

Assessing the greenhouse gas effects of harvested wood products manufactured from managed forests in Canada

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We developed a life-cycle analysis (LCA) system to quantify the carbon dynamics for Canadian-made harvested wood products (HWP). We considered the carbon stocks of HWP in use and in landfills/dumps, emissions reduced by substituting HWP for non-wood construction materials, HWP production emissions and methane emissions from decomposing wood disposed of in landfills. Carbon dynamics analyses were conducted for five HWP production scenarios. Results indicate structural panels have the highest potential in mitigating greenhouse gases (GHG) emissions, followed by lumber and non-structural panels. Net GHG effects of Canadian-made HWP were evaluated by integrating HWP carbon dynamics with forest carbon analysis using four forest management units (a total of 2.21 million ha of forests managed for timber production) from Ontario, Canada, as a case study. If HWP substitution benefits were estimated using the average displacement factor, and the wood obtained by increasing harvesting (relative to the baseline harvest scenario) in these four management units is used for structural panel, lumber, non-structural panel and business as usual HWP production, 0, 21, 39 and 84 years are needed to achieve net emission reductions, respectively; net emission reductions were, respectively, estimated to be 112, 93, 66 and 21 Mt CO₂-equivalent in 100 years. Our results suggest harvesting sustainably managed forests in Canada to produce long-lived solid HWP can significantly contribute to GHG mitigation.

Introduction

Forests play an important role in the global carbon (C) cycle (Pan *et al.*, 2011; Grassi *et al.*, 2017). In 2009, at the 10th Conference of Parties of the UN Framework Convention on Climate Change in Copenhagen, delegates agreed to include forest management in mandatory reporting of national greenhouse gas (GHG) inventories, and to add harvested wood products (HWP) originating from forest management as an additional C pool for countries having country-specific activity data (HWP production, end uses, service lives and end-of-life disposals) and analytical methods (UNFCCC, 2009). To accommodate this change, the Intergovernmental Panel on Climate Change (IPCC) revised its methods and guidelines for estimating anthropogenic GHG emissions by sources and removals resulting from land use, land use change and forestry (IPCC, 2014). Canada has 347 million ha of forest that account for 9 per cent of global forests, and has long been one of the principal HWP producers in the world (NRCan, 2016). Therefore, even though Canada is no longer a signatory to the Kyoto Protocol, it is important to assess the great

potential that Canada's forests and HWP have to influence atmospheric GHG emissions (Sikkema *et al.*, 2013).

A HWP life cycle starts with forest harvesting and ends when the HWP fully decomposes or is combusted. Supported by life-cycle inventory data, HWP life-cycle analysis (LCA) is used to quantify physical C stocks, flows and emissions of HWP, emissions associated with fossil fuel use and reduced GHG emissions from using HWP to substitute for non-wood materials and by using wood to replace fossil fuels (Pingoud *et al.*, 2010; Lippke *et al.*, 2011; Chen *et al.*, 2014; Lundmark *et al.*, 2014). Because a HWP life-cycle is a highly complex system (Lippke *et al.*, 2011; FAO, 2016; Hoberg *et al.*, 2016), life-cycle inventory data are often lacking to support comprehensive analyses required to develop a LCA system. As suggested by Soimakallio *et al.* (2016), most HWP life-cycle analyses (LCA) have been either methods focused (Pingoud *et al.*, 2010; Butarbutar *et al.*, 2016) or case studies of a particular HWP type, end use or life stage (Perez-Garcia *et al.*, 2005; Athena, 2012a, b; Nepal *et al.*, 2016).

Accurately assessing HWP GHG mitigation potential requires a systems perspective to include all the C stock, emission, and

removal components related to forest management, and HWP production, uses, and post-service disposals (Ximenes *et al.*, 2012; Lemprière *et al.*, 2013; Smyth *et al.*, 2016). A consequential life-cycle analysis (CLCA) approach is developed by taking such a systems perspective, and in the past decade it has been widely used to assess the GHG mitigation potential of forest bioenergy (McKechnie *et al.*, 2011; Tittmann and Yeh, 2013; Macintosh *et al.*, 2015; Ter-Mikaelian *et al.*, 2015). The CLCA approach is useful to investigate how physical C stocks and emissions and material substitution effects would change in the future if, for example, forest harvesting, HWP production and HWP end uses change (Earles and Halog, 2011; Helin *et al.*, 2013; Nepal *et al.*, 2016).

In this study, we first conducted a HWP LCA that encompasses the entire life-cycle of all major HWP produced in Canada, starting with harvested wood and ending after disposal of the retired HWP. Five HWP production scenarios (business as usual, lumber, structural panel, non-structural panel, and pulp and paper) were defined (in section Methods, Development of HWP-CASE and FORCARB-ON2 models), based on primary wood use in manufacturing HWP. The LCA results are presented as age-dependent C curves by production scenario for HWP made from wood harvested in Canadian forests, which dynamically illustrate HWP C stocks/emissions components as the percentages of input wood C entering a HWP manufacturing process. To conduct the HWP CLCA, we integrated the HWP LCA C dynamics with the C budget of the forests harvested to produce HWP for four forest management units from Ontario, Canada, as a case study. We defined a baseline harvesting scenario and an increased harvesting scenario (in section Methods, Assessment of harvested wood product net greenhouse gas effects) to simulate forest carbon changes resulting from different levels of harvesting. In the baseline scenario, historical harvest rates (the average ratios of actual annual harvest volume to the allowable harvest volume projected in the government approved forest management plans of the four management units for the period 1990–2009) were used to simulate forest harvest for 100 years into the future. For the increased harvest scenario, harvest rates were 95 per cent of the annual allowable harvest volume. In the case study, only the additional wood harvested in the increased harvest scenario was used in our analysis (based on assumptions described in section Methods, Assessment of harvested wood product net greenhouse gas effects). Our objectives included (a) quantifying HWP life-cycle C dynamics (C stocks, flows and emissions) for Canadian HWP by production scenario and HWP life-cycle stage, (b) assessing how much GHG emissions are reduced by substituting HWP for alternative energy-intensive construction materials, (c) combining (a) and (b) to describe overall LCA emission/removal dynamics by HWP type, and (d) as a case study, assessing the net GHG effects of the five HWP production scenarios by integrating HWP LCA C dynamics with the C budget of Canadian forests harvested to produce the HWP.

Methods

The HWP life cycle, starting with harvested wood and ending when the wood/HWP is burned or decomposes, was divided into three life-cycle stages: production, end use and post-service disposal, a method commonly used in HWP life-cycle analysis. The industrial and natural processes within and among the three life-cycle stages dynamically and inter-actively determine wood C flows and emissions. Figure 1 outlines the C

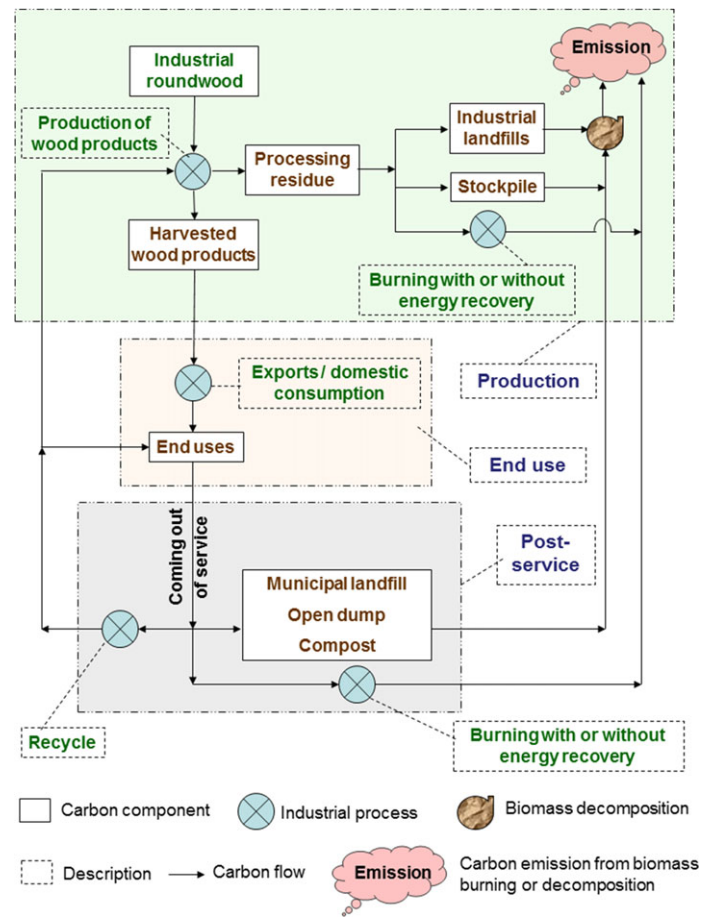


Figure 1 Conceptual model of the components used to estimate harvested wood product life-cycle carbon flows, stocks and emissions (modified from Chen *et al.*, 2013).

components and C flows and emissions over a HWP life cycle, and illustrates the steps taken to estimate HWP life-cycle C stocks and emissions. The HWP LCA analysis system we developed includes six HWP life-cycle C components: C stocks of HWP in use, C stocks of wood/HWP in landfills/dumps, HWP production emissions, landfill methane emissions from wood/HWP decomposition, emissions reduced by using wood to substitute for fossil fuels and emissions reduced by using HWP to replace non-wood materials in construction. Industrial roundwood and finished HWP provide temporary C storage, and were not included as a part of the cumulative HWP LCA C stocks. The C stocks and stock changes of the forests harvested to produce HWP were not considered for HWP LCA, but were included when conducting CLCA to assess HWP net GHG effects. The methods used to estimate the carbon stocks, flows and emissions of these three HWP life-cycle stages are summarized in section Methods (Estimating harvested wood products carbon stocks/emissions by life-cycle stage – a summary), with more detailed descriptions provided in the Supplementary data.

Estimating harvested wood products carbon stocks/emissions by life-cycle stage – a summary

To quantify the conversion from harvested wood to HWP, historical Canadian forest harvesting and HWP production data (Supplementary data: 1. Data sources for forest harvest, harvested wood product production, and international trade) were used to allocate harvested wood

to HWP production processes. Based on the data obtained from the FAOSTAT database (<http://www.fao.org/faostat/en/#data/FO>), industrial roundwood export from and import to Canada were 2.1 and 2.6 per cent, respectively, of the domestic industrial roundwood produced in Canada between 1961 and 2014. Due to the lack of data for the use of exported Canadian industrial roundwood in HWP manufacture, as well as the end uses and post-service disposal of the HWP produced, in HWP LCA we assumed the small fraction of exported Canadian industrial roundwood was used the same as that consumed domestically. Simulation of the HWP production life-cycle stage is detailed in Supplementary data: 2. Harvested wood product manufacturing, including converting industrial roundwood and mill residue to HWP, quantifying waste wood materials production and disposal, and estimating the C stocks of waste wood materials disposed of in dumps and industrial landfills. Manufacturing of HWP often results in fossil fuel-based emissions from fossil fuels and electricity consumed during forest management activities, wood harvesting and transport, and HWP manufacturing. These fossil fuel-based emissions were estimated based on a series of Canada-specific HWP manufacturing analyses (Table S2, Supporting data).

Harvested wood product end uses vary among HWP consuming countries. To simulate HWP end-use life-cycle stage, recent production and international trade data for Canadian-made HWP were obtained for the period 1997–2014, and used to calculate the export fractions of all Canadian HWP by HWP category and major consuming country. These historical HWP export fractions were applied to future Canadian-made HWP to estimate domestic and foreign HWP consumption. Since the United States and Canada have been the most important consumers of Canadian HWP, and due to a lack of end use data for Canadian-made HWP consumed by other countries, we allocated Canadian-made HWP to end uses based on statistical analysis of HWP end uses in the United States and Canada (for details see Supplementary data: 3. Harvested wood product end use).

Detailed methods descriptions for post-service HWP C stocks and emissions are provided in Supplementary data: 4. Post-service harvested wood product carbon flows, stocks, and emissions. In summary, retired HWP were assumed to be discarded via various disposal options. Harvested wood products disposed of in dumps and municipal landfills decompose at different rates, with C storage changing and C emissions accumulating from HWP decomposition. Methane emissions from decomposing HWP discarded in landfills were estimated by considering rates of HWP decomposition, methane generation and capture, and the oxidation of uncaptured methane when it reaches the top layer of waste covering soil in landfills.

We followed the revised IPCC guidelines for preparing national GHG inventories (IPCC, 2014) to track the C stocks and emissions of the HWP originating from Canada's managed forests; i.e. analysis included the HWP exported to other countries but excluded imported HWP. For completeness, we also included the C dynamics analysis for HWP disposed of in landfills, even though the revised IPCC guideline recommendation was to treat the HWP C as an instantaneous carbon dioxide (CO₂) emission, with landfill methane emissions accounted for by the waste sector (IPCC, 2014). Carbon released as CO₂ from burned or decomposed wood biomass was deducted from HWP C stocks, while landfill methane emissions from HWP decomposition were accounted for separately because of methane's 28 times global warming potential compared with CO₂ over a 100-year period (IPCC, 2013). As they are additions to the global C cycle, emissions associated with fossil fuel and electricity consumption for forest harvesting, wood transport and HWP manufacturing were included in analyses.

Emissions reduced by using HWP to replace non-wood materials in construction are a critical component in HWP C modelling (Salazar and Meil, 2009; Sathre and O'Connor, 2010; Lippke et al., 2011; Brunet-Navarro et al., 2016; Gustavsson et al. 2017). We used published data to estimate average displacement factors for solid HWP, which are defined

as tonnes of CO₂ equivalent (tCO₂eq) emissions reduced per tonne of C (tC) contained in HWP used to substitute for non-wood construction materials. Displacement factors were developed by comparing fossil fuel-based emissions released from manufacturing HWP with those from alternative materials. In this study, C stocks of HWP in use and in landfills, HWP recycling and re-use, landfill HWP C dynamics and forest C budgets were simulated separately.

Analysis of harvested wood products life-cycle carbon dynamics

We considered four categories of HWP: lumber, structural panels (plywood and oriented strand board), non-structural panels (medium density fibreboard and particleboard) and a single pulp and paper category for pulp, paper, and paper products. To simplify the analysis, we assumed that: (a) HWP manufacturing is completed within a year of wood harvesting, (b) HWP are put into various end uses in the year of HWP production and (c) retired HWP are disposed of in the year they are retired. These assumptions make it possible to merge the three HWP life-cycle stages to produce a combined HWP LCA emissions/removal curve for a HWP type. The curve reflects the sum of the C stocks of HWP in use and wood/HWP in landfills/stockpiles, plus the substitution benefits from using HWP in residential construction and from using collected landfill methane to recover energy, minus HWP production emissions and methane emissions from wood/HWP decomposing in landfills. In HWP LCA, forest C stock changes due to forest harvesting are not considered, but they are included in HWP CLCA as described in section Methods (Assessment of harvested wood product net greenhouse gas effects). Unless otherwise specified, all the LCA emissions/removals components are presented as CO₂eq, and landfill methane emissions were converted to CO₂eq by considering their global warming potential.

Wood substitution effects on reducing fossil fuel-based emissions

Based on 21 studies, Sathre and O'Connor (2010) produced an average displacement factor of 2.1 tC, or 7.7 tCO₂, of reduced emissions for each tC contained by HWP used to replace non-wood construction materials. However, those studies have different system boundaries, e.g. they may or may not (a) include forest and HWP C analysis, (b) account for wood biomass used for energy for HWP production and (c) consider the end-of-life HWP fate, landfill HWP C dynamics, landfill methane emissions and methane collected to produce energy. These life-cycle stage C stocks and emissions components of HWP change over time, depending on end uses and end-of-life disposals, and thus need to be quantified dynamically. Using a single displacement factor that ignores dynamic changes in these components, therefore, is likely to result in inaccurate HWP LCA.

We calculated average displacement factors for HWP used to replace non-wood materials in construction based on published data, by comparing the embodied emissions of HWP with those of alternative materials. Embodied emissions are defined as the GHG emissions that result from energy used to extract and transport raw material, manufacture and deliver products, and in some cases include those from construction (Börjesson and Gustavsson, 2000; Glover et al., 2002; Lippke et al., 2004; Upton et al., 2008). Since the CO₂ emissions from burning mill residues were accounted for by reducing wood C stock during HWP manufacture, they were excluded from the embodied emissions when developing the displacement factor. Thus, we reduced the reported HWP embodied emissions by up to 50 per cent, depending on how they were reported in the cited studies, assuming that, in Canada, more than 50 per cent of the energy consumed by the HWP industries is generated from wood processing residue (Meil et al., 2009). Our analyses show that, on

average, for each tC contained in HWP used to substitute for alternative materials in residential and non-residential construction, emissions are reduced by 9.56 and 3.64 tCO₂, respectively (Table 1). However, the published substitution studies have a mix of non-wood materials replaced by HWP, from which we could only produce a single displacement factor. In comparison, Lippke *et al.* (2004) and Upton *et al.* (2008), for example, estimated separate displacement factors for using HWP to substitute for steel and concrete in residential construction. Chen *et al.* (2013, Table 16) estimated that 62.1 per cent of total Canadian HWP was consumed by the construction sector: 28.8 per cent for new residential construction, 26.6 per cent for residential repair and remodelling and 6.7 per cent for non-residential construction. Of the 62.1 per cent of total Canadian HWP consumed by construction, 64 per cent were estimated to replace non-wood materials (Chen *et al.*, 2014). Wood and non-wood materials can both be used for residential repair and remodelling. Thus, some of the HWP used for residential repair and remodelling replaced non-wood materials, and we assumed the same percentage (64 per cent) of HWP consumed by this end use replaced non-wood materials. Based on these ratios, a weighted average displacement factor for all HWP used to replace alternative materials in construction was estimated to be 8.91 tCO₂ of reduced emissions per tC contained in HWP used in substitution.

Emissions reduced by substituting HWP are permanent and cumulative. Reductions are estimated by multiplying the displacement factor and the amount of HWP used to replace non-wood materials in construction. Although not considered in the present study, the producers

of alternative non-wood materials may find new markets and/or end uses for their products, and thus the substitution effects may be lower than estimated.

Because the substitution benefits highly depend on the displacement factors, we also produced a low- and high-end of the range displacement factors for HWP used in residential and non-residential constructions, estimated as the average minus and plus the standard deviation of the values listed in Table 1. And similar to the weighted average calculation in Table 1, we then produced a weighted average low- (2.51 tCO₂eq/tC in HWP) and high-end (15.33 tCO₂eq/tC in HWP) displacement factor for all HWP used in construction that substitute for non-wood materials. These displacement factors were used to produce a range of substitution benefits for HWP used in construction.

Wood used to produce energy for the HWP industry reduced fossil fuel-based emissions by 2.0 tCO₂eq per tC in wood, and collecting landfill methane to replace fossil fuels in electricity generation can reduce 3.4 tCO₂eq of emissions per tC in methane (Chen *et al.*, 2014). Again, the use of mill residue by HWP industries to produce energy is embedded in HWP production emissions analysis, i.e. the use of wood energy reduces use of other energy sources and thus fossil fuel-based production emissions.

Development of HWP-CASE and FORCARB-ON2 models

Analysis and synthesis of the Canada-specific data were used to develop HWP-CASE, a model for the Comprehensive Assessment of the carbon

Table 1 Displacement factors for harvested wood products (HWP) used to substitute for non-wood materials in construction (tonnes of reduced CO₂ emissions per tonne of carbon contained in HWP used for substitution)

Construction type	Data source	End use	Displacement factor
Residential construction	Buchanan and Levine (1999)	Single family house	12.83
	Cha <i>et al.</i> (2011)	Single family house	7.92
	Glover <i>et al.</i> (2002)	Single family house ¹	5.87
	Gustavsson <i>et al.</i> (2006)	Apartment building	4.03
	Koch (1992)	Residential construction	8.07
	Börjesson and Gustavsson (2000)	Multi-story apartment	3.30
		Single family house ¹	8.80
			21.27
	Scharai-Rad and Welling (2002)	Single family house	16.98
	Salazar and Meil (2009)	Single family house	5.17
	Smyth <i>et al.</i> (2016)	Single family house	23.10
		Multi-family house	1.06
	Average displacement factor for residential construction	9.56	
Non-residential construction	Buchanan and Levine (1999)	Hostel building	3.67
		Office building	4.40
		Industrial building	5.87
	John <i>et al.</i> (2009)	Multi-story building ¹	4.77
	Robertson <i>et al.</i> (2012)	Office building	4.07
	Scharai-Rad and Welling (2002)	Warehouse	5.50
		Shed ¹	0.88
			1.39
	Smyth <i>et al.</i> (2016)	Office building	0.73
	Average displacement factor for non-residential construction	3.64	
Weighted average ² displacement factor for all construction			8.91

¹Values represent estimates for using HWP to substitute for two alternative materials: steel and concrete.

²Calculated based on the estimates of Canadian-made solid HWP used in construction following Chen *et al.* (2013, Table 16).

Stocks and Emissions for Canadian HWP (Chen et al., 2014). The production approach elaborated by the IPCC (2014) is implemented in HWP-CASE to track the C flows, stocks, and emissions over the HWP life-cycle. Inputs to the model are annual or decadal values for harvested merchantable wood volume, organized by primary wood use category (saw logs, pulpwood, etc.). The simulation results consist of cumulative annual/decadal C stocks of HWP in use, C stocks of wood residue/HWP discarded in landfills/dumps, HWP production emissions, landfill methane emissions, substitution benefits from using HWP in construction and from using collected landfill methane to produce energy, and the combined LCA GHG emissions/removals.

To conduct HWP LCA, we defined the following production scenarios based on primary use of wood to produce different types of HWP: (a) business as usual (BAU) HWP, (b) lumber, (c) structural panels, (d) non-structural panels and (e) pulp and paper. In the BAU HWP scenario, the harvested wood was divided among the production processes of all major HWP types based on past use (Chen et al. 2013, Figure 3b). Scenarios (b)–(e) are hypothetical, in which the harvested wood is primarily used to produce a particular type of HWP. These scenarios were chosen to allow the LCA C stocks/emissions of different HWP to be contrasted under boundary conditions. Results can also be used to produce LCA estimates for scenarios with mixed use of harvested wood and to help decide how to maximize the climate change mitigation potential. The wood conversion in non-structural panel and pulp and paper production scenarios were assumed to be the same as those in Table S1 (Supporting data), because the wood conversion is simple: almost all of it is converted to HWP or energy. However, Meil et al. (2009) estimated that, in Canada, large fractions of wood used to produce lumber and structural panels were converted to pulp chips, which consist of relatively good quality wood fibre. Thus, to maximize solid HWP production in the hypothetical lumber (b) and structural panel (c) production scenarios, we assumed half the pulp chips estimated by Meil et al. (2009) were used to produce structural panels and the other half for non-structural panels, respectively. Because some wood biomass, such as bark, is not suitable for producing HWP, and diverting the biomass use from energy production will increase the use of other energy sources such as fossil fuels and purchased electricity (i.e. greater HWP production emissions), we assumed the same fractions of wood biomass were used to produce energy for all HWP production processes, as indicated in Table S1 (Supporting data). The wood C flows for the five production scenarios (based on Table S1 in Supporting data) and the described assumptions about using wood chips to produce solid HWP are listed in Table 2.

To quantify the changes in HWP C emissions/removals components, we ran HWP-CASE for a period of 100 years, assuming 100 units of wood C entering the HWP production processes in the first year and no input wood for the remaining 99 years. This way the HWP C emissions/removals components estimated using HWP-CASE can be considered as time-dependent percentages of the input wood C by HWP life-cycle stage. An overall LCA C curve for each HWP production scenario was estimated by merging substitution benefits and the C curves of the three life-cycle stages. These C curves can be used to estimate HWP LCA C emissions/removals by HWP types based on HWP age.

To assess the net GHG effects, HWP LCA needs to be combined with the C budget of the forest from which the wood is harvested (Eriksson et al., 2007). FORCARB-ON2 is a new version of Ontario's forest carbon budget model FORCARB-ON (Chen et al., 2010) that is designed to estimate the C stocks for Ontario's Crown (i.e. public owned) forests managed for timber production. Forest C is estimated based on empirical relationships among C pools (live tree, standing dead tree, forest floor, down dead wood, understory vegetation and soil) and merchantable volume density and/or forest age (Chen et al., 2010, Table 1). Among other changes, major updates implemented in FORCARB-ON2 include a component to simulate disturbance (harvest, fire)-caused C fluxes and post-disturbance C dynamics of forest floor, standing dead tree, and down dead wood C pools; a modification of the post-disturbance soil C module; and a revised approach for including residual live trees in clear-cut harvest areas in post-harvest simulation.

Assessment of harvested wood product net greenhouse gas effects

To assess the net GHG effects of forest harvesting and the production and use of HWP, the HWP LCA of C stocks/emissions need to be integrated with the C balance of the forests harvested to produce those products (Skog, 2008; Pingoud et al., 2010; Sathre and O'Connor, 2010; Helin et al., 2013; FAO, 2016). Canada has 347 million ha of different types of forests (NRCAN, 2016). To account for various forest characteristics such as dominant tree species, growth and yield, natural disturbances and forest management, HWP CLCA needs to be conducted for the specific forests harvested to produce the HWP.

As a case study of assessing the CLCA GHG effects of Canadian-made HWP, we used four adjacent boreal forest management units from north-western Ontario, Canada: the Crossroute, Dog River-Matawin, Sapawe and

Table 2 Carbon flows by harvested wood product (HWP) production scenario (percentages of wood carbon delivered to HWP mills)

HWP and residue disposal options	HWP production scenarios				
	BAU ¹	Lumber ²	Structural panel ²	Non-structural panel ³	Pulp and paper ³
Lumber	32.1	43.1	0.0	0.0	0.0
Structural panel	4.3	13.7	72.7	0.0	0.0
Non-structural panel	5.4	22.7	4.3	88.9	0.0
Pulp and paper	22.8	0.3	0.0	0.0	60.5
Burned for energy	26.2	15.4	20.3	10.9	35.5
Industrial landfill	2.3	1.0	0.3	0.2	4.0
Stockpile	0.1	0.7	1.8	0.0	0.0
Decay/burned as waste	6.8	3.1	0.6	0.0	0.0
Total	100.0	100.0	100.0	100.0	100.0

¹Business as usual carbon flows of harvested wood used to produce the four major types of HWP (Chen et al., 2013, Figure 3b).

²Modified based on the carbon percentage for producing lumber and structural panels in Table S1 in Supporting data by assuming pulp chips are used as additional wood biomass input to produce more non-structural panels using the carbon percentages of non-structural panel production.

³Same as in Table S1 in Supporting data.

Wabigoon Forests. These forest management units cover an area of 2.41 million ha, of which 2.21 million ha is managed for timber production. The most common tree species are black spruce (*Picea mariana* (Mill) B.S.P.), jack pine (*Pinus banksiana* Lamb.) and trembling aspen (*Populus tremuloides* Michx.). Forest inventory information, growth, natural succession, and fire return intervals, as well as projected maximum annual allowable harvest areas and volumes, were obtained from government-approved forest management plans. In each management unit inventory, forests are classified by forest unit (aggregates of forests with similar species composition and productivity) in 10-year age class intervals. This information was used to run FORCARB-ON2 for a simulation period of 100 years.

For the FORCARB-ON2 simulation, we defined a baseline harvesting scenario and an increased harvesting scenario. In the last two decades, forest harvesting in Ontario has been well below the maximum allowable harvest level. Thus, in the baseline scenario, forests were projected to be harvested in the future at mean rates of actual annual harvest volume from 1990 to 2009; these harvest volumes captured recent fluctuations in forest harvesting in Ontario, from their peak in early 2000s to a low in 2008. These historical harvesting rates are management unit-specific and were estimated as 56.0, 59.1, 42.6 and 68.7 per cent of the 1990–2009 maximum allowable harvest volume for the Crossroute, Dog River Matawin, Sapawe, and Wabigoon Forests, respectively, calculated using the procedure described in Ter-Mikaelian *et al.* (2016). In the increased harvesting scenario, harvesting rates for the four forest management units were increased to 95 per cent of the allowable harvest volume, assuming 100 per cent would be infeasible due to lack of merchantable wood and inaccessible areas. These harvesting rates were applied uniformly to all forest stands available for harvesting in the forest management plans. The harvested wood is reported as decadal harvest area and volume, specified by forest unit and age class. For both scenarios, other forest dynamics specified in forest management plans, such as growth and yield, natural succession, fire disturbance, post-fire and post-harvest succession, as well as areas protected from harvest for various purposes and conversion between forest and non-forest land, were assumed the same as in the forest management plans.

FORCARB-ON2 simulations provided decadal changes of the forests and the forest C stocks (by C pools). The forest C stock differences between the baseline and the increased harvesting scenarios were determined by the C stock changes in the areas affected by increased harvesting: in the baseline scenario, these areas are not harvested and the forests continue to undergo natural processes of growth; in the increased harvesting scenario, they are harvested and regenerated (artificially or naturally, as specified in the respective forest management plans). The wood from harvesting the same area in both scenarios (defined in the baseline scenario) is assumed to be used to produce HWP based on the BAU production scenario to meet market demand – when calculating differences of HWP LCA C emissions/removals between the scenarios these HWP cancel each other out. Therefore, to simplify the analysis in the case study of net GHG effects of HWP, we only considered the additional wood harvested in the increased harvesting scenario (relative to baseline). And we assumed the additional wood from increased harvesting was used to produce HWP following the production scenarios defined in section Methods (Development of HWP-CASE and FORCARB-ON2 models). Decadal harvest volumes obtained from the increased harvest were used to run HWP-CASE for 100 years (as in FORCARB-ON2 forest C simulation), to obtain the cumulative HWP LCA C stocks/emissions.

Equation (1) was used to estimate the net GHG effects for the HWP produced from using the additional wood harvested in the increased harvesting scenario:

$$GHG_{net}(t_0 + t) = GHG_{HWP-inc}(t_0 + t) + \Delta FC(t_0 + t) \quad (1)$$

in which t_0 is the year when increased forest harvesting starts, t is the number of years after t_0 ; $GHG_{HWP-inc}(t_0+t)$ is the HWP LCA emissions/

removals at year t , including the C stocks of HWP in use and wood/HWP in landfills, HWP production emissions, methane emissions from HWP decomposition in landfills, reduced emissions from substituting HWP for non-wood construction materials, and reduced emissions from using collected landfill methane released from decomposing HWP to generate electricity (Chen *et al.*, 2013); $\Delta FC(t_0+t)$ is the difference in forest C stocks between the increased and the baseline harvesting scenarios at year t , and $GHG_{net}(t_0+t)$ is the net GHG effects of the HWP produced from increased forest harvesting.

Note $GHG_{HWP-inc}(t_0+t)$ in Equation (1) is different from the HWP age-dependent LCA C dynamics described in the previous sub-section, although calculated using the same LCA method: the former was calculated with continuous annual harvest for a 100-year simulation period, while the latter was calculated assuming 100-unit of wood carbon entering HWP life-cycle in year 1 and no input wood in the remaining 99 years.

The change in forest C between the increased and the baseline harvesting scenarios, $\Delta FC(t_0+t)$, is calculated using Equation (2):

$$\Delta FC(t_0 + t) = FC_{inc-har}(t_0 + t) - FC_{baseline}(t_0 + t) \quad (2)$$

where $FC_{baseline}(t_0 + t)$ and $FC_{inc-har}(t_0 + t)$ are the forest C stocks of the baseline and the increased harvesting scenarios at year $t_0 + t$, respectively. Here, $FC_{baseline}(t_0) = FC_{inc-har}(t_0)$, i.e. at the beginning t_0 , both harvesting scenarios have the same forest C stock. The change in forest C stock, $\Delta FC(t_0 + t)$, is determined by the C stock changes in the area affected by increased harvesting as analysed before.

We also used the time to C sequestration parity to assess the net HWP GHG effects. Similar to Ter-Mikaelian *et al.* (2015), time to C sequestration parity was defined as the time required for the combined forest C stocks of the increased harvesting scenario and the LCA emissions/removals of the HWP produced from additional harvesting to equal the forest C stocks of the baseline scenario. In Equation (1), the number of years, t_{csp} , when $GHG_{net}(t_0 + t_{csp}) = 0$, is the time to C sequestration parity.

Figure 2 is a stand scale analysis used to illustrate the concept of net GHG effects and carbon sequestration parity for HWP originated from sustainably managed forests. The short thick solid curve on the top-left corner represents the forest carbon stocks before the stand is harvested at t_0 . Wood harvested at t_0 is assumed to be used to produce HWP in the year

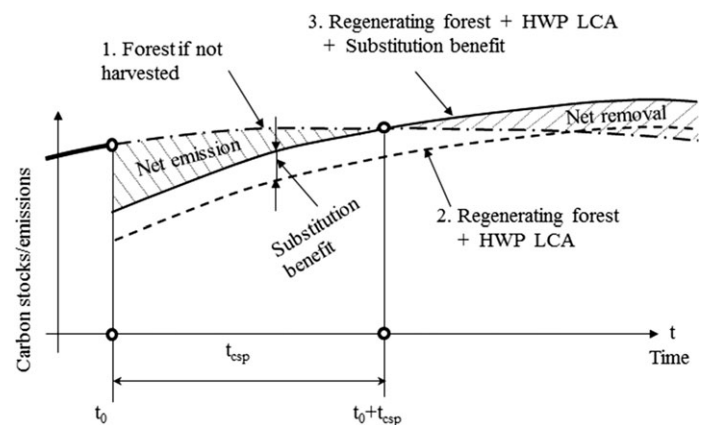


Figure 2 Net greenhouse gas effects and time to carbon sequestration parity for harvested wood products (HWP) that originated from a sustainably managed forest stand. The three curves represent different combinations of forest and HWP carbon analysis. The shaded area on the left between curves 1 and 3 represents cumulative net emissions, while the shaded area on the right represents cumulative net emission reduction. LCA: life-cycle analysis of carbon stocks and emissions; t_0 : the time when the forest is harvested; t_{csp} : time to carbon sequestration parity.

of harvesting. The three curves starting from t_0 represent different combinations of forest and HWP carbon stocks/emissions analysis: curve 1 shows the forest C stocks if the forest is not harvested; curve 2 represents the carbon stocks of the regenerating forest and the life-cycle C stocks/emissions of the HWP produced from harvesting the forest; and curve 3 illustrates the carbon stocks of the regenerating forest, the life-cycle C stocks/emissions of the HWP produced from harvesting the forest, and the substitution benefit of reduced emissions (the gap between curves 2 and 3) from using some of the HWP to substitute for non-wood materials in construction. The difference between curve 1 and 3 determines the net GHG effects of the HWP originated from the forest stand: the shaded area at left between these curves represents the cumulative net GHG emissions from harvesting the forest to produce HWP; the net GHG emissions decreases to zero at t_0+t_{csp} , when C sequestration parity is achieved; and the shaded area at right illustrates the cumulative net GHG emissions reduction resulting from producing and using the HWP that originated from this forest stand. At a forest management unit scale, where different forest stands are harvested every year, the C curves are more complicated, but the concept and analysis of net GHG effects of HWP are the same.

A sensitivity analysis was conducted (Supplementary data: 6. Sensitivity analysis) to evaluate how the key HWP LCA parameters incorporated in HWP-CASE affect HWP LCA C stocks/emissions (including HWP substitution effects).

Results

Life-cycle carbon dynamics of Canadian harvested wood products

The C dynamics of Canadian-made HWP by HWP production scenario are illustrated in Figure 3. Results are presented as relative values of five HWP LCA C components (i.e. C stock of HWP in use, C stock of HWP in landfills, substitution effects, HWP production emissions and landfill methane emissions) for 100 units of harvested wood C, as well as the overall HWP LCA C emissions/removals. Comparing the overall C emissions/removals between the five production scenarios can help to identify preferred wood use for maximum GHG mitigation potential. For all HWP production scenarios (Figure 3a–e), the in-use HWP C stocks decrease relatively faster in the first a few decades; consequently, the C stocks of HWP in landfills increase rapidly over the same period, stabilizing when the fractions of in-use HWP C stocks approach zero. Production emissions occur with forest harvesting and wood transport, and when HWP are manufactured, appearing in year 1 and remaining stable, while landfill methane emissions are cumulative and increase as wood/HWP decomposes over time. The substitution benefit of solid HWP is the emissions reduction achieved when using HWP to replace non-wood materials in construction, with small additions from landfill methane collected over time that was assumed to be used to produce electricity. For the paper and paper products scenario (Figure 3e), the small increasing substitution benefit is obtained from landfill methane-based energy production. In Figure 3a–e, two additional curves were produced to illustrate the range of HWP LCA emissions/removals, in which HWP substitution benefits were estimated using low- and high-end displacement factors calculated in the section Methods (Wood substitution effects on reducing fossil fuel-based emissions).

Figure 3f shows that the combined LCA emissions/removals changed from –157, –145, –125, –95 and –42 (per 100 unit of input wood C) in year 1, to –122, –105, –82, –44 and +35 in

year 100, for structural panel, lumber, non-structural panel, BAU HWP, and pulp and paper production scenarios, respectively, in which the substitution benefits for the first four production scenarios were estimated using the average displacement factor (Table 1). A value below –100 indicates the combined LCA C emissions/removals exceed input wood C stock, while a positive value reflects C emissions equivalent to having all the input wood C released back to the atmosphere with additional C emissions equal to the positive value. Four additional curves are presented to show HWP C stock/emission changes without substitution effects included (except for pulp and paper production scenario that did not include material substitution); the results show that the combined LCA emissions/removals change from –67, –71, –69 and –55 (per 100 unit of input wood C) in year 1 to –31, –30, –23 and –1 in year 100 for structural panel, lumber, non-structural panel and BAU HWP production scenarios, respectively.

These LCA emissions/removals curves were produced to illustrate HWP C dynamics over HWP life-cycle, i.e. from forest harvesting until after the retired HWP are disposed of, but do not include changes in forest C stocks due to harvest. However, assessing net GHG effects of HWP also requires forest C stock changes to be considered, as reported in the following section.

Net greenhouse gas effects of harvested wood products – a case study of consequential life-cycle analysis

Changes in total forest C stocks of the four forest management units differed between the baseline and the increased harvesting scenarios: in the baseline scenario, the C stock decreased from 1359 million tonnes of C (Mt C) in the beginning to 1353 Mt C by year 20, increased to 1426 Mt C by year 80, and thereafter became relatively stable; in comparison, in the increased harvesting scenario, the C stock decreased to 1309 Mt C by year 30, and then continuously increased to 1373 Mt C in year 100 (Table S4, Supplementary data).

Average decadal wood volume harvested from the four forest management units in northwestern Ontario were projected as 18.5 million m³ when the forest is harvested at historical rates (baseline harvesting scenario) and 30.3 million m³ at 95 per cent of the maximum allowable harvest volume (increased harvesting scenario), which is equivalent to an additional 11.8 million m³ of wood harvested every 10 years. The additional wood volumes were first converted to C (Table S4, Supplementary data), and following the approach described in Methods (Assessment of harvested wood product net greenhouse gas effects), we estimated the net GHG effects for the HWP produced from using the additional harvested wood (Figure 4). Using the decadal cumulative analysis results obtained from FORCARB-ON2 and HWP-CASE simulations (Table S4, Supplementary data), the time to C sequestration parity reported in Table 3 were estimated using linear interpolation based on the net GHG effects of the two adjacent decades where the net GHG effects change from positive (net emission) to negative (net removal). When the average displacement factors was used to estimate HWP substitution benefits and the additional harvested wood was primarily used to produce structural panels, no additional time was required to reach C sequestration parity (Table 3) (i.e. the HWP produced were immediately C neutral), because the combined LCA C

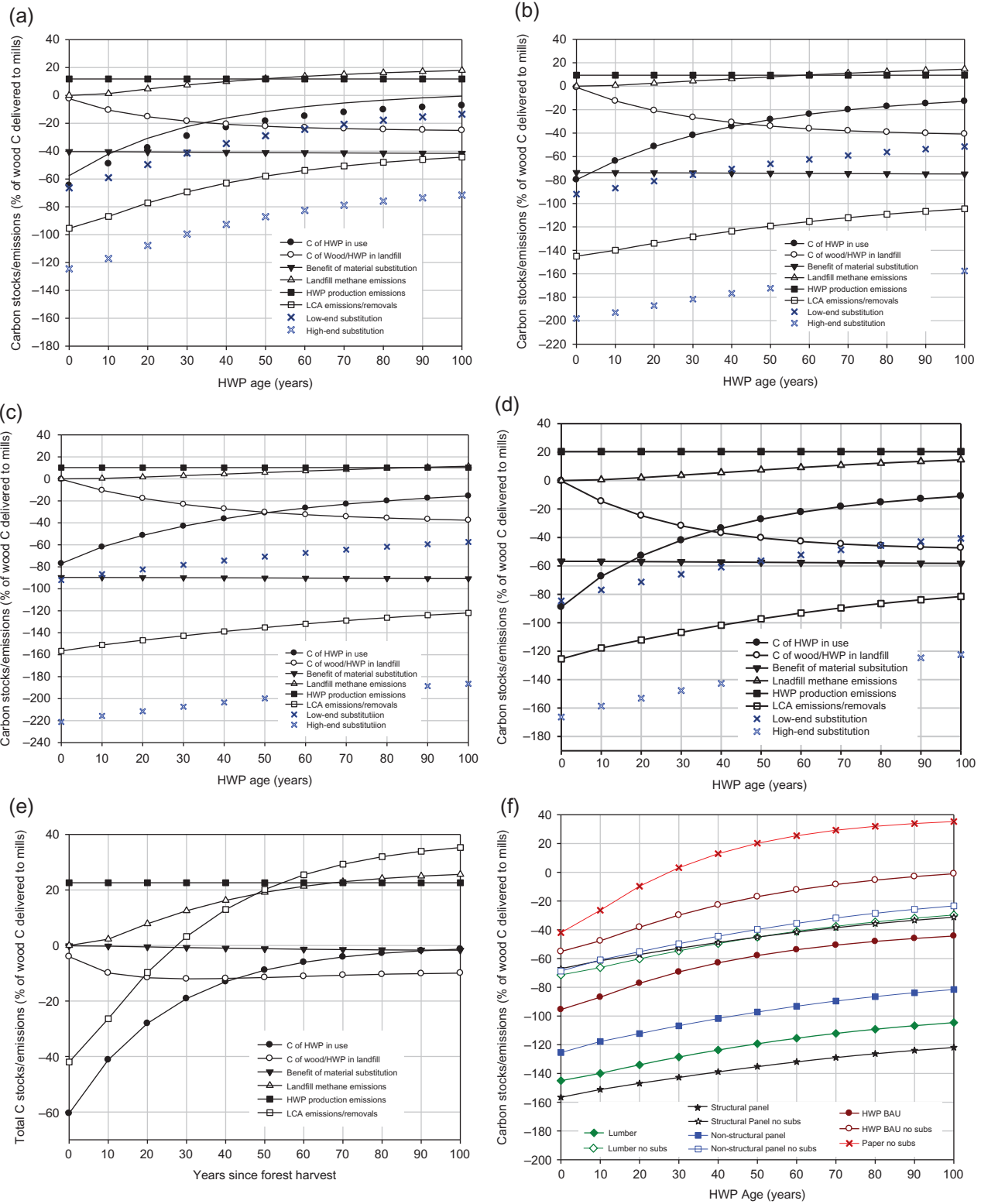


Figure 3 Life-cycle harvested wood product (HWP) carbon (C) stocks and emissions presented as percentages relative to C in wood delivered to mills in five HWP production scenarios: (a) business-as-usual (BAU) HWP, (b) lumber, (c) structural panels, (d) non-structural panels, and (e) pulp and paper (no substitution considered), and (f) comparison of net HWP life-cycle analysis (LCA) emissions/removals for the five scenarios (curves specified with a 'no subs' suffix in legend do not include HWP substitution benefits). Positive and negative values represent LCA emissions and removals, respectively. 'Low-end substitution' and 'high-end substitution' in (a)–(e) represent HWP LCA emissions/removals with HWP substitution benefits estimated using the low-end and high-end displacement factors, respectively.

emissions/removals of the structural panels produced exceeded forest C stock decreases from the increased harvesting. Time to C sequestration parity for lumber, non-structural panel, and BAU HWP production scenarios were estimated as 21, 39 and 84 years, respectively (Table 3). Similarly, when the low-end displacement factor was used, the time to C sequestration parity were estimated to be 75, 78 and 89 years for structural panel, lumber, and non-structural panel, respectively, while C sequestration

parity could not be achieved in the 100-year simulation period for the BAU HWP. However, if the high-end displacement factor was used, the times to C sequestration parity were estimated to be 43 years for BAU HWP, with the other three solid HWP production scenarios were estimated to immediately be C neutral.

After 100 years of increased harvesting and using the wood to produce HWP, with substitution benefits estimated using the average displacement factor, reductions in net GHG remissions were estimated as 112, 93, 66 and 21 Mt CO₂eq for structural panel, lumber, non-structural panel and BAU HWP production scenarios, respectively (Table 3, Figure 4). Since the four forest management units have 2.21 million ha of forest area managed for timber production, when the substitution benefits were estimated using the average HWP displacement factor, the annual per ha emissions reduction ranges from 0.10 to 0.51 Mt CO₂eq for the 100-year period. Using the additional harvested wood to produce pulp and paper products provided the least favourable GHG profile, never achieving C sequestration parity and increasing GHG emissions to 66 Mt CO₂eq after 100 years (Figure 4).

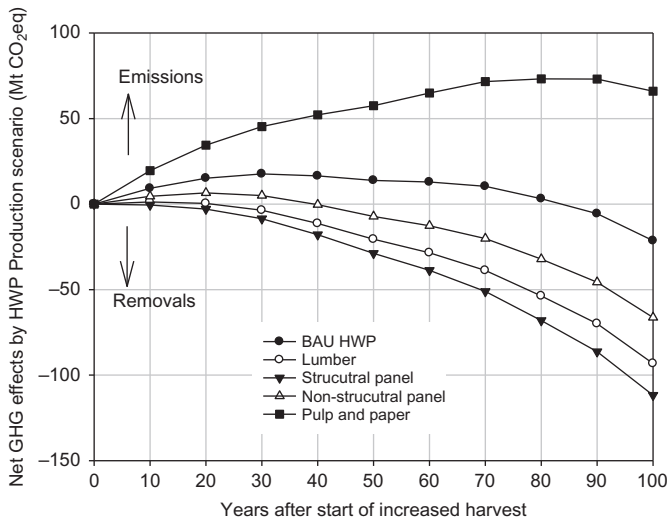


Figure 4 Carbon sequestration parity and consequential life-cycle analysis of net greenhouse gas (GHG) effects of the five harvested wood product (HWP) production scenarios for increasing harvesting from historic rates to 95 per cent of allowable harvest; results are combined for the four forest management units used for the case study. The harvested wood specified in the baseline harvesting scenario was assumed to be used to produce HWP following a business as usual (BAU) production scenario (Table 2) for both the baseline and increased harvesting scenarios so the GHG effects of these HWP cancel each out in scenario comparisons; the additional wood harvested in the increased harvesting scenario was assumed to be used in the same proportions as the five HWP production scenarios (Table 2). Reduced emissions from using HWP to substitute for non-wood materials in construction were estimated using the average displacement factor (Table 1).

Table 3 Time to carbon sequestration parity and greenhouse gas (GHG) emissions reductions by harvested wood product (HWP) production scenario after 100 years of increased harvest. The harvested wood specified in the baseline harvesting scenario was assumed to be used to produce HWP in a business as usual (BAU) production scenario (Table 2) for both the baseline and the increased harvesting scenarios, thus the GHG effects of these HWP cancel each other out in comparisons; the additional wood harvested in the increased harvesting scenario was assumed to be used in the same proportions as in the four HWP production scenarios (Table 2)

Displacement factor ²	Time to carbon sequestration parity (years)			Net greenhouse gas effects ¹ (Mt CO ₂ eq)				
	Structural panel	Lumber	Non-structural panel	BAU HWP	Structural panel	Lumber	Non-structural panel	BAU HWP
Low-end	75	78	89	— ³	-33.0	-28.5	-16.3	14.1
Average	0	21	39	84	-111.8	-93.3	-66.2	-21.4
High-end	0	0	0	43	-190.6	-158.1	-116.1	-56.8

¹Negative values indicate reduced GHG emissions.

²Represent the low-end (2.51 tCO₂/tC), average (8.91 tCO₂/tC), and high-end (15.33 tCO₂/tC) displacement factors used in estimating HWP substitution benefits.

³Carbon sequestration parity is not achieved within the 100-year simulation period.

a lack of data for use of paper and paper products to substitute for other materials, we assumed that they provide no direct substitution benefits. As a result, harvesting forests to produce pulp and paper only contributes to increased GHG emissions.

The sensitivity analysis results (Table S5, Supplementary data) reveal that HWP LCA C stocks/emissions are most sensitive to the fraction of logs (relative to pulpwood) in total wood harvested, and the conversion efficiencies from logs to solid HWP—together, these parameters determine the fraction of harvested wood C contained in finished solid HWP. The second set of sensitive parameters are fractions of solid HWP used in construction, fraction of solid HWP used in construction that substitute for non-wood materials, and HWP displacement factor—together, these parameters determine the reduced emissions from HWP substitution. Thus, our results support the general conclusion of [Pingoud et al. \(2010\)](#) that the production efficiency of converting input wood to solid HWP and the fraction of HWP used in long-lived end uses such as construction are the highest contributors to differences in mitigation benefits. Converting more harvested wood to solid HWP and using more solid HWP in long-lived end uses result in more and longer-term C storage in HWP in use and greater material substitution benefits. In addition, if disposed of in landfills, a larger fraction of retired solid HWP will not decay or will decay very slowly, while the degradable fraction decomposes more slowly than that of paper and paper products ([Skog, 2008](#)); as a result, solid HWP can retain more C in landfills and produce relatively less methane emissions.

The sensitivity analysis also helps to explain differences in LCA among the HWP production scenarios. Lumber and structural panel production scenarios have similar production efficiencies, respectively, converting 79.5 and 77.0 per cent of input wood to finished solid HWP. In the structural panel scenario, a significantly larger fraction of HWP used in construction (76.8 per cent of total production) than does lumber because a larger fraction of non-structural panels are produced in the latter (as defined in Table 2); as a result, for the lumber scenario only 63.7 per cent of finished HWP are used in construction. Consequently, using wood to produce structural panels produces better mitigation results than using it to produce lumber. Though the non-structural panel production scenario has the highest production efficiency (converting 88.9 per cent of input wood to product), only 40.1 per cent of the non-structural panels are used in construction ([Chen et al., 2013](#), Table 16), providing less C stocks in HWP in use with generally shorter service lives and less substitution benefits, and consequently more C stocks in HWP in landfills. Thus, in terms of mitigating GHG emissions, the non-structural panel scenario is less efficient than the structural panel and lumber production scenarios.

The four hypothetical HWP production scenarios (lumber, structural panel, non-structural panel, and pulp and paper) and a BAU production scenario were used to investigate the GHG mitigation potential of different wood uses. In future HWP manufacturing, the fractional shares of harvested wood use will certainly differ from any of these scenarios. However, our results indicate that producing solid HWP, especially structural panels and lumber, can help to reduce GHG emissions, if the forests harvested to provide the wood are managed sustainably and the assumed substitution benefits actually occur.

Our results suggest that reduced emissions by using HWP to replace non-wood construction materials are a critical component in estimating the mitigation potential of harvesting forests

to produce solid HWP (Figure 3), which is also supported by the sensitivity analysis (Supplementary data: 6. Sensitivity analysis). Other studies (e.g. [Lippke et al., 2011](#); [Ximenes et al., 2012](#); [Oliver et al., 2014](#); [Butarbutar et al., 2016](#); [Nepal et al., 2016](#); [Smyth et al., 2016](#)) have also indicated the importance of including substitution effects when assessing the GHG mitigation effects of HWP. Based on a meta-analysis, [Sathre and O'Connor \(2010\)](#) found that HWP displacement factors range from -8.4 to 55.1 tCO₂eq of reduced emissions per tC in HWP used to substitute for non-wood construction materials, with the low- and high-end values representing extreme scenarios that are unlikely; most displacement factors were in the range of 3.7 – 11.0 tCO₂eq of reduced emissions per tC for HWP used to replace non-wood construction materials. Variations among published displacement factors result from diverse assumptions, life-cycle system boundaries and the energy source used to produce construction materials ([Sathre and O'Connor, 2010](#)). For example, some studies include the C dynamics and methane emissions from HWP disposed of in landfills, resulting in significantly smaller displacement factors; while others consider recycling and re-use of retired HWP resulting in larger displacement factors. [Sathre and O'Connor \(2010\)](#) concluded the consensus was that GHG emissions are reduced if HWP replace non-wood construction materials.

Similar to [Smyth et al. \(2016\)](#), we used a comprehensive approach to estimate displacement factors based on embodied emissions of HWP and alternative construction materials, with forest C balance, C stocks of HWP in use, and HWP post-service disposals considered separately. However, the average displacement factor produced by [Smyth et al. \(2016\)](#) for Canadian-made HWP is 1.98 tCO₂eq (or 0.54 tC) of reduced emissions per tC in HWP. This is significantly smaller than the 8.91 tCO₂eq of reduced emissions per tC in HWP reported in our study. Reasons for this difference are numerous. First, the displacement factor produced by [Smyth et al. \(2016\)](#) is applicable to all HWP produced using the additional harvested wood; in comparison, our displacement factors are applicable to the HWP that substitute for non-wood construction materials, and thus valid for the use of HWP in construction only. We assumed that 62 per cent of the HWP produced from increased harvesting was used in construction, of which 64 per cent was used to substitute for non-wood materials, resulting in reduced emissions. If [Smyth et al. \(2016\)](#) had used our assumptions, their displacement factor values would be higher. Furthermore, we reduced the published embodied emissions by up to 50 per cent to calculate HWP displacement factors, based on [Meil et al. \(2009\)](#) who estimated that mill residue-based energy accounted for more than 50 per cent of the total energy consumed by Canadian HWP industries. In contrast, in references used by [Smyth et al. \(2016\)](#) to estimate their displacement factor, e.g. [Lippke et al. \(2004\)](#) and [Gustavsson et al. \(2006\)](#), the ratio of bioenergy in the embodied energy is far less than 50 per cent of total energy consumption. Therefore, the displacement factors used by [Smyth et al. \(2016\)](#) might underestimate substitution effects for Canadian-made HWP.

Residential construction in the United States and Canada has been the largest consumer of solid HWP manufactured in North America ([Skog, 2008](#); [Chen et al., 2013](#)). [Lippke et al. \(2004\)](#) and [Upton et al. \(2008\)](#) conducted life-cycle analyses for typical wood- and non-wood-based single family house construction in the United States; using their embodied emissions values, and

assuming mill residue-based energy accounts for 50 per cent of the total embodied energy for Canadian-made HWP, we calculated displacement factors in their studies of 8.8 and 21.3 tCO₂eq of reduced emissions per tC in HWP used to substitute for steel and concrete in residential construction, respectively. These factors are comparable to or greater than our result of 9.56 tCO₂eq of reduced emissions per tC in HWP.

We purposely chose the same forest management units for the case study as were used by [Ter-Mikaelian et al. \(2015\)](#), who investigated the GHG mitigation potential from using wood biomass to replace coal in power generation in Ontario. We also used the same baseline and increased harvesting scenarios, as well as the definition of time to C sequestration parity so that the GHG effects of using wood to substitute fossil fuels and using wood to produce HWP were directly comparable. If the additional harvested wood from the four forest management units was used to replace coal in power generation in Ontario, [Ter-Mikaelian et al. \(2015\)](#) estimated the time to C sequestration parity to be 91 years with reduced emissions of 14.7 Mt CO₂eq after 100 years of increased harvesting. In comparison, we estimated that 0, 21, 39 and 84 years are needed to reach C sequestration parity if the additional wood from increased harvesting is primarily used for structural panel, lumber, non-structural panel, and BAU HWP production scenarios, respectively. The GHG emission reductions after 100 years of increased harvesting were estimated to be 111.8, 93.0, 66.2 and 21.4 Mt CO₂eq for the same production scenarios, respectively, when HWP substitution benefits were estimated using the average displacement factor. In addition, in [Ter-Mikaelian et al. \(2015\)](#), wood was assumed to substitute for coal, the most GHG intensive fossil fuel, and the distance to transport harvested wood to the pellet plant, and pellets to power generation station were assumed reasonably short; thus, they reported a higher GHG emissions reduction potential relative to other fossil fuel substitution with a displacement factor of 4.05 tCO₂eq per tC in live tree wood harvested to replace coal in power generation, which is greater than the low-end value we chose from the range of displacement factor values. However, even HWP substitution benefits were estimated using the low-end displacement factor, using the wood to produce structural panel and lumber appears notably better in net GHG effects than using the wood for power generation in [Ter-Mikaelian et al. \(2015\)](#), while using the wood for non-structural panel is only slightly better. Therefore, using wood to produce solid HWP (especially structural panel and lumber), and using the HWP in long-lived end uses is a much better option for GHG mitigation, a conclusion generally supported by other studies ([Eriksson et al., 2007](#); [Gustavsson et al., 2006](#); [Knauf, 2015](#); [Pingoud et al., 2010](#); [Smyth et al., 2016](#)).

Time to C sequestration parity is one way to evaluate mitigation activities based on when the net emissions reductions are obtained ([Pingoud et al., 2012](#)). An activity might be acceptable if it increases atmospheric GHG in the near term, but provides mitigation benefits within a set timeframe. In this study, we estimated that it requires 0, 21 and 39 years to reach C sequestration parity when the additional wood from increased harvest was used to produce structural panels, lumber, and non-structural panels, respectively, and the average HWP displacement factor was used. These estimates are comparable to that reported by [Ximenes et al. \(2012\)](#), who projected net GHG emissions reduction 30–50 years after harvesting forests

to produce HWP, compared with a no-harvest scenario in Australia. In a boreal forest study, [Pingoud et al. \(2012\)](#) estimated C sequestration parity occurred 0 and 36 years after harvesting.

In our study, the forest C stock of all four forest management units combined eventually stabilizes at ~1426–1427 for the baseline and 1358–1337 Mt CO₂eq for the increased harvesting scenario, with the increased harvesting reducing forest C stocks by 55 Mt CO₂eq by year 100. After factoring in forest C decreases due to harvesting, the net GHG emissions removed by increasing harvesting and using the wood for structural panel, lumber, or non-structural panel production are 111.8, 93.3 and 66.2 Mt CO₂eq, respectively, by year 100. Thus, over the long term, sustainable forest harvesting reduces GHG emissions more than forest conservation but at a possible cost of increased emissions in the short- to medium term depending on HWP type and end use. This conclusion echoes findings by [Ximenes et al. \(2012\)](#) and [Gustavsson et al. \(2017\)](#) that, in the long term, producing HWP and bioenergy by harvesting sustainably managed forests mitigates GHG more than conserving forests.

For this study, we used FORCARB-ON2, an improved version of Ontario's forest C budget model (FORCARB-ON) that was used to estimate future HWP C stocks (HWP in use and in landfills) based on projected harvesting in Ontario ([Chen et al., 2008, 2010](#); [Ter-Mikaelian et al., 2013](#)). Due to a lack of data and/or methods, HWP production emissions, landfill methane emissions, as well as reductions in GHG emissions from substituting HWP for non-wood construction materials, were not included in FORCARB-ON. In the last two decades, more HWP life-cycle data have become available, allowing comprehensive analysis to support the development of modelling approaches and parameters that can be used to quantify HWP C stocks and flows from harvesting to disposal ([Chen et al., 2013](#)), and to estimate production emissions ([McKechnie et al., 2014](#)) for HWP produced in Ontario and Canada. These new modelling approaches and parameters have been used to develop the HWP module in FORCARB-ON2, and the stand-alone model HWP-CASE, with the latter used to quantify the C stocks and emissions for Canadian HWP produced between 1900 and 2010 ([Chen et al., 2014](#)). In general, HWP manufacturing, end uses and post-service disposal are more accurately and more completely simulated in HWP-CASE and FORCARB-ON2. In particular, in FORCARB-ON it was assumed that 85 and 90 per cent of retired solid HWP and paper and paper products, respectively, were disposed of in landfills ([Chen et al., 2008](#)). However, from 2000 to 2010, 67 per cent of retired Canadian-made solid HWP were estimated as being discarded in landfills, but in 2009 only 12 per cent of Canadian paper and paper products were estimated as disposed of in landfills with another 67 per cent recycled ([Chen et al., 2013](#)). Therefore, for the same amount of harvested wood, the C stock of HWP disposed of in landfills estimated using FORCARB-ON2 or HWP-CASE will be much less than that from FORCARB-ON.

Our results suggest that after 100 years of increased harvest, the GHG mitigation potential from using the additional harvested wood to produce lumber, structural panels and non-structural panels, when HWP substitution effects were estimated using the average displacement factor, are from 4.5 to 7.6 times that if the wood is used to replace coal in power generation (as per [Ter-Mikaelian et al. 2015](#)). Producing solid HWP also reduces time to

C sequestration parity relative to harvesting trees for bioenergy. Thus, GHG mitigation can be enhanced by using more wood harvested from sustainably managed forests to produce solid HWP, especially structural panels and lumber. Nepal *et al.* (2016) demonstrated the potential for reducing GHG emissions by increasing HWP uses in low-rise non-residential construction in the United States when the C stocks of forest and HWP LCA were integrated. The interest in tall wood buildings (e.g. Karacabeyli and Lum, 2014) also provides an avenue for increased use of wood in construction. In the 2015 editions of Canada's National Building Code, one of the significant changes is allowance for constructing wooden buildings up to six stories high, compared with previous restrictions to four stories. This change is expected to increase the use of HWP in mid-rise building construction, contributing to reduced GHG emissions in Canada.

Supplementary data

Supplementary data are available at *Forestry* online.

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Conflict of interest statement

None declared.

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