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California's harvested wood products: A time-dependent assessment of life cycle greenhouse gas emissions

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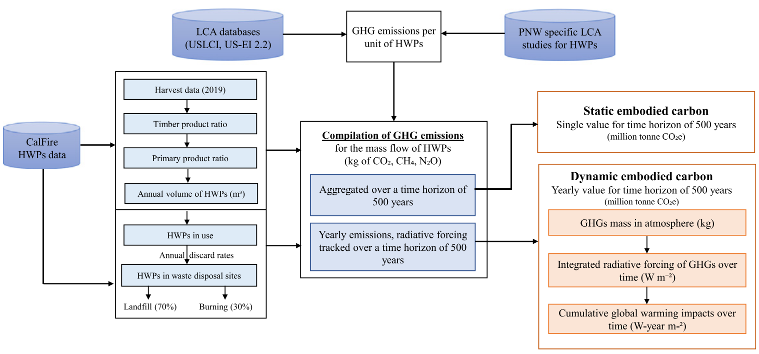
HIGHLIGHTS

- Assessment of embodied carbon of HWPs manufactured and used in California
- Static and dynamic LCA performed for cradle-to-grave life stages
- Including timing of GHG emissions helps in better understanding of the impacts.
- Global warming impacts were greatly influenced by the end-of-life disposal scenarios.

GRAPHICAL ABSTRACT



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ABSTRACT

Following life-cycle assessment (LCA) methodology, this study presents a state-level estimation of embodied carbon of wood products harvested in 2019 from California and subsequently processed, manufactured, transported, used, and disposed at the end-of-life (EoL). In a conventional static approach to LCA, all GHG emissions were aggregated and considered to occur at year 0 of the given time horizon (500 years in this study) and used a static characterization factor (CF). In dynamic LCA, GHG emissions occurring in different years were considered, and their global warming impact (GWI) was determined using a time-dependent CF over the selected time horizon of 500 years. Four scenarios were developed to examine the impact of EoL choices on GWI. It was found that dynamic GWI for all scenarios ranged from 0.27 to 0.93 million tonne CO_2e, which were 45–73 % lower than those estimated with static LCA approach, indicating that the static LCA approach could lead to an underestimation of the benefits of substituting wood for non-wood products, compared to those based on dynamic LCA approach. This analysis also demonstrated that the choice of EoL treatment option is a key factor affecting the estimated GWI as it directly determines the annual emission of GHGs released into atmosphere and subsequently their warming effect depending on the time harvested wood products (HWPs) spend in the horizon of assessment. Overall, the dynamic LCA performed in this study enabled more robust interpretations of embodied carbon by including temporal boundaries associated with the HWPs life cycle.

1. Introduction

1.1. Driving forces for harvested wood products (HWP)

The anthropogenic greenhouse gas (GHG) emissions have increased at an unprecedented rate in recent years and have put the earth system at

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risk of climate change (Steffen et al., 2015). Keeping the climate change within the safe limits would require net-zero GHG emissions by the year 2050 (IPCC, 2018). The global building operation and construction sector together accounts for more than 38 % of the global emissions (GlobalABC, 2019; UNEP, 2021), and therefore offers huge potential to mitigate GHG emissions (UNEP, 2021). Although significant efforts have been made to reduce emissions of building operations through energy efficiency, increasing urbanization and related construction activities are counteracting these efforts (GlobalABC, 2019; Hart et al., 2021). Thus, using materials with lower embodied carbon (EC) is a key strategy for reducing GHG emissions of the buildings and construction sectors over the long term (Andersen et al., 2021).

EC is an environmental metric for comparing different materials used especially in the built environment (Think Wood, 2021). Several studies suggest that harvested wood products (HWPs) have low or negative EC than their functionally equivalent non-wood materials such as steel and concrete (Bergman et al., 2014; Hart et al., 2021; Hawkins et al., 2021; Nepal et al., 2016; Saade et al., 2020). Forests sequester atmospheric carbon, and when trees are harvested, a significant amount of that carbon is transferred to durable HWPs (e.g., building materials), which remains stored for long time. In 2019, approximately 40 % of U.S. roundwood production was used in the manufacturing of durable HWPs, and 66 % of that was consumed in the construction industry (Brandeis et al., 2021). After their useful service life, HWPs are discarded, and the fate of carbon stored in HWP depends on its end-of-life (EoL) choices. For example, discarded HWP can be landfilled, where a large fraction of carbon remains stored indefinitely, or they could be recycled, fossil fuel-intensive materials or they could be burned with energy capture, replacing fossil fuel (US EPA, 2020) or simply burned without energy capture. The flow of carbon from forests through the HWPs is often considered carbon neutral if the carbon lost from harvests is recovered by biomass regrowth on forest land over time through sustainable forest management practices (Brashaw and Bergman, 2021; Jakes et al., 2016; Lippke et al., 2014). Assessment of EC of HWPs from a life-cycle perspective would provide decision makers a basis to compare their climate change mitigation potential to that of their functionally equivalent non-wood materials.

1.2. Significance for California region

This study estimates EC of wood products harvested in 2019 in California, U.S. Although California has a relatively small wood products industry (Marcille et al., 2020), it was chosen as study region because it has the largest economy of the 50 states in the U.S. and is a pioneer in addressing climate change via forest sector. For example, California Air Resource Board (CARB) has targeted a GHG reduction of 15–20 million t CO_2e by year 2030 from natural and working lands including forests (CalFire, 2023). In addition, the California region is vulnerable to wildfires; around 10,000 fires burned 4.2 million acres in California in 2020 (Jerrett et al., 2022; Redmore et al., 2021). To mitigate fire risk, California has implemented several measures such as fuel treatment practices, harvesting small-diameter trees, and expanding the use of HWPs (Cabiyo et al., 2021; Forest Climate Action Team, 2018). The state has set a goal to reduce wildfire risk on 1 million acres of land per year (Forest Climate Action Team, 2018). A sustainable expansion of the HWPs in durable uses would benefit the environment by delaying the release of carbon into atmosphere while also substituting for fossil fuel-intensive materials.

1.3. Embodied carbon and LCA

The EC, also termed embodied GHG emissions or carbon footprint, has become a popular environmental metric to compare GWI of different products or materials. The GWI is a relative measure to estimate how much a given mass of GHG once released into the atmosphere contributes to the warming effect over a period (Andersen et al., 2021; Hart et al., 2021; Hawkins et al., 2021). The term EC, embodied GHG emissions and GWI are used interchangeably in this paper. For HWPs, EC refers to the fossil carbon dioxide equivalents (CO_2e) of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emitted over their life cycle. The characterizations factors (CF), a number that quantitatively represents the GW potential of a GHG relative to CO_2, developed by IPCC are used in estimating the total EC of a product or process. The latest CFs are available in the IPCC AR6 WG I report (Forster et al., 2021).

The EC is estimated using LCA methodology guided by ISO standards (ISO 14040, 2006; ISO 14044, 2006). There are four steps in an LCA: (1) goal and scope definition of the product system under study, (2) life-cycle inventory (LCI) of all material, energy inputs and environmental emissions, (3) life-cycle impact assessment (LCIA) using CFs derived from various environmental models, and (4) interpretation for decision-making. This study used the LCA methodology adapted to building and construction sector provided in ISO 21930 (ISO 21930, 2017) and EN 15978 (EN 15978, 2011). Fig. 1 presents a layout of the scope of building LCAs. The life-cycle stages of HWPs are listed chronologically and grouped into modules of manufacturing (A1-A3), construction and installation (A4-A5), use (B), end-of-life (C), and benefits from recovery, reuse, and recycling (D). Increasing the scope of assessment would allow for better evaluation of the impacts of HWPs (Fig. 1).

The life-cycle of HWPs was considered starting with the harvesting of trees and ending when it is disposed in landfills or burned at the end of its useful life. Alternatively, HWP at EoL may end up being recycled or reused but this was outside the scope of current analysis. Estimating the EC of HWPs life cycle depends on the associated LCI data (quantity of materials inputs and energy used and GHG emissions released). However, most of the HWP LCA studies have a limited scope of assessment (Hawkins et al., 2021). Often, the LCA studies are case based focusing on the manufacturing (production) stage of a specific HWP. In a recent meta-analysis of LCA studies on embodied GHG emissions of wood buildings, nearly 50 % studies excluded EoL stages and over 90 % of the studies excluded the stage D (Andersen et al., 2021). A systems perspective covering all life stages would offer more accurate assessment of climate change mitigation potential of HWPs, which this study aims to evaluate, using a case of wood products harvested in California in 2019.

This analysis focuses on materials or product level EC assessment, which includes emissions associated with wood materials usage beyond buildings such as furniture, shipping container, repair and remodeling of buildings, railroad ties etc., but excludes emissions associated with installation/assembly (module A5) and building use (stage B).

1.4. Timing of embodied carbon emissions in HWP life-cycle

The commonly used static LCA approach treats all life-cycle GHG emissions as a single pulse emitted at the beginning of the time horizon of assessment. By doing so, this approach simplifies the effect of the timing of emissions; primarily intended to make policy development simpler. The time horizon refers to a point in time up to which radiative forcing of the GHGs is evaluated when estimating the EC. Here, the static CF (Table 1) are used in estimating the total EC of HWPs.

If all GHG emissions occurred at year 0; there would be no time inconsistency. However, in reality, emissions occurs at various stages of HWPs' life-cycle beyond year zero, such as when they are discarded after building demolition (Dyckhoff and Kasah, 2014; Hawkins et al., 2021; Reap et al., 2008; Sohn et al., 2020). Therefore, considering the time of emissions in would be relevant for the EC assessment (Collinge et al., 2013) of HWPs. To account or compare the EC of HWPs, three-time components would need to be considered. First, timing of the GHG emissions released over the life-cycle duration; warming effect for emissions released in the beginning would be different than those released a year or a decade later. Second, the time horizon of assessment which demarcates the time beyond which radiative forcing effect of GHGs wound not be considered. The third component is the CF for the time horizon analyzed, which would be different at different times rather than a static number for the entire period of assessment. Considering these time components in LCA (dynamic LCA) would help better understand the EC over entire life-cycle, especially in the case

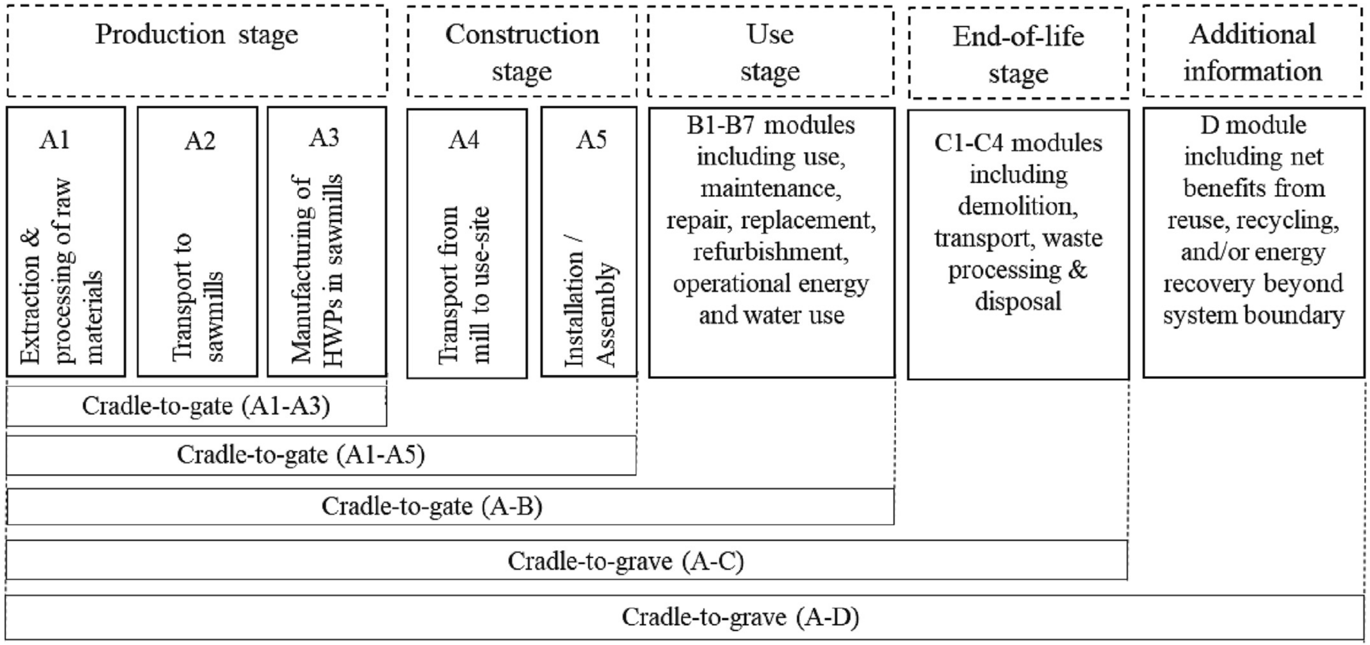

Fig. 1. Life stages for LCA of building and construction products (adapted from ISO 21930, 2017).

Table 1
Global warming potentials (GWP) of GHGs for different time horizons based on the IPCC AR6 WG I report (Forster et al., 2021).

GHGs	Lifetime (year)	20 years	100 years	500 years	GWP 500 used in this study
CO_2		1	1	1	1
CH_4	11.8 ± 1.8	82.5 ± 25.8	29.8 ± 11	10.0 ± 3.8	7.9
N_2O	109 ± 10	273 ± 118	273 ± 130	130 ± 64	132

Note: All GWP values include carbon cycle responses and are with tropospheric adjustments for radiative efficiency of the GHGs.

of long-lived HWPs (Collinge et al., 2013; Ganguly et al., 2020). Two types of dynamic LCA (dLCA) approaches are discussed in the literature: 1) dLCA applied at inventory step with static CF in impact assessment step (fixed time horizon), and 2) dLCA including time dependent inventory and impact assessment with a fixed endpoint of time horizon. Fig. 2 illustrates these two approaches using a product of 50-year lifespan assessed over a time horizon of 100 years.

In Fig. 2a, dLCA is applied at inventory step with static CF in impact assessment (LCIA) step. And the emissions occurring at year 25 and 50 are evaluated up to year 125 and 150, respectively, the time period covered by the assessment (LCIA step) in estimating GWI exceeds 100 years. So, the LCIA results of two products having different time profile of emissions would not be comparable even with the same time horizon for assessment (e.g., 100 years). The second approach (Fig. 2b) avoids this inconsistency by defining a fixed endpoint of time horizon to better represent the effect of timing of GHG emissions. This approach combines the dynamic LCI and dynamic LCIA with development of time-dependent CFs (Bergman, 2012; Breton et al., 2018; Dyckhoff and Kasah, 2014; Levasseur et al., 2010). Although GHGs emissions would have lingering effects beyond 100 years, but because of the 100-year temporal boundary, GWI after this point would not be considered (grey part in Fig. 2b). This allows a time consistent comparison of two or more products or scenarios for a given time

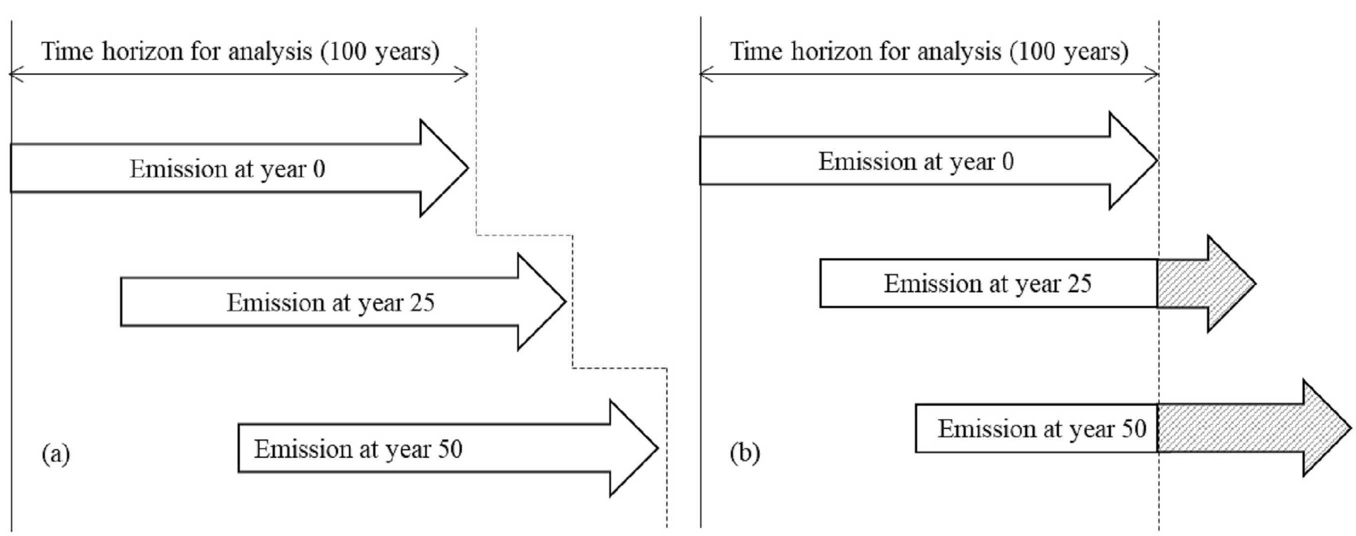

Fig. 2. Considerations on period of assessment selected for embodied carbon assessment of a product (50-year lifespan) evaluated using: (a) fixed time horizon (b) fixed endpoint on the time horizon. Adapted from Levasseur et al. (2010).

horizon. Another advantage of this approach is that course of LCIA results (GWI) over time can be evaluated for two or more products or scenarios.

The primary goal of this study was to perform a dynamic LCA for accounting EC of the HWPs produced and used in California covering stages A through C (Fig. 1). Dynamic GWI assessment of HWPs was performed considering four end-of-life (EoL) scenarios (discussed in the next section). A comparison of the estimated GWI impact with both static and dynamic approaches was made to understand the differences in the estimated results.

2. Methods

This study performed static and dynamic LCAs covering cradle-to-grave life-cycle stages to assess the EC associated with the 2019 timber harvests volumes and related wood products manufactured, transported, used, and disposed in California. In doing so, the volumes of harvested timber were first converted into five different HWP classes (softwood lumber, non-structural panels, softwood plywood, other industrial products, and softwood utility poles), commonly processed and manufactured in California, utilizing timber product ratios and primary products ratios (Table S1) provided in the California forest ecosystem and HWP carbon inventory report (Christensen et al., 2021). Next, the quantities of each class of HWP going to different end-uses were estimated based on data from the same report which was based on McKeever (Mckeever, 2009; McKeever and Howard, 2011) and McKeever and Howard (2011) which is a national data set used for all National Forest Service regions for the distribution of primary products to end uses for all regions. The mean life of wood products remaining in these end-uses varied from 9 years to 144 years (Table S2). For each end-use system, a first order (exponential) decay function was used to estimate annual discards of wood products for EoL treatment (Skog, 2008). The IPCC Guidelines for National Greenhouse Gas Inventories for 2019 provided basis for this assumption (Rüter et al., 2019).

All emissions occurring in A1-A4 modules were assumed to occur in year zero, the initial year of the study. Emissions occurring in A5, B, and D stages (Fig. 1) were not considered. GHG emissions were estimated and tracked for the EoL (stage C) of the HWPs. Emissions associated with wood discarded in each future years (C1-C4 modules) were accounted and tracked up to 500 years. A 500-year time horizon was used to integrate the radiative forcing caused by GHG emissions releasing from different life stages. This large time horizon was chosen to allow for an approximately complete removal (95 %) of HWPs from their useful service lives, which could take very long time given the exponential rate of wood discards. In addition, the choice of longer time horizon allowed seeing the intensity and timing of GHG emissions through the HWPs life-cycle stages, the time those GHGs remain in the atmosphere, and the long-term climate impacts (Hawkins et al., 2021; De Rosa et al., 2018) of timber harvested in 2019. The sections below describe the dynamic LCA framework used in the study, and then how the dynamic inventory of GHG emissions from different stages of the HWP life-cycle was created.

2.1. Dynamic LCA framework

The GWI of each year's GHG released into the atmosphere during the life of a HWP was estimated for a 500-year time horizon using Eq. (1). This estimation was based on scientific model adopted by the IPCC sixth assessment report (Forster et al., 2021) and described by (Gasser et al., 2017; Joos et al., 2013).

$$GWI_i(t) = \underbrace{\frac{AGWI_i(t)}{AGWI_r(t)}}_{(A)} = \underbrace{\frac{\int_0^t [RF_i(t)\, dt}{\int_0^t [RF_r(t)\, dt}}_{(B)} = \underbrace{\frac{\int_0^t RE_i [C_i(t)\, dt}{\int_0^t RE_r [C_r(t)\, dt}}_{(C)} \quad (1)$$

Part A of Eq. (1) denotes Aggregated Global Warming Impact (AGWI, W m^{-2} year kg^{-1} GHG) for time-horizon; where *i* is the emitted GHG (CO_2, CH_4, and N_2O) and *r* is reference GHG (CO_2). The part B expands the equation further indicating that AGWI is the integrated radiative forcing (RF) for the chosen time horizon (*t*). The term RF is expanded in part C. The RF for a GHG *i* is estimated by multiplying its radiative efficiency (RE_i) in the atmosphere and its concentration (C_i) emitted at a time *t*. The components of part C of the Eq. (1) for three GHGs are further described below.

The radiative efficiency (RE, W m^{-2} kg^{-1} GHG) is the radiative forcing per unit mass of the GHG in atmosphere. The RE value of CO_2, CH_4, and N_2O is 1.33×10^{-5} W m^{-2} ppb^{-1}, 3.88×10^{-4} W m^{-2} ppb^{-1}, and 0.0032 W m^{-2} ppb^{-1}, respectively. Therefore, to convert the RE value from Wm^{-2} ppb^{-1} to mass unit (W m^{-2} kg^{-1}), it is multiplied by (M_a/M_i) x (10^9/T_m) as shown in Eq. (2).

$$RE_i\ (kg) = RE\ (ppb) \quad \frac{M_a}{M_i} \quad \frac{10^9}{T_m} \quad (2)$$

Here, M_a = mean molecular weight of air (28.97 kg mol^{-1}); M_i = molecular weight of GHG (kg mol^{-1}); T□ = total mass of the atmosphere (5.1352×10^{18} kg^{-1}).

For the RE of CH_4 in Eq. (2), a correction factor of 0.54 was applied to include the indirect radiative forcings of CH_4 on production of ozone (+0.39) and stratospheric water (+0.15) (Etminan et al., 2016). So, the direct effect of CH_4 was increased by 0.54. For RE of N_2O, a correction was made to include the atmospheric reduction in CH_4 (−36 molecules of CH_4 per +100 molecules of N_2O) (Etminan et al., 2016; Forster et al., 2021).

The C_i(t) in Eq. (1) represents atmospheric mass of a GHG (kg) decayed at time *t* following its emission at time 0. For CH_4 and N_2O, the C*i*(t) is described by first-order decay Eq. (3).

$$c(t) = \mathrm{e}^{\quad t/\tau_i} \quad (3)$$

Here τ is the perturbation time (years) for the GHG emission to decay 1/e times the initial pulse of emission. The perturbation time of 11.8 years for CH_4 meant that after this time released CH_4 had reacted with OH (hydroxyl) radicals to get oxidized into CO_2 and water (H_2O). Therefore, relative impact caused by CH_4 changes over time in comparison to that of CO_2. The perturbation time for N_2O was 109 years. For CO_2, the C*i*(t) is described by Eq. (4) using a background concentration of 399 ppm (Etminan et al., 2016) where concentration of CO_2 decayed in the atmosphere following a pulse emission is estimated by sum of exponentially decaying functions for different carbon sinks such as oceans and plants. The equation also includes an infinite fraction (a_0) which remains in the atmosphere. The τ is lifetime (years) of CO_2 perturbation in those sinks.

$$c(t) = a_0 + \Sigma_{i=1}^{3} a_i \quad e^{\quad t/\tau_i} \quad (4)$$

Here, a_0 = 0.2173; a_1 = 0.2240; a_2 = 0.2824; a_3 = 0.2763; τ_1 = 394.4 years; τ_2 = 36.54 years; τ_3 = 4.304 years.

AGWI (part A, Eq. (1)) estimated with this method is also referred to as cumulative radiative forcing (CRF) or dynamic characterization factor (DCF) for a unit GHG pulse emission since its release at time 0. Instantaneous value of this characterization factor needs to be estimated per time step (Eq. (5)) to understand at course of impact over the time horizon (Levasseur et al., 2010).

$$DCF_i(t)_{instantaneous} = \int_{t\quad 1}^{t} RE_i [C_i(t)\, dt \quad (5)$$

With $DCF_{instantaneous}$ values estimated for every year, we could calculate the global warming impact (GWI) by multiplying with the dynamic inventory of GHG emissions per HWPs life cycle (Eq. (6)).

$$GWI(t) = \Sigma_i GWI_i(t) = \Sigma_i \Sigma_{j=0}^{t} [g_{i\ j} * [DCF_{i\ t\quad j} \quad (6)$$

Here, GWI(t) = global warming impact for the time horizon studied; g = dynamic inventory result for a GHG; i = each GHG in the dynamic inventory; DCF = instantaneous dynamic characterization factor; j = any point of time from 0 to 500.

2.2. Dynamic inventory of GHG emissions by HWPs life cycle stages

In dynamic LCA, the emissions were distributed over the HWPs lifecycle from forest harvesting to the EoL of the products is shown in Fig. 3. Landfilling and burning were the two options considered for the EoL of the HWPs. In the PNW region, forest harvesting is mainly ground-based (56 %) and cable-based (44 %) (Oneil and Puettmann, 2017). Around 11 million m^3 of wood was harvested in California in year 2019 (Christensen et al., 2021). Ecoinvent and DATASMART (US-EI 2.2) databases available in SimaPro software (9.3) were used in the study (LTS, 2021; Pre Consultants, 2021; Wernet et al., 2016). The LCI processes from these databases were used to disaggregate the GWI values (CO_2e) into kg of GHG emissions (CO_2, CH_4, N_2O). PNW forest harvesting (A1) emissions included all forestry operations (site preparation, thinning, fertilization, and harvesting), and raw material extraction processes for the inputs, and were modeled using the LCI unit process available for PNW region “softwood logs with bark, average intensity harvesting” (Table 2) comprising low, medium, and high intensity harvesting of 42 %, 46 %, and 12 %, respectively. These operations also included the cradle-to-gate productions of fertilizers and fuels consumed. Around 68 % of the harvested wood (i.e., log) was assumed to be transported 110 km in trucks to sawmills (A2) with the remaining 32 % left onsite (i.e., logging slash or forest residues).

The distance between harvesting sites and sawmills was an average transportation distance reported by several studies from the PNW region (Johnson et al., 2005; Mason et al., 2008; Oneil and Puettmann, 2017; Puettmann, 2020). The GHG emissions from the combustion of diesel fuel were modeled based on a truck transportation unit process available in US-EI database. The carbon stored in forest residues (32 %) left on forest site after harvest was considered emitted in the first year of harvest, so there were no GHG emissions estimated for the use of residues. The GHG emissions from HWPs manufacturing in the mills (A3) were modeled based on the information from LCA studies specific to the PNW region or from the US aggregate studies where no region-specific LCA studies were available (Table 2). Both static and dynamic LCI of GHG emissions were prepared considering year 2019 as the first year of study or accounting, i.e., $t = 0$ (Table S3 and S4). For all the HWP categories, the GHG emissions inventory was obtained from databases as shown in Table 2. Non-structural panels category comprised a combination of particleboard (55.8 %), medium density fiberboard (MDF 38.5 %), and hardboard (5.7 %) (Brandeis et al., 2021). For this study, GHG emissions of wood pallets (Alanya-Rosenbaum et al., 2021) were included in ‘other industrial products’ category.

According to the 2019 California HWP carbon inventory report (Christensen et al., 2021), 71 % of logs received by the sawmills were transformed into one of the primary wood products, while 29 % were assumed to be used for on-site energy use. The emissions from wood boiler system were based on a LCA study of wood-fired boilers from Consortium for Research on Renewable Industrial Materials (Puettmann and Milota, 2017). Manufactured HWPs from mills were assumed to be transported to the end-use sites (A4) such as construction, installation, assembly, or other use within 500 km using trucks (Bergman et al., 2013; Han et al., 2015; Sahoo et al., 2019, 2021). Modules A1 to A4 emissions were assumed to occur in the first year of the HWPs life cycle, which meant that the impact

Table 2
LCI database unit processes used for HWPs GHG emissions inventory.

HWP life stage	LCI unit processes
Harvesting	Softwood logs with bark, harvested at average intensity site, US PNW NREL, m^3/US
Transportation	Transport, combination truck, diesel powered, tkm/NREL/US
Manufacturing of HWPs	
Lumber	Softwood dry rough lumber, at kiln, US PNW NREL/kg/US
Plywood	Softwood plywood, at plywood plant, US PNW/kg/US
Utility poles	Softwood utility pole, PCP treated/m^3/RNA
Non-structural panels[a]	Particleboard, average, softwood, particleboard mill/m^3/RNA Medium density fiberboard (MDF), at MDF mill/m^3/RNA Fiberboard hard, at plant/m^3/US- US-EI
Other industrial products	Pallet/m^3/US- US-EI based on (Alanya-Rosenbaum et al., 2021)

PNW = Pacific Northwest; NREL = National Renewable Energy Laboratory; PCP = Pressure-treated pentachlorophenol; RNA = North America region; US-EI = US-Ecoinvent.

[a] The GHG emissions of non-structural panels represented the 2019 production share including particleboard, MDF, and hardboard in 55.8 %, 38.5 %, and 5.7 %, respectively.

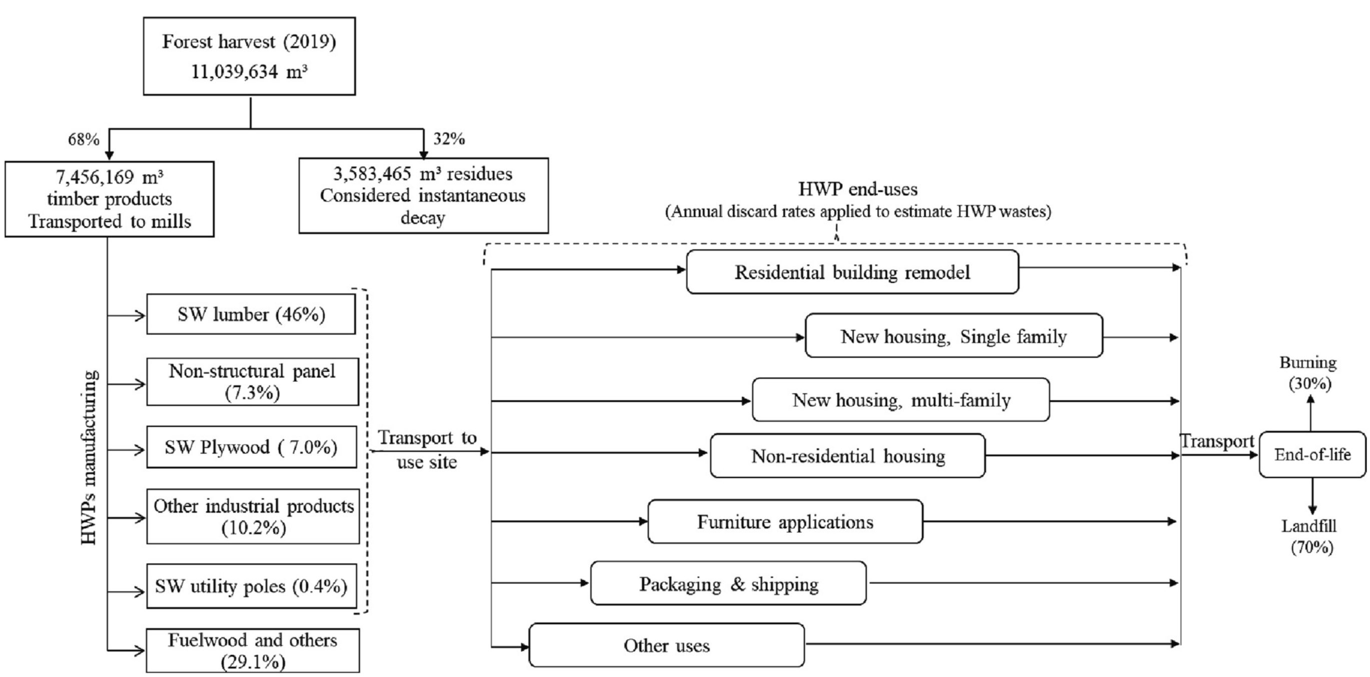

Fig. 3. Covered harvested wood products system based on California HWP carbon inventory for 2019 reporting period (Christensen et al., 2021).

of these emissions was assessed throughout the time horizon (up to 500 years).

The GHG emissions associated with the construction, installation, or any required assembly of the HWPs (A5) were not included because of the lack of sufficient data available for all HWPs going into different end-uses. GHG emissions from HWPs in use stage (B), including maintenance, repair, refurbishments etc., are not included in the study as these use modules are shown to have insignificant environmental impacts compared to the whole life-cycle (Dixit, 2019). However, the amount of HWPs discarded from use each year was estimated using first order (exponential) discard rates for different end-uses. Table S2 provides end-use types for different HWPs, their lifespan, and annual discard rates (based on the California's HWP carbon inventory report) (Christensen et al., 2021). Discard rates refer to rate at which wood leaves the end-use site for a particular HWP. The discarded quantities of HWPs were sent to EoL (C) including modules of demolition (C1), transportation (C2), waste treatment (C3), and final disposal (C4). No GHG emissions were considered for the C1 and C3 modules. The transportation of discarded HWPs to the disposal sites (C2) was considered to occur within a 50 km distance given most use sites are within the urban environment along with disposal sites. The emissions resulting from the transportation of the discarded wood products were estimated.

The national data on EoL disposal options for wood included landfilling (67 %), burning (16 %) and recycling (17 %) (US EPA, 2019). However, for California's wood waste disposal options, a recent study by Cabiyo et al. (2021) considered landfilling (65 %), burning (25 %), and uncollected waste (10 %). This study evaluated four scenarios representing different combinations of landfilling (70 %) and burning (30 %) options. Table 3 provides details of the four EoL disposal scenarios evaluated in this study. A scenario was defined depending on whether landfill gas was captured, and energy was recovered. In two EoL scenarios (1 and 3), energy recovery from burning of wood waste was included in the form of electricity generation (Xu et al., 2021), where the amount of electricity generated from HWP waste burned yearly (30 %) was estimated using factor 0.882 kWh/kg of woody biomass (oven-dry basis) (Xu et al., 2021). Emissions saved from energy recovery were estimated based on the GHG emissions per kWh of electricity in California electricity mix. However, in all evaluated scenarios, energy recovery from landfill gas was not included mainly due to the lack of available data. GHG emissions from landfilling and combustion of wood waste were based on Waste Reduction Model (WARM) of U.S. EPA (US EPA, 2020).

According to WARM, the anaerobic conditions in the landfills lead to decomposition of around 12 % of the carbon contained in the wood with the remaining non-decomposed fraction (88 %) representing a permanent carbon sink. A first-order model was used to estimate the gas quantity generated from decomposable portion (12 %) of wood waste (Micales and Skog, 1997; Wang et al., 2020). The generated landfill gas from wood consists of 50 % CO_2 and 50 % CH_4. Around 10 % of the CH_4 gets oxidized into CO_2 as it reaches the surface, and the estimated composition of landfill gas at surface become 55 % CO_2 and 45 % CH_4 (Salazar and Meil, 2009). While CO_2 emitted was considered a part of natural carbon cycle (not included in current analysis), the CH_4 was counted as an anthropogenic GHG because it is not generated in natural decomposition (US EPA, 2020). Although energy recovery from landfill gas was not included, landfill gas capture and flare were included in two scenarios (baseline and scenario 3) to gain a better understanding of the effects of such practice on the estimated GWI.

3. Results and discussion

3.1. Metrics of dynamic global warming impacts

The cradle-to-grave GWI of HWPs with baseline scenario in EoL was analyzed using dynamic LCA framework. Fig. 4 represents the estimated three metrics of dLCA for each GHG (CO_2, CH_4, N_2O), resulting from wood products harvested (and subsequently transported, processed, used, and disposed) in California over a time horizon of 500 years. These metrics are (i) mass of GHGs remaining in the atmosphere (kg); (ii) integrated radiative forcing caused by the GHGs (W-year m^{-2}); and (iii) aggregated (cumulative) GWI of the individual GHGs (W-year m^{-2}). The representation of metrics for each GHG allowed us to see how the modeled effects vary over time.

3.1.1. Mass of GHGs remaining in the atmosphere

In case of CO_2 (Fig. 4a), the mass remaining in the atmosphere at any time was higher than that of CH_4 (Fig. 4b) and N_2O (Fig. 4c). This could be explained by more complexity in the decay model of CO_2 which involves interactions of three differently timed absorptions by three sinks and a constant part (a_0, Eq. (4)). However, the abundance of CO_2 decreased rapidly, and the rate of decay declined towards a long-term equilibrium almost never reaching zero.

3.1.2. Integrated radiative forcing

Figs. 4(d, e, f) show the integrated radiative forcing (IRF) effect caused by the mass of GHGs present in the atmosphere over the time horizon of 500 years. Because the radiative forcing effect is concentration-dependent, the trends observed in IRF were like the trends of GHG abundances (Fig. 4a-4c). The GHG emissions from modules A1 to A4 were assumed to occur in year 0 and therefore contributed significantly to IRF, since their impact persisted throughout the period of assessment. The IRF of CH_4 (Fig. 4e) was high because a large fraction (70 %) of discarded HWPs after use was going into landfills where methane gas causing high radiative forcing was emitted. Considering longer time horizon, this outcome indicates substantial contribution to GWI from EoL options. The influence of EoL emissions would be reduced with a shorter period of assessment, therefore a longer time horizon is recommended for HWP LCAs (Cardellini et al., 2018; Dyckhoff and Kasah, 2014). The shorter period of assessment here refers to a period less than the maximum mean life of HWPs included in this study. For example, the HWP having mean life of 144 years would have a reduced influence of EoL emissions in the GWI if the assessment period was 100 years.

3.1.3. Aggregated global warming impact

Fig. 4 (g, h, i) represent the amount of additional heating effect per unit area that would occur as a result of the IRF caused by each GHG up to a particular year on the time horizon. The AGWI of CH_4 (Fig. 4h) initially increased rapidly, but the rate of increase slowed with the decay of remaining CH_4 and there was no significant increase after around 150 years (time when mean life of all HWPs was ending). There was a similar trend observed for N_2O as well (Fig. 4i). In contrast, the rate of increase of AGWI for CO_2 (Fig. 4g) was almost linear as the mass of CO_2 in the atmosphere approached equilibrium. Thus, these results support the fact that the time-dependent analysis of the GWI of individual GHG emissions is relevant for long-lived HWPs. Future wood product designs would likely favor

Table 3
Percentages of the wood waste treated by different EoL disposal scenarios analyzed in the study.

Scenario	Landfill without gas capture	Landfill with gas capture and flare	Burning without energy recovery	Burning with energy recovery
Baseline	–	70 %	30 %	–
Scenario 1	70 %	–	–	30 %
Scenario 2	70 %	–	30 %	–
Scenario 3	–	70 %	–	30 %

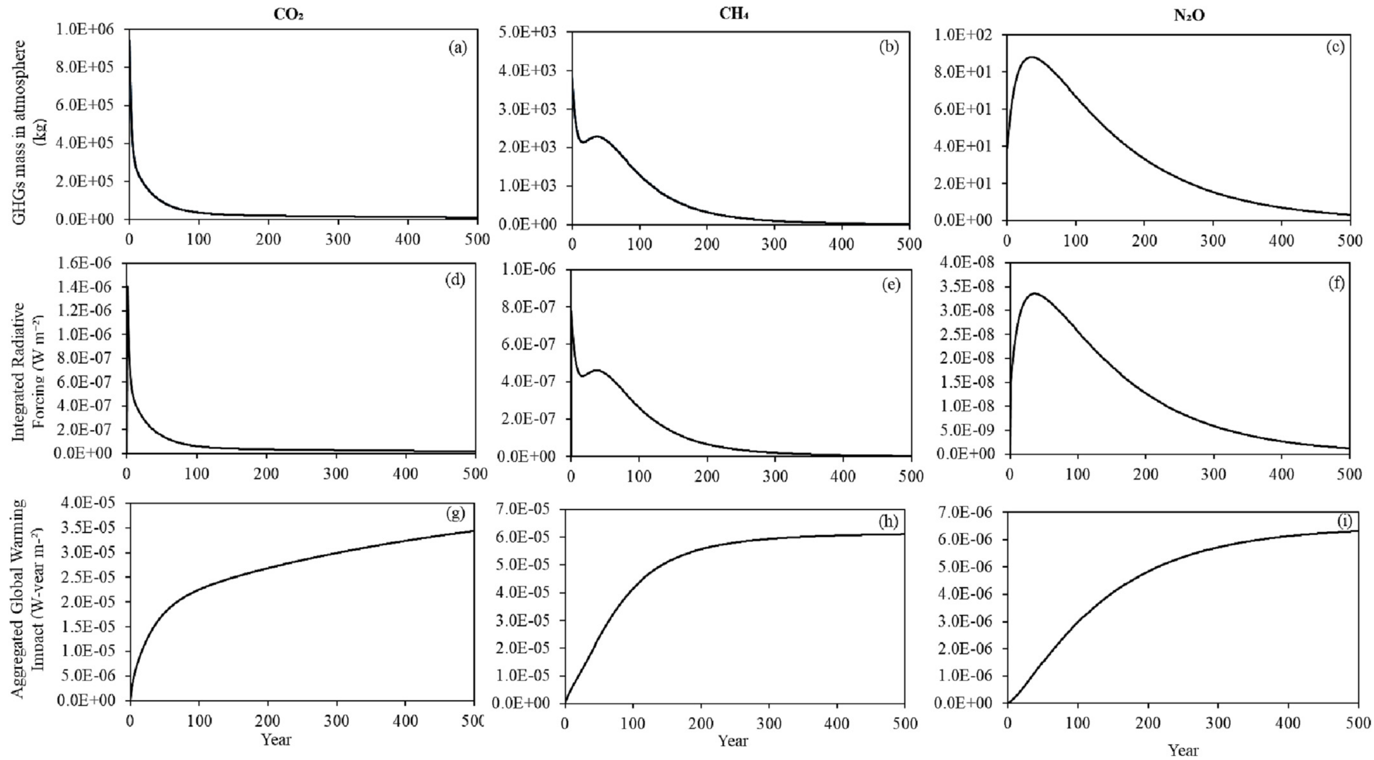

Fig. 4. The estimated GWI metrics of cradle-to-grave dynamic LCA of all HWPs originating in 2019 from California over a time horizon of 500 years.

longer-lived products and/or more sustainable alternatives over short-lived applications.

3.2. Static LCA and dynamic LCA results

The estimated GWIs of California's HWPs with static LCA approach are shown in Table 4. As the timing of the GHG emissions was not considered in the static LCA, only the percent contribution of each module to the total GWI can be reported (Table 4). In all scenarios, A3 module showed the largest among the modules considered in year '0' (A1-A4). The EoL stage contribution was highest in scenario 2 (45 %) where no capture of landfill gas and burning of waste was considered. Contrary to this, scenario 3 which includes landfill gas capture as well as energy recovery from burning has the lowest percent share of EoL stage in total GWI.

Fig. 5(a-d) shows the estimated GWIs of HWPs using dynamic and static LCA approach. The grey and dotted green lines represent the GWIs estimated with dynamic LCA approach. The green dotted lines represent the estimated GWIs of the HWP system at any year 't' within the 500-year time horizon. The grey line shows the contribution of GHG emissions to dynamic GWIs cumulated up to the year 't' to the end of 500-year time horizon. In static LCA, the GWI (black line in Fig. 5) was a single aggregated value characterizing the entire time horizon of 500 years.

Table 4
Static LCA results with percent share of HWP modules in total GWI of different scenarios.

	Baseline	Scenario 1	Scenario 2	Scenario 3
Total GWI (million tonne CO_2e)	1.16	1.54	1.75	0.95
A1	5.0 %	3.8 %	3.2 %	6.1 %
A2	2.8 %	2.1 %	1.8 %	3.4 %
A3	66.1 %	49.9 %	42.9 %	80.8 %
A4	10.2 %	7.7 %	6.6 %	12.5 %
C2	1.0 %	0.8 %	0.7 %	1.3 %
C4 landfill	13.6 %	48.6 %	43.8 %	16.7 %
C4 burning	1.3 %	−12.8 %	0.9 %	−20.8 %

3.2.1. Baseline scenario

Fig. 5a (green and grey lines) shows the dynamic GWI of baseline scenario, which indicates that the dynamic GWIs at year 500, 100, and 20 were 0.31, 0.73, and 0.88 million t CO_2e, respectively. Overall, the GWI showed a downward trend from year 0 to 500. This was mainly because of the assumed landfill gas capture in this scenario which reduced the release of CH_4 emissions. Further, the results clearly show that selection of longer time horizon (500 years) helped in better understanding of the GWI of HWPs.

3.2.2. Scenario 1

The estimated GWI in scenario 1 (Fig. 5b) was 0.85, 2.17, and 1.29 million t CO_2e at year 500, 100, and 20, respectively. The GWI at year 100 was highest because a large quantity of emissions which occurred at year 0 had their radiative forcing causing high GWI around this time. A higher impact near 100 years was attributed to emissions of EoL stage because a large quantity of HWP were going to landfill disposal and burning around that time. Following this, the GWI showed declining trend because of the landfill gas capture in the scenario.

3.2.3. Scenario 2

The cumulated GWI was highest in scenario 2 (0.93 million t CO_2e) because of no landfill gas capture and no energy recovery assumed in this scenario. Like scenario 1, the GWI was high around year 100 because of the high radiative forcing contribution by CH_4 emissions from anaerobic decomposition of wood waste at landfill sites. This was the period (year 45–100) around which 52 % of the HWPs reached their mean service life resulting in more waste (i.e., discarded wood) ending up in landfills.

3.2.4. Scenario 3

The dynamic GWI of scenario 3 were shown to be the lowest (0.27 million t CO_2e) among all scenarios. Evidently, this was due to the assumption that landfill gases were captured and flared, and wood waste was burned to generate energy. As a result of the emissions savings from generated electricity, the dynamic LCI showed negative emissions, resulting in a reducing

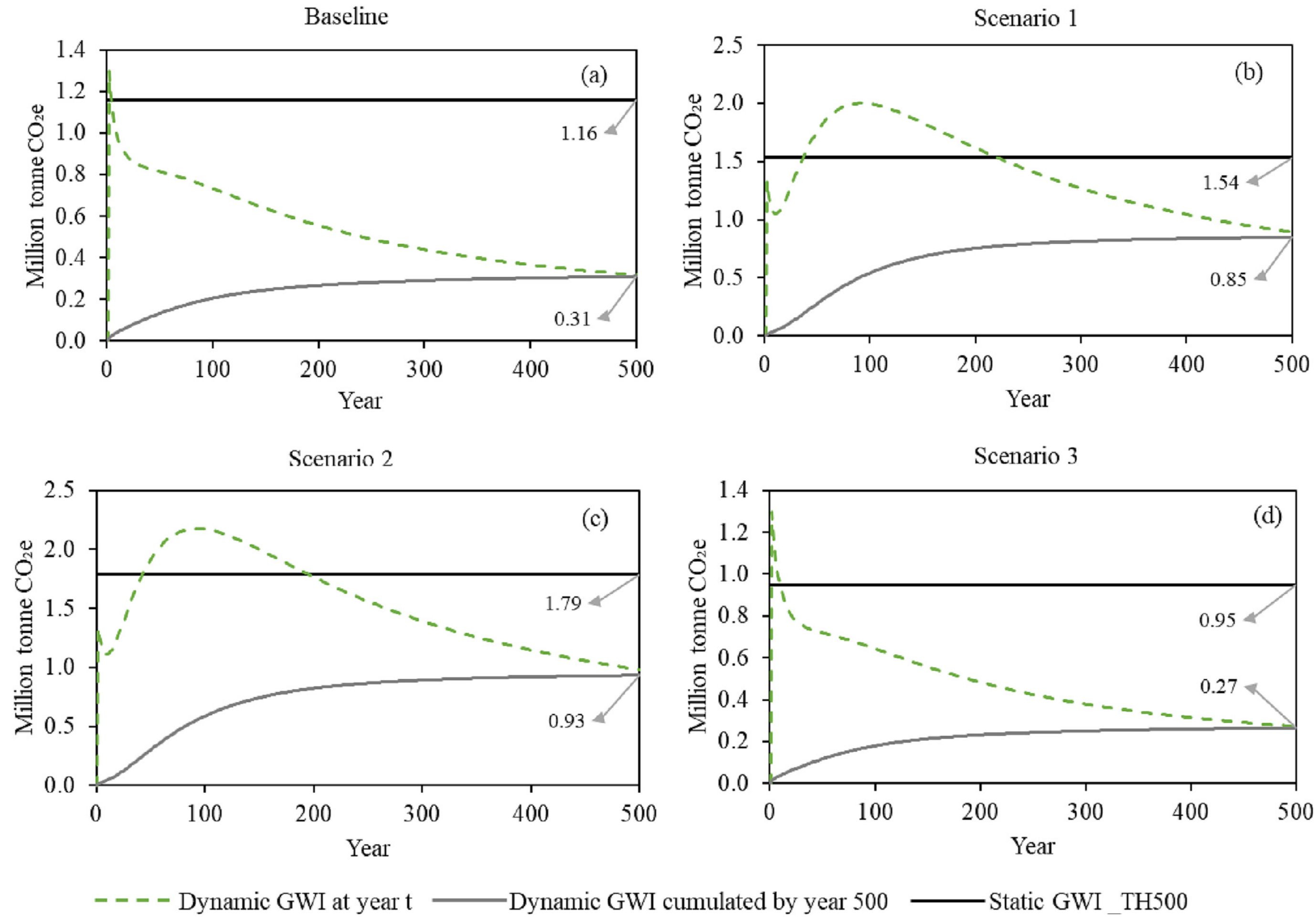

Fig. 5. Comparison of dynamic and static GW impact assessment results for a state-level analysis of California.

warming effect. The GWI could further be lower if energy recovery from captured landfill gas was considered.

As clearly seen in the Fig. 5 (a-d), the dynamic GWIs were significantly lower than those estimated with the static LCA in all scenarios of the EoL stage. The observed rationale behind this difference was time considerations for GHG emissions and their atmospheric decay based on the time spent during the horizon of assessment in dynamic LCA. Since wood was removed from use annually, landfill gas was generated annually until the end of the time horizon (due to exponential discard rate). Whereas in the static LCA approach, timing of the emissions was not considered, and all emissions were assumed to occur at time '0'.

Similar results can be found in recent LCA studies on wood products. A study on dynamic GHG effects of wood-based panels industry (Wang et al., 2020) showed how static LCA approach can underestimate the emissions reduction potential of forestry sector and how dynamic LCA is more relevant in assessing climate mitigation potentials. Another study on wood cork (Demertzi et al., 2018) showed contrasting differences in static and dynamic LCA results depending on whether the biogenic carbon uptake was included. In contrast, an LCA study on glulam reported higher GWI impacts from dynamic LCA compared to static LCA (Cardellini et al., 2018). However, this study considered different EoL disposal scenarios (35 % burning with energy recovery, 33 % recycling, and 32 % landfilling with no capture of emitted methane).

In the present study, it was interesting to note that, the dynamic GWIs were higher in the period 45–200 years and then significantly lower as compared to those of static LCA in scenario 1 (Fig. 5b) and scenario 2 (Fig. 5c). The major similarity in these two scenarios was no saving of GHG emissions considered from the capture of landfill methane from HWP waste. Lower dynamic GWIs in baseline scenario and scenario 3 throughout the time horizon were attributed to annual GHG emissions savings from landfill methane gas capture and additional negative radiative forcing (i.e., higher positive outcome) from electricity generation in scenario 3. Overall, the dynamic LCA approach could play an important role in decision making by improving the understanding of the sensitivity of conclusions to given time horizon and choices in EoL stages.

The differences in the estimated dynamic and static GWI values suggest considerable implication for estimating the potential substitution benefits of wood products over alternative non-wood products. Most of the studies reporting substitution benefits of using wood products in place of alternative non-wood products (e.g., Leskinen et al., 2018; Sathre and O'Connor, 2010), are based on static LCA results. However, as demonstrated above, a static approach could result in underestimation of the substitution benefits. With the help of a time-dependent assessment of the GWIs like in this study; it would be possible to estimate more precise substitution benefits of HWPs.

3.3. Influence of end-of-life stage on dLCA results

This section discusses the influences of including EoL stage on the dynamic LCA results. Fig. 6a shows dynamic GW impacts associated to A1-A4 modules of HWPs. Since A1-A4 modules occurred in the first year of the study, all the radiative forcing was caused in the beginning of time horizon and then decayed over time. The GW impacts in Fig. 6 (b) were associated to GHG emissions of A1-C4 modules with baseline scenario at the EoL. The GWI trajectory illustrates the influence of including GHG emissions from EoL life stages. A broader scope of assessment, therefore, can provide a better understanding of the overall embodied carbon of the HWPs, especially when comparing against alternative products.

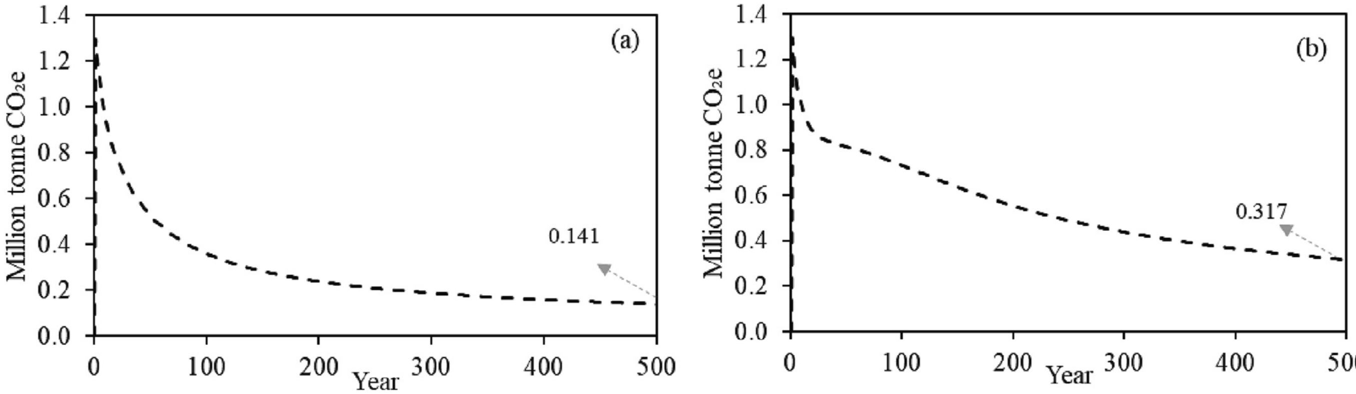

Fig. 6. State-level dynamic GW impacts for the Californian HWP system pertaining to (a) A1-A4 modules (b) A1-C4 modules including baseline scenario for disposal at end-of-life.

This analysis demonstrated that the EoL option directly determines the mass of GHGs released into atmosphere and subsequently their warming effect depending on the time they spend in the horizon of assessment. Though EoL scenarios can be subjected to uncertainties, but a yearly profile of impacts can assist in identifying when and where in the life cycle stages, impacts reach peak levels, and how they vary as EoL scenarios change. So, it is only in the longer time horizons, that the effects of EoL options became more evident. In future studies, considering the recycling and composting options and evaluating the emissions trade-offs in future studies would help in gaining a deeper understanding of the potential GW impacts of all possible EoL scenarios.

4. Limitations and future works

This paper presented temporal effects of GHG emissions with a cradle-to-grave system boundary of HWPs. Although several useful inferences were drawn from the results, there were some limitations that can be used as a basis for future research. These include: 1) the dynamics of carbon emissions and savings as a result of utilizing forest residues as a feedstock for either energy or wood-based products were not considered; 2) GWIs due to biogenetic carbon stored in HWPs over the time horizon were not calculated. Inclusion of biogenic carbon would have resulted in lower estimated GWIs than reported in the study; 3) the current analysis is based on the exponential function and associated half-lives in estimating HWP discard rates, which suggest faster decay rates in earlier years. However, there has been growing criticism that the exponential function may not be appropriate to model discard rates of long-lived wood products as most of these products go out of use in later years than earlier years; 4) the analysis considers limited sets of EoL scenario. Additional scenarios combining landfilling, burning, recycling, composting and energy recovery could provide a more complete picture of the GWIs of HWPs; 5) an area of future research could be studying cascading utilization of wood wastes before final disposal. Lastly, 6) the background temporal variations in the CF used in LCA studies which provided GHG emissions inventory for inputs like grid electricity were not considered. A continued progress in data quality and additional research in future works would lead to even greater accuracy in climate impact assessments of HWPs. In addition to these limitations, future studies can improve on the data availability for construction, installation, or assembly phase.

5. Conclusions

This study presented a state-level and time-dependent embodied carbon assessment of HWPs, using the case of wood products harvested in 2019, and subsequently processed, manufactured, used, and disposed of in California. A cradle-to-grave life-cycle inventory with time profile of three GHG emissions (CO_2, CH_4, N_2O) during HWPs life-cycle was evaluated over a long horizon of 500 years. It was found that dynamic LCA approaches help to better understand different GW metrics (mass of GHGs, integrated RF, and GW impacts) over time, while static LCA approaches only report aggregated GWIs. Such time-dependent analysis of the GWIs of individual GHG emissions is specifically relevant for long-lived products. The key finding of the analysis was that the estimated GWIs with two approaches can differ greatly depending on the scope of assessment and selected EoL options. Among the evaluated EoL scenarios, the dLCA results were most favorable for scenario 3 that considered energy recovery from burning wood waste and landfilling with collection and flaring of the landfill methane gas. The results indicated that static LCA could underestimate the substitution benefits of avoided GHG emissions from using wood products in place of non-wood materials. Despite certain limitations discussed above, the dynamic LCA approach used in this study would be useful to natural resource managers in California and other wood-producing states interested in assessing the relative importance of time-horizon on climate impact mitigation strategies.

CRediT authorship contribution statement

Poonam Khatri: Methodology, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Visualization. **Prakash Nepal:** Conceptualization, Methodology, Formal analysis, Investigation, Writing – review & editing, Supervision. **Kamalakanta Sahoo:** Methodology, Formal analysis, Investigation, Writing – review & editing. **Richard Bergman:** Conceptualization, Methodology, Formal analysis, Investigation, Writing – review & editing, Supervision. **David Nicholls:** Conceptualization, Investigation, Writing – review & editing, Funding acquisition, Supervision. **Andrew Gray:** Conceptualization, Investigation, Funding acquisition, Supervision.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.163918.

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Corrigendum

Corrigendum to 'California's harvested wood products: A time-dependent assessment of life cycle greenhouse gas emissions' [Sci. Total Environ. 886 (2023) 163918]

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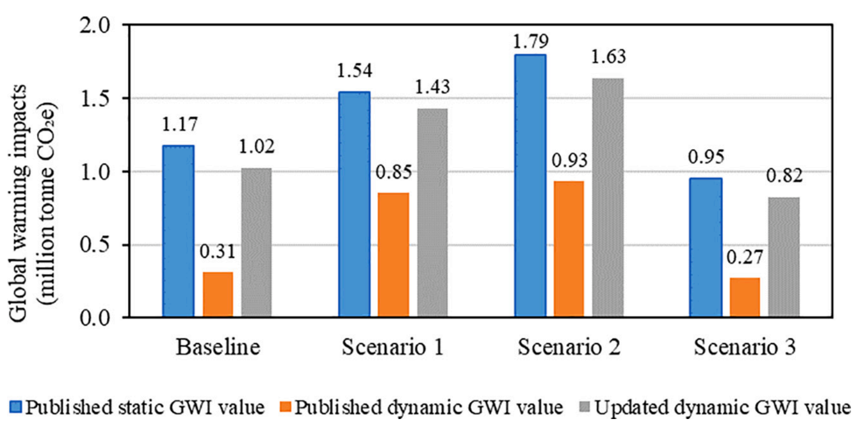
The authors regret that the printed version of the above article contained an error in the calculation of dynamic characterization factor in Eq. (5) and the correction caused an increase in the dynamic global warming impact (GWI) values. The updated dynamic GWI at 500 years are plotted against the published values for all scenarios below:

The value and percentage of GWI in a sentence in the abstract would now be read with updated values like below:

It was found that dynamic GWI for all scenarios ranged from 0.27 to 0.93 (updated values: 0.82 to 1.63) million tonne $CO_{2}e$, which were 45–73 % (updated values: 7–13 %) lower than those estimated with static LCA approach, indicating that the static LCA approach could lead to an underestimation of the benefits of substituting wood for non-wood products, compared to those based on dynamic LCA approach.



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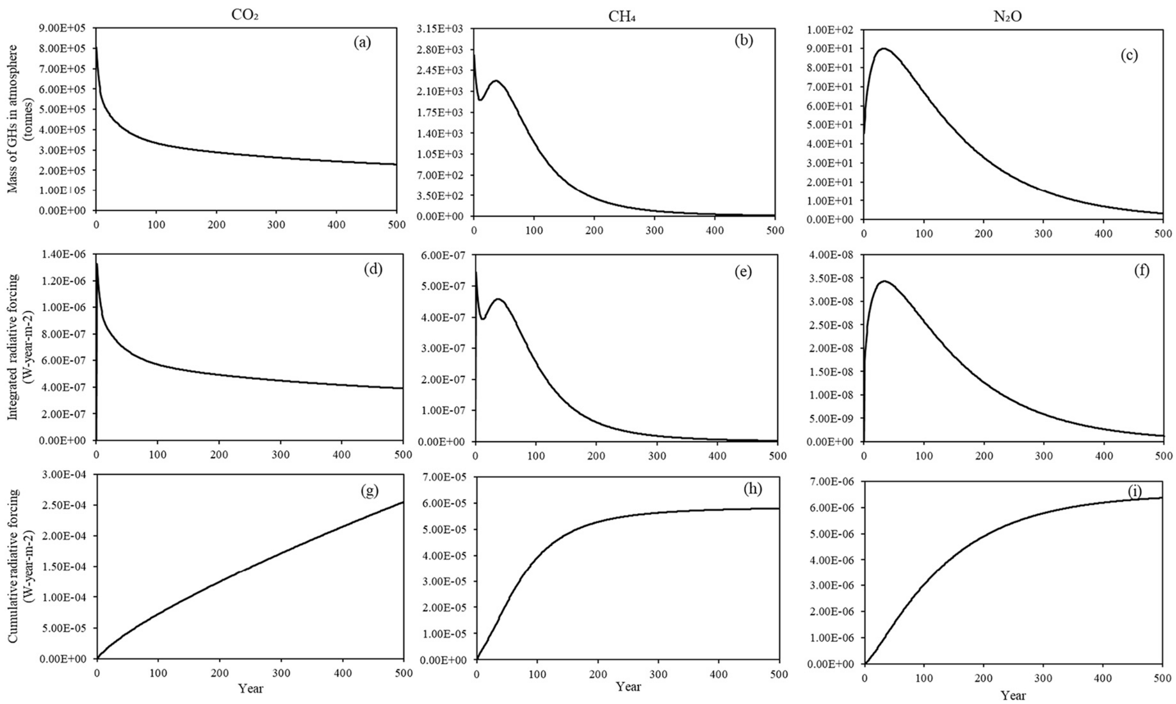

Fig. 4. GW impact metrics of cradle-to-grave dynamic LCA of HWPs over a time horizon of 500 years.

Overall, the updated dynamic GWI value in all scenarios remained less than static GWIs, but the magnitude of values and the percentage differences between scenarios have reduced. Therefore, showing the updated GWI values is of great importance. The dynamic GWI still proves beneficial in the context that it generates more detailed information on the GWI over time. The GWI variations while comparing different scenarios and trends of greenhouse gas (GHG) emissions, trends of radiative forcing all help in overcoming the time-inconsistency of static LCA approach. More details on these global warming metrics are available in the results Sections 3.1 and 3.2 in the article.

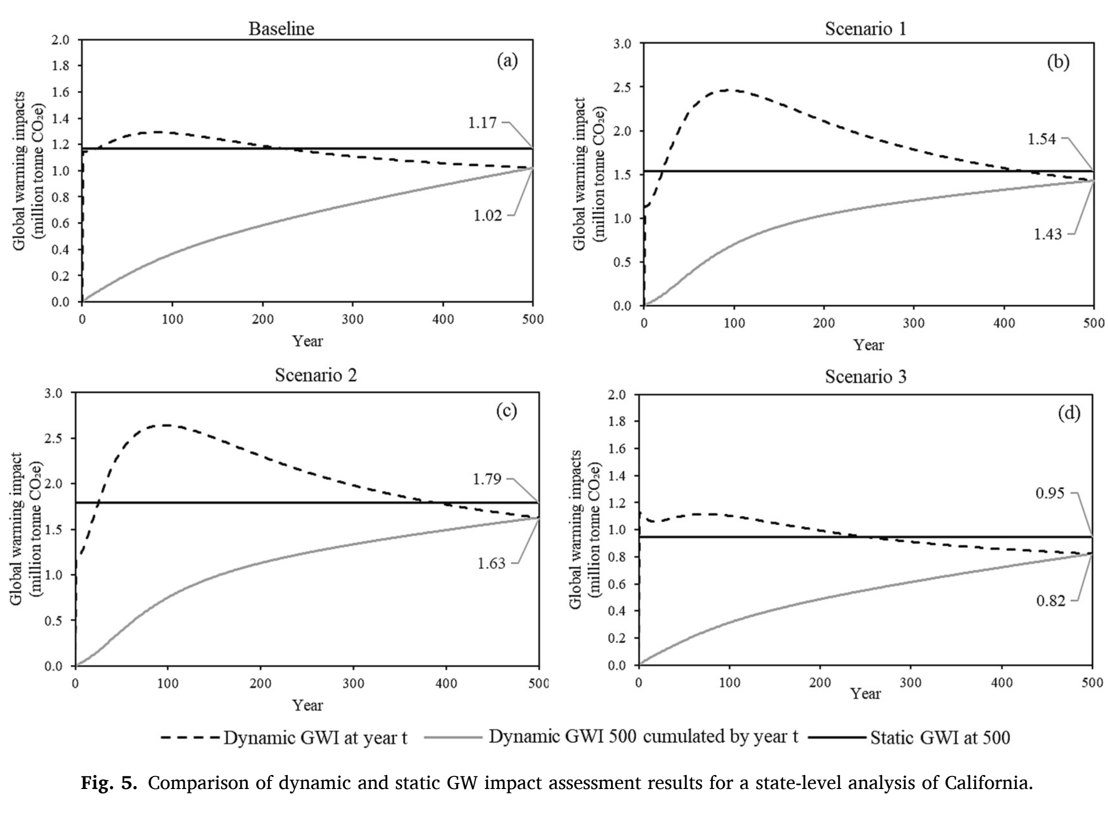

Fig. 5. Comparison of dynamic and static GW impact assessment results for a state-level analysis of California.

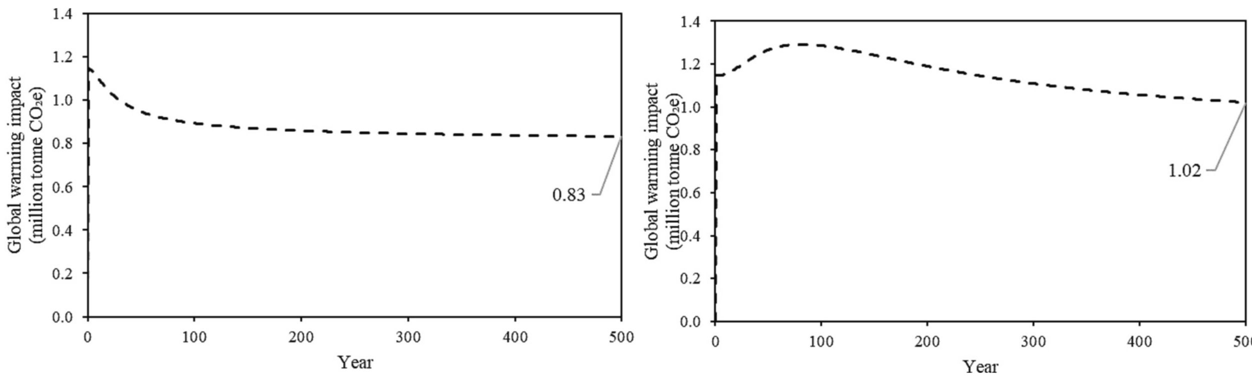

Fig. 6. State-level dynamic GW impacts for the Californian HWP system pertaining to (a) A1-A4 modules (b) A1-C4 modules including baseline scenario for disposal at end-of-life.

The authors would like to apologize for any inconvenience caused.