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Climate change substitution factors for Canadian forest-based products and bioenergy

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A R T I C L E I N F O

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A B S T R A C T

Evaluating the climate change mitigation potential of the forest sector requires a holistic approach based on forest carbon (C) sequestration, C storage in harvested wood products (HWP) and substitution on markets. High uncertainty is associated with substitution factors, that express avoided fossil greenhouse gas (GHG) emissions from the use of forest-based products in replacement of GHG-intensive materials and fossil fuels. Few studies have focused on the development of substitution factors in Canada, resulting in the use of unrepresentative generic data. Here, we provide a framework to reduce uncertainties related to substitution factors for primary wood products in a Canadian context. A life cycle assessment framework is used to quantify fossil GHG emissions for a baseline and a wood-intensive scenario. For solid product substitution, we focused on the construction sector and analyzed a range of innovative wood buildings with steel and reinforced concrete as alternative materials. We found non-weighted averages of 0.80 tC/tC for sawnwood and 0.81 tC/tC for panels. For energy substitution, we analyzed cases with different specifications on biomass product, facility type and alternative fossil fuel source in non-residential heat production and biofuel transportation sectors. We found a non-weighted average of 0.80 tC/tC for non-residential heat production and 0.51 tC/tC for biofuel transportation, that can be interpreted as 0.91 tC/tC for heavy fuel oil, 0.69 tC/tC for light fuel oil and 0.68 tC/tC for natural gas substitution. These results provide a benchmark for substitution factors in Canada, to help guide forest management strategies for climate change mitigation.

1. Introduction

Cement production, land use change and fossil fuel combustion are the principal causes of the increase in anthropogenic greenhouse gas (GHG) emissions worldwide (WMO, 2019). As engineering technologies continue to improve, there is an anticipated increase in forest-based products and bioenergy (Babí Almenar et al., 2023). This growth can be attributed to the climate benefits they offer in comparison to traditional building materials (such as concrete and steel) or fossil fuels (Churkina et al., 2020; Kouchaki-Penchah et al., 2022). Following the Paris Agreement, Environment and Climate Change Canada (2022) envisions a removal potential of 30 Mt CO_2eq by 2030 through Forestry (LULUCF) and Natural Climate Solutions, representing 13 % of the global Canadian target. In fact, end use of construction materials made up more than a third of the carbon footprint of Canada's second largest city – Montreal (Elliot & Levasseur, 2022). The contribution of the forest sector can be presented as three main levers: (i) increasing forest areas and thus, carbon (C) sequestration, through afforestation/reforestation, (ii) sustainable management of existing forests to increase C stocks in forest and harvested wood products (HWP), and (iii) increasing HWP and bioenergy markets share to substitute GHG intensive materials and fossil fuels (IPCC, 2014). Among all C pools, several studies have highlighted the substitution effect as the one providing the highest long-term climate benefits (Beauregard et al., 2020; Lippke et al., 2011; Paradis et al., 2019).

The substitution effect is defined as the climate benefit associated with the replacement of GHG intensive materials, or fuels, by forest-related products or bioenergy, that can deliver the same service while reducing atmospheric CO_2 emissions (Schlamadinger & Marland, 1996). It generates permanent C offsets, and is quantified by substitution factors (SF) that express the net GHG emission reduction per quantity of

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Abbreviations

C	Carbon
CLT	Cross Laminated Timber
EOL	End of life
EPD	Environmental Product Declaration
GHG	Greenhouse gas
GLT	Glue Laminated Timber
HWP	Harvested wood product
HTL	Hydrothermal liquefaction
LCA	Life cycle assessment
MDF	Medium-density fibreboard
ODW	Oven dry weight
OSB	Orientated strand board
SF	Substitution factors

wood use, a positive value resulting in emission reductions. Here, the substitution effect equates to the net fossil emissions (non-biogenic) divided by the biogenic carbon stored in the product (Smyth et al., 2014), and substitution factors are in units of tonnes of carbon of emission reduction per tonne of carbon used in wood product (tC/tC) (Smyth et al., 2017). The life cycle assessment (LCA) methodology framed by ISO14040 and 14,044 standards (ISO, 2006a, (ISO., 2006b) is used to compare life cycle GHG emissions of the wood and non-wood alternatives on a functionally equivalent basis.

A number of LCA studies comparing wood products to their functionally equivalent alternatives found that the use of wood tends to reduce fossil fuel consumption and associated GHG emissions (e.g., Bergman et al., 2014, Kouchaki-Penchah et al., 2023a). Wood products generally require less energy and fossil fuel resources for manufacturing operations (e.g., raw material extraction, transportation, transformation), and a large proportion of the energy use can be provided by biomass residues. Indeed, wood harvesting and processing generates a large amount of co-products that can be used as biofuels to replace fossil fuels, increasing the benefits of the substitution. Forest bioenergy can generate positive, neutral or negative effects on climate change mitigation, depending on key elements such as biomass sources, conversion processes, combustion efficiency and fossil fuels substituted (Laganière et al., 2017; Xie et al., 2023), but using residual biomass feedstock from forest industry processes generally results in positive impacts (Berndes et al., 2016).

A broad range of studies about the substitution effect can be found in the literature, from micro-level studies based on a specific product to macro-level studies that assess substitution at the regional or market level. Results are highly variable due to the diversity of studies. Many factors can influence the SF value, such as methodological choices (e.g., system and temporal boundaries, or allocation procedures (Hurmekoski et al., 2021; Sathre and O'Connor, 2010) and physical factors (e.g., location, industrial processes, energy sources, type of alternative material compared (Leskinen et al., 2019; Xie et al., 2023). A *meta*-analysis conducted by Sathre & O'Connor (2010) based on 21 studies found an average value of 2.1 tC/tC, with a range of values going from −2.3 to 15.0 tC/tC, while values from 0.5 to 1 tC/tC are associated with forest bioenergy substitution. Negative values illustrate the combination of the worst-case scenarios from the different studies analysed, in which the wood option results in more GHG emissions. However, such scenarios are judged less probable and unrepresentative of real-life cases. The literature review by Leskinen et al. (2019) analysed 51 studies and 433 different substitution factors and found an average of 1.2 tC/tC with 95 % of the values ranging between −0.7 and 5.1 tC/tC. The average value can be separated in 0.8 tC/tC for the material production stage, and 0.4 tC/tC for the impact of energy recovery substituting fossil fuels. Leskinen et al. (2019) also found an average SF for structural and non-structural wood in construction of respectively 1.3 and 1.6 tC/tC including energy recovery. The more recent review by Hurmekoski et al. (2021) analyzed quantitative results from 44 papers on wood product substitution. Results ranged from 0.27 to 1.16 tC/tC with an average of 0.55 tC/tC. If average values reported by Sathre & O'Connor (2010), Leskinen et al. (2019), and Hurmekoski et al. (2021) put forward a general agreement that wood substitution can generate GHG emissions reduction, their values are not suitable to be used in a specific case study (e.g., evaluation of the climate change mitigation potential of the Canadian forest sector) because they include a wide range of studies with different methodological choices and geographical locations.

A study conducted by Smyth et al. (2017) estimated substitution factors at the national level in Canada. They developed SF for primary wood products sawnwood and panels resulting in respectively 0.54 tC/tC and 0.45 tC/tC. For bioenergy, two feedstock scenarios were evaluated, they found 0.47 tC/tC for constant supply (fixed biomass feedstock) and 0.89 tC/tC for constrained supply (feedstock to match with each region's heat demand). Product SFs were based on six HWPs (softwood lumber, hardwood lumber, plywood, particleboard, OSB and MDF) and on six different end uses (single-family-home, multi-family home, multi-use building, flooring, furniture and decking) to then be weighted using national consumption statistics. The greatest benefits in terms of emissions reduction were observed by the substitution of steel and concrete in buildings, and by the replacement of fossil fuels in bioenergy heat or combined heat and power plants. Xu et al. (2018) derived SF from Smyth et al. (2017) and targeted only steel and concrete substitution in single-family homes, multi-family homes, and multi-use buildings to find SF values of 2.1 tC/tC for sawnwood and 2.2 tC/tC for panels. All these SFs are now used in provincial and national forest management studies on climate change mitigation (Beauregard et al., 2020; Smyth et al., 2020; Moreau et al., 2022) but uncertainties remain about methodological choices and assumptions that can highly impact the variability of the factors and therefore affect their validity. For instance, Moreau et al., (2023) shows how the choice of substitution factors in integrated studies markedly impacts the outcomes. The selected value of the substitution factor becomes a determining factor, as it has the potential to render a forest management strategy advantageous or disadvantageous.

Smyth et al. (2017) highlight the lack of comprehensive data from comparative LCA studies on end uses as a main source of uncertainty for product SF, leading to data generalizations that might differ significantly across various projects (e.g., bill of material and GHG emissions of buildings). These values are highly variable and should be based on multiple building samples for each end use (e.g., single-family home, multi-family home, multiuse building). Material systems, construction techniques, building functionalities and geographical locations are unique to each project and influence the bill of materials and associated GHG emissions (Milaj et al., 2017). Furthermore, the wood materials evaluated by Smyth et al. (2017) do not encompass all wood product types. Notably, engineered wood products such as Cross Laminated Timber (CLT), Glue Laminated Timber (GLT), and Laminated Veneer Lumber (LVL) were not included in their analysis. As the markets for these products are in expansion, and as they present high mechanical and structural properties, they should be considered as high potential substitutes to steel and concrete so that their substitution effect should be assessed. Additionally, using LCA data from studies conducted outside of Canada and before 2006, as seen in Smyth et al. (2017), could lead to increased uncertainty.

Regarding energy substitution, Smyth et al. (2017) provide energy substitution factors for each forest management unit (FMU) in Canada, based on provincial energy fuel mix and heat and electricity consumption. However, bioenergy deployment is mainly guided by governmental strategies aimed at replacing particular fossil fuels (Beauregard et al., 2020), indicating a research gap in bioenergy substitution factors tailored to specific fuel substitutions. Moreover, there is a wide range of bioenergy products that can be assessed as substitutes.

To effectively evaluate the GHG mitigation potential of the forest sector at the provincial or national level, a combination of different approaches is required, as carbon accounting must consider forest products and substitution on markets. Strategic decisions concerning forest management for climate change mitigation must be based on reliable data. The uncertainty associated with substitution factors can be reduced by providing more detailed, local, and diverse results. The objective of this research is to create an array of substitution factors at a micro-level for construction and bioenergy systems, emphasizing the use of HWP tailored to the Canadian context. These substitution factors could then be used in provincial and national-level studies. Initially, this involves reviewing up-to-date Canadian LCA studies that compare biomass products with their alternatives. Subsequently, specific substitution factors are calculated from these studies to match with the holistic methodology used in Canada to assess the climate change mitigation potential of the forest sector.

2. Materials and methods

2.1. Analytical framework

Several differences in methodology relative to the calculation of substitution factors are found in the literature (see SM1 for materials SF and SM2 for energy SF). The definition of the frameworks and their utilization context are important to ensure applicability and validity. In our study, we followed the methodology presented in Smyth et al. (2014), Smyth et al. (2017), and Moreau et al., (2023), in which SFs are used within a broader assessment framework that considers carbon stock changes in forests and HWPs, and where each primary wood product is modelled with a specific half-life time and an end-of-life scenario. However, while the broader assessment frameworks quoted in these studies apply weighted SFs, we estimated unweighted SFs as per the approach taken by Leskinen et al. (2019). Please refer to Taylor et al. (2024) for a comparison between SFs and weighted-average SFs. Biogenic carbon emissions are not included in the scope of this study, as they are already accounted for in forest and HWP carbon models. Substitution factors reported in this study, that consider only fossil GHG emissions, should not be used without considering this broader framework.

The methodology to calculate product and energy substitution factors within this framework is presented in Fig. 1. The first step consists in measuring the substitution effect at the product level, according to at least two scenarios of material/energy consumption that provide the same function: baseline and wood-intensive. The baseline scenario represents the traditional assembly or consumption of materials/energy, while the wood-intensive scenario promotes the use of wood. The substitution factor *SF* is then calculated for these two scenarios with Eq.1, from Sathre & O'Connor, (2010). It expresses the avoided fossil GHG emissions per quantity of wood in units of carbon (tC/tC).

$$SF = \frac{GHG_{non-wood} - GHG_{wood}}{WU_{wood} - WU_{non-wood}} \quad (1)$$

where.

GHG_{wood}: Fossil emissions from wood-intensive [t C].

$GHG_{non-wood}$: Fossil emissions from baseline [t C].

WU_{wood}: Amount of wood in wood-intensive [t C].

$WU_{non-wood}$: Amount of wood in baseline [t C].

Fossil GHG emissions for both scenarios were taken from the LCA studies collected and are based on ISO14040 and 14,044 standards (ISO, 2006a; ISO, 2006b), ensuring a functionally equivalent comparison. The C content of GHG emissions (GHG_{wood} and $GHG_{non-wood}$) was calculated from CO_2eq using the molecular ratio of 12/44, assuming that CO_2eq represents exclusively CO_2 emissions. The quantity of wood used has been estimated from the list of materials available for each study, and a C content ratio of 50 % was assumed over the product oven-dry weight to generate the amount of carbon in HWP products (WU_{wood} and $WU_{non-wood}$) (Sathre & O'Connor, 2010). Only the amount of wood contained in the final product was considered in the calculation. Most of the SF calculated in this study are associated only with the production stage, which included fossil emissions from forestry, harvesting, extraction of raw materials, transportation, and product manufacturing. When possible, the construction and use stages, as well as the end-of-life stage (transport only), were considered in the calculation, as it brings more completeness and representativeness in the SF values by avoiding potential trade-offs that may exist during different product system stages. The subsequent step of the methodology consists of estimating an average SF by sub-category (e.g., construction structural, construction non-structural, furniture, etc.) of HWP end uses by comparing an overall mix of HWP end uses to a mix of alternative products. The calculation of weighted SF was not included in the scope of this study due to the lack of data regarding consumption statistics of HWP end uses. This could be done for specific cases when integrated in holistic approaches based on forest, products, and markets. Here, we calculate SF by end-use and by sub-category. Product and energy substitution present differences in sub-category and end-use classification. Their calculation was thus separated. Product substitution refers to three primary wood products of the HWP basket (sawnwood, panels, and pulp and paper), and to seven informal sub-categories e.g., construction, furniture, packaging, paper, chemical products, and textiles. A variety of the main end uses are shown in Fig. 2. In this study, we focused mainly on construction end uses shown in green due to limitations regarding data availability in the literature. Additional LCA studies are needed to satisfy all end uses and sub-categories.

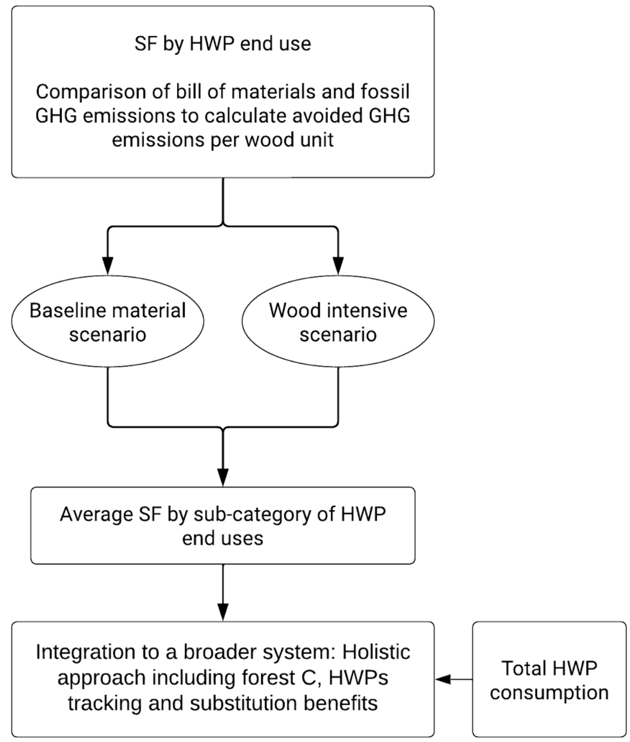

Fig. 1. Overview of the methodology for the calculation of substitution factors (C = Carbon; HWP = Harvested wood product; GHG = Greenhouse gas; SF = Substitution factor).

The energy substitution methodology is illustrated in Fig. 3, focusing on bioenergy as the primary wood product in the HWP basket. Unlike product substitution, it is divided in sub-categories that correspond to the output of the forest HWP model. Therefore, to improve precision, SF should be calculated separately by these sub-categories, rather than combined (Xie et al., 2023). End uses of bioenergy systems should be

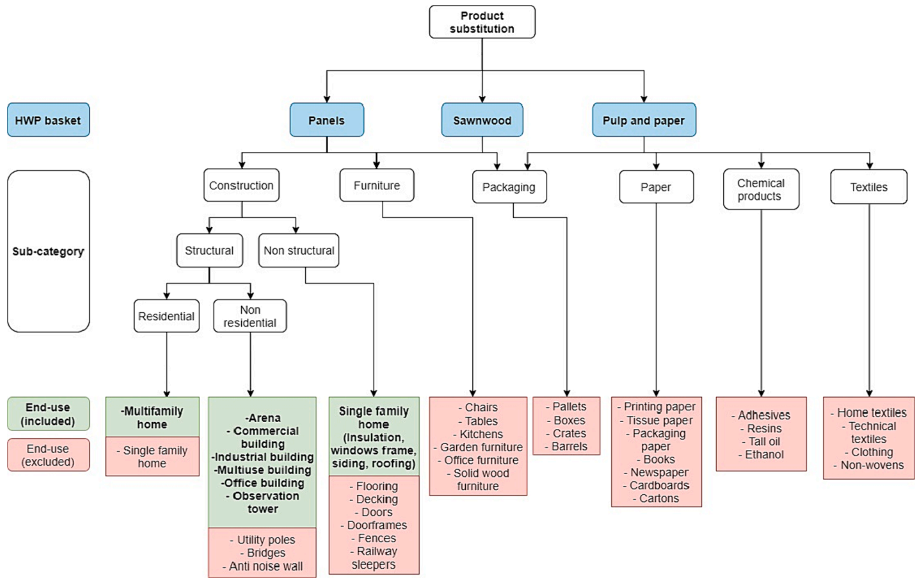

Fig. 2. Overview of the framework for product substitution (HWP = Harvested wood product).

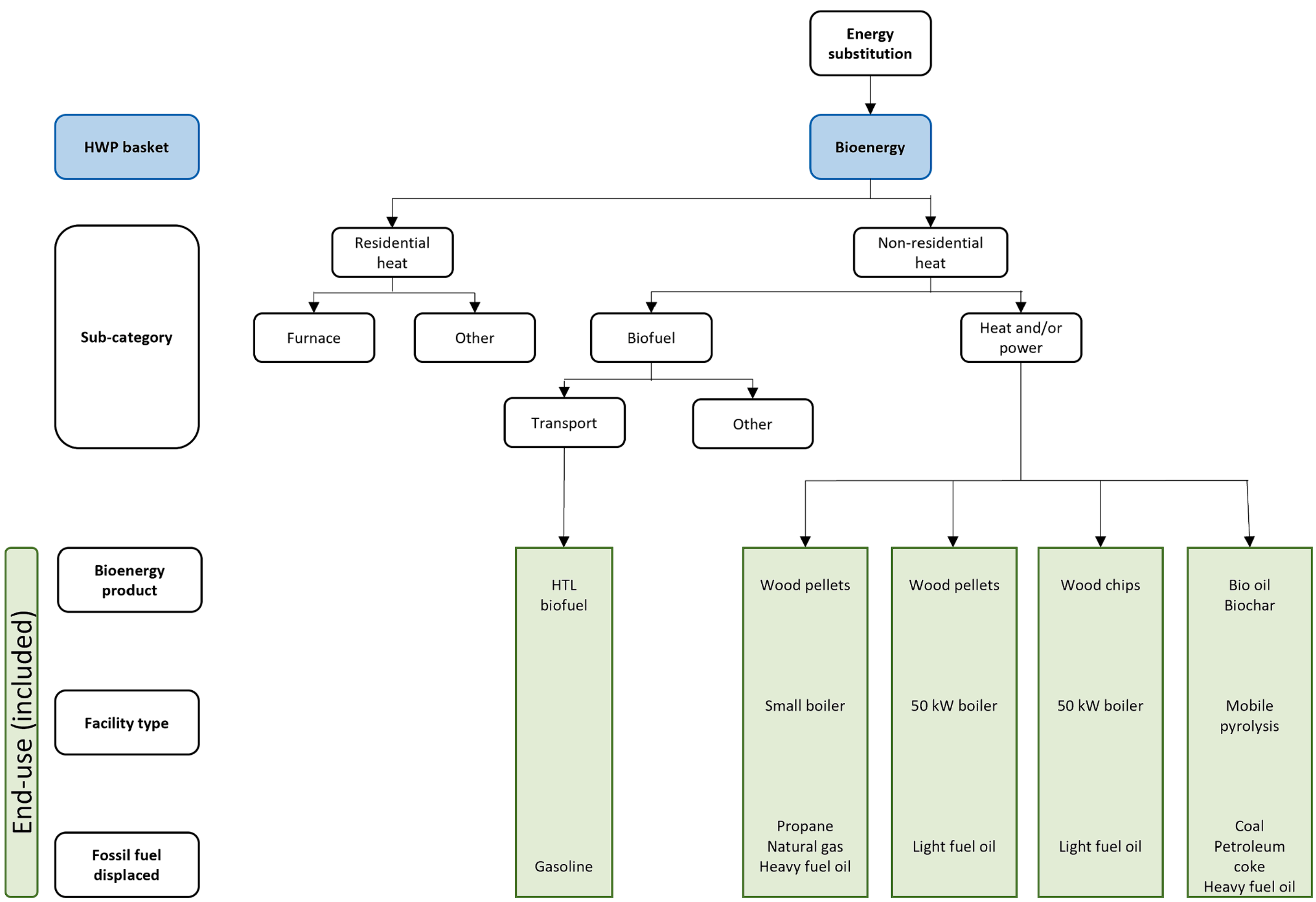

Fig. 3. Overview of framework for energy substitution for non-residential heat combinations covered by the case studies available (HTL = Hydrothermal liquefaction).

seen as a combination of three different components that greatly influence the SF value, i.e., the type of bioenergy product, the facility type, and the fossil fuel displaced. Given the extensive range of potential combinations, this study selectively addresses those depicted in Fig.3. Here, we focus on analyzing a range of alternatives for non-residential heat and one case for biofuel transportation, as per LCA data availability.

2.2. Substitution factors for material products

Sawnwood and panels can be found in a broad range of manufacturing sectors (construction, furniture, packaging, etc.). According to Chen et al. (2014), the construction industry is the main user of solid HWP in Canada, where 64 % of sawnwood is processed, as of 77 % for structural panels and 41 % for non-structural panels. Also, government strategies tend to encourage the use of wood in construction for its lower carbon footprint, therefore, the construction sector represents a great avenue for substitution (Meyer et al., 2024). In this study, we analyzed a variety of buildings and focused mainly on new constructions in the residential and non-residential sectors to generate substitution factors for sawnwood and panels. Each wood-intensive scenario is compared to a functionally equivalent scenario from an alternative material. Most of the studies are related to structural applications and compare wood to steel or reinforced concrete. However, one study includes the comparison of non-structural HWP end uses to fiberglass, brick, asphalt, and vinyl.

For each study, the amount of wood used was estimated for both alternatives from the bill of materials. Types of HWP found in these studies are listed in SM3.1 and include engineered wood products like CLT and GLT. Oven-dry weight, moisture content, and wood ratio of each product were used to calculate the total carbon content of wood in buildings, as most of the wood quantities were available in volume (m^3). Data regarding wood materials were taken from a variety of North American environmental product declarations (EPD), or the Athena Sustainable Materials Institute. When data were not available, a moisture content of 15 % was assumed to calculate the oven-dry weight. The ratio of sawnwood and panels was calculated from the difference in the amount of wood used between the baseline and the wood-intensive scenario.

Additional distinctions within each study were made between their system boundaries. Each study was categorized based on EN 15978 (European Standard, 2011), which is widely used in North America for building life cycle assessments. Therefore, not all of the studies presented here can be compared with each other since they have different boundaries. In our study, we have included modules A1-A3 for all studies. Additionally, whenever possible, we have included the construction stage (A4-A5) and use stage (B1-B7) as reported in SM3.2. They are the insulation, siding, roofing and windows case study, one of the multistorey apartment case studies, one of the multiuse building case studies, two office building case studies, the arena case study and two office building case studies. However, we have excluded modules C1-C4 from the study as end-of-life (EOL) biogenic emissions are generally analyzed through the use of a harvested wood product model. This model tracks the biogenic carbon through the different products' life cycle, including end-of-life. In three studies, the separation of the EOL stage from global LCA result was not possible and was kept in the SF calculation. However, the EOL impact of these studies only includes fossil GHG emissions, and has a very low contribution, so that the impact on the SF value is almost negligible and double counting of biogenic carbon is avoided. All comparisons were made from functional units that include an equivalent building floor area, a number of storeys and, when the use stage is considered, a lifetime. Some studies had considered all construction materials while others, structural materials only. While considering all materials is preferable to enhance data accuracy, Alain (2015) found that structural materials account for the majority of GHG emissions relative to a building. Only one study included operational energy and water use (modules B6 and B7).

Data needed for the calculation of substitution factors (bill of materials and GHG emissions) were obtained through diverse references i.e., literature, *Programme de vitrine technologique* (PVT) and Gestimat database. The PVT is a special program from the government of Quebec, which aims to reduce GHG emissions by supporting projects which design and build innovative wooden building solutions. Studies from this program are identified as PVT-1 to PVT-7 in SM3.2 and conform to the GHG quantification protocol of structural materials (Roberge, 2018). This protocol includes third-party review of calculations and results of GHG emissions and structural design.

Gestimat is an online tool developed by CECOBOIS that estimates and analyzes GHG emissions due to manufacturing (modules A1-A3) of different building structure scenarios (e.g., wood, steel, reinforced concrete). In the Gestimat tool, it is possible to either enter detailed material quantities for a given project, or to specify the main building restrictions (e.g., end use, number of storeys, floor area, construction systems, construction components, etc.) and the database will automatically estimate the quantity of structural materials needed. GHG emissions are estimated by materials. The Gestimat database includes 33 different construction materials mainly from wood, steel and concrete. Each of these materials is linked to the most local manufacturing emissions factors, provided by the International Reference Centre for the Lifecycle of Products, Processes and Services (CIRAIG) through diverse EPDs and the ecoinvent version 3 database (Wernet et al., 2016). None of the other lifecycle stages as construction, operation and end-of-life are included, only modules A1-A3 are considered.

Within the Gestimat database, three different typical buildings are modelled (office building of 1 storey, office building of 4 storeys and industrial building). For each building, two wood construction techniques (light frame, and post and beams), and two alternative scenarios (steel and reinforced concrete) were analyzed. They are identified as GES-1 to GES-3 in SM3.2. As they are prospective building approaches, they should be interpreted differently. SM3.2 brings the description of the studies analyzed in this study coming from the literature, PVT and Gestimat database.

2.3. Substitution factors for energy

Energy substitution was estimated for six end uses for which wood biomass substitutes fossil fuels. All references come from the literature and are based on comparative LCA studies located in Canada that analyze the environmental impact of the use of woody biomass over fossil fuels on an energy production equivalent unit (e.g., production of 1 GJ). Substitution factors are specific to each end use, including the types of biomass product (e.g., pellets, chips, bio-oil, biochar and hydrothermal liquefaction biofuel), the facility type, and the alternative fossil fuel sources (light fuel oil, heavy fuel oil, natural gas, coal, petroleum coke, and gasoline). Five of the cases were in the context of heat production for non-residential applications, and one for biofuel transportation.

To calculate energy SF, we used Eq.1 from Sathre & O'Connor (2010). For each case, at least two scenarios were compared, one with bioenergy from wood biomass and another with fossil fuels. The amount of biomass needed to fulfill the energy demand (functional unit) was taken within the study's data, as well as the resulting fossil GHG emissions reductions. GHG emissions from biogenic sources were excluded as they are generally considered through the use of a harvested wood products model. Therefore, for bioenergy systems, only emissions from pre-consumption of the supply chain, i.e., extraction, transport, and pretreatment of the fuel were considered. Emissions from combustion were included only for the fossil fuel scenarios. The carbon content for each bioenergy scenario was calculated from the oven-dry weight of the biomass, with moisture content information available in each study.

The values of energy SF are influenced by three main factors: the type of bioenergy product, the conversion efficiency of the facility, and the type of fuel displaced. The six case studies were about 1) bio-oil and biochar from mobile pyrolysis substituting a mix of heavy fuel oil, coal and petroleum coke, 2) hydrothermal liquefaction biofuel substituting gasoline, 3) wood chips from harvest residues substituting light fuel oil, 4) wood pellets from harvest and mill residues substituting light fuel oil, 5) wood pellets from mill residues substituting heavy fuel oil, and wood pellets from mill residues substituting natural gas. Details of each case study related to these factors are listed in SM3.3.

3. Results

3.1. Substitution factors for harvested wood products

SM3.4 presents substitution factors for each study as well as the data needed for their calculation, i.e., the difference of wood use between the wood-intensive and the baseline scenarios expressed in oven-dry tonne (ODT), and net avoided fossil GHG emissions in t CO_2 eq. Overall, 18 different studies relative to the construction sector were analyzed, which resulted in SF varying from 0.29 to 1.86 tC/tC, with an average of 0.80 tC/tC (sawnwood and panels combined). The ratio of sawnwood and panels was calculated based on the increase in the amount of wood use between wood-intensive and baseline scenarios, where most of the studies showed a high ratio of sawnwood use versus panels.

The majority of studies are related to structural construction of multistorey buildings with steel or reinforced concrete (RC) as baseline material. Only one study (Salazar et al., 2009) compared non-structural components (insulation, siding, roofing, windows) with different baseline materials (fibreglass, brick, asphalt, vinyl) for a single-family house. Therefore, the results of this study are highly focused on structural construction, more specifically on the substitution of steel and reinforced concrete. The SFs proposed should not be used to generalize all types of wood use and all baseline materials.

3.1.1. Product substitution by end use

SF values vary depending on the quantity of wood use and the net avoided emissions. The type of wood construction i.e., light wood frame, post and beams, and mass timber, tends to influence the quantity of wood use and thus, impacts SF values. Light frame wood constructions generally use a small amount of wood but are limited in the number of storeys. In comparison, mass timber constructions generally use a large amount of wood, resulting in a lower SF, as illustrated in Salazar et al. (2009), PVT-1, and Robertson et al. (2012), which show the three lowest SF (0.29, 0.4, and 0.3 tC/tC respectively). However, high SF values might also be reached in mass timber constructions, as shown by Essoua & Lavoie (2019) with 2.12 tC/tC and Grann (2014) with 1.13 tC/tC, because they displace a large amount of reinforced concrete.

Average SF were calculated by grouping the studies within eight types of end uses, as illustrated in Fig. 4. Each type of end use is associated with an average value and a range of SF, based on the number of studies associated shown in parentheses. Adding studies for a given end use improves understanding of the uncertainty of the results. The multiuse building end-use relates to a mix of residential and commercial applications in the same building, mostly condominiums with commercial areas on the ground floor.

3.1.2. Product substitution by sub-category

Substitution factors per sub-category of end uses are presented in Fig. 5. SF for non-structural construction is 1.08 tC/tC, but the value relies on one single study. The average SF for structural construction is 0.79 tC/tC, based on 17 studies (Fig. 5(A)). Within these 17 studies, 5 are for residential buildings and 12 are for non-residential building, with average SF of 0.86 tC/tC and 0.78 tC/tC respectively (Fig. 5(B)).

The calculation of weighted SF for sawnwood and panels was out of the scope of this study due to the lack of availability of HWP consumption data. However, to get an overview of the difference between SF for sawnwood and panels, the avoided GHG emissions were attributed to each primary product from the mass ratio resulting in 0.81 tC/tC for panels and 0.80 tC/tC for sawnwood.

3.1.3. Difference between structural material and whole building results

Studies including only structural materials generate higher substitution factors, with an average of 0.87 tC/tC compared to 0.72 tC/tC for those considering the whole building. These results show that SF calculated from studies considering only building structures could be overestimated. This difference can be explained by the addition of other non-structural materials (aluminum, brick, gypsum and isolation) to respect national building codes. Therefore, the contribution of engineered wood products to the carbon footprint of buildings should be considered in absolute terms, rather than relative terms. Fig. 6 compares

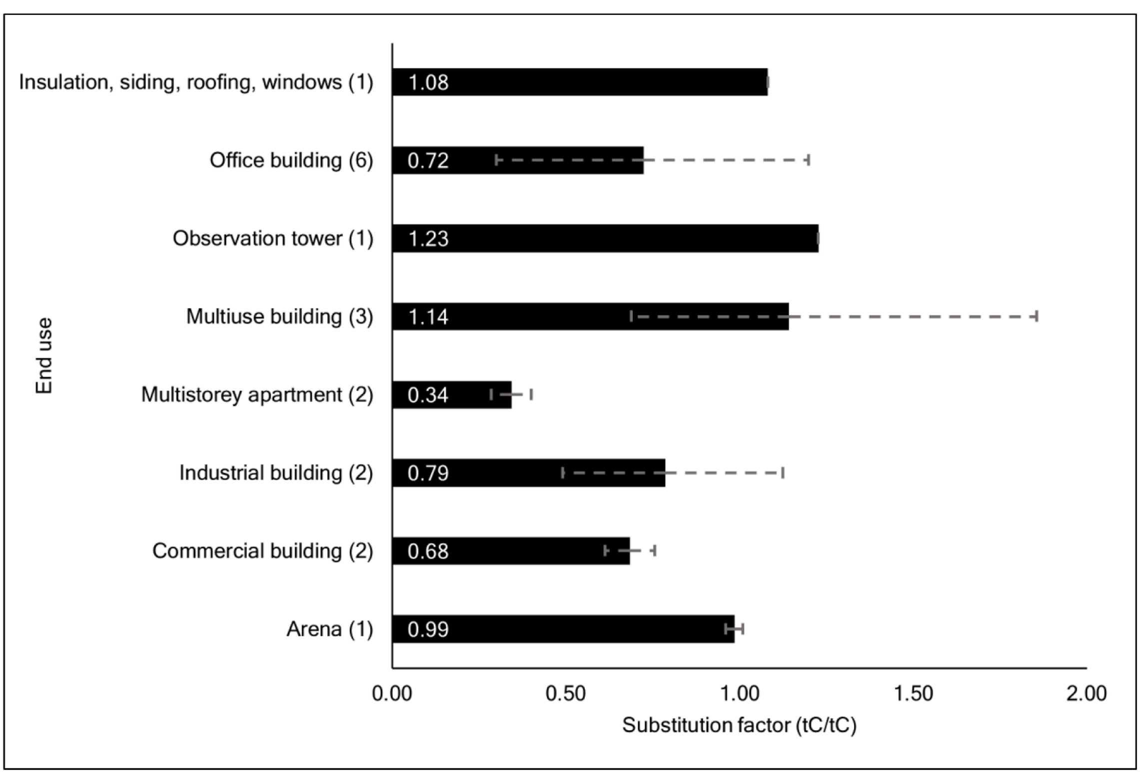

Fig. 4. Average product substitution factors by end use (tC = tonnes of carbon) with range indicated by error bars.

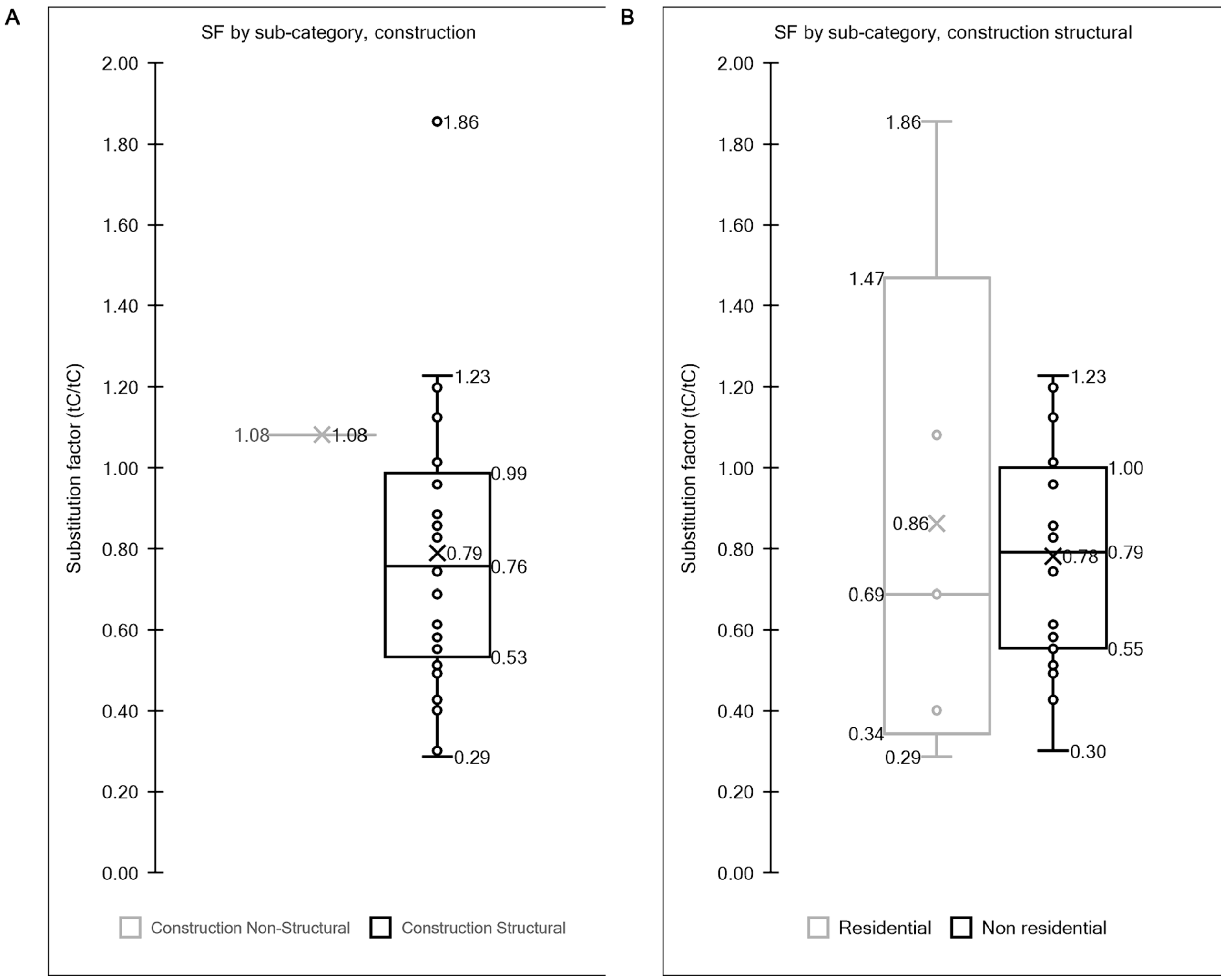

Fig. 5. A (left) Product SF by sub-category (tC = tonnes of carbon). B (right) Product SF by sub-category in structural construction.

substitution factors for studies considering the whole building and those including only structural materials.

3.2. *Substitution factors for bioenergy*

3.2.1. *Bioenergy substitution by end use*

Substitution factors for bioenergy are presented in SM3.5 for six different studies. SF values express the efficiency to displace the emission of fossil carbon by the use of wood biomass, in units of tC/tC. For example, a SF of 0.72 tC/tC means that for every kgC of burned biomass, 0.72 kgC of fossil GHG emission is avoided. Results vary between 0.51 and 1.15 tC/tC, only one study leading to a SF higher than 1. Wood biomass has a lower energy content per unit mass than most fossil fuels (Transition énergétique Québec, 2019). Therefore, to generate a specific amount of energy, it generally requires a greater amount of biomass than for other fuels resulting in SF lower than 1.

The amount of biomass needed to fulfill a functional unit of energy production varies depending on the energy and moisture content of the bioenergy product and the efficiency of the facility type. Here, we analyze pellets, chips, bio-oil and biochar, and hydrothermal liquefaction biofuel, all coming from harvest and/or wood mill residues except for Dias et al. (2017), for which pellets are made from short rotation willow (SRW). The facility type influences the combustion conversion efficiency of the bioenergy process, a lower efficiency leading to a higher use of biomass. Here, efficiency varies from 75 % (Dias et al., 2017) to 94 % (CIRAIG, 2014). Avoided fossil GHG emissions depend on the type of fossil fuel displaced and the emissions from biomass feedstock conversion processes.

Average and range of values for energy SF are presented by end use type in Fig. 7. The SF value increases when bioenergy substitutes higher carbon intensity fossil fuels. The highest value comes from the substitution of a mix of fossil fuels (petroleum coke, coal and heavy fuel oil) by bio-oil and biochar in a mobile pyrolysis unit for cement production. That is because the fast pyrolysis process generates a co-product (syngas) that is used for drying wood chips and harvested residues instead of using fossil fuel.

3.2.2. *Bioenergy substitution by sub-category*

Energy SF are presented by sub-category in Fig. 8, with an average of 0.80 tC/tC for non-residential heat and/or power, and 0.51 tC/tC for biofuel for transport. Based on five studies, four bioenergy products (pellets, chips, biochar and bio-oil) and six types of fossil fuels (coal, light fuel oil, heavy fuel oil, propane, petroleum coke and natural gas), the category of non-residential heat and/or power gives a fair range of SF values for small scale boilers and industrial furnaces. However, propane substitution should be calculated, as it is a fuel targeted by provincial strategies (FQCF, 2013). Also, more research should be done for biofuel transport, as only one study was analyzed.

SF by sub-category could be calculated by applying a weighting factor for each end use, based on provincial or national consumption. As this step was out of scope for this study, it is important to see these results as values for specific substitution cases and not a realistic average.

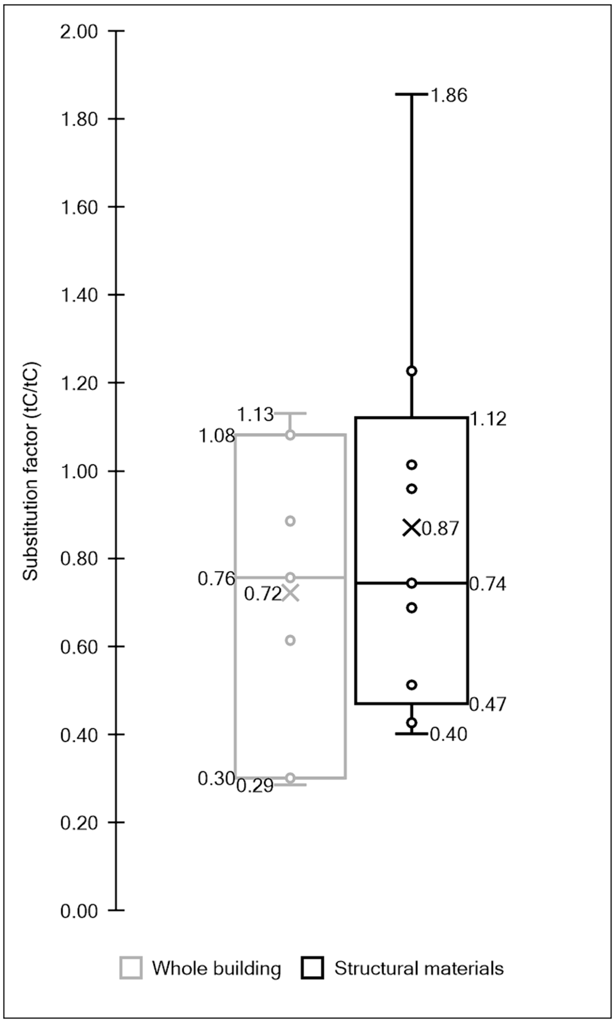

Fig. 6. Difference between considering only structural materials and whole building on substitution factors for sawnwood and panels in construction (tC = tonnes of carbon).

4. Discussion

The assessment of climate benefits from substitution by forest-based products requires a holistic approach combining micro-level studies that focus on the substitution of a specific product to macro-level studies, in which econometric factors and policy measures are integrated (Gustavsson et al., 2006a,b). As changes in wood consumption create an impact on forest ecosystems, the quantification of the substitution benefit should always be done in a broader framework that includes the tracking of C in forests and HWPs (Smyth et al., 2017; Xie et al., 2023). Therefore, the interpretation and use of our results should not be done without considering the broader framework as presented in Smyth et al. (2014).

4.1. Material product substitution

The average HWP substitution factor of 0.80 tC/tC found in this study is lower than the average of 2.1 tC/tC reported in the *meta*-analysis from Sathre & O'Connor (2010). It can be explained by differences in methodologies since Sathre & O'Connor (2010) use different system boundaries, including forest C emissions and end-of-life scenarios with energy recovery. Using Sathre & O'Connor's (2010) methodology in Smyth et al.'s framework could lead to double counting of GHG flows. SF calculated by Sathre & O'Connor (Sathre & O'Connor, 2010) include GHG emission reductions generated by co-products, which increases the SF value, in opposition to our methodology that only take into account the quantity of wood found in the final product. Here, we calculated product SF mainly for the production stage, as only a few studies include the use stage and transportation to landfill, the latter being negligible. Our results are consistent with the average value of 0.8 tC/tC reported by Leskinen et al. (2019) for SF considering only the production stage. However, the value reported by Leskinen et al. (2019) contains a larger variety of end uses than our study, which focuses on the construction sector.

One of the main objectives of this study was to compare our results with the latest Canadian study on SF development, which have been done by Smyth et al. (2017). We observed that our average SF values for sawnwood and panels are different from those reported by Smyth et al. (2017). The difference in the SF values between our study and Smyth et al. (2017) can be attributed to the differences in our methodologies. Smyth et al. (2017) considered end uses that we did not include, such as flooring, furniture, and decking, in their SF calculations. These end uses generate fewer greenhouse gas (GHG) emission reductions while being associated with higher factors. Our approach, more aligned with Xu et al. (2018), excluded these end uses and focused on SF calculations for single-family homes, multi-family homes, and multi-use buildings, which led to higher SF values, specifically 2.1 tC/tC for sawnwood and 2.2 tC/tC for panels.

To provide a more comprehensive comparison, we conducted further analysis using data available in Smyth et al. (2017). We calculated SF by end use for single-family homes (1.91 tC/tC), multi-family homes (2.35 tC/tC), and multi-use buildings (2.49 tC/tC). This analysis revealed that the amount of wood used in the buildings referenced in Smyth et al. (2017) was lower than in most of the buildings considered in our study, resulting in higher SF values in our research. This discrepancy underscores the importance of employing multiple end-use comparison scenarios, as generalizing the amount of wood used may lead to an overestimation of the SF.

Developing specific SF for sawnwood and panels is a complex task due to the wide range of potential end uses for these primary products, each of which can substitute various alternative materials, resulting in a broad spectrum of SF values. The representation of these end uses and their associated weightings must rely on realistic and reliable data. Smyth et al.'s (2017) approach, which limited the number of end uses to six and utilized data on Canadian wood consumption that may not accurately reflect the current and future market dynamics, raises valid concerns. Notably, the share of wood construction for non-residential applications in the province of Quebec has seen substantial growth, a trend that Smyth et al.'s weightings may not capture. Furthermore, our study did not delve into the weighting of substitution factors. In developing SFs for sawn wood and panels, we recognize the complexity due to the myriad of potential end uses, each with their own alternative materials and resultant range of SF values. Accurate representation of these uses demands robust data. While Smyth et al. (2017) provided a valuable framework by concentrating on a select number of uses, we note that market dynamics, particularly in Quebec's non-residential wood construction, have evolved since. Acknowledging this, our study has refrained from applying weighted SFs. Consequently, we recommend interpreting our SF values within the specific context of construction applications where wood substitutes steel or reinforced concrete. This deliberate focus ensures our SF values are applied in scenarios they most accurately represent, without overextension to broader wood usage contexts.

Most of the studies included in our analysis were based on a functional unit that includes only structural materials. However, wood constructions are subject to different restrictions in the national building code that could influence the type and the quantity of finishing elements needed. The choice of structural materials (wood, steel or concrete) might affect other aspects such as the envelope or finishing materials and thus, influence LCA results. Moreover, envelope materials might

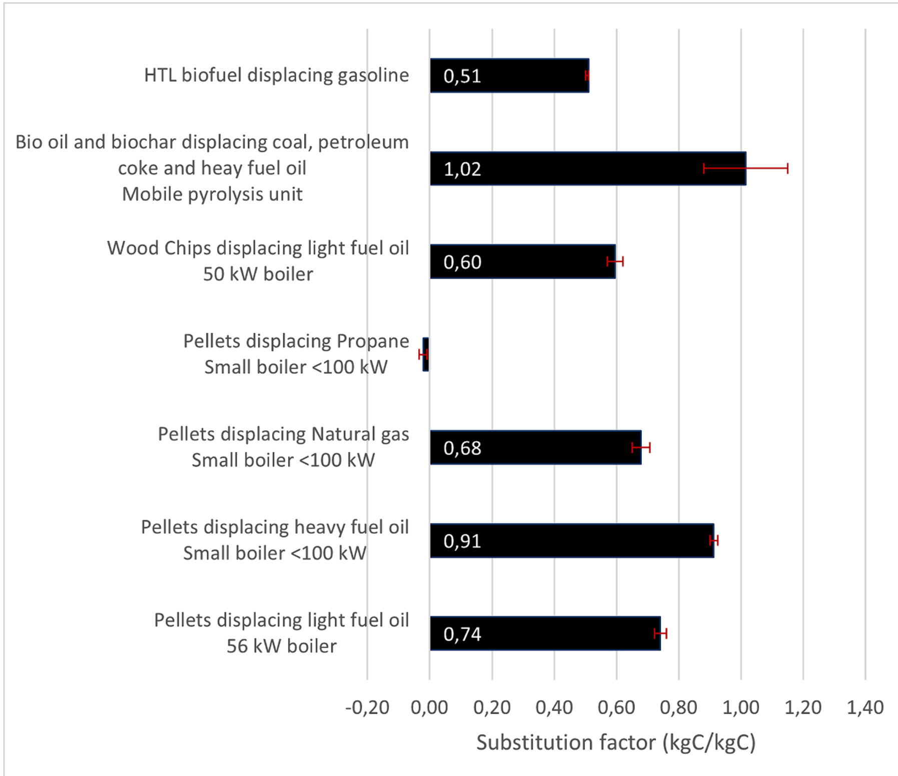

Fig. 7. Energy substitution factors for specific bioenergy end uses.

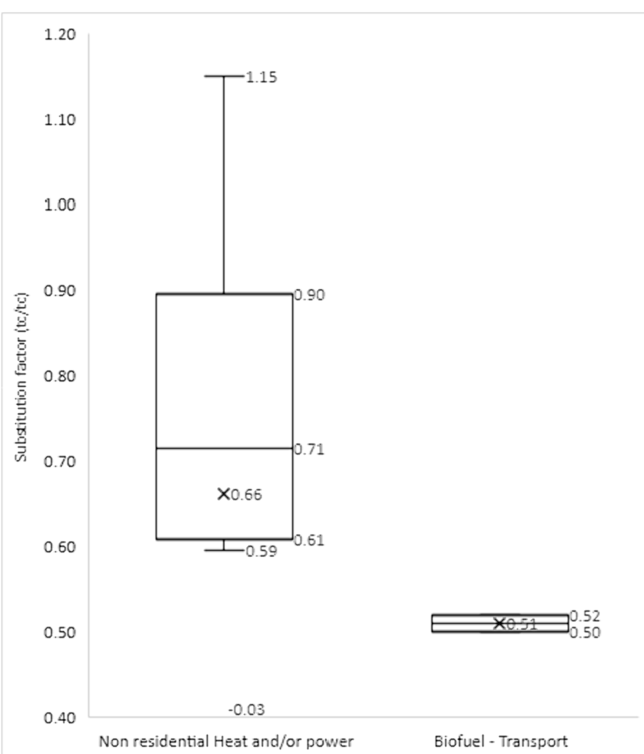

Fig. 8. Energy substitution factors by sub-category.

also influence energy efficiency of the building and change the environmental footprint over its lifecycle. Essoua & Lavoie (2019) show a GHG emissions reduction of 65 % when considering only structural materials and a 20 % reduction for the whole building, as more finishing elements are needed. Further research should be done to better understand the impact of finishing materials on LCA results of wood buildings. Studies used for SF calculation should be as inclusive as possible, including more materials in the functional unit.

4.2. Energy substitution

As biomass has a lower energetic content than fossil fuels, its use engenders substantial biogenic C emissions and creates a C debt (Mitchell et al., 2012). The net benefit of bioenergy occurs only when the C debt is reimbursed, i.e., when the forest reaches preharvest C levels (Laganière et al., 2017). Three main factors determine whether or not a bioenergy system can be beneficial for the climate i.e., the sources of biomass used, the conversion efficiency and the type of fossil fuel displaced (Berndes et al., 2016; Serra et al., 2016). Residues from harvesting, milling, construction and demolition are better biomass sources than mature forests and green trees, particularly when including biogenic carbon (Kouchaki-Penchah et al., 2023b). The substitution of high carbon intensity fossil fuels and the use of highly efficient conversion modes should also be prioritized. Our SFs include only fossil GHG emission reductions. The monitoring of biogenic C being undertaken in the broader framework (Smyth et al., 2014), where different sources of forest biomass and bioenergy system are considered. Alternatively, an integrated approach can be used (Kouchaki-Penchah et al., 2023b, 2024), considering different forest management strategies within the energy system. This approach takes into account biogenic CO_2 flows and market competition among different pathways.

A source of uncertainty for our results is that there are many ways to convert forest biomass into energy, while only a few cases were analyzed in this study. The process stages (e.g., grinding, transport, drying, densification, gasification, roasting, hydrolysis) of forest bioenergy vary between the different end product types and greatly influence associated fossil GHG emissions. These different processes, particularly gasification and hydrolysis, aim to increase the energy content of the final product but also generate more process GHG emissions. From the different bioenergy final products included in our analysis, process emissions vary between 0.73 kg CO_2eq/MJ for pellets and 20 kg CO_2eq/MJ for HTL biofuel. Bioenergy end uses with high process emissions tend to be associated with a lower substitution factor, e.g., in our analysis, the HTL biofuel have the highest process emissions and the lowest SF. However, these emissions can be reduced by employing other bioenergy products as it is demonstrated in Padilla-Rivera et al. (2017), where co-products are used in the process stages, which decreases the input needs of fossil fuels leading to increases SF value.

Our results for energy SFs align within the range reported by Smyth et al. (2017); however, we recognize that comparing these figures directly is not methodologically sound due to differences in study design. While Smyth et al. did not differentiate between types of bioenergy products and their associated supply chain emissions, our study considers these factors more explicitly. This difference in approach can influence the SF values significantly, and any comparisons should be made with an understanding of these underlying methodological distinctions.

However, methodological assumptions employed by Smyth et al. (2017) can be questioned. No distinction was made between the type of bioenergy products (e.g., pellets, chips, HTL biofuel, bio-oil, biochar, etc.) and fossil GHG emissions generated in the different supply chains. Process emissions associated with grinding, loading and transportation were not included in the scope of the study for bioenergy, in opposition to fossil fuels, for which emission factors included extraction and transportation of raw materials. These assumptions tend to overestimate the value of the substitution factor. Furthermore, energy SF developed in Smyth et al. (2017) is based on provincial average energy fuel mix and more applicable for general bioenergy use, which leads to higher uncertainty. Reducing uncertainty can be done by generating weighted SF for each sub-category, which are based on specific SF for each type of end use.

In the current context, the deployment of bioenergy is often driven by political strategies based on an energy transition plan, which implies targeted substitutions on specific fuels and sectors. For example, in the province of Quebec, the development of the bioenergy sector is framed by a plan (FQCF, 2013) that focuses on non-residential heating and targets the substitution of three fossil fuels i.e., heavy fuel oil, light fuel oil and propane. Thus, developing specific SF for these pathways will increase the precision in studies aiming at assessing the climate change mitigation potential of the forest sector. Beauregard et al. (2020) found a specific SF of 0.95 tC/tC for substitution of heavy fuel oil, which is consistent with our result of 0.91 tC/tC. Moreover, we found an average SF of 0.69 tC/tC for light fuel oil substitution and 0.68 tC/tC for natural gas substitution. Further research would be needed to calculate a specific SF for propane substitution.

4.3. Limitation and future research

The results presented in our study come with several limitations, as the calculation of substitution factor by end use was done using a limited number of studies. For product SF, all scenarios of materials consumption and GHG emissions comparison were associated with the construction sector. Further research would be necessary to cover the other sectors i.e., furniture, packaging, paper, chemical products and textiles, which are sectors for which wood consumption is expected to increase in the future and thus, represent a high substitution potential. For energy SF, our results are limited to sub-categories of non-residential heat and/or power and biofuel transport. Further research should be done to cover all sub-categories. Also, for most end uses, only one study was analyzed, which increases the uncertainty of the SF value. Adding more studies would increase the range of values leading to a more accurate average.

Weighting substitution factors to HWP basket (sawnwood, panels, pulp and paper, and bioenergy) was out of the scope for this study, but is necessary to obtain more representative values that can then be integrated in the broader framework. This step would require a market analysis to better understand the current and future final use of wood and develop a method for weighting and separating the substitution impact between the different primary products (Xie et al., 2023). Doubts should be raised concerning the partitioning of avoided emissions between sawnwood and panels on a mass basis. This is a simplistic approach and more research could be done to enhance precision. Moreover, as wood substitution is a projected change in our society, SF should be weighted based on projected wood consumption statistics and not current, or past statistics.

Another limitation of our study concerns the variety in terms of rigour from the different studies used. Some are rigorous LCA studies that have been through a peer-review process, while others are internal studies for informal purposes. Data from the Gestimat database relied on several prospective building techniques assumptions and were included to bring more perspective and comparison scenarios. Ideally, substitution factor should be based only on existing studies to reduce uncertainty. Moreover, regional SF should be calculated for each province inside Canada, as they all present different energy mixes that impact GHG emissions for production and use stages. Our data sources mainly come from Quebec and British Colombia making it is difficult to generalize our results on a national basis and further research should be done to generate results for each province.

Substitution factors should not be constant over time, as they depend on many evolving factors such as technological development, changes in product design or industry processes, that could influence GHG emissions and bills of materials, thus resulting in different SF (Leskinen et al., 2019). Steel and concrete industries are very proactive in implementing strategies to reduce their carbon footprint. With different strategies, such as energy efficiency, fossil fuel substitution and carbon capture and storage (CCS), it is likely that these industries could reduce their product carbon intensity up to 50 % before 2050 (IEA, 2018, 2020). Therefore, regressive substitution factors should be considered when they are used for long-term projections. Additionally, as our understanding of forest bio-geo-physical properties evolves, so will their quantifiable climate impacts. SF such as ours would gain robustness by being updated, for example following works on the climate impact of albedo modification induced by harvesting boreal Canadian trees (Bright & Lund, 2021).

Beyond the rigor presented in the calculation method, great importance must be given to the context in which the factors are used to ensure that they are used correctly.Moreover, the factors should be applied only to the proportion of wood that will actually be used as a substitute for other materials or energies. Adding more forest-based products and bioenergy to the market does not necessarily engender a decrease in other competing markets, (e.g., adding 1 MJ of biofuel to the market does not necessarily lead to the reduction of 1 MJ of fossil fuel). This concept, named as carbon leakage, can be defined as the loss of environmental benefits generated by unforeseen consequences of an action or a decision (Brown et al., 1997). A better understanding of this concept for the Canadian context would be required, as its exclusion could lead to an overestimation of carbon benefits from substitution (Kallio et al., 2018).

5. Conclusion

Forest-related management strategies present a high potential of climate change mitigation and can be evaluated by examining different scenarios about forest C sequestration, HWP dynamics and substitution benefits (Smyth et al., 2014). We analyzed the quantification of the substitution effect for Canada, based on the method presented in Smyth et al. (2017). Several uncertainties remain about substitution factors, which quantify fossil GHG emission reductions engender by the use of wood-based products and bioenergy instead of high carbon intensity materials and fuels over the same functional unit. For product substitution factors, we analyzed the comparison of 18 different wood-intensive scenarios relative to the construction industry (17 buildings and one observation tower) and produce a range of values between 0.29 tC/tC and 1.86 tC/tC with non-weighted average values for sawnwood (0.80 tC/tC) and panels (0.81 tC/tC). Energy substitution factors are influenced by the type of bioenergy product, the conversion efficiency of the facility and the type of fuel displaced. Here, we examined six cases including different biomass products (pellets, chips, bio-oil, biochar and hydrothermal liquefaction biofuel) and alternative fossil fuels (light fuel oil, heavy fuel oil, natural gas, coal, petroleum coke and gasoline). We found an average substitution factor of 0.80 tC/tC for non-residential heat production and 0.51 tC/tC for biofuel transportation. When interpreted by the type of fossil fuel displaced, we found 0.91 tC/tC for heavy fuel oil, 0.69 tC/tC for light fuel oil, and 0.68 tC/tC for natural gas substitution.

Substitution factors presented in this study serve to disclose the range of variation in climate benefits of Canada's HWP's to be integrated in broader studies in combination with forest and HWP carbon dynamics. Further research should be done to calculate substitution factors for other end uses and sub-categories that were not considered in this study and scale the results using national wood consumption statistics and future projections.

CRediT authorship contribution statement

Thomas Cardinal: Writing – original draft, Visualization, Validation, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Charles Alexandre:** Writing – review & editing, Formal analysis. **Thomas Elliot:** Writing – original draft, Visualization, Validation, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Hamed Kouchaki-Penchah:** Writing – review & editing, Visualization, Formal analysis. **Annie Levasseur:** Writing – review & editing, Validation, Supervision, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Thomas Cardinal reports financial support was provided by Natural Sciences and Engineering Research Council of Canada. Annie Levasseur reports financial support was provided by Centre interuniversitaire de recherche en opérationnalisation du développement durable funded by Fonds de recherche du Québec and by the Canada Research Chair program.

Data availability

Data is available in supplementary material

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2024.111940.

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