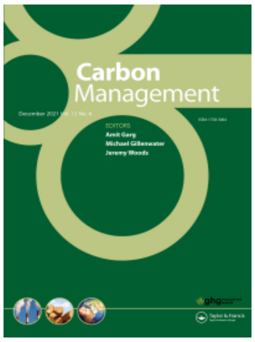

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REVIEW ARTICLE

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Carbon impact of wood-based products through substitution: a review of assessment aspects and future research perspectives in life cycle assessment

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ABSTRACT

Reducing carbon emissions is a top priority for combating climate change, and the use of wood products is one important strategy toward this direction. However, the impact pathways of wood products remain subjective to uncertainties, and there is a lack of consensus over the methodology for assessing impacts. This review focuses on the accounting of benefits, when wood-based products substitute non-wood products. The carbon impact of substitution is measured through the substitution factor (SF), which is derived from a comparative estimation of greenhouse gas (GHG) emissions of wood and non-wood products, using life cycle assessment (LCA). The calculation of SF is influenced by several factors such as system boundaries, functional unit, life cycle stages, product types, substitution assumptions, and end-of-life considerations. This review addresses the previously mentioned challenges and provides a summary of SFs for longer-lived wood products, categorized by product type, system boundary, and country. The findings show that SFs for wood products are higher in construction applications than in interior or furniture uses, with regional variations reflecting differences in the substitution effect. Among product categories, the sawnwood category exhibits the highest SF, followed by engineering wood products and wood-based panels. GHG emissions estimates are sensitive to whether biogenic carbon is accounted for, which in turn influences the respective SFs. Different biogenic carbon accounting methods yield varying outcomes, making this a divisive issue in LCA. Additionally, this review identifies sources of variability and uncertainty in SFs estimation and highlights a range of challenges linked to LCA aspects. Therefore, this review emphasize precautions within the LCA domain to ensure a more realistic estimation of carbon impacts while managing variability and uncertainties.

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Substitution; substitution factors; carbon; wood-based products; life cycle assessment; GHG emissions

Introduction

The reduction of greenhouse gas (GHG) emissions is the fundamental component of mitigating climate change. One of the key strategies to combat climate change is to use wood products due to their carbon storage ability [1] and potential to reduce GHG emissions [2]. Carbon sequestration occurs in forests, and when wood from sustainably managed forests is used in long-lived products, this carbon remains stored for extended periods, contributing to atmospheric CO_2 reduction. Besides carbon storage and sequestration, substituting GHG and energy-intensive materials like steel, concrete, bricks, and fossil fuels [3–7] with wood, can reduce GHG emissions.

The substitution effect of wood refers to the avoided GHG emissions that occur when wood-based products replace carbon-intensive alternatives. These benefits are realized when an increase in wood usage leads to a corresponding decrease in the production and use of non-wood products [8]. In many

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cases, such benefits are amplified when the market share of wood use increases [9] and this applies to both virgin and reused/recycled wood use. The magnitude of substitution benefits is closely linked to both the volume of wood usage and the substitution factor [9], which quantifies the GHG emissions avoided due to wood product substitution [3]. However, the effectiveness of substitution varies widely depending on factors such as product lifespan, material availability, cost, and prevailing market conditions [10–12]. Regulatory frameworks and market structures play a role in shaping the scale and impact of wood product substitution [13–15] and thereby influencing associated carbon emissions [16]. In practice, product substitution is much more complex, shaped by additional factors like the rebound effect and policy schemes [17]. Although substitution factors (SFs) provide insight into potential emission reductions [2], they can also be overestimated [3,11]. Yang et al. [18] point out that substitution can reduce fossil-based emissions, but it may not fully compensate for the biogenic emissions associated with wood use. Moreover, current global industry standards and emissions reporting guidelines do not yet account for avoided emissions [19]. National GHG inventories do not include avoided emissions from substitution due to their hypothetical nature, associated risks of inconsistency and double counting [20]. ISO 13391–3:2025 highlights that substitution potentials are not included in GHG inventory reporting methodology at the organizational level [21].

However, wood substitution could cut global CO_2 emissions by 14–31% and reduce global fossil fuel consumption by 12–19% [22]. The greatest climate benefit comes from substituting GHG-intensive products with wood [23], with substitution providing a permanent impact on CO_2 reduction [24–26]. However, there is still limited understanding of the wood product substitution effects at the market, country, and global level [13,27], and a framework for scaling these effects has yet to be developed [27]. Yang et al. [18] used forest-based functional unit to create an SF database for wood use in the EU, emphasizing that from a market perspective, substitution is more likely to occur at the building level rather than at the wood product level. Leskinen et al. [3] assessed substitution impacts at the market level by comparing the overall production mix of forest products to a mix of competing products, multiplying the product volumes by their respective SFs. In their study focused on the EU, they identified the largest substitution benefits in sawnwood used for construction because of large market volume and relatively large SF of sawnwood. But, it remains unclear whether wood products always replace an opposing fossil product, or whether wood products instead only complement the market, as this is determined by demand rather than supply [11]. ISO 13391–3:2025 also notes that calculating substitution potential relies on counterfactual scenarios and currently does not consider effects of market dynamics and changing consumption patterns or whether displacement occurs [21]. To upscale substitution potential, it is essential to ensure the substitute product provides equivalent functionality, expand system boundaries to capture wider impacts, and incorporate dynamic assumptions and realistic scenarios that consider market factors. This knowledge gap limits efforts to maximize the climate benefits of wood product substitution.

Substitution impacts are typically quantified using life cycle assessment [28], a standard method for evaluating the environmental impacts of products across their entire life cycle [29,30]. As outlined by ISO 14040 (2006) and ISO 14044 (2006) standards, LCA consists of four phases: goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and result interpretation (see Figure 1). In practice, SFs are calculated based on GHG emissions obtained from comparative LCAs of wood products and non-wood alternatives [20,33]. However, variations in system boundaries, substitution assumptions, and timeframes lead to wide variation in SF estimates [34]. The timing of biogenic carbon flows where emissions and sequestration happen at different times complicates evaluations and weakens the precision of impact assessments.

Furthermore, there is no universally accepted method for calculating substitution factors [35]. Approaches range from single SF values and direct comparisons to average, weighted, and unweighted methods, all contributing to the wide variability [20]. Additionally, beyond LCA uncertainties, the actual likelihood of substitution depends on external factors like market trends, consumer demand, and policy incentives [9]. SFs can change over time as wood-based products and the products they replace can evolve in terms of GHG emission profiles [21]. Given the wide range of substitutes, uncertainty in substitution effects is inevitable [12]. Variability in the wood product market [36], and differing market assumptions [37] can contribute to uncertainty in the SFs. Notably, most studies do not apply

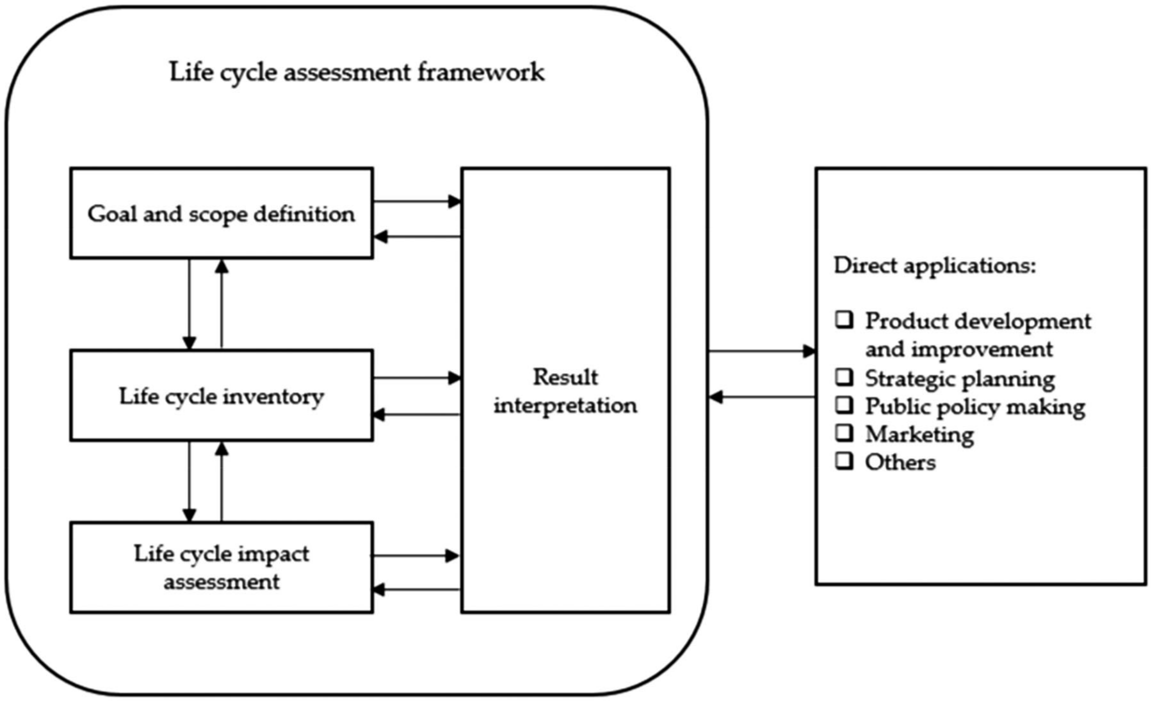

Figure 1. LCA methodology based on ISO 14040 and ISO 14044 [31,32]: framework and applications.

discounting factors to address uncertainty and the temporal aspects of substitution. Recent research highlights the influence of calculation methods on SF values. For example, Han et al. [35] examined three methods for calculating SF (single SF, replacement rate-based SF, and wood-intensity-based SF), showing how the selected method influences SF values, when comparing wood-based furniture to non-wood alternatives. Similarly, Yang et al. [18] applied a unit-weighted SF for intermediate products and a volume-weighted SF for wood products (measured in tC/tC) by employing two complementary methods: a supply-oriented approach and a demand-oriented approach. Further, Schulte et al. [11] analyzed the substitution of non-wood products with wood, and demonstrated that the extent of emission reductions in multi-family housing construction using the timber light-frame system depends on the size of the flats. The calculation of SFs takes into account factors like material weight, carbon content, the frequency with which wood replaces non-wood materials, and GHG emissions per product's functional unit [35]. Inconsistencies in SF calculation methods and units across studies contribute to uncertainty. Previous studies have rarely analyzed gaps in assessments, differences in SFs, sources of variability, and uncertainties. Some researchers recommend applying discounting factors to account for uncertainties and the conditional effects of substitution. For instance, Werner et al. [38] proposed regional and likelihood-based discounting, while Pingoud et al. [39] explored how delayed benefits and methodological issues affect substitution assessments. Valatin [40] recommended accounting for rebound and leakage effects when quantifying the carbon substitution benefits of HWP, especially about reductions in fossil fuel use.

Further research is necessary to include a wider range of wood-based product categories and regions. Profiling product-specific SFs allows for prioritizing wood-based products in combating climate change and promoting sustainability. Addressing the existing gaps, inconsistencies and uncertainties will lead to proposals for improving the ability to quantify the contributions of wood products in meeting the sustainable development goals. Therefore, this review aims to specifically profile SFs and GHG emissions factors associated with wood-based products in the construction and furniture sectors. Additionally, it presents what is known, identifies gaps and uncertainties in LCA. Finally, the review offers precautionary measures from the researcher's perspective to extend knowledge in the LCA domain. The boundaries of this review, as well as some general considerations regarding wood product substitution, are presented below.

Wood product substitution

Wood product substitution is determined by the relative emissions of wood versus the non-wood materials it replaces [41], as well as the likelihood of this substitution effect occurring. In addition to these factors, there are two major sources of uncertainty influencing estimates of substitution impacts. The first source of uncertainty lies in the comparison of the impacts of wood and non-wood products through LCAs, which can vary depending on the products being replaced, their production processes, and carbon footprints [3]. The second source of uncertainty relates to the feasibility of substituting non-wood products with wood in practice. The substitution effect is further constrained by supply and demand dynamics in the wood market [9,11], and shifts in market trends can alter substitution patterns and their impact on emissions [16]. Yang et al. [42] stress the importance of accounting for substitution ratios and market conditions when estimating the effect of wood substitution.

There is already a body of literature on the SFs at the product level, while a review by Hurmekoski et al. [43] estimated the average and range of the potential impacts of large-scale material substitution. However, Hurmekoski et al. [27] stressed that there is no single, established method of determining market-level substitution impact estimates. Sathre and O'Connor [25] reported SFs for wood products ranging from 2.3 to 15.0 tC/tC, with the majority falling between 1.0 and 3.0 tC/tC. These authors estimated a mean substitution effect of 2.1 kg C/kgC wood product. According to Geng et al. [44] each tC in wood used in the furniture sector for material substitution reduces 1.46 tC of emissions [44]. Knauf et al. [26] reported a material substitution of 1.5 tC/tC. Taverna et al. [45] also estimated a material SF of 0.8 tC/tC for Switzerland. Specifically, Leskinen et al. [3] estimated an average SF of 1.3 kg C/kg C for structural construction and 1.6 kg C/kg C for non-structural construction or an average SF of 1.2 kg C/kgC. Cardinal et al. [46] reported a non-weighted average of 0.80 tC/tC for sawnwood and 0.81 tC/tC for wood-based panels (WBPs) or an average of 0.80 tC/tC (sawnwood and WBPs combined). Boiger et al. [47] reported an average SF of 0.4141 tC/tC wood for industries using wood for material applications, excluding wood used for energy purposes, in Austria. Petersen and Solberg's [48] reported that substituting wood for steel avoids 36–530 kg CO_2e per m^3 of timber, while substituting wood for concrete avoids 93–1062 kg CO_2e, provided the wood is not landfilled after use. In Norway, using glulam in place of steel avoids 0.24–0.31 tCO_2e/m^3 of sawn wood input, increasing to 0.40–0.97 tCO_2e/m^3 with forest carbon sequestration included [49]. Thus, the variation in SFs is substantial [50], highlighting the impact of differences in LCA methodologies, system boundaries, assumptions, data, production techniques, geographic regions, product types, emission intensities, and end-of-life treatment [12,20,25,50,51]. This variability presents a challenge in producing consistent estimates of SFs. Therefore, to improve climate change mitigation estimates, SFs that are specific to both the product and the country are necessary [52].

Classification of wood products and boundaries of the review

Figure 2 depicts the classification of wood products and the scope of the review. Harvested-wood products (HWPs) are categorized into three groups: sawnwood, wood-based products (WBPs), and paper/paperboard [53].

Sawnwood is used to make rafters, joists, planks, beams, boards, scantlings, laths, boxboards, lumber, sleepers, wooden flooring, and moldings [53]. The WBPs category includes an aggregate of veneer sheets, plywood, particleboard, and fiberboard [53]. They can be manufactured in a wide range of sizes and shapes [54], and are also known as value-added products as they have a great scope of engineering properties [55]. The structural WBPs (plywood, oriented strand board) are manufactured by laminating various wood-based materials to improve the panel's strength, stiffness, and stability [54]. WBPs can also be classified according to whether they are used for structural or non-structural panels, whether they are exterior or interior grade panels, and the type of wood and materials used, which range from fiberboards to laminated beams [56]. WBPs are categorized according to the manufacturing procedure (wet or dry) [57]. WBPs are categorized into four groups: (a) veneer-based material, (b) laminates, (c) composite materials and (d) wood-nonwood composites, depending on the variation and relative size of wood elements utilized in panel production [54]. Engineering wood products, is another category and comes

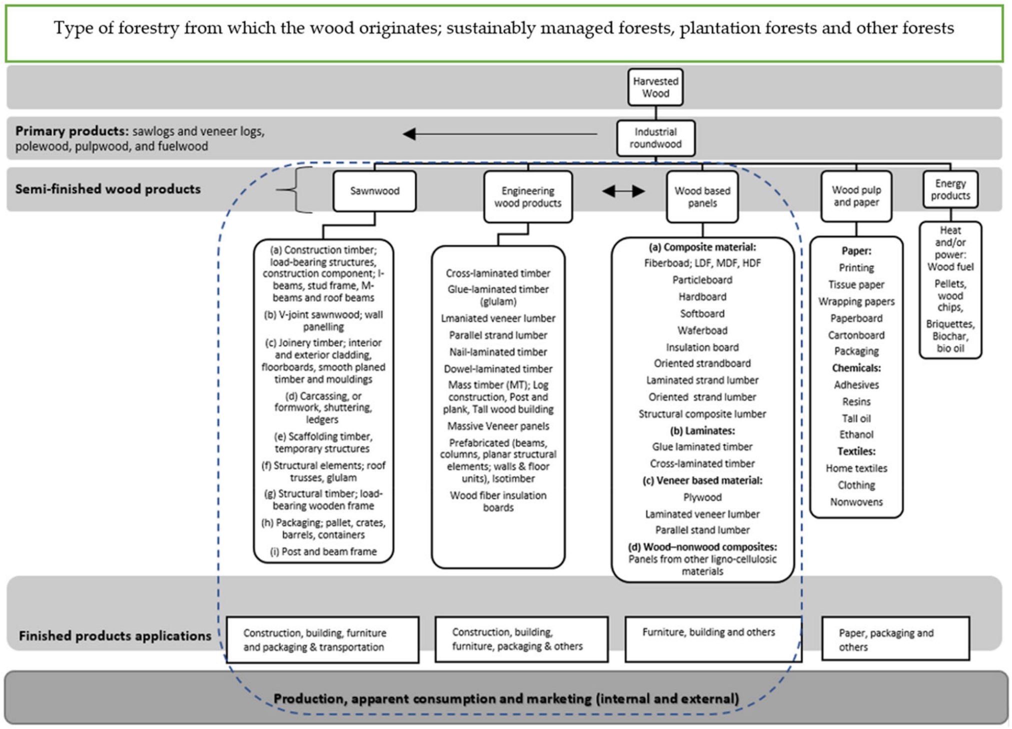

Figure 2. **Classification of wood-based products and boundaries of the review.**

in a number of sizes and specifications, including cross-laminated timber (CLT), glue-laminated timber or glulam (GLT), laminated veneer lumber (LVL), and others [58,59]. These products serve construction purposes like roofs, walls, flooring, and beams [58]. Furthermore, the classification includes wood pulp, paper products, and energy-related products. Moreover, there is a growing category of emerging wood products, including biochar, wood vinegar, and other innovative materials that do not fit into existing classifications. These may represent a new and developing area within wood-based product classifications.

Wood products are utilized across a wide range of sectors, including construction (for structural elements and engineering components), furniture (for residential and office items), packaging (pallets, boxes, and crates), paper and pulp (paper and cardboard), and bioenergy (fuel and raw materials for chemicals and textiles).

This literature review focuses particularly on sawnwood, WBPs, and engineering wood products to review their specific SFs in the construction and furniture sectors. The dotted line in Figure 2 highlights the central focus of this review.

Definition of substitution factor

The SF, often used interchangeably with the displacement factor, measures how much GHG emissions are avoided when a wood-based product replaces a non-wood product with the same function [3,60]. While wood products are typically assumed to fully substitute non-wood products, perfect 1:1 substitution is rare, and wood products may substitute each other without a separate SF calculation [37,60]. The SF is numerically quantified as the GHG emissions avoided in kg of carbon divided by the mass of carbon in the wood product [34,61]. LCA is used to calculate SFs by comparing at least two functionally equivalent products within the same scope and system boundary [20,33]. Thus, substitution potential is calculated based on functional units, not mass alone. In practice, equal mass or volume does not guarantee functional equivalence [12]. Deciding the mass, volume, size of each product to be analyzed and compared based on functional equivalence is challenging due to variations in material properties and applications, often relying on hypothetical equivalencies. Moreover, avoided emissions cannot be directly measured or verified but they can only be modeled using assumptions [12,37]. SF calculations focus on fossil emissions while excluding biogenic carbon to prevent double-counting and overlook certain emissions [33].

Substitution effect and substitution factor calculation

The substitution effect of wood products is a widely debated issue, lacking a consistent assessment method [62] and characterized by high uncertainty due to the reliance on underlying assumptions and limited availability of LCA data [33]. Lundmark et al. [63] classified substitution effects into three categories:: (i) avoiding emissions linked to the production and disposal of energy-intensive non-wood materials, including the full lifecycle from raw material extraction to end-of-life disposal; (ii) the effects of importing and exporting wood products; carbon accounts of the effects both in-country and abroad; and (iii) avoiding the use of fossil fuels due to energy recovery from fuel wood and residues from wood processing, chemical pulp processing, waste wood, and paper.

At the product level, the substitution effect is quantified using a SF, which specifies how many units of fossil carbon are avoided per unit of biogenic carbon contained in a wood product [37]. This is done by multiplying the volume of wood products by the SF [25]. Substitution benefits can be scaled based on either the amount of wood contained in the final product, or the amount of wood harvested to produce the given product [3]. At present, there is no single and established method for estimating substitution impacts at the market level [27]. In LCA, impacts of wood and non-wood products are compared using a functional unit, though this does not imply or measure the existence or rate of substitution on the market level [37]. Leskinen et al. [3] applied production volumes of forest products and their respective SF to determine substitution impact at market level or when upscaling product-level GHG benefits to regions or markets. Yang et al. [18] used 20 forest-based functional units to assess substitution impacts of wood use in the EU. Han et al. [35] examined three methods for calculating SFs to compare wood-based and non-wood furniture: (1) single SF, focusing on carbon storage or life-cycle GHG emissions; (2) replacement rate-based SF, considering how often wood furniture replaces non-wood, factoring in durability and market trends; and (3) wood-intensity-based SF, comparing the amount of wood versus non-wood material used in products. Yang et al. [18] utilized a unit-weighted SF for intermediate products and a volume-weighted SF for wood products (tC/tC). Although SFs can be expressed in different units, such as tC/tC, $kgCO_2e/kg$, tCO_2e/tCO_2e, and $MtCO_2e/year$, tC/tC appears to be the most transparent and comparable option [25]. Substitution based on mass, volume, or density is considered unrealistic [36]. A positive SF implies that the use of wood products would decrease GHG emissions, whereas a negative value implies the opposite [2]. The higher the factor (positive or negative), the more emissions can be avoided or produced [64].

According to Sathre and O'Connor [25], SF can be aggregated as follows:

$$\mathrm{SF} = \frac{GHG\ non_{wood} - GHG\ wood}{WUwood - WUnon_{wood}} \tag{1}$$

where: *GHGnon-wood* is non-wood GHG emissions; *GHGwood* is wood GHG emissions (expressed in mass units of carbon corresponding to the CO_2e of the emissions); *WUnon-wood* is amounts of non-wood used, applied to functionally equivalent product volumes; *WUwood* is the amounts of wood used, applied to functionally equivalent product volumes (expressed in mass units of C contained in the wood).

SF can also be stated as follows [65]:

$$\mathrm{SF} = \frac{GHG\ non_{wood} - GHG\ wood}{WUwood} \tag{2}$$

The substitution potential on the building level is calculated in two steps. In the first step, the difference in GHG emissions between building minerals and timber building is calculated. In the second step, the resulting difference is divided by the GHG emissions of the mineral building. The substitution is expressed through substitution potential SF_G as follows [64]:

$$SFG = \frac{GHG\ \text{building minerals} - GHG\ \text{building timbers}}{|GHG\ \text{building minerals}|} \left[\frac{\text{kgCO2e}}{\text{kgCO2e}}\right] \tag{3}$$

where: *GHG building minerals* are GHG emissions of the building to be replaced (mineral building);

GHG building timber are GHG emissions of the building, which replaces the substituted building (timber building).

The SF for material substitution (SFma) can be calculated as follows [66]:

$$SFma = \frac{(\text{C emissions material use of nonwood product} - \text{C emissions material use of wood product})}{C\ content\ of\ wood\ product} \tag{4}$$

The SF cannot be specified in a single indicator value; instead, it is expressed as a range between minimum and maximum values, as it depends on the type of product being compared [64,67].

To account for the entire value chain, the SF is calculated as follow [62]:

$$SFtotal = SFp + SFeol \tag{5}$$

where: SF_{total} is the total SF; SF_P is the SF from the material production stage; SF_{EoL} is the SF comprising emissions from the end-of-life stage.

According to Xie et al. [9] the substitution benefit is calculated as follows:

$$Substitution\ benefit = Eavoided = Csubstitution * \ DF \tag{6}$$

where: *Eavoided* is the avoided emissions; *C substitution* is the carbon contained in the end-uses of justified substitution; *DF* is the corresponding displacement factor.

Hurmekoski et al. [37] specified the overall substitution impact for the production stage (SI_P) as follows:

$$SI\ Pt = \sum DFWit * \ Sit \tag{7}$$

where: *DFWi* is the volume weighted DF for wood product i (tC/tC); *Si* is the supply of intermediate wood product i (MtC/yr); *t* is the year.

Likewise, the total substitution impact during the end-of-life phase (SI_EOL) is calculated as follow [18]:

$$SI\_EOFt = \sum DF\_EOFit * \ OFit * 44/12 \tag{8}$$

where: SI_EOLt is the substitution impact of the total wood supply during end-of-life stage, (Mt CO2e/yr), and t = year; DF_EOLi = the end-of-life DF for wood product i (tC/tC), OFi = the annual outflow of wood product i from the wood product pool (MtC/yr), and t = year.

The SF takes into account different life cycle stages of wood products, including production, use, and disposal [23]. Focusing only on emissions during production may mistakenly favor a product with higher GHG emissions during its whole lifecycle [12]. Leskinen et al. [3] recommend that SFs should encompass four components, i.e. production, use, cascading, and end-of-life to fully capture the emissions throughout a product's entire life cycle. These components influence GHG emission estimates (summarized in Table 1) and contribute to the variation in SF values based on the type and lifespan of the product [68]. To promote a standardized approach, the ISO recently released ISO 13391-3:2025 for assessing greenhouse gas dynamics and displacement effects of wood and wood-based products [21]. However, this standard does not account for market dynamics or shifts in consumer behavior, which can influence SFs. For example, a drop in sustainable wood product prices may increase their use and reduce demand for non-wood products. Conversely, the substitution effect also depends on consumer preferences; a price drop in a displaced product might draw in consumers who favor that product, but it won't impact those who prefer alternative products [17]. Ultimately, substituting one product for another is influenced not just by how much their emissions differ, but also by the scale of production and consumption of the products [3]. Therefore, to upscale the estimate of the substitution impacts at a regional or market level requires an understanding of market dynamics and detailed substitution processes [3].

Methodology

A literature search was accomplished through the Web of Science core collection and Scopus databases with specific search algorithms, after several keywords and connectors combination trials. We applied a search string containing the following keywords to search the literature: ("wood product*" OR "harvested wood product*" OR "wood-based product*") AND ("wood use*") AND ("displacement factor*" OR "substitution factor*" OR "substitution benefit*" OR "substitution effect*" OR "substitution impact*") AND ("Life cycle assessment*" OR "LCA*"). The keywords were separated by inter and intra-group boolean operators of "OR" and "AND" and extended them with an asterisk (*) to retrieve the precise hits. Then, we conducted a four-step literature selection process according to the following inclusion criteria (see Figure 3): (i) studies from the period of 2000–2024; (ii) studies that cover long-lived wood-based products through life cycle assessments; (iii) studies that provide GHG emission data for a wood-based product and a functionally equivalent non-wood product or presented product SFs. Only studies that clearly defined their methodological approach and provided sufficient data for comparative GHG emission analysis were included. Finally, a total of 130 articles (100 of which accounted for SFs) were selected as the focus of this review's data extraction procedure. Data from the selected studies were systematically extracted and compiled into a structured Excel spreadsheet. The extracted data included the following parameters: study location (country), wood product category, scope or specific sector of application, type of substituted material, functional unit used, life cycle stages considered, assumptions and scenarios analyzed, end-of-life treatment, reported SF values, unit of measurement used, and GHG emissions data. Consequently, SFs in the reviewed studies were calculated by comparing the GHG emissions of wood-based products with those of non-wood products serving the same function. The difference in emissions is then divided by the additional amount of wood required to perform the same function. At the building level, the substitution potential is calculated in two steps following Hafner et al. [64]. First, the difference in GHG emissions between a mineral-based building and a timber-based building is calculated. Second, by dividing this emission difference by the GHG emissions of the non-wood building. Where possible, both single SF values and ranges of SF estimates were calculated. Studies presenting only one scenario allowed for a single SF calculation, whereas those with multiple scenarios or variable assumptions enabled the derivation of both upper and lower SF estimates, reflecting the range of potential outcomes.

To identify patterns across the selected studies, a thematic analysis was performed, and the resulting categories were presented in tabular format. Themes were developed based on similarities in LCA system boundary, product types, application sectors, LCA focus, GHG emissions, and substitution-related carbon impacts. Additional sub-themes addressed geographic focus, functional unit, assumptions and end-of-life considerations. This thematic framework enabled a structured synthesis of the findings and supported the identification of existing gaps, methodological inconsistencies, and key drivers of SF variability. Subsequently, we created sub-categories based on the system boundary to refine the synthesis. Finally, we summarized the existing gaps, variabilities and uncertainties associated with carbon impact pathways. Therefore, based on the summaries of the results, we formed the analysis to ensure that it appropriately implies the reviewed studies.

Table 1. Factors affecting the SFs calculation.

LCA stages	Wood product	Substituted non-wood product
Raw material acquisition	Type of forestry from which the wood originates	Place and methods for extracting the raw material
	Harvesting methods	Processing the raw material (different for concrete, steel and plastic but usually non-renewable and/or fossil)
	Transports	Transports
Production	Sawmill processing, wood product manufacturing	Non-wood product manufacturing
	Transports	Transports
Use phase	Use, maintenance, repair, replacement, refurbishment	Use, maintenance, repair, replacement, refurbishment
Waste disposal (EoL scenarios)	Reuse, recycling, energy recovery, landfill	Reuse, recycling, energy recovery, landfill

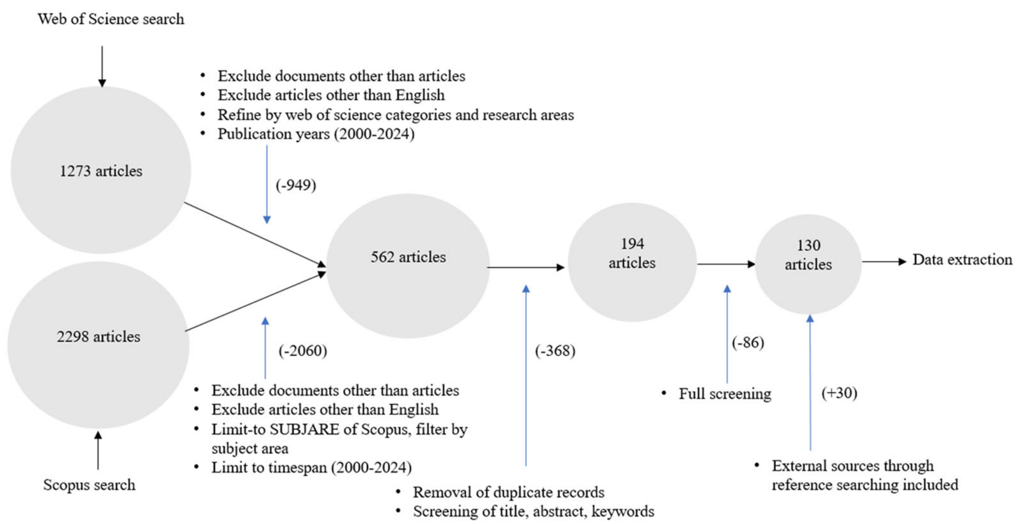

Figure 3. **Process followed in the selection of the articles for the review.**

Results and discussion

The below section discusses the various themes extracted through the systematic review of the literature to provide nuanced insights of the domain.

Distinct substitution factors of long-lived wood-based products'

The magnitude of SFs can vary widely based on several factors, including the type of wood product, its application, the materials it replaces, and the system boundary of the LCA used in the evaluation. A cradle-to-gate LCA only considers emissions from raw material extraction to the point where the product leaves the manufacturing facility, often underestimating long-term carbon impacts of WBPs, especially for long-lived applications like buildings. In contrast, a cradle-to-grave includes use-phase and end-of-life treatment, capturing broader environmental implications. This broader approach typically shows higher impact reductions, particularly when including biogenic carbon storage and end-of-life recycling. Tables 2–4 show the SFs for long-lived wood-based products used in construction and furniture sectors across various countries, focusing on LCA scopes. With a cradle-to-gate LCA scope (Table 2), sawnwood used in construction, furniture, and utility poles shows SFs ranging from 0.514 to 3 tC/tC, with the highest SF of 3 tC/tC for reinforced wood doors in the USA, indicating significant carbon savings compared to steel doors. Boiger et al. [147] indicated that using sawnwood in applications is the most efficient approach to significantly lowering GHG emissions. Leskinen et al. [3] found that sawnwood used in construction offers the largest substitution benefits due to its large market volume and high SF. However, assessing substitution effects using intermediate products can result in misinterpretations, as the calculations rely on oversimplified assumptions [12].

WBPs like particleboard, plywood, and fibreboard show SFs between 0.45 to 1.528 kgC/kg C, substituting materials like polyurethane foam and plasterboard. Engineering wood products such as CLT and glulam, used in mid-rise buildings, have SFs ranging from 0.16 to 1.662 kg C/kg C, with lower SFs for CLT in residential and commercial buildings (0.185-0.696 kg CO_2e/m^2 in the USA and China). In China, construction-related HWPs have SFs of 3.48 tC/tC [94]. In Germany, wood products used in construction and furniture show SFs between 1.1 to 2.4 tC/tC [26].

When evaluated under a cradle-to-grave LCA (Table 3), sawnwood SFs range from 0.431 to 7.5 tC/tC, depending on the specific wood product and substituted material (e.g. steel, concrete). The SF is higher for construction applications and lower for interior work. SFs for plywood, particleboard, and furniture range from 0.77 to 1.92 tC/tC. Insulation materials like fibreboards show SFs between 0.398 to 1.17 kg CO_2e/kg, depending on the substituted material. Engineering wood products like CLT and glulam have SFs ranging from 0.06 to 1.95 kg CO_2e/kg, varying with the building type (e.g. residential, commercial).

Table 2. Carbon SFs for the long-lived wood-based products (cradle-to-gate, construction process, and cradle-to-usage).

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit (FU)	System boundary	SF	Unit	Country	Source
Sawnwood	Sawnwood	Construction	Not specified	–	Cradle-to-gate	1.1	tC/tC	Finland	[16,69]
		Furniture				0.9			
	Wooden fishing vessel	Boat	Steel fishing vessel	1 fishing vessel (19.81 m)	Cradle-to-gate	0.7	kg CO_2 e/kg	India	[70]
	Sawnwood in construction	Construction, furniture	Not specified	–	Cradle-to-gate	1.1	tC/tC	Finland	[16]
	Sawnwood (softwood lumber)	Buildings, furniture, flooring, decking	Concrete/steel/Linoleum	Each end-use product	Cradle-to-gate	0.54	tC/tC	Canada	[71]
	Reinforced wood doors	Residential doors	Steel doors	–	Cradle-to-gate	3	tC/tC	USA	[72]
	Wooden utility poles	Utility poles	Steel	–	Cradle-to-gate	1.6	tC/tC	USA	[73]
	Sawn timber	Multi-storey building material	Precast reinforced concrete	$1m^3$	Cradle-to-gate	0.514	tC/tC	Lithuania	[74]
			Site-cast concrete			0.144–0.331			
WBPs	Particle board (spruce)	Building materials	Polyurethane foam board	Per kg	Cradle-to-gate	1.528	kg C/kg C	Norway	[75]
	Plywood (birch)		Plaster board			− 0.275			
						−0.301		Germany	
	Particle board (spruce)		Polyurethane foam board			1.501			
	Wood panels	Panels	Not specified	–	Cradle-to-gate	1.1	tC/tC	Finland	[16,69]
	Wood	Furniture	Not specified	Per m^3	Cradle-to-gate	0.4512	tCO_2e/m^3	Austria	[47]
		Construction	Not specified			0.4141			
	OSB, plywood, particleboard, MDF	Buildings, furniture, flooring, decking	Plastic/high-density polyethylene	Each end-use product	Cradle-to-gate	0.45	tC/tC	Canada	[71]
	Wooden Kitchen cabinets: sawn wood, particle board	Kitchen furniture	Kitchen furniture made of metal	–	Partially processed materials	1.1	tCO_2e/m^3	Countries in Western Europe	[76]
Engineering wood products	Cross laminated beam (spruce)	Building materials	Steel beam	Per kg	Cradle-to-gate	1.662	kg C/kg C	Norway	[75]
						1.621		Germany	
	Mass-timber building (CLT 30% & glulam 4%)	Mid-rise low-energy residential building	Mainstream concrete building	Per m^2 of floor area; over 50 years	Cradle-to-usage	0.63	kg CO_2e/m^2	Chile	[77]
					Cradle-to-construction	0.395			
	Two-floor GLT	Low-rise standard public buildings	Two-floor precast reinforced concrete	Two-floor (765 m^2)	Production value chains	1.43	tC/tC	Lithuania	[78]
	Five-floor GLT		Five-floor precast reinforced concrete	Five-floor (1913 m^2)		1.34			
	Mass timber: CLT and glulam	Office building	Structural steel framing	1 m^2 of floor area	Cradle-to-gate	0.9	$kgCO_2e/m^2$	USA	[79]
	Timber frames	Industrial buildings	Steel	Per m^2	Cradle-to-gate	0.034	kg CO_2e/m^2	Norway	[80]
			Concrete			0.228			
	Wooden building (Planed timber, Plywood, Glulam)	Building structures	Concrete building	Per unit floor area (m^2)	Design phase of building	0.65–0.72	kg CO_2e/m^2	Sweden	[81]

(*continued*)

Table 2. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit (FU)	System boundary	SF	Unit	Country	Source
	Mass timber building (CLT)	Residential building	Concrete building	1 m^2 of floor area	Cradle-to-gate	0.25	$kgCO_2e/m^2$	China	[82]
	Mass timber building (CLT and glulam structural elements)	Mixed-use office & apartment	Concrete building	1 m^2 of floor area	Cradle-to-site	0.185	kg CO_2e/m^2	USA	[83]
	Hybrid CLT building, with fireproofing	Mid-rise commercial buildings	Reinforced concrete building	1m^2 of total floor area	Cradle-to-gate	0.26	$kgCO_2e/m^2$	USA	[84]
	Hybrid CLT building, with charring					0.27			
	Structural system of load-bearing walls made of cross laminated timber	Residential buildings: a semi-detached house	Conventional masonry: load-bearing frame of reinforced concrete and walls made of light clay bricks	1 m^2 of heated floor area	Cradle-to-gate	0.35	kg CO_2e/m^2	Italy	[85]
	Panel wood construction	Two-floor cubic structure	Masonry construction	–	Cradle-to-gate	1.55	$kgCO_2e$/FU	Austria	[86]
	Cork oak-based flooring that includes CLT	Flooring applications	Concrete flooring	11 m^2	Cradle-to-gate	0.268	$kgCO_2e$/FU	USA	[87]
	Massive wood material (Massiv–Holz–Mauer, MHM) wall system	Building design (wall system)	Traditional brick wall	1 m^2 exterior wall	Cradle-to-gate	0.52	tCO_2e/t	Italy	[88]
	CLT	Low- & mid-rise commercial building	Concrete	0.25m^3	Cradle-to-gate	0.696	kg CO_2e/FU	USA	[89]
	CLT wall and floordeck	Mid-rise urban construction	Concrete and steel	–	Cradle-to-gate	0.16	kg C/kgC	Global	[90]
	CLT			Floor space of 30 m^2/capita		0.28			
	CLT			Floor space of 79.1 m^2/capita		0.48			
	CLT	Mid-rise residential buildings	Reinforced concrete	156 m^2 & 25 m height	Production & construction stages	0.5	tCO2e/FU	Australia	[86]
	GLT	Multi-storey building material	Precast reinforced concrete	1m^3	Cradle-to-gate	0.355	tC/tC	Lithuania	[74]
			Site-cast concrete			(−0.014–0.172)			
	Mass timber building stand structure	Residence building	Steel	1 m^2 of GFA	Cradle-to-construction site	0.28	kg CO_2e/m^2	USA	[91]
	Mass timber structures	Mid-to high-rise mass timber construction	Concrete structures	Per m^3 mass timber use	Production & construction stages	1.1	t CO_2e/m^3	USA	[92]
Combined/unspecified wood products	HWPs	–	Material substitution	–	Cradle-to-gate	2.9	MtC/year	Japan	[93]
	HWP	Construction	Construction production	Per m^2	Cradle-to-gate	3.48	tC/tC	China	[94]
		Furniture	Furniture production			1.36			
		Combined	Nonwood			2.9			

(*continued*)

Table 2. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit (FU)	System boundary	SF	Unit	Country	Source
	[1]Softwood based glued timber (glulam, CLT); [2]Wood based panels like particleboard, MDF, OSB; [3]Roundwood (poles, fences, buildings, treated); [4]Plywood, overlaid; [5]Wooden flooring (one layer, multi layers), laminate flooring; [6]Doors (interior, exterior)–only framing/ construction; [7]Wooden window frames; [8]Wooden furniture (solid wood); [9]Wooden furniture (panel based); [10]Wooden kitchen furniture	Buildings, furniture, flooring, windows, packaging	[1]Concrete, steel, bricks; [2] Gypsum board, plaster, concrete, brick typewalls; [3] Steel,concrete, aluminum; [4] Aluminum profiles, glass-fiber-plastic; [5]Ceramic tiles, plastic flooring, wall to wall carpet; [6]Steel, aluminum, PVC; [7] PVC, aluminum; [8] Glass, plastic, metal; [9] Glass, plastic, metal; [10] Glass, plastic, metal	–	Cradle-to-gate	1.1–2.4 (average SF 1.50)	tC/tC	Germany	[26]
	Wood product (board, fiber, soft polystyrene)	Not specified	Not specified	–	Cradle-to-gate	1.28	tC/tC	France	[95]

Table 3. Carbon SFs for the long-lived wood-based products (Cradle-to-grave).

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	End of life treatment	SF	Unit	Country	Source
Sawnwood	Coniferous sawnwood	Construction	Not specified	Per t	Not specified	1.2	tC/tC	Finland	[37]
	Non-coniferous sawnwood					1			
	Veneer based composite pole (VBC)	Utility poles for power transmission	Steel	A 12.5 m long (VBC, steel & concrete) utility pole	Landfilling & energy recovery for VBC poles; material recycling for steel poles & landfilling for concrete poles	0.476–0.558	kg CO_2e/FU	Australia	[96]
			Concrete			0.66–0.71			
	Sawnwood in construction	Construction	Not specified	–	Energy	1.92	tC/tC	Finland	[16]
	Sawn timber hardwood, pressure vessel treated softwood, painted softwood	Interior work: Claddings	Claddings: cement, stone, aluminium, zinc, steel, HPL	–	Landfill, energy recovery, material recycling	0.431–0.902	kg CO_2e/kg HWP	Europe	[97]
	Wooden pallets	Packaging	Plastic pallets			0.442–0.861			
	Wood windows	Interior work: windows	Windows: PVC, aluminium			2.21–5.53			
	Wooden filing racks/ shelves	Office furniture	Filing steel racks/shelves			0.66–0.728			
WBPs	Plywood & veneer	Construction	Not specified	Per t	Not specified	0.78	tC/tC	Finland	[37]
	Particle board					0.77			
	Hardboard					0.7			
	Officefurniture	Furniture factory	Not specified	Per m^3	Landfill	0.043	Mg/m^3 Carbon equivalent	France	[98]
	Kitchenfurniture					0.069			
	Homefurniture					0.043			
	Chairs					0.043			
	Beds					0.043			
	Plywood in construction	Construction, furniture	Not specified	–	Energy	1.92	tC/tC	Finland	[16]
	Furniture replacement					0.9			
	Insulation material: wood fibre board	Interior insulation material	Rock wool, glass wool, Polystyrene	–	Landfill, energy recovery, material recycling	−0.398–1.17	kg CO_2e/kg HWP	Europe	[97]
Engineering wood products	CLT and glulam	4-story office building	Structural steel framing	1 m^2 of floor area	Landfill	0.825	kg CO_2e/FU	USA	[79]
					Particleboard	0.8			
					Re-use	1.01			
	CLT and timber frame	Office & administration timber building	Brick, sand-lime brick, porous concrete and reinforced concrete	1 m^2 gross external area (GEA)	Not specified	0.06–0.48	kg CO_2e/FU	Germany	[64]
		Agricultural timber buildings				0.05–0.37			
		Non-agricultural timber buildings				0.14–0.44			
		Non-residential timber buildings				0.13–0.46			
		Residential building				0.675	kgCO_2e/FU	Malaysia	[99]

(*continued*)

Table 3. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	End of life treatment	SF	Unit	Country	Source
	CLT and steel plates as a hybrid component		Concrete and brick Concrete prefabricated	Entire building frame	Bricks-reusing, steel-recycling, CLT- reuse and energy recovery, Concrete-landfill	0.555 Brick work 0.316			
	Light steel frame	0.54							
	GLT-concrete slabs (hybrid)	Floor slabs	Lightweight steel composite decking Precast concrete panel Cofradal slab composite	$1m^2$ of slab structure (floor area)	Metal deck-reused & 5% landfill, GLT-reused & 20% landfill, Concrete-landfill	0.34 0.19 0.29	kg CO_2e/FU	Malaysia	[100]
	Mass timber building; CLT	High-rise mass timber building	Conventional concrete building	1 m^2 of floor area	CLT-30% land fill & reused and energy recovery, concrete-45% landfill and 55% recycle	0.016	CO_2e/m^2	USA	[101]
	Wooden flooring	Flooring systems	Laminated flooring Ceramic flooring Concrete flooring	1 m^2 per flooring	Energy recovery Landfill Landfill	0.17 0.516 0.58	kg CO_2e/FU	Malaysia	[102]
	Timber-frame	Single-family house	Concrete-masonry-conventional house	1 m^2 of heating area	Land fill	1.38	kg CO_2/m^2/yr	Uruguay	[103]
	CLT	12-story building	Reinforced concrete building	Comparison buildings	Reuse, recycling and energy recovery	0.7	kg CO_2e /FU	USA	[104]
	CLT, timber frame, and prefabricated timber	Single/two-family-house Multi-story residential buildings	Mineral (brick, sand-lime brick, porous concrete or reinforced concrete)	$1m^2$ gross external area	Not included	0.35–0.56 0.09–0.48	kg CO_2e /m^2	Germany & Austria	[67]
	CLT construction Timber frame	Residential building	Mineral; concrete	1 m^2 per GEA	–	0.26–0.52 0.16–0.27	kg CO_2e/m^2 GEA	Germany & Austria	[105]
	Wood flooring	Flooring	Ceramic tile	$1m^2$ of floor area	Landfill, energy recovery	0.16–2.85	t CO_2e/m^3	China	[106]
	CLT design 50 % Timber design with increased bio design 69%	Hypothetical building	Concrete design	$1m^2$ of living area for 50 years	Incineration with energy recovery	0.42–0.99 0.39–1.41	kgCO_2e/m^2	Sweden	[107]
	CLT	Mid-rise residential building	Concrete building	2799.3 m^2 area	Demolition, disposal and recycle	0.393–0.423	tCO_2/FU	China	[108]
	Woodchip paved walk ways	Paved walkways	Asphalt-paved walkway Artificial turf walkway	1 m^2/year	Disposed chips used for energy	1.45 1.61	kgCO_2e/m^2 /year	Japan	[109]
	CLT facades	Low- & mid-rise commercial building	Concrete	$0.25m^3$	(a) CLT recycle (b) CLT landfill	0.45–0.54	kg CO_2e/FU	USA	[89]
	ACQ-treated lumber decking	Size deck surface	Wood plastic composite decking	Per deck per year (29.7 m^2)	Reuse and landfill	0.65–0.84	lb CO_2e/yr	USA	[110]
	Borate-treated lumber framing	Single story house	Galvanized steel framing	206.7 m^2 & 60.96 m wall framing	Landfill	0.445	lb CO_2e/FU	USA	[111]
	Wood building frames	Multi-storey building	Concrete building frames	–	Landfill, energy recovery	2.3–7.4	tC/tC	Sweden	[112]
	CLT	Building construction	Concrete floors in steel structural systems	Per t	Not reported	1.58	tCO_2e/t	Global	[113]
			Reinforced concrete		Landfill, recycling	1.95	kg C/kg C	Australia	[114]

(*continued*)

Table 3. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	End of life treatment	SF	Unit	Country	Source
	CLT as main structural material	Nine-storey building featuring CLT panels		9 story building with 1558 m^2					
	CLT building	Mid-rise apartment buildings	Concrete building	1 m^2 of building	Recycling, incineration, landfill	0.497–0.676	$kgCO_2e/m^2$	Norway	[115]
	Wood structural framing	Multistory apartment building	Reinforced concrete frame	Apartment building	Energy recovery	4.4–7.5	tC/tC	Sweden	[116]
	Wood: Truss and flooring Exterior cladding Interior coverings Other end-use products	Building product	Not specified	Per m^3	Landfill	0.169 0.024 0.024 0.024	Mg/m^3 Ce	France	[98]
	CLT building	Residential building	Concrete slab with light gauge steel studded walls	–	Landfill, reuse, energy recovery	0.46 (0.38-0.56)	kg C/kg C	Canada	[117]
	Wood-framed buildings	Apartment building	Concrete framed buildings	–	Energy recovery	1.9–5.6 0.4–3.3	tC/tC	Sweden Finland	[118]
	Wood-framed buildings	Apartment building	Concrete framed buildings	–	Energy recovery	0.1–7.3	tC/tC	Sweden	[119]
	CLT	Mid-rise apartment buildings	Concrete building	Use of 1 m^2 of building	Landfill, incineration with energy recovery, recycling	0.50–0.68	$kgCO_2e/m^2$	Norway	[115]
	CLT building and modular timber structures	Residential building	Concrete building	6 story (1686 m^2)	Recycling or energy recovery	1.10–1.82	tC/tC	Sweden	[120]
	CLT, timber	Multistorey buildings	Reinforced concrete frame Steel frame	–	Timber: landfill & energy recovery, Concrete: landfill	0.51–0.58 0.85–1.02	tC/tC	United Kingdom	[121]
	Wood flooring, floorboards in solid oak	Flooring materials	Carpet in wool, carpet in polyamide, vinyl, and linoleum	Per m^3 flooring	Energy	0.1–15.5	tCO_2e/m^3	Norway	[122]
			linoleum vinyl, Carpet in polyamide Wool carpets			0.1–1.9 0.2–2.3 0.9–2.5 11.8–15.5			
	Flooring: laminate	Interior work: Laminates	Flooring: PVC	–	Landfill, energy recovery, material recycling	1.19–1.52	kg CO_2e/ kg HWP	Europe	[97]
	Flooring: parquet (solid, multi-layer)	Construction: interior, parquet	Flooring: ceramic tiles, artificial stone			−0.0164–0.924			
ceramic composite house	Wooden house (spruce)	House of 100 m^2 of floor 1	Aerated concrete blocks 0.47	Per m^2/year		0.62	Kg CO_2e/m^2 /year	Ukraine & Slovakia	[123]
Combined or unspecified wood products	Sawnwood, pulp: solid wood panel, glulam column, wood joist ceiling/floor, wood fiber insulation board, exposed beam structure, wood	(a) Building constriction structure: exterior walls, columns, storey ceiling/floor, insulation, roof, (b) Building finish: wall	Brick cavity masonry, steel column, reinforced concrete floor, rockwool, aerated concrete steep roof, concrete palisade, interior plastering,	1t	Energy generation from wood	1.42	tC/tC	Europe	[38]

(*continued*)

Table 3. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	End of life treatment	SF	Unit	Country	Source
	palisade, spruce panelling, oak staircase, 3-layer parquet, raw wood siding, architrave, wood furniture, formwork panels, spruce panelling	and ceiling covering, stairs, floor coverings, facades fittings, furniture, (c) Wood products: packaging, auxiliary construction materials	precast concrete stairs, glazed ceramic tiles, exterior plastering, architrave frame, steel furniture, polypropylene, aluminium formwork, interior plastering						
	Laminated timber board, glulam pillar, ceiling of wood beams, wood fibre insulation panel, unlined joists, wood palisade, profiled board, wooden staircase, three-layer parquet flooring, wood panels, particleboard	Building: Construction, furniture, energy	Two-layered brick wall, steel pillar, ceiling of reinforced concrete, mineral wool, porous concrete pitched roof, concrete palisade, interior plasterwork, ready-made concrete staircase, ceramic tiles, exterior plasterwork, steel	1t	Incineration for energy recovery	1.29	tC/tC	Switzerland	[124]
	Lumber, plywood, LVL, OSB, (trusses, glulam beams, I-joists	Residential construction Above ground wall designs Floor and roof assemblies	Steel Concrete Steel Concrete Steel	–	Landfill, incineration, recycling or reuse	0.21 0.24 0.25 0.44 0.61	kg CO_2/FU	Minneapolis, USA Atlanta, USA Minneapolis, USA Atlanta, USA Minneapolis, USA	[125]
	Wood-based house (lumber, wood panels)	Single/multifamily house	Steel and concrete based house	–	Burning, landfill, energy recovery	0.39	tC/tC	Canada	[126]
	Wood products	Single-family house	Concrete Steel	– –	Landfill	2.2 0.9	tC/tC	USA	[127]
	Wooden buildings, wooden piles, guardrails, furniture: sawnwood, plywood, chips,	Building construction, civil engineering, furniture	Non-wooden buildings, cement and sand piles, metal guardrails, metal furniture	–	Not reported	0.75	tC/tC	Japan	[128]
	Roof beams, sawn wood in glulam laminated wood	Beams	Steel	1 m^2 roof	Landfill, energy, with or without carbon fixation on forest land	0.24–0.97	tCO_2e/m^3	Norway	[49]
	Wood	Floor coverings	Natural stone	1 m^3 floor	Biofuel	1.26	t CO_2e/m^3	Norway	[129]
	Sawnwood, plywood, particlreboard	–	Concrete and steel		Bioenergy	1.2	tC/tC	Finland	[130]
	Core & shell: Exterior walls: wood frame, CLT, log wall; Interior walls (load bearing): wood	Construction: core & shell on residential buildings	Exterior walls: brick masonry, reinforced concrete, brick cavity walls, sand-lime brick,	–	Landfill, energy recovery, material recycling	1.11–1.58	kg CO_2e/ kg HWP	Europe	[97]

(*continued*)

Table 3. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	End of life treatment	SF	Unit	Country	Source
	frame, wood construction; Interior walls (non-load bearing): wood frame with different claddings; Storey ceilings without and thermal requirements: wooden beam construction, solid timber element, solid timber element, wood laminated veneer lumber, wood joist ceiling		breeze concrete, aerated concrete; Interior walls (load bearing): brick masonry, reinforced concrete, sand-lime brick, breeze concrete; Interior walls (non-load bearing): plaster board; Storey ceilings: reinforced concrete						
	Wooden buildings (wood frame, wood-solid, brick-insulation, brick-single-shell, concrete insulation)	Single-family building	Pairs of wood and nonwood alternatives for low energy and passive house standard (difference in wood usage)	Gross floor area of 221 m^2	–	0.54	$kgCO_2/kgCO_2$	Austria	[131]

Table 4. Carbon SFs for long-lived wood-based products (Different wood-based product combinations and/or unspecified system boundary).

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	SF	Unit	Country	Source
Sawnwood	Sawn wood	Building construction	Concrete, steel, plaster	–	0.16–0.24	tC/tC	Germany	[132]
	Heavy timber	Residential buildings	Concrete	–	0.60	tCO_2e/tCO_2e	Global average produced	[9]
	Lightweight wood frame			–	1.76			
	Heavy timber	Non-residential buildings	Concrete	–	1.22			
	Lightweight wood frame		Steel	–	1.11			
	Harvested wood products: sawn wood	Construction dwelling	Not specified	–	0.55 (0.27–1.16)	tC/tC	Global	[43]
	Sawnwood and industrial roundwood	–	Steel, concrete and plastics	–	0.04	Mg/m^3Carbon equivalents	South Korea	[133]
	Timber	Construction	Concrete and bricks	–	1.66	tC/tC	Germany	[134]
	Sawnwood and panels	–	–	–	0.45–2.00	tC/tC	Mexico	[135]
	Long-lived wood products (sawnwood)	–	Not specified	–	1.1	tC/tC	Europe	[136]
	Wood structures	Non-residential buildings	Concrete	–	4.54	–	Canada	[137]
			Steel	–	0.59	–		
	Sawnwood: Lumber	General use	Not specified	–	0.45	tC/tC	Canada	[138]
		Building sector		–	2.2			
	Wooden buildings, doors, poles, etc.	Building products	Non wood products	–	0.25–5.60	tC/ tC	China	[139]
	Wooden building	Building structures	Concrete building	Per unit floor area	0.65–0.72	kg CO_2e/FU	Sweden	[81]
	Harvested wood products	–	Energy/material	–	0.50–0.78	t CO_2/m^3	Austria	[140]
	Wood based products	Material	Material substitution	–	1.50	tC/tC	Germany	[26,66]
	Wood product	Buildings, furniture, flooring, doors, windows	Non-wood product	1t	2.1 (−2.3–15)	tC/tC	Sweden	[25]
	Houses with wood-based frame	Residential construction	Steel frame-based building	1.5 million housing per year	0.17–0.21	$MtCO_2e$/FU	USA	[141]
	Houses with wood-based wall		Concrete wall-based building		0.24–0.50			
	Sawnwood, wood based panels	Not specified	Concrete, Steel, plastics	–	0.90	tC/tC	Finland	[142]
	Solid wood products (panels)	Single-family, multi-family, & multi-use building	Concrete and Steel	–	2.20	tC/tC	Canada	[143]
	Sawnwood			–	2.10			
	Sawnwood	Construction sector	Not specified	–	0.80	tC/tC	Canada	[46]
WBPs	MDF, timber, veneer, plywood, glue board, particle board	Bedroom, kitchen, living room, dining furniture, bed, wardrobe	PVC, steel, glass, aluminium, polypropylene, polyurethane rigid foam	Per t	1.46	tC/tC	China	[44]
	Chair (Particleboard and MDF)	Wood based office furniture products	Steel, particleboard and MDF	–	1.27	tC/tC	Republic of Korea	[35]
	Desk (Particleboard and MDF)		Steel, particleboard and MDF	–	1.03			
	Cabinet (Particleboard)		Steel particleboard	–	7.60			

(*continued*)

Table 4. Continued.

Wood product category	Specific wood product	Specific sector or purpose	Substituted product	Functional unit	SF	Unit	Country	Source
	Medium-lived wood products (panels)	–	Not specified	–	0.55	tC/tC	Europe	[136]
	Lumber, plywood, veneer, OSB	Low-rise non-residential buildings	Unclear	–	2.03	t CO_2e/tCO_2e	USA	[144]
	OSB, plywood, particleboard, MDF	General use	Not specified	–	0.54	tC/tC	Canada	[138]
		Building sector		–	2.10			
	Wood panels	Construction sector	Not specified	–	0.81	tC/tC	Canada	[46]
Engineering wood products	Average timber light-frame	Multi-family housing construction	Concrete frame	461,000 dwelling units	0.24	MtCO_2e/year	Sweden	[11]
	Cross-laminated timber				0.09			
	Cross-Laminated Timber	High-rise building construction	Not specified	–	0.16–1.95	kgC/kgC	Global	[13]
Combined or unspecified wood products	12 wood products: laminated timber board, glulam pillar, ceiling of wood beams, wood fibre insulation panel, unlined joists, wood palisade, profiled board, wooden staircase, three-layer parquet flooring, wood panels, particleboard	Construction or interior works	Concrete, mineral wool, & bricks, interior works, ceramic tiles or steel\two-layered brick wall, steel pillar, ceiling of reinforced concrete, mineral wool, porous concrete pitched roof, concrete palisade, interior plasterwork, ready-made concrete staircase, ceramic tiles, exterior plasterwork, steel	Material substitution	0.261 (in Switzerland) and 0.335 abroad	MtCO_2e/FU	Switzerland	[1]
	Wood products	Unclear	Steel, plastic or concrete		1.14	tC/tC	Germany	[65]
	CLT, sawnwood, panels, glulam	–	Concrete, steel, stone wool, glass wool, plasterboard, plastics, crushed stone, mortar, lightweight bloacks, aluminium	–	0.89–1.14	tC/tC	Sweden	[28]
	Structural (building, internal or external wall, wood frame, beam) and non-structural construction wood products	Construction of dwelling	Not specified	–	Average SF 1.2 (1.3 structural & 1.6 non-structural)	kg C/kgC	Global	[3]
	Wood based products	Not specified	Not specified	–	1.3–2.4	tC/tC	Finland	[145]
	Wood product	Not specified	Not specified	–	2.1	tC/tC	Denmark	[146]
	Sawnwood	Construction sector	Steel or reinforced concrete as baseline material	18 different studies	0.80	tC/tC	Canada	[46]
	Panels				0.81	tC/tC		

Wooden flooring has SFs between 0.16 to 2.85 tCO_2e/m^3, and wood frames can reach SFs up to 7.5 tC/tC, depending on usage, substituted materials, and end-of-life treatment and location. For example, the SFs for timber-based building systems vary greatly depending on how the wood products are managed at the end-of-life [148]. Extending the lifetimes of products would provide the greatest climate benefit [149]. The combined or unspecified wood products category shows SFs ranging from 0.21 to 2.2 tC/tC, with significant variation depending on product type and location.

When considering different wood-based product combinations or unspecified LCA scope (Table 4), sawnwood shows SFs from 0.04 to 2.2 tC/tC, with higher SFs in construction and residential sectors when substituting materials like concrete, steel, and plastics. WBPs have an SFs around 0.54 to 2.1 tC/tC, while particleboard, MDF, timber, and veneer panels used in furniture products have SF of 1.03 to 7.60 tC/tC. The SFs for CLT and other engineering wood products in high-rise buildings range from 0.09 to 1.95 kgC/kgC globally. SFs for combined or unspecified wood products typically range from 0.89 to 1.14 tC/tC when replacing materials like concrete, steel, and mineral material in construction.

Overall, the result shows that product types, production process, wood applications, and LCA stages determine the wood products that most effectively reduce emissions. The degree of substitution depends partially on the amount of energy and chemicals needed during manufacturing [150]. Consequently, SFs vary depending on the wood product type, its application, and the material it replaces. SFs are higher for long-lived structural wood products used in construction compared to those used in interior applications or as furniture. This is likely the case, as it has been noted that wood products used in construction provide the greatest reduction in carbon emissions compared to furniture manufacturing [94,116]. Boiger et al. [47] also found that the greatest GHG emission reductions per unit of wood are anticipated in the furniture and construction sectors. However, it is important to note that not all wood used in construction contributes to substitution potential, and overestimating the effect could occur if substitution does not actually lead to a reduction in CO_2 emissions. The substitution effects of wood in construction influenced by building type, making it difficult to establish an average SF; however, wood construction demonstrates moderately high SF values, particularly under optimistic calculations [12]. The average substitution benefit, at least in principle, be increased by changing the product portfolios or the end use of wood products [69,151]. The choice, quantity and nature of materials affect environmental impacts [152]. For example, the substitution potential of a building structure is influenced by its individual materials, construction, geometry, and design [64]. Consequently, the SFs for timber-based building systems vary greatly depending on how the wood products are managed at the end of their life [148]. The SFs also vary significantly across countries like Germany, Sweden, Canada, and South Korea, which reflect regional differences in material substitution effects. It is also important to recognize that SFs are not static and can evolve with advances in technology and changes in market dynamics. For instance, improvements in wood processing efficiency, adoption of low-carbon manufacturing energy sources, and innovations in product design can enhance the substitution potential of WBPs. Furthermore, regional differences in forest management practices, wood availability, and end-of-life infrastructure impact the realized substitution benefits. For example, countries with well-established recycling systems can achieve higher SFs compared to those relying primarily on landfilling. Therefore, the broad variation in the individual product SF can be attributed to several factors, such as the type of wood product under consideration, the material being substituted, life cycle stage accounted, assumptions made about production technology, efficiency, energy mix in manufacturing, and end-of-life options (landfilling, incineration, and recycling), and thus, generalizations are not straightforward [13,94].

Product-specific GHG emission factors associated with the life cycle of wood-based products

The product-specific GHG emission factor represents the total GHG emissions associated with the entire life cycle of a wood-based product. The GHG emission factor is the carbon footprint of the product itself, typically expressed in units of CO_2e per unit of product, carbon content per unit mass of a material or FU. The emission factor varies depending on the LCA system boundaries applied during the analysis, as different boundaries yield varying results. Table 5 presents GHG emissions factors associated with the life cycle of various wood-based products, which are divided into three main categories: sawnwood, WBPs, and engineering wood products. It highlights notable variations in emission factors depending on the product type, location, and the scope of its LCA. Sawnwood, commonly used in manufacturing various end-use products, exhibits varying GHG emissions [71], ranging from 0.0383 kg CO_2e/kg to 1480 kg CO_2e/m^2, depending on the product, LCA scope, and region. Notably, wooden flooring and treated lumber, particularly in the USA and Malaysia, show higher emissions.

WBPs like particleboard, plywood, MDF, and OSB show GHG emission factors ranging from 124.5 kg CO_2e/m^3 to 655 kg CO_2e/m^3, typically when assessed from cradle-to-gate. Emissions vary significantly based on the type of wood products and their treatment. For example, manufacturing particleboard from wood waste emits 276 kg CO_2e/ton, while using fresh wood results in 282.01 kg CO_2e/ton [160]. A study found that particleboard production emits 333 kg CO_2e/m^3 in Brazil and 215 kg CO_2e/m^3 in Spain [163]. MDF production emissions range from 227 kg CO_2eq/m^3 [158] (gate-to-gate) to 679 kg CO_2e/m^3 (cradle-to-grave) in Iran, depending on whether biogenic carbon is included. The study from Iran shows GHG emission ranges 345–655 kg CO_2e/m^3 (with biogenic carbon) and 679 kgCO_2e/m^3 (without biogenic carbon) for poplar-based MDF.

Engineering wood products, including CLT, GLT, and others, have emission factor ranging from 24.1 kg CO_2e/m^3 to 547 kg CO_2e/m^3. For instance, using CLT in buildings results in annual GHG emissions of 54 kg CO_2e/m^2 (excluding biogenic carbon) and 49 kg CO_2e/m^2 (including biogenic carbon) under a cradle-to-grave analysis. CLT production emissions are estimated at an average of 152.0 kg CO_2e/m^3 [185], with significant variation. DLT has the lowest carbon emissions (118 kg CO_2/m^2) compared to CLT (130 kg CO_2/m^2) and NLT (133.5 kg CO_2/m^2) during manufacturing and construction [173]. CLT construction-related GHG emissions range from 0.05 to 6.3 tCO_2e/m^2 of floor area [185]. GLT frame construction offers lower emissions [78]. However, the cradle-to-gate accounting of glue-laminated solid wood results in higher emissions, with approximately 547 kg CO_2e/m^3 in Germany [172].

The overall GHG emissions of long-lived wood-based products are influenced by factors such as material types, product lifespan, energy consumption, transportation, and the mix of wood species [68]. Changes in these factors can significantly affect emission estimates. GHG emissions also vary based on end-of-life treatment and whether biogenic carbon is considered, leading to a wide range of estimates. Regional differences in species composition and wood density, which affect drying energy and carbon emissions, also contribute to these variations [186]. Extending product lifespan, such as by reusing CLT panels, can reduce emissions [176], as can sourcing wood locally and using lighter species in production [181]. This complexity underscores the diverse GHG emission factors across different wood-based products with a substantial variation across countries.

Challenges associated with LCA aspects of wood-based products

Conducting LCAs presents a range of challenges, when applied to wood-based products. These challenges arise from the inherent complexities of wood products and their diverse applications, as well as from methodological issues within the LCA framework itself. A detailed discussion of these challenges is provided below.

Challenges related to goal and scope definition

The LCA results depend on how they are framed and modelled [187,188]. Under defined goals, unarticulated or incomplete scope definition, purpose and context of the LCA are a pressing challenge as they lead to unclear results and affect how the results might be used. Different purposes lead to confusion when reporting the results [189]. Variation in LCA scopes, functional units and system boundaries can dramatically alter LCA results, and thus, impair applicability and comparability [190,191]. The outcomes of the LCAs for a product can vary depending on the applied life cycle phases. Decisions about where to set system boundaries for assessments can have a significant impact on the results as it leads in different impacts [151]. For example, "cradle-to-gate" system boundaries used frequently in LCA studies. However, the cradle can be set at different points in the system, and the scope changes greatly depending on where the cradle begins and what is included in the evaluation. Differences in system boundaries, inclusion and exclusion of different phases, and assumptions led to a wide range of results [192,193]. It is important to document all assumptions in the goal and scope section. Thus, the choice of

Table 5. Product-specific GHG emission factor associated with the life cycle of long-lived wood-based products.

Wood product category	Specific wood product	Functional unit	GHG emission factors	System boundary	Location	Source
Sawnwood	Softwood lumber	1 m^3	61.99 kg CO_2e/m^3	Cradle-to-gate	Canada	[153]
	Sawnwood	1 kg	0.0383 kg CO_2e/kg of wood	Cradle-to-gate	Spain	[154]
	Wood pallet	1 unit of pallet (22.35 kg)	2.12 kg CO_2e/FU			
	Hardwood lumber production	1 m^3	88 kg CO_2e/m^3	Gate-to-gate	Pakistan	[155]
	Wooden flooring	1 m^2 per flooring	1480 kg CO_2e/m^2	Cradle-to-grave	Malaysia	[102]
	Woodchip paved walkways	1 m^2 /year	(a) 2.62 kg-CO_2e/m^2/year; (b) 0.2 kg CO_2e/m^2/year	(a)Cradle-to-construction; (b) Cradle-to-grave	Japan	[109]
	Alkaline copper quaternary treated lumber decking	Per 30 m^2	2853 lb-CO_2e/FU	Cradle-to-grave	USA	[110]
	Borate-treated lumber perimeter wall framing	30.5 linear meters	2245 lb-CO_2e/FU	Cradle-to-grave	USA	[111]
	Wood floor (solid parquet: (a) 8 mm; (b) 10 mm; (c) 22 mm)	1 m^2	(a) 7.1 kg CO_2e/m^2; (b) 5.9 kg CO_2e/m^2; (c) 4.4 kg CO_2e/m^2	Cradle-to-grave	Germany	[156]
	Wood floor (multilayer parquet)	1 m^2	12.7 kg CO_2e/m^2			
	Wood floor (solid floorboard)	1 m^2	0.2 kg CO_2e/m^2			
	Wood floor (wood blocks)	1 m^2	2.8 kg CO_2e/m^2			
	Roof (wood)	–	(a) 1754 kg CO_2e; (b) 222 kg CO_2e	(a) Production phase; (b) End of life phase	Norway	[157]
WBPs	Particle board (spruce)	1 m^3	343.55 kg CO_2e/m^3	Cradle-to-gate	Norway	[75]
	Plywood (birch)	1 m^3	436.04 kg CO_2e/m^3			
	MDF	1 m^3	227 kg CO_2e/m^3	Cradle-to-gate	Iran	[158]
	OSB	1 m^3	127 kg CO_2e/m^3	Cradle-to-gate	Brazil	[159]
	Particleboard(wood waste)	1t	276 kg CO_2e/t	Production phase	Republic of Korea	[160]
	Particleboard(fresh wood)	1t	282.01kgCO_2e/t			
	MDF poplar-based	1m^3	345–655 kg CO_2e/m^3 (with biogenic carbon) and 679 kgCO_2e/m^3 (without biogenic carbon)	Cradle-to-grave	Iran	[161]
	Plywood, indoor use	1 m^3	391.81 kg CO_2e/m^3	Cradle-to-gate	Slovakia	[162]
	Plywood, outdoor use	1 m^3	515.06 kg CO_2e/m^3			
	Three-layer solid wood panel	1 m^3	335.43 kg CO_2e/m^3			
	Particleboard	1 m^3	333 kgCO_2e/m^3	Gate-to-gate	Brazil	[163]
		1 m^3	215 kgCO_2e/m^3		Spain	
	Furniture: office cabinet with a sliding door, 900 mm width, 480 mm depth, and 1600 mm height	Per 1 office cabinet	122 kg CO_2e/office cabinet	Cradle-to-grave	Brazil	[164]
	Veneer based composite pole for power transmission	12.5 m utility pole	(a) 63.22 kg CO_2e/FU (b) 74.995 kg CO_2e/FU	Cradle-to-grave; (a) incineration, (b) landfilling	Australia	[96]
	Childhood furniture set: a baby cot convertible into a bed a study desk and abed side table	Total weight of 173.9 kg	164.9 kg CO_2e/FU	Production process	Spain	[165]
	Convertible cod into childhood bed	1 kg of product ready for use	810 gCO_2e/kg product	Cradle to-gate	Spain	[166]
	Kitchen cabinet		3269 gCO_2e/kg product			
	Office table		4842 gCO_2e/kg product			

(*continued*)

Table 5. Continued.

Wood product category	Specific wood product	Functional unit	GHG emission factors	System boundary	Location	Source
	Living room furniture		1509 gCO_2e/kg product			
	Headboard		2426 gCO_2e/kg product			
	Youth room accessories		886 g CO_2e/kg product			
	Wine crate		434 gCO_2e/kg product			
	Wooden modular playground		1439 gCO_2e/kg product			
	Ventilated wooden wall		537 gCO_2e/kg product			
	Wood fibre insulation	–	(a) 424 kg CO_2e; (b) 0.86 kg CO_2e	(a) Production phase; (b) End of life phase	Germany	[157]
Engineering wood products	I-joists	1 m^3	218.55 kg CO2e per m^3	Cradle-to-gate	Canada	[153]
	Cross laminated beam (spruce)	1 m^3	77.88 kg CO_2e/m^3	Cradle-to-gate	Norway	[75]
	CLT wall	1 m^2	69.9 kg CO_2e/m^2	Production, use & EoL stage	Republic of Korea	[167]
	(a) Timber wall panel, (b) Circular wall panel	1 m^2	(a) 116 kg CO_2e/m^2; (b) 122 kg CO_2e/m^2	Cradle-to-cradle	United Kingdom	[168]
	CLT panel	1 m^3	142 kg CO_2e/m^3	Production, use & EoL stage	Republic of Korea	[169]
	Timber: residential building	Per unit area per year	10.55 kg $CO_2/(m^2{\cdot}a)$	Product stage to EoL stage	Finland	[170]
	Timber: public buildings	Per unit area per year	58.69 kg $CO_2/(m^2{\cdot}a)$	Product stage to EoL stage	USA	[170]
	CLT	1 m^3	158.7 kg CO_2e /m^3	Cradle-to-gate	USA	[171]
	Glue-laminated solid wood (glulam)	1 m^3	547 kg CO_2e/m^3	Cradle-to-gate	Germany	[172]
	CLT (foundation)	Floor area of 180 m^2	(a) 1188 kg CO_2e/FU; (b) 1286 kg CO_2e/FU	(a) Production phase; (b) End of life phase	Sweden	[157]
	Wooden framework	–	(a) 46 kg CO_2e; (b) 1.05 kg CO_2e		Norway	
	Wood panel	–	(a) 38 kg CO_2e; (b) 0.86 kg CO_2e		Norway	
	CLT (internal components)	–	(a) 211 kg CO_2e; (b) 291 kg CO_2e		Sweden	
	Two-Floor GLT low-rise public buildings	1 m^2	0.0874 tCO_2e/m^2	Production phase	Lithuania	[78]
	Five-Floor GLT low-rise public buildings	1 m^2	0.0899 tCO_2e/m^2			
	Mass timber building using CLT and glulam	–	(a) 415 tCO_2e; (b) 251 tCO_2e; (c) 338 tCO_2e; (d) −408 tCO_2e	(a) Cradle-to-gate; (b) End-of-life (landfill); (c) End-of life (particleboard); (d) End-of-life (re-use)	USA	[79]
	CLT	1 m^2	130 kg CO_2/m^2	Production phase	Malaysia	[173]
	NLT	1 m^2	133.5 kg CO_2/m^2			
	DLT	1 m^2	118 kg CO_2/m^2			
	CLT and steel plates as a hybrid component	Entire frame of residential building	9100 kg CO_2e/FU	Cradle-to-grave	Malaysia	[99]
	Mass-timber building (CLT 30% and glulam 4%)	1 m^2 of floor area	90 kgCO_2e/m^2 101 kg CO_2e/m^2 131 kg CO_2e/m^2	Material production Cradle-to-construction Cradle-to usage, excl. biogenic carbon	Chile	[77]
	GLT-concrete slabs (hybrid)	1 m^2 of slab structure (floor)	8500 Kg CO_2e/FU	Cradle-to-grave	Malaysia	[100]

(*continued*)

Table 5. Continued.

Wood product category	Specific wood product	Functional unit	GHG emission factors	System boundary	Location	Source
	Mass timber building stand structure	$1m^2$ of GFA	198 kg CO_2e/FU	Cradle-to-construction site	USA	[91]
	Mass timber building (CLT)	1 m^2 of floor area	221.3 kg CO_2e/FU	Cradle-to-gate	China	[82]
	Mass timber building (CLT panel, glued solid timber and OSB)	1 m^2 single-story residential building	57.08 kg CO_2e/m^2	Product stage	Slovakia	[174]
	Mass timber building (CLT and glulam)	1 m^2 of floor area	3153 kg CO_2e/m^2 floor area	Cradle-to-grave	USA	[101]
	Glulam, indoor use	1 m^3	235.89 kg CO_2e/m^3	Cradle-to-gate	Slovakia	[162]
	Glulam, outdoor use	1 m^3	256.03 kg CO_2e/m^3			
	CLT	1 m^3	113–375 kg CO_2e/m^3	Cradle-to-gate	USA	[175]
	CLT	1 m^3	252 kg CO_2e/m^3	Cradle-to-gate	Japan	[176]
	(a) Hybrid CLT building, with fireproofing (b) Hybrid CLT building, with charring	1 m^3	(a) 333.52 $kgCO_2e/m^3$ (b) 327.53 $kgCO_2e/m^3$	Cradle-to-gate	USA	[84]
	Wooden single-family house	$180.4m^2$ (house $150.4m^2$ & garage 30 m^2): (1 m^2 living area in a 100 years)	566.7 $kgCO_2e/m^2$ or 5.7 $kgCO_2e/m^2$/year	Cradle-to-grave	Sweden	[177]
	Single-family house; concrete slab & thermo-treated wood, wood frame & cellulose insulation	Gross floor area of 180 m^2 (main building 150 m^2 & garage 30 m^2)	2 kg CO_2e per m^2/year	Cradle-to-grave	Sweden	[178]
	Structural system of load-bearing walls of CLT	1 m^2 of heated floor area	224 kg CO_2e/FU	Cradle-to-gate	Italy	[85]
	CLT, timber frame & prefabricated timber: residential buildings	$1m^2$ GEA	77–207 $kgCO_2e/m^2$ GEA	Product stage and EOL stage	Germany & Austria	[67]
	CLT, timber frame and prefabricated timber: multi-story residential buildings	$1m^2$ GEA	18–178 kg CO_2e/m^2 GEA			
	CLT construction; residential building	$1m^2$ GEA	160 kg CO_2e/FU	Product stage and EOL stage	Germany & Austria	[105]
	Timber frame; residential building	$1m^2$ GEA	182–248 kg CO_2e/FU			
	Cork oak-based flooring (cork-CLT flooring)	11 m^2 (roughly 100 m^2)	574 kg CO_2e/FU	Cradle-to-gate	USA	[87]
	OSB	$1m^3$	124.5 kg CO_2e/m^3	Cradle-to-gate	Canada	[179]
	Glulam	1 m^3	112.01 kg CO_2e/m^3			
	CLT	$1m^3$	77.21 kg CO_2e/m^3			
	I-joist	1 m^3	66.11 kg CO_2e/m^3			
	Massive wood material wall system	1 m^2 of exterior wall	35.23 kg CO_2e/m^3	Cradle-to-gate	Italy	[88]
	CLT	$0.25m^3$	24.1 kg CO_2e/FU	Cradle-to-gate	USA	[89]
		$0.25m^3$	(a) 300 kg CO_2e/FU; (b) 221 kg CO_2e/FU	Cradle-to-grave: (a) Landfill (b) Recycle		
	Glued-laminated wood product	1 ^{m3}	102 kg CO_2e/m^3	Cradle-to-gate	Canada	[180]
	CLT	1 m^3	156.7- 185.69 kg CO_2e/m^3	Cradle-to-gate	USA	[181]
	CLT building	1 m^2	(a) 454.2 kg CO_2e/m^2 (without biogenic carbon) or 288.5 kg	(a) Cradle-to-grave (b) Material production	Norway	[115]

(*continued*)

Table 5. Continued.

Wood product category	Specific wood product	Functional unit	GHG emission factors	System boundary	Location	Source
			CO_2e/m^2 (with biogenic carbon); (b) 340 $kgCO_2e/m^2$			
	CLT building	8 stories, no basement; 3374 m^2 net floor area	(a) 203.4 $kgCO_2e/m^2$; (b) 268.0–280.5 $kgCO_2e/m^2$ (excl. module D) (b) 52.4–92.4 $kgCO_2e/m^2$ (incl. module D)	(a) Production (b) Cradle to-grave	Sweden	[182]
	Multistory building with CLT	4 stories, no basement; 1058 m^2 net floor area	(a) 120 $kgCO_2e/m^2$; (b) 510 $kgCO_2e/m^2$; (c) 220 $kgCO_2e/m^2$	(a) Cradle-to-gate, excl. carbon stock (b) Cradle-to-gate, incl. Carbon stock (c) Cradle-to-grave, excl. Carbon stock	Sweden	[183]
	CLT building	Per m^2 of gross floor area per year	54 $kgCO_2e/m^2$ per year (excl. biogenic carbon) and 49 $kgCO_2e/m^2$ per year (incl. biogenic carbon)	Cradle-to-grave	Australia	[114]
	Wood-frame construction	1 m^2 GFA	130.72 $kgCO_2e/m^2$ GFA	Cradle-to-grave	Austria	[131]
	Wood-solid construction		114.67 $kgCO_2e/m^2$ GFA			
	Low energy scenario pairs with wood frames		201.25 $kgCO_2/m^2$ GFA			
	Passive house scenario pairs using solid wood		260.10 $kgCO_2/m^2$ GFA			
	CLT	1 m^3	155.6–158.6 kg CO_2e/m^3	Different scopes combination	Global	[184]
	Wooden (Spruce) house of 100 m^2 of floor 1	1 m^2 /year	8.87 kg CO_2e/m^2 /year	Cradle-to-grave	Ukraine and Slovakia	[123]

method depends on the goal and scope of the study including, the modeling perspective and functional unit selected [194].

Challenges related to assumptions

LCA is often hindered by erroneous claims and biased assumptions, particularly regarding product lifespans and end-of-life scenarios, making it difficult to verify facts. The choice of assumptions can significantly influence the conclusions drawn [151]. In particular, end-of-life modeling assumptions have been shown to significantly impact the results of LCA studies [195,196], and standard assumptions may lead to inaccurate results [197]. Even small differences in assumptions can drastically alter conclusions and hinder the applicability of an LCA [198]. For instance, assuming a uniform product lifespan across different regions ignores factors like climate, maintenance, and user behavior that greatly affect environmental impacts. Similarly, generic end-of-life treatment routes may not reflect local end-of-life management infrastructures, leading to misrepresentations of recycling rates, landfill emissions, or energy recovery potentials. For example, not considering methane emissions at landfall and ignoring dynamic substitution can lead to a substantial overestimation of the potential of wood-use options [199].

Challenges related to LCA data

LCA is challenging to execute due to difficulties in gathering data, which can be time-consuming and subject to obsolescence, availability, and quality issues. These challenges, along with reliance on proxy data or inaccurate activity data, significantly influence LCA results [105]. The use of proxy data, in particular, introduces substantial uncertainty and may not accurately reflect the actual results [158]. The use of non-local databases, due to a lack of local LCI, may lead to inaccurate results. Traditional LCAs often depend on industry-average data or more generic secondary data from commercial databases, but these datasets are based on methodological assumptions that can result in generalized or misleading outcomes. Additionally, LCA and market data are available only for well-established product groups, and these values are likely to change over time [151]. Thus, using proxy data and lacking country-specific data in existing life cycle databases remains a significant barrier [94,200,201] and can introduce large uncertainties into the analysis [94]. Therefore, when selecting data for LCA, it is essential to account for its technological, geographical, and temporal relevance to ensure more robust and representative outcomes.

Challenges related to LCA tools or LCA software, and databases

LCA results can vary depending on the tools and software used, which often exhibit distinct differences [81]. Variations in LCA results may arise from the use of different commercial software and databases. For example, much of the data in Ecoinvent (SimaPro) and GaBi (Sphera) is based on European averages or global estimates [81]. Changes in software calculation methods, such as default values or settings, can also impact LCA results [202]. Additionally, differences in LCI data between databases due to differing methodological choices and variable cut-off criteria. A notable example is a case study on MDF production in China, where the same input data yield varying carbon footprint results under different methodologies [203]. Another challenge is the lack of transparency in LCA databases and tools.

Challenges related to LCIA methods

LCIA compares environmental impacts, but it is sensitive to the choice of impact metric [198], and varies depending on the LCIA methods used. LCIA methods are often site-dependent, context- and location-specific, with a diverse range of impact categories. Many LCIA methods focus on continental Europe, and their features vary according to scope and modeling objectives. These methods also differ in the impact categories they address, such as climate change, eutrophication, or human toxicity, which can influence the interpretation of results. The most evaluated impact is linked to global warming potential, with a particular focus on GHG emissions [204]. Variations in system boundaries and characterization models further contribute to discrepancies in results, including differences in the units of measurement used. As a result, LCIA outcomes often lack uniformity and are difficult to compare directly. Importantly, LCIA methods do not provide value judgments or allow for straightforward comparisons between impact categories due to the differing nature of the units involved.

Moreover, the normalization and weighting phases of LCIA introduce a level of subjectivity, as they often depend on value-laden assumptions or regional policy priorities. Impact indicators may also be calculated and reported differently depending on the characterization method applied. One of the key limitations is the absence of a universally accepted single-score output, which makes it difficult to synthesize and communicate the overall environmental impact of a product or system. The methodological differences can cause the same data to produce different results. This challenge is also common in carbon footprint methodologies such as PAS 2050, the GHG Protocol, and ISO 14067 for product carbon footprints [203].

Challenges related to the choice of biogenic carbon accounting approach

The absence of a global standard for measuring and accounting for biogenic carbon creates challenges for accurate carbon accounting. The classical LCA commonly neglects the impacts of biogenic carbon [205], and does not address temporary carbon storage, their timing and delayed emissions because it is assumed that CO_2 is emitted as a single emission after a certain storage period [107,206,207]. However, emissions and sequestration of biogenic CO_2 commonly occur at different points in time, posing a varied amount of effect on climate change [208]. Thus, the choice of the biogenic carbon accounting approach (0/0, −1/+1, −1/0, and −1/+1*) roots the variability of the LCA results [61,209]. While standards like ISO 21930:2017 and EN 16449:2014 guide wood and wood-based products, they are not consistently applied in practice. ISO 21930:2017, for instance, stipulates that biogenic carbon should be recorded as a negative emission (carbon removal) when incorporated into the product system, and as a positive emission when released (e.g. through combustion). In the construction sector, standards such as ISO 21930, ISO 21931, EN 15978, and EN 15804 commonly adopt the −1/+1 approach. However, many studies fail to fully comply with these standards, and the lack of harmonized verification frameworks further undermines LCA quality and fair comparison. This inconsistency underscores the need for stricter and more uniform applications of ISO standards when dealing with biogenic carbon in building product LCAs.

A comparison of biogenic carbon accounting methods is presented in Table 6. The first approach is the 0/0 method, which considers neither sequestration in the production stage nor releases of biogenic carbon at the end of life, thus assuming carbon and climate neutrality of wood products [209]. While easy to apply, this method can obscure the actual benefits or drawbacks of using biogenic materials. The second approach is the −1/+1 method, which tracks biogenic carbon throughout product's lifetime irrespective of the end-of-life treatment [157,209]. The −1/+1 approach provides an overview of all biogenic carbon flows. However, −1/+1 approach, when applied selectively to certain life-cycle stages, can yield a net negative global warming score, potentially leading to inaccurate or misleading conclusions [210]. When wood comes from sustainably managed forests, (e.g. replanted at the same rate it is harvested), the emissions (+1) are balanced by the prior sequestration (–1). Thus, biogenic carbon storage in wood product can be added, as a negative value, to the GWP indicator [211], and discharged at the end-of-life [105]. However, the uptake (–1) and release (+1) may occur over different time scales, meaning carbon neutrality is not always immediate. The mismatch of the biogenic carbon balance is a major source of variability in the −1/+1 method and of deviation to the results based on the 0/0 approach [209]. It is simple to apply an −1/+1 kg CO_2e for inputs and output of biogenic carbon but it does not consider the potential positive or negative effects of the temporary storage of the biogenic carbon in the product stock [115]. The third approach is the −1/0 method, which includes biogenic carbon as a credit, ignoring its end-of-life. While easy to apply, it risks overestimating the climate benefits of wood products by ignoring eventual carbon release. The fourth approach is the −1/+1* variation, which considers landfills and recycling as partially permanent sequestration of biogenic carbon, and thus fewer

Table 6. Comparison of biogenic carbon accounting methods in wood-based products.

Method	Carbon storage/ sequestration	Emissions at the end of life	Accounts for landfill/ recycling	Accounts for land use changes
0/0	No	No	No	No
−1/+1	Yes	Yes	No	No
−1/0	Yes	No	No	No
−1/+1*	Yes	Yes	Yes	No

emissions are accounted for in the end-of-life stage [61,209]. It assumes that some biogenic carbon will remain stored indefinitely or re-enter the product system, thereby reducing the amount considered as emitted. This approach introduces additional assumptions that can vary based on end-of-life management practices. Therefore, differences in approaches to biogenic carbon continue to make it difficult to compare findings [61]. Similarly, the end-of-life (EoL) allocation also affects LCA results at various levels [208]. At the product level, applying different EoL allocation methods increases the heterogeneity of LCA results [107,175,208]. Thus, the assessments must include realistic and region-specific EoL scenarios. Moreover, there is a lack of international integration for GHG emissions calculation [170], and no consensus on how to account for temporary carbon storage, as all approaches involve value choices that can lead to different results. Moreover, the current methods for accounting biogenic carbon fail to integrate land use, land-use changes, or carbon storage credits [210]. Kanellos et al. [204] highlighted the necessity of incorporating land-use and land-cover data into assessments. One way to achieve this is by integrating annually updated national GHG inventory data into LCA studies [212].

Challenges related to time frame

The timeframe used in LCA modeling influences the results due to time-related assumptions and EoL scenarios [109,208]. However, LCA studies often give little attention to the time profile of GHG emissions over a product's life cycle [48]. Different time horizons yield different metric values, leading to ambiguous comparisons. For instance, the impact on global warming, when modeled over different time frame such as 20, 100, or 500 years, results in different outcomes, making direct comparisons difficult [181]. The shorter timeframes may fail to capture long-term impacts effectively [107], while longer timeframes introduce higher uncertainty [213].

Challenges related to LCA precision

Current LCAs rely on a static basis and commonly fail to integrate temporal considerations, with only a few considering dynamic effects over time [61,213]. Static LCA measures all life cycle stages impacts at static, single points in time. A static metric accounts equally for GHG emissions and uptake, regardless of the time profile [214]. In terms of metrics, the static LCA overestimates the long-term cumulative impact on climate change and fails to demonstrate that the result is an accumulation of negative and positive emissions [215]. Using static LCA overestimates the reduction in emissions in the short term, but over the long term, it underestimates the reduction in emissions [216]. In contrast, dynamic LCA methodologies provide a more accurate representation by evaluating the timing and evolution of both emissions and sequestration throughout the life cycle [217]. However, dynamic LCA is more complex and requires advanced modeling techniques [218]. Its results are also highly sensitive to the selected time horizon [219], which can influence the timing and magnitude of reported benefits [220]. The use of dynamic biogenic carbon accounting can affect outcome interpretations by redistributing environmental burdens between early and later life cycle stages [221]. For instance, applying dynamic LCA to building materials has been shown to better inform short- and long-term climate change mitigation actions [222]. Overall, incorporating temporal dynamics into LCA can substantially change the results and improve the precision and relevance of environmental assessments [223]. Another challenge lies in the type of LCA used: attributional LCA focuses on the processes directly involved in a product's life cycle and fails to quantify the overall system-wide change in emissions or removals resulting from an intervention, whereas consequential LCA aims to account for the broader system-wide effects, including all processes impacted by an intervention [224].

Challenges related to LCA uncertainty

Uncertainty is a critical challenge that can undermine the quality and potential applications of LCA results [225]. Many LCA case studies either omit uncertainty analyses entirely or include only minimal sensitivity evaluations [226]. Uncertainties in LCA can arise from various sources, including input parameters such as data values, service life estimates, characterization factors, and material quantities. Additionally, uncertainties stem from the accounting methods used to model these parameters. The level of uncertainty in LCA inventory data is relatively high, as is the degree of modeling uncertainty for endpoint damage categories. Methodological variation introduces uncertainties, as seen in the Ecological

Scarcity 2013 method, which is based on EU data, compared to the globally focused ReCiPe method [99]. Sensitivity to LCA methodology is rarely addressed [214].

Challenges related to LCA conducting and reporting

There is a lack of uniformity in assessment methods. Fragmentation, inconsistency, variability, and complexity are common in LCA studies, which undermine the comparability and reliability of results. Another related challenge is that LCA studies use a variety of units to communicate results, which hampers interpretation and makes cross-comparison difficult [227]. Using product category rules; environmental product declarations, which (in theory) allow for comparability between different products and materials fulfilling the same function [192].

Implications and future agenda

As outlined in Table 7, this review highlights several key challenges in LCA studies, including inconsistencies, data limitations, and unverified assumptions, all of which undermine comparability and reliability of LCA. In response, Table 7 presents a synthesized research agenda and implications for advancing LCA practice. Future research in LCA should prioritize the standardization of scope, system boundaries, and functional units to improve comparability across studies. Adherence to ISO 14040/14044, ISO 21930, EN 15804, and ILCD Handbook guidance are essential for consistency. Transparent assumption modeling, particularly in lifespan, end-of-life, and allocation scenarios are equally critical. Sensitivity analyses should be standard practice to assess the influence of assumptions on results. Furthermore, improving the quality, availability, and specificity of LCI data, especially through localized and primary sources, will help ensure more accurate and relevant outcomes. Moreover, encouraging open data sharing and the use of dynamic, region-specific datasets can reduce reliance on generic assumptions and improve the contextual relevance of LCA outcomes. In relation to SF calculation, to achieve more accurate and convincing results, estimations must be grounded in specific contexts to assess targeted product substitution, highlighting the need for advances in existing estimation methods [17]. Recent developments, such as ISO 13391-3:2025 and BS EN 18027:2025, offer comprehensive guidelines for the LCA of wood-based and bio-based products and their comparison with fossil-based alternatives. Future research should align closely with these standards to ensure consistent and coherent SF calculations. These future directions are not only grounded in persistent gaps identified in the literature but are essential steps toward strengthening the scientific credibility and practical utility of LCA in the context of wood-based products and beyond.

Sources of variability and uncertainty in LCA and SF assessments

LCA and SF are subject to various sources of variability and uncertainty. Table 8 highlights the main phases of LCA, the types of variability and uncertainty, and strategies for addressing these challenges. Variations in SFs arise due to differences in LCA system boundary definitions, assumptions, the inherent heterogeneity of wood and non-wood products, production techniques, the evaluated life cycle stages, data quality, methodological approaches, and whether biogenic carbon is included in the analysis [3,249]. Substitution assumptions, in particular, introduce significant uncertainty [37], as the evaluation of substitution effects relies on specific assumptions that can lead to substantial variation in outcomes [66].

Estimates of GHG emission reductions attributed to substitution are highly sensitive to both the underlying assumptions and the parameters used, contributing further to uncertainty [37,216]. These variations are caused by: (i) differences in estimation methods; (ii) uncertainties in input factors, like service life, characterization factors, and quantities [68,188]; (iii) uncertainties in emission calculations, such as definitions, model structure, and system boundaries; and (iv) uncertainties regarding product end-of-life scenarios [250].

The IPCC identifies major uncertainty factors as model accuracy, activity data, emission factors, parameters, and methodological choices. Uncertainty is also tied to spatial, temporal, and technical variability, as well as potential errors. Major categories of uncertainty include those related to model accuracy and completeness, process parameters, data variability, and differences in approaches and databases [4,246].

Table 7. Life cycle assessment aspects, challenges and future research perspectives.

LCA aspects	Challenges	Future research perspectives
Goal and scope definition: purpose and context; declared (functional) unit; system boundaries	• Under defined goals or not explicitly articulated or incomplete scope definition • LCA scope variation, not declaring ISO used, and inconsistent between studies. • Lack of explaining the purpose and context of the study [189] • Lacking strict adherence to ISO and European standards • In current LCA practice, functional units are not commonly used; variation and inconsistent, incomparable functional units are widespread • Different system boundaries limit the comparability of LCA results. • Results vary depending on LCA boundaries • Lack of uniform system boundary • Inherent cut-off error in LCA, and multi-functionality problem [228] • There is no commonly accepted approach how allocations made within LCA [12] • Use of mass or physical allocation without considering the causal relationship	• Describing the motivation for doing an LCA study, its purpose, and intended audience [229] • Declaring a clear scope definition in compliance with ISO 14040/14044 standards • Specifying life cycle scopes, assumptions made, and system boundaries [202] • System boundary should reflect the same functions or realities in each scenario [230] • Using specified and quantifiable functional unit according to ISO 14040, 14044, and LCA handbook of ILCD, for example 1 m^3, 1 t, or 1 m^2 as functional units for wood products [229] • Using a clear cut-off criterion, and identifying suited allocation method; avoid allocation by expanding the product system according to ISO 14044 (2006), or considering system expansion and substitution in LCA [228] • In Europe, EN 15804 standard (EN 15804:2012 + A2:2019/AC:2021) specifies product category rules (PCRs) for the environmental product declarations (EPDs) of construction products. • EN 15978 provides a framework for assessing the environmental performance of a whole building. • ISO 21930 provides the principles and requirements for the development of EPDs for building products. • ISO 21931 defines principles, framework, and overall approach for assessing the environmental performance of entire construction works (buildings, infrastructure), beyond just individual products. • Harmonizing verification frameworks and standards for LCA quality and fair comparison
Assumptions	• Inaccurate claims and biased assumptions • Assumptions of end-of-life modelling and parameters [195,196] • Lifespans vary greatly, and assumptions may yield inaccurate results [197] • There are no independent scenarios in LCA • Scenario modelling is often exploratory, focusing on what if scenarios	• LCAs can be improved using numerous alternative assumptions on system boundaries [48] • Considering entire life cycle of the materials using deliberate and realistic assumptions [197] • Use of scenarios to estimate most likely future impact • The end-of-life scenario should be grounded in current practices and technology • Undertaking a sensitivity analysis to validate assumptions and uncertainty, demonstrate how different assumptions affect the outcome [77,197,231]
LCA data	• There is no consistent method for collecting data • Data obsolescence problems, availability and quality – uncertainty, incomplete data, reliance on proxy data, missing or inaccurate activity data [105] • Although the same input data is used, different carbon footprint methods yield different results [203]	• Following the directions provided by LCA ISO standards for data collection • Quality LCAs require quality LCI data [232] • Connecting primary data up the supply chain, validating, relating data to unit process and functional unit, data aggregation and refining system boundaries

(*continued*)

Table 7. Continued.

LCA aspects	Challenges	Future research perspectives
	• Data quality and availability vary by life cycle stages • Absence of country-specific inventory data • Differences in the emissions factors used • Lack of published inventory data; data privacy and confidentiality • LCA data is time sensitive, and collecting data for all unit processes is pressing • Use of non-local databases, lack of country-site-specific data [94,200,201] • Dependence on generic or secondary data, and less specific data	• LCA data should be geographically, technically, and timely representative • Using large language models to address missing foreground flow data and inconsistency in background data matching [233]. • Input output LCI delivers the simple, better result and faster solution with more expanded system boundary • Assessing data quality by applying a scoring pedigree approach helps minimize potential bias, in line with the ecoinvent data quality guidelines [153] • Understanding the background of secondary datasets and accounting for them • Leveraging background databases like Ecoinvent for emission factors of research region [99]
LCA tools or LCA software, databases and datasets	• LCA depends on the employed tools and offers distinct differences [81] • Different in LCA software, databases and datasets • The lack of transparency in LCA databases and tools • Databases vary in their cut-off criteria and system modelling approach • Default value or default settings [202]	• Using consistent & up-to-date databases, such as Ecoinvent, Sphera, and PEF through ELCD • Exercising care when applying commercial software tools and data sources [81] • Utilizing localization method to obtain more representative results • In Ecoinvent database, in case no specific data is available, using proxy data but with cautious, and important to check reliable references that present some similarities
LCIA	• Potential impacts are anticipated rather than actual, as most data is sourced from databases rather than specific site data • Lack of clearly stating the impact assessment methods used • LCAs outcomes vary depending on the methods used • LCA results are quite sensitive to the choice of climate impact metric [198] • LCIA methods are site-dependent, context- and site-specific with varied numbers of impact categories • Misconceptions about the choice of time horizons and characterization of climate impacts of HWPs [234] • LCIA methods have a continental focus, primarily on Europe, and their features vary depending on their scope and modelling objectives • Variations in characterization models, and hyperregionalized approaches [227] • Limited impact categories and impact indicators accounted differently • LCIA methods to some extent based on subjective values • Weighting can introduce bias into the results as it involves assigning subjective importance to different environmental impacts • Different weighting methods give different results • Lack of a single score for environmental impact indicators	• Assessing highest possible number of environmental indicators [187] • Providing LCA results for different impact categories • Using the most recent LCIA methods (e.g. Impact World + v2.01) • Applying at least two LCIA methods to compare the impact results to each other (e.g. CML and TRACI), and (e.g. ReCiPe, IMPACT2002+, IMPACT world+) [235] • Checking the influence of the different impact assessment methods • Use of normalization, weighting and aggregation to provide a weighted single score index to compare the scores. However, for comparative LCAs, ISO discourages weighting to ensure that LCA results remain as objective as possible
Choice of biogenic carbon accounting approach	• Approach for calculating GHG emissions lacks international integration [170]	• Accounting biogenic carbon following the ISO 21930:2017 • Accounting for biogenic carbon and timing of emissions

(*continued*)

Table 7. Continued.

LCA aspects	Challenges	Future research perspectives
	• Misunderstanding and confusion related to biogenic carbon in the calculations • Different biogenic carbon accounting methods result in varying implications: 0/0, −1/+1, −1/0, and −1/+1* [61,209] • Emissions and sequestration of biogenic CO_2 occur at different time [208] • No agreement on how to account for temporary carbon storage and delayed emissions [206] • EoL allocation influences LCA outcomes at the different level [4,38]	• Showing biogenic carbon separately [202] • Harmonizing LCA techniques for biogenic carbon • Characterize climatic impacts from biogenic carbon if it is temporarily stored [115] • Dynamic approach for assessing biogenic carbon uptake is the most robust & transparent [210] • Considering time horizon, storage period, and rotation period [107,236] • Fossil and biogenic carbon flows need to be tracked across time [27]. Otherwise, a realistic solution to ensure reliability is to solely account for fossil carbon
Time frame	• Timeframe significantly influences the LCA outcomes [109] • Choosing different time horizons • Failing to take the time profile of GHG emissions into account • Dynamic inventory/impact assessment only works for cradle-to-grave assessments	• Time profile of GHG emissions should be given more consideration in LCA [48] • A longer time perspective such as a default timeframe of 100 years is necessary [74,107] • Use of dynamic LCA to study effects over time. Fixed time horizon for cradle-to-gate assessment is necessary
Precision in LCA	• Most LCA studies rely on static data, overlooking changes over time in inputs, outputs, and environmental impacts [213] • Static LCA commonly fails to integrate temporal considerations [61,213] • Static metric accounts equally for GHG emissions and uptake, irrespective of time horizon [214] • Static approach does not show that the result is an accumulation of negative and positive emissions [215] • Static LCA underestimates climate warming effects or overestimates mitigation contributions [216] • Dynamic approach is complex and requires sophisticated modelling [218] • Dynamic LCA results are highly sensitive to the choice of a time horizon [219], and the impacts may fluctuate due to the time dependency of the LCI data	• Tracking biogenic and fossil carbon separately on a year-to-year basis [175] • Using dynamic LCA to estimate climate effects within a given timeframe [216]. It incorporates temporal impacts in assessment • Static approaches enable comparison with a boundary which is constant through time and provide highly ambitious ideal references [218] • Integrating static comparisons with dynamic approaches [218] • Use of dynamic LCA [61], with a dynamic metric provides more specific, realistic, better resolution and complementary information [107,215] • Dynamic LCA is consistent when inventory and impact are time-dependent, and flexible enough to handle difference in timing [107,215] • Using dynamic LCA to reflect GHG emissions and uptakes with their timing [237] • Addressing methods for time-distributed biogenic carbon accounting [221] • Developing dynamic and spatially explicit LCA models
Uncertainty	• Variations, uncertainty, and inconsistence are widespread in the LCA • Complexity and a lack of knowledge about LCI data uncertainty • Uncertainties due to input parameters, service life, and characterization factors • Variation between different methods • Sensitivity of results to the LCA methodology is rarely addressed [214] • High levels of subjective interpretation involved in the LCA process	• Focusing on mid-point damage categories due to minimum overall uncertainty • Weighting in LCA can improve the relevance and accuracy of the results [238] • Using Monte Carlo analysis to estimate a mean standard deviation [85,239,240]. • Using multiple-criteria decision-making to compare alternatives [173] • Harmonize guide to uncertainty analysis • Avoid drawing a definitive conclusion without conducting sensitivity and uncertainty analyses [110,214], and communicating uncertainty.

(*continued*)

Table 7. Continued.

LCA aspects	Challenges	Future research perspectives
LCA conducting and reporting	• Lack of uniformity, inconsistent units, and non-compliance with ISO standards • Use of diverse types of units to communicate the LCA results [227] • Lack of harmonization in LCA methods causes inconsistencies in how different countries or regions apply, interpret, and regulate LCA practices.	• Identify significant issues/hot-spots and assess importance of assumptions • LCA reporting must be adhered to the ISO standards • Using product category rules; EPDs [192]. • Contribution and sensitivity analysis should be included in interpretation [241] • Increasing harmonization, rigour, and ensuring robust compliance [202] • Harmonizing the impact reporting metrics and units • Exploring approaches that perform well in different settings • Executing evaluation of series of checks: relevance; accuracy; completeness; consistency; calibration; validation; transparency; uncertainty and sensitivity checks

For example, using datasets with varying levels of specificity can lead to substantial differences in results [251]. Furthermore, variations in methods, assessment components, assumptions, and system boundaries can heavily influence LCA outcomes, raising concerns about the reliability and applicability of conclusions drawn [187,202]. Ultimately, the variability in LCA results highlights the challenges of making broad or universal conclusions [252].

Conclusion and future perspectives

The carbon impact of substituting non-wood products with wood-based products is quantified by comparing the lifecycle GHG emissions between products. However, estimating their impact is complex due to varying methods, particularly in how biogenic carbon is accounted for in LCA. Different approaches, such as 0/0, $-1/+1$, $-1/0$, and $-1/+1^{*}$, influence LCA results, as they involve value-based choices, meaning no method is entirely objective. These variations lead to differences in GHG emission estimates, which ultimately affect the estimation of SFs, which measure the amount of CO_2 emissions avoided per unit of substituted product. Accounting for biogenic carbon is particularly complex compared to fossil carbon. Fossil carbon emissions move in one direction, from fossil stocks to the atmosphere, whereas biogenic carbon is more intricate. First, it binds to biomass and leaves the atmosphere. After that, it is either released back into the atmosphere or moved to temporary storage. To ensure accurate LCA results, both flows must be included in the LCA calculations; ignoring either leads to incorrect results.

SF is calculated using different methods, including single SF, direct comparison, average SF, weighted and unweighted averages. The calculation of SFs depends on factors like material weight, carbon content, how often wood replaces non-wood, market conditions, and emissions per functional unit. Additionally, factors like the type of products, LCA focus, and assumptions about production technology and end-of-life options influence SF values. The diversity of these variables and underlying assumptions in LCA models lead to variations in the estimation of GHG profiles, which ultimately lead to variation in SF values. Moreover, there are no standardized rules for determining SFs, resulting in further variability in assessments. This makes calculates SFs a dynamic process, with values that fluctuate based on methodological choices and case-specific substitution assumptions. Thus, SFs are context-specific [3], and therefore, should be reported as a range between minimum and maximum values rather than fixed values to reflect their inherent uncertainties. Typically, SFs are higher for wood products used in construction than for those used in interior applications or furniture, with regional variations further influencing outcomes. Therefore, upscaling substitution benefits requires not only harmonized LCA comparisons but also scenario-based modeling that capture realistic market conditions and dynamics.

Table 8. Sources of variability and uncertainty of GHG substitution effects and LCA related issues.

LCA phases associated with variability and uncertainty	Sources of variability and uncertainty	Techniques to address variability and uncertainty issues in LCA aspects
Goal and scope definition Life cycle inventory LCIA (Choice of LCIA methods; choice of impact categories and classification; midpoint and damage characterization; normalization and weighting)	• Uncertainties: • Methodological choices (functional unit, boundaries, allocation methods, technology level, LCIA methods, time horizon, weighting method) • Unjustified assumptions, bias introduced, end-of-life uncertainties [195] • LCI data and different databases • Data gaps, LCI data location, unrepresentativeness (time, geographical, technical coverage) • Input parameters (service life, characterization factors, quantity), scenarios, model uncertainties • Parameter uncertainty (measurement errors, analytical imprecision, calculation errors) • Inaccurate input flow and emission factors • Regional differences in emission factors • Model and process parameters, interpretations • Characterization methods and factors • Inaccurate normalization data • Choice of weighting method • Variabilities: • Inventory variation (data variability, methodologies, databases) • Variability between sources and objects (unreliability, incompleteness, time-sensitive, varying geographic regions) • Spatial and temporal variability (e.g. variation at primary data and characterization level) • Material substitution variability • Substitution ratios and market compositions [42] • Differences in product design and manufacturing • Assumptions-related variability	• Methodological: • Adhere to ISO standards (ISO 14040 and ISO 14044) • For the calculation of SFs, comparable function units are required. • Using process flow and matrix for data inventory • Gathering more & better data for estimating [3] • Data gaps can be improved using proxy data and sensitivity analysis • Adding a time dimension to product system mapping for end-of-life uncertainty [195] • Communicate and characterize uncertainty clearly [242] • Using structured pedigree matrix approach to estimate uncertainty related with LCI data [225] • Characterization and analysis: • Midpoint-oriented characterization to minimize subjectivity and uncertainty • Focusing on mid-point impact categories to minimize overall uncertainty • Choice of reference value is important in normalization • Using several weighting methods [243] • Conduct sensitivity analysis and uncertainty analysis considering input parameter, and model uncertainties, as well as spatial, temporal, and technological variability • Conduct an uncertainty importance analysis to identify critical data points that exhibit both high influence and significant uncertainty [244] • Statistical and modeling techniques: • Correlation and regression analysis for parameter and model uncertainties • Scenario analysis for choices and temporal variability, and measures how results change when scenario changes • Scenario analysis quantified by resampling different decision scenarios [244] • Sensitivity analysis for input data and modeling choices, e.g. perturbation analysis [243], and identify how variations in parameters influence outcomes • Breakeven analysis [243] • Machine learning methods to overcome incompleteness or uncertainty in data [225] • At the LCIA level, parameter; data uncertainty can be evaluated using Monte Carlo analysis [243,245], and multi-criteria decision analysis [246] • Model uncertainty can be evaluated using sensitivity analysis [243], and model formulations [244] • Non-linear modeling to address model uncertainty • Non-probabilistic methods [225] • Dynamic and spatialized modeling, e.g. dynamic LCA modeling to address model uncertainty, spatial, and temporal variability. Dynamic LCA incorporates temporal impacts in assessment • Uncertainty propagation using Taylor series, probabilistic approach • Ranking correlation coefficients, regression coefficients, probability density function for variability of a specific parameter estimation [247] • Empirical evaluation: • Empirical evaluation to test model validity and uncertainty • Expert judgment can be relied upon in situations where statistical analysis is not feasible [248]

The diverse range of functional units, LCA focus, LCIA methods, and databases used in LCA studies make it challenging to achieve consistent and fair comparisons [170]. The primary sources of uncertainty include factors associated with the model, process parameters, data variability, and the application of different approaches and databases. While some variation is unavoidable due to the differences in wood and non-wood products, as well as the inherent uncertainties in LCA data, it is crucial to establish consensus on key principles. These include LCA methodological choices, consistent functional units, allocation methods, LCA system boundary, LCA software, metrics, and data sources. Special attention should be given to incorporating biogenic carbon and land-use-related emissions and removals into LCA calculations. This review, therefore, stresses the importance of harmonizing LCA methods, establishing consistent approaches for uncertainty analysis, and standardizing the calculation of SF. One possible approach is to focus solely on GWP-fossil emissions and exclude biogenic carbon to ensure a more equitable comparison when only the product stage is considered. Alternatively, if biogenic carbon uptake is included using the −1 method, biogenic carbon emissions should also be incorporated using the +1 method. However, concentrating on specific life-cycle stages and using the −1/+1 approach could lead to a net negative global warming score, which may result in misleading or incorrect conclusions [210]. Another potential solution is to standardize the impact assessment method and LCA focus to account for emission timing and end-of-life factors, and using tools like pedigree matrix method, sensitivity analyses, probabilistic and non-probabilistic methods and scenario analysis to improve results. To enhance the reliability of SF calculations, the review suggests using ranges rather than single values and taking regional and contextual differences into account.

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Data availability statement

The dataset used in this investigation is available within the article (Tables 1–8).

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