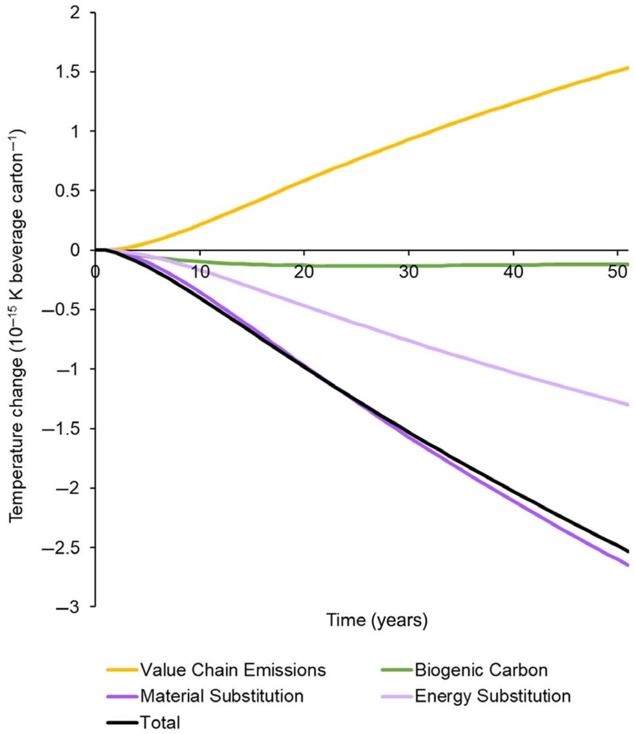

FIGURE 7 Temperature change per unit of beverage carton from a system perspective, including fossil value chain emissions, biogenic carbon fluxes, and substitution. Substitution effects assume a moderate PET bottle replacement rate ($R = 0.48$) and a fossil-intense marginal Uruguayan electricity and Swedish energy mix displacement (mix in 2020), according to Hagberg et al. (2017) and adapted from MIEM (2019). *Note*: End-of-life emissions from beverage carton incineration are included in energy substitution, which also considers forgone energy credits from avoided PET combustion. The results are shown from a landscape perspective. PET, polyethylene terephthalate

renewable-intense ES induced a temperature change of around $-0.7 \cdot 10^{-15}$ K beverage carton^{-1}. Thus, differences in ES, based on differing marginal energy mixes, induced a temperature change difference of about $0.8 \cdot 10^{-15}$ K beverage carton^{-1} by 2050.

Overall, the temperature change range caused by altered replacement rate of the beverage carton (Figure 8) was substantially larger than of changing substituted marginal energy mix (Figure 9).

4 | DISCUSSION

4.1 | Time dynamic climate effects including biogenic carbon and substitution

The climate assessment in the present study expanded on previous LCAs on wood products by including fossil value chain emissions with biogenic fluxes (Røyne et al., 2016) and substitution effects, assessed in a time dynamic approach (Breton et al., 2018). The study also provided new information on substitution factors, since pulp and paper or chemicals are often omitted in LCA studies, as stated by Soimakallio et al. (2016), and geographical regions such as South America are rarely considered, as mentioned by Leskinen et al. (2018). The climate effects found (Figures 3, 6, and 7) confirm findings in a review by O'Sullivan et al. (2016) and in LCAs by Falkenstein et al. (2010) and Markwardt et al. (2017) dealing with climate effects of beverage cartons.

Along all value chain emissions, industrial processing activities from the pulp and paperboard mill had the largest climate warming effect, mainly due to high-energy consumption and use of chemicals. At this life cycle stage, climate efficiency can be substantially improved by carbon capture and storage, for example, by tall oil manufacturing or lignin extraction (Kuparinen et al., 2019). The fossil emissions from plantation operations were minor, with harvesting being the most emitting operation. Climate effects from transportation were minor considering all value chain emissions, contradicting findings by Judl et al. (2011). However, the emissions ratio found for the value chain of the eucalyptus pulp product was similar to that reported by Silva et al. (2015), Corcelli et al. (2018), and Sun et al. (2018). A 1-year time gap was assumed between production and incineration including the half-life of the beverage carton, adapted from Rüter et al. (2019). This led to minor influence of biogenic carbon storage in the beverage carton on the time dynamic climate effects. Alternatively, a landfill scenario could have been assumed as an end-of-life alternative, but landfilling is not common practice in the geographical area of the study (Sweden). In other geographical areas, it may be important to consider a landfill scenario. However, this would probably increase the uncertainty in the results, as carbon dynamics are quite variable (Sathre & O'Connor, 2010) and as ES is not possible as long as methane collection is not applied.

Substitution effects from material displacement (Figure 8) and offsetting energy (Figure 9) were substantial contributors to the overall climate impact. In contrast, total biogenic carbon sequestration and retention from standing biomass and soil in the eucalyptus plantation, and in the HWP, barely offset the dominating climate warming effect from all fossil value chain emissions, including the end-of-life incineration (Figure 7). This contradicts findings by Markwardt et al. (2017) that biogenic carbon can have a "significant role in the impact category climate change" of a beverage carton's life cycle. However, the magnitude of temperature cooling by the modeled eucalyptus plantation was similarly strong to that found in an

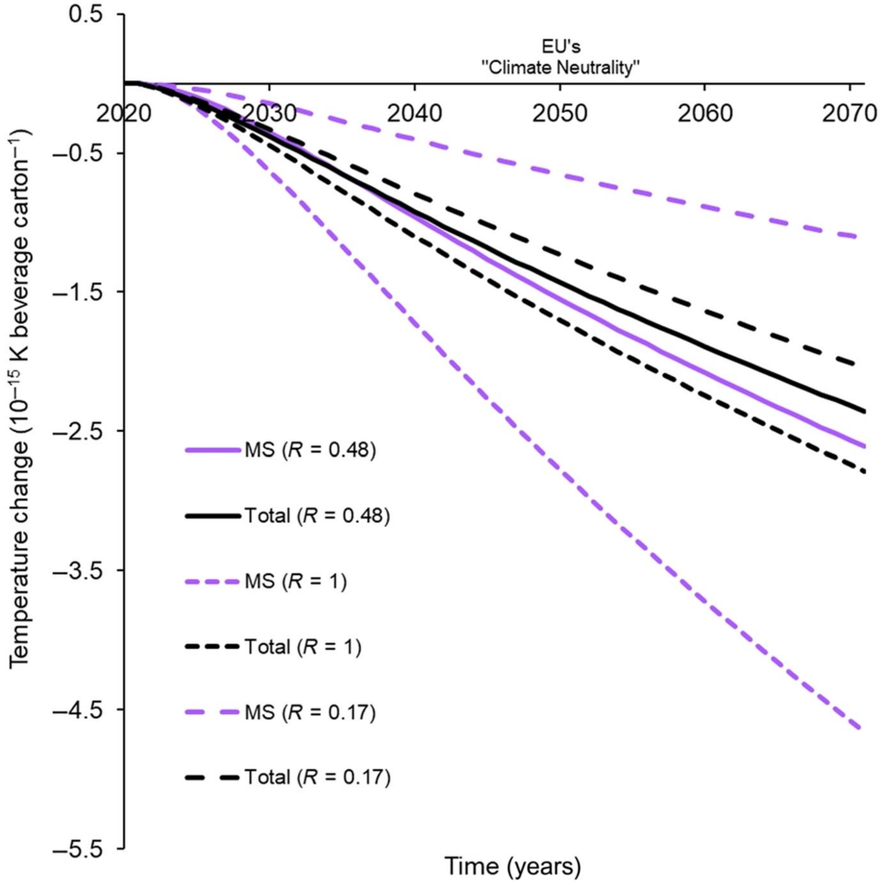

FIGURE 8 Temperature change in sensitivity analysis on changing the replacement rate (R), influencing material substitution (MS). The entire climate effect (Total) includes biogenic carbon, value chain emissions, and both energy and material substitution. The baseline represents moderate replacement ($R = 0.48$), while the other scenarios represent high ($R = 1$) and low ($R = 0.17$) replacement. *Note*: Energy substitution includes end-of-life emissions and forgone energy credit from avoided polyethylene terephthalate combustion. The results are shown from a landscape perspective

earlier time dynamic temperature change study of a eucalyptus product system (Porsö et al., 2016).

The calculated SFs accounted for production and the end-of-life stage. The range of values obtained was moderately higher to that reported by Soimakallio et al. (2016) for paperboard products (mean value −1.4 g C_{fossil} g^{-1} $C_{biogenic}$) and to the results from the meta-analysis by Leskinen et al. (2018) for packaging materials (−1.0 to −1.5 g C_{fossil} g^{-1} $C_{biogenic}$). However, Holmgren and Kolar (2019) obtained an even lower SF of −0.7 g C_{fossil} g^{-1} $C_{biogenic}$ for pulp and paper products, possibly because they applied a proxy value. In contrast, higher values found in this study are mainly due to the large ES effects both in Uruguay and Sweden. The difference to the literature values highlights the overall variability in calculation of SFs and substitution effects, as discussed in more detail below. In addition, varying SFs among studies can emphasize the uncertainty connected to input parameters in LCAs of forest products which should be subjected to uncertainty analysis, for example, via Monte Carlo simulation (Sahoo et al., 2019). In this study, however, uncertainty was examined for the most climate influential part of the life cycle (i.e., the substitution effect) by sensitivity analysis of changing material replacement rates and varying marginal energy mixes.

4.2 | Methodological climate impact criteria and limitations

4.2.1 | Temporal assumptions and decisive factors for biogenic carbon effects

Time dynamic climate effects involve various assumptions, among which the chosen time horizon can be decisive for the results. In this study, 100 years on a hectare basis was chosen, to facilitate comparisons with other studies dealing with the climate effects of eucalyptus plantations. In addition, 50 years per unit of beverage carton was included, to highlight potential implications for the EU's committed climate neutrality target by 2050 (EC, 2020b). In general, defining shorter time horizons such as 50 years means truncating climate effects, which can disguise long-term benefits from biogenic carbon storage or a change where climate cooling becomes a warming effect. Thus, shorter time horizons can highlight immediate possible contributions of a (wood) product system for required action to meet short- or medium-term climate policy targets. Moreover, shorter time horizons lower the uncertainty in the results, due to decreased potential for future changes in, for example, technology, benchmark product characteristics, production processes, or end-of-life scenarios (Peñaloza et al., 2019). Overall, it is

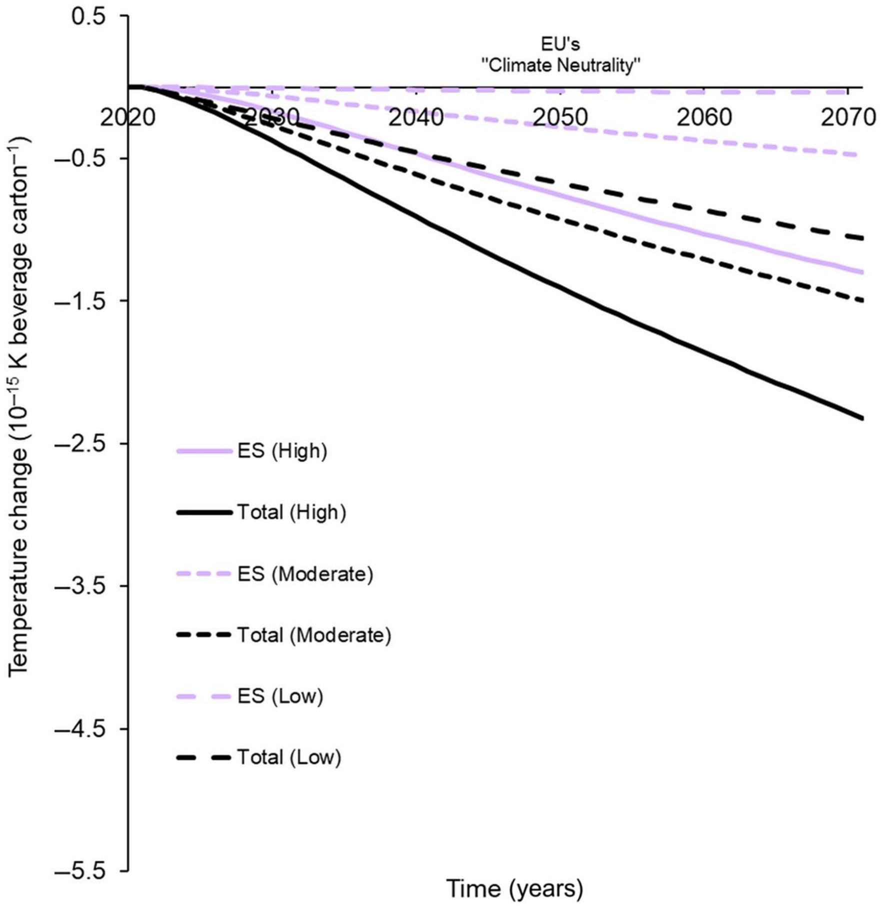

FIGURE 9 Temperature change in sensitivity analysis on changing the marginal Uruguayan electricity and marginal Swedish energy mix, influencing energy substitution (ES). The entire climate effect (Total) includes biogenic carbon, value chain emissions, and both energy and material substitution. The baseline represents a fossil- intense ES (marginal mix for 2020), while the other scenarios represent a moderately fossil-based ES (marginal mix for 2030) and a renewable-intense ES (marginal mix for 2040). *Note*: Energy substitution includes end-of-life emissions and forgone energy credit from avoided polyethylene terephthalate combustion. The results are shown from a landscape perspective

important to be aware of the consequences of a chosen time horizon for the climate effect results and consider whether an alternative choice might yield a different outcome (Lueddeckens et al., 2020).

Timing of biogenic carbon uptake before or after harvest is another important aspect for time dynamic climate assessments. In this study, accounting before harvest was applied, since it "represents reality better as the trees have to grow before they can be harvested" (Peñaloza et al., 2019) and since the objective was to advance knowledge of a product system's climate effects. In a landscape perspective modeling context, however, assumptions on former land use can be more decisive for the climate effects. In any case, there is consensus that applying a dynamic life cycle inventory and dynamic characterization increases the accuracy of LCA, regardless of the timing assumptions made (Lueddeckens et al., 2020).

A sustainable forestry system was assumed, for which a theoretical landscape perspective (i.e., a series of time-shifted stands) was applied (Berndes et al., 2016). This meant that all biogenic carbon from standing biomass was taken up from the atmosphere in year zero and no further fluxes were assessed. Thus, carbon equilibrium in the standing biomass was maintained over the rotation periods, in accordance with Sathre and O'Connor (2010). However, including the built-up phase of the plantation, which occurred before the start of the time horizon assessed here, could have substantially influenced overall biogenic carbon effects. This is because major biogenic carbon fluxes occur during this period due to biomass growth in the plantation, but were not assessed to ensure constant input–output flow in the time dynamic life cycle inventory. In contrast to a stand perspective, a landscape perspective can account for the simultaneous occurrence of silvicultural and subsequent manufacturing operations. It can thus be more realistic, increasing the policy relevance of the model. However, assumptions about production improvements (e.g., due to breeding, fertilization, etc.) and threats from future climate change were omitted here, and could add further policy relevance if included in the assessment.

Land use change and land use reference substantially influenced climate effects of biogenic carbon from standing biomass and the soil (Peñaloza et al., 2019). In this study, a LUC from grassland to eucalyptus plantation was assumed, while another option could be to rely on natural regeneration (continuation of grassland) (Helin et al., 2013; Koponen et al., 2018; Soimakallio et al., 2015). In any case, it is important to note that choosing the land use reference can cause a climate warming effect, for example, due to LUC from natural forest to plantations (Sathre & O'Connor, 2010). This highlights the major importance of avoiding conversion of natural forest into managed forest, as the former is considered superior in removing carbon from the atmosphere (Lewis et al., 2019). Similar to LUC from natural forest to plantations, conversion of grassland into eucalyptus plantations directly affects SOC stocks. In this study, SOC levels reached a quasi-long-term equilibrium, as also assumed in other studies on managed forests (Ericsson et al., 2013; Sathre & O'Connor, 2010). However, whether SOC levels under eucalyptus plantations remain constant over time is still uncertain (Behtrong et al., 2012; Cavalett et al., 2018; Fialho & Zinn, 2014; McMahon et al., 2019; Sandoval López et al., 2020), and thus implications for the climate. In addition, future biogenic carbon dynamics will probably be increasingly influenced by ongoing climate change, as climate-driven risks may fundamentally compromise (managed) forest carbon sinks (Anderegg et al., 2020), especially in regions such as South America (Payn et al., 2015).

From a product system perspective on the beverage carton, biogenic carbon played a minor role in the time dynamic climate effects. In another context where biogenic carbon from the plantation or forest could differ more substantially from a system perspective, defining the land use reference using more accurate data could be more important. In that case, time dynamic climate effects from LUC and biogenic carbon should consider the implications of indirect LUC (Faraca et al., 2019) and albedo (Sieber et al., 2019). However, calculation of these is still associated with great uncertainty and was thus omitted. Further research is needed on other forestry systems (e.g., Nordic forests) to increase knowledge on temporal and spatial variations in the role of biogenic carbon in time dynamic climate effects of wood product systems.

4.2.2 | The role of substitution effects

Substitution effects can be regarded as permanent (Sathre & O'Connor, 2010), in contrast to the temporary climate benefit from biogenic carbon storage. However, this approach requires decisive assumptions concerning both material and ES.

Material substitution effects may be substantially lowered if technologies of production or the properties of the substituted product (i.e., the PET bottle) improve (Leskinen et al., 2018). These developments could be accounted for in modeling by, for example, reducing the mass used for the PET bottle (Markwardt et al., 2017) or changing replacement rate of the wood product (Hammar et al., 2020). From a climate mitigation perspective, it is best to aim for replacing those nonrenewable materials which are most carbon intense and which are unlikely to become more efficient in their production (Suter et al., 2017; Verkerk et al., 2020). However, technological improvements in nonrenewable materials could also stimulate wood product manufacturers to improve production efficiency in their technology, for example, by improving heat recovery in paper mills (Corcelli et al., 2018).

Inherent properties of the materials, such as the LHV, can also be decisive for the climate effects during end-of-life incineration. Since the LHV of a PET bottle is higher than that of a beverage carton, avoided PET bottle incineration involves a forgone credit from ES which is greater than the credit derived from incinerating the beverage carton. Thus, the forgone credit from ES can result in a climate warming effect when regarded from a system perspective. In addition, climate cooling effects of ES from beverage carton incineration in Sweden are likely to decrease when the displaced future marginal energy mix becomes increasingly based on renewable energy via, for example, wind and solar energy. Such a development is likely for the entire EU, considering the commitment to ambitious renewable energy targets (Faraca et al., 2019). ES may eventually lead to a climate warming effect (Figure 9). However, the energy mix substituted is a normative definition and can differ between, for example, the current mix, the marginal mix, or a changing mix (Knauf et al., 2016). In any case, bioenergy from combustion of wood products can be important in balancing future energy fluctuations from solar and wind power (Arasto et al., 2017). By applying both a climate mitigation and a renewable energy supply perspective, bioenergy can thus be a crucial contributor to achieving aligned policy ambitions, such as the EU's climate neutrality target.

However, to increase the policy relevance of substitution in a growing bioeconomy, "leakage effects" should be considered in the climate assessment. This includes potential emissions shifts induced by shifted activities from sustainable forestry to less sustainably managed forestry (Leskinen et al., 2018) or cross-sectoral shifts of avoided climate burdens (Harmon, 2019).

In general, substitution effects of wood products are a highly debated issue and there is currently no consistent basis for assessment. Some highlight the potentially strong climate mitigation effect of substitution (Leskinen et al., 2018; Sathre & O'Connor, 2010), while others argue the opposite (Harmon, 2019; Leturcq, 2020). In fact, it is claimed that the long-term climate mitigation benefits from substitution may have been overestimated by two- to 100-fold, for example, through the frequent assumption of keeping SFs constant over time and omitting "leakage effects" (Harmon, 2019). In this context, the strongest climate mitigation effect for wood products may not be induced by increasing harvest levels (Leturcq, 2020). Irrespective of this, effective substitution should be focused on increasing wood application in the construction sector, but accounting for substitution credit will still only be valid if an "increase in wood product consumption implies verifiably a global reduction in non-wood productions" (Leturcq, 2020). Without policies ensuring the absence of "leakage effects," substitution benefits from wood products can be limited (Harmon, 2019).

This study confirmed the great variability in substitution effects and associated SFs. Sensitivity analysis on varying replacement rates and differing marginal energy mixes was conducted to alleviate the shortcomings of static substitution over time. However, to improve the validity of substitution effects (of wood product systems), dynamic substitution factors covering time-dependent changes (in e.g., energy mixes or technological advances in a product) should be used in future assessments, as implemented by Brunet-Navarro et al. (2021).

4.2.3 | Cascading wood use

Directly connected to substitution effects is cascading use of wood products. Whether cascading leads to definite increased climate cooling effects is debated (Berndes et al., 2016; Höglmeier et al., 2014, 2015; Leskinen et al., 2018; Sathre & Gustavsson, 2006; Suter et al., 2017). In a meta-analysis, Thonemann and Schumann (2018) found that around half of the studies reviewed indicated a positive effect from cascading on the climate and the other half a mixed effect. Moreover, perceptions vary on whether additional cascading steps naturally lead to additional climate benefits. In this regard, Faraca et al. (2019) concluded that the largest climate benefits are obtained in the first cascade step, "when the quality of the resource is at its highest point." This highlights the importance of choosing quality-oriented recycling. In the present study, a pulp product was investigated, which limited cascading use. Nevertheless, it was shown that the energy recovery step brought a substantial potential benefit to the overall climate effects of the product.

Irrespective of its controversial climate implications, cascading wood use can bring great benefits in terms of land use (Höglmeier et al., 2015; Sathre & Gustavsson, 2006; Suter et al., 2017). In fact, it is key to consider other impact categories apart from climate effects, such as water consumption (Ferraz et al., 2019) or biodiversity aspects (Pozo & Säumel, 2018) when assessing the environmental sustainability of a eucalyptus wood product system. Since only 8% of the managed forests in South America are under PEFC or FSC certification (Sikkema et al., 2017), there is an urgent need for scrutiny of environmental sustainability in LCA of the European wood supply (O'Sullivan et al., 2016).

4.2.4 | Suitability of the climate metric chosen

The climate metric chosen can substantially influence the outcome of a time dynamic climate impact assessment (Breton et al., 2018). Apart from the AGTP, alternative climate metrics also accounting for time dynamics are, for example, the GWP_{bio} (Cherubini et al., 2011), or time-dependent radiative forcing (Sathre & Gustavsson, 2012), which are further "up" the cause–effect chain from GHG emission to climate change impacts (Breton et al., 2018; Myhre et al., 2013). The focus in this study was on AGTP, which accurately incorporated time-dependent carbon fluxes of biomass-based systems. Thus, the metric is reliable and flexible, and provides a better understanding of biogenic carbon effects on the climate in a long- and short-term perspective, as also stated by Peñaloza et al. (2019) and Garcia et al. (2020). Regarding substitution effects, the AGTP metric functioned well in accounting for potential time-dependent emission savings. In future assessments, AGTP would therefore be a suitable metric to include recommended dynamic substitution factors to account for future changes concerning substitution-related aspects, such as improved production efficiency.

4.3 | Potential relevance for climate policy making

Biogenic carbon and substitution are both regarded as important pillars in climate-smart forestry for policy making of bio-based products in a growing bioeconomy (Nabuurs et al., 2018). This study revealed a potential temperature change range from $-0.8 \cdot 10^{-15}$ to $-1.8 \cdot 10^{-15}$ K per unit of beverage carton until the year 2050, based on all results from the sensitivity analysis performed. Considering an average annual output volume of a European liquid packaging board mill of approximately one million tons (Stora Enso, 2019), a potential temperature change range from $-2.9 \cdot 10^{-10}$ K to $-6.2 \cdot 10^{-10}$ K could thus be reached for 2050 by producing and using beverage cartons over PET bottles. As CO_2 emissions intensity from the European paper industry has decreased by approximately 25% since 1990 already (Corcelli et al., 2018), the results of this study can thus highlight the additional climate benefit of a rapid replacement of fossil-based PET bottles by the use of bio-based beverage cartons.

5 | CONCLUSIONS

The forest sector can play a pivotal role in mitigating climate warming, but assessing the actual climate effects of a wood product system is complex and requires a holistic approach. This study revealed that inclusion of biogenic carbon and substitution effects improved the holistic and time dynamic climate effect assessment of a bioeconomically promising wood product, a beverage carton. The results showed that substantial climate warming from value chain emissions was barely offset by biogenic carbon from the eucalyptus plantation, including standing biomass and SOC, as well as short carbon storage within the HWP. In contrast, effects from MS and displacing marginal energy mixes converted the time dynamic temperature change into considerable climate cooling. Sensitivity analysis involving varying replacement rates of the beverage carton and differing marginal energy mixes showed great variability of substitution effects for both substitution factors and temperature change. Potential relevance for climate policy making was evident from a climate cooling effect of $-0.8 \cdot 10^{-15}$ to $-1.8 \cdot 10^{-15}$ K per unit of beverage carton by 2050. Production and use of wood-based beverage cartons instead of PET bottles can therefore contribute to mitigating climate warming effects. Further holistic time dynamic climate assessments on alternative forestry systems (e.g., Nordic forests) are needed to advance knowledge on the role of biogenic carbon, including dynamic assessment of substitution effects.

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CONFLICT OF INTEREST

The authors declare that they have no conflicts of interest.

AUTHOR CONTRIBUTIONS

MS performed the LCA modeling and led the writing process. TH made substantial contributions to the methodological approach, the conception of the study, and interpretation of data. JS contributed considerably by raising important aspects about the study's concept and discussion. MSe modeled primary data into soil organic carbon stocks using the dynamic soil carbon model Yasso 15. PAH made substantial contributions to conception and design of the study.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available in the Supporting Information of this article.

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REFERENCES

Anderegg, W. R. L., Trugman, A. T., Badgley, G., Anderson, C. M., Bartuska, A., Ciais, P., Cullenward, D., Field, C. B., Freeman, J., Goetz, S. J., Hicke, J. A., Huntzinger, D., Jackson, R. B., Nickerson, J., Pacala, S., & Randerson, J. T. (2020). Climate-driven risks to the climate mitigation potential of forests. *Science*, *368*(6497). https://doi.org/10.1126/science.aaz7005

Arasto, A., Chiaramonti, D., Kiviluoma, J., van den Heuvel, E., Waldheim, L., Maniatis, K., & Sipilä, K. (2017). *Bioenergy's role in balancing the electricity grid and providing storage options—An EU perspective*. IEA Bioenergy.

Behtrong, S. T., Pineiro, G., Jobbágy, E. G., & Jackson, R. B. (2012). Soil C and N changes with afforestation of grasslands across gradients of precipitation and plantation age. *Ecological Applications*, *22*(1), 76–86.

Berndes, G., Abt, B., Asikainen, A., Cowie, A., Dale, V., Egnell, G., Lindner, M., Marelli, L., Paré, D., Pingoud, K., & Yeh, S. (2016) *Forest biomass, carbon neutrality and climate change mitigation. From Science to Policy/European Forest Institute* (Vol. *3*). EFI.

Bernstad Saraiva, A., Valle, R., Bosquê, A., Berglin, N., & Schenck, A. (2017). Provision of pulpwood and short rotation eucalyptus in Bahia, Brazil—Environmental impacts based on lifecycle assessment methodology. *Biomass and Bioenergy*, *105*, 41–50. https://doi.org/10.1016/j.biombioe.2017.06.004

Breton, C., Blanchet, P., Amor, B., Beauregard, R., & Chang, W.-S. (2018). Assessing the climate change impacts of biogenic carbon in buildings: A critical review of two main dynamic approaches. *Sustainability*, *10*, 2020. https://doi.org/10.3390/su10062020

Brunet-Navarro, P., Jochheim, H., Cardellini, G., Richter, K., & Muys, B. (2021). Climate mitigation by energy and material substitution of wood products has an expiry date. *Journal of Cleaner Production*, *303*, 127026. https://doi.org/10.1016/j.jclepro.2021.127026

Cavalett, O., Norem Slettmo, S., & Cherubini, F. (2018). Energy and environmental aspects of using eucalyptus from Brazil for energy and transportation services in Europe. *Sustainability*, *10*(11), 4068. https://doi.org/10.3390/su10114068

Cherubini, F., Peters, G. P., Berntsen, T., Strøman, A. H., & Hertwich, E. (2011). CO_2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. *GCB Bioenergy*, *3*, 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x

Cintas, O., Berndes, G., Cowie, A. L., Egnell, G., Holmström, H., & Ågren, G. I. (2016). The climate effect of increased forest bioenergy use in Sweden: Evaluation at different spatial and temporal scales. *Wiley Interdisciplinary Reviews: Energy and Environment*, *5*(3), 351–369. https://doi.org/10.1002/wene.178

Corcelli, F., Ripa, M., & Ulgiati, S. (2018). Efficiency and sustainability indicators for papermaking from virgin pulp—An emergy-based case study. *Resources, Conservation and Recycling*, *131*, 313–328. https://doi.org/10.1016/j.resconrec.2017.11.028

Doka, G. (2013). *Treatment of waste polyethylene terephthalate, municipal incineration, RoW. ecoinvent 3.6 dataset documentation*.

EC. (2011). *Cogeneration, or Combined Heat and Power (CHP). Technology information sheet cogeneration. SETIS—Information for decision making*. Brussels: European Commission.

EC. (2016a). *Cascades. Study on the optimised cascading use of wood. Final report*. Brussels: European Commission.

EC. (2016b). *Regulation (EC) No 1935/2004 of the European Parliament and of the Council of 27 October 2004 on materials and articles intended to come into contact with food and repealing Directives 80/590/EEC and 89/109/EEC*. Brussels: European Commission.

EC. (2018). *A sustainable bioeconomy for Europe. Strengthening the connection between economy, society and the environment: Updated bioeconomy strategy*. Publications Office of the European Union.

EC. (2020a). *Proposal for a regulation of the European Parliament and of the Council establishing the framework for achieving climate neutrality and amending Regulation (EU) 2018/1999 (European Climate Law)*. Brussels: European Commission.

EC. (2020b). *Internal market, industry, entrepreneurship and SMEs. Pulp and paper industry*. Retrieved June 29, 2020, from https://ec.europa.eu/growth/sectors/raw-materials/industries/forest-based/pulp-paper_en

ECN. (2021). *Phyllis2. Database for biomass and waste. Black liquor (#1395)*. Energy Research Centre of the Netherlands.

EFBW. (2020). *Bottled water packaging. Packaging lightweight. Weight reduction per bottle in grammes*. Retrieved August 10, 2020, from https://www.efbw.org/index.php?id=52

Eliasson, P., Svensson, M., Olsson, M., & Ågren, G. I. (2013). Forest carbon balances at the landscape scale investigated with the Q model and the CoupModel—Responses to intensified harvests. *Forest Ecology and Management*, *290*, 67–78. https://doi.org/10.1016/j.foreco.2012.09.007

Ericsson, N., Porsö, C., Ahlgren, S., Nordberg, Å., Sundberg, C., & Hansson, P.-A. (2013). Time-dependent climate impact of a bioenergy system—Methodology development and application to Swedish conditions. *GCB Bioenergy*, *5*(5), 580–590. https://doi.org/10.1111/gcbb.12031

EUNOMIA. (2020). *PET market in Europe. State of play. Production, collection and recycling data*. Brussels: European Federation of Bottled Waters, Petcore Europe, Plastics Recyclers Europe.

Eurosta. (2020). *Average distance on which goods are carried, 2018*. Brussels: European Commission.

FAO & CMCC. (2017). *Global soil organic carbon database (at 30 arcsec). Geocarbon; toward an operational global carbon observing system*. Rome: Food and Agriculture Organization of the United Nations, Euro-Mediterranean Center on Climate Change Foundation.

FAO. (2019). *Pulp and Paper Capacities|Capacités de la pâte et du papier|Capacidades de pasta y papel 2018–2023*. Rome: Food and Agriculture Organization of the United Nations.

Faraca, G., Tonini, D., & Astrup, T. F. (2019). Dynamic accounting of greenhouse gas emissions from cascading utilisation of wood waste. *The Science of the Total Environment*, *651*(Pt. 2), 2689–2700. https://doi.org/10.1016/j.scitotenv.2018.10.136

Ferraz, S. F. B., Rodrigues, C. B., Garcia, L. G., Alvares, C. A., & Lima, W. P. (2019). Effects of Eucalyptus plantations on streamflow in Brazil: Moving beyond the water use debate. *Forest Ecology and Management*, *453*, 117571. https://doi.org/10.1016/j.foreco.2019.117571

Fialho, R. C., & Zinn, Y. L. (2014). Changes in soil organic carbon under eucalyptus plantations in Brazil: A comparative analysis. *Land Degradation & Development*, *25*(5), 428–437. https://doi.org/10.1002/ldr.2158

Fröhlich, T. (2017). *Polyethylene terephthalate production, granulate, bottle grade, RER. ecoinvent 3.6 dataset documentation*.

Gabrielle, B., Nguyen The, N., Maupu, P., & Vial, E. (2013). Life cycle assessment of eucalyptus short rotation coppices for bioenergy production in southern France. *GCB Bioenergy*, *5*(1), 30–42. https://doi.org/10.1111/gcbb.12008

Garcia, R., Alvarenga, R. A. F., Huysveld, S., Dewulf, J., & Allacker, K. (2020). Accounting for biogenic carbon and end-of-life allocation in life cycle assessment of multi-output wood cascade systems. *Journal of Cleaner Production*, *275*, 122795. https://doi.org/10.1016/j.jclepro.2020.122795

Geng, A., Yang, H., Chen, J., & Hong, Y. (2017). Review of carbon storage function of harvested wood products and the potential of wood substitution in greenhouse gas mitigation. *Forest Policy and Economics*, *85*, 192–200. https://doi.org/10.1016/j.forpol.2017.08.007

Giraldo, L., & Hyman, B. (1996). An energy process-step model for manufacturing paper and paperboard. *Energy*, *21*(7/8), 667–681. https://doi.org/10.1016/0360-5442(96)00016-3

Giuntoli, J., Agostini, A., Edwards, R., & Marelli, L. (2015). *Solid and gaseous bioenergy pathways: Input values and GHG emissions. Calculated according to the methodology set in COM(2010) 11 and SWD(2014) 259*. Joint Research Center, European Commission.

González-Goméz, M. (2019). Dynamics of pulp exports from South America to the European Union. *Austrian Journal of Forest Science*, *136*(4), 313–330.

Hagberg, M., Gode, J., Lätt, A., Ekvall, T., Adolfsson, I., & Martinsson, F. (2017). *Miljövärdering av energilösningar i byggnader (etapp 2). Metod för konsekvensanalys*. Stockholm: ivl - Svenska Miljöintitutet.

Hammar, T., Hansson, P. A., Seleborg, M., & Stendahl, J. (2020). *Climate effects of a forestry company—Including biogenic carbon fluxes and substitution effects*. Report. Uppsala: Swedish University of Agricultural Sciences (SLU).

Hammar, T., & Levihn, F. (2020). Time-dependent climate impact of biomass use in a fourth generation district heating system, including BECCS. *Biomass and Bioenergy*, *138*, 105606. https://doi.org/10.1016/j.biombioe.2020.105606

Harmon, M. E. (2019). Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. *Environmental Research Letters*, *14*(6), 65008. https://doi.org/10.1088/1748-9326/ab1e95

Head, M., Bernier, P., Levasseur, A., Beauregard, R., & Margni, M. (2019). Forestry carbon budget models to improve biogenic carbon accounting in life cycle assessment. *Journal of Cleaner Production*, *213*, 289–299. https://doi.org/10.1016/j.jclepro.2018.12.122

Helin, T., Sokka, L., Soimakallio, S., Pingoud, K., & Pajula, T. (2013). Approaches for inclusion of forest carbon cycle in life cycle assessment—A review. *GCB Bioenergy*, *5*, 475–486. https://doi.org/10.1111/gcbb.12016

Höglmeier, K., Steubing, B., Weber-Blaschke, G., & Richter, K. (2015). LCA-based optimization of wood utilization under special consideration of a cascading use of wood. *Journal of Environmental*

Management, *152*, 158–170. https://doi.org/10.1016/j.jenvman.2015.01.018

Höglmeier, K., Weber-Blaschke, G., & Richter, K. (2014). Utilization of recovered wood in cascades versus utilization of primary wood—A comparison with life cycle assessment using system expansion. *International Journal of Life Cycle Assessment*, *19*(10), 1755–1766. https://doi.org/10.1007/s11367-014-0774-6

Holmgren, P., & Kolar, K. (2019). *Reporting the overall climate impact of a forestry corporation—The case of SCA*. Sundsvall: SCA - Svenska Cellulosa Aktiebolaget.

Hurmekoski, E., Jonsson, R., Korhonen, J., Jänis, J., Mäkinen, M., Leskinen, P., & Hetemäki, L. (2018). Diversification of the forest industries: Role of new wood-based products. *Canadian Journal of Forest Research*, *48*(12), 1417–1432. https://doi.org/10.1139/cjfr-2018-0116

Indufor. (2013). *Study on the wood raw material supply and demand for the EU wood-processing industries*. Final report. European Commission, Enterprise and Industry Directorate General.

IPCC. (2014) Summary for policymakers. In O. Edenhofer, Y. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel, & J. C. Minx (Eds.), *Climate change 2014: Mitigation of climate change. Contribution of working group III to the fifth assessment report of the intergovernmental panel on climate change* (pp. 8–12). Cambridge and New York, NY: Cambridge University Press.

IPCC. (2019). *N_2O emissions from managed soils, and CO_2 emissions from lime and urea application. 2019 refinement to the 2006 IPCC guidelines for National Greenhouse Gas Inventories. Chapter 11*. Intergovernmental Panel on Climate Change.

ISO. (2006a). *ISO 14040: Environmental management—life cycle assessment—Principles and framework*. International Organization for Standardization.

ISO. (2006b). *ISO 14044: Environmental management—Life cycle assessment—Requirements and guidelines*. International Organization for Standardization.

Järvenpää, M., Repo, A., Akujärvi, A., Kaasalainen, M., & Liski, J. (2018). *Soil carbon model Yasso15—Bayesian calibration using worldwide litter decomposition and carbon stock data* [Manuscript in preparation]. Geoscientific Model Development.

Joos, F., Colin Prentice, I., Sitch, S., Meyer, R., Hooss, G., Plattner, G.-K., Gerber, S., & Hasselmann, K. (2001). Global warming feedbacks on terrestrial carbon uptake under the Intergovernmental Panel on Climate Change (IPCC) Emission Scenarios. *Global Biogeochemical Cycles*, *15*(4), 891–907. https://doi.org/10.1029/2000GB001375

Joos, F., Roth, R., Fuglestvedt, J. S., Peters, G. P., Enting, I. G., von Bloh, W., Brovkin, V., Burke, E. J., Eby, M., Edwards, N. R., Friedrich, T., Frölicher, T. L., Halloran, P. R., Holden, P. B., Jones, C., Kleinen, T., Mackenzie, F. T., Matsumoto, K., Meinshausen, M., … Weaver, A. J. (2013). Carbon dioxide and climate impulse response functions for the computation of greenhouse gas metrics: A multi-model analysis. *Atmospheric Chemistry and Physics*, *13*(5), 2793–2825. https://doi.org/10.5194/acp-13-2793-2013

Judl, J., Koskela, S., Mattila, T., & Jouttijärvi, T. (2011). The climate change implications of offshoring Finnish pulp production to South America. *International Journal of Life Cycle Assessment*, *16*(9), 878–885. https://doi.org/10.1007/s11367-011-0308-4

Klein, D., Wolf, C., Schulz, C., & Weber-Blaschke, G. (2015). 20 years of life cycle assessment (LCA) in the forestry sector: State of the art and a methodical proposal for the LCA of forest production. *International Journal of Life Cycle Assessment*, *20*(4), 556–575. https://doi.org/10.1007/s11367-015-0847-1

Knauf, M., Joosten, R., & Frühwald, A. (2016). Assessing fossil fuel substitution through wood use based on long-term simulations. *Carbon Management*, *7*(1–2), 67–77. https://doi.org/10.1080/17583004.2016.1166427

Koponen, K., Soimakallio, S., Kline, K. L., Cowie, A., & Brandão, M. (2018). Quantifying the climate effects of bioenergy—Choice of reference system. *Renewable and Sustainable Energy Reviews*, *81*, 2271–2280. https://doi.org/10.1016/j.rser.2017.05.292

Kuparinen, K., Vakkilainen, E., & Tynjälä, T. (2019). Biomass-based carbon capture and utilization in kraft pulp mills. *Mitigation and Adaptation Strategies for Global Change*, *24*(7), 1213–1230. https://doi.org/10.1007/s11027-018-9833-9

Leskinen, P., Cardellini, G., González-García, S., Hurmekoski, E., Sathre, R., Seppälä, J., Smyth, C., Stern, T., & Verkerk, P. J. (2018). *Substitution effects of wood-based products in climate change mitigation. From Science to Policy 7*. European Forest Institute.

Leturcq, P. (2020). GHG displacement factors of harvested wood products: The myth of substitution. *Scientific Reports*, *10*(1), 20752. https://doi.org/10.1038/s41598-020-77527-8

Levasseur, A., Lesage, P., Margni, M., Deschênes, L., & Samson, R. (2010). Considering time in LCA: Dynamic LCA and its application to global warming impact assessments. *Environmental Science & Technology*, *44*(8), 3169–3174. https://doi.org/10.1021/es9030003

Lewis, S. L., Wheeler, C. E., Mitchard, E. T. A., & Koch, A. (2019). Restoring natural forests is the best way to remove atmospheric carbon. *Nature*, *568*, 25–28. https://doi.org/10.1038/d41586-019-01026-8

Lueddeckens, S., Saling, P., & Guenther, E. (2020). Temporal issues in life cycle assessment—A systematic review. *International Journal of Life Cycle Assessment*, *25*(8), 1385–1401. https://doi.org/10.1007/s11367-020-01757-1

Markwardt, S., Wellenreuther, F., Drescher, A., Harth, J., & Busch, M. (2017). *Comparative life cycle assessment of Tetra Pak® carton packages and alternative packaging systems for liquid food on the Nordic market*. Final report. Tetra Pak International SA.

McKinnon, A., & Piecyk, M. (2010). *Measuring and managing CO_2 emissions of European chemical transport*. Brussels: European Chemical Industry Council.

McMahon, D. E., Vergütz, L., Valadares, S. V., Silva, I. R., & Jackson, R. B. (2019). Soil nutrient stocks are maintained over multiple rotations in Brazilian Eucalyptus plantations. *Forest Ecology and Management*, *448*, 364–375. https://doi.org/10.1016/j.foreco.2019.06.027

MIEM. (2019). *Energy balance 2019*. Ministerio de Industria, Energía y Minería (MIEM).

Montes del Plata SA. (2021). *Cellulose & energy—Clean and renewable energy*. Retrieved May 3, 2021, from https://www.montesdelplata.com.uy/english/cellulose-energy-4

Muncke, J. (2020). *Food packaging health, food packaging materials*. Retrieved September 7, 2020, from http://www.foodpackagingforum.org/food-packaging-health/food-packaging-materials

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., & Zhang, H. (2013). *Anthropogenic and natural radiative forcing. Climate Change 2013: The physical science basis*. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change, Cambridge, UK and New York, NY.

Nabuurs, G.-J., Verkerk, P. J., Schelhaas, M.-J., González Olabarria, J. R., Trasobares, A., & Cienciala, E. (2018). *Climate-smart forestry: Mitigation impacts in three European regions. From science to policy 6*. European Forest Institute.

NTM. (2020). *NTM.Calc 4.0*. Retrieved August 6, 2020, from https://www.transportmeasures.org/NTM.calc/v4/basic/index.html#/

O'Sullivan, O. S., Hopkinson, L., Crockatt, M. E., Laird, P., & Bebber, D. P. (2016). Stable forest carbon stocks support current assumption of biogenic carbon neutrality in the case of European-manufactured beverage cartons. *International Journal of Life Cycle Assessment*, *21*(1), 118–128. https://doi.org/10.1007/s11367-015-0989-1

Payn, T., Carnus, J.-M., Freer-Smith, P., Kimberley, M., Kollert, W., Liu, S., Orazio, C., Rodriguez, L., Silva, L. N., & Wingfield, M. J. (2015). Changes in planted forests and future global implications. *Forest Ecology and Management*, *352*, 57–67. https://doi.org/10.1016/j.foreco.2015.06.021

Peñaloza, D., Røyne, F., Sandin, G., Svanström, M., & Erlandsson, M. (2019). The influence of system boundaries and baseline in climate impact assessment of forest products. *International Journal of Life Cycle Assessment*, *24*(1), 160–176. https://doi.org/10.1007/s11367-018-1495-z

Porsö, C., Mate, R., Vinterbäck, J., & Hansson, P.-A. (2016). Time-dependent climate effects of eucalyptus pellets produced in Mozambique used locally or for export. *BioEnergy Research*, *9*(3), 942–954. https://doi.org/10.1007/s12155-016-9746-y

Pozo, P., & Säumel, I. (2018). How to bloom the green desert: Eucalyptus plantations and native forests in Uruguay beyond black and white perspectives. *Forests*, *9*(10), 614. https://doi.org/10.3390/f9100614

Røyne, F., Peñaloza, D., Sandin, G., Berlin, J., & Svanström, M. (2016). Climate impact assessment in life cycle assessments of forest products: Implications of method choice for results and decision-making. *Journal of Cleaner Production*, *116*, 90–99. https://doi.org/10.1016/j.jclepro.2016.01.009

Rüter, S., Matthews, R. W., Lundblad, M., Sato, A., & Hassan, R. A. (2019). *Harvested wood products. Chapter 12. 2019 Refinement to the 2006 IPCC guidelines for National Greenhouse Gas Inventories*. Intergovernmental Panel on Climate Change.

Sahoo, K., Bergman, R., Alanya-Rosenbaum, S., Gu, H., & Liang, S. (2019). Life cycle assessment of forest-based products: A review. *Sustainability*, *11*(17), 4722. https://doi.org/10.3390/su11174722

Sandoval López, D. M., Arturi, M. F., Goya, J. F., Pérez, C. A., & Frangi, J. L. (2020). Eucalyptus grandis plantations: Effects of management on soil carbon, nutrient contents and yields. *Journal of Forest Research*, *31*(2), 601–611. https://doi.org/10.1007/s11676-018-0850-z

Sathre, R., & Gustavsson, L. (2006). Energy and carbon balances of wood cascade chains. *Resources, Conservation and Recycling*, *47*(4), 332–355. https://doi.org/10.1016/j.resconrec.2005.12.008

Sathre, R., & Gustavsson, L. (2012). Time-dependent radiative forcing effects of forest fertilization and biomass substitution. *Biogeochemistry*, *109*, 203–218. https://doi.org/10.1007/s10533-011-9620-0

Sathre, R., & O'Connor, J. (2010). Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environmental Science & Policy*, *13*(2), 104–114. https://doi.org/10.1016/j.envsci.2009.12.005

SCB. (2019). *Total tillförd och återvunnen mängd förpackningar uppdelat efter förpackningsslag, tabellinnehåll och år* [Total amount of packaging put on the market and recycled broken down by type of packaging]. Swedish Statistics Service.

Sieber, P., Ericsson, N., & Hansson, P.-A. (2019). Climate impact of surface albedo change in life cycle assessment: Implications of site and time dependence. *Environmental Impact Assessment Review*, *77*, 191–200. https://doi.org/10.1016/j.eiar.2019.04.003

Sikkema, R., Dallemand, J. F., Matos, C. T., van der Velde, M., & San-Miguel-Ayanz, J. (2017). How can the ambitious goals for the EU's future bioeconomy be supported by sustainable and efficient wood sourcing practices? *Scandinavian Journal of Forest Research*, *32*(7), 551–558. https://doi.org/10.1080/02827581.2016.1240228

Silva, D. A. L., Raymundo Pavan, A. L., Augusto de Oliveira, J., & Ometto, A. R. (2015). Life cycle assessment of offset paper production in Brazil: Hotspots and cleaner production alternatives. *Journal of Cleaner Production*, *93*, 222–233. https://doi.org/10.1016/j.jclepro.2015.01.030

Simons, A. (2019). *Transport, freight, lorry >32 metric ton, EURO5. GLO. ecoinvent 3.6 dataset documentation*.

Soimakallio, S., Cowie, A., Brandão, M., Finnveden, G., Ekvall, T., Erlandsson, M., Koponen, K., & Karlsson, P.-E. (2015). Attributional life cycle assessment: Is a land-use baseline necessary? *International Journal of Life Cycle Assessment*, *20*(10), 1364–1375. https://doi.org/10.1007/s11367-015-0947-y

Soimakallio, S., Saikku, L., Valsta, L., & Pingoud, K. (2016). Climate change mitigation challenge for wood utilization—The case of Finland. *Environmental Science & Technology*, *50*(10), 5127–5134. https://doi.org/10.1021/acs.est.6b00122

Stora Enso. (2019). *Annual report 2019*. Helsinki, Stockholm: Stora Enso Oyj, Stora Enso AB.

Sun, M., Wang, Y., Shi, L., & Klemeš, J. J. (2018). Uncovering energy use, carbon emissions and environmental burdens of pulp and paper industry: A systematic review and meta-analysis. *Renewable and Sustainable Energy Reviews*, *92*, 823–833. https://doi.org/10.1016/j.rser.2018.04.036

Suter, F., Steubing, B., & Hellweg, S. (2017). Life cycle impacts and benefits of wood along the value chain: The case of Switzerland. *Journal of Industrial Ecology*, *21*(4), 874–886. https://doi.org/10.1111/jiec.12486

Swedish Standard Institute (2004). *Packaging—Requirements for packaging recoverable in the form of energy recovery, including specification of minimum inferior calorific value 13.030.99; 55.020(13431)*. Swedish Standard Institute.

Thonemann, N., & Schumann, M. (2018). Environmental impacts of wood-based products under consideration of cascade utilization: A systematic literature review. *Journal of Cleaner Production*, *172*, 4181–4188. https://doi.org/10.1016/j.jclepro.2016.12.069

Timander, P. (2011). *Fertilization in Eucalyptus urophylla plantations in Guangxi, southern China, Alnarp*. Alnarp: Swedish University of Agricultural Sciences (SLU).

Trzciński, G., Moskalik, T., & Wojtan, R. (2018). Total weight and axle loads of truck units in the transport of timber depending on the timber cargo. *Forests*, *9*(4), 164. https://doi.org/10.3390/f9040164

UNFCCC. (2015) *Paris agreement*. Bonn: United Nations Framework Convention on Climate Change (UNFCCC).

Verkerk, P. J., Costanza, R., Hetemäki, L., Kubiszewski, I., Leskinen, P., Nabuurs, G. J., Potočnik, J., & Palahí, M. (2020). Climate-smart forestry: The missing link. *Forest Policy and Economics*, *115*, 102164. https://doi.org/10.1016/j.forpol.2020.102164

von Falkenstein, E., Wellenreuther, F., & Detzel, A. (2010). LCA studies comparing beverage cartons and alternative packaging: Can overall conclusions be drawn? *International Journal of Life Cycle Assessment*, *15*(9), 938–945. https://doi.org/10.1007/s11367-010-0218-x

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): Overview and methodology. *The International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Wolf, C., Klein, D., Weber-Blaschke, G., & Richter, K. (2016). Systematic review and meta-analysis of life cycle assessments for wood energy services. *Journal of Industrial Ecology*, *20*(4), 743–763. https://doi.org/10.1111/jiec.12321

Zhao, Q., Ding, S., Wen, Z., & Toppinen, A. (2019). Energy flows and carbon footprint in the forestry-pulp and paper industry. *Forests*, *10*(9), 725. https://doi.org/10.3390/f10090725

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