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POLYTECHNIQUE MONTRÉAL

affiliée à l'Université de Montréal

Improvement of Biogenic Carbon Accounting in the Life Cycle of Wood used in Construction in Canada

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Département de mathématiques et de génie industriel

Thèse présentée en vue de l'obtention du diplôme de *Philosophiae Doctor*

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présentée par **Marieke HEAD**

en vue de l'obtention du diplôme de *Philosophiæ Doctor*

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DEDICATION

For Amelia and Lentil,

“Wat geweldig dat niemand ook maar één moment hoeft te wachten met het verbeteren van de wereld.” (“How wonderful it is that nobody need wait a single moment before starting to improve the world.”)

Anne Frank

“We must reinvent a future free of blinders so that we can choose from real options.”

David Suzuki

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The end of the road! I sometimes never thought that this moment would actually happen!

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RÉSUMÉ

Au Canada, le bois est généralement utilisé comme matériau de construction. Cependant, il existe des limites dans la méthodologie de comptabilité des impacts climatiques des matériaux de construction en bois en analyse de cycle de vie (ACV). Typiquement, en ACV, l'hypothèse de la neutralité du carbone biogénique est utilisée pour la biomasse et pour les produits issus du bois, puisque le carbone séquestré par la biomasse est égal au carbone qui est émis éventuellement par cette biomasse en fin de vie. Étant donné la nature dynamique des émissions de dioxyde de carbone et de l'effet de serre, le paradigme simpliste qu'un bilan de carbone neutre est égal à la neutralité de carbone est remis en question.

Il y a une quantité croissante de preuves scientifiques indiquant que les impacts climatiques réels des produits issus du bois dépendent de plusieurs facteurs, dont le temps de stockage et le timing des émissions, le type d'aménagement forestier et le type de traitement en fin de vie. L'objectif global de ce travail doctoral est de développer une méthode qui comptabilise systématiquement la captation, l'émission et le stockage de carbone biogénique dans les ACV du bois utilisé dans les bâtiments en 1) développant des profils de flux de carbone qui sont différentiés temporellement en tenant compte de la dynamique du carbone forestier, 2) développant des profils de flux de carbone différentiés temporellement du point de récolte jusqu'à la fin de vie, et 3) mettant en pratique l'ACV dynamique aux profils de flux de carbone différentiés temporellement pour une ACV berceau-au-tombeau d'un produit issu de bois.

La plupart des ACV ne considèrent pas l'aménagement forestier des produits issus du bois. Ce travail de recherche vise à améliorer la comptabilité du carbone biogénique de la phase forestière du cycle de vie des produits issus de bois au Canada. Ceci implique la modélisation spécifique des flux de carbone en fonction des espèces d'arbres, des conditions de croissance et des pratiques d'aménagement forestier des forêts canadiennes aménagées. En général, les résultats démontrent que pour la plupart des paysages forestiers, la récolte de bois dans la forêt canadienne boréale engendre des émissions nettes négatives. Ce travail produit aussi des flux de carbone, appelé « ecosystem carbon costs » (ECC) pour la plupart des espèces de conifères utilisées dans la construction canadienne. Ces ECC qui peuvent être utilisés pour modéliser le carbone des écosystèmes forestiers associés avec le produit issu de bois. Les conséquences de ces résultats sont

que la récolte durable de bois provenant de la plupart des paysages forestiers canadiens génère une séquestration nette, au-delà de ce qui a déjà été séquestré dans le bois récolté en soit. Compte tenu des effets bénéfiques de la récolte durable des forêts sur le bilan de carbone biogénique global des produits issus de bois, la dynamique du carbone des forêts devrait toujours être incluse dans les ACV des produits issus de bois.

La comptabilité du carbone biogénique n'est présentement pas considérée à travers la durée de vie des produits issus de bois dans les études d'ACV. Ce travail vise à améliorer la comptabilité du stockage et des flux de carbone dans les produits issus de bois à longue durée de vie en ACV. Le carbone biogénique du bois rond façonné est suivi à travers la production de produits issus de bois, la vie du bâtiment et sa fin de vie. À partir de ces étapes, les stocks et flux de carbone vers l'atmosphère ont été estimés. Les résultats démontrent que le degré de délai des émissions de fin de vie est principalement dépendant des paramètres tels que le type de produit issu de bois, la région où le bois est utilisé et la durée de vie. Ce travail développe des profils de carbone biogénique qui permettent la modélisation des ACV berceau-au-tombeau dynamique des produits de construction issus de bois canadiens. Les résultats impliquent que le carbone biogénique du traitement du bois jusqu'à la fin de vie peut avoir des émissions de carbone variables, qui dépendent des paramètres spécifiques du bâtiment.

Après avoir considéré les émissions, le stockage et la prise du carbone biogénique dans les ACV de produits issus de bois, l'élément final du projet est d'intégrer le timing des émissions de gaz à effet de serre. L'objectif de ce travail est de calculer une base de données d'inventaire de cycle de vie qui est différentié temporellement ce qui rend les impacts de changement climatique dynamique selon des contextes d'utilisation différents à travers le Canada. Les résultats incluent tous les éléments de ce travail de recherche, qui permet l'évaluation d'impacts de changement climatiques du berceau-au-tombeau. À cet effet, ils permettent de prononcer un verdict sur la pertinence de la neutralité de carbone biogénique des produits issus de bois. Dans la majorité des cas, les impacts nets de changements climatiques du cycle de vie des produits issus de bois sont négatifs. Ceci implique que l'hypothèse de neutralité de carbone pour le carbone biogénique serait conservatrice et a tendance à surestimer les impacts de changement climatique du cycle de vie.

Les cadres établis dans cette recherche doctorale permettent l'évaluation complète du berceau-au-tombeau des impacts de changements climatiques des produits issus de bois dans le contexte de

l'industrie de la construction canadienne. Les résultats des impacts des changements climatiques en soit démontrent que la plupart des produits issus du bois ont une séquestration de carbone nette sur le cycle de vie et par conséquence, les émissions de carbone biogénique du cycle de vie ne s'annulent pas. Ces conclusions s'ajoutent à celles déjà présentes dans la littérature et permettent de dissiper le mythe que les produits issus de bois devraient être considérés carboneutres en ACV.

ABSTRACT

In Canada, wood is commonly used as a building material throughout the construction sector. However, the climate impacts of wood construction materials currently have limitations in how they are accounted for in life cycle assessment (LCA). Typically in LCA, a biogenic carbon neutral assumption is used for biomass and wood products, due to the fact that the carbon sequestered by biomass is equal to the carbon eventually released by that biomass. Given the dynamic nature of carbon dioxide emissions and the resulting effect on the greenhouse gas effect and subsequently on climate change, the simplistic paradigm that carbon neutral equals climate neutral is being questioned.

There is an increasing body of scientific evidence that the actual climate impacts are dependent on many factors, such as storage time and emissions timing, the type of forestry management practiced, and the end-of-life treatment of the wood product. The overall object of this PhD work is to develop a method that consistently accounts for the uptake, emission and storage of biogenic carbon in the life cycle assessments of wood used in buildings by 1) developing temporally differentiated carbon flux profiles of the forestry carbon dynamics, 2) developing temporally differentiated carbon flux profiles from the point of harvest through to end-of-life, and 3) applying dynamic life cycle assessment to cradle-to-grave temporally differentiated carbon flux profiles of wood products.

Most wood LCAs do not consider the forest management of wood products. This research work aims to improve the biogenic carbon accounting of the forestry phase of the life cycle of softwood products in Canada. This involves specifically modelling carbon fluxes as a function of tree species, growing conditions and forest management practices, from Canadian managed forests. Overall, the results show that for most forest landscapes, harvesting wood in the Canadian boreal forest results in net negative emissions. The research work also yields carbon fluxes, termed as ecosystem carbon costs (ECC) for most softwood species used in Canadian construction that can be used to model the forestry ecosystem carbon associated with the wood product. The implications of these results are that the sustainable harvesting of wood from most Canadian forest landscapes show a net sequestration, beyond what is already sequestered in the harvested wood itself. Considering the beneficial effects of sustainably harvesting forests on the overall biogenic carbon balance for wood

products, forestry carbon dynamics should always be included in the life cycle assessments of wood products.

Biogenic carbon accounting is currently not considered throughout the lifespan of wood products in LCA studies. This work aims to improve the accounting of carbon storage and fluxes in long-life wood products in LCA. Biogenic carbon from harvested roundwood logs were tracked through wood product manufacturing, building life and end-of-life phases, and carbon stocks and fluxes to the atmosphere were estimated. The results show that the degree of postponement of end-of-life emissions is highly dependent upon the wood product type, region and building lifespan parameters. This work develops biogenic carbon profiles that allows for modelling dynamic cradle-to-grave LCAs of Canadian wood building products. The implications of the results are that the biogenic carbon from wood processing to end-of-life can have variable positive carbon emissions, which are dependent on the specific building parameters.

The final element in considering the biogenic carbon emissions, storage and uptake in wood product LCAs, is to integrate the timing of greenhouse gas emissions. The objective of this work is to calculate a database of temporally differentiated life cycle inventories (LCI) and dynamic climate change impacts of wood products, for different use contexts across Canada. The results encompass all the elements of this research work, allowing for the cradle-to-grave climate change impacts of wood products to be evaluated. In doing so, they allow for a verdict to be made on the relevance of biogenic carbon neutrality of wood products. In all but potentially the most outlying cases where ECC scores are positive or have very low levels of sequestration, the overall net life cycle climate change impacts of wood products are negative. This implies that using a carbon neutrality assumption for biogenic carbon would be a conservative assumption by overestimating overall life cycle climate change impacts.

The frameworks established within this doctoral research allow for a full cradle-to-grave assessment of climate change impacts of wood products in the context of the Canadian construction sector. The climate change impact results themselves show that most wood products have net life cycle carbon sequestration and thus life cycle biogenic carbon emissions do not cancel themselves out. These findings add to the mountain of evidence in the literature that help in dispelling the myth that wood products should be considered biogenic carbon neutral in LCA.

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LIST OF SYMBOLS AND ABBREVIATIONS

AB	Alberta
BC	British Columbia
BL	Building lifespan
BSI	British Standards Institute
CBM-CFS3	Carbon Budget Model of the Canadian Forest Service (software)
CBM-FHWP	Carbon Budget Model Framework for Harvested Wood Products (software)
CH_4	Methane
CLT	Cross-laminated timber
CO_2	Carbon dioxide
CRD	Construction, renovation and demolition
DCCI	Dynamic climate change impacts
DF-L	Douglas fir-Larch
DLCA	Dynamic life cycle assessment
ECC	Ecosystem carbon cost
GHG	Greenhouse gases
GTP	Global temperature change potential
GWI_{cum}	Cumulative global warming impact
GWI_{inst}	Instantaneous global warming impact
GWP	Global warming potential
GWP_{bio}	Global warming potential of biomass-derived CO_2
HWP	Harvested wood products
IPCC	Intergovernmental Panel on Climate Climate (of the United Nations)
LCA	Life cycle assessment

LCI	Life cycle inventory
LCIA	Life cycle impacts assessment
LVL	Laminated veneer lumber
MB	Manitoba
N_2O	Nitrous oxide
NB	New Brunswick
NL	Newfoundland
NS	Nova Scotia
ON	Ontario
OSB	Oriented strand board
PCR	Product Category Rules
PE	Prince Edward Island
QC	Quebec
SK	Saskatchewan
SPF	Spruce-pine-fir
TAWP	Time-adjusted warming potential
TH	Time horizon
U.S.	United States

LIST OF APPENDICES

CHAPTER 1 INTRODUCTION

Currently, buildings are estimated to contribute up to one-third of global carbon emissions, through their construction and operation (UNEP-SBCI, 2009). In order to minimise a building's environmental impacts, it is essential to properly assess the impacts of different design choices. One common means of evaluating the environmental impacts of design choices is life cycle assessment (LCA). LCA is a method used to evaluate the potential environmental impacts of products, which in a building context could be used to evaluate the impacts of energy use, but could also be used to evaluate the impacts of using various types of building materials.

In Canada, wood is commonly used as a building material throughout the construction sector. However, the climate impacts of wood construction materials currently have limitations in how they are accounted for in life cycle assessment. Since biomass is considered to be part of the fast domain of the carbon cycle, the carbon fluxes between the atmosphere and biomass have been differentiated from the carbon fluxes originating from fossil sources. As such, the carbon from biomass, referred to as biogenic carbon, is said to have a net carbon balance of zero, meaning that the carbon sequestered by biomass is equal to the carbon eventually released by that biomass. This net zero *carbon* balance has been equated to a net zero *climate change* impact. Given the dynamic nature of carbon dioxide emissions and the resulting effect on the greenhouse gas effect and subsequently on climate change, the simplistic paradigm that carbon neutral equals climate neutral is being questioned.

There is an increasing body of scientific evidence that the actual climate impacts are dependent on many factors, such as storage time and emissions timing, the type of forest management practiced, and the end-of-life treatment of the wood product. However, to date there has been a lack of consensus on the issue surrounding the climate neutrality assumption. In addition, this issue has not yet been approached in a comprehensive and holistic manner that would consistently account for the climate impacts of wood over the entire life cycle of a building. As such, the aim of this research is to develop a method that reliably accounts for the uptake, emission and storage of biogenic carbon in the life cycle assessments of wood used in buildings in the Canadian context.

CHAPTER 2 LITERATURE REVIEW

In order to contextualise the dissertation, the literature review will be presented in two main sections. The first section, on life cycle assessment and biogenic carbon, will introduce the reader to these concepts such as to give context to the biogenic carbon accounting issue. In the second section, wood products and methodological issues related to the calculation of their climate change impacts will be explored. This will address in detail the forestry, end-of-life disposal and timing aspects of the building life cycle, where biogenic carbon has important implications on climate change impacts.

2.1 Life Cycle Greenhouse Gas Assessment

At the very core of this research project, is life cycle assessment (LCA), a well-known and well-accepted tool that is used to assess the environmental impacts of a product throughout its product life cycle. In the context of this project, current LCA methodology concerning the accounting of biogenic carbon in climate change impact assessment, will be further broadened and developed. This section will introduce the reader to LCA, such as to provide context to the discussion on the accounting of biogenic carbon in wood products and buildings in later sections.

2.1.1 Introduction to Life Cycle Assessment

In its simplest terms, life cycle assessment (LCA) is a structured method that accounts for all resource inputs and emissions at every life cycle stage of a product or system, from its inception, resource extraction to its final disposal (European Commission, 2010c). It is a comprehensive approach that considers all quantifiable environmental impacts and in doing so strives to avoid the displacement of one environment impact for another (Bjørn et al., 2018d, p. 12). LCA can also be considered a decision-making tool, by comparing the environmental impacts of different production or process alternatives. The ISO 14044 standard on LCA defines four main phases (ISO, 2006a):

1. Goal and Scope Definition
2. Life Cycle Inventory (LCI)
3. Life Cycle Impact Assessment (LCIA)
4. Interpretation

2.1.1.1 Goal and Scope Definition

In the goal and scope definition, the LCA is framed and defined. More specifically, the context for the study and the intended audience are identified (Bjørn et al., 2018a, p. 68) and the product(s) or system(s) are described. The identification and description of the product(s) or system(s) includes determining the functional unit, system boundaries, how multifunctional processes will be treated, and the environmental impact categories (LCIA – see below) to be covered (Bjørn et al., 2018c, p. 76; European Commission, 2010c).

2.1.1.2 Life Cycle Inventory (LCI)

The objective of the life cycle inventory (LCI) phase is to collect and model data regarding all the processes in the studied product(s) or system(s) (European Commission, 2010c). In particular, in the LCI phase, all material, energy, emission flows flowing into and out of the product system (known as elementary flows) are considered. These are summed for all processes within the product life cycle and form the inventory for the studied product(s) or system(s) (Bjørn et al., 2018b, p. 118).

Some processes have more than one function and yield more than one product (coproducts), and thus they can be considered multifunctional (Bjørn et al., 2018c, p. 89). The process' inventory must be attributed to the product considered by a given study's function and functional unit. The ISO 14044 standard presents a hierarchy for solving multifunctional solutions (Bjørn et al., 2018c, p. 90; ISO, 2006b):

1. Attempt to subdivide multifunctional process inventory if separate inventories are available for each coproduct
2. Identify the most probable alternative production route for the other coproduct and expansion system boundaries to include this coproduct.
3. Allocate process inventory to primary product based on causal physical relationship, such as mass.
4. Allocate process inventory to primary product based on representative physical relationship such as energy density.
5. Allocate process inventory to primary product based on another parameter, such as economic relationship.

2.1.1.3 Life Cycle Impact Assessment (LCIA)

In the life cycle impact assessment (LCIA) phase, elementary flow inputs and outputs collected and modelled in the LCI phase are converted into potential environmental impacts and damages (European Commission, 2010c). The potential environmental impacts and damages that are calculated are those defined in the goal and scope definition. The conversion of environmental inputs and outputs to interpretable impact categories (referred to as LCIA), is accomplished using impact assessment methods. Since the early nineties, there have been several methods developed to quantify emissions into tangible potential environmental impacts (European Commission, 2010a). These methods combine extensive research and modelling of various environmental issues and damages. At their core, these methods can be distilled to usable conversion factors known as characterisation factors, which enable the conversion of resource inputs and emissions to mass equivalents of a reference substance (at the Midpoint level, e.g. kg CFC11-eq.[1]) or to recognised damage units (at the Endpoint level, e.g. DALY[2]). LCA methods are available to the LCA practitioner in the form of datasets and within LCA software programs.

2.1.1.4 Interpretation

Finally, in the interpretation phase, the results of the LCA are evaluated such that questions posed in the goal definition are answered (European Commission, 2010c). The relative environmental impact information can be used to make purchasing decisions, strategic decisions and can influence product design. Though the phases are typically carried out in the order given, LCA is actually an iterative process (European Commission, 2010c; ISO, 2006a). Based on preliminary results from the interpretation phase, for example, the goal and scope may be reviewed and adapted, which leads to changes in the subsequent LCI and LCIA phases.

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[1] Kilogram equivalents of trichlorofluoromethane (CFC11), a reference substance for impact category, *Ozone depletion potential*

[2] Disability Adjusted Life Years: measure of human life years lost to illness or death, expresses human health damage.

2.1.2 The Carbon Cycle

Carbon is contained within all living organisms, in soil and in fossilised organisms. Cumulatively, carbon-based emissions are part of the global carbon cycle. The carbon cycle consists of carbon reservoirs, as well as the exchanges of carbon fluxes between them (Ciais et al., 2013).

Two main domains have been identified within the carbon cycle, based upon the turnover rates within the reservoirs. The first domain is considered the fast domain, in which a large amount of rapid exchanges take place. These exchanges occur between the atmosphere, oceans, vegetation, soils and freshwater. The second domain is considered the slow domain, in which carbon exchanges occur slowly (in periods of over 10,000 years) between geological formations and the atmosphere. In nature, without human intervention, the fluxes occurring within the slow domain are small. Since the advent of the industrialised era, carbon has been extracted from the geological reservoirs in the form of fossil fuels. The extraction and combustion of these fossil fuels has led to a large transfer of carbon from the slow domain of the carbon cycle to the fast domain, and thus the perturbation of the global carbon cycle (Ciais et al., 2013).

2.1.3 The Greenhouse Gas Effect and Climate Change Metrics

Through mechanisms such as combustion, decomposition and weathering, carbon-containing substances emit greenhouse gases such as carbon dioxide and methane. Greenhouse gases (GHG) are substances which tend to trap heat in the atmosphere, increasing the greenhouse gas effect. The greenhouse gas effect is a mechanism caused by the combination of solar radiation and greenhouse gases. One third of this energy is reflected back towards space, while the two-thirds is absorbed by the surface of the earth and the atmosphere (Cubasch et al., 2013). The absorption of this solar energy is crucial to the life on earth, which would otherwise be entirely frozen.

The accumulation of additional heat has been directly correlated to an increase in global temperatures and to overall change in the earth’s climate. Several authors have proposed different metrics for quantifying climate change impacts. The most commonly used metric used by the Intergovernmental Panel on Climate Change (IPCC) is the global warming potential (GWP). The GWP is defined as the time integrated radiative forcing as a result of a pulse emission of a greenhouse gas relative to a pulse emission of carbon dioxide (Myhre et al., 2013):

$$GWP_i(H) = \frac{\int_0^H RF_i(t)dt}{\int_0^H RF_{CO_2}(t)dt} = \frac{AGWP_i(H)}{AGWP_{CO_2}(H)} \tag{1}$$

Where *H* is the time horizon, *t* is time, *RF* is radiative forcing, *i* is a given greenhouse gas, and *AGWP* is absolute global warming potential.

The GWP indicator is a midpoint indicator. All substances having a global warming potential have been normalised with the impact score of the reference substance, carbon dioxide (Forster et al., 2007).

Another metric that has been proposed is the Global Temperature change Potential (GTP) (Shine et al., 2005). GTP measures the impact of climate change further down in the cause and effect chain. It is defined as the *"change in global mean surface temperature at a chosen point in time in response to an emission pulse relative to that of CO_2 (Myhre et al., 2013)*. Mathematically, GTP can be expressed as:

$$GTP_i(t) = \frac{AGTP_i(t)}{AGTP_{CO_2}(t)} = \frac{\Delta T_i(t)}{\Delta T_{CO_2}(t)} \tag{2}$$

Where *t* is time, *i* is a given greenhouse gas, *AGTP* is absolute global temperature potential and ΔT is the change in temperature.

While GTP and GWP both express results in terms of kg CO_2-equivalents, there are differences between these metric types. GTP yields results that more closely model actual impacts than the radiative forcing approach. In addition, GTP is usually appropriate for use in assessing the impacts at the end of a targeted period of time than other metrics. However, since GTP is based on the response time of the climate to greenhouse gas emissions, the metric can result in more uncertainty than a metric like GWP that is further upstream in the cause and effect chain (Levasseur et al., 2016a; Levasseur et al., 2016b; Myhre et al., 2013).

These metrics can be expressed at different time horizons, allowing for short- and long-term climate change impacts to be quantified. Typically, and historically, the GWP 100-year time horizon has been the default indicator used in quantifying the climate change impacts of greenhouse gas emissions. However, the latest recommendations are to use two indicators in order to cover short-term and long-term climate change. For the short-term climate change effects that target warming rates, GWP100 is recommended as it considers GHGs that decay rapidly. For the long-term climate change effects that target long-term temperature rise, GTP100 is recommended as it considers similar impacts to those of GWP250 or GWP500, for example (Levasseur et al., 2016b).

2.1.4 Biogenic Carbon and Neutrality

The concepts of fast and slow domains within the carbon cycle were introduced in section 2.1.2. Plant matter and organisms currently or recently living, also known as biomass, are considered to be part of the fast domain of the carbon cycle. As a result of the speed at which biomass typically cycles between reservoirs, a distinction has been made for carbon emissions from so-called biogenic sources. The reason for this has to do with the fact that biogenic carbon emissions originate from biomass that has previously but recently sequestered carbon dioxide from the air.

Since the amount sequestered and the amount released are more or less identical, several papers and guidelines on LCA and carbon footprinting have assumed that the net carbon balance and thus the climate change impacts are zero (Johnson, 2009; Rabl et al., 2007; Searchinger et al., 2009). However, several publications have recently shown that this assumption could lead to accounting errors (Garcia & Freire, 2014; Røyne et al., 2016; Searchinger et al., 2009; Vogtländer et al., 2013). Depending on the origin of wood and how it is harvested, forest products can result in emissions, net zero emissions or sequestration (Berndes et al., 2016). Røyne et al. (2016) found that including biogenic carbon accounting could increase the climate change impacts by up to 44% by considering the climate impacts of both biogenic and fossil carbon in end-of-life processes of wood products. In particular, accounting errors occur as a result of the application of the carbon neutrality assumption. First, if the neutrality assumption were applied to managed forests, it would mean that there would be no difference to the carbon footprint whether a tree was left standing or harvested (Johnson, 2009). Second, carbon neutrality can also stem from a consideration of carbon from the forest assessment scale or a given harvest pattern, in which harvest is balanced by regrowth of new trees (Lemprière et al., 2013; McKechnie et al., 2011). Third, the carbon neutrality assumption does not consider the time needed to offset carbon emissions, as it may take years to counteract the carbon that has accumulated in the atmosphere since the release of a greenhouse gas (Cherubini et al., 2011; Helin et al., 2013; Lemprière et al., 2013; McKechnie et al., 2011; Zanchi et al., 2010).

Several authors have developed approaches for improving the accounting of biogenic carbon of bioenergy (Cherubini et al., 2011; Kendall et al., 2009; Repo et al., 2011; Zanchi et al., 2010). Other authors have used a carbon neutrality factor in order to quantify the relative greenhouse gas emissions savings of bioenergy (Schlamadinger et al., 1995; Zanchi et al., 2010). McKechnie et al. (2011) integrated forest carbon analysis and life cycle assessment in order to evaluate greenhouse gas emissions of the use of forest biomass for bioenergy over time. The development of approaches in life cycle assessment for characterising biogenic carbon have been largely in the area of forest biomass used in bioenergy. However, a few studies have included methods for biogenic carbon accounting for wood products (Cherubini et al., 2012; Pingoud et al., 2011).

2.1.5 Emissions Timing

Typically in LCA, emissions are summed over the whole life cycle and characterised in LCIA without considering when in time the emissions took place. However, over twenty years ago researchers began acknowledging that the timing of emissions can have an effect on environmental impacts (Owens, 1997; Reap et al., 2008). Since the turn of the century, several methods have been proposed to address emissions timing (BSI, 2011b, 2011c; European Commission, 2010b; Fearnside, 2002; Guest et al., 2013b; Kendall, 2012; Levasseur et al., 2010; Moura Costa & Wilson, 2000; Vogtländer et al., 2013). However, the climate benefits of regarding emissions timing have been controversial (Brandão & Levasseur, 2011; Brandão et al., 2013). As such, LCA studies do not typically include the impacts of carbon storage or the timing of carbon emissions and uptakes, as shown by Røyne et al. (2016) who found that the majority of LCA studies do not include the timing of emissions. Emissions timing is especially relevant in the case of biomass life cycles, particularly when biomass is used in applications within the anthroposphere with relatively long lifetimes, such as buildings. The biogenic carbon uptake in the biomass and its eventual release as greenhouse gas emissions can take place over relatively long timescales, which can have an impact on the climate change potentials.

In conventional LCAs, emissions and impacts time horizons do not necessarily have to cover the same sequence of years. However, the climate change outcomes may differ depending on when the accounting period begins (Levasseur et al., 2013), the choice of which should be guided by the objective and context of a study (Berndes et al., 2016). As such, at the outset of a life cycle assessment, time horizons for both the product life cycle emissions as well as the impact assessment are chosen. Although the time horizon is often chosen relative to the lifespan of the product considered, this is not always the case. In fact, the choice of a time horizon is considered to be subjective (Fearnside, 2002). With GWP impacts, typically 20, 100 or 500 year time horizons are chosen (Forster et al., 2007), each corresponding to a different value perspective. Jørgensen, and Hauschild (2012) found that both short and long time horizons are necessary for addressing both the acute and long-term climate impacts of temporary carbon storage. However, as shown by Levasseur (2015), the choice of time horizon has an important effect on climate change results.

2.1.5.1 Temporal Approaches at the Life Cycle Impact Assessment Level

Several methods have been proposed over the course of the last fifteen years to address the issue of emissions timing. Brandão et al. (2013) presented a review of the most viable approaches, as a result of an expert workshop that was convened on the topic of carbon sequestration and temporary storage. A brief introduction to the most relevant methods is given below, and summarised in Table 2.1.

Two similar approaches using what is known as the tonne-year baseline, were developed around the same time (in the year 2000). The tonne.year baseline approach uses the radiative forcing (climate impact) of 1 tonne of carbon dioxide released at time zero, over a 100-year time horizon. The integral of this baseline curve is calculated in terms of tonne.years.

Fearnside et al. (2000) proposed a method known as the Lashof method, which attempted to evaluate the effects of delaying emissions following temporary storage. The method makes use of the curve of cumulative atmospheric mass loading of carbon as a result of a pulse emission, as a function of time. The integral of the curve gives the cumulative atmospheric mass loading in tonne.years. When an emission is delayed, this curve is offset by the corresponding number of years that the emission is delayed (see Figure 2.1). The part of the curve extending past the fixed time horizon is no longer considered and corresponds to the benefits related to the storage of that quantity of carbon.

The other method based on the tonne.year approach is known as the Moura-Costa method (Moura Costa & Wilson, 2000). The Moura-Costa method translates the tonne-years into years of carbon storage, and further allocates this stored carbon credit per year. Thus, despite having similar baselines, the Lashof and Moura-Costa approaches differ substantially in terms of the amount of emission benefit attained. They are also both very dependent upon the time horizon chosen.

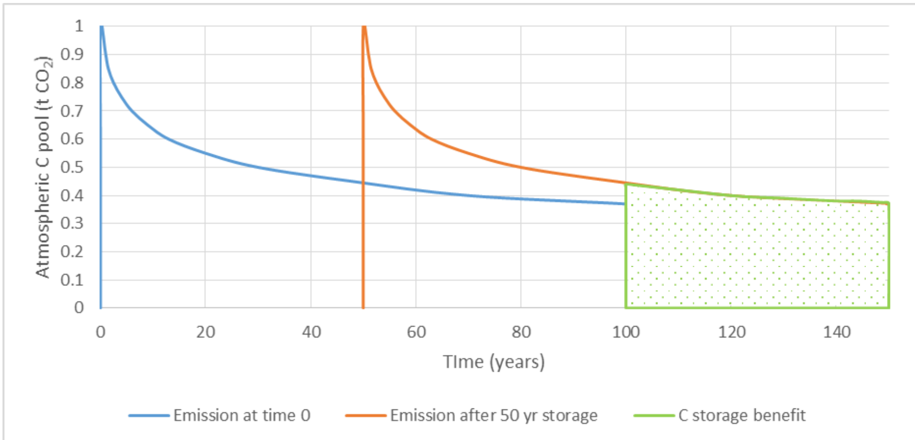

Figure 2.1 – Lashof approach example. Storage of 1 tonne of CO_2 for a period of 50 years for a 100-year time horizon. The blue line indicates an emission at time 0, while the red line indicates an emission following a storage period. The green shape represents the carbon benefit achieved from the storage period, beyond the 100-year time horizon (source: based on PAS2050 figures)

The approach uses weighting factors, which are related to the timing of the emissions (BSI, 2008). If the emissions occurred within 2-25 years of the product manufacture, the weighting factor was calculated using a linear approximation of the Lashof method, characterised by:

$$Weighting\ factor = \frac{0.76 \times t_0}{100} \tag{3}$$

t_0 = number of years the full carbon storage benefit of a product exists after the products is created.

Whereas, the weighting factor for emissions occurring beyond 25 years is calculated as:

$$Weighting\ factor = \frac{\sum_{i=1}^{100} x_i}{100} \tag{4}$$

i = each year of carbon storage, x = proportion of total storage remaining in any year i

The British Standards Institute (BSI) published an update to the standard in 2011 (BSI, 2011a), which now states that the emissions timing is optional.

The European Union's Joint Research Council published a series of guidelines for life cycle assessment in 2010, known collectively as the ILCD handbook (European Commission, 2010a). The ILCD handbook method follows the same emissions delay approach used in the Lashof method, where emissions occurring after the defined time horizon are subtracted from the amount of carbon originally emitted. Unlike the Lashof method, the ILCD handbook method uses a linear approximation, which is carried by multiplying avoided emissions by the GHG 100-year characterisation factor, the number of years delayed (years that carbon is stored) and the factor -0.01 kg CO_2-eq/year.kg CO_2-eq. Any emissions occurring after 100 years are not included in the LCA results but are documented separately.

One of the more recent approaches developed to account for emissions timing has been the Dynamic LCA method (Levasseur et al., 2010). This method, which is based on the IPCC's absolute global warming potential (AGWP) equation integrated continuously through a fixed time horizon, allows for the calculation of a radiative forcing impact at any point in time. Dynamic characterisation factors (DCF) are used in combination with a temporally differentiated emissions inventory to calculate the instantaneous global warming impact, GWI_{inst},

$$GWI_{inst}(t) = \sum_{i=k}^{t}\left[kgCO_2(k) \cdot DCF_{CO_2}(t-k)\right] + \sum_{i=k}^{t}\left[kgCH_4(k) \cdot DCF_{CH_4}(t-k)\right] + \sum_{i=k}^{t}\left[kgN_2O(k) \cdot DCF_{N_2O}(t-k)\right] \quad (5)$$

Each summed expression represents the radiative forcing of all pulse emissions of a given GHG in a given year. The pulse emissions (in kg GHG) in year k are multiplied by GHG-specific characterisation factors for every time step (t – k).

The cumulative radiative forcing (GWI_{cum}) of all GHG pulse emissions is calculated for all years from 0 – t and is represented by,

$$GWI_{cum}(t) = \sum_{k=0}^{t} GWI_{inst}(t) \quad (6)$$

Cumulative forcing results are translated into DLCIA (dynamic life cycle impact assessment) scores in terms of kgCO_2-eq., by dividing the GWI_{cum} by the cumulative radiative of 1 kg of CO_2 emitted at time zero to the given time horizon TH,

$$DLCA = \frac{GWI_{cum}(TH)}{\int_0^{TH} RE_{CO_2} \cdot C(t)_{CO_2} dt} \quad (7)$$

The denominator of the DLCA equation represents the cumulative radiative forcing of a single 1 kg CO_2 pulse emission. TH represents the time horizon chosen for the study. In essence, the equation results in the global warming impacts at every point of time where emissions occur, for every GHG, over the course of the life cycle of the product in question. An Excel-based tool called $DYNCO_2$ (Levasseur, 2013) was developed to provide a calculation platform for Dynamic LCA.

Kendall (2012) developed an approach known as the Time-Adjusted Warming Potential (TAWP), which is based on the IPCC GWP model. This method is identical to the Dynamic LCA method, though instead of allowing flexibility in time horizons, the method fixes five specific time horizons (20, 30, 50, 100, 500 years).

GWP_{bio} is a concept that combines the Bern carbon cycle model and a Gaussian growth curve in order to quantify the climate impact of biomass energy emissions, which was originally developed by Cherubini et al. (2011) to calculate characterisation factors for bioenergy. The approach is dependent on the number of years required to regrow the biomass and uses three fixed time horizons: 20, 100 and 500 years. Guest et al. (2013b) further adapted this method to include a product lifetime for the use phase of the biomass being used as bioenergy. The model oversimplifies a few aspects about the forestry phase, including species, growing region and type of forest management. In addition, the model is based on the end-of-life combustion of the biomass, which excludes any other end-of-life waste management outcomes, such as recycling or landfilling.

One of the most recent methods of approaching emissions timing and temporal incongruity in LCAs was proposed by Yuan et al. (2015). The method was developed in order to include several perspectives not necessarily taken into consideration in existing methods and consists of a procedure, involving: 1) calculating the temporal scale of LCA, 2) compiling the temporally differentiated life cycle emissions, 3) modelling the actual environmental fate of emissions, 4) discounting emissions to selected reference time point, 5) aggregating discounted emissions at the reference time point.

In addition to these general timing approaches, other authors have proposed methods for accounting for the delay of a specific greenhouse gases, such as Sevenster (2014) who developed a linear approximation of the dynamic LCA method for methane. Despite the practicality of such an approach for specific activities, such as the release of methane emissions from landfill, approaches with broader range of applications are more useful for modelling a variety of life cycle scenarios.

Of the approaches presented, the Lashof and Moura-Costa approaches are simple, focused on temporary storage but are not necessarily appropriate for consistent application to life cycle assessment. The ILCD and PAS2050 approaches are also quite user-friendly and straightforward, however due to their reliance on a 100-year time horizon, the fact that they ignore sequestration dynamics and that they do not provide GHG-specific emission delay credits, they make poor choices for in-depth application to the life cycle of wood products. Despite purporting to address temporal issues in LCA, Yuan et al. (2015) approaches time using a discounting methodology but does not actually provide specific means for addressing time horizons or emission delays. Finally, the DLCA and TAWP approaches both consider sequestration dynamics, are GHG-specific, and offer the choice of time horizon (DLCA: any, TAWP: choice of 5), which makes these methods applicable to a large range of cases and complexity levels.

The climate change outcomes may differ depending on when the accounting period begins (Levasseur et al., 2013), the choice of which should be guided by the objective and context of a study (Berndes et al., 2016). In testing four different approaches that consider emissions timing and temporary carbon storage, Røyne et al. (2016) found that the approaches had climate benefits ranging from 8% (for the GWP_{bio} approach - Guest et al. (2013b)) to 70% (for a discounting approach using a 1% discount rate) of the life cycle impacts of a wood construction. The ILCD approach (European Commission, 2010b) and the dynamic life cycle approach (Levasseur et al., 2010) have climate benefits somewhere in between these two extremes (31-38%). The emissions timing methods described above are summarised in Table 2.1, in terms of their calculation approach, time horizon and advantages and disadvantages.

Table 2.1 – Summary of emissions timing methods. CAML = cumulative atmospheric mass loading, TH = time horizon

Method	**Calculation approach**	**Time Horizon**	**Advantages**	**Disadvantages**	**Main reference**
Lashof	-Uses curve of CAML of carbon as a result of a pulse emission, as a function of time -Integral of CAML = tonne.years -Emissions delay = offset curve -Uses temporal cut-off	Any	Simple	Choice of TH crucial has large impact on results and conclusions	Fearnside et al. (2000)
Moura-Costa	-Uses equivalence factor yearly crediting (tonne.year) -Based on CAML value of single CO_2 emission -Uses temporal cut-off	Any	Simple	May not be suitable for LCAs	Moura Costa, and Wilson (2000)
ILCD	-Linear approximation of Lashof method, subtracting the emission that takes place beyond the time period -Uses temporal cut-off	100 years	Simple Linear	-Does not consider sequestration dynamics -Credits for delayed emissions same for all GHG emissions.	European Commission (2010c)
PAS2050	-Linear approach based on Lashof method for two different times: 2-25 years and 25+ years after product manufacture -Uses temporal cut-off	100 years	Simple Linear	-Does not consider sequestration dynamics -Credits for delayed emissions same for all GHG emissions.	BSI (2008, 2011c)
Dynamic LCA (DLCA)	Calculation of global warming impacts at every point of time where emissions occur, for every GHG, over the course of the life cycle of the product in question.	Any	-Considers sequestration dynamics -GHG-specific approach -TH consistent (emissions & impacts) -Choice of any TH	Choice of TH and temporally differentiated emissions can make choice for decision-maker more difficult (Dyckhoff & Kasah, 2014; Levasseur et al., 2013)	Levasseur et al. (2010)
TAWP	Calculation of global warming impacts at every point of time where emissions occur, for every GHG, over the course of the life cycle of the product in question. -Characterisation factors calculated for each emission year	20, 30, 50, 100, 500 years	-Considers sequestration dynamics -GHG-specific approach -Limits choice of TH	-Only considers 5 TH – could limit outcomes	Kendall (2012)
Yuan et al. (2015)	Proposed a five-step method, which is distinguished by the use of discounting	N/A	Addresses temporal differentiation of emissions	-No specific method for dealing with TH and emission delays -Requires high level detail on timing of emissions	Yuan et al. (2015)

2.1.5.2 Temporal Differentiation of the Life Cycle Inventory

In order to practically apply most of the temporal approaches to the life cycles of products, the specific timing of each individual emission must be determined. Several authors have explored the *temporal differentiation* of life cycle inventories (LCI) in order to be able to apply temporal methods (Beloin-Saint-Pierre et al., 2014; Collet et al., 2013; Collinge et al., 2013; Pinsonnault et al., 2014;

Yuan et al., 2015). Collet et al. (2013) developed a method for determining whether considering dynamic life cycle inventories has an overall impact on LCIA results, and whether they are worth applying. Beloin-Saint-Pierre et al. (2014) proposed the enhanced structural path analysis (ESPA) method in order to apply temporal differentiation on a mass scale (database-level) to elementary and process flows. Within the context of global warming impacts, Pinsonnault et al. (2014) sought to determine the sensitivity of adding temporal information to LCIA results by using the ESPA method (Beloin-Saint-Pierre et al., 2014). They generated temporally differentiated LCIs for every product in the ecoinvent 2.2 database and found that temporal information can be particularly relevant for products containing biomass. In their temporal LCIA method, Yuan et al. (2015) also described steps in order to temporally differentiate at the LCI level. More recently, Tiruta-Barna et al. (2016) and Cardellini et al. (2018) developed methods that generate temporally differentiated LCIs using software programs.

2.2 Carbon Accounting for Wood used in Building and Construction

2.2.1 Whole Wood Life Cycle Considerations

LCA practitioners and industry instead tend to rely on existing perspectives and approaches that have reached a certain consensus within the LCA community, such as product category rules (PCR) and normative standards. Table 2.2 gives an overview of the PCRs and normative standards covering wood and buildings as well as their treatment of both biogenic carbon and emissions timing.

Table 2.2 – Product category rules (PCR) and normative standards covering wood and buildings

Region	PCR/ Standard	Biogenic CO_2	Allocation	Timing	LUC
North America	North American Structural and Architectural Wood Products (FPInnovations, 2015);	Inputs as -1 and outputs as 1, characterised as fossil C = net zero, carbon storage can be provided separately	Multi-output processes: Δ$ coproducts: low: mass, volume high (>10%): economic Allocate by physical flows for biogenic carbon, water, energy	Presented as separate GWP indicator with reference methodology	LUC reported separately
	North American Pressure-treated Wood Products (ASTM/ICC-ES, 2016);		Physical relationship between co-products: by mass No relationship: equal division among co-products	Not stated	Not mentioned
United Kingdom	PCR for Construction Products EPD (BRE, 2014).	Inputs as -1 and outputs as 1, characterised as fossil C = net zero, no carbon storage	Has its own allocation preference hierarchy (decision tree)	Not stated	Included as per IPCC guidelines for national GHG inventories
	PAS 2050 (BSI, 2011c);	Inputs as -1 and outputs as 1, carbon storage can be provided separately	Avoid allocation, otherwise allocate by physical property (mass, etc.), as a last resort by economic or other relationship.	Reported separately using method provided	GHGs allocated to goods/services for 20 years after the land use change
Norway	Wood and wood-based products for use in construction (The Norwegian EPD Foundation, 2013b);	Inputs as -1 and outputs as 1, carbon storage can be provided separately	Allocation by volume or mass if subdivided or little value difference. If not subdivided and large value differences, economic allocation shall be used.	Can be calculated by recognised method, but reported separately	None mentioned
	Building boards (The Norwegian EPD Foundation, 2013a);		Allocation according to mass or volume. At large value differences (>20%), economic allocation shall be used.		LUC reported separately
France	AFNOR BP X30-323 (ADEME & AFNOR, 2011).	Inputs as -1 and outputs as 1, carbon storage can be provided separately	Avoid allocation, otherwise allocate by physical property (mass, etc.), as a last resort by economic or other relationship.	Time-weighted average for storage and delay 0-100 years.	Included as per IPCC guidelines for national GHG inventories
Germany	Product Category Rules for Building-Related Products and Services (Institut Bauen und Umwelt, 2014)	Not stated	Δ$ coproducts: low: mass, volume otherwise economic allocation Allocate by physical flows for biogenic carbon, water, energy	Not stated	None mentioned
Austria	Solid wood products (Bau-EPD, 2017);	Add to total emissions		Not stated	Optional

Table 2.3 – Product category rules (PCR) and normative standards covering wood and buildings (continued and end)

Region	**PCR/ Standard**	**Biogenic CO_2**	**Allocation**	**Timing**	**LUC**
Europe	Round and sawn timber (CEN 16485, 2014);	Inputs as -1 and outputs as 1, storage can be provided separately	Allocate by biogenic carbon regardless of allocation procedure used for other flows in LCA	Can use recognised method, but reported separately	LUC reported separately
	ILCD Handbook (European Commission, 2010b);	Inputs as -1 and outputs as 1, storage can be provided separately	Avoid allocation, otherwise allocate by physical property (mass, etc.), as a last resort by economic or other relationship.	Excluded from most studies, unless study goals require it. Use PAS2050 procedure	Included as per IPCC guidelines for national GHG inventories
	Product Environmental Footprint (PEF) Guide (European Commission, 2012a);	Inputs as -1 and outputs as 1, but report separately; storage can be provided separately		Not considered	GHGs allocated to goods/services for 20 years after the land use change
International	Buildings (EPD International, 2018a);	Biogenic emissions reported separately from fossil, Storage report separately	Allocate by biogenic carbon regardless of allocation procedure used for other flows in LCA	Not stated	LUC reported separately
	Construction products and services (EPD International, 2017);			Not stated	LUC reported separately
	Products of wood, cork, straw and plaiting materials (EPD International, 2018b);			Not stated	LUC reported separately
	GHG Protocol (WRI/WBCSD, 2011);	Inputs as -1 and outputs as 1, but report separately, carbon storage not addressed	Avoid allocation, otherwise allocate by physical property (mass, etc.), as a last resort by economic or other relationship.	Not included in results but can be reported separately	LUC reported separately
	ISO/TS 14067 (2013);	Inputs as -1 and outputs as 1, but report separately, storage can be provided separately		Not included in results but can be reported separately	LUC reported separately, calculated as per IPCC guidelines for national GHG inventories

As shown in Table 2.2, the current PCRs and normative standards have a range of approaches for both the treatment of biogenic carbon and emissions timing. This range of perspectives can result in large differences between climate change impacts. However, despite these differences most approaches recommend accounting for carbon uptakes as -1 kg CO_2-eq and carbon emissions as 1 kg CO_2-eq, thus considering biogenic CO_2 emissions the same as fossil CO_2 emissions. In the context of Canadian wood products, guidance from the North American Structural and Architectural Wood Products PCR is recommended due to its relevance to North America.

The EN 15804 Product Category Rules developed for construction products (European Commission, 2012b), defines a set system boundary of which elements of the life cycle and which processes should be attributed to the product system for construction LCAs. As such, it identifies modular A-D life cycle phases to be considered in construction product LCAs. The modules cover different stages of the life cycle and can be summarised as follows: product stage: A1-A3, construction process stage: A4-A5, use stage: B1-B5, end-of-life stage: C1-C4, future, reuse, recycling or energy recovery potentials: D. Most life cycle assessment studies of building products and buildings have been conducted such to position the relative importance of the main life cycle phases of a building, typically the construction materials, the energy consumption during the use phase and the end-of-life phase of the building (Buyle et al., 2013). In building comparisons, some studies assume that operational energy consumption is equivalent and can be excluded from the building LCA. However, in their review of methodological practices for building LCAs, Miller et al. (2016) concluded that unless the thermal properties and building envelope permeability of the buildings being compared are exactly the same, the operation energy should be included. However, within the LCA, this does not preclude the practitioner from focusing analyses on the building material impacts.

The system boundaries of wood products involve several different processes throughout the product life cycle. At the forestry management phase, this involves the growth and planting of seedlings, the application of fertilisers, the use of harvest machinery, forest road construction and maintenance, the management of harvest residues and the transport of logs to the sawmill. In assessing the climate change impacts of wood construction materials, many studies have not considered biogenic carbon accounting due to the carbon neutrality argument (Puettmann & Wilson, 2005; Takano et al., 2014; Zabalza Bribián et al., 2011). However, biogenic carbon may play a role on climate change impacts throughout the life cycle of a wood product used in construction, specifically at the points of carbon exchange with the atmosphere, such as sequestration, decomposition or segregation from the carbon cycle such as during long-term carbon storage (see Figure 2.2).

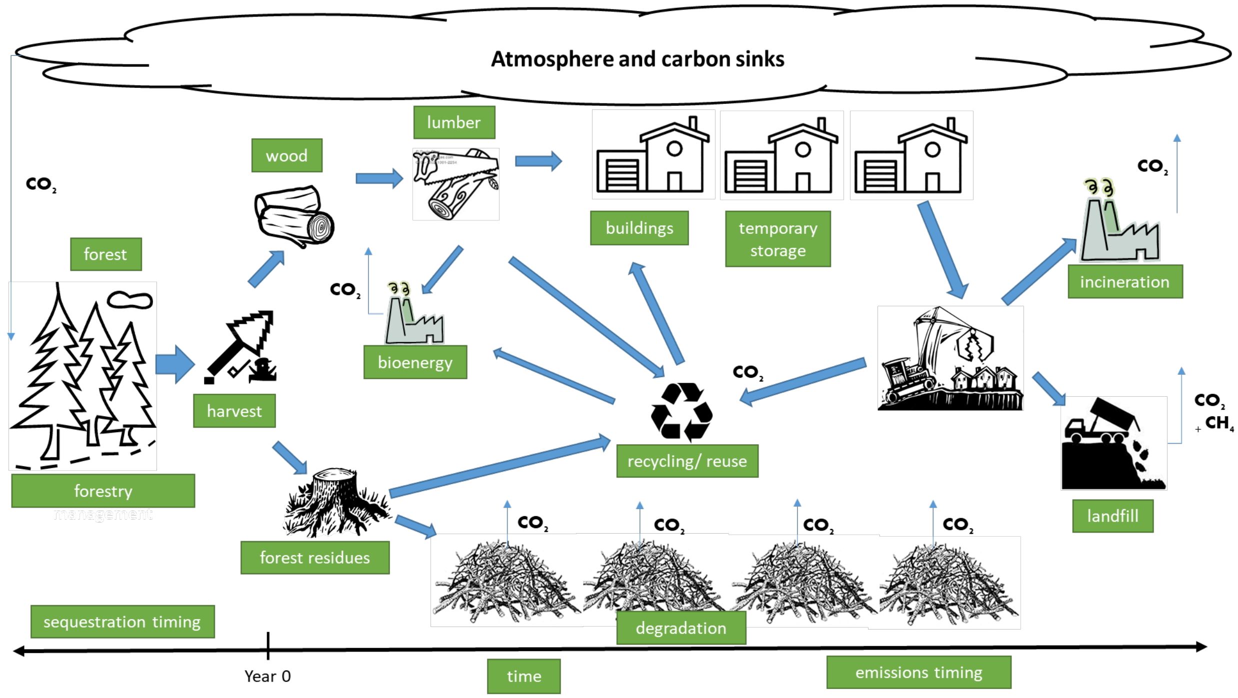

Figure 2.2 – Overview of the life cycle of wood used in a building, from a biogenic C perspective (thicker blue arrows indicate flow of biomass between stages, thinner blue arrows with "CO_2" or "CH_4" indicate absorption or emission of those GHGs

As exemplified by Smyth et al. (2014), the emissions of forestry products (E) can be expressed as:

$$E = F + P + D \tag{5}$$

Where, *F* is the net emissions from the forest, *P* is the emissions from harvested wood products (HWP) during or at the end of their lives and *D* is displaced emissions from substitution of other materials.

However, to apply this relationship from the perspective of calculating the absolute life cycle impacts of wood used in buildings it can adapted as:

$$E(t) = F + U + EoL$$

to include, forestry carbon dynamics (F), carbon storage during building use (U), end-of-life treatment (EoL), emissions timing (t) and to exclude displaced emissions from material substitution.

The overall life cycle emissions of wood used in construction is dependent upon forestry conditions and management, the storage of wood in buildings and the specific end-of-life options, all related to the timing of those emissions. These aspects will be elaborated upon further in this section.

2.2.2 Forestry Phase

2.2.2.1 Forests and Carbon Balance

An improved understanding of the forest carbon balance is key to assess the climate impacts of wood production. Forests can either act as net carbon *sinks* or net carbon *sources*. If trees are healthy and forests are managed correctly, forests can be carbon sinks meaning that they absorb more carbon dioxide than they release. If forests are burned or degraded, they can also act as carbon *sources*, meaning that they have a net carbon release (NRC, 2015). Although most forests tend to be net carbon sinks, this source-sink interaction adds significant complexity to the forest carbon balance. Several papers have examined this source and sink interaction on a macro scale, determining whether or not global forests are net sequesterers of carbon (Kindermann et al., 2008; Kirschbaum, 2003a, 2003b; Pan et al., 2011). In particular, a study by Pan et al. (2011) found that the global forests were a carbon sink, estimated at 1.1 ± 0.8 Pg carbon per year – roughly the same amount of carbon emitted as carbon dioxide by the entire European Union in 2013. Historically, Canadian forests have been carbon sinks, though recent natural disturbances (notably large wildfires and insect epidemics) have affected this especially since 2000 (Natural Resources Canada, 2018; Stinson et al., 2011). As such, it important to consider natural disturbances in forestry carbon dynamics of Canadian forests.

2.2.2.2 Parameters Affecting Forestry Carbon

The carbon dynamics of forests become increasingly complex when forestry activities are considered. These dynamics are crucial to the understanding of the climate impacts of wood products. The net carbon balance of forest products depend on several factors: the quantity, form and timing of emissions (Hall, 2011); whether land use change emissions have been accounted for (Lamers & Junginger, 2013); plant growth factors including the forest biome, species, productivity, forest management (Cowie et al., 2013; Lamers & Junginger, 2013); and the type of biomass used such as roundwood logs, residual biomass, dead wood and trees grown on highly productive or marginal land (Lamers & Junginger, 2013). The scale of assessment also affects the carbon balance, with a stand-level perspective considering the carbon dynamics of a small land parcel with trees of the same age class, whereas a landscape or national level looks at a larger region that encompasses several forest stands (Cowie et al., 2013). Peñaloza et al. (2018) found that the choice of system boundary in the forest can have a large influence on the carbon emission assessment, as with a stand approach carbon uptake occurs throughout the rotation period whereas with landscapes approaches it occurs within a single (year 0). In addition, the choice of the reference scenario is important as it can have a large influence on the carbon outcomes and it should consider what would have happened if the human intervention had not taken place (Berndes et al., 2016; Lamers & Junginger, 2013; Peñaloza et al., 2018). Two types of baselines exist for benchmarking changes in forest carbon stocks: a reference point in the past or future against which carbon stocks can be compared and an anticipated baseline which allows for parsing out of natural disturbances and processes (US EPA, 2010). An anticipated baseline is used for estimating carbon emissions and uptakes for specific forest products, thus comparing a natural vs. a human intervention state.

These and other parameters that have shown to have an effect on forestry carbon, are identified and categorised in Table 2.3, using Lorenz, and Lal (2010) as a basis. For each parameter, a description and an indication of the possible range of variability in the parameters is given. In addition to having specific relationships to forestry carbon and resulting greenhouse gas emissions, these parameters will also have interactions with some of the other parameters.

Given the number of parameters and studies covering the interactions of these parameters with carbon dynamics, efforts have been made to develop databases to distill this information. In Canada, several database initiatives have been established, including the development of the National Forestry Database (Canadian Forest Service, 2013) and the Forest Ecosystem Carbon Database (Shaw et al., 2005). The Forest Ecosystem Carbon Database, in particular, which was developed for large-scale carbon modellers and analysts, contains data for over 700 forestry plots throughout Canada for 60 different variables. Such forestry carbon databases allow for the modelling of virtually all forestry situations across Canada.

Table 2.4 – Overview of forestry parameters, with range of possibilities *(parameters: Lorenz, and Lal (2010), categories: author)*

Parameter	Description	Possible range of options
Baseline Characteristics		
Natural forest or plantation	Forestry can either be on natural forest or plantations	Natural, semi-natural, plantation (Lorenz & Lal, 2010)
Tree species	The species of trees and composition	Approximately 20 tree species within Canadian boreal forest (Osborne & O'Reilly, 2015)
Region / climate	The region and specific climate of the forestry region.	Fifteen terrestrial ecozones across Canada (Kull et al., 2014)
External Disturbances & Interactions		
Fire damage	Fires can damage aboveground biomass and sometime other pools	Fires can be stand-replacing (large) or non-stand replacing (smaller) (Binkley et al., 1997)
Insects, disease, herbivore destruction	Forests can be ravaged by insect infestation, disease and herbivores.	Various insect epidemics affect Canadian forests, ranging from beetles, to moths to caterpillars (Stinson et al., 2011). Diseases and pathogens affecting forests include various bark, leaf and root syndromes. Herbivores such as deer, elk, moose and hare may also affect vegetation (Ayres & Lombardero, 2000).
Climate change effects	In the boreal forest, climate change may affect species composition, increase fire and insect disturbances and generally affect forest health (Bonan, 2008)	The range of climate change effects is unknown, but could cause more frequent forest fires (Kurz et al., 2008b), more frequent insect epidemics (Kurz et al., 2008a), cause changes in the species type and growth (Taylor et al., 2017)
Land use and land use change	Initial harvest of virgin forest causes loss of carbon Land use has interactions with albedo (Schwaiger & Bird, 2010)	Afforestation, reforestation and land occupation for managed forests can be examined (Michelsen et al., 2012).
Albedo effect	Land cleared in areas with seasonal snowfall has increased sun reflection from the snow surface which increases albedo.	There are complex interactions between forest and albedo (Bright et al., 2015). Albedo can be temporary or more permanent, depending on LULUC (Bright et al., 2012)
Silviculture Management		
Rotation period	Time a forest is allowed to grow before harvest	Rotation periods can vary as a function of species (Luyssaert et al., 2007) and management practice (Cherubini et al., 2011; Kula & Gunalay, 2012; Nunery & Keeton, 2010)
Forestry density	The number of trees or biomass for a given area can vary	Forest density varies depending on species and management practices (Kauppi et al., 2010)
Thinning	Select trees and branches are cut within a forest area	Thinning can vary in intensity (amount biomass removed) and in frequency (Law et al., 2013)
Use of fertilisers	Fertilisers can be applied to forestry	Application of fertilisers have been associated with increased biomass growth (Binkley et al., 1997). Some forestry practices apply fertilisers, others do not.
Site preparation	Means of preparing land for planting	Can involve manual, mechanical or chemical interventions (Lorenz & Lal, 2010)
Harvest techniques	The types of harvest can vary, from selected harvest to clear-cutting	Range from high impact to low impact and options in between (Harmon et al., 2009; Lorenz & Lal, 2010). Spatial arrangement may impact tree species composition (Harmon et al., 2009)
Treatment of residues	Biomass that remains after harvest	Residues are either left to decompose or removed for use (Achat et al., 2015)
Measurement Context		
Assessment scales	Forests can be examined from the standpoint of different sized systems, termed assessment scales	Assessment scales can range from regional, to supply area, a single landowner, multiple plots and single plots (Galik & Abt, 2012)
Forest carbon pools	Include aboveground biomass, roots, litter, dead trees, soil.	Studies typically include aboveground biomass, but can include any other of the pools (Lorenz & Lal, 2010)

2.2.2.3 High-level Forestry Carbon Research Methods

High-level forestry carbon research methods encompass stand alone methods that have been developed using mathematical relationships incorporating climate change modelling, global carbon models or forestry growth models. These models are usually accompanied by characterisation factors or model inputs so that the user can easily use the model. Often the research methods have a more macro or top-down perspective, and as such they tend not to consider details such as species, growing region, natural disturbances and other forestry details.

Tellnes et al. (2017) reviewed a series of emerging biogenic carbon accounting methods, including a flexible parametric model for forests (De Rosa et al., 2017), characterisation factors for biogenic CO_2 emissions with atmospheric decay known as GWP_{bio} (Cherubini et al., 2011; Guest et al., 2013b) and an approach based on the global carbon cycle (Vogtländer et al., 2013). De Rosa et al. (2017) proposed a simplified parametric stand-level model that can be applied to forest systems around the world and allows for a dynamic life cycle inventory of carbon fluxes. While this approach is flexible and versatile, it may result in a high degree of uncertainty due to the variation in tree species, climatic conditions and forest management practices possible in the world's forests. Nevertheless, this model has been identified by other authors as a simple model that is usable in LCA (Peñaloza et al., 2018). Laganière et al. (2017) developed a greenhouse gas bioenergy model from a landscape perspective that calculates the net difference between three different wood biomass feedstocks (forest residues, salvaged trees and green trees) versus fossil fuel substitution. Vogtländer et al. (2013) developed a method for considering the carbon storage of harvested wood products (HWP) in which the temporary storage of carbon in buildings or long-life products is only considered to have a net benefit on emissions if the building replacement rate is exceeded.

Tellnes et al. (2017)'s review mentioned above also specifically analysed the methods on twelve data input requirements. While the De Rosa et al. (2017) is widely applicable and flexible, it requires several data inputs, including wood species, rotation period, biomass annual increment, wood density, carbon content, aboveground/belowground ratio, biomass conversion and expansion factor, share of aboveground and belowground slashes and percent wood debris harvested. Granted, many of these data are easy to obtain and in addition, for the last four requirements, reference values are made available by the authors. Vogtländer et al. (2013)'s method requires the biomass annual increment, the biogenic carbon emissions per year over complete life cycle, the carbon content of wood, the ratio of aboveground to belowground biomass, and the share of aboveground and belowground slashes. For the GWP_{bio} method (Guest et al., 2013b), Tellnes et al. (2017) found that relatively few data inputs are required. However, despite adding a timing element to carbon emissions, GWP_{bio} still supports a carbon neutrality approach. This does not allow for forest carbon dynamics to be modelled beyond forestry rotation periods. For the context of calculating the forestry carbon dynamics of a wide range of species and regions across Canada where the specific forest stand location is unknown, these approaches are not optimum. This context is based modelled using approaches as described in the forestry carbon accounting models.

2.2.2.4 Forestry Carbon Accounting Models

Within the context of forest science and forest management, several forest carbon models have been developed to assess the dynamics of forest carbon. Forest carbon accounting models consider a whole group of software models, calculators, spreadsheets and other tools that allow for specific and user-defined forests calculation, considering the interaction of carbon flows between pools as a result of growth and decay. These models tend to require more specific modelling parameters than the high-level research methods presented above. A large degree of variation exists both in how forest carbon calculators consider ecological, forest management and life cycle processes, as well as the ease of use and input data requirements and the types of outputs available. A review of 12 calculators applicable to North American or global forests by Zald et al. (2016), divided the calculators into three classes: low, intermediate and high system complexity. As another cross-section, 6 groups were identified depending on the specific perspective, objectives and types of calculations possible by the models. Table 2.4 gives an overview of this classification as well as a summary of key characteristics and applications of the models and calculators.

Table 2.5 – Classification of forest carbon models and calculators, adapted from Zald et al. (2016), L = low (red), I = intermediary (yellow), H = high (green)

Tools	**System class**	**Group**	**Types of forest C pools**	**Scale of analysis**	**Data type**	**Management and Disturbances**	**Post-harvest emissions**	**Applications**
CCTv4.0	L	1A	Many regions and forest types	Large areas	Little user data required	None	None	Estimate regional or national C pools
COLEv3.0	L	1A	Many regions and forest types	Large areas	Little user data required	None	n/a	Estimate regional or national C pools
AFOLU-CC	L	1B	Many regions and forest types	Large areas	Little user data required	None	None	Calculate C in international forest development projects
FICAT	L	1B	Many regions and forest types	Large areas	Little user data required	Management, no natural disturbance	Included	Calculate C in international industrial forest sector projects
FORPLAN	I	2	Oregon, Washington	Stand-level	Web GIS user interface	Management, no natural disturbance	None	Stand-level C and financial assessments of management activities
THPGGEC	I	3	California	Stand-level	User data required		Included	Forest sector C assessments of stand-level management
GTR-NE-343	I	4	U.S. (except Alaska)	Stand-level	Unlinked lookup tables	Simulate management not natural disturbance	Included	National forest sector C assessments of stand-level management
FORGATE	I	4	Maine	Stand-level	Linked lookup tables	Limited management, no natural disturbances	Included	Regional forest sector C assessments of stand-level management
LMS	H	5	U.S.	Stand-level	User data	Wide range of management, no natural disturbance	Included	Decision support tool
FSCC	H	5	Oregon Cascades	Stand, landscape forest sector	User inventory data not required	Flexible user defined management and natural disturbances	Included	Decision and discussion tool to assess how management activities, natural disturbance effect forest sector C dynamics
CBM-CFS3	H	6	Global with focus on Canada	Stand, landscape forest sector	User inventory data required	Flexible user defined management and natural disturbances	Included	Decision-support tool to assess how management activities, natural disturbance effect forest sector C dynamics
CR-FVS	H	6	U.S., excluding interior Alaska	Stand, landscape forest sector	User inventory data required	Flexible user defined management and natural disturbances	Included	Decision-support tool to assess how management activities, natural disturbance effect forest sector C dynamics

Of the twelve calculators, two score the highest in the review classification: CBM-CFS3 and CR-FVS. Although both tools have very similar attributes, CBM-CFS3 is specialised in the calculation of carbon dynamics in Canadian forests, while the CR-FVS is focused on the United States. The CBM-CFS3 software was developed by the Canadian Forest Service and the Canadian Model Forest Network, to meet the forestry industry's need for an operational-scale carbon accounting tool and to provide country-scale forestry carbon accounting (Canadian Forest Service, 2015). As such, CBM-CFS3 is most suitable to forest carbon dynamics calculations in the Canadian context.

2.2.3 Manufacturing and Use Phase

Once wood has been harvested it will be manufactured into usable building materials, referred to as harvested wood products (HWP) and used in the construction of a building. The wood will remain in the structure of the building throughout the lifespan of the building or until it is replaced through renovation, at which time demolition waste will be sent for disposal. During the period that the wood is in the building, the embodied carbon in the wood is stored and thus prevented from being released into the atmosphere as a greenhouse gas. In fact, carbon can remain stored in harvested wood products for long periods of time, such as in the case of wood buildings, and are dependent upon the specific use and life of those products (Lemprière et al., 2013).

From an energy perspective, the use phase represents an important stage in the life cycle of the building (Buyle et al., 2013). Throughout a typical lifetime of 50 to 100 years, a building uses energy, mainly in the form of electricity and fossil fuel combustion, to provide heating, cooling and electrical power to the building occupants. The use of fossil energy carriers results in the continual emission of fossil greenhouse gas pulse emissions, which contribute to the overall climate change impacts of a building's life cycle. However, due to the focus on wood construction materials, the use phase is not considered here.

2.2.3.1 Multifunctionality in Wood Product Production

Several processes within wood product life cycles are multifunctional, and thus as outlined in section 2.1.1.2.1, this needs to be solved in order to calculate the environmental impacts of a process for a given coproduct. A recent study by Sandin et al. (2015), showed that while allocation methods can influence results by several orders of magnitude, the type of allocation used is of greater importance if the primary product (focus on a given LCA study) is not dominant in terms of mass. De Rosa et al. (2018) considered a system expansion approach in their LCA of structural timber, finding that forestry products can be highly sensitive to the methodological choices used for multifunctional systems. In a study on the influence of end-of-life modelling of construction materials, Sandin et al. (2014) looked at a consequential approach where substitution/system boundary expansion is applied as well as an attribution approach where a cut-off is applied. Results suggest approach used for solving multifunctionality can have a significant impact in terms the absolute LCA scores obtained in comparisons of construction materials. Other studies have examined wood products from a systems perspective, looking at the climate change effects of shifting between wood end uses such as bioenergy and wood products (Cintas et al., 2017; Gustavsson et al., 2017; Smyth et al., 2014). For solving multifunctionality these studies have all opted for a substitution approach, given that the goals of these studies are to compare wood uses within a larger forestry system. However, the PCRs recommend that biogenic carbon should allocated by physical flows for individual wood products.

2.2.3.2 Wood Product Models

Several institutions and authors have also developed models that simulate the carbon balance of wood production, use and end of life of wood products. Brunet-Navarro et al. (2016) conducted a comparative review of 41 wood product models. They evaluated the models using characteristics of model content and model use and classified them into groups A, B or C according to their scores on complexity and user support. Group A models are likely the best options as they scored high both for the complexity of the model structure and user-friendliness, whereas groups B and C have little or no user support (B covers simple models, and C complex ones). Brunet-Navarro et al. (2016) found that for climate change mitigation calculations that the more complex models are needed. Nine models were classified as group A, ranging from meeting 4 to all 7 of the model content characteristics. Using a process of elimination, we eliminate all the models that do not include recycling or substitution effects and that exclude more than one characteristic. The remaining models are compared in Table 2.5.

Table 2.6 – Comparison of wood product models (source: based on Brunet-Navarro et al. (2016))

Model	Comment	Advantages	Disadvantages
CO2FIX3		Has a graphical user interface	-Not sure if products can be tracked without forestry if another forestry model is chosen -Parameters seem already set and general
CBMFPS3 / CBM-FHWP	Developed by the Canadian Forest Service Replaced by CBM-FHWP	-Flexible -Designed with CBM-CFS3 in mind	No graphical user interface
Carbon Object Tracker		Uses CBM-CFS3 inputs	Does not seem readily available
CAPSIS	Based in France		seems to be a complete forestry calculator and wood product model - forestry calculator specific to France, not sure if it is possible to use CAPSIS for HWP model

Due to its flexibility and pairing with the CFS-CFS3 forestry ecosystem software, CBM-FHWP is the most suitable software for modelling wood product carbon.

2.2.3.3 Timing in Use Phase

The use phase of buildings also involves timing and storage issues. Within this life cycle phase, the storage of carbon within the wood used in a building structure is of particular relevance. Another issue of relevance for overall carbon accounting (including fossil carbon sources) is the issue of the timing of emissions resulting from building energy use throughout the occupancy period of the building.

One of the main critiques of the neutrality assumption is that it ignores the questions of temporary carbon storage and delayed emissions, which can result in potential climate benefits. A more accurate portrayal of climate impacts can be achieved by estimating carbon fluxes from harvested wood products (HWP) over time (Lemprière et al., 2013). As such, HWPs used in long-lived products and buildings are stored for multiple decades, thus keeping the carbon contained within the wood within the anthroposphere and avoiding greenhouse gas emissions. On a general level, several authors (Chen et al., 2008; Guest & Strømman, 2014; Pan et al., 2011; Stinson et al., 2011) have attempted to make estimations of the overall quantity of carbon stored in HWP in various geographic regions. Most of these studies compared the carbon storage of harvested wood products to the current standing forests for a particular year or range of years. Globally, long-term carbon storage in harvested wood products and landfills is estimated to be increasing (Lemprière et al., 2013). This trend is positive, considering that Werner et al. (2010) and Smyth et al. (2014) found that wood used in long-lived products is more beneficial climatically than using wood for bioenergy.

A few authors have proposed means of considering the climate change impacts of carbon storage on a product or building level. Vogtländer et al. (2013) proposed a method for considering the carbon storage of HWPs in which the temporary storage of carbon in buildings or long-life products is only considered to have a net benefit on emissions if there is global growth of forest area and of wood application in the building industry. Levasseur et al. (2013) showed that the use of the dynamic LCA approach (see section 2.1.5.1 and Table 1.4) could consistently calculate climate change impacts and benefits of biogenic carbon throughout the life cycle of products, including temporary carbon storage during the product life. Guest et al. (2013b) related the temporary carbon storage of HWPs with the forestry rotation periods of that wood, enabling the forestry phase of the life cycle to be linked to the use of wood in long-life buildings. Guest et al. (2013a) found that biomass with shorter rotation periods but stored for longer durations had climate benefits, whereas biomass with longer rotation periods stored for shorter periods of time had climate change impacts. In addition, they found that storing biogenic carbon in long-life products does not always translate to climate change benefits. Regardless of the approach taken, the temporary storage of carbon should be accounted in biomass LCAs.

2.2.4 End-of-Life Disposal of Harvested Wood Products (HWP)

In order to cover the entire carbon balance of wood, it is logical to include the management of materials at the end-of-life. In the first stage of the carbon balance, carbon dioxide is sequestered in trees. The carbon remains in the wood after harvest and throughout the life of the wood product. At the end of the life of the product, the carbon can be released through combustion, degradation or it can be permanently stored in landfills. The research conducted on the topic of wood and wood product disposal has been predominantly conducted in the field of waste management rather than in the field of climate change or life cycle assessment (Barlaz, 2006; Micales & Skog, 1997; Wang et al., 2013; Ximenes et al., 2015; Ximenes et al., 2008). Although this research has been mostly concentrated on wood landfilling, other disposal options for wood will be addressed in this research. Wood is most often treated by combustion, landfilling or recycling.

In the context of buildings and construction, the end-of-life stage can occur several decades in the future, at which point the waste scenarios for demolished building materials is unknown. Sandin et al. (2014) examined the effects of future waste management scenario assumptions on the outcomes of environmental impacts of building materials. Their results suggest that the assumptions made about waste management scenarios of the future, such as type of disposal, level of technology and type of LCA approach (attributional vs. consequential), may have significant impacts in terms of the relative environmental impacts of the end-of-life phase of material alternatives. The most common end-of-life waste management technologies used for wood are discussed below.

2.2.4.1 Landfilling

In North America, landfilling is still a very common way of managing municipal and construction waste. Wood comprises of roughly 7% of all unrecovered waste sent to landfill in Canada (Howe et al., 2013), and 40% of construction, renovation and demolition (CRD) wood (Kelleher Environmental & Guy Perry and Associates, 2015). Conditions within modern landfills, are predominantly anaerobic due to their design both in preventing moisture and precipitation from entering and in the use of cover materials to prevent exposure to air. Typically only a minimal amount of aerobic decomposition occurs, in cases when waste is not immediately covered (Larson et al., 2012). Although the anaerobic decomposition of organic materials emits greenhouse gases (principally methane), several studies (Barlaz, 2006; Chen et al., 2008; Micales & Skog, 1997; Wang et al., 2013; Ximenes et al., 2015; Ximenes et al., 2008) demonstrate that wood degrades very slowly in landfill sites. Since wood consists of a complex lignin matrix that integrates cellulose and hemicellulose and the conditions of most landfills are anaerobic, only a small proportion of wood is degraded. A review of several wood decomposition in landfill studies (Micales & Skog, 1997; Wang et al., 2011; Wang et al., 2013; Ximenes et al., 2015) shows that a range of 0-8% of carbon contained in wood for most wood types is emitted as a gas at landfill sites. According to Micales, and Skog (1997), it is estimated that only 0-3% of carbon contained in wood is emitted as a gas at landfill sites. As such, a large portion of the carbon contained in the wood can be theoretically long-term stored, which may have potential climate benefits.

2.2.4.2 Incineration

Another common end-of-life treatment of wood is incineration. Like decomposition, incineration can take a few different forms. Worldwide, the most common form of wood incineration is through very rudimentary open fires. In very simple terms, the carbon in wood in the presence of oxygen and fire, is transformed into water and carbon dioxide (Kondratiev, 2015). With the current carbon neutrality paradigm, the (biogenic) carbon dioxide released from the combustion reaction is considered to not have a climate impact, such as shown in several life cycle assessment studies of wood products (Liamsanguan & Gheewala, 2008; Zabalza Bribián et al., 2011; Zhang et al., 2010).

On the other side of the spectrum, wood can also be treated in controlled and engineered incineration plants. At their most efficient, the energy generated through combustion can be captured and used, both to power an incineration plant itself and to provide useful electricity and heat to neighbouring energy users (UNEP, 2015). In this way, the energy produced through incineration can substitute fossil fuels and be used for producing useful energy.

2.2.4.3 Reuse and Recycling

Another possible disposal route for wood is through recycling and reuse. As in the case of the other end-of-life disposal options, recycling and reuse can be done through both highly technical and organised means but also through informal treatment. At the highly technical extreme, wood demolition waste can be centrally treated to produce wood chips or other wood products (Husum et al., 1999). At the informal extreme, used wood can be repurposed for other uses. For example, one could dismantle a table and reuse the pieces of wood for building another structure. In all cases, the reuse or recycling of wood displaces a need for virgin wood material in the economy. In North America, practically all wood residues occurring during manufacturing have been eliminated and redirected towards products such as particle board or on-site energy production (Howe et al., 2013). In terms of post-consumer wood waste, there are estimates that unrecovered wood debris from waste streams total 1.75 million tonnes annually in Canada.

In general, it is thought reuse and recycling of wood that keeps the embodied carbon within the anthroposphere for a longer period of time and thus have overall climate benefits (Lemprière et al., 2013). However, the consequences for the carbon accounting of the reuse and recycling of wood can become complex, due to the way in which the benefits of recycling are treated. The modelling of recycling processes in life cycle assessment has been debated for several years and has resulted in the proposal of several different modelling approaches (European Commission, 2010c). Several of the LCA, carbon footprinting and PCR guidelines also have guidance on recycling (Table 2.6).

Table 2.7 – Overview of recycling approaches

Approach	**Description**	**Notes**	**Reference**
ILCD handbook	-2 attributional (A) and 2 consequential (C) approaches* (A): recycling burdens/benefits allocated to system generating waste and system using secondary material: →If $ material >0: inventory solved by allocation →If $ material <0: is treated as a waste product and thus no benefits or burdens associated with that material.	-Allocation between two life cycles must be between co-function of EOL waste of primary product and future products – often unknown -Too many options for users (Lindfors et al., 2012)	European Commission (2010b)
PAS2050	-2 method options based on if the recycled material has same properties as the virgin material it is replacing: 1) *Recycled-content*: allocates recycling benefits (of avoiding virgin material) to 2nd life cycle when properties secondary material≠ virgin material 2) *Closed-loop approximation*: allocates recycling benefits to 1st life cycle when properties secondary material= virgin	-Secondary wood does not have the same properties as virgin wood, thus the recycled content approach is suitable for calculating the GHG emissions of recycling	BSI (2011c)
PCR for North American wood products	- If properties of recycled material *change*: use *open-loop* recycling, where the recycling burdens allocated to new downstream products. -If properties of recycled material are the *same*: use *closed-loop* recycling, where a substitution is used for virgin material.		FPInnovations (2013)

*attributional approaches assess the processes within a system for a given functional unit, whereas consequential approaches describe the consequences of often large-scale decisions/policies by considering all processes affected

In terms of the climate implications of wood recycling, in particular, a few authors have published on this topic. In their study on the LCA of particle board, Wilson (2010) considers carbon storage in the carbon balance and predicts that recycling processes will keep carbon out of the atmosphere even longer then the service life of the initial product. However, Werner et al. (2006) conclude in their study on end-of-life alternatives of wood products that there is no method of modelling post-consumer wood that would account for all situations of wood use. Kim, and Song (2014) used system expansion to deal with recycled materials in particleboard manufacturing, thus accounting for the avoidance of virgin materials through the use of recycled wood. They also calculated the carbon benefit of recycling based on the carbon storage of wood during the service life of the wood, as well as the extended period of storage attained through recycling. This calculation also accounts for the effects of progressively diminishing storage through material degradation as a result of several rounds of recycling. Although the approaches used for the treatment of wood recycling differ, it is clear that it has significant implications in terms of biogenic carbon accounting.

2.2.4.4 Timing in End-of-Life

The end-of-life disposal of a building also involves some timing issues, related to the emission of greenhouse gases. As noted in section 2.2.4, landfilling and recycling have potential temporal issues in terms of how and when emissions are released. The temporal profile of incineration is typically characterised by a single pulse emission or a few emissions occurring within a short time period.

In the case of the landfilling of wood, there are several simultaneous fates for carbon, including the release of methane, carbon dioxide and volatile organic compounds (VOCs) to the air, the release of carbon to landfill leachate and the storage of carbon in the landfill. The climate impacts of these fates are dependent on both the technology utilised at a given site (i.e. landfill gas collection), as well as the specific timing of these emissions and storage profiles (ICF International, 2015).

With recycling, emissions timing is related to the duration of the subsequent life cycles of the products in which the secondary material will be used. As previously discussed (Wilson, 2010), recycling can further prolong the temporary carbon storage period. Very recently, Faraca et al. (2019) modelled dynamic climate change impacts of the cascading use of wood waste in secondary products, finding that in general more cascade steps resulted in larger climate change impact benefits. Similar to the storage of wood in buildings, recycling can further postpone an eventual release of carbon by storing carbon in future products. Recycling also involves losses, as wood will progressively degrade over time and through multiple recycling rounds, as acknowledge by Kim, and Song (2014). These fluxes can also be accounted for on a temporal scale.

CHAPTER 3 PROCESS FOR THE RESEARCH PROJECT AS A WHOLE AND GENERAL ORGANISATION OF THE DOCUMENT INDICATING THE COHERENCE OF THE ARTICLES IN RELATION TO THE RESEARCH GOALS

3.1 Research Problem

Within LCA studies, the use of wood products has long been assumed to have a net zero biogenic carbon balance. As such, biomass is expected to release the same amount of carbon dioxide upon final disposal as is sequestered during its growth. However, there is an increasing body of scientific evidence that indicates that applying a carbon neutrality principle for wood products may lead to carbon accounting errors, which would influence overall climate change results. There are several important issues which could lead to different outcomes in terms of environmental impacts:

1. The carbon dynamics of various forestry management parameters, such as the tree species and region and type of forestry management, are not currently considered in the calculation of climate impacts of forestry products.
2. The climate impacts of the end-of-life treatment of wood are not consistently treated in LCA. Waste management processes for wood differ in terms of the climate impacts depending on the amount of carbon emitted (such as from combustion or landfill decomposition), the amount of carbon stored (such as in landfills) and the form in which the carbon is emitted (carbon dioxide or methane).
3. Temporary storage of harvested wood products in the technosphere, including extended lifetime into secondary products due to wood recycling, are not always included in the climate impacts of wood products and are rarely evaluated with respect to a relevant time horizon for decision-making.

So far there is no well-established method for calculating the carbon balance and climate impacts of wood products that simultaneously addresses all above mentioned issues, beyond the use of the simplifying assumption of carbon neutrality. For the building sector, this hinders the opportunity to obtain robust carbon footprints of wood products, especially in the case of comparisons with other building materials. This research will provide a framework that will attempt to consistently address the biogenic carbon balance over the entire life cycle of wood used in buildings in the Canadian context.

3.2 Thesis Objectives

The main objective of this project is to develop a method that consistently accounts for the uptake, emission, storage and timing of biogenic carbon in the life cycle assessments of wood used in buildings. Three specific objectives have been identified:

1. Develop temporally differentiated carbon flux profiles of the forestry carbon dynamics phase of softwood products at the landscape level as a function of tree species, growing conditions and forest management practices across the Canadian managed forest.
2. Develop temporally differentiated carbon flux profiles from the point of harvest through wood product manufacturing, building life and end-of-life phases, for wood products across Canada.
3. Apply dynamic life cycle assessment to cradle-to-grave temporally differentiated carbon flux profiles of wood products, developing life cycle inventories and climate change scores for a range of product types for use by designers, architects and LCA practitioners in respect to a relevant time horizon for decision making.

3.3 General Methodology

This section presents an overview of the methodology applied to develop a method that consistently accounts for the uptake, emission, storage and timing of biogenic carbon in the life cycle assessments of wood used in buildings. The research is comprised of eleven phases divided amongst the three sub-objectives of the research. A flow diagram illustrating this general methodology is show in Figure 3.1.

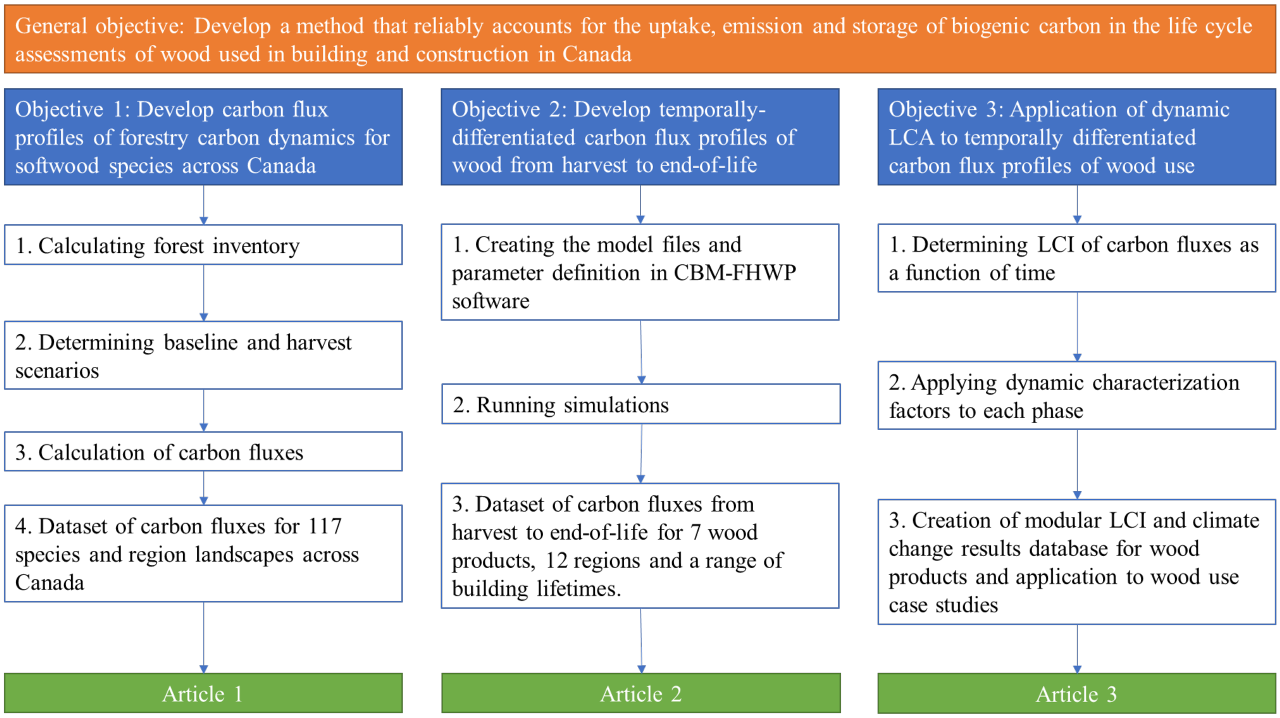

Figure 3.1 – General methodology flow diagram

Further explanation of the phases is detailed below for each sub-objective and in-depth methodology is detailed in Chapter 4, 5 and 6 for each of the sub-objectives. A general discussion of the research, including research limitations and future research as well as conclusion are provided in Chapters 7 and 8.

3.3.1 Develop carbon profiles of forest carbon dynamics for softwood species across Canada

The first article, provided in Chapter 4, focuses on quantifying the effects of consistently and sustainably harvesting wood from Canadian forests. More specifically it involves developing carbon profiles or ecosystem carbon costs of harvesting wood from the start of forest management in a forest landscape. The methodology consists of four main phases: 1) calculating yield curves, area affected by disturbance and age classes by species/region, 2) determining baseline and harvest scenarios, 3) calculation of carbon fluxes, 4) database of carbon fluxes for 117 species and region landscapes across Canada. The database of carbon fluxes (termed Ecosystem Carbon Costs – ECC) at the landscape level brings a new level of quantification of the effects of harvesting sustainably across a wide range of provinces and terrestrial ecosystem for a variety softwood tree species.

In the first phase, the forest inventory was calculated for the simulations that are carried out using forest carbon software. This involved calculating yield curves for 12 species of softwood trees using a theoretical yield curve model. Temperature and precipitation inputs into the model were derived from intersecting climate maps with species and region maps and calculating mean values. The areas disturbed and the calculation of age classes were calculated using literature and mapping information.

Then, a baseline scenario was established, which considers wildfire as a part of the natural forestry system. A harvest scenario was also modelled with both wildfire and a very small annual harvest rate across each landscape. It is important to note that the scenarios were run consecutively, that is, baseline followed by harvest, such to replicate the beginning of historical forest management after a natural forest state. The carbon fluxes were calculated by inputting the calculated forest inventory into forest carbon software and simulating carbon stocks. ECC are calculated taking the interannual differences minus the carbon content of the wood harvested. Finally, a database of ecosystem cost curves is made available for 117 species/region landscapes, with reporting of values after 100 years of historical harvest.

3.3.2 Develop temporally differentiated carbon flux profiles of wood from harvest to end-of-life

In the second article, provided in Chapter 5, temporally differentiated carbon fluxes from the harvesting to end-of-life are developed for a range of wood product types, regions and building lifespans. The methodology consists of three phases: 1) creating the model files and parameter definition in CBM-FHWP software, 2) running simulations, 3) dataset of carbon fluxes from harvest to end-of-life for 7 wood products, 12 regions and a range of building lifetimes.

The first phase involves creating the model files and defining parameters for simulating seven wood products in the CBM-FHWP software, by using sawmill mass balances and the fate of coproducts and waste products throughout use and end-of-life phases of the life cycle. Model files are adaptable to different provinces/territories and building lifespans. In the second phase the models are simulated using CBM-FHWP. Since this software does not have a graphical user interface, simulations are carried out using command files and are compiled using additional MATLAB code. Finally, a dataset of carbon fluxes is calculated for both CO_2 and CH_4 emissions (in kg C) for seven wood product types, 12 provinces/territories and building lifetimes ranging from 1-150 years.

3.3.3 Application of dynamic LCA to temporally differentiated carbon flux profiles of wood use

In the third article, provided in Chapter 6, dynamic life cycle assessment is applied to temporally differentiated carbon profiles of the life cycle phases of a wood building product. The methodology consists of three phases: 1) determining life cycle inventory (LCI) of carbon fluxes as a function of time, 2) applying dynamic characterisation factors with respect to a time horizon relevant for the decision, 3) creation of modular LCI and climate change impact database for wood products and application to wood use case studies.

In the first phase, life cycle inventories for all life cycle phases of wood products were temporally differentiated. For the forestry phase, this involved including the ecosystem carbon costs, carbon uptake in wood and forest management as single carbon fluxes in year 0. Sawmill and construction and transport emissions were calculated as single fluxes occurring in year 1. Embodied emissions from processing through to end-of-life from research objective 2 (article 2) were also considered. Then in the second phase, dynamic characterisation factors were applied to each of the temporally differentiated carbon profiles, for all life cycle phases. The third phase involved creating a modular LCI and climate change impact database for wood products and applying it to four wood use cases using common wood mixes. For forest carbon (carbon uptake and ecosystem carbon cost), this involves calculating a weighted average of dynamic climate change emissions based on forest species and region. For the other life cycle phases, sample case specifications were chosen and the dynamic climate change emissions corresponding to those specifications were used for those cases. Finally, a database of dynamic LCI and climate change impacts for all life cycle phases in all of their permutations is created and made available for modular construction of custom cradle-to-grave LCAs.

CHAPTER 4 ARTICLE 1: FORESTRY CARBON BUDGET MODELS TO IMPROVE BIOGENIC CARBON ACCOUNTING IN LIFE CYCLE ASSESSMENT

4.1 Introduction to Article 1

The manuscript presented in this chapter demonstrates the calculation of dynamic carbon profiles of forestry carbon dynamics representing the impacts of harvest wood on the forest. This work that lead to the writing of this article was conducted in collaboration with Pierre Bernier from the Canadian Forest Service. The authors of this article are Marieke Head, Pierre Bernier, Annie Levasseur, Robert Beauregard and Manuele Margni. It was submitted on May 23rd, 2018 to the *Journal of Cleaner Production* and was approved for publication on December 12th, 2018.

Following the review and publication of Article 1 "Forestry carbon budget models to improve biogenic carbon accounting in life cycle assessment", just weeks before submitting this dissertation, an error was discovered in the calculation of the ecosystem carbon costs (ECC). Although the overall conclusion that most landscapes show net carbon sequestration at 100 years still holds true, the specific ECC values differ slightly from the published values. In addition, certain landscapes (namely Engelmann spruce) have more positive emissions on average. A more detailed explanation of this error and some updated results can be found in a corrigendum (see Appendix A). There are plans to submit an official correction to these values to the *Journal of Cleaner Production* in the coming months. Although the Supplementary Material can be found with the published article online (https://www.sciencedirect.com/science/article/pii/S0959652618338320), it is recommended to use the updated values in Appendix A. Appendix B provides an overview of the input parameters used in modelling forestry carbon.

4.2 Manuscript

4.2.1 Introduction

Wood is commonly used as a building material throughout the North American construction sector. The life cycle assessment (LCA) methodology is increasingly used to assess and compare the potential environmental impacts of construction materials while considering all life cycle stages, from raw materials extraction to end-of-life (ISO, 2006a, 2006b). However, the climate impacts of wood and wood construction materials currently have limitations in how they are accounted for in life cycle assessments. Since biomass is considered to be part of the fast domain of the carbon cycle, the carbon fluxes between the atmosphere and biomass have been differentiated from the carbon fluxes originating from fossil sources (Ciais et al., 2013). Being part of the fast domain of the carbon cycle, the carbon from biomass, known as biogenic carbon, is thus said to have a net carbon balance of zero, meaning that the carbon sequestered by biomass is equal to the carbon eventually released by that biomass. Several publications have shown that this assumption could lead to accounting errors (Garcia & Freire, 2014; Røyne et al., 2016; Searchinger et al., 2009; Vogtländer et al., 2013), incentives to clear-cut forests (Searchinger et al., 2009) or the creation of a temporal shift in carbon uptake and release causing an increase in cumulative radiative forcing (Helin et al., 2013). Moreover, this assumed net zero carbon balance has also been equated to a net zero climate change impact. Given the dynamic nature of biogenic carbon emissions and sequestration, the simplistic paradigm that carbon neutral equals climate neutral is also being questioned (Cherubini et al., 2011; Levasseur et al., 2012; Zanchi et al., 2010).

Net carbon neutrality is often argued based on the instantaneous oxidation approach used in early guidelines published by the Intergovernmental Panel on Climate Change (IPCC, 1997), whereby the carbon in harvested wood is considered emitted in the year of harvest. These guidelines also assumed that the net amount of carbon stored in harvested wood products was constant over time. However, the storage of carbon in harvested wood products is thought to be increasing as products are kept in use or are stored in landfills upon disposal (Lemprière et al., 2013). By using the instantaneous oxidation argument in Canada, where wood is used extensively in buildings, the greenhouse gas inventory emissions are overestimated (Dymond, 2012; Smyth et al., 2014). From a life cycle assessment perspective, considering a net carbon neutrality through all the life cycle stages of a wood product is an overly simplistic assumption on both the life cycle inventory and the potential impacts on climate change. In the forestry phase, a carbon neutrality approach ignores the site-specific carbon dynamics (Coursolle et al., 2012; McKechnie et al., 2011), and prevents that the carbon fluxes specific to forest ecosystems or forest management be factored into life cycle assessments. Within the use phase of a wood product, none of the temporary carbon storage or emission delays would be considered either (Brandão & Levasseur, 2011). Finally, at the end-of-life, the carbon neutrality approach can yield considerably different results compared to methods that quantify biogenic carbon of waste disposal options such as landfilling, recycling and incineration (Christensen et al., 2009; Levasseur et al., 2013; Muñoz & Schmidt, 2016).

Forests can act as either net carbon sinks or net carbon sources with respect to the atmosphere. Under normal growth conditions and in the absence of significant disturbances, forests are typically net carbon sinks as they absorb more carbon dioxide than they release to the atmosphere. When forests undergo stand-replacing disturbances such as fires or insect outbreaks, they usually become carbon sources as they release more carbon dioxide than they absorb from the atmosphere (Natural Resources Canada, 2016). Although most of the world's forests tend to be net carbon sinks (Pan et al., 2011), this source-sink interaction adds significant complexity to the forest carbon balance. From the perspective of life cycle assessment, the source-sink interactions of a managed forest need only to be benchmarked to a natural state in order to account for the human influence on harvested wood (Böttcher et al., 2008; Cao et al., 2016; Keith et al., 2009; Lessard, 2013).

Work on forest carbon dynamics has largely focused on the carbon balances for national greenhouse gas accounting (Kauppi et al., 2010; Kindermann et al., 2008; Kurz et al., 2009; Kurz et al., 2013; Luyssaert et al., 2007). Recently a few authors have considered the lack of consistent forestry carbon accounting in LCA. In a recent review of approaches, Helin et al. (2013) found large differences in how forest carbon stocks are considered in LCAs. Out of the 26 studies reviewed, eleven considered all aboveground and belowground carbon pools in modelled forestry carbon stocks, while in another cross-section nine studies (of the 26) used a IPCC Tier 3 approach (IPCC, 2006a). Some authors proposed simplified approaches that can easily be applied to forest systems around the world. While these approaches are flexible and versatile, they may result in a high degree of uncertainty due to the variation in tree species, climatic conditions and forestry management practices possible in the world's forests. Only five studies (of the 26) were based on national forest inventory data, an approach which allows for tracking carbon exchanges over time through various forestry management scenarios. One of these last five is that of McKechnie et al. (2011) in which the authors present a framework to integrate life cycle inventory (LCI) and forest carbon modelling. They evaluated a regional-level forest-based bioenergy case in Ontario (Canada) using the FORCARB2 model. Although McKechnie et al. (2011) focused on a Canadian case study, they focused on one region within the province of Ontario and presented aggregated results for all tree species. While the FORCARB2 model makes use of robust empirical estimates of aboveground forest carbon pools, the model cannot simulate natural disturbances such as wildfires (Zald et al., 2016). Our work aims to improve on the cases presented above and to model forest carbon for several species and regions across Canada through the calculation of a natural forest state that includes wildfires.

The work we presented below is part of a larger research project on the use of wood as a building material within the context of Canadian forests. The extent and slow growth rates of Canadian forests and the prevalence of natural stand-replacing disturbances make these forests and forestry management different from that of other forestry regions. The frequent natural stand-replacing wildfires in these forests are an integral part of their natural dynamics (Boulanger et al., 2014; Stocks et al., 2002). In comparison to forests in other regions, the slow growth rate of boreal trees results in much lower biomass volume accumulation on a given area over time (Bogdanski, 2008; Brandt et al., 2013; Jarvis & Linder, 2000) and thus long intervals between successive harvests.

This research work aimed at improving biogenic carbon accounting in the life cycle assessment of softwood products by specifically modeling carbon fluxes of the forestry phase as a function of tree species, growing conditions and forest management practices. More specifically, the objective of this work was to quantify the net impact of harvest activities on the ecosystem carbon costs of forest ecosystems in Canada. This was achieved by calculating the carbon fluxes of harvested forests covering a range of climatic conditions and disturbance rates found across the Canadian managed forest, and for softwood tree species commonly used in Canadian building construction. The resulting carbon fluxes were then allocated to the units of wood harvested in a given landscape, allowing for the calculation of carbon fluxes of cradle-to-gate wood harvest in LCA.

4.2.2 Methods

4.2.2.1 Modeling forestry carbon fluxes for softwood harvest

From the perspective of a building planner choosing a building material, a wood product could be made from many different softwood tree species and could originate from any Canadian managed forest. As such, the scope of the forest carbon flux calculations covered the most common softwood tree species harvested commercially for building materials across Canada. This translated into the creation of several landscapes that are specific for a given species, Canadian province (or territory) and terrestrial ecozone. Each of these landscapes was subjected to disturbances, resulting in changes in the carbon stored and emitted from that forest that are specific to each landscape.

The terms used in this text to refer to carbon dynamics are defined below:

Biomass: Biomass is the mass of all living vegetation which includes both aboveground (stem, stump, branches, bark, seeds and foliage) and belowground (roots) portions of trees. In the context of this paper, dead trees (snags) and non-tree biomass (moss, shrubs) are excluded.

Total carbon stocks (TCS): Refers to the sum of carbon mass across all ecosystem carbon pools (in $tC\cdot ha^{-1}$) in a given area of forest, including all biomass and dead organic matter (DOM) (including soil).

Net carbon flux (NCF): The net result of the uptake of carbon through photosynthesis and carbon losses through plant respiration or decomposition. At the forest landscape level, *NCF* includes carbon losses due to fire as well as the removal of carbon through the harvesting of wood. In this study, positive values of *NFC* represent a net forest C loss to the atmosphere.

Ecosystem carbon cost (ECC): Within the context of this study, the net carbon flux of the forest to the atmosphere attributed to the harvesting activity. *ECC* is calculated at the landscape level and expressed per unit of wood (cubic meters or tonnes) harvested per year. Positive values of *ECC* represent net forest C losses to the atmosphere.

Time to ecosystem cost neutrality: The number of simulated harvest years required for the *ECC* to cross the zero line and reach carbon neutrality.

Sequestration: A net carbon flux between the atmosphere and the forest that results in increased *TCS* over one or more years. In the context of this paper, the forest is considered from a landscape-level meaning that it also includes the effect of fire. Sequestered C can be transferred out of the forest and remain sequestered as wood products.

The modelling of forest ecosystem carbon dynamics was carried out using the CBM-CFS3 software (Kurz et al. 2009). CBM-CFS3 is a carbon budget model developed by the Canadian Forest Service (Natural Resources Canada) that keeps track of carbon fluxes within user-defined forest landscapes, as driven by tree growth, natural disturbances and forest management (Figure 4.1). The

CBM-CFS3 can be used to simulate forest carbon dynamics in forests of any composition across Canada, but does not account for non-tree plant species in its calculations. The model can be used at the level of stands and of landscapes, which allows for the landscape-level approach suitable to this research (Zald et al., 2016). A landscape-level perspective considers a much larger forest area than the stand-level perspective and consists of stands of differing ages, disturbance histories, species compositions and site conditions, across which disturbances take place. The CBM-CFS3 model can simulate natural disturbances (Zald et al., 2016), which allows for the estimation of a natural baseline state in the forest, as well as of transient states where forest management is a recent addition to the landscape dynamics.

The three main inputs to the model were areas per tree species and age class, volume over age (yield) curves and area affected annually by disturbance type (Figure 4.1). The model provides annual estimates of C stocks by ecosystem reservoir, the sum of which give annual values of *TCS*. We calculated annual values of *ECC* as the interannual difference in *TCS* of the forest landscapes, minus the carbon contained in the harvested wood. The resulting values were expressed in $tC{\cdot}ha^{-1}{\cdot}yr^{-1}$ as well as in $tC{\cdot}m^{-3}$ of harvested wood yr^{-1} or $tC{\cdot}t^{-1}$ harvested wood, using the LCA sign convention of representing a flux to the atmosphere as a positive value. More details on these calculations can be found in section 4.2.2.5.

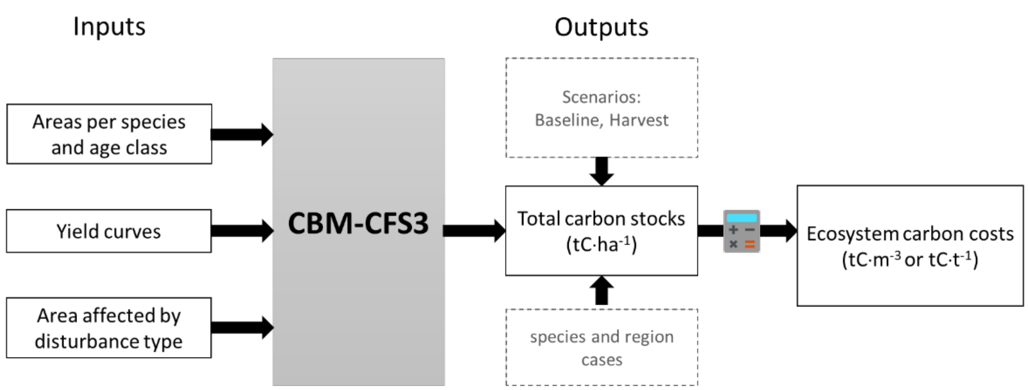

Figure 4.1 – Schematic overview of inputs and outputs used with CBM-CFS3

Vectorial maps of commercial softwood tree species distribution across Canada were created using the 250-m resolution Canada-wide maps of tree composition from Beaudoin et al. (2014). The boundaries of the tree species maps were set by drawing vector polygons of the pixel clusters representing the forested areas. These maps were then overlaid with a map of the Canadian managed forest area, of provincial and territorial boundaries and of ecozones (Ecological Stratification Working Group, 1996). The intersections of 14 softwood tree species, 12 Canadian provinces and territories, and 13 terrestrial ecozones yielded 266 landscapes that range in area from 30 to 4 800 000 ha.

The age frequency distribution for each species and terrestrial ecozone were calculated using the 250m-resolution stand age and forest composition maps of Beaudoin et al (2014) taken in 2001 as a frequency distribution of pixel counts per 10-year age increment. The surface area of each age class in each landscape was calculated as the product of age frequency per 10-year age increment and the total surface area of each landscape. These age distributions were used as a starting point for the simulations.

4.2.2.2 Yield curves

A yield curve is an empirical relationship that predicts the wood volume of a stand of a given tree species as a function of the stand age. Such curves are required inputs for the CBM-CFS3 model. For our analysis, we used the national yield curves of Ung et al. (2009) that had been parametrised for most commercial tree species in Canada using data obtained in field plots across the country:

Eq. 1 $\ln V = v_{10} + v_{11}T + v_{12}P + \left(\frac{v_{20}+v_{21}T+v_{22}P}{A}\right)$

Eq. 2 $V = \ln(V)C_d$

Where,
V = Gross total volume of live merchantable trees ($m^3 \cdot ha^{-1}$); a merchantable tree has a diameter at breast height (dbh) greater than 9 cm
v_{ij}= species-specific coefficients for 25 species
T = mean annual temperature (°C)
P = total precipitation (mm)
A = Plot age (yr)
C_d = correction factor

We calculated the values of mean annual temperature (T) and total precipitation (P) for each of the defined landscapes using 1981-2010 climate normal maps (McKenney et al., 2016). The resulting values were used as inputs in equations 1 and 2, with A ranging from 0 to the maximum natural lifespan (Burns & Honkala, 1990) of each species. Considering the mathematical basis of the yield curve model and the calculated coefficients, landscapes with atypical yield curves were discarded (23 cases) yielding a new total of 243 landscapes to be simulated.

4.2.2.3 Area affected by disturbance type

The disturbances defined in the CBM-CFS3 simulations in the model are fire and harvest, hence only those disturbances were considered in our analysis. Values of mean percent annual area burned within each ecozone were first calculated as the area-weighted mean of the homogeneous fire regime zones defined by Boulanger et al. (2014). For each of our 243 landscapes, we then calculated a value of mean annual area burned as the product of its ecozone annual burn rate and the area of the landscape. Values of mean annual area harvested were calculated using published historical area-based harvest rates by forest management units across Canada (Gauthier et al., 2015). Based on the proportional area of each unit, a weighted mean of harvest intensity was calculated by ecozone. These harvest intensities were then multiplied by the areas of each landscape to obtain mean annual harvested area by landscape. The complete list of the harvest rates and climatic data can be found in the Supplementary Material.

4.2.2.4 Creation of simulation scenarios

Two management scenarios were developed for this study:

- The *baseline scenario* simulates the carbon fluxes between the forests and the atmosphere under natural no-harvest conditions, where a proportion of each landscape is subjected to a constant annual burn rate for a 1000-year period (Boulanger et al., 2014). For the purpose of this work, the forest is assumed to have reached an approximate steady-state at 1000 years.
- The *harvest scenario* includes both the regular natural disturbance regimes of the baseline scenario as well as an annual harvest based on the regional harvest rates reported in Gauthier et al. (2015), and continues from the steady-state point of the baseline scenario for a simulation period of 100 years (thus from year 1001-1100) adding forestry harvesting activities. As shown by McKechnie et al. (2011), a 100-year period is consistent with long-term forestry management. It also reflects the historical period of

management across most of Canada's forests. The harvest scenario does not include the collection of harvest residues.

4.2.2.5 Calculation of carbon fluxes

CBM-CFS3 simulations were initialised across all 243 species and region landscapes at year 0 using forest composition and age class distribution for the year 2001 from Beaudoin et al. (2014), region- and species-specific yield models from equations 1 and 2, and disturbances based on Boulanger et al. (2014) and Gauthier et al. (2015) as described above. For a given landscape, for each year, the model calculated the mean value of carbon stocks per hectare as well as the mean value of carbon in the harvested wood, also expressed per hectare.

As mentioned above, the baseline scenario at year 1000 was taken as an approximate steady-state reference point for each of the landscapes, which was followed directly by the harvest scenario for 100 years (thus from year 1001 to year 1100). The ecosystem carbon costs *($ECC_{harvest}$)* of harvest in the forest ecosystem were calculated as the annual intervals or the partial derivatives of the *total carbon stocks* (TCS), subtracted by the carbon contained in the wood harvested annually, for each harvest simulation year or 0 $\leq$ t $\leq$ 100 years:

Eq. 3 $$ECC_{harvest,t}\left(\frac{tC}{ha\ yr}\right) = \frac{\partial}{\partial t}TCS - C\ content\ of\ wood\ harvest$$

To convert ECC results from tC·ha^{-1}·yr^{-1} to tC·m^{-3} wood and tC·t^{-1} wood, the amount of wood harvested each year in a given landscape was first converted to m^3 wood·ha^{-1}·yr^{-1} and t^{-1} wood·ha^{-1}·yr^{-1}.:

Eq. 4a $$Wood\ harvested\left(\frac{m^3 wood}{ha\ yr}\right) = \frac{C\ content\ of\ wood\ harvest\ (\frac{tC}{ha\ yr})}{\frac{0.5tC}{t\ wood} \times wood\ density_{species}\left(\frac{t\ wood}{m^3 wood}\right)}$$

Eq. 4b $$Wood\ harvested\left(\frac{tonne\ wood}{ha\ yr}\right) = \frac{C\ content\ of\ wood\ harvest\ (\frac{tC}{ha\ yr})}{\frac{0.5tC}{t\ wood}}$$

The ecosystem carbon costs for the harvest scenario over a landscape was then converted to ecosystem carbon costs per volume of wood harvested, such to allocate the carbon fluxes to the amount of wood harvested instead to the managed forest surface:

Eq. 5a $ECC_{harvest}\ \left(\frac{tC}{m^3 wood}\right) = \frac{ECC_{harvest}\left(\frac{tC}{ha\,yr}\right)}{Wood\ harvested\left(\frac{m^3 wood}{ha\,yr}\right)}$

Certain results were also calculated per mass of wood harvested in order to assess the proportion of carbon content that ecosystem carbon costs represent:

Eq. 5b $ECC_{harvest}\ \left(\frac{tC}{twood}\right) = \frac{Net\ relative\ C\ flux\left(\frac{tC}{ha\,yr}\right)}{Wood\ harvested\left(\frac{t\,wood}{ha\,yr}\right)}$

The ecosystem carbon costs, as calculated by equation 5a, were plotted for all 243 landscapes. We then examined all curves and eliminated from the analysis landscapes in which carbon stocks still increased during the period with harvest or followed an erratic trajectory, as well as landscapes whose combination of species and geographic range was deemed to be commercially irrelevant to the Canadian wood industry. The remaining 117 landscapes, provided in the Supplementary Material, were used for the analysis.

4.2.2.6 Aggregated results over larger regions

In addition to providing detailed results for specific species and regions, we also calculated aggregated results to account for the perspective of a wood user who might not know the species or regional origin of a given wood product. The aggregation is based upon the calculation of the weighted mean of all landscapes, where the weights are based on harvest volumes of species by regions across Canada (National Forest Inventory, 2013). Weighted means were calculated for each species, each province as well as for Eastern and Western Canada wood markets.

4.2.2.7 Evaluation against monitored data from flux towers

Carbon flux results obtained using the CBM-CFS3 simulations were validated against empirical CO_2 flux measurement data. For several years, the Canadian Carbon Network measured CO_2 exchanges across a network of forest sites in Canada using eddy covariance flux towers (Coursolle et al., 2006; Margolis et al., 2006). One of the sites near Chibougamau, Quebec, is covered by a mature forest dominated by black spruce (*Picea mariana*) and jack pine (*Pinus banksiana*) (Bergeron et al., 2007; Bernier & Paré, 2013; Bernier et al., 2010; Margolis et al., 2006). The history of the site includes wildfire between 1885 and 1915 affecting 74% of the site and harvest in the 1960s affecting 17% of the site area (Margolis et al., 2016). Carbon flux measurements from this flux tower have been already been used in many analysis, including a comparison of ecosystem carbon models (Bernier et al., 2010), and the calculation of carbon debt from bioenergy use (Bernier & Paré, 2013).

The flux tower on-site gathered high-frequency measurements of vertical wind velocity, air temperature, water vapour density and CO_2 concentrations in the air, which were then transformed into estimates of CO_2 fluxes as NEE (net ecosystem exchange) for half-hour intervals from 2003 to 2010 in terms of $\mu mol\ CO_2 \cdot m^{-2} \cdot s^{-1}$ (Coursolle et al., 2006). We averaged for each year the half-hourly NEE values for every year from 2004 to 2010 (measurements for 2003 only covered from June to December). To obtain a carbon flux in $tC \cdot ha^{-1} \cdot yr^{-1}$, the annual averages were multiplied by the ratio of the molar mass of CO_2 over the molar mass of C. An average over the period 2004-2010 was calculated such as to smooth out the interannual variability of the flux tower data and used it as a benchmark to evaluate model results.

These values were plotted alongside modelled ecosystem carbon costs of black spruce and jack pine landscapes within the Quebec Boreal Shield, for the baseline scenario as well as the harvest scenario. Although the modelled landscapes describe an annual disturbance and flux tower stands have been subject to infrequent disturbances, an attempt was made to manage the inherent differences between these two datasets. This was done by offsetting the simulation period of the modelled landscape curves to correspond with the number of years since the disturbance events of the flux tower stands. For example, wildfire affected the stands surrounding the flux tower at year 0, which was used as the start year for the modelled post-fire baseline landscape, and a wildfire scenario was run for 118 years to estimate carbon stocks at year 118. Similarly, harvest occurred at year 65 and thus the harvest scenario was run for a further 100 years (year 65-year 165) as to estimate carbon stocks at year 118.

4.2.3 Results and Discussion

The data points in the figures presented in this section can be found in the Supplementary Material.

4.2.3.1 Total carbon stocks

The baseline scenario reached an approximate steady-state in total carbon stocks after a landscape-level spin up period of 1000 years, following an initial spin-up designed to reach an initial equilibrium among carbon pools at the site-level (Figure 4.2). The spin-up refers to an initialisation step executed by the CBM-CFS3 that assigns values to the dead organic matter (DOM) pool which is not measured as part of the regular forest inventory. A constant harvest regime from years 1001 to 1100 was imposed on the landscapes following the realisation of the baseline steady-state. The carbon stock values on the harvest curves were used as inputs to the calculation of ecosystem carbon cost for each cubic meter of harvested wood (Eq. 3).

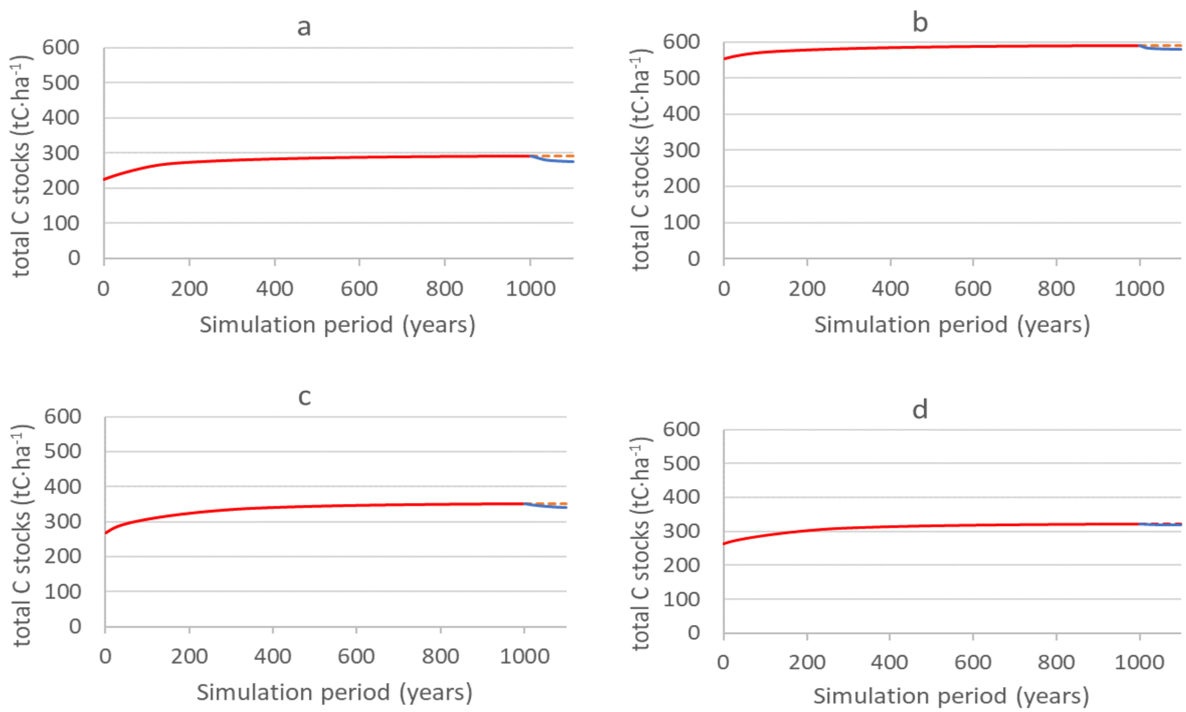

Figure 4.2 – Total carbon stocks of the baseline period from 0-1000 years (red line), followed by a harvest period from 1001-1100 years (blue). The dotted red lined represents the approximate steady-state value of the baseline period at 1000 years. a) Balsam fir (Abies balsamea), Quebec, Boreal Shield East, b) Lodgepole pine (Pinus contorta, British Columbia, Pacific Maritime, c) Western larch (Larix occidentalis), Alberta, Subhumid Prairies, d) White spruce (Picea glauca), New Brunswick, Atlantic Maritime.

The baseline curve shows a rapid increase in total carbon stocks in the first few hundred years, followed by a flattening out as the forest carbon approaches a steady-state, as younger trees are re-established in the landscapes (Figure 4.2). The total carbon stocks of the harvest regime decreased from the baseline steady-state as expected and similarly to the results of Lessard (2013) for a landscape modelled in the Quebec Boreal Shield. This sample of four landscapes in Figure 4.2 also shows that the total carbon stock curves for the harvest regime vary by species type and ecozone, as a result of differences in the growth rates of the trees and the proportion of biomass affected by disturbances.

4.2.3.2 Ecosystem carbon costs of harvest activity

Using the total carbon stocks of just the harvest period (from 1001-1100 years) as well as the carbon content of the wood harvested annually per hectare, the ecosystem carbon costs were calculated using eq. 5 for all the landscapes. The ecosystem carbon costs curves are shown for four landscapes (Figure 4.3).

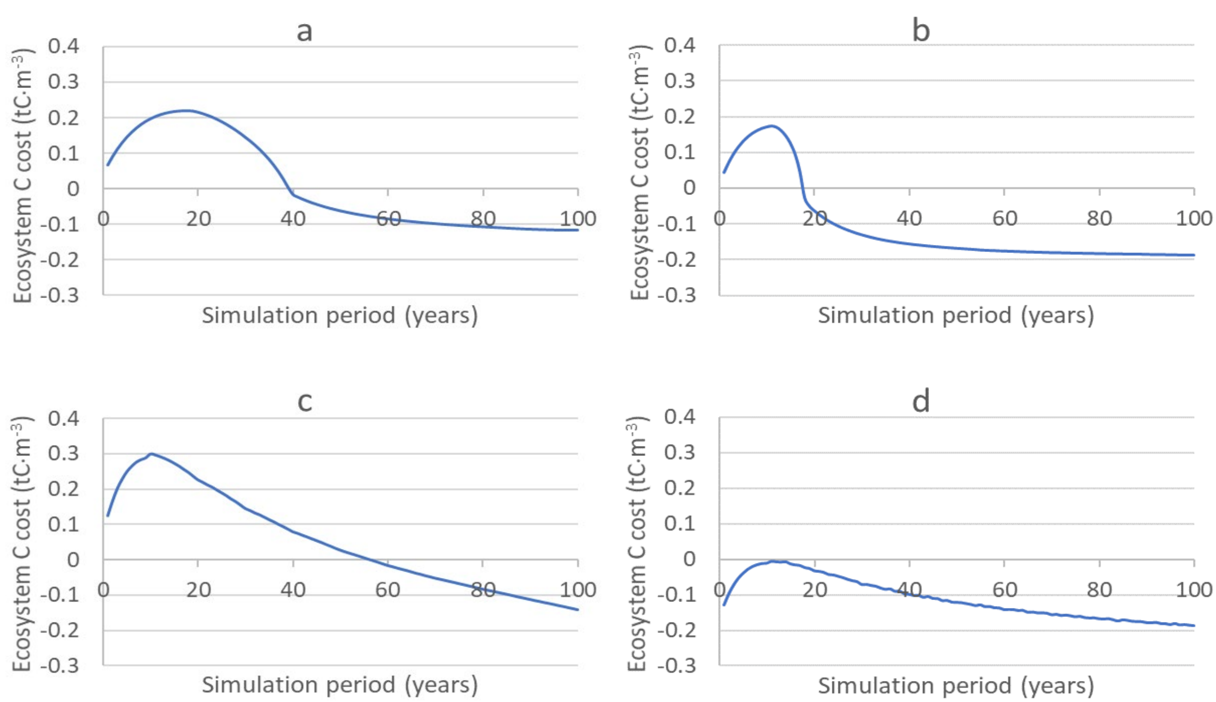

Figure 4.3 – Ecosystem carbon cost, as net annual loss of carbon from the forest ecosystem to the atmosphere per cubic meter of wood harvested that year ($tC{\cdot}m^{-3}$ wood) for four sample landscapes, as calculated with eq. 5a. The 0-100 period corresponds to the 1000-1100 period in Figure 4.2. a) Balsam fir (Abies balsamea), Quebec, Boreal Shield East; b) Lodgepole pine (Pinus contorta), British Columbia, Pacific Maritime; c) Western larch (Larix occidentalis), Alberta, Montane Cordillera; d) White spruce (Picea glauca), New Brunswick, Atlantic Maritime.

Positive values represent a net C loss to the atmosphere, while negative values represent a net C gain from the atmosphere.

While the calculations of total carbon stocks (Figure 4.2) include the transfer of carbon to harvested wood, here, those of *ECC* represent only the net loss in carbon to the atmosphere and are further expressed per unit of wood harvested (Figure 4.3 and Eq. 3). For all four sample landscapes (a-d), the *ECC* increases rapidly in the first decade and are followed by a decrease.

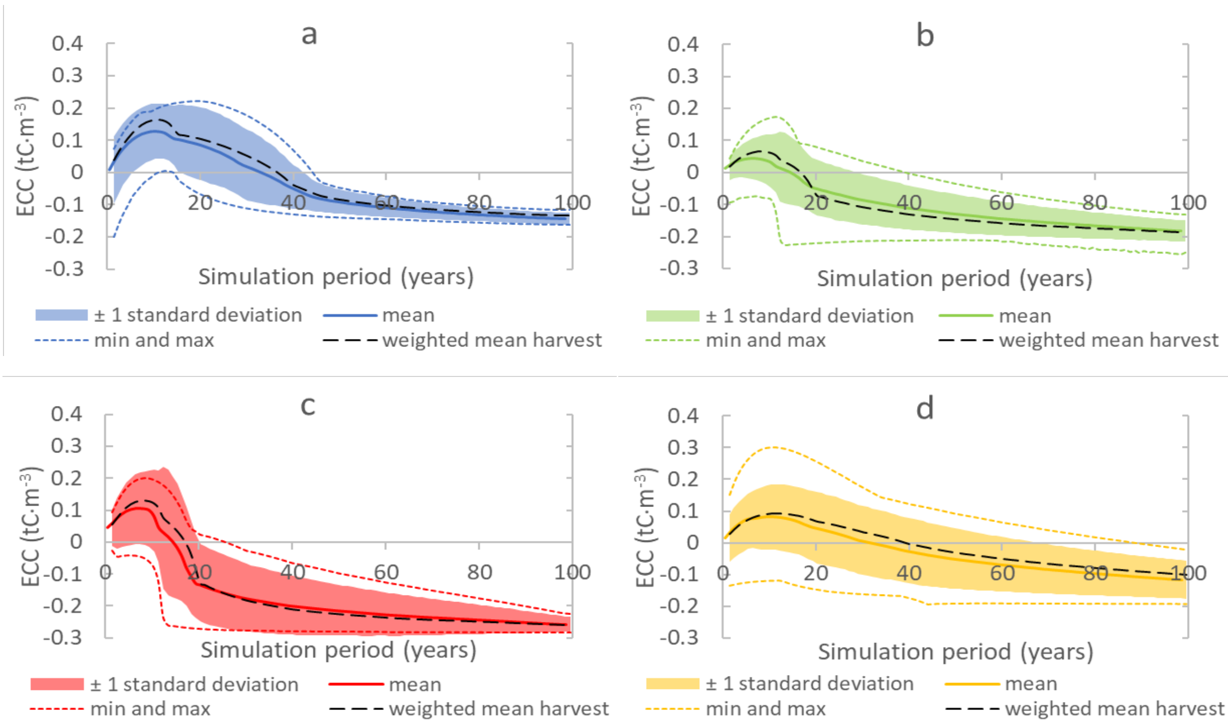

Figure 4.4 – Ecosystem carbon cost (ECC) by cubic metre of wood harvested for four common softwood tree species across Canada. The curves represent the interannual intervals of the harvest activity minus the carbon contained in the harvested wood, divided by the carbon content of the annual harvest volume (see eq. 5a). a) Balsam fir (Abies balsamea), all occurrences, b) Lodgepole pine (Pinus contorta), all occurrences, c) Western larch (Larix occidentalis), all occurrences, d) White spruce (Picea glauca), all occurrences. The dark centre lines show the mean species and regions, the lighter bands ± 1 standard deviation, the dotted lines the minimum and maximum and the black dotted lines are the weighted average based on harvest volumes.

While Figure 4.3 illustrated the curves for four individual landscapes, the curves in Figure 4.4 show the statistical spread of the landscapes for each species, which includes the landscapes featured in Figure 4.3. As with the individual landscapes (Figure 4.3), the statistical spread curves (Figure 4.4) show increased values of *ecosystem carbon cost* over the first few years of simulation, followed by a decrease over the 100-year simulation period. After a few decades, the curves cross the zero line as the landscape becomes a net carbon sink. This *time to ecosystem cost neutrality* varies between landscapes, with the mean for each species ranging from 16-60 years. At the two extremes, are a small number of landscapes that have either curves entirely with negative ecosystem carbon cost values or curves with positive ecosystem carbon costs that never reach carbon neutrality. The variability in time to ecosystem cost neutrality is affected by the shape and amplitude of the curves. The shapes of the curves are determined by the species-specific coefficients used in the yield curve equations (see equations 1 and 2), while the amplitude of the curves is related to the harvest rates and the climatic data used for creating the yield curves. The shape of the balsam fir and white spruce curves are similar, as the ecosystem carbon costs decrease steadily over time, while the lodgepole pine and western larch curves exhibit much steeper and rapid decreases.

Higher temperatures and precipitation tend to result in more biomass accumulation per year per hectare of forest and thus these landscapes have increased carbon sequestration capacity. This increase in carbon sequestration capacity means that landscapes are less affected by disturbances and thus have lower forest-to-atmosphere carbon fluxes. For example, in the case of balsam fir, the highest ecosystem carbon costs are associated with relatively low temperatures and precipitation (1.0-2.2°C and 800-1000 mm), while the lowest fluxes were found where temperature and precipitation were highest (6.4-6.9°C and 900-1400 mm). The complete list of the harvest rates and climatic data can be found in the Supplementary Material.

The weighted Canada-wide mean ecosystem carbon costs based on production volumes for each species across simulation years are close to the calculated mean curves (Figure 4.5a and 4.5b).

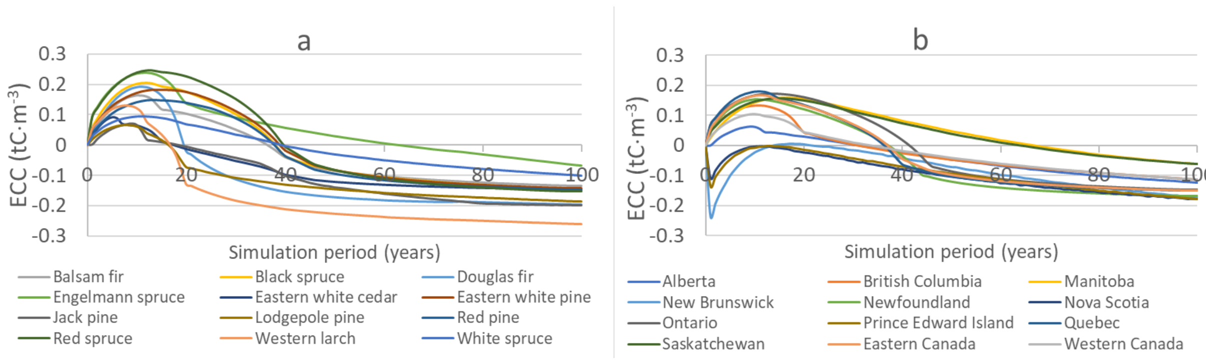

Figure 4.5 – Weighted mean of ecosystem carbon costs of harvest activity, by a) tree species b) provinces. Weights are harvested volumes.

The ecosystem carbon cost curve of each landscape is affected by the species-specific coefficients used in the yield curve equations, as well as regionally-specific climatic data and annual harvest rate, yielding significant differences among the weighted averages by species and by region. By species (see Figure 4.5a), the ecosystem carbon costs per m^3 of wood are highest for the spruce species (Engelmann spruce, white spruce, black spruce and red spruce), and lowest for certain pine species (lodgepole pine and jack pine) as well as western larch and eastern white cedar. When calculated by region (Figure 4.5b), the ecosystem carbon costs are highest for the western Canada provinces (Manitoba, Saskatchewan, Alberta and British Columbia) and lowest for Atlantic Canada (Prince Edward Island, Nova Scotia, New Brunswick and Newfoundland).

After 100 years of simulated harvest activity, the ecosystem carbon cost per m^3 (or tonne) of harvested wood is negative (net sequestration) for almost all landscapes (Figure 4.6). When the results are presented by species (Figure 4.6a and b), the median values range from -0.26 to -0.098 $tC{\cdot}m^{-3}$ (-0.48 to -0.27 $tC{\cdot}t^{-1}$), whereas the boxes range from -0.27 to -0.032 $tC{\cdot}m^{-3}$ (-0.50 to -0.090 $tC{\cdot}t^{-1}$) and outliers range from -0.28 to 0.15 $tC{\cdot}m^{-3}$ (-0.73 to 0.043 $tC{\cdot}t^{-1}$). The weighted mean based on harvest volumes ranged from -0.26 to -0.069 $tC{\cdot}m^{-3}$ (-0.48 to -0.19 $tC{\cdot}t^{-1}$). The species with the highest ecosystem carbon costs are spruce (such as white spruce, black spruce, Engelmann spruce), due to both higher rates of harvest in combination with lower temperatures and precipitation in the growing regions. In addition, the spruce species show more variation across regions than other species. For white spruce, this could be caused by the larger number of landscapes (24), whereas the larger variation within black spruce and Engelmann spruce could be as a result of large geographical range of those species. In other cases, such as jack pine (18 landscapes) which is harvested across Canada, the species are less sensitive to variations in temperature and precipitation. The interquartile ranges of the remaining species are clustered together. Despite these few trends, the spread of data shows that the region of origin of the species could be important information for pinpointing a more precise ecosystem carbon cost.

When the results are presented by region, the statistical spread of the data shifts somewhat (Figure 4.6c and d). The median values per province range from -0.22 to -0.059 $tC \cdot m^{-3}$ (-0.55 to -0.15 $tC \cdot t^{-1}$), whereas the boxes range from -0.24 to -0.030 $tC \cdot m^{-3}$ (-0.66 to -0.083 $tC \cdot t^{-1}$) and outliers range from -0.28 to 0.015 $tC \cdot m^{-3}$ (-0.73 to 0.043 $tC \cdot t^{-1}$). The weighted mean based on harvest volumes ranged from -0.18 to -0.034 $tC \cdot m^{-3}$ (-0.50 to -0.085 $tC \cdot t^{-1}$). For two Atlantic Canada provinces (New Brunswick and Nova Scotia), most of the interquartile range (representing the middle 50% of values) showed more net sequestration than any of the other provinces. This is related to both very low levels of harvest in the New Brunswick and Nova Scotia landscapes with their higher mean annual temperatures and precipitation levels, are characteristic of coastal forests and show the most negative values of ECC at 100 years illustrating the strong carbon sink per unit of harvested wood. On the other end of the spectrum, Manitoba and Saskatchewan landscapes with their low mean annual temperatures and precipitation show either less sequestration (small negative values) or slightly net emissions (small positive values) per unit of wood harvested. All data points shown in Figure 4.6 are available in table form in the Supplementary Material.

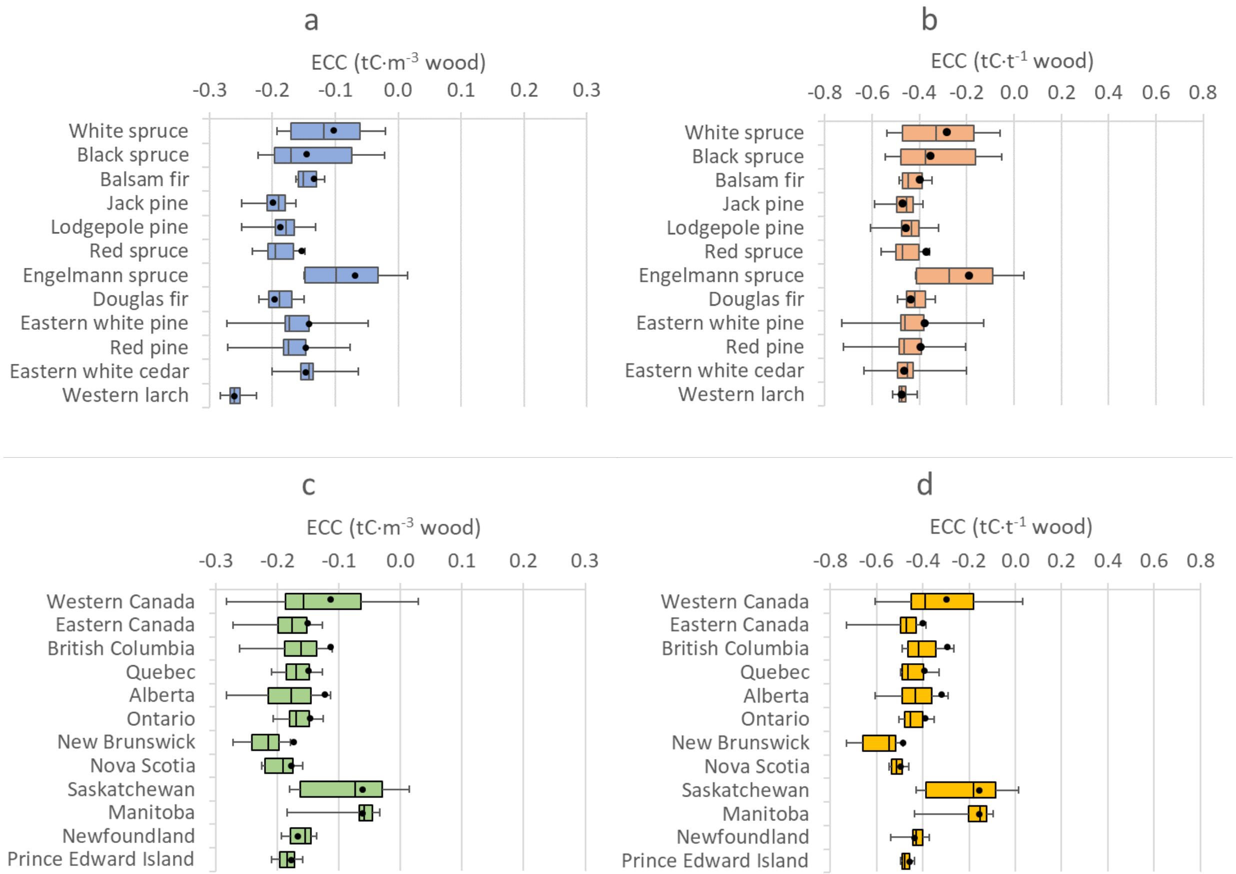

Figure 4.6 – Ecosystem carbon costs at year 100 of simulation, for a) per tree species in $tC{\cdot}m^{-3}$ wood harvested, b) per tree species in $tC{\cdot}t^{-1}$ wood harvested, c) per region in $tC{\cdot}m^{-3}$ wood harvested, d) per region in $tC{\cdot}t^{-1}$ wood harvested. The carbon content of the dry wood ranges from 0.175 $tC{\cdot}m^{-3}$ wood harvested (Eastern white cedar) to 0.300 $tC{\cdot}m^{-3}$ wood harvested (Western larch). The lower and upper error bars show the minimum and maximum values, while the lower bound of the box shows first quartile value, the middle line the median value and the upper bound the third quartile value. The round markers indicate the weighted mean values according to approximate annual wood harvest volumes.

Table 4.1 – Ecosystem carbon costs at year 100 of simulation, in tC·m^{-3} wood harvested

	mean	std dev	wgt mean by harvest volume	mini-mum	Q1	median	Q3	maxi-mum	C in wood
Western larch (Larix occidentalis)	-0.257	0.024	-0.261	-0.283	-0.267	-0.261	-0.251	-0.225	0.300
Eastern white cedar (Thuja occidentalis)	-0.141	0.038	-0.147	-0.200	-0.155	-0.142	-0.135	-0.064	0.175
Red pine (Pinus resinosa)	-0.171	0.052	-0.148	-0.271	-0.183	-0.175	-0.147	-0.076	0.201
Eastern white pine (Pinus strobus)	-0.162	0.059	-0.141	-0.272	-0.179	-0.173	-0.142	-0.047	0.200
Douglas fir (Pseudotsuga menziesii)	-0.187	0.036	-0.197	-0.222	-0.205	-0.189	-0.169	-0.150	0.244
Engelmann spruce (Picea engelmannii)	-0.083	0.081	-0.069	-0.149	-0.149	-0.098	-0.032	0.015	0.195
Red spruce (Picea rubens)	-0.191	0.029	-0.154	-0.232	-0.207	-0.195	-0.167	-0.148	0.218
Lodgepole pine (Pinus contorta)	-0.182	0.032	-0.187	-0.249	-0.195	-0.178	-0.165	-0.131	0.215
Jack pine (Pinus banksiana)	-0.194	0.023	-0.199	-0.249	-0.208	-0.191	-0.180	-0.162	0.222
Balsam fir (Abies balsamea)	-0.144	0.020	-0.134	-0.163	-0.158	-0.150	-0.130	-0.117	0.175
Black spruce (Picea mariana)	-0.138	0.074	-0.130	-0.223	-0.197	-0.171	-0.074	-0.022	0.220
White spruce (Picea glauca)	-0.116	0.060	-0.104	-0.193	-0.171	-0.118	-0.061	-0.021	0.195
Prince Edward Island	-0.184	0.021	-0.046	-0.209	-0.197	-0.184	-0.172	-0.160	
Newfoundland	-0.162	0.022	-0.166	-0.193	-0.179	-0.154	-0.145	-0.135	
Manitoba	-0.072	0.044	-0.049	-0.184	-0.067	-0.059	-0.046	-0.041	
Saskatchewan	-0.095	0.063	-0.034	-0.180	-0.162	-0.074	-0.030	-0.021	
Nova Scotia	-0.193	0.028	-0.178	-0.225	-0.220	-0.191	-0.175	-0.155	
New Brunswick	-0.218	0.041	-0.174	-0.272	-0.242	-0.215	-0.197	-0.150	
Ontario	-0.166	0.026	-0.108	-0.207	-0.181	-0.169	-0.148	-0.117	
Alberta	-0.173	0.061	-0.068	-0.283	-0.215	-0.178	-0.146	-0.048	
Quebec	-0.165	0.030	-0.114	-0.210	-0.185	-0.169	-0.148	-0.081	
British Columbia	-0.151	0.072	-0.051	-0.262	-0.189	-0.162	-0.137	0.015	
Eastern Canada	-0.176	0.034	-0.115	-0.272	-0.199	-0.176	-0.152	-0.081	
Western Canada	-0.138	0.074	-0.055	-0.283	-0.187	-0.157	-0.064	0.015	

Forestry management is relatively recent in Canada, as compared to Fenno-Scandinavia, where forests have been extensively exploited for centuries (Kurz et al., 2013). However, some regions across Canada, such as the Maritime provinces, Eastern Ontario and Western Quebec were settled by Europeans earlier than other parts of Canada and were subjected to wood harvest prior to the 20^{th} century (Kurz et al., 2013). Those areas would have forestry management legacies of 100 years or more, with the ecosystem carbon cost per unit of wood harvested at or nearing its steady-state and at net sequestration (see Figure 4.4). For other regions with a more recent forest management history, our results suggest that the ecosystem carbon costs attributed to wood harvesting would result in ecosystem carbon costs higher than the value at 100-years of harvest.

The reference natural disturbance used in all scenarios was wildfire, which is the most widespread and frequent disturbance type in most of Canada's forests. However, other natural disturbance types, most notably insect outbreaks, and locally, windstorms, have also had particularly large impacts on the net emissions of forests between 2002 and 2008 (Stinson et al., 2011), potentially turning Canada's forests in net carbon sources. In fact, climate change itself, through a feedback loop could increase the incidence of both wildfire and insect outbreaks (Kurz et al., 2008a). The complex modelling involved in forecasting both climate scenarios and future disturbances in Canadian forests, are out of the scope of this paper but should be considered in future research (but see (Boucher et al., in press)).

The choice of methodological approaches also has an influence on the overall ecosystem carbon costs. A landscape approach was chosen in order to be able to model the forestry carbon dynamics when the specific site of the forest and provenance of harvested wood is unknown. It also allowed for the inclusion of fire disturbance and thus modelled forests were subjected to very small but constant rates of annual fire and harvest disturbances. A similar exercise could have been accomplished using a stand-level perspective, but the detail required to model specific stands would have limited the geographical scope of the results. Also, although forest residues from harvest are typically left on-site in Canada (Thiffault et al., 2015), these could also be collected and thus would be considered a co-product of wood harvesting. The utilisation of forest residues for bioenergy, for example, could influence how the ecosystem carbon costs are allocated and thus the result of the ecosystem carbon costs per m^3 of harvested wood.

This work represents a first attempt at modelling the ecosystem carbon costs of harvesting wood across multiple species and regions at the product level. In modelling the forestry ecosystems for most commercially important species across Canada, much of the input forest inventory data was developed using more macro level national forest inventory data and peer-reviewed models. While this data allowed for broad-reaching coverage of most commercial wood, it does have limitations in terms of not having a finer level of detail and granularity that would be expected from the study of a particular forest stand. The model makes use of a theoretical yield curve, which gives a reasonable estimate of the annual biomass accumulation. However empirical forest inventory data for a specific forest stand would almost inevitably better reflect the biomass volume of the forest.

Another limitation is the level of aggregation chosen for the landscapes, across which the temperature and precipitation values have been averaged. Smaller landscapes that more specifically reflect the provenance of harvested wood would allow for more representative mean temperature and precipitation values for the yield curve equations. Furthermore, the wildfire and harvest disturbances are aggregated by ecozone, without considering how those disturbances could affect species differently, while smaller homogeneous fire regimes zones have been shown to better represent the distribution of fire regimes across Canada (Boulanger et al., 2014). Finally, the harvested wood volumes used for calculating the weighted means were calculated, in the absence of more specific statistical data, by combining data from different sources and by using some assumptions to handle data gaps.

The results counter the prevailing assumption of forestry products having a net zero biogenic carbon flux and indicates that carbon footprint and LCA studies should also include the impacts from harvest activities that can further influence the carbon cost of harvested wood. The overall carbon balance of a harvested forest should also consider the carbon content of the wood itself (0.175 - 0.300 $tC{\cdot}m^{-3}$ dry wood). Typically, LCAs of wood products are done from a static point of view. The authors plan to demonstrate how the ecosystem carbon cost results can be integrated into a temporally dynamic life cycle inventory in an upcoming study.

4.2.3.3 Validation of results using flux tower measurement data

The validation of the modelling results was carried out by comparing modelled C fluxes of black spruce and jack pine landscapes with flux tower measurement data (Figure 4.7).

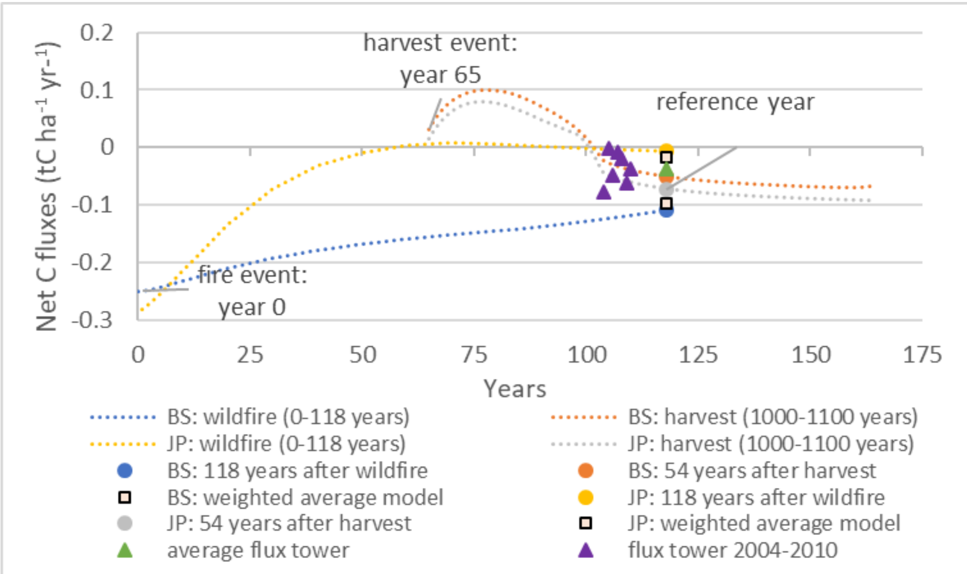

Figure 4.7 – Modelled scenarios (baseline and harvest) for black spruce (Pinea mariana) and jack pine (Pinus banksiana) vs. flux tower data. BS = black spruce (Picea mariana), JP = jack spine (Pinus banksiana), wildfire (0-118 years) = landscape affected by annual fire starting from fire event in 1900 to present, harvest (1000-1100 years) = landscape affected by 100 years of annual fire and harvest following wildfire, 118 years after wildfire = point at present day 118 years after fire, 54 years after harvest = point at present day 54 years after harvest, weighted average model = weighted average of baseline and harvest scenarios, average flux tower = average of flux tower data from 2004-2010, flux tower 2004-2010 = annual averages from 2004-2010. Flux tower data from Fluxnet Canada / Canadian Carbon Program (Coursolle et al., 2012).

The annual flux tower (2004-2010) and the average flux tower data lie at the midpoint between the weighted average of the black spruce and the jack pine modelled results (Figure 4.7). This demonstrates that despite inherent differences in spatial scales between the two result types, at least for black spruce and jack pine in the Quebec Boreal Shield, the models reflect the ecosystem carbon costs that have been measured at flux tower test sites. As such, the flux tower data provides at least a partial validation of the modelled results. A complete validation of the modelled landscapes would require having access to widespread flux tower data. However, this would be difficult to obtain presently due to the limited geographical scope of the flux tower sites and available length of the measurement record.

4.2.4 Conclusion

The ecosystem carbon costs per m^3 of wood harvested in most forest landscapes in Canada shows net sequestration infirming the carbon neutrality assumption. The weighted mean ecosystem carbon costs from a 100-year-old harvested forest, based on harvested volume by species, range from -0.26 to -0.069 $tC \cdot m^{-3}$. The spruce species tend to have higher and more variable ecosystem carbon cost scores, while the remaining species tend to have lower scores and less variability. By province, the weighted mean ecosystem carbon costs range from -0.18 to -0.034 $tC \cdot m^{-3}$. The Atlantic provinces (New Brunswick and Nova Scotia, in particular) show the most sequestration, whereas the ecosystem carbon costs are highest in the Prairies (Manitoba and Saskatchewan). The mean *time to ecosystem cost neutrality* for each species ranges from 16-60 years. These results show that sustained wood harvest in Canadian forests at current wildfire and harvest rates result in net sequestration benefits. Flux tower measured data at given test sites confirms that simulated results reflect the ecosystem carbon costs. As such, the results of this research work show that harvesting softwood tree species at current rates in Canadian forests, mostly has net carbon sequestration on the forest ecosystem.

Though carbon dynamics of forest management have long been considered in forestry research, this has to the authors' knowledge, not yet been extended to life cycle assessment. Despite the knowledge in the LCA community that biogenic carbon should not be considered neutral, the typical assumption has been that that the carbon sequestered in wood during its growth is the only carbon that has been sequestered, i.e. not considering the effects of wood harvest on the forest ecosystem. By this research work we provided evidence that in addition to account for the sequestration of the carbon embodied in wood, a wood product life cycle assessment should also account for the ecosystem carbon cost.

This research work also demonstrates the feasibility of using a forest carbon budget model to generate regionalised *cradle-to-gate* inventories of forest ecosystem carbon dynamics for harvested wood products across Canada. Together these inventories form a database covering 12 softwood tree species across 10 provinces of the Canadian boreal forest. The database could be used as is within decision-making tools, such as building information models for designing green buildings. These data can also be used as a part of a *cradle-to-grave* life cycle assessment by converting the ecosystem carbon costs into CO_2 emissions and expressing them in a life cycle inventory along with the carbon fluxes occurring at other life cycle stages. In doing so it will be important to evaluate the choice of ecosystem carbon cost values to use within a dynamic life cycle assessment, in order to ensure an equitable allocation of sequestration benefits to the wood users.

4.2.5 Acknowledgements

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CHAPTER 5 ARTICLE 2: TEMPORALLY DIFFERENTIATED BIOGENIC CARBON ACCOUNTING OF WOOD BUILDING PRODUCT LIFE CYCLES

5.1 Introduction to Article 1

This following manuscript details how biogenic carbon is tracked from harvested roundwood logs through wood product manufacturing, building life and end-of-life phases, by considering the carbon fluxes between the wood product and the atmosphere as temporally differentiated life cycle inventories. The work for this article was done in collaboration with Michael Magnan and Werner Kurz at the Pacific Forestry Centre of the Canadian Forest Service. The authors of this article are Marieke Head, Michael Magnan, Werner Kurz, Annie Levasseur, Robert Beauregard and Manuele Margni. The manuscript was submitted on January 29th, 2019 to the *International Journal of Life Cycle Assessment*. Supplementary Material has been produced to be published with this article. Since most of the Supplementary Material consists of a large number of carbon flux profile data, only the input data is provided in this dissertation (see Appendix C).

5.2 Manuscript

5.2.1 Introduction

The 2003 Good Practice Guidance of the Intergovernmental Panel on Climate Change (IPCC) introduced methodologies for the estimation and reporting of carbons stocks and fluxes in harvested wood products (IPCC, 2003). Until then, it was assumed that the sum of carbon additions to the harvested wood products pools from current harvest was equal to the sum of carbon losses from the wood products that were harvested in prior years and that the size of the total HWP carbon pool was constant (IPCC, 1997). Instead of tracking the details of the fate of harvested carbon, the IPCC made the simplifying assumption that inputs are equal to outputs, thus effectively treating the carbon from wood harvest as instantly oxidised, ignoring any time delays and storage benefits associated with harvested wood products (HWP). Carbon storage in harvested wood products in Canada delays the emission of greenhouse gases (Apps et al., 1999), and results in lower carbon biogenic carbon emissions from Canada's forest products industry in national greenhouse gas reporting (ECCC, 2018).

In life cycle assessment (LCA) of individual products, it is necessary to determine the impact of harvest on overall emissions. One simplifying assumption is net biogenic carbon neutrality, which assumes that carbon harvested is off-set by a similar amount of carbon that is regrown in the forest resulting in a net zero impact on the greenhouse gas balance in the forest (Johnson, 2009; Searchinger et al., 2009). However, there are several ways in which the carbon contained in harvested biomass is not necessarily cancelled out by an equal sequestration of carbon dioxide in biomass regrowth. The carbon neutrality assumption does not consider the time needed to regrow the forest and offset carbon emissions, as it may take years to decades to counteract the carbon that has accumulated in the atmosphere since the release of a greenhouse gas (Cherubini et al., 2011; Helin et al., 2013; Lemprière et al., 2013; McKechnie et al., 2011; Zanchi et al., 2010). This time delay is very scale dependent: in the extreme case of a single stand, regrowth may require decades to centuries, while at the landscape level, annual regrowth may balance all harvest losses. The biogenic carbon balance should be better accounted for in LCA, by considering the carbon uptake and emissions throughout every life cycle stage from the forest to the end-of-life of a product.

Several authors have highlighted the need to incorporate the biogenic carbon of wood products in product LCA (Brandão & Levasseur, 2011; Buyle et al., 2013; Helin et al., 2013; Lemprière et al., 2013; Xie, 2015). Most LCA guidelines and standards covering wood products now also tend to stipulate specific measures for biogenic carbon accounting. Of eight surveyed LCA guidelines and standards on greenhouse gas emissions and wood products (BSI, 2012, 2014a, 2014b; EPD-Norway, 2013; FPInnovations, 2013; ISO, 2007, 2013; NEN, 2014), all but ISO 14067 (ISO, 2013) take the position that biogenic carbon uptakes and emissions should be accounted for in LCA. However, they only provide the very simplified assumption that the uptake of carbon in forest should be considered a negative emission (i.e. removal) and that the release of carbon should be considered a positive emission. In the case of certain long-life products such as building materials, the carbon contained in the wood is sequestered through long product lives, which can amount to delaying emissions for several decades or centuries in some cases. In addition, the long-term storage of carbon in landfills has been identified as having potential climate benefits (Wang et al., 2011). One of the main critiques of the neutrality assumption is that it ignores the questions of temporary carbon storage and delayed emissions, which can result in potential climate benefits. The storage of carbon in products is currently not considered in many LCA studies as there has been no consensus on how to account for it (Brandão et al., 2013).

Forestry science has been considering the carbon balance of harvested wood products throughout their use phases for a few decades (Apps et al., 1999). Brunet-Navarro et al. (2016) reviewed 41 wood product models and classified them based on their functionality and performance. A wood product model can either estimate and evaluate the fate of biogenic carbon in different wood product classes or it can be used to estimate the carbon emissions from wood product use and end-of-life (Brunet-Navarro et al., 2016). From a single wood product perspective, the latter is most relevant and requires a model that can track carbon, including the allocation of co-products, the consideration of time and the ability to handle various end-of-life treatment options. Such a carbon accounting model could be used in life cycle assessment to consider biogenic carbon storage and fluxes in wood products.

The objective of this study is to improve the biogenic carbon accounting of long-life wood products in LCA, by tracking biogenic carbon from harvested roundwood logs through wood product manufacturing, building life and end-of-life phases, by considering the carbon fluxes as an inventory between the wood product and the atmosphere through time. This tracking is also used to test dynamic inventories through the use of policy scenarios that increase recycling rates and landfill gas collection. This improvement of biogenic carbon accounting in wood products will provide building designers with a more accurate portrait of the climate impacts of wood products, and support more informed decisions related to material selection. While the proposed method is applicable to any geographical region, in this paper the method is applied to the Canadian building sector. To cover the products commonly used for structural elements in the Canadian building sector, seven types of softwood products, across 12 Canadian provinces and territories with building lifetimes varying from 0 to 150 years, are considered.

5.2.2 Methods

5.2.2.1 Wood product model

A team at the Canadian Forest Service (ECCC, 2018) developed the Carbon Budget Model Framework for Harvested Wood Products (CBM-FHWP). CBM-FHWP allows for the dynamic construction, validation, simulation and analysis of a system that describes and quantifies the flow of carbon in harvested wood products through time and space. Within this flexible framework, users must define all aspects of the models they create, which includes the definition of the space (i.e. the origin of the harvested wood and the region of product use), the carbon stocks, the physical state, the mass of carbon, the flows as well as the model time step size. Until now the model has been mostly used for tracking the carbon of harvested wood products from a macro perspective for different geographical regions (Dymond, 2012). CBM-FHWP is currently most extensively used by the Canadian Forest Service to calculate the contribution of harvested wood products to Canada's greenhouse gas balance for the national inventory reports submitted to the UNFCCC every year (ECCC, 2018). The specific perspective of this research project, its focus on individual wood products throughout their life cycles, will be a new application of the modelling framework.

5.2.2.2 Model scope and system boundaries

In all, seven wood product models (lumber, plywood, glulam, oriented strand board (OSB), laminated veneer lumber (LVL), cross laminated timber (CLT) and I-joists), which correspond to products that would commonly be used in the construction of buildings in Canada, were built and simulated in CBM-FHWP. The models consider carbon from the roundwood log delivered to the sawmills, the sub-division of logs into products, the use of the co-products (use in bioenergy, external manufacturing or disposal), the storage of the wood carbon over the lifetime of the wood product, and the end-of-life processing, including a half-life approach for modelling the fate of the degradable carbon in landfills (Figure 5.1). These models were run for building life years from 0-150 years[3] for 12 Canadian provinces and territories (excluding Nunavut, for a total of 2352 cases). The model is focused on the biogenic carbon contained in the roundwood log input required for a

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[3] 1-10 years: annual increments, 10-50 years: 5-year increments, 50-150 years: 10-year increments

given wood product, and as such other wood product life cycle emissions are already included in life cycle inventory databases and thus they are not considered in this study.

The focus of this research is on creating temporally differentiated biogenic carbon profiles for the life cycle of wood used in buildings. As such, the outputs of this work are in the form of life cycle inventories that can be subsequently used in life cycle impact assessment.

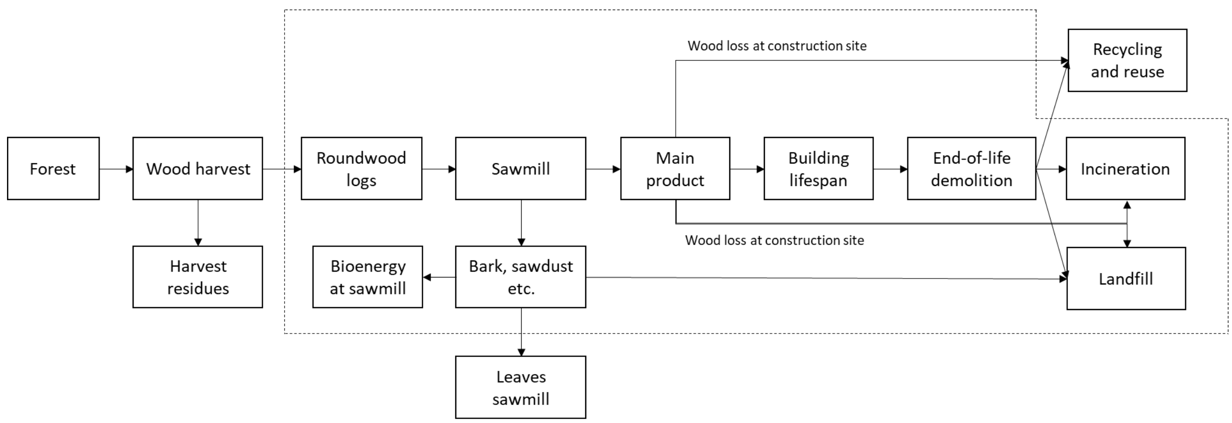


Figure 5.1 – System boundaries for wood product carbon flows. The processes contained within the dotted line are included in the model. The forest ecosystem and upstream forest harvest activities are developed in a previous study (Head et al., 2019a). Our implementation of the model treats the "Leaves sawmill" and "Recycling and reuse" processes as being outside of the system scope. The figure only includes the biogenic carbon contained within the wood

5.2.2.3 Creating the model files and parameter definition

In CBM-FHWP, products are modelled as a series of text file line entries following the six dimensions (space, stocks, physical states, mass, flows and time) (Magnan, 2013). At the flow level, *pools* and *events* are defined such that carbon can move through each life cycle and each co-product in succession. The partitioning of co-products at each event is done as proportions of a total of 100%.

Seven separate models were developed for each of the seven wood products, each model using the lumber model as a template. Each of the models begins with roundwood logs as input but has different co-product outputs and fates at the manufacturing (sawmill) phase. Mass balances of the seven wood products were obtained from Athena Sustainable Materials Institute reports (ASMI, 2012a, 2012b, 2012c, 2012d, 2013a, 2013b, 2013c, 2018a, 2018b, 2018c, 2018d). The mass balances are provided as green wood and oven dried wood, and therefore a unit conversion step was required to account for carbon content. The carbon content for each pool was then calculated as a proportion of all pools flowing in or out of an event (Table 5.1).

Table 5.1 – Co-product outputs of sawmills for seven wood product types (% mass flows). CLT= cross-laminated timber, glulam= glue laminated timber, I-joist= engineered wood joist, LVL= laminated veneer lumber, OSB= oriented strand board, off-spec= off-specification, by-products= unspecified co-products

	Lumber	CLT	Glulam	I-joist	LVL	OSB	Plywood
Main product	43.1%	54.0%	50.3%	55.0%	47.3%	79.3%	49.8%
Bark	8.9%	9.0%	8.7%	6.7%	11.3%		
Planer shavings	6.3%		2.9%	2.1%			
Sawdust	5.6%	4.4%	4.8%	1.9%			
Pulp chips	34.5%	32.5%	32.5%	21.1%	28.7%		19.4%
Trim ends	0.6%		0.3%	0.2%			
Chipper fines	0.2%	0.2%	0.2%	0.1%			
Wood waste	0.7%		0.3%	0.3%		0.3%	
Off-spec				2.4%	2.6%		
Peeler cores				3.4%	10.1%		9.0%
Wood for hogfuel				5.8%		17.4%	21.5%
By-products				1.0%		2.9%	
Veneer							0.3%
Total	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%

Beyond the manufacturing phase, the structure of the models was identical for the use phase and end-of-life waste management of the main wood product. At the building construction site, the lumber is again divided into two co-products: the main building product and the waste occurring at the construction site. The amount of waste occurring at the construction site is taken from Wang et al. (2013) as a waste factor of 10% at construction sites in North America. The remaining carbon (90%) is assumed to be embedded in the building. The wood remaining in the building is modelled as 100% in the building for every year up until the designed building life year, at which point 0% remains in the building and the carbon is moved to end-of-life treatment.

Both the construction site waste and the building demolition waste are treated via the same four end-of-life treatment options: landfilling, incineration, recycling and use as firewood. In Canada, the majority of construction wood waste is landfilled, with a small proportion being recycled despite specific municipal and provincial policies discouraging the landfilling of construction waste (MDDEP, 2011). The proportions of waste going to different treatment options were based on an Environment Canada report on construction waste (Kelleher Environmental & Guy Perry and Associates, 2015) (Table 5.2).

Table 5.2 – End-of-life fate of clean wood (lumber) and composite/engineered wood (CLT, glulam, I-joist, LVL, OSB, plywood). "Construction" refers to waste occurring at the construction site at the beginning of a building's life, whereas "demolition" is waste occurring at the end of a building life

Jurisdiction	Solid wood				Composite/engineered wood			
	Construction		Demolition		Construction		Demolition	
	Recycled	Landfilled	Recycled	Landfilled	Recycled	Landfilled	Recycled	Landfilled
Canada	18%	82%	21%	79%	26%	74%	23%	77%
British Columbia	30%	70%	42%	58%	41%	59%	44%	56%
Alberta	8%	92%	9%	91%	13%	87%	10%	90%
Saskatchewan	1%	99%	1%	99%	2%	98%	1%	99%
Manitoba	4%	96%	4%	96%	6%	94%	5%	95%
Ontario	16%	84%	17%	83%	24%	76%	19%	81%
Quebec	21%	79%	27%	73%	30%	70%	29%	71%
New Brunswick	2%	98%	2%	98%	4%	96%	2%	98%
Nova Scotia	40%	60%	47%	53%	51%	49%	49%	51%
Prince Edward Island	0%	100%	0%	100%	0%	100%	0%	100%
Newfoundland	0%	100%	0%	100%	0%	100%	0%	100%
Northwest Territories	0%	100%	0%	100%	0%	100%	0%	100%
Nunavut	0%	100%	0%	100%	0%	100%	0%	100%
Yukon	0%	100%	0%	100%	0%	100%	0%	100%

5.2.2.4 Treatment of outputs from system

There are a few places in this model where outputs are utilised in other processes, such as the production of bioenergy and the use in other material life cycles (Figure 5.1). When a process produces more than one useful product, it can be termed multifunctional. As such, only the flows directly related to that product in question must be accounted for in the calculation of its environmental impacts. ISO 14044 recommends a specific hierarchy for solving for multifunctional processes (European Commission, 2010b). As a first priority, subdivision should be attempted, by dividing black box processes into single operation unit processes. Second, substitution should be attempted by either expanding the system boundaries to include another function that is not within the product system or by subtracting an alternative production process. Third, if neither subdivision or substitution is possible, the allocation of process burdens can be done by partitioning the process flows according to some chosen criterion. There is a preference for a physical means of allocation such as allocation by mass, energy content, stoichiometry, etc. However, in some cases allocation by economic value is appropriate.

The literature shows that the methodological choices surrounding multifunctional systems, particularly wood and forestry products, can have a significant impact on LCA scores (De Rosa et al., 2018; Sandin et al., 2014; Sandin et al., 2015). For this research work we applied the guidance provided by the European EN16485 product category rule standard (Round and sawn timber — Environmental Product Declarations —Product category rules for wood and wood-based products for use in construction), which recommends allocating biogenic carbon according to the carbon content of the product.

A variety of different co-products were modelled as outputs at the manufacturing stage for the modelled wood products, including bark, shavings, sawdust, pulp chips, trim ends, chipper fines, peeler cores and off-specification product. The Athena Reports (ASMI, 2012a, 2012b, 2012c, 2012d, 2013a, 2013b, 2013c, 2018a, 2018b, 2018c, 2018d) specify end uses of all these co-products in a Canadian context, and we further streamlined these into three different fates:

Leaves sawmill: This refers to co-products that are sold off to other facilities to be used as a raw material. Given that these co-products (and their carbon content) are used by third parties, the carbon in the co-product is allocated to other systems (cut-off from the main product system) and also shares the burden of the processes with the main product.

Landfilling: The landfill fate is modelled as an average Canadian landfill as the Athena reports have not specified the geographic locations of all sawmills. The specifics of the treatment of landfills will be described with the other end-of-life options for the wood emerging from building demolition. The carbon released from landfilling sawmill co-products is allocated to the main wood product.

Bioenergy: The co-products can also be used for bioenergy at the sawmills. The bioenergy transforms the carbon embedded in the co-product into CO_2 and (negligible) CH_4 emitted from the combustion of the material. Carbon emitted through the combustion of co-products for bioenergy at the sawmill is allocated to the main wood product.

The proportions of each co-product that is going to each fate is provided in the Supplementary Material.

5.2.2.5 End-of-life waste management

Landfilling

Wood accounts for around 7% of all unrecovered waste sent to landfill in Canada (Howe et al., 2013). Conditions within modern landfills are predominantly anaerobic due to their design both in preventing moisture and precipitation from entering the landfill and in the use of cover materials to prevent exposure to air. Typically, only a minimal amount of aerobic decomposition occurs, in cases when waste is not immediately covered (Larson et al., 2012). Although the anaerobic decomposition of organic materials emits greenhouse gases (Larson et al., 2012), several studies (Barlaz, 2006; Chen et al., 2008; Micales & Skog, 1997; Wang et al., 2011; Wang et al., 2013; Ximenes et al., 2015; Ximenes et al., 2008) demonstrate that wood degrades very slowly in landfill sites. Since wood consists of a complex lignin matrix that integrates cellulose and hemicellulose and the conditions of most landfills are anaerobic, only a small proportion of wood is degraded.

According to Micales, and Skog (1997), it is estimated that only 0-3% of carbon contained in wood is emitted as a gas at landfill sites. Wang et al. (2011) compared wood degradation of several types of wood products in laboratory-scale landfills for 440-1347 days until methane production could no longer be detected. For most wood types, degradation calculated as carbon conversion percentages, ranged between 0-7.9% (degradation for hardwood OSB was 19.9%). More recently, Wang et al. (2013) found through field studies in the United States that the degradation of wood in a landfill is dependent upon the type of wood product. They found that after leaving wood in the landfill for 1.5-2.5 years, 5-23% of the carbon contained in the wood is degraded for engineered wood such as oriented-strand board (OSB), whereas for hardwood and softwood lumber very little (0-9%) of the carbon was degraded. In a study examining wood degradation in landfills in

Australia, Ximenes et al. (2015) found that temperate species experienced only 0-8% carbon loss after 16-44 years of being recovered.

Taking into consideration the variation of wood degradation in landfills found in the literature, which varies across wood types, wood species and local climatic conditions, we elected to make use of the landfill models supported by the CBM-FHWP. The CBM-FHWP models the degradation of carbon in landfills using the first order decay method, a method used by the IPCC (IPCC, 2006c):

Eq. 1 $$DDOCm = DDOCm_0 \cdot e^{-kt}$$

Where, t is time (years), $DDOCm$ is the mass of the degradable organic carbon that will decompose under anaerobic conditions in a landfill at time t, $DDOCm_0$ is the mass of DDOC at time 0, k is the decay rate constant ($years^{-1}$).

Since the decay rate constant, k, is influenced by several factors such as climate, landfill engineering and waste composition, it is difficult to obtain values that are specific to both province/territory and wood type (lumber, OSB, etc.) (Krause et al., 2016). Two sets of k values were used to model the degradation of carbon in landfills, 1) a value of 0.03 $years^{-1}$ for average wood landfilled in Canada and 2) specific k values for each province and territory ranging from 0.003-0.083 $years^{-1}$ (ECCC, 2017). An overview of the values is shown in Table 5.3. The degradable organic carbon has three possible fates for the resulting greenhouse gas emissions: capture of CH_4 without flaring (to be used as energy) – 16.8%, capture of CH_4 with flaring and direct emission of CO_2 (17.2%) and landfill gas to the atmosphere (66% of which 90% CH_4, 10% CO_2). The proportion of carbon emitted as CO_2 and CH_4 emissions from the combustion of landfill gas for energy production (capture without flaring) was modelled as 99.995% and 0.005%, respectively (IPCC, 2006b). Carbon emissions from capture with flaring are 99.7% CO_2 and 0.3% CH_4 (ECCC, 2017).

Table 5.3 – Decay constant, k ($year^{-1}$) for different regions and landfill types (ECCC, 2017)

Region	decay constant ($years^{-1}$)
Degradable wood in MSW landfills	
British Columbia	0.083
Alberta	0.012
Saskatchewan	0.012
Manitoba	0.019
Ontario	0.046
Quebec	0.059
New Brunswick	0.059
Nova Scotia	0.075
Prince Edward Island	0.061
Newfoundland & Labrador	0.078
Yukon	0.002
Northwest Territories	0.003
Degradable wood in wood waste landfills	
Canada, average	0.03

Recycling

The possibility for recycling construction wood waste in Canada is highly dependent upon the population density of the city or region. Larger urban centres are more likely to have a higher capacity for construction waste recycling than smaller cities or rural areas, where the economics of recycling these waste materials is unfavourable (Kelleher Environmental & Guy Perry and Associates, 2015). Recycling rates are also dependent upon the end-of-life classification of the wood. Solid (or untreated) wood tends to be recycled and have higher market values than engineered and treated woods that can contain adhesives, paints and preservatives (Kelleher Environmental & Guy Perry and Associates, 2015). However, the consequences for the carbon accounting of the reuse and recycling of wood can become complex, due to the way in which the emissions benefits of recycling are treated.

In terms of the climate implications of wood recycling, in particular, a few authors have published on this topic. In their study on the LCA of particle board, Wilson (2010) considers carbon storage in the carbon balance and predicts that recycling processes will keep carbon out of the atmosphere even longer than the service life of the initial product. However, Werner et al. (2006) conclude in their study on end-of-life alternatives of wood products that there is no method of modelling post-consumer wood that would account for all situations of wood use. Kim, and Song (2014) used system expansion to deal with recycled materials in particle board manufacturing, thus accounting for the avoidance of virgin materials through the use of recycled wood. They also calculated the carbon benefit of recycling based on the carbon storage of wood during the service life of the wood, as well as the extended period of storage attained through recycling. This calculation also accounts for the effects of progressively diminishing storage through material degradation as a result of several rounds of recycling. Although the approaches used for the treatment of wood recycling differ, it is clear that the recycling of wood can have significant implications in terms of biogenic carbon accounting.

The carbon content of the demolition wood sent to recycling is tracked, however in the model it is treated with a cut-off approach. The cut-off approach, whereby the subsequent fate of the recycled material is excluded from the scope of the system, was chosen for recycling in this study for a few reasons. First, the third parties purchasing the recycled wood material could be using it for a multitude of purposes and thus it could be substituting a variety of intermediary materials that would otherwise be made with virgin materials. Second, the timing of the ultimate disposal of the material as well as the number of product life cycles that the wood will be part of is unknown. Third, the carbon becomes part of another product, the “responsibility” for which belongs to that product life cycle. Finally, since the objective of this study was to provide temporally differentiated life cycle inventories, including the effects of subsequent life cycles would necessitate an impact assessment (LCIA) and thus go beyond the scope of this work. However, choosing a cut-off approach may have a few implications in terms of the carbon fluxes attributed to the primary product life cycle and how these carbon emissions are characterised as climate impacts. By not accounting for this carbon at the point that it leaves the primary product life cycle, certain attributes of the future material use are not considered. For example, the wood could be sold to a waste management company to be chipped and used as a daily landfill cover. Aside from the financial transaction having taken place, there is very little difference to this fate than if the material had simply been landfilled. Nevertheless, this approach has been chosen for its flexibility, as it would allow for solutions to the multifunctionality in post-processing calculations that could be added to the model results.

Incineration and use as firewood

Incineration only accounts for a very small proportion of waste management in Canada (Kelleher Environmental & Guy Perry and Associates, 2015), taking place in only a few municipalities. A quick survey of four specific municipalities, showed that the incinerators do not even accept construction and demolition waste. As such, 0% of wood waste is assumed to be incinerated in the model. However, in order to consider incineration either for additional regions outside of Canada or to investigate the impacts of incineration in Canada, incineration has been left as a waste management option in the model. The carbon emitted by the incinerator is modelled in this work to be mostly CO_2 (99.999905%) with a negligible amount of CH_4 (0.000095%) (Doka, 2016). Since the heat created by incinerators can be harnessed for energy purposes, some incinerators generate usable electricity or heat. If desired, this could be accounted for separately.

Another waste management option included in the model is the potential for wood waste to be used as residential firewood. While construction wood waste is not treated via this method throughout municipalities, the option is left in the model for potential marginal cases, such as in remote areas, where individuals use construction wood waste as firewood. The carbon emitted by using waste wood as firewood, modelled as emissions from residential conventional stoves and fireplaces for CO_2 (97% of C) and CH_4 (3% of C) (Environment Canada, 2004). The combustion of firewood creates heat, which can substitute directly or indirectly for other residential heat sources, such as electrical, oil or natural gas heating systems. This substitution can be accounted for outside of the FHWP model if this waste management is used.

5.2.2.6 Additional scenarios

In the context of buildings and construction, the eventual end-of-life phase of a building product can occur several decades in the future. In the meantime, proportions of waste going to different treatment options, and the waste management technology itself can evolve significantly, at which point the waste scenarios for demolished building materials is unknown. Sandin et al. (2014) examined the effects of future waste management scenario assumptions on the outcomes of environmental impacts of building materials. Their results suggest that the assumptions made about waste management scenarios of the future, such as type of disposal, level of technology and type of LCA approach (attributional vs. consequential), may have significant impacts in terms of the relative environmental impacts of the end-of-life phase of material alternatives.

Some additional policy scenarios were considered which involve varying a key parameter through time as a policy target is reached, starting from the year 2020. These scenarios use existing base cases with static parameters and add the carbon emissions related to material or fuel substitution.

The scenarios are defined as:

REC70%: This scenario models the 70% construction waste recycling target by 2025 set by the Quebec government (MDDEP, 2011). The recycling rate increases at the rate calculated based on construction waste recovery in Montreal, the city with the largest population in Quebec (City of Montréal, 2015).

LFG80%: This scenario models an 80% landfill gas capture rate, based on the 75% capture rate by 2020 set by the British Columbia government (Province of British Columbia, 2008). Given the explicit targets for 75% LFG capture by 2020, we have assumed that an average of 80% capture rate by 2030 is possible, given that there is an average of 7 years of life left in British Columbia landfills (Bonam, 2009).

The scenarios are run for only a select set of case parameters in order to show the full range of results without rerunning all 2352 cases (see Table 5.4).

Table 5.4 – Base case parameters for testing end-of-life policy scenarios for wood products

Parameter	Low	Medium	High
Engineered wood product content (EWP)	Lumber (0% EWP)		I-Joist (100% EWP)
Recycling rates (provinces)	Northwest Territories (0%)	Ontario (16-24%)*	Nova Scotia (40-51%)*
Landfill half-lives (provinces)	British Columbia (8.3 y^{-1})	Saskatchewan (57.8 y^{-1})	Northwest Territories (346.6 y^{-1})
Building life	1	50	150

* depending on type of wood product and whether wood is from a construction or a demolition site

5.2.2.7 Running the simulations

The default procedure for running results for a given set of parameters, involves using batch files that call up the CBM-FHWP software to convert the files to a usable format, import them and then run the simulation. In the case that several sets of parameters need to be run at once, CBM-FHWP is flexible enough to easily allow for customisable runs and thus can process several parameters sets at once. The scenarios were run and the results exported into a spreadsheet, to facilitate analyses. The results are calculated in terms of carbon transfers between HWP carbon pools, along a time scale from 0 to 300 years. The simulation period for the time scale presented was chosen for a few reasons. First, it allows for the modelling of the longest building lifespans (150 years) as well as sufficient time (an additional 150 years) for end-of-life emissions to propagate and eventually diminish to negligible quantities. Second, a consistent simulation period reduces the number of permutations of parameters that need to be simulated. The raw results are provided in the Supplementary Materials.

5.2.3 Results

Tracking carbon in different carbon pools through time can be complex, especially in combination with variable building products, provinces and buildings lifespans. The fate of the carbon contained in the original roundwood logs input into the sawmill is illustrated through all carbon pools over 300 simulation years for four different building lifespans (Figure 5.2).

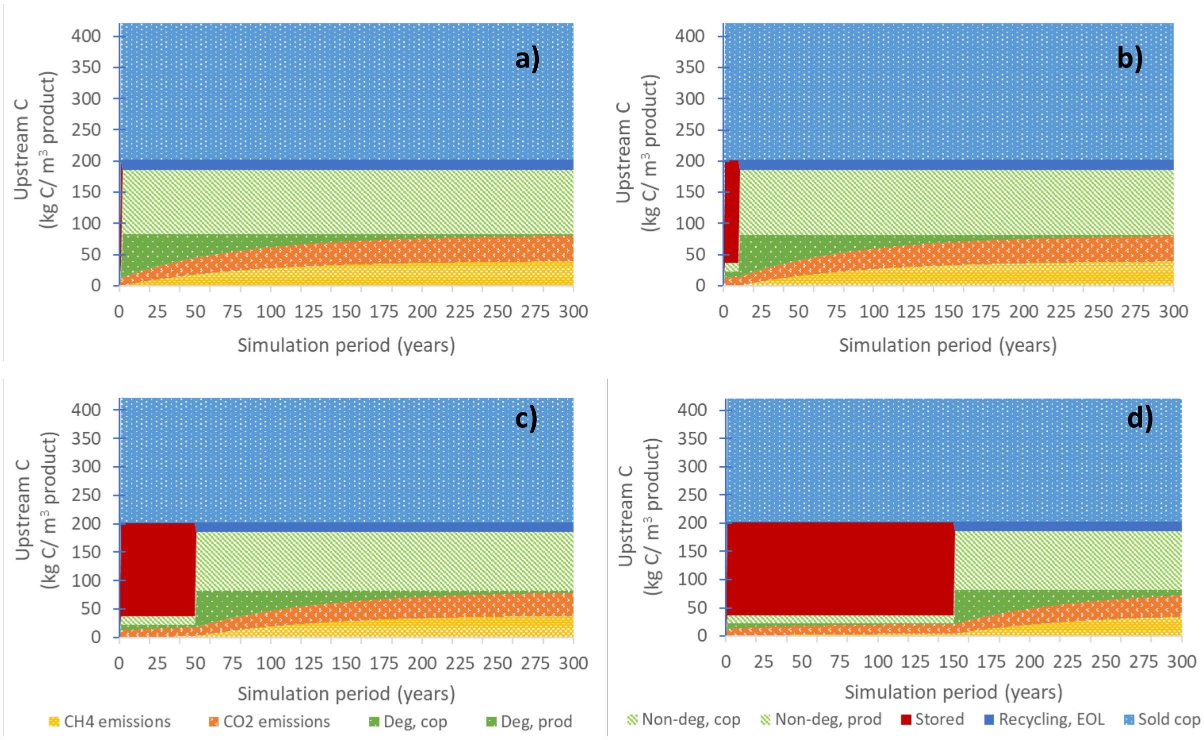


Figure 5.2 – Change in carbon pools over time, for 1 m^3 lumber, Alberta, for four buildings lifespans (a) = 1 year, b) = 10 years, c) = 50 years, d) = 150 years). Deg, prod/cop= degradable portion of carbon in landfilled lumber main product/sawmill co-products, Non-deg, prod/cop= non-degradable portion of carbon in landfilled main product/sawmill co-products. Stored= carbon stored in building, Recycling, EOL= recycling of wood at end-of-life, Sold cop= co-products at sawmill sold to third parties

Roughly 50% of the carbon of the roundwood log needed for 1 m^3 of lumber is transferred to other uses through the sale of sawmill co-products as for example, fibreboard, or through post-sawmill recycling processes (Figure 5.2). The building life, which is represented by the amount of carbon stored in the product (in solid red), affects how much the end-of-life processes, including the storage of carbon in landfill and CO_2 and CH_4 emissions are delayed. Within the simulation period of 300 years, a longer building life means that overall fewer cumulative C emissions from the wood product manufacture and end-of-life are released within the assumed time horizon. If for example, a typical 100-year simulation period had been chosen, the end-of-life emissions at demolition for a building lifespan of 100+ years would be entirely excluded as the simulation period would end prior to these emissions. As such, the choice of the time horizon is key to determine the life cycle carbon emissions. The carbon emissions from manufacturing, use and end-of-life can vary as a function of wood product type, province or territory and building lifetime (Figure 5.3).

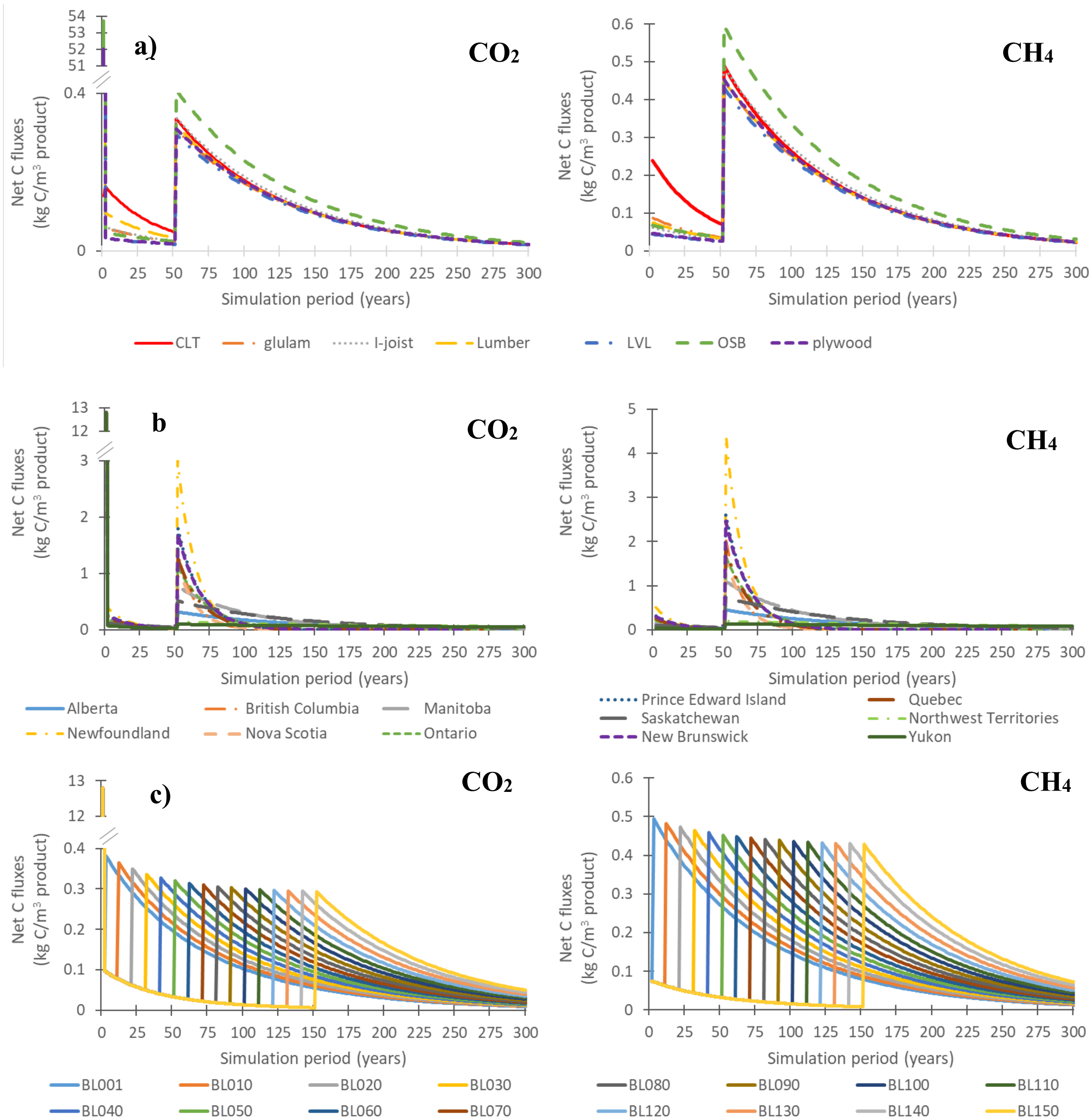
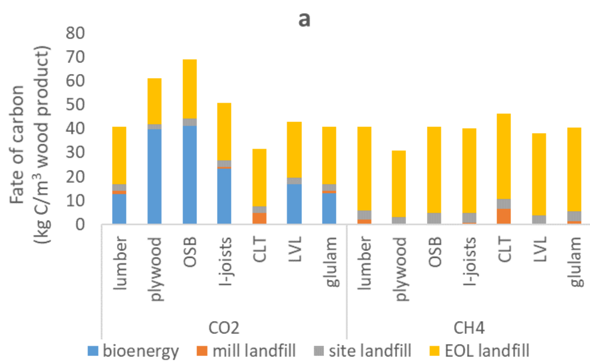


Figure 5.3 – Net carbon fluxes (kgC m^{-3} product · year^{-1}) of CO_2 and CH_4 across: a) seven building products (Alberta, building life = 50 years), b) provinces (lumber, building life = 50 years), c) building life (BL) years (Alberta, lumber)

As shown in Figure 5.3a, the highest emission pulses (reaching 53.7 $kgC{\cdot}m^{-3}$ product for OSB) occur in year 1 due to sawmill co-product treatment via mostly biofuel and landfilling and construction site waste landfilling. Figure 5.4 shows the contribution of these emissions to the total life cycle carbon emissions, for both CO_2 and CH4. When emissions results are examined across wood product types (Figure 5.3a), CLT stands out in terms of the year 1 emissions, especially CH_4 emissions, due to a larger proportion of the finished product (13%) that is deemed to be off-specification and thus is sent to landfill. In terms of the initial post-demolition emissions (year 50), most wood products cluster between 0.29-0.333 and 0.40-0.49 $kgC{\cdot}m^{-3}$ product, for CO_2 and CH_4, respectively. The only wood product that does not follow this trend is OSB, with demolition year emissions of 0.41 and 0.60 kg $C{\cdot}m^{-3}$ product, CO_2 and CH_4, respectively. The reason for this difference is related to large proportion of roundwood log inputs at the sawmill going to the main product (79% for OSB vs. 43-53% for the other wood products) and not the co-products. Given the type of wood product, this makes sense as OSB is made of layered wood strands, which makes it more forgiving in terms of product specifications and thus fewer co-products are produced at the manufacturing stage. A more detailed emissions contribution by life cycle stage is shown in Figure 5.4, which shows the amount of carbon (in $kgC{\cdot}m^{-3}$ wood product) going to various end-use fates for CO_2 and CH_4, for a) all emission outputs and b) all landfill outputs.



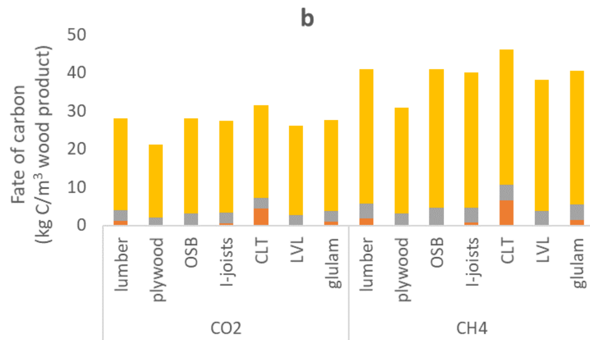


Figure 5.4 – End fates of carbon for seven wood products at the sawmill (bioenergy and mill landfill), at the construction site (site landfill) and end-of-life (EOL landfill). A) shows all end-of-life emissions, while b) shows just landfilling emissions. The EOL landfill proportion are calculated using landfilling rates for Alberta (91.5%). The wood (as carbon) sold from the sawmill is cut-off from the system and is not considered here, and differs by wood type (in $kgC{\cdot}m^{-3}$ (% of total roundwood logs): lumber=219 (52%), plywood=105(36%), OSB=7 (3%), I-joists=151 (42%), CLT=178 (46%), LVL=181 (48%), glulam=214 (52%))

For a given wood product (lumber), the region in which the material is demolished and treated at end-of-life also affects both the CO_2 and CH_4 emissions occurring at and beyond the demolition year (Figure 5.3b). This is a function of two factors: both the percentage of materials recycled as opposed to landfilled and the degradation rates of wood in the landfills (related to the climate of the region). For example, emissions are particularly high (3.0 and 4.4 $kgC{\cdot}m^{-3}$ CO_2 and CH_4, respectively) in the case of lumber disposal in Newfoundland due to the combination of a higher half-life (carbon degrades at a faster rate) and zero percent recycling of construction wood, whereas there are lower CO_2 and CH_4 emissions for disposal in Yukon (CO_2: 0.10 $kgC{\cdot}m^{-3}$, CH_4: 0.13 $kgC{\cdot}m^{-3}$) (due to very slow degradation in cold dry climates) and Nova Scotia (high wood recycling rates). As shown in Figure 5.3c and as was also shown in Figure 5.2, the building lifetime also has a big effect in how the emissions are delayed, with fewer cumulative emissions taking place with higher building lifespans due to the prolonged storage of wood during the use phase.

The application of an increased construction and demolition waste recycling policy (REC70%) and an increased landfill gas collection policy (LFG80%) are compared in terms of CO_2 and CH_4 fluxes (as kg C) (Figure 5.5). Results are shown in three cross-sections using lumber with building lifespan of 50 years as a default, such as to highlight examples with a) high recycling rates (Nova Scotia), b) low landfill half-lives (British Columbia) and c) different building lifespans (Ontario). Since the policy scenarios change the recycling rates and landfill gas capture rates, the three cross-sections (a, b, c) were chosen in order to be able to best highlight differences relative to the baseline.

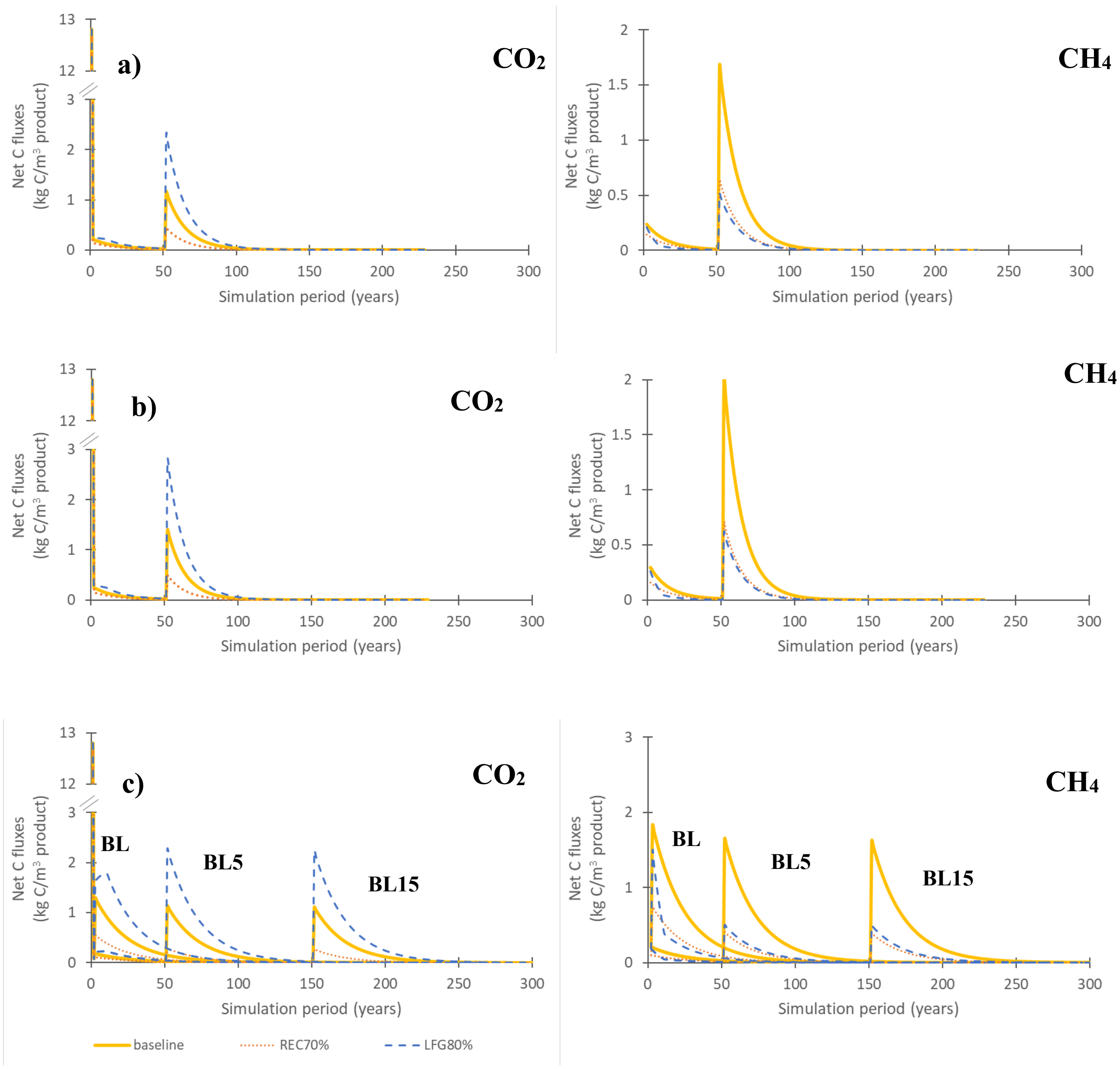


Figure 5.5 – Net carbon fluxes (kg C/m^3 product) of CO_2 and CH_4, comparing increased recycling policy (REC70% scenario) and increased landfill gas collection (LFG80% scenario) to the baseline scenario for lumber across a) high recycling rates (Nova Scotia), b) low landfill half-lives (British Columbia) and c) different building lifespans (Ontario). BL1 = building lifespan 1 year, BL50 = building lifespan 50 years, BL150 building lifespan 150 years

In general, increased construction and demolition waste recycling policy (REC70%) and an increased landfill gas collection policy (LFG80%) have a noticeable influence on the results

(Figure 5.5). The initial emissions released as CO_2 from sawmilling and construction site wastes in year 1 are identical across all scenarios, which is due to the very recent implementations of the policy scenarios, which do not yet have an effect on net carbon fluxes. In terms of CO_2 emissions, the increased landfill-gas capture scenario (LFG80%) has the largest C fluxes, as methane is captured, transformed into energy and ultimately flared to CO_2 emissions. For CH_4 emissions, the baseline scenario has the highest C fluxes and REC70% and LFG80% scenarios have quite similar and lower net C fluxes due to a either a decease in materials going to landfill (REC70%) or an increased capture of CH_4 (LFG80%). While the results in Figure 5.5 report the life cycle inventory in terms of kgC, an exerpt of Figure 5.5a is also reported in Figure 5.6 in terms of $kgCO_2$ and $kgCH_4$.

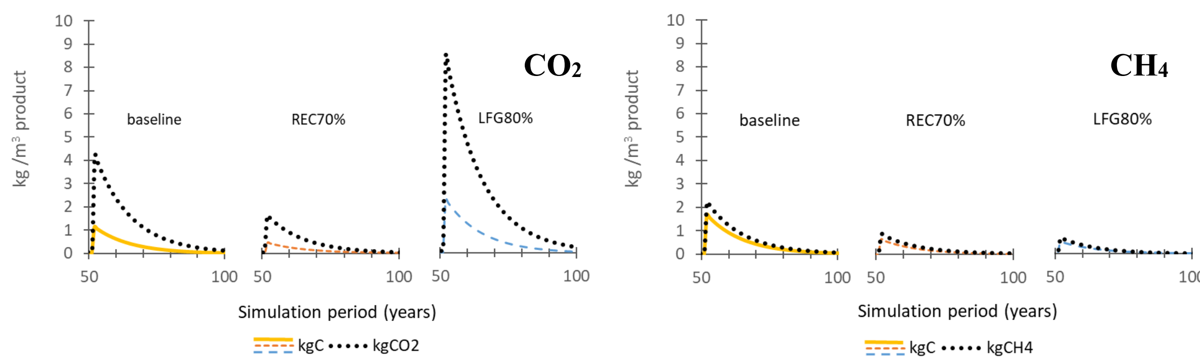


Figure 5.6 – Excerpt of Figure 5.5a (50-100 years) curves comparing kgC results with $kgCO_2$ and $kgCH_4$, for each baseline, REC70% scenario and LFG80% scenario curves

Due to the stoichiometric ratios between C and CO_2 and CH_4 (44/12 and 16/12, respectively), the emissions curves reported in terms of kg CO_2 and kg CH_4 are higher than the per kg C curves, particularly in the case of CO_2. It is important to stress that these emissions are still at the inventory level and would require emissions characterisation (LCIA) to be expressed in terms of kg CO_2-eq.

When recycling rates increase (REC70%, Figure 5.5), lower net C fluxes than the baseline scenario results for both CO_2 and CH_4 emissions (end-of-life peaks of 0.3-0.5 $kgC{\cdot}m^{-3}$ and cumulative emissions of 22.1 kg $C{\cdot}m^{-3}$ – see Table 5.5), due to less wood being sent to landfill. Instead, the recycling of this wood shifts the accountability of the carbon onto other product systems. Though it would seem that an increase in landfill gas capture (LFG80%, Figure 5.5) only shifts CH_4 fluxes to CO_2 fluxes as methane is captured and is combusted, CH_4 emissions have a much higher global warming potential than CO_2 (25 kg CO_2-eq./kg CH_4 vs. 1 kg CO_2-eq./kg CO_2, Ciais et al. (2013)). As such, increased landfill gas collection would reduce overall impacts on climate change at the life cycle impact level, namely from the conversion of CH_4 to CO_2 but also from potential fossil energy substitution that could take place from the generation of energy from landfill gas.

Table 5.5 – Initial post-demolition peak and cumulative CO_2 and CH_4 emissions for curves in Figure 5.5, by province (NS = Nova Scotia, BC = British Columbia, ON = Ontario), scenario, and building lifespan. NS = Figure 5.5a, BC = Figure 5.5b and ON = Figure 5.5c

province	Scenario	building lifespan (years)	CO_2 ($kgC·m^{-3}$)		CH_4 ($kgC·m^{-3}$)	
			initial post-demo peak	cumulative	initial post-demo peak	cumulative
NS	Baseline	50	1.2	32.7	1.7	26.9
	REC70%	50	0.4	22.1	0.6	11.3
	LFG80%	50	2.3	50.9	0.5	8.8
BC	Baseline	50	1.4	34.6	2.0	29.6
	REC70%	50	0.5	22.1	0.7	11.3
	LFG80%	50	2.8	54.4	0.6	9.7
ON	Baseline	1	1.3	42.4	1.8	41.1
	REC70%	1	0.6	25.9	0.7	16.9
	LFG80%	1	1.6	67.1	1.5	16.4
	Baseline	50	1.1	42.4	1.6	41.1
	REC70%	50	0.3	22.1	0.4	11.3
	LFG80%	50	2.3	70.5	0.5	13.1
	Baseline	150	1.1	42.4	1.6	41.1
	REC70%	150	0.3	22.1	0.4	11.3
	LFG80%	150	2.2	70.4	0.5	13.1

The choice of building life has a slight effect on the amplitude of the initial post-demolition emission peaks (Figure 5.5c), which as discussed previously, differs between CO_2 and CH_4 emissions. For CO_2, the baseline scenario has a peak of 1.3 $kgC·m^{-3}$ with building demolition after year 1, but only peaks at 1.1 $kgC·m^{-3}$ at demolition after 50 and 150 years (Table 5.5). In the case of demolition at year 1, the CO_2 peak is only recorded at year 3, a delay that can be accounted for by the time required for landfilled wood to begin decomposition. In contrast, LFG80%, has a peak of 1.6 $kgC·m^{-3}$ for a building lifespans of 1 year, but increases to 2.3 and 2.2 $kgC·m^{-3}$, for building lifespans of 50 and 150 years due to the increased CO_2 from CH_4 combustion. This shift is most evident in comparison with the CH_4 results for LFG80%, where the initial post-demolition emission peak at demolition after 1 year is 1.5 $kgC \cdot m^{-3}$, then drops off to 0.5 $kgC \cdot m^{-3}$ after 50 and 150 years once the policy landfill gas capture targets have been reached. In terms of cumulative emissions, the baseline scenario has constant CO_2 and CH_4 emissions for all building lifespans (42.4 and 41.1 $kgC \cdot m^{-3}$, respectively). Similar to the initial post-demolition emission peaks, the application of the policy scenarios changes the cumulative emissions totals from one building lifespan to another. For REC70%, for example, cumulative CO_2 emissions are 25.9 $kgC \cdot m^{-3}$ at BL1, vs. 22.1 $kgC \cdot m^{-3}$ at BL50 and BL150, while CH_4 emissions are 16.9 $kgC \cdot m^{-3}$ at BL1, vs. 11.3 $kgC \cdot m^{-3}$ at BL50 and BL150. The dynamic nature of recycling and landfill gas capture rates, means that the timing of the building demolition relative to the waste management practices in place at that given time, has impacts on the resulting emissions. For example, when the wood is sent to waste management treatment after 1 year of use, the cumulative net carbon fluxes of LFG80% are similar to those of the baseline. In the first year of LFG80% (2020), the methane capture is still the same as it is in the baseline scenario. LFG80% only begins to distinguish itself from the baseline as landfill gas capture rates begin to increase, as as they gradually approache 80% methane capture over a period of 10 years.

5.2.4 Discussion

As expected, the results show that biogenic carbon emissions are delayed through the storage of carbon in buildings. Keeping the carbon stored in the technosphere by postponing the eventual disposal of wood products has a number of advantages. First, the climate may soon hit a tipping point, meaning that small changes in radiative forcing from emissions could trigger a strong response in the dynamics of the climate, irreparably changing its state (Lenton, 2011). As such, it would be beneficial to avoid the release of as many emissions as possible on the short-term (Brandão et al., 2013; Lenton, 2011). Second, postponing the eventual disposal of wood products may allow for the capacity for recycling to increase and for recycling markets to improve. For example, the Canadian government is currently developing the Clean Fuel Standard, which, when implemented, will aim to reduce greenhouse gas emissions from fuels by 30% by 2030. Some of the fuel pathways currently considered are to be produced from wood residues – this could be a potential treatment pathway for wood construction and demolition waste (IISD, 2017). Finally, it would allow for landfill gas capture rates to improve, specifically if gas was utilised for energy instead of simply flaring CH_4 to CO_2. As older landfills close, it may become attractive to move towards CRD-specific (construction, renovation, demolition) landfills that do not emit as much landfill gas, due the landfilling of dry materials only. Kelleher Environmental, and Guy Perry and Associates (2015) found that there are already a number of municipalities with CRD-specific landfill sites.

As was aluded to the in the previous paragraph, the temporary storage of wood in buildings has the potential for climate benefits. This is especially true when considering the timing of emissions over the entire life cycle of a wood building product. To consider the effects of storage, the temporally-differentiated carbon emissions profiles developed in this work would need to be characterised to climate change potential (in kg CO_2-eq) using a temporal LCA method, which could be done in future work. The chosen time horizon, the period of consideration for the emissions and environmental impacts of a product, are an important consideration in LCA, especially those covering considering long-lived products and temporal emissions profiles. As such, the emissions or impacts of emissions occurring beyond the time horizon are not accounted for in the environmental impacts (Levasseur et al., 2011). Typically in LCA, a time horizon of 100 years is used, although with the development of alternative global warming potential characterisation factors any time horizon can be chosen (Myhre et al., 2013). If, for example, a 100-year time horizon was chosen, beginning the year of the building construction and the wood remains in the building for 100 years, the end-of-life emissions of the wood beyond this point would not be included in the life cycle assessment. As such, as Levasseur (2015) found, the choice of time horizon has an important effect on climate change results. In future research, the effects of selecting different time horizons could be explored with emission characterisation in LCIA.

In this paper, a cut-off approach was used for treating the multifunctionality of both the sawmill processes and the demolition waste. This approach may have some limitations, such as it does not consider additional benefits and burdens from valorizing co-products to be used in other product systems. That being said, the impacts from considering second product life cycles would be expected to vary significantly, based on the wide range of possible uses for products (bioenergy feedstock, mulch, fibreboard, reuse as a wood product, etc.). This could have significant contributions to the life cycle impacts of wood products. Further research is warranted in order to validate a cut-off approach and to examine substitution effects of wood co-products.

The recycling of demolition wood generally decreases the carbon fluxes emitted by a given product, since the carbon is cut-off from the product life cycle and is shifted to a second product. However, its contributions depend on the particular circumstances of the building case. In the case of isolated northern regions, such Yukon and Northwest Territories, where low temperatures and precipitation result in very low landfill decomposition levels and subsequently low landfill emissions and recycling markets are far away, landfilling may be preferable to recycling. However, this is dependent upon the system that is being modelled. While sending secondary wood to markets further south for recycling (e.g. in British Columbia) could mean more emissions in material transport than is avoided by sourcing virgin wood, local recycling could be beneficial, especially if it avoids having to to transport other materials to the north (e.g. housing materials, firewood). The evaluation of these types of effects could be included in future work on wood co-product substitution.

The uncertainties in this study are primarily associated with the proportions and fate of the sawmill co-products. The Athena reports are based on the mass balances and co-product of averaged surveyed sawmills, for which no statistical data is provided. As such, stastical ranges could not be calculated for each of the seven wood product types. Also due to the averaging of sawmill data and the lack of specific geographical content, any sawmill co-products sent to landfill were modelled using an average Canadian landfill. However, due to the small quantities of co-products treated via landfilling, having access to more information on the location of sawmills and sawmill landfills would likely not have much impact on the overall results.

While this model has been developed in this research for the Canadian building context, the flexible nature of CBM-FHWP would allow modelling wood product life cycle cycles specific to other regions. In order to model for the context of another region, the following parameters would need to be adapted: the mass balance and fate of wood production at sawmill, the EOL fate of wood used in construction and background data on landfill half-lives. The adaption of these elements would allow the model to track the fate of biogenic carbon for a region of the user’s choice.

5.2.5 Conclusion

In general, the results show that temporary storage of carbon in buildings delays emissions in the short-term. Most wood products have similar emissions profiles, although CLT and OSB deviate somewhat, primarily due to differences in product specifications. CLT has higher emissions from waste treatment at the sawmill (CO_2: 0.16 kgC·m^{-3} vs 0.03-0.10 kgC·m^{-3}, CH_4: 0.24 kgC·m^{-3} vs. 0.04-0.09 kgC·m^{-3}) due to a large proportion of off-specification product going to landfill. OSB has higher demolition year emissions due to the main product containing a larger proportion of carbon relative to wood inputs, as very few co-products are produced in manufacturing. The province or territory where the building is constructed also has a large influence on the initial post-demolition emissions (CO_2 range: 0.10-3.0 kgC·m^{-3}, CH_4 range: 0.13-4.4 kgC·m^{-3}), due to variability in recycling rates and landfill gas decay rates. Higher recycling rates result in lower carbon fluxes, due to to fewer materials going to landfill causing CH_4 and CO_2 emissions. The coldest and driest regions have the longest landfill half-lives (347 years^{-1} in Yukon landfills), which result in a much slower degradation in the landfill and thus a shift of CH_4 and CO_2 emissions over time, closer to or beyond the time horizon relevant for the decision. The longer building lifepans have both the effect of delaying emissions and an effect on the amplitude of the initial C flux at the year of demolition.

The policy scenarios show the effects of implementing annual changes to the treatment of waste on the resulting carbon fluxes. Both the recycling policy (REC70%) and the landfill gas policy (LFG80%) showed significant changes in emissions after demolition, particularly for those building lifespans that extend beyond the policy target years for the scenarios. In the case of the landfill gas capture scenario, an increased capture shifts CH_4 to CO_2 as landfill gas is combusted through energy utilisation and flaring. The recycling policy reduces the overall CO_2 and CH_4 emissions as it shifts the carbon from the landfill to other product systems, that become accountable for the eventual carbon emissions.

This research demonstrates the use of a harvested wood product model that has been designed for the regional scale, to generate carbon fluxes for single wood products. The resulting carbon fluxes can be expressed as greenhouse emissions and used to determine a dynamic life cycle inventory for modelling the manufacturing, use and end-of-life phases of a cradle-to-grave LCA of single wood products. More explicitly, the results could be integrated in an LCA tool, such a building information model (BIM), such that biogenic carbon fluxes would be automatically modelled for a given wood product. In considering emissions timing, these data would allow building designers to make more informed choices on building material selection. The results could also be used to develop large scale scenarios of the building sector such to inform climate strategy.

5.2.6 Acknowledgements

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CHAPTER 6 ARTICLE 3: DYNAMIC GREENHOUSE GAS PROFILES OF WOOD USED IN CANADIAN BUILDINGS

This chapter provides the article manuscript that details the calculation of a database of life cycle inventories (LCI) and dynamic climate change impacts (DCCI) of wood building products, for different use contexts across Canada. The authors of this article are Marieke Head, Annie Levasseur, Robert Beauregard and Manuele Margni. The manuscript was submitted to *Building and Environment* on August 2nd, 2019. The Supplementary Material submitted to the journal with this manuscript is large and contains the modular database of life cycle inventory and dynamic climate change impacts, as well as additional information, tables and figures. The database files are large and in spreadsheet form and cannot be feasibly included in this dissertation, however most of the additional information, tables and figures in the Supplementary Material can be found in Appendix D.

6.1 Manuscript

6.1.1 Introduction

Wood building materials are very common in the Canadian construction sector, particularly in residential applications. Recently wood has been increasingly promoted, due to its reported low climate change impacts compared to other building materials (Province of British Columbia, 2009; Quebec Ministry of Forests Wildlife and Parks, 2008). However, the literature reports a wide range of climate change impact scores, making comparisons with non-wood materials difficult. Røyne et al. (2016) analyzed how climate change impacts are calculated using nine different metrics in 101 peer-reviewed life cycle assessments (LCA) of forestry products (published between 1997 and 2013), finding that 87% of studies considered biogenic carbon emissions to be neutral, 3% of studies considered emissions timing and 0% gave credits for temporary or permanent carbon storage.

The zero-impact assumption of biogenic carbon neutrality considers that the same amount of carbon that is sequestered in biomass during its growth is eventually reemitted into the atmosphere, i.e. any biogenic carbon dioxide uptakes or emissions could be assigned zero climate change impacts as they cancel out. However, several authors have challenged this assumption, because of several inaccuracies in the calculation of climate change impacts (Garcia & Freire, 2014; Røyne et al., 2016; Searchinger et al., 2009; Vogtländer et al., 2013), such as not considering unsustainable harvest rates or the fact that the atmosphere does not differ between the source and type of CO_2 molecules.

Several methods have been proposed over the course of the last fifteen years to evaluate the benefits of temporary carbon storage, but the benefits of delaying carbon emissions is still debated (Brandão & Levasseur, 2011; Brandão et al., 2013). A few authors have proposed means of considering the climate change impacts of carbon storage on a product or building level. Vogtländer et al. (2013) proposed a method for considering the carbon storage of wood products in which the temporary storage of carbon in buildings or long-lived products is only considered to have a net emissions benefit if the building replacement rate is exceeded. Levasseur et al. (2013) showed that the use of a dynamic LCA approach, whereby the timing of emissions is considered through both temporal emissions profiles and time-dependent characterisation factors (Levasseur et al., 2010)**,** could consistently calculate climate change impacts and benefits of biogenic carbon throughout the life cycle of products, including temporary carbon storage during the product life. In their review of life LCAs of forestry products, Røyne et al. (2016) found that the majority of LCAs do not address the dynamics of biogenic carbon emissions. Furthermore, De Rosa et al. (2018) found that since forestry activity and forest products highly affect the timing of CO_2 uptake and emissions, the time dependence of CO_2 fluxes should be addressed in any LCA that involves forestry products.

As such, to put into practice most of the temporal approaches to the life cycles of products, the dynamic profiles of each emissions type should be created. Several authors have explored the temporal differentiation of life cycle inventories (LCI) in order to be able to apply dynamic methods (Beloin-Saint-Pierre et al., 2014; Collet et al., 2013; Collinge et al., 2013; Pinsonnault et al., 2014; Yuan et al., 2015). Collet et al. (2013) developed a method for determining whether considering dynamic LCIs has an overall impact on life cycle impact assessment (LCIA) results, and whether they are worth applying. Beloin-Saint-Pierre et al. (2014) proposed the enhanced structural path analysis (ESPA) method in order to apply temporal differentiation on a mass scale (database-level) to elementary and process flows. Within the context of global warming impacts, Pinsonnault et al. (2014) sought to determine the sensitivity of adding temporal information to LCIA results by using the ESPA method (Beloin-Saint-Pierre et al., 2014) and found that temporal information can be particularly relevant for products containing biomass. In their temporal LCIA method, Yuan et al. (2015) also describe steps in order to temporally differentiate at the LCI level.

Since the development of methods for considering timing, several authors have applied these methods to the LCAs of forestry products (Daystar et al., 2017; Demertzi et al., 2017; Faraca et al., 2019; Fouquet et al., 2015; Peñaloza et al., 2018). Fouquet et al. (2015) explored methodological challenges of building LCA, including biogenic carbon accounting and emissions timing. Most recently, some authors studied specific methodological issues. Peñaloza et al. (2018) explored how system boundaries and baselines for forest carbon dynamics affects climate change impacts of forestry products when temporal LCIA is used. Faraca et al. (2019) used temporal LCIA from a waste management perspective, thus determining the climate change impacts of wood reuse using a cascading approach.

The recent literature shows that though it is important to consider the emissions timing for wood products in long-lived products, such as buildings, reviews like Røyne et al. (2016) show that very few studies consider timing, for many reasons including the lack of data. As such, the objective of this study is to calculate a database of LCI and dynamic climate change impact (DCCI) scores of Canadian wood products, covering the twelve most common softwood species from managed forests across the country, for the production of seven common wood construction materials, used in buildings in 12 provinces and territories for building lifespans ranging from 1-150 years. To do so, we take into account the temporally differentiated greenhouse gas emissions profiles of wood harvest on Canadian forests (Head et al., 2019a), the temporary carbon storage and end-of-life emissions of wood used in buildings (Head et al., 2019b, unpublished results), as well as the emissions of activities occurring in intermediary life cycle phases such as wood manufacturing and transport. Additionally, we performed a sensitivity analysis and we compared results with those of another study on wood products. Finally, we provide a complete modular database of LCI and DCCI scores of Canadian wood products that can be used by LCA practitioners and building designers to assess case-specific building & construction projects.

6.1.2 Materials & Methods

6.1.2.1 Dynamic Life Cycle Assessment (DLCA)

The DLCA method developed by Levasseur et al. (2010) was used to assess the climate change impacts of wood product life cycles over time. The method uses a dynamic characterisation factor for calculating radiative forcing impacts of temporally differentiated greenhouse gas emission pulses.

$$DCF_g(t)_{instantaneous} = \int_{t-1}^{t} RE_g \cdot \left[C_g(t)\right] dt \qquad (1)$$

Where $DCF_g(t)$ is the dynamic characterisation factor used to calculate radiative forcing at year t following the emission of a g greenhouse gas pulse (in $W \cdot yr \cdot m^{-2} \cdot kg^{-1}$), RE_g is the instantaneous radiative forcing per unit mass increase in the atmosphere for a given greenhouse gas g, also called radiative efficiency (in $W \cdot m^{-2} \cdot kg^{-1}$) and $C_g(t)$ is the atmospheric load of a given greenhouse gas t years after an emission pulse ($kg \cdot kg^{-1}$).

For carbon dioxide, $C_g(t)$ is further characterised by the Bern carbon cycle-climate model,

$$C_{CO_2}(t) = a_0 + \sum_{i=1}^{3} a_i e^{-t/\tau_i} \qquad (2)$$

Where a_0, a_i and τ_i are coefficients for which the most up to date values were used (Joos et al., 2013). While C(t) for non-CO_2 greenhouse gases (GHG) is characterised by,

$$C(t) = e^{-t/\tau_j} \qquad (3)$$

Where τ_j are derived for each non-CO_2 GHG, CH_4 and N_20 in Myhre et al. (2013).

Solving for DCF using RE_g and $C_g(t)$ for carbon dioxide and for non-CO_2 GHG, results in

$$DCF_{CO_2}(t)_{instantaneous} = RE_g \left[a_0 t + \sum_i a_i \tau_i \left(1 - exp\left(-\frac{t}{\tau_i}\right)\right)\right] \qquad (4)$$

$$DCF_{non-CO_2GHG}(t)_{instantaneous} = RE_g \cdot \tau_j \left[1 - exp\left(-\frac{t}{\tau_j}\right)\right] \qquad (5)$$

These dynamic characterisation factors are used in combination with a temporally differentiated emissions inventory to calculate the instantaneous global warming impact, GWI_{inst}. GWI_{inst} the sum of the radiative forcing that occurs at time t caused by the GHG emissions occurring previously. It is calculated by multiplying each emission by the dynamic characterisation factor for the interval of time between the pulse emission (at time i) and time t.

$$GWI_{inst}(t) = \sum_{i=0}^{t}\left[kgCO_2(i) \cdot DCF_{CO_2}(t-i)\right] + \sum_{i=0}^{t}\left[kgCH_4(i) \cdot DCF_{CH_4}(t-i)\right] + \sum_{i=0}^{t}\left[kgN_2O(i) \cdot DCF_{N_2O}(t-i)\right] \quad (6)$$

The sum of GWI_{inst} calculated for all years from 0-t is the cumulative radiative forcing caused by all GHG emissions over a 0-t time period and is represented by,

$$GWI_{cum}(t) = \sum_{k=0}^{t} GWI_{inst}(k) \quad (7)$$

Finally, the results are converted into DLCIA (dynamic life cycle impact assessment) scores in terms of $kgCO_2$-eq., by dividing the GWI_{cum} by the cumulative radiative of 1 kg of CO_2 emitted at time zero to the given time horizon TH,

$$DLCA = \frac{GWI_{cum}(TH)}{\int_0^{TH} RE_{CO_2} \cdot C(t)_{CO_2} dt} \quad (8)$$

Where the denominator of the equation represents the cumulative radiative forcing of a single 1 kg CO_2 pulse emission. TH represents the time horizon chosen for the study. As the choice of the TH might have a large impact on the results of a study (Daystar et al., 2017; Levasseur et al., 2010), the results in this study are calculated at 100, 250 and 500 years (TH_{100}, TH_{250}, TH_{500}), in order to capture the timespan of forestry, building lifespan and landfill emission propagation.

6.1.2.2 Wood Product Life Cycle and Temporally Differentiated Emission Profile

The wood life cycle considered in this paper can be summarised by the system boundaries diagram shown in Figure 1. In addition to showing the process flow of the life cycle, the relative timing of each unit process along an emissions timescale is provided.

Building and construction product category rules (PCR) provide specific guidance for LCAs of building and construction products. Most of these PCRs refer to the European Directive, EN 15804 (European Commission, 2012b), which identifies modular A-D life cycle phases for construction products: Product stage: A1-A3; Construction process stage: A4-A5; Use stage: B1-B5; End-of-life stage: C1-C4; Future, reuse, recycling or energy recovery potentials: D. An LCA of a product used in a building or construction project should consider all modules from at least A through C.

Given the focus of this work on building materials and the wide variety of building energy consumptions, the use stage (modules B1-B5) is excluded from the system boundaries (as is shown by the dotted box around the Building Life unit process in Figure) and should be added by the users of the database we developed according to their context.

Only greenhouse gas emissions (CO_2, CH_4, N_2O) are considered and characterisation of emissions is done using DLCA as described in section 2.1. The functional unit is: 1 m^3 of wood product used in buildings or construction projects in Canada. Since several wood products are considered, it is important to stress that the functionality of each wood product and the product lifespan are not equivalent as they are used differently within buildings. As such, the discussion of specific results is aimed at highlighting tendencies and not to compare different wood products.

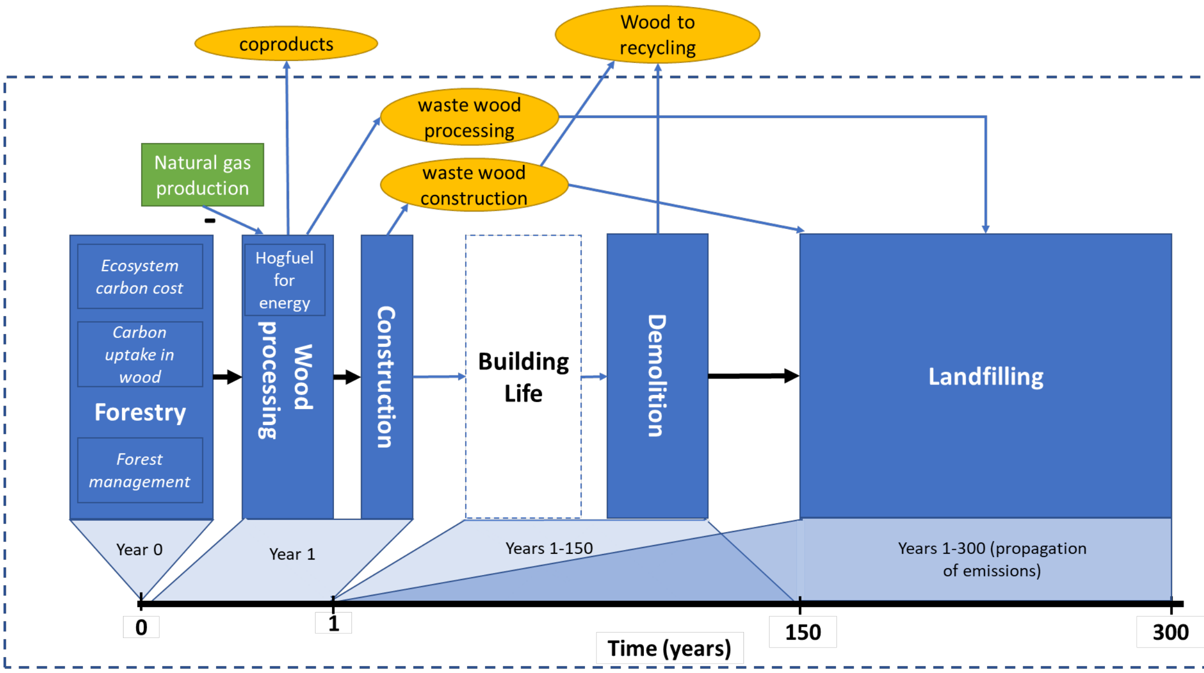

Figure 6.1 – System boundaries of wood products along a life cycle timeline. The dotted large square denotes the system boundary. Thick black arrows show transport flows, while blue arrows do not include transport. Building Life appears in the timeline, but no elementary flows are considered. Yellow circles indicate elementary and intermediary flows. The green box with "Natural gas production" is a unit process with an arrow directed towards "Wood processing" (and a negative sign) to indicate the substitution of hogfuel used as bioenergy at the wood processing unit process.

6.1.2.2.1 Considerations for DLCA

The DLCA is applied on a temporally differentiated emissions profile, which requires assessing the relative timing of each of the emissions to year zero within a unit process. For example, in the case of a building, the manufacturing emissions occur in year 1, while operation emissions from heating, air conditioning and electricity consumption from the use phase occur annually during the building lifespan (e.g.1-75 years). Each individual unit process of a product life cycle has its own emissions profile which should be differentiated relative to itself, i.e. the start time zero is relative to the first emission fluxes of that individual process (Beloin-Saint-Pierre et al., 2014). These process-specific profiles can be subsequently placed relative to one another modularly on an absolute life cycle timescale. A modular approach allows for the endless permutations of all life cycle parameters, which in the case of this study, could include species, growing region, time since harvest began historically, wood product type, building location and building lifespan. The life cycle phases considered in the main results are described in the sections below.

6.1.2.2.2 Forestry

6.1.2.2.2.1 Ecosystem carbon costs

In addition to the carbon uptake in harvested wood, the act of harvesting wood also has a carbon cost depending on the management context. The ecosystem carbon cost (ECC) is defined as the net carbon flux from the forest to the atmosphere that can be attributed to the harvesting activity and is calculated at the landscape level. The landscape level perspective taken here considers a much larger geographical area where only a small portion of trees are harvested every year and thus age class distribution within the forest is variable. In Canada, forestry is managed through annual allowable cuts determined by provincial governments to ensure that harvest rates remain at sustainable levels (FPInnovations, 2010). ECC of harvesting wood from 117 forestry landscapes were developed in Head et al. (2019a). For each landscape, curves from 0-100 years of historically managed forests were developed. As shown in Figure 6.2, the ECC varies significantly from 0-100 years, where carbon fluxes per m^3 harvested wood to the atmosphere are positive for the first few decades of forest management, followed by decreasing carbon fluxes such that most landscapes show a net negative carbon flux per m^3 harvested wood (sequestration) at 100 years. Weighted average ECC for the twelve examined species over their annual production vary from -0.24 to 0.094 $tC \cdot m^{-3}$ at 100 year of historical forest management. Note that these weighted averages differ somewhat from the specific species chosen for the sensitivity analysis described in section 2.4.1.

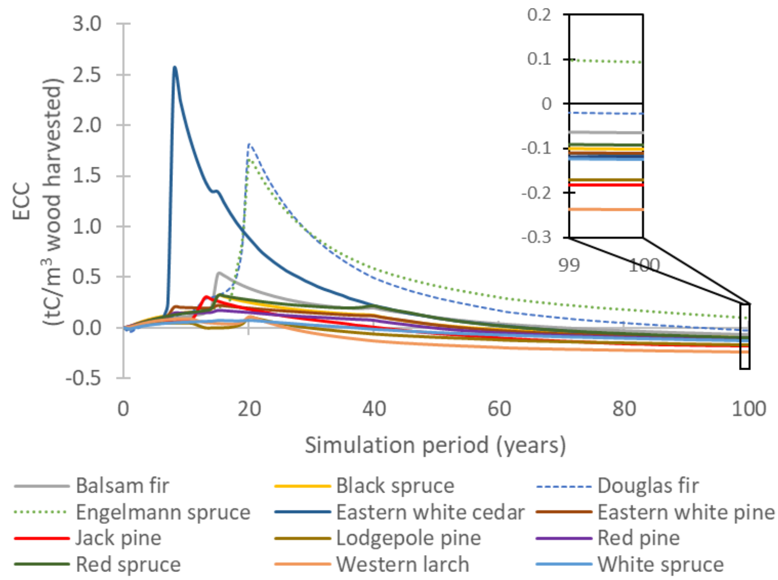

Figure 6.2 – Weighted average ecosystem carbon cost (ECC) from 0-100 years of historical forest management for 12 tree species

ECC values after 100 years of forest management were used as a default in this study based on the historical legacy of Canadian forests that began on average at the beginning of the 20th century (Kurz et al., 2013). ECC values are considered at year 0 of the product life cycle (reference year for the LCA).

6.1.2.2.2.2 Carbon uptake in wood

Plants and trees use sunlight to capture atmospheric carbon dioxide in a process called photosynthesis. These carbon dioxide molecules are transformed into oxygen that is released to the atmosphere and simple sugars, which are used by the plant for maintaining vital plant functions and for plant growth. The simple sugars are converted into carbon that is locked into the accumulating biomass. In the LCA of wood products, it is important to consider the uptake of carbon dioxide by tree biomass. The carbon contained in the harvested wood is not accounted for by the ECC and thus must be considered separately. Due to the landscape perspective taken by the ECC and the use of a single carbon emission at time zero, the carbon content of the wood is also represented by a single pulse at year 0 (reference year for the LCA).

6.1.2.2.2.3 Forest management

Within the context of a life cycle of wood products, the actual site preparation, extraction and transportation processes occurring during forest management need to be considered. The ecoinvent v3.4 database unit process *Sawlog and veneer log, measured as solid wood under bark CA-QC* was used and carbon uptake was set to zero in order to avoid double counting with previously calculated carbon uptake (ecoinvent, 2017). In the absence of other province-specific data we assumed this process being a reasonable approximation for forestry management practices across Canada. Greenhouse gas emissions from diesel and other activities are assumed to take place in year 1, following the harvest of wood.

6.1.2.2.3 Wood processing

Once the logs have been harvested and transported to wood processing sites, the logs are transformed into wood products at one or more facilities, depending on the product type. The LCIs were modelled using the Athena Sustainable Materials Institute wood product reports (ASMI, 2012a, 2012b, 2012c, 2012d, 2013a, 2013b, 2013c, 2018a, 2018b, 2018c, 2018d, 2018e) for seven different wood products: cross-laminated timber (CLT), laminated veneer lumber (LVL), glulam, softwood lumber, plywood, oriented-strand lumber (OSB) and I-joists.

The latest versions of the Athena reports (2018) use an economic allocation approach as advised by the North American wood product PCR (FPInnovations, 2015) and also includes results using mass allocation. Due to confidentiality with wood product processors, the prices and allocation factors used to generate the economic allocation LCIA results are not included in the report. This makes the economic allocation calculation difficult considering the variation of market prices that exist relative to those used by Athena. In addition, for some products (e.g. CLT, I-joist) coproducts are transferred free of charge despite being integral to a secondary product or bioenergy for the third-party receiver. Using economic allocation in this case would allocate 100% of impacts to the main product, despite the coproduct being used as a material input for other processes. Also, the Athena reports show a range of differences between economic and mass allocation results for global warming potential impact category of 0-45%. Mass allocation by kg of oven dry wood, is also proportional to allocation by carbon content, as oven dry wood is 50% carbon. For these reasons, mass allocation was used between main products and sold coproducts. The emissions of burning hogfuel (waste wood) on site are considered, as are the avoided emissions CO_2, CH_4 and N_2O from the ecoinvent unit process *Heat, district or industrial, natural gas (CA-QC), which was* determined to substitute an equivalent MJ of bioenergy from the hogfuel combusted. Landfill emissions from waste wood are also considered as an average of Canadian landfill conditions and are modelled as landfilling is modelled for end-of-life wood products as described in section 2.2.5.

6.1.2.2.4 Construction and Demolition

For the building and demolition of the building, emissions were assumed to come from the use of (diesel burning) construction equipment. The energy consumption was expressed in $kWh \cdot m^{-2}$ floor space (80 $kWh \cdot m^{-2}$ for building and 10 $kWh \cdot m^{-2}$ for demolition) and were converted to $kWh \cdot tonnes^{-1}$ wood, all using the data supplied in Gustavsson et al. (2010).

6.1.2.2.5 End-of-Life

In the life of wood products, wood is discarded at two separate times: at the construction site and at the demolition site. As is described in Head et al. (2019b, unpublished results), 10% of the wood input is assumed to be discarded at the construction site. The 90% of wood that remains in the building, is discarded upon the demolition of the building after a given lifespan. End-of-life wood waste is mostly treated through landfilling, with a small amount of recycling that is variable between provinces.

As described in Head et al. (2019b, unpublished results), landfill gases are treated as per the average Canadian landfill and thus gases are captured and flared, captured and used as bioenergy and directly emitted to atmosphere. Wood recycling is treated using a cut-off approach, the details of which are described in Head et al. (2019b, unpublished results). The carbon flux profiles cover two GHG emissions (CO_2 and CH_4) seven wood products (see those listed in section 6.1.2.2.3), twelve Canadian provinces and territories and building lives from 1-150 years (with intervals: annual 1-10 years, every 5 years 10-50 years, every 10 years 50-150 years).

6.1.2.2.6 Transportation

Two transportation legs were modelled – the transportation of new products from the wood processing to the building site (assumed to be 500 km) and the transportation from the building site to the landfill (assumed to be 30km). These were modelled using a >32t ton truck (Euro 3 classification) from ecoinvent v3.4.

6.1.2.3 Application of DLCIA to wood product cases

DLCIA was applied to the temporalised LCIs described above, creating a DCCI database that covers the use of softwood in Canadian buildings. To model a life cycle from forest to end-of-life with a modular database, a few decisions must be considered. First, an absolute life cycle timescale needs to be created. We took the normative choice to set the year zero of the product life cycle at the time of wood harvesting. Thus, the emissions related to the forest phase takes place at year 0, the building construction phase within year 1, with the timescale ending at year 500 for illustrative purposes. DLCIA for the carbon uptake is considered for a duration of 500 years, whereas for other phases DLCIA is calculated up to 500 years (that is, 500 years minus the timing of a given emission). The modularised LCI and DCCI should be integrated as shown in Table 6.1 (additional details are given in the Supplementary Material).

Table 6.1 – Relative time placement of unit processes into cradle-to-grave LCA of wood product, BL = building lifespan

Unit processes	Subprocesses	Relative time placement	Other
Forestry	Carbon uptake in wood	Start at 0	Multiply by roundwood logs needed per m^3 of product*
	Ecosystem carbon cost	Start at 0	Multiply by roundwood logs needed per m^3 of product*
	Forest management	Start in year 1	
Wood processing		Start in year 1	
Construction and Demolition	Building construction	Start in year 1	
	Building demolition	Start at end of BL	
End-of-life		Start at end of BL	Note: database considers all biogenic carbon emissions from wood processing through to end-of-life (start at year 1)
Transportation	Transport wood to site	Start in year 1	
	Transport construction site waste to end-of-life	Start in year 1	Multiply by 0.1 to consider that 10% of wood disposed during construction (Head et al., 2019b, unpublished results)
	Transport demolition waste to end-of-life	Start at end of BL	Multiply by 0.9 to consider that 90% of wood remaining in the building at demolition (Head et al., 2019b, unpublished results)

*See Supplementary Material for a list of ratios of m^3 roundwood logs/m^3 wood product

While there are endless possibilities for combinations of different modules, certain species and species mixes are more prevalent in combination with certain wood products over others (see Supplementary Materials). In order to illustrate how dynamic product life cycles inventories and impact assessment can be developed using modular phases, four illustrative wood product case studies were constructed. The wood product life cycles start with four wood mixes: Eastern SPF (spruce-pine-fir), Western SPF, Douglas fir-Larch and Cedar (see Supplementary Material for more details on common wood product types). These four wood types are associated with wood products typically produced with those wood mixes. The parameters of these product life cycles are given in Table 6.2.

Table 6.2 – Parameters for base cases of four cradle-to-grave wood product cases (AB = Alberta, BC = British Columbia, SK = Saskatchewan, MB = Manitoba, ON = Ontario, QC = Quebec, NB = New Brunswick, NS = Nova Scotia, PE = Prince Edward Island, NL = Newfoundland, CLT = cross-laminated timber)

Short case name	E SPF	W SPF	DF-L	Cedar
Full name	Eastern spruce-pine-fir	Western spruce-pine-fir	Douglas fir-Larch	Cedar
Species	Balsam fir, Black spruce, Jack pine, Red spruce, White spruce	Engelmann spruce, Lodgepole pine, White spruce	Douglas fir, Western Larch	Eastern white cedar
Growing provinces*	SK, MB, ON, QC, NB, NS, PE, NL	AB, BC	AB, BC	MB, ON, QC, NB, NS, NL
Wood product	Lumber	CLT	LVL	Lumber
Application	Building	Building	Building	Decking
Building site	Quebec	British Columbia	Alberta	Nova Scotia
Product lifespan	50 years	100 years	25 years	15 years

*The Canadian Wood Council considers that Western Canada constitutes AB and BC only. All other provinces are considered Eastern Canada for purposes of wood supply.

6.1.2.4 Sensitivity Analysis

Alternative variants of the base case are assessed in a sensitivity analysis such as to determine their effect on the four case studies. Further details on these two cases are available in the Supplementary Material.

6.1.2.4.1 Using different ECC values:

The default calculations are done using ECC emissions after 100 years of constant sustainable harvest (Figure 2). In this case, ECC are extended to 300 years to consider what the impact of continuing constant sustainable harvest has on climate change impacts of the wood product life cycle. The default cases consider wood mixes, however for this sensitivity analysis case, individual landscapes representing the top landscapes of each mix are used:

- E SPF: Black spruce, Quebec, Boreal Shield
- W SPF: Lodgepole pine, British Columbia, Montane Cordillera
- DF-L: Douglas fir, British Columbia, Montane Cordillera
- Cedar: Eastern white cedar, Quebec, Boreal Shield East

Five different points along x-axis of the extended ECC curves are considered within the dynamic LCA of the wood use life cycles: 50, 75, 100 (default), 150 and 200 years.

6.1.2.4.2 All wood waste from building is incinerated

The base case considering landfilling and recycling EOL is compared to a case in which all building site waste is 100% incinerated, with substitution of electricity generation of the grid mix as well as the marginal energy production for each region (AESO, 2018; National Energy Board, 2019; Nova Scotia Power, 2017) and a case where no energy is substituted. Information on the unit processes used for each wood product case can be found in the Supplementary Material. The electricity generated by the incineration of wood is calculated using a lower heating value of 18.9 $MJ\cdot kg^{-1}$ (Kostiuk & Pfaff, 1997) and a best available technology efficiency of 30% (Stantec, 2011).

6.1.3 Results & Discussion

6.1.3.1 Greenhouse gas emissions LCI database

A modular greenhouse gas emissions LCI database was developed, covering all the unit processes described in section 2.3. The database files are included in the Supplementary Material. A graph of the cradle-to-grave emissions profiles (in kg GHG$\cdot$ m^{-3} lumber) for a single case (Eastern SPF) is shown in Figure 3. The two curves show the emissions of the decaying wood in the landfill. A table of this emissions profile, with unit process contributions is available in the Supplementary Material.

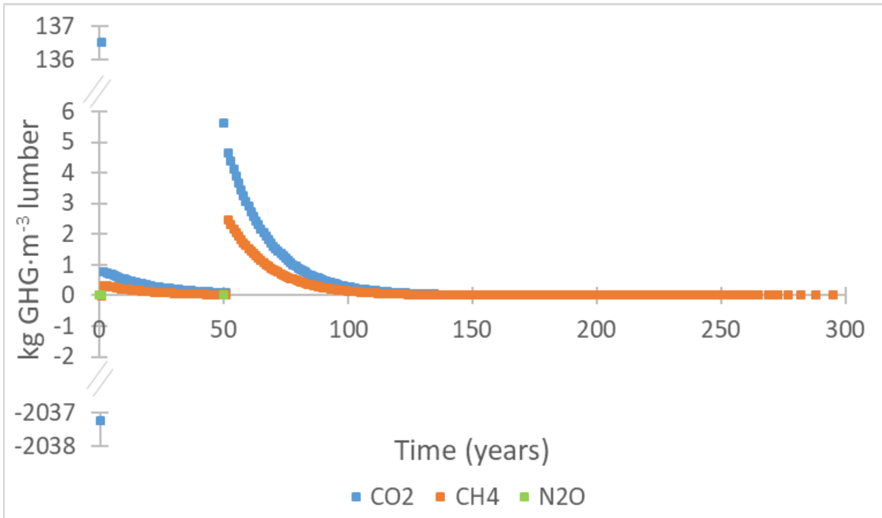

Figure 6.3 – Overall cradle-to-grave GHG emissions (kg CO_2, kg CH_4, kg N_2O) profile for Eastern SPF. Each point indicates a pulse emission in a given year. Outlier points are shown in the extensions to the graph in the positive and negative axis.

6.1.3.2 DCCI database

The application of DLCA to the temporally differentiated gate-to-gates LCI of each life cycle phase yielded a database of DCCI expressed in instantaneous radiative forcing (GWI_{inst}). The database is separated into different files, covering forestry carbon, forest management, wood processing, construction and demolition processes, transport, and end-of-life phases. The database extends to 1000 years of emissions impacts for each phase. While wood processing, construction, demolition and transportation (as defined in Figure 1 and section 2.3) are generic averages for wood used in Canada, the carbon uptake in wood and ECC during the forest management life cycle stage, can be specified by the user as well as building specifications, such as wood product type, location of the building and expected building lifespan (as defined in section 2.2.5). The full database is available in the Supplementary Materials.

6.1.3.3 Cradle-to-grave DCCI of wood product case studies

Results for GWI_{inst}, GWI_{cum} and DLCA for four wood product cases are shown in Figure 4. Note that any mention of specific results is not meant to be understood as direct comparisons between wood products.

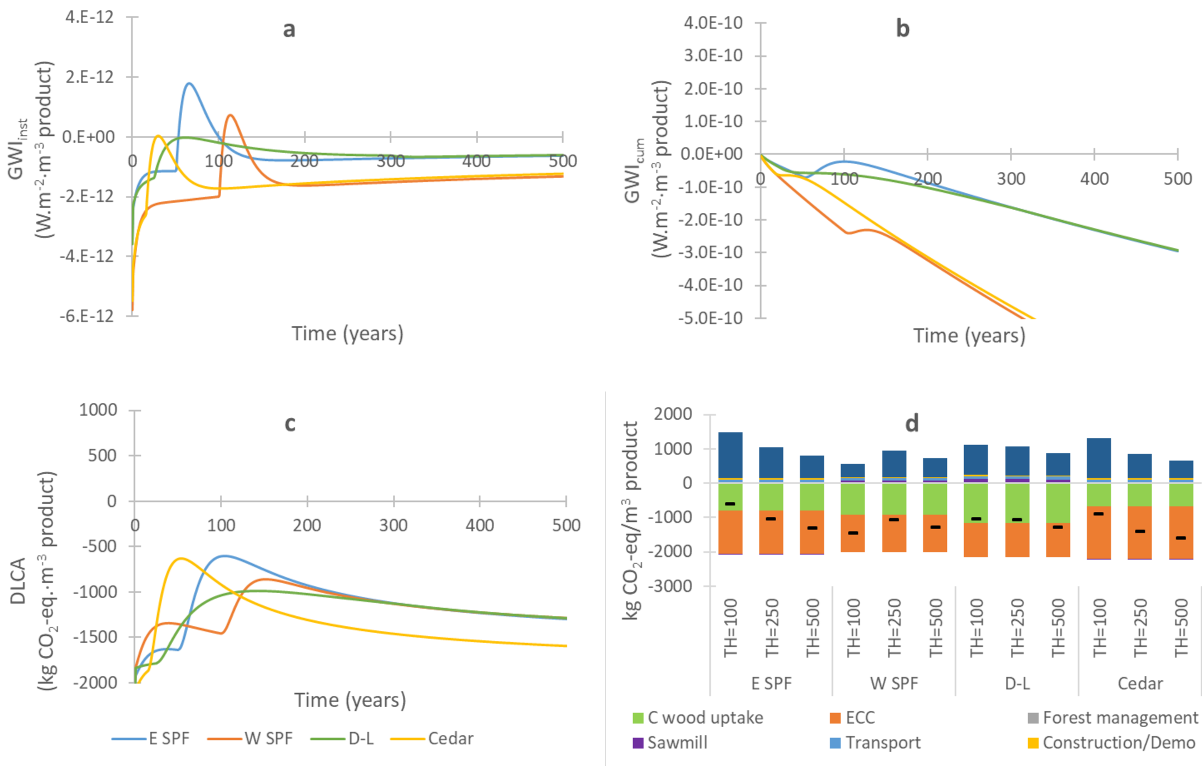

Figure 6.4 – Results for four case studies: E SPF, W SPF, DF-L, Cedar. Curves in a, b and c are shown for TH 0 to 500 years. a) GWI_{inst},($W \cdot m^{-2} \cdot m^{-3}$ product), b) GWI_{cum} ($W \cdot m^{-2} \cdot m^{-3}$ product), c) relative warming potential (kg CO_2-eq$\cdot m^{-3}$ product), and d) relative warming potential of E SPF, W SPF, DF-L, Cedar for three different TH, TH_{100}, TH_{250}, TH_{500} (kg CO_2-eq$\cdot m^{-3}$ product). Case studies: E SPF = Eastern spruce-pine-fir, W SPF = Western spruce-pine-fir, DF-L = Douglas fir-Larch, ECC = ecosystem carbon cost.

GWI_{inst} results (Figure 4a) show initial negative radiative forcing values at year 0 due to the carbon uptake of the harvested wood and the ECC. This differs between wood mix types, as each combines different landscapes (species and regions) and wood carbon densities. The sharp curve is then followed by an increase in radiative forcing, which is dependent on emissions from activities related to forest management, wood products processing, transportation and building construction and most notably the carbon released at the end-of-life. At the building lifespan for each wood use case, a large and sudden increase in the upward curve is visible which is characterised by demolition processes, transportation of waste to a disposal site and the subsequent end-of-life disposal of the wood, as was shown in the sample emissions profile in Figure 3. Following the increase is a slow decrease in the curves, as emissions taper off from landfills and as the GHGs in the atmosphere decay or taken by carbon sinks. Though their radiative forcing values beyond TH_{300} are relatively similar, Western SPF and Cedar have the lowest radiative forcing scores, while Eastern SPF and Douglas fir-Larch have the highest.

With the GWI_{cum} curves (Figure 4b) the individual phase contributions to the radiative forcing values are no longer visible, however the cumulative radiative forcing over time is shown which smooths out effects of large instantaneous values. The spread amongst the four wood use cases is much greater, with the Western SPF and Cedar cases having a negative cumulative radiative forcing curves and the Eastern SPF and Douglas fir-Larch cases having positive ones.

In terms of the relative warming potential, DLCA (Figure 4c and 4d), the curves are clustered more in the negative axis than with GWI_{inst}. Though its emissions are some of the highest of the four cases in the first 100 years, the Cedar case has a low DCCIs beyond TH_{100} despite having a relatively low product lifespan (15 years). This is result of a few factors. First, the ECC impact has a very low negative impact (-1541 kg CO_2-eq·m^{-3} lumber). Second, end-of-life emissions occur relatively soon in the temporally differentiated profile, which means that emissions taper off sooner, despite the fact that landfill emissions are relatively higher since decay rates in landfills in Nova Scotia are much higher than most provinces in Canada (half-life=9.242 years, decay rate constant k=0.075 $year^{-1}$). The Cedar curve reaches -1373 kg CO_2-eq·m^{-3} lumber TH_{500}. The other three wood product cases differ in the few 200 years but converge to a very similar curve afterwards. At TH_{100}, Western SPF has the lowest DCCI (at -1172 kg CO_2-eq·m^{-3} CLT), as no end-of-life emissions have yet taken place. Eastern SPF has the highest DCCI at TH_{100}, due to end-of-life emissions having taken place starting at 50 years coupled with a relatively high landfill decay rate in Quebec (k =0.059 $year^{-1}$, half-life=11.8 years). In contrast, Douglas fir-Larch has a much lower DCCI at TH_{100} despite having a shorter product lifespan (25 years), due to the low decay rates in landfill sites in Alberta (half-life=57.762 years, landfill decay rate constant k=0.012 $year^{-1}$)

6.1.3.4 Sensitivity Analysis

6.1.3.4.1 Using different ECC values

ECC varies depending on how long in the past the forest has been exploited (forest management time horizon). Figure 6.5 show the ECC variability for the top tree species for each wood case which tend to converge at a constant negative ECC at around -200 tC/m^3 after 300 years of forest management. We analyzed the sensitivity of applying ECC values selected at 50, 75, 100 (default), 150 and 200-year time horizons on the DLCA results for the four cases (Figure 6.6).

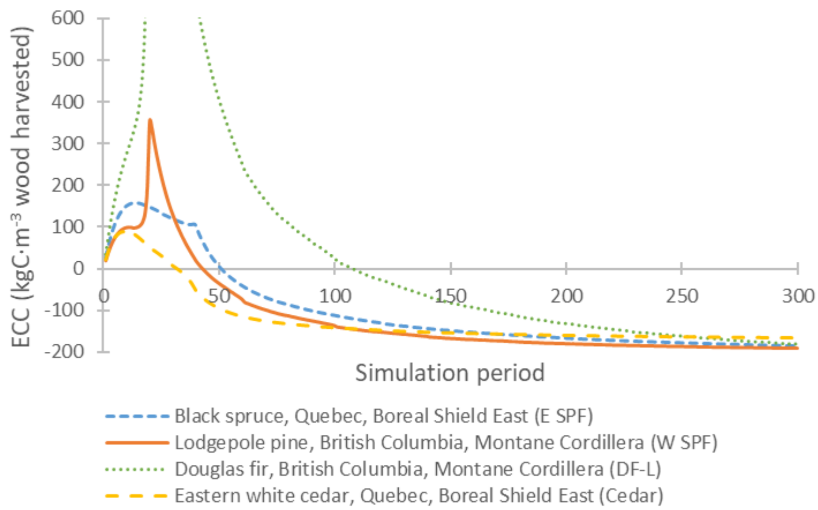

Figure 6.5 – Ecosystem carbon costs (ECC) extended to 300 years of forest management for 4 species. E SPF = Eastern spruce pine fir, W SPF = Western spruce pine fir, DF-L = Douglas fir-Larch. Peak for Douglas fir curve extends to 1120 kg C·m^{-3} wood harvested (not shown in graph)

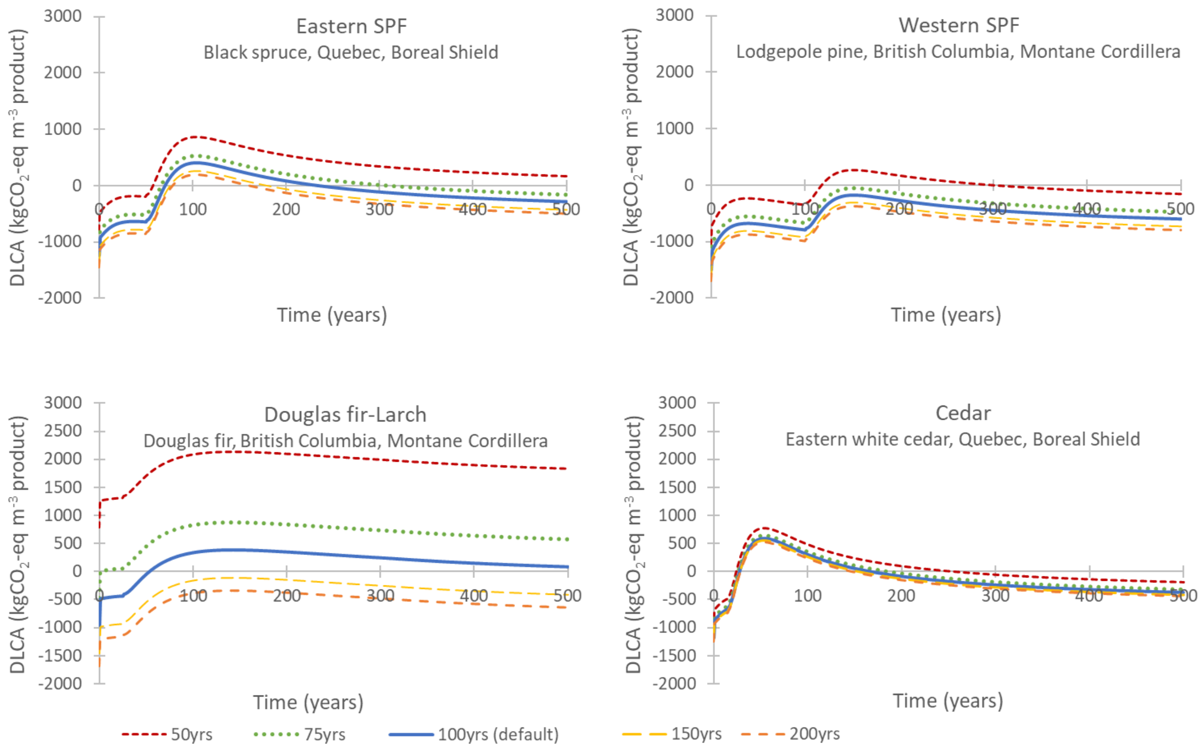

Figure 6.6 – Comparing dynamic LCA results for four cases per m^3 wood product (Eastern SPF, Western SPF, Douglas fir-Larch, Cedar using top tree species) over a variable TH 0 to 500 years, using different ecosystem carbon cost (ECC) values determined after: 50, 75, 100 (default), 150 and 200 years of historical forest management.

Varying the years since historical forest management has an impact on the overall life cycle climate change potentials (Figure 6.6). For all four wood product cases, the shorter the time since historical forest management, the higher net emissions of the curve, thus at 50 years DCCIs are the highest whereas at 200 years they are the lowest. The spread between the scenarios is highly variable between wood species, DF-Larch having the largest spread between the different ECC scenarios, while Cedar has the least amount of difference between ECC scenarios. The differences between wood species result from the differences in ECC values of the scenarios along the ECC curve.

At TH_{100}, DLCA results are inconclusive on carbon neutrality of the wood life cycles. It is important to reiterate that the ECC and C wood uptake values used in these results are for single landscapes and not mixes and thus results differ from those shown in Figure 4. Neither E SPF nor Cedar wood product cases have any ECC values at which carbon neutrality is attained at TH_{100}. In the case of W SPF and DF-L carbon neutrality is only achieved for certain ECC values, all but ECC 50 yrs for W SPF and ECC 150 yrs and ECC 200 yrs for DF-L. Overall carbon neutrality is observed beginning at TH_{200} for the default ECC (100 years), with the exception of DF-L, for which carbon neutrality does not occur until TH_{500}.

6.1.3.4.2 All wood waste from building is incinerated

The default waste management of demolition waste, characterised by landfilling and recycling, is compared with wood incineration with electricity generation substitution using the grid mix and the marginal energy of the regions where the wood is used and treated at end-of-life and wood incineration without energy substitution (Figure 6.7).

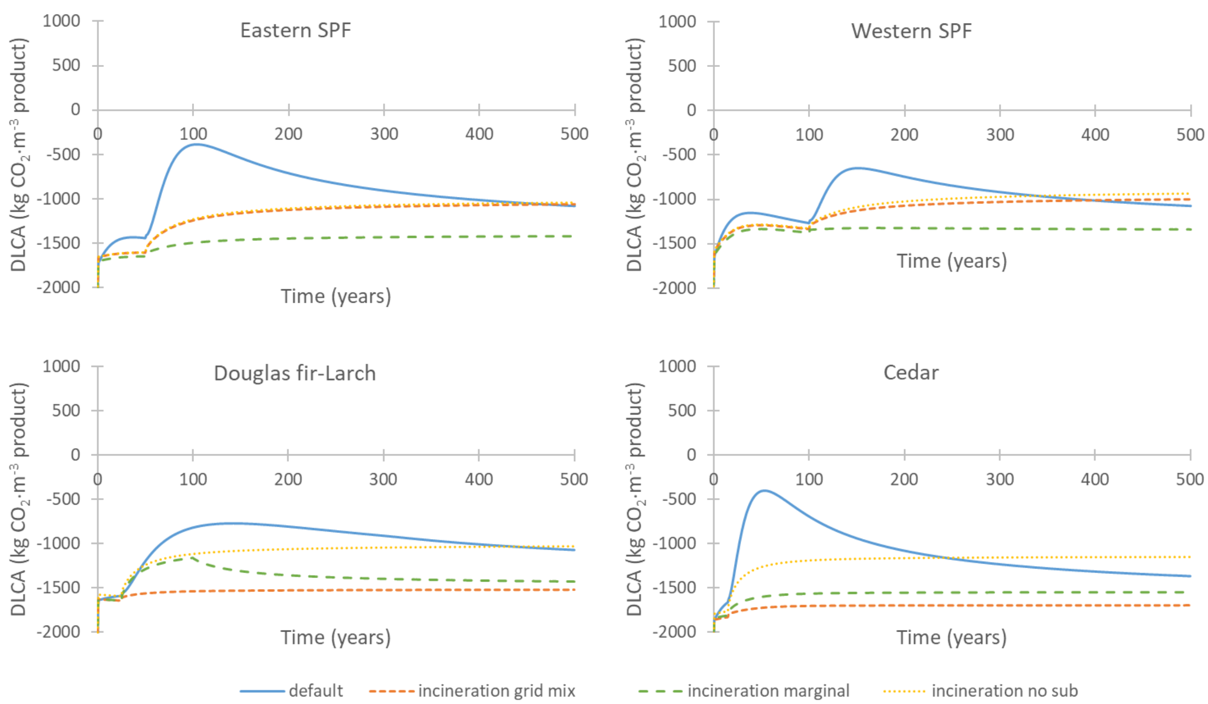

Figure 6.7 – DLCA in kg CO_2-eq·m^{-3} product of four wood cases, comparing default (blue line) with incineration with grid mix substitution (red dotted line), incineration with marginal electricity substitution (green dotted line) and incineration with no energy substitution (yellow line).

The DLCA results for each of the four wood product cases shows that the default scenario has least amount of sequestration throughout the first few hundred years for all four wood product cases.

This is due to few factors: first, 25-37% of carbon in the product is degraded over the timeline, second about 50% of the carbon is emitted as methane, and finally methane has a higher characterisation factor than CO_2 (27.75 times for biogenic according to the Intergovernmental Panel on Climate Change (IPCC) 5th Assessment Report (Myhre et al., 2013)). In the case of incineration, the carbon in the wood is almost entirely emitted as CO_2. The differences between the default scenario and the incineration scenarios at TH_{100} are primarily related to the building lifespan of each wood product case. Beyond about TH_{200}, the default scenario has a lower impact than some of the incineration scenarios, dependent upon the type of energy substitution. The relative DCCI scores for incineration with electricity generation substitution via grid mix and marginal energy production are dependent on the region and energy type. For E SPF and W SPF the incineration with grid mix substitution show less C sequestration than the incineration with marginal energy, as electricity production in Quebec and British Columbia (E SPF and W SPF wood product cases, respectively) are dominated by hydroelectricity production, which tends to have very low carbon intensities. In these two cases, the marginal energy is natural gas and thus incineration has much more benefit than the other waste management treatments. In the case of DF-L and Cedar, the grid mixes (Alberta and Nova Scotia, respectively) are dominated by coal energy production. The relative benefits of electricity substitution vs. no substitution is highly dependent on the type of energy production that is being avoided.

6.1.3.5 Net life cycle climate change impacts of wood products

The four case studies presented in this paper represent only a very small fraction of the total number of life cycle cases that can be modelled with the modular database. In fact, with the combination of forestry landscapes, wood product types, building regions and building lifespans available, the database allows for the calculation of over 270 000 life cycles. Although the case studies demonstrated the use of the database and some indications of DCCI for wood products, no definitive conclusions on the net DCCI of wood products or the relevancy of biogenic carbon neutrality can be draw from such a small sample size. As such, net DCCI of biogenic carbon (ECC and C wood uptake and end-of-life were summed) for all 117 forestry landscapes, seven wood product types, twelve building regions for a selection of five building lifespans (BL001, BL010, BL050, BL100, BL150) at TH_{100}. A statistical spread of life cycle DCCI results was developed for each wood product along the five building lifespans, each sampling the DCCI of 1404 different assembled life cycles. Results in Figure 6.8 are shown for lumber, though results for the six wood products (plywood, glulam, CLT, LVL, OSB, I-joist) are available in the Supplementary Material.

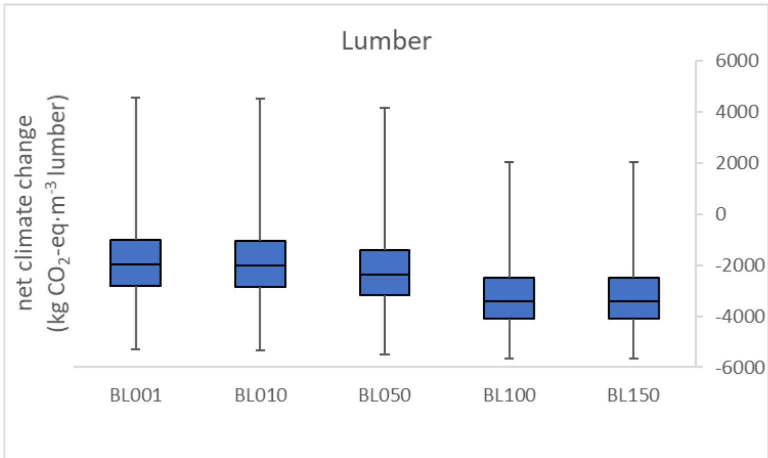

Figure 6.8 – Net life cycle climate impacts of biogenic carbon (ECC, C wood uptake, EOL emissions) for lumber (kg CO_2-eq·m^{-3} lumber). BL = building lifespan, 001, 010, 050, 100, 150 are building lifespan years.

The statistical spread of the results shows that most life cycle cases (boxes) are clustered in the negative axis (Figure 8). However, the outliers at the upper extreme show that certain life cycles yield net positive climate change impacts. The building lifespan has tangible impact on the overall net DCCI, as shown where the box and whisker for longer lifespans yield more net sequestration. It is important to note that at TH_{100}, where end-of-life emissions have not yet been propagated, statistical spreads for BL100 and BL150 are identical. Also, of note is the smaller range of upper and lower values for BL100 and BL150, which is also related to the absence of end-of-life emissions at 100 years.

6.1.3.6 Comparison with other studies

The results from this study are compared with a study by Levasseur et al. (2013) for both carbon sequestration before and after wood harvest (Figure 6.9).

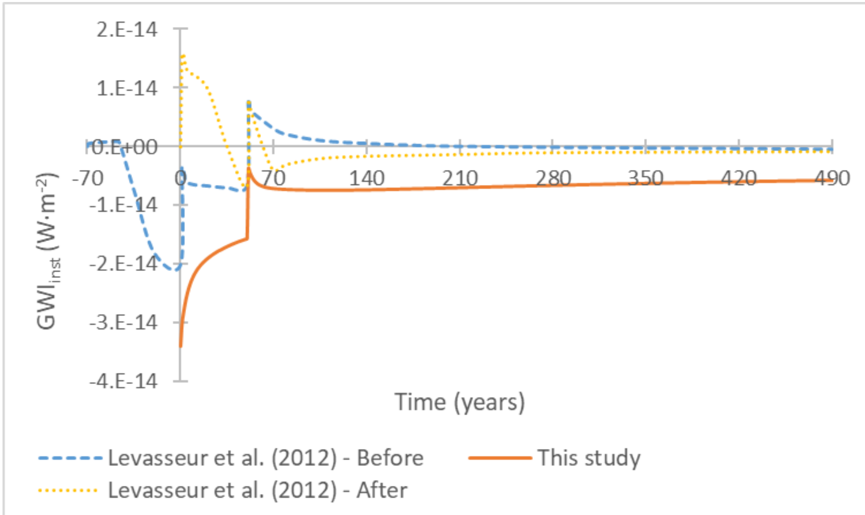

Figure 6.9 – GWI_{inst} results comparing black spruce (Quebec) from this study with Levasseur et al. (2013) for incineration (no energy substitution)

The results for incineration (with no energy substitution) differ between the two perspectives (sequestration timing before and after) from Levasseur et al. (2013) and this study. One major difference is the way in which the forestry phase of the wood product is considered. Illustrative of this are the stark differences in the first hundred years between Levasseur et al. (2013) and this study. Levasseur et al. (2013) consider that the carbon is taken up from a stand perspective (afforestation compared to natural regeneration of forest). The perspective taken for the sequestration timing (before or after) has significant impacts on the curves in the first years of the timeline. When the sequestration timing is considered to occur after the wood is harvested, positive radiative forcing results in the first few decades, before it becomes a net sequestration at year 27. In contrast, the "timing before" perspective results in a negative radiative forcing over the course of the 70-year rotation period. Since the chair lifespan is less than the rotation period (50 vs. 70 years), the sequestration effects in the forest are not visible after 50 years as a result of the beginning of the end-of-life emissions.

In this study, the forestry phase is considered from a landscape perspective, and as such both the ECC and the carbon uptake in the wood is considered in year 0 (the year of harvest), resulting in a single negative pulse emission. As a result of the magnitude of this emission and the fact that the forest management and sawmill emissions are small, the curve remains a negative radiative forcing even at 50 years when wood is incinerated. After 50 years, the curves for all three cases (Levasseur et al. (2013) before and after, this study) slowly flatten out as CO_2 and CH_4 emissions degrade. All three curves result in negative radiative forcing at 490 years, though this study has a lower radiative forcing. As such, the perspective of the forestry carbon dynamics has a significant impact on the overall results of the LCA.

Though this study provides a reasonable estimate of the climate change impacts of wood products in Canada across a multitude of parameters, there are still limitations that can be identified. First, the tracking of carbon through manufacture, use and end-of-life is tied specifically to the average carbon contents of the wood products as reported by Athena SMI (ASMI, 2012a, 2012b, 2012c, 2012d, 2013a, 2013b, 2013c, 2018a, 2018b, 2018c, 2018d, 2018e) and not that of the specific species. This was done in order to limit the already very large database, representing seven wood products, twelve build regions and a range of building lifespans from 1-150 years. The limitation of this approach, however, is that when modelling for a particular tree species, the carbon content of the wood per m^3 may deviate somewhat from the carbon flows resulting from the wood carbon modelled in the later stages of the wood product life cycle.

Second, the base case in this study considers the current (2012 survey) waste management fates of construction wood for each province. Though these rates are not expected to change drastically in a few years, the longer building lifespans would imply that waste management would occur in several decades' time at which time completely different end-of-life fates could be expected. These could be as a result of changes in climate or energy policy, such the federal Clean Fuel Standard (Environment and Climate Change Canada, 2017), which is currently under development by the Canadian government or the Quebec regulation that aims to ban wood landfilling in the coming years (MDDEP, 2011). These regulations could mean that wood that was once landfilled could be diverted from landfills to be used a biofuel. Another possibility is that with wood gaining popularity in the building industry (such as in British Columbia, where the building code has been changed to allow for higher story construction with engineered wood), the cost of wood increases, meaning that wood inputs for lower-grade products (such as particle board) shift from virgin wood or wood processing coproducts to post-consumer wood streams such as construction and demolition wood.

Third, though the first order decay method used in this study is recommended by the IPCC for calculating landfill emissions (IPCC, 2006c), the theoretical model may yield higher degradation rates than in practice. Several authors have shown empirically that wood degrades very slowly and that only up to 8% of the carbon in the wood may be emitted as gas at landfill sites (Barlaz, 2006; Chen et al., 2008; Micales & Skog, 1997; Wang et al., 2011; Wang et al., 2013; Ximenes et al., 2015; Ximenes et al., 2008). Using empirical decomposition rates would be expected lead to fewer emissions from the end-of-life phase and thus would mean that the life cycle climate change impacts of wood products would have even more carbon sequestration. However, due to poor model correlation between theoretical and empirical models, work would need to be done to adjust empirical degradation rates to consider climate conditions in different regions.

In general, wood products show negative climate change impacts, however, since this study was done from an attributional perspective, a negative impact may encourage increased use, which should be assessed from a consequential perspective. A significant increase of wood production could lead to increased pressure on provincial forestry ministries to increase annual allowable cuts, which would increase harvest rates in landscapes, meaning that the ECC curve would “restart” and thus cause a positive climate change emission in the first decades of the new harvest rates. In the unlikely case that demand exceeds annual allowable cuts, potential “leakage” issues could occur, meaning that wood may be sourced from non-sustainably managed forests outside Canada to meet demand. Though some of this wood may come from countries with sustainable wood stocks (the United States, Sweden or Finland) it is likely that increased wood demand would occur in those countries as well, and demand may push for wood to be sourced from countries such as Russia, where forestry practices are often unsustainable (Kissinger et al., 2012; Wyatt, 2013).

At the end-of-life of the base case, wood is recycled at different rates, depending on the construction and demolition recycling rates in each of the wood product case regions. As was mentioned in section 2, a cut-off approach was chosen for wood recycling. There can be large variation in secondary uses for recycled wood, with a large proportion seeming to be downcycled as mulch for landscaping or as daily landfill cover (Kelleher Environmental & Guy Perry and Associates, 2015). These types of secondary uses do not seem to differ greatly from having landfilled the wood in the first place. In addition, they would only substitute for another waste material and not a virgin material. The Clean Fuel Standard could change secondary product fates for recycled wood streams, by using them as biofuel feedstocks. In providing useful energy, the wood feedstocks would substitute fossil fuels. If a readily available, low emissions energy alternative was available via construction wood waste, this could in turn also increase construction wood recycling rates across Canada. The modelling of the emissions savings and potential for increasing recycling rates would be interesting to consider in future research.

6.1.4 Conclusion

The results show that the dynamic life cycle of wood use in building products has overall net negative climate change (sequestration) at TH_{100} for most wood product cases, though certain outlier cases yield net positive emissions. At this time horizon, climate change impacts range from -1264 to -388 kg CO_2-eq·m^{-3} wood product for the wood product cases, affected mainly by wood product lifespans (100 and 50 years, respectively of these two outer limits). Users of the climate change impacts database should consider that outcomes are very dependent on a number of factors, including: the time horizon relevant to the decision in question, the ECC of the wood, the carbon content of the wood, the building lifespan and the end-of-life waste management treatment and thus should ensure that the database parameters chosen reflect the wood product and building project being modelled.

The use of different ECC values resulted in more overall carbon sequestration the longer a forest has been managed. At present, Canadian forests have had a historical management legacy of about 100 years, though in Eastern Canada management started prior to the 20th century and forest management began less than 100 years in Western Canada. The difference between the ECC value choice is highly dependent on the shape of the ECC curves, as shown by the Douglas fir-Larch case having far greater differences between the different years on the curve than the Cedar case. The use of different waste management scenarios can impact the climate change scores, as was shown with comparisons of the default end-of-life management (landfilling + recycling) to incineration.

This research work resulted in a database of LCI and DCCI of Canadian wood products, throughout a range of geographical, temporal and typological contexts. This modular database can be used to develop thousands of different combinations of wood product cases, which allows users of the database to calculate the climate change impacts of specific wood product cases. In addition to allowing the calculation of small set of wood product cases, the database could also be integrated into an LCA tool, such as a building information model (BIM), which would provide building designers with the climate change results needed to more make informed choices on building material selection. The database results could also be used to inform climate policy on building and other long-life wood products.

6.1.5 Acknowledgements

The authors would like to thank project partners Cecobois, Canadian Wood Council, Desjardins, GIGA, Hydro-Québec and Pomerleau as well as the Natural Science and Engineering Research Council of Canada (CRD 462197-13). They would also like to thank Anthony Crockford for helping with some code to run large calculations. The authors declare that they have no actual or potential competing financial interests.

CHAPTER 7 GENERAL DISCUSSION

The objective of this research work was to challenge the carbon neutrality assumption of wood products by developing and applying a framework that accounts for the uptake, emissions, storage and timing of biogenic carbon in the Canadian softwood life cycle. This chapter provides a discussion for how the research work has achieved the sub-objectives and main objective of the doctoral research. In addition, some limitations of the research are identified, and some recommendations are made for future directions that this research topic can take.

7.1 Achievement of research objectives

The main objective of this dissertation *Improvement of Biogenic Carbon Accounting in the Life Cycle of Wood used in Construction in Canada* was achieved through the specific sub-objectives defined for this research.

7.1.1 Sub-objective 1: Develop carbon profiles of forest carbon dynamics for softwood species across Canada

The methodology presented in Chapter 4 (article 1) applies forest carbon dynamics used in forestry science to model the net carbon fluxes of wood harvesting activities in LCA. By allowing a natural (baseline) forest to be established, and then applying a harvest regime of constant sustained harvest through time, the effects of harvest activity on the forest ecosystem can be quantified in terms of net carbon fluxes to the atmosphere. In modelling this net carbon flux (termed in the article ecosystem carbon cost – ECC) for several species covering the managed forest area of Canada, the ECC of most commonly harvested softwood species in Canada is calculated.

The updated results (section 4.1 and Appendix A) find that the ECC per m^3 of wood harvested shows net sequestration in most forest landscapes in Canada, assumed to be exploited for 100 years under a regime of stable harvest. Exceptions to this trend include Engelmann spruce, for which most landscapes have net positive emissions, as well as single outlier landscapes for white spruce, black spruce, eastern white pine and Douglas fir. The weighted mean ecosystem carbon costs from a 100-year-old harvested forest, based on harvested volume by species, range from -0.24 to 0.094 $tC \cdot m^{-3}$ wood harvested.

The implications of these results are that the sustainable harvesting of wood from most Canadian forest landscapes show a net sequestration, beyond what is already sequestered in the harvested wood itself. For many landscapes, the ECC score is close to the amount of carbon sequestered in the wood, which ranges from 0.175-0.300 $tC \cdot m^{-3}$ wood harvested. As such, the beginning of the wood product life cycle is characterised by a net sequestration that for some landscapes can be twice the carbon content of the wood. As will be discussed in section 7.1.3, most ECC values used to calculate the life cycle climate change impacts of wood products in buildings ensure that the scales are tipped sufficiently that the overall climate change results of wood products result in a net carbon sequestration.

Considering that sustainably harvesting forests on the overall biogenic carbon balance for wood products is not zero, forestry carbon dynamics should always be included in the life cycle assessments of wood products. The research work yields ECC values for most softwood species used in Canadian construction that can be used to model the forest ecosystem carbon uptake associated with the wood product.

7.1.2 Sub-objective 2: Develop temporally differentiated carbon flux profiles of wood from harvest to end-of-life

Chapter 5 (article 2) outlines how a carbon tool used for calculating the carbon stocks of wood at a country level can be used to model the harvest to end-of-life (gate-to-grave) biogenic carbon emissions of wood products. Using a model designed to track wood carbon through the anthroposphere, the harvest to end-of-life carbon fluxes of seven wood products were modelled with end-of-life for 12 provinces and building lifespans ranging from 1-150 years. A key deliverable for this research work was a set of carbon profiles that covers gate-to-grave biogenic carbon fluxes of harvested wood products, including wood processing coproduct emissions, construction site waste end-of-life, the emissions delay related to the building lifespan and the eventual end-of-life of the final product. This covers all of the net carbon emissions (in kg C for CO_2 and CH_4) from the wood products, in 2352 combinations of wood products, building region and building lifespan which can be used by LCA practitioners and building designers for custom calculations.

The results find that using wood products in buildings temporarily stores carbon and thus delays emissions in the short-term. Results vary depending on wood product type, end-of-life location and building lifespan time. Though most wood products have a relatively similar emissions profile, certain products have higher emissions due to differences in coproduct treatment (e.g. cross-laminated timber (CLT)) has a low proportion of coproducts than other wood products). Where in Canada a building is built and eventually demolished and disposed of can also have a measurable impact on the end-of-life emissions, as a result of local waste management practices (i.e. recycling rates) and landfill decay rates. Higher construction and demolition wood recycling rates in provinces such as in Nova Scotia (40-47%) and British Columbia (30-42%) reduce the amount of materials going to landfill and thus reduce the stored carbon that can be degraded as CO_2 and CH_4 emissions. Instead this proportion of the wood is transferred to secondary product life cycles, where its fate is allocated to those product life cycles. Landfill decay rates and thus resulting CO_2 and CH_4 emissions are lowest where the climate is colder and drier (Yukon: 0.002 $years^{-1}$) and highest where the climate is warmer and wetter (British Columbia: $k = 0.083$ $years^{-1}$). Finally, the building lifespan can have a large influence on how long emissions are delayed and the amount of cumulative emissions taking place within a chosen time horizon.

The models were also run to consider a dynamic inventory, where emission fluxes can change over a given period. This is relevant for environmental policy, for example, where a change may lead to reduced annual emissions in the period leading up to a policy target year. The two policy scenarios (REC70%: 70% recycling by 2025 and LFG80%: 80% landfill gas capture by 2030) demonstrate significant decreases in end-of-life emissions compared to the status quo, especially when building lifespans extend beyond the policy target years for each respective scenario. REC70% results in lower overall CO_2 and CH_4 emissions due to less wood being sent to landfill and thus more wood being accounted for in other product systems. LFG80% results in a shift from CH_4 to CO_2 fluxes as a result of a proportion of landfill gas being collected and combusted to yield CO_2 emissions via energy utilisation and flaring.

The implications of the results are that the biogenic carbon emissions from wood processing to the end-of-life phases of the life cycle can have variable positive carbon emissions (as CO_2 and CH_4), which are dependent on the specific building parameters. These carbon emissions can be used within the development of a cradle-to-grave emissions inventory and in calculating life cycle climate change impacts. Users wanting to use these emissions profiles in the calculation of the climate change impacts of wood products, are recommended use the LCI and dynamic climate change impact data published along with article 3 (see chapter 6 and section 7.1.3).

7.1.3 Sub-objective 3: Application of dynamic LCA to temporally differentiated carbon flux profiles of wood use

In article 3 (Chapter 6), a modular database of gate-to-gate life cycle inventories and dynamic climate change impacts of wood building products was developed. This database allows for the modelling of dynamic climate change impacts by LCA practitioners, for thousands of wood product specifications, covering Canadian forest landscapes (as defined in sub-objective 1), a range of building specifications (sub-objective 2), as well as additional life cycle phases needed to develop a cradle-to-grave LCA of a wood product.

This database of life cycle inventories and dynamic climate change impacts (reported as instantaneous radiative forcing – GWI_{inst}) is provided as Supplementary Material to the article and four wood product case studies are used to demonstrate how the calculated climate change impact data can be used modularly to develop cradle-to-grave LCAs as well as to provide a discussion on the implications of climate change impacts of wood products. The case study results show that the dynamic cradle-to-grave climate change impacts of wood products used in buildings has overall net negative climate change impacts at a time horizon of 100 years, ranging from -1264 to -388 kg CO_2-eq·m^{-3} wood product (for Western SPF and Eastern SPF cases, respectively). The climate change impacts of the different wood product cases are affected by the species type (ECC and carbon uptake), the building lifespan and the end-of-life waste management treatment. The dynamic nature of the emissions during carbon storage and landfilling are best captured by DLCA which considers the timing of each pulse emission taking place.

The sensitivity of the database choices was tested in two sensitivity analysis cases. Using ECC values earlier and later in the historical forest management legacy timeline, shows that more overall carbon sequestration (net negative climate change impacts) occurs the longer a forest has been managed. End-of-life waste management involving landfilling and recycling, results in less net sequestration than incineration, regardless of the type of energy substitution considered.

The results of this third article encompass all the elements of this research work, allowing for the cradle-to-grave climate change impacts of wood products to be evaluated. In doing so they allow for a verdict to be made on the relevance of biogenic carbon neutrality of wood products. In all but potentially the most outlying cases where ECC scores are positive or have very low levels of sequestration, the overall net life cycle climate change impacts of wood products are negative. This implies using a carbon neutrality assumption for biogenic carbon would be a conservative assumption in the case of wood sourced from Canadian forests by overestimating overall life cycle climate change impacts. On the upstream forestry end of the life cycle, it ignores the benefits that current sustainable forest harvesting has on the wood product level. Further downstream, carbon neutrality disregards the temporary carbon storage benefits of wood in buildings as well as the potentially permanent carbon storage benefits that landfills provide at the end-of-life. With landfills, the amount of CO_2-equivalent emissions, despite the CH_4 emissions released from landfill, simply do not add up to the emissions captured by the trees during growth. As such, compared to net biogenic carbon neutrality, our approach considers two additional carbon sequestration sources in the life cycle that mean that the carbon neutrality principle can be deemed as an oversimplification of carbon emissions.

7.1.4 Discussion of overall research objective

The frameworks established within this doctoral research allow for a full cradle-to-grave assessment of climate change impacts of wood products in the context of the Canadian construction sector. The climate change impact results themselves show that most wood products have net life cycle carbon sequestration and thus life cycle biogenic carbon emissions do not cancel themselves out. These findings add to the mountain of evidence in the literature that help in dispelling the myth that wood biomass and wood products should be considered biogenic carbon neutral.

The Intergovernmental Panel on Climate Change (IPCC) has recognised the important role of forestry and forestry products in climate change mitigation (Edenhofer et al., 2014; IPCC, 2003, 2006d). However, there is often insufficient information on how to achieve sequestration or negative climate change emissions from forestry, especially as it relates to the life cycle assessment of specific forestry products. This doctoral work has contributed to these data gaps in quantifying the climate change mitigation potentials of wood products.

Forestry science has been able to model carbon dynamics at the regional scale and at the single stand-level scale, but often times this modelling does not translate well in the context of modelling the forestry carbon dynamics of a wide variety of forestry products and thus this aspects is not included in product LCAs. This work uses forestry modelling to calculate the carbon dynamics of sustainably harvesting wood in Canada. The carbon fluxes (as ECC) derived from this work allow for the benefits of harvesting in Canadian forests to be attributed to a wide range of softwood products.

Although the temporary storage of carbon in long-life products has been considered by other researchers (see section 2.1.5.1), the authors are not aware of any studies that actually provide operationalised emission flux data on a product basis and certainly no studies cover the Canadian wood products context. This work provides ready-to-use full temporally differentiated carbon flux profiles that model carbon fluxes from wood processing to end-of-life that can be used to model different building use contexts.

And finally, most literature that considers the life cycle climate change impacts of forestry products, calculate LCA results for a few sample cases only. The value of this work is that it considers the dynamic climate change impacts for a wide spectrum of wood product specifications that can be assembled modularly by the user. This research work provides the tools needed to determine the climate change mitigation potential that can be achieved through wood products. It also provides a steppingstone for further analysis of large-scale climate change mitigation potentials through the stimulation of continued and increasing wood use in Canada.

7.2 Limitations to obtained results

While the objectives of the research were achieved, the methodologies that were developed throughout this research and the results that were obtained rest on the validity of a number of assumptions. As such, there are still a few limitations associated with this research, which are explored below.

Harvest and wildfire rates over the 100-year simulation period are based on kNN maps from 2001 due to an absence of historical data to this level of detail. However, in reality, these rates would certainly change over time. The mountain pine beetle posed a large threat to forests, particularly in

Western Canada, in the decade after kNN maps were produced (Natural Resources Canada, 2018; Stinson et al., 2011). Recent disturbances are likely to have diminished the forest's capacity as a carbon sink and thus harvest activities would expect to have a less sequestering effect on the forests with higher ECC values than the ECC values that article 1 (and Appendix A) present. Climate change may exacerbate wildfire and insect epidemics (Kurz et al., 2008a; Kurz et al., 2008b) and may have an effect on the growth rate of certain tree species (D'Orangeville et al., 2018; Taylor et al., 2017), meaning that a sustained harvest in a landscape may result in higher ECC values. Harvest rates may have changed in response to the increased natural disturbances, due to the decrease in non-burned forest in landscapes and of the annual allowable cuts prescribed by the provincial forestry ministries.

Not all softwood species and regions are covered by the studied landscapes. Several landscapes with atypical stock and stock change curves were discarded. This is due to limitations in the theoretical model used for yield curves, those having been approximated mathematically, they may not yield logical results for all species and climate regions. Access to empirical growth curves specific to species and regions may allow additional landscapes to be included. Though this limitation is very real in the representation of ecological landscape of Canada, its importance for the current study is marginal since these landscapes are not large contributors to the production of softwood lumber for the construction sector.

While only the mean values of the Ung et al. (2009) growth curve model and input climate data (McKenney et al., 2016) were used for calculating the growth curves, these also included standard deviations and minimum and maximum values. A statistical spread of values for each landscape could be calculated, however limits to the CBM-CFS3 software made such calculations extremely time-consuming and not feasible. These technical limitations also prevented broader uncertainty analyses to be conducted using methods such as Monte Carlo simulations.

In addition, the CBM-CFS3 model also has a few limitations that have been identified in literature. Kurz et al. (2013) acknowledge that as with all models, CBM-CFS3 does not account for all ecosystem processes. Most significantly, the model does not account for all processes that influence soil and dead organic material carbon dynamics and thus results show far less carbon stock variability than is observed empirically (Metsaranta et al., 2017). This could lead to far more variability in ECC values, which would in turn increase the span of dynamic climate change impacts possible for Canadian wood products.

Mass balances for wood products at the wood processing stage in the model may not be as up to date as possible. Athena SMI has been in the process of updating their wood LCI reports from 2012/2013. At the time of the modelling conducted in this research, only some of the updated reports had been published. In addition, the updated LCI reports did not always have the detail level of the earlier versions. As such, the mass balances often used a combination of old (2012/2013) and new (2018) LCI data in order to reflect both the detail level desired and the need to have the most updated data possible.

Landfill gas fate proportions (oxidation, bioenergy, flaring) is modelled for all of Canada due to very sparse data available at the province level. As such, bioenergy and flaring technologies may be overrepresented in certain regions and underrepresented in others. Emissions from harvest to end-of-life represent the current end-of-life fate of wood in Canadian provinces and territories. Significant increases in recycling rates would require recalculation of the wood product models.

The dynamic characterisation factors used in the calculation of climate change impacts (article 3) used IPCC 5th Assessment Report (AR5) parameters, and are based on atmospheric carbon dioxide, methane and N_2O concentrations from 2014. In addition, based on the approach used in the DynCO_2 tool (Levasseur, 2013), no ozone or stratospheric water interactions between N_2O and CH_4 were modelled. The decision to keep characterisation factors in line with AR5 coefficients and models may influence results in the future. Atmospheric concentrations of CO_2, CH_4 and N_2O have increased since 2014 and it also expected that climate models will be refined by the time AR6 is released in 2021/2022. This could have an effect on the dynamic characterisation factors as well as the climate change impacts calculated in this study. For relevancy and simplicity, the LCIA only includes CO_2, CH_4 and N_2O emissions. In the carbon uptake, ecosystem carbon cost (ECC) and in the harvest to end-of-life carbon profiles, only CO_2 and CH_4 models were considered. The other life cycle phases derived from databases also consider other GHGs, such as HCFCs, CFCs, etc. In all cases, all of these other GHG emissions, except for N_2O represented far less than 1% of climate change impacts thus these were excluded from the LCIA.

It is important to note that this work only covers the climate change impact categories of life cycle assessment. Harvesting wood in Canada may have fewer positive impacts in other life cycle impact assessment categories, such as the midpoint Land Use and its effects on the endpoint Ecosystem Quality. For example, current harvest practices in Canada have had an effect on woodland caribou and wolverine populations (Bowman et al., 2010; Wittmer et al., 2007). As such, it is vital that any communication of positive effects of wood use and current forestry practices provide a caveat that benefits are only in terms of carbon sequestration and not in terms of all environmental impact categories. Despite positive climate benefits, the selection of wood vs. other materials may result in trade-offs in other environmental impact categories.

7.3 Recommendations

As discussed in the limitations section (7.2), the research work has several limitations that should be considered in future research. Recommendations for how to proceed with these limitations are discussed below, in terms of both future research directions as well as in terms of recommendations of how the research outcomes should be used.

7.3.1 Interesting future research directions

One of the main limitations in this work is the basis that was used for the wildfire and harvest rates. As was described above in section 7.2, wildfire and harvest rates were calculated using kNN maps from 2001. These rates were used both retrospectively and prospectively to model the forest through time. As was discussed previously in this work, natural disturbances since 2001 have been particularly damaging to forest carbon in Canada. This increase in natural disturbance rates would likely have a significant impact on the natural baseline and have an impact on the harvest rates. In addition to changing the baseline natural disturbance occurring, this would also likely change the harvest rates and have an influence on ecosystem carbon cost values. Though it is unlikely that empirical wildfire and harvest rates prior to 2001 (especially those dating from the early 20th century) will become available, work could be done to acquire more details for wildfire and harvest rates after 2001. Forestry research and advances in satellite imaging in the last decades could allow for the extrapolation of natural disturbance and harvest rates.

Effects of climate change in the future could also have significant implications on forest carbon dynamics and the ECC values. Warming temperatures and changes in precipitation patterns could mean a further increase in natural disturbances such as insect epidemics and wildfires. As described above, this would change the ECCs. These changing temperatures and precipitation patterns could also affect the amount of biomass accumulated in the forest (affecting the growth curve) and thus affect the amount of carbon harvested and left as forest residues, and ultimately the ecosystem carbon cost values. Since building with wood is likely continue to become more popular in the future, more robust predictions of ECC values will be invaluable. Although a dynamic model would be advisable, technical limitations of the modelling software (CBM-CFS3), limitations to the theoretical growth model and the number of parameters affected by climate change would be expected to make this a challenging prospect. Instead, climate projections as measured by the IPCC could be used to calculate a range of ECC scenarios for the future.

Research on the topic of adaptation of tree species and ecosystems to climate change would also be valuable to guide future forest management policy in using more adaptable species to improve the resilience of forests to climate change.

7.3.2 Recommendations on using research deliverables

This research has yielded carbon flux profiles for the forestry phase (as ECC) and for harvest to end-of-life phases, as well as a modular database of dynamic LCI and climate change impacts for all life cycle phases. Though these data are useful at the research level, their current spreadsheet form makes them less convenient for the building designer. These data could be inputted into a building information model (BIM) for architects or a simple spreadsheet calculator could be developed that would allow the user to select tree species, growing region, wood product type, region of building site and building lifespan.

CHAPTER 8 CONCLUSION AND RECOMMENDATIONS

This research work and its contributions have enabled the achievement of the main research objective of developing a framework that consistently accounts for the uptake, emission, storage and timing of biogenic carbon in the life cycle assessments of wood used in buildings. The contributions of this research work are as follows:

1. Forestry carbon dynamics were used to develop the net carbon fluxes (termed ecosystem carbon costs) of harvesting activities for common softwood tree species across Canada. The ecosystem carbon costs of wood harvested in the present-day show mostly net carbon sequestration (after a historical forestry legacy of 100 years) indicating that harvesting wood in Canada generally has a net climate change benefit.
2. To determine the manufacturing, use and end-of-life carbon balance of wood products, temporally differentiated carbon flux profiles were developed for a range of wood products in different use contexts across Canada. Profile results indicate that the degree of postponement of end-of-life emissions is dependent on the wood product type, the building region and expected building lifespan.
3. A modular gate-to-gate LCI and DCCI database for wood building products was developed, to allow LCA practitioners to calculate DCCI results for thousands of wood product specifications with respect to a chosen time horizon relevant for decision-making. The results show that the dynamic cradle-to-grave climate change impacts of wood products used in buildings generally has overall net negative climate change impacts and are affected by the species type, the building lifespan and the end-of-life waste management treatment.

The results of this doctoral research enable the modelling of full cradle-to-gate climate change impacts of wood products used in construction in Canada. Climate change impact results demonstrate that most wood products have net carbon sequestration for their product life cycle and as such biogenic carbon emissions are not net zero. These results add to the growing evidence that wood biomass and wood products should not be considered biogenic carbon neutral.

Applying a carbon neutrality assumption for biogenic carbon would be considered a conservative assumption by overestimating overall life cycle climate change impacts. In the forestry phase of the life cycle, it ignores the benefits that sustainable forest harvesting has on both Canada's carbon emissions balance as well as on the wood product level. Carbon neutrality also disregards the temporary carbon storage benefits of wood in buildings as well as the potentially permanent carbon storage benefits that landfills provide at the end-of-life.

The limitations of this work are primarily related to the specific contexts of the results, that is that the database and the work underpinning it can only be applied to the Canadian building and wood products context. In particular, the modelling of forestry considers both Northern boreal tree species, as well as the natural baseline presence of wildfire and the very low harvest rates characterised by Canadian forests and forestry management practices. Also, the nature of the carbon model used for modelling the post-harvest biogenic carbon means that small updates to factors such as landfill gas capture or recycling rates need to be done at the model level, which does not allow for quick customisation of these factors by the user.

Future work on this topic could lead to the refining of carbon flux factors to consider a wider array of tree species, wood product types, and future climate change effects on forest growth yields and natural disturbances. In projecting results onto larger policy or structural changes considering increased wood use in a low carbon future, work could be done to adapt results to a consequential perspective. This framework could also be used to model forestry and other life cycle phases of wood products in the context of other countries that have very different forestry management, building sector practices and end-of-life waste management contexts than Canada.

In piecing together these contributions, this dissertation work provides an operational and much needed database of climate change potentials for a plethora of wood product specifications for use in the Canadian construction sector. In doing so, it allows designers, architects and LCA practitioners to model products that are specific to their context, with respect to time horizon that is relevant to decision-making. Also, crucially the overall climate change impacts also allow for a pronouncement on the relevance of using the biogenic carbon neutrality principle for long-life wood product applications, thus demonstrating that biogenic carbon should be accounted for in wood product LCAs.

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APPENDIX A CORRIGENDUM OF ECOSYSTEM CARBON COST VALUES (ARTICLE 1)

Following the review and publication of Article 1, *Forestry carbon budget models to improve biogenic carbon accounting in life cycle assessment*, an error was discovered in the calculation of the ecosystem carbon costs (ECC).

In the calculation of ECC, the tC harvested wood are subtracted at every simulation year in order to isolate the forest carbon from the carbon in the harvested wood (see section 4.2.2.5). In article 1, the tC harvested wood was calculated accounting for both annual wildfire disturbances and harvest, starting from the time when forestry management has initiated, that is, from 0-100 simulations years. No wildfire disturbance prior to the 0-100 year has been considered, to consider the pseudo steady-state condition of the natural forest used as a starting point for ECC calculation, that is 1000 years of forest growth with wildfire only. However, the tC harvested wood should actually be calculated similarly to the forest carbon also accounting for natural wildfire disturbances when modeling the baseline state over the 1000 preceding the harvest period. Disturbances in CBM-CFS3 are calculated using the area affected by disturbance and thus tC values can vary depending on the distribution of age classes in the landscape, and thus are affected by any prior simulation of disturbance. By considering a baseline state with wildfire disturbance (for a period of 1000 years) as a starting point for the 100 years forestry management simulation, the tC harvested are slightly different than the tC that would be harvested without considering wildfire disturbance. A different starting point will also lead to different ECC results after 100 years of forest management as shown in Figure A.1, where we compare results for balsam fir landscapes.

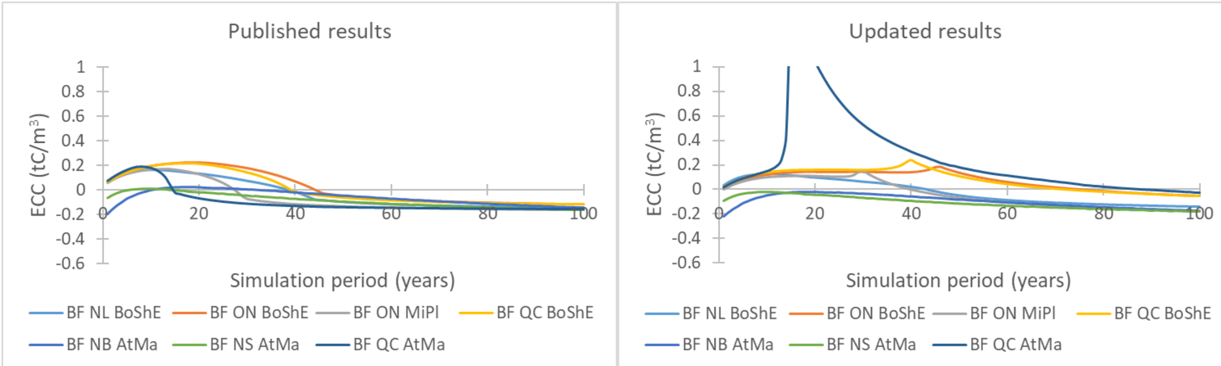

Figure A.1 – Comparison of balsam fir (BF) ECC curves (0-100 years of forest management), in $tC \cdot m^{-3}$ wood harvested, published results vs. updated results

The shape of the ECC curves changes in the updated results for the first few decades of the simulation delaying neutrality at a later point, i.e. certain landscape curves cross the zero line at a later point in the simulation period. In addition, the peaks of the curves are sharper in the updated results, particularly for balsam fir in Quebec Atlantic Maritime region (BF QC AtMa). However, at the 100-year mark, the trend towards negative ECC values is maintained with some notable differences that are detailed in Figure A.2.

The trend in the results shown for balsam fir are also observed for all tree species considered in article 1. Published vs. updated ECC values at 100 years are plotted in a scatter plot to compare these across all 117 landscapes (Figure A.2).

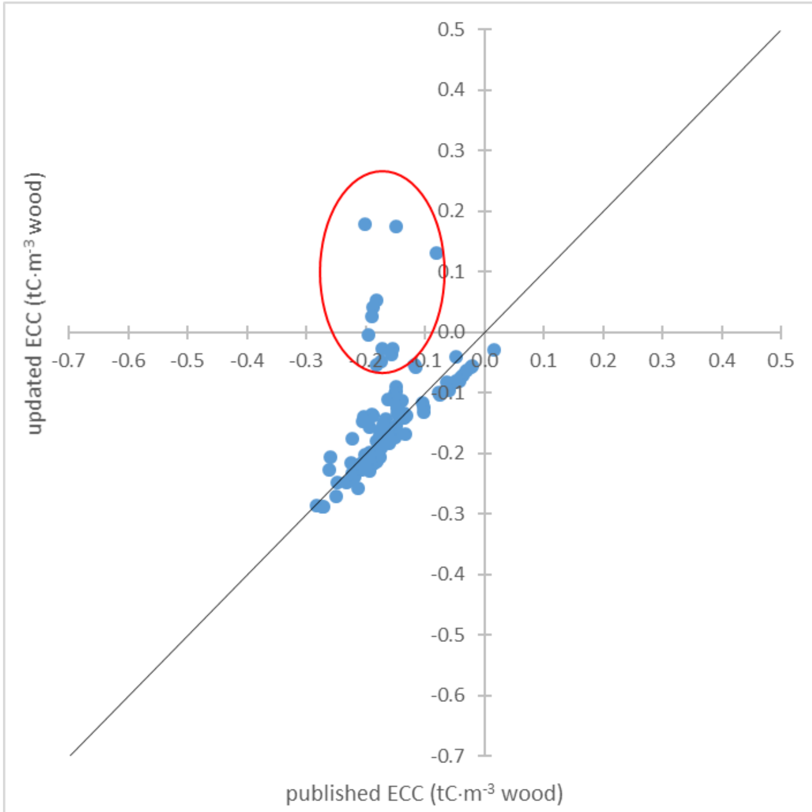

Figure A.2 – Comparison of ECC values for 117 landscapes (12 tree species) at 100 years, published vs. updated results. Diagonal line shows points where values of x axis equal those of the y-axis. Red circle highlights landscapes that do not follow the trend around the x=y line. A single point not visible on the plot is at x=-0.147, y=1.025 (Engelmann spruce, British Columbia Pacific Maritime)

The scatter plot shows that most landscapes still have negative ECC values with the updated values and that almost all the data points follow the line where x=y. Thirteen (13) of the 117 landscapes (circled in red) do not follow the general trend thus showing that these updated ECC values are much greater than the published values.

Table A.1 shows that these thirteen landscapes (in blue) cover a range of tree species, however only a few of regions are represented in this sample (Quebec Atlantic Maritime, Quebec Mixedwood Plains, British Columbia Montane Cordillera and British Columbia Pacific Maritime). These regions are characterised by relatively low wildfire rates and relatively high harvest rates. A forest with very low wildfire rates that has reached a steady state, has much more pronounced carbon fluxes that occur if a relatively high harvest rate is introduced. Net sequestration can be eventually reached, as is shown for Douglas fir in British Columbia Montane Cordillera (Figure A.3), however it is reached after the 100 years of forest management.

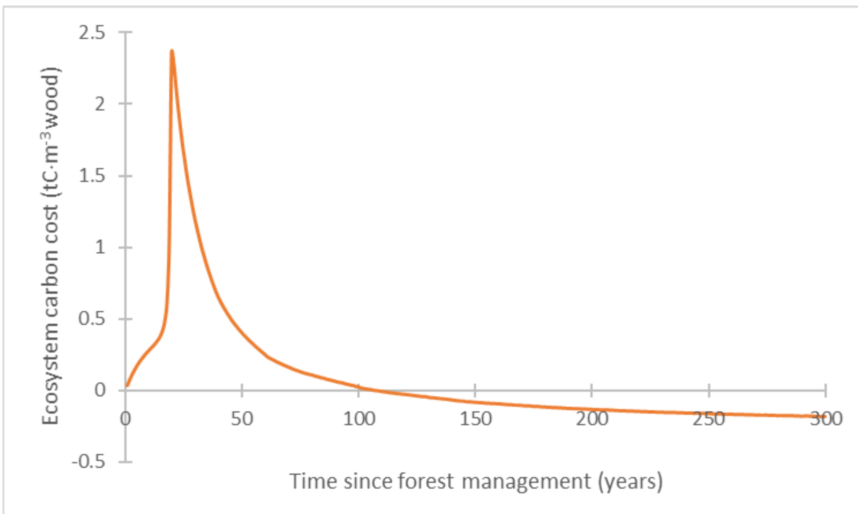

Figure A.3 – Ecosystem carbon cost (ECC), in tC·m^{-3} wood harvested, from 0-300 years

Both the published and the updated ECC results at 100 years are provided in Table A.1, with a guide to the landscape codes used provided in Table A.2.

Table A.1 – Ecosystem carbon cost (ECC) values, tC·m^{-3} wood harvested, published and updated ECC values at 100 years. Blue landscapes indicate the 13 landscapes that do not follow the general trend described above Table A.2 provides a guide to the landscape codes.

Landscape	Publishe	Updated	Landscape	Published	Updated	Landscape	Published	Updated
BF NL BoShE	-0.143	-0.142	EWP ON BoShE	-0.138	-0.113	RP QC BoShE	-0.147	-0.122
BF ON BoShE	-0.117	-0.057	EWP NB AtMa	-0.272	-0.288	RP MB BoShW	-0.076	-0.099
BF ON MiPl	-0.163	-0.172	EWP NL BoShE	-0.148	-0.133	RS QC BoShE	-0.148	-0.090
BF QC BoShE	-0.117	-0.054	JP NB AtMa	-0.223	-0.240	RS ON MiPl	-0.207	-0.227
BF NB AtMa	-0.150	-0.173	JP SK BoPl	-0.180	-0.196	RS NS AtMa	-0.219	-0.238
BF NS AtMa	-0.161	-0.183	JP QC AtMa	-0.198	-0.142	RS QC MiPl	-0.203	-0.139
BF QC AtMa	-0.155	-0.027	JP ON MiPl	-0.201	-0.212	RS ON BoShE	-0.151	-0.101
BS NL BoShE	-0.154	-0.111	JP AB TaShW	-0.180	-0.195	RS NL BoShE	-0.167	-0.144
BS QC MiPl	-0.201	0.178	JP MB BoShW	-0.184	-0.199	RS NB AtMa	-0.232	-0.249
BS MB BoPl	-0.041	-0.075	JP QC MiPl	-0.210	-0.218	RS QC AtMa	-0.195	-0.005
BS NS AtMa	-0.223	-0.232	JP AB BoShW	-0.249	-0.270	RS PE AtMa	-0.193	-0.214
BS SK BoPl	-0.022	-0.058	JP PE AtMa	-0.209	-0.227	WL BC PaMa	-0.260	-0.205
BS MB BoShW	-0.059	-0.090	JP NS AtMa	-0.225	-0.245	WL AB MoCo	-0.283	-0.286
BS QC BoShE	-0.152	-0.111	JP NL BoShE	-0.192	-0.199	WL BC MoCo	-0.262	-0.226
BS SK BoShW	-0.074	-0.102	JP AB MoCo	-0.206	-0.146	WL AB BoPl	-0.225	-0.215
BS QC AtMa	-0.188	0.041	JP ON BoShE	-0.181	-0.179	WS BC BoPl	-0.037	-0.071
BS NB AtMa	-0.208	-0.219	JP AB TaPl	-0.164	-0.179	WS NS AtMa	-0.180	-0.210
BS ON MiPl	-0.192	-0.156	JP SK TaShW	-0.162	-0.176	WS AB BoShW	-0.133	-0.168
DF AB MoCo	-0.222	-0.175	JP AB BoPl	-0.189	-0.205	WS AB BoPl	-0.103	-0.116
DF BC MoCo	-0.189	0.027	JP SK BoShW	-0.175	-0.190	WS ON MiPl	-0.181	-0.213
DF AB BoPl	-0.150	-0.172	JP QC BoShE	-0.169	-0.150	WS BC BoCo	-0.102	-0.125
ES BC PaMa	-0.148	1.025	LP BC BoPl	-0.148	-0.154	WS AB MoCo	-0.162	-0.111
ES BC BoPl	0.015	-0.028	LP BC MoCo	-0.189	-0.136	WS QC BoShE	-0.161	-0.182
ES BC MoCo	-0.149	0.175	LP BC TaPl	-0.201	-0.203	WS PE AtMa	-0.176	-0.206
ES AB BoPl	-0.048	-0.040	LP BC PaMa	-0.187	-0.137	WS NB AtMa	-0.187	-0.217
EWC QC BoShE	-0.135	-0.141	LP AB TaPl	-0.178	-0.180	WS NL BoShE	-0.193	-0.230
EWC NS AtMa	-0.155	-0.172	LP AB BoPl	-0.173	-0.173	WS AB TaPl	-0.101	-0.131
EWC ON BoShE	-0.135	-0.140	LP AB BoShW	-0.249	-0.247	WS QC AtMa	-0.081	0.132
EWC QC AtMa	-0.148	-0.097	LP BC BoCo	-0.157	-0.160	WS SK TaShW	-0.030	-0.062
EWC NB AtMa	-0.200	-0.212	LP AB MoCo	-0.213	-0.257	WS SK BoPl	-0.021	-0.055
EWC MB BoPl	-0.064	-0.082	LP AB TaShW	-0.178	-0.181	WS ON BoShE	-0.156	-0.172
EWC QC MiPl	-0.156	-0.037	LP SK BoShW	-0.131	-0.138	WS SK BoShW	-0.062	-0.096
EWC NL BoShE	-0.135	-0.133	RP QC MiPl	-0.183	-0.053	WS QC MiPl	-0.178	-0.177
EWP QC MiPl	-0.182	0.054	RP ON BoShE	-0.147	-0.125	WS BC TaPl	-0.064	-0.096
EWP MB BoShW	-0.047	-0.082	RP PE AtMa	-0.160	-0.178	WS BC MoCo	-0.169	-0.161
EWP ON MiPl	-0.176	-0.170	RP QC AtMa	-0.175	-0.048	WS AB TaShW	-0.048	-0.081
EWP QC BoShE	-0.142	-0.114	RP NB AtMa	-0.271	-0.288	WS MB BoPl	-0.059	-0.096
EWP QC AtMa	-0.173	-0.026	RP NS AtMa	-0.202	-0.220	WS BC PaMa	-0.167	-0.152
EWP NS AtMa	-0.179	-0.201	RP ON MiPl	-0.176	-0.163	WS MB BoShW	-0.043	-0.081

Table A.2 – Overview of landscape species and region codes

Code	Species	Co de	Province	Code	Terrestrial Ecozones
BF	Balsam fir (Abies balsamea)	AB	Alberta	AtMa	Atlantic Maritime
BS	Black spruce (Picea mariana)	BC	British Columbia	BoCo	Boreal Cordillera
DF	Douglas fir (Pseudotsuga menziesii)	MB	Manitoba	BoPl	Boreal Plains
EWC	Eastern white cedar (Thuja occidentalis)	NB	New Brunswick	BoShE	Boreal Shield East
EWP	Eastern white pine (Pinus strobus)	NL	Newfoundland	BoShW	Boreal Shield West
ES	Engelmann spruce (Picea engelmannii)	NS	Nova Scotia	MiPl	Mixedwood Plains
JP	Jack pine (Pinus banksiana)	ON	Ontario	MoCo	Montane Cordillera
LP	Lodgepole pine (Pinus contorta)	PE	Prince Edward Island	PaMa	Pacific Maritime
RP	Red pine (Pinus resinosa)	QC	Quebec	TaCo	Taiga Cordillera
RS	Red spruce (Picea rubens)	SK	Saskatchewan	TaPl	Taiga Plains
WL	Western larch (Larix occidentalis)			TaShW	Taiga Shield West
WS	White spruce (Picea glauca)				
Code	Terrestrial Ecozones				

The statistical spread of the updated data is provided in Figure A.4, which would replace the results shown in Figure 6 of the published article (shown in Chapter 4). The data is also available in table form (replacing Table 1 of the published article), which can be found in Table A.3.

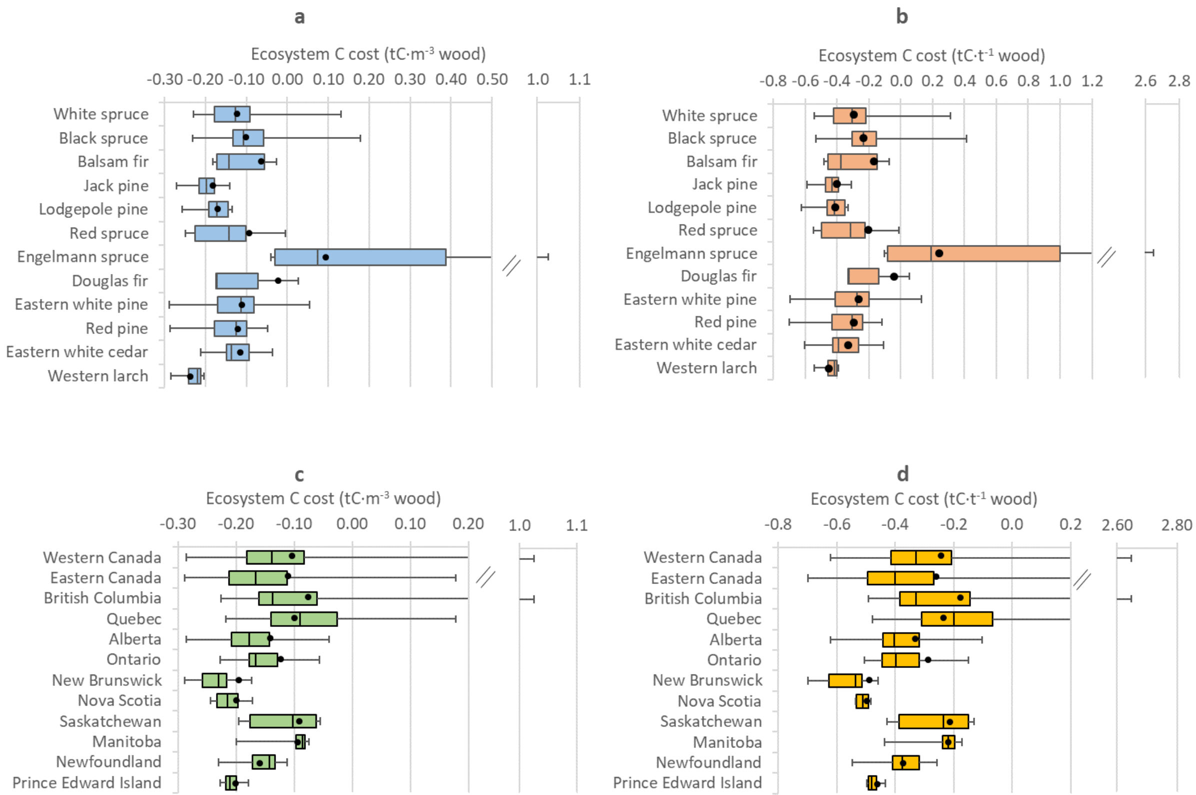

Figure A.4 – Updated results from article 1 (Figure 4.6). Ecosystem carbon costs at year 100 of simulation, for a) per tree species in $tC{\cdot}m^{-3}$ wood harvested, b) per tree species in $tC{\cdot}t^{-1}$ wood harvested, c) per region in $tC{\cdot}m^{-3}$ wood harvested, d) per region in $tC{\cdot}t^{-1}$ wood harvested. The carbon content of the dry wood ranges from 0.175 $tC{\cdot}m^{-3}$ wood harvested (Eastern white cedar) to 0.300 $tC{\cdot}m^{-3}$ wood harvested (Western larch). The lower and upper error bars show the minimum and maximum values, while the lower bound of the box shows first quartile value, the middle line the median value and the upper bound the third quartile value. The round markers indicate the weighted mean values according to approximate annual wood harvest volumes.

The spread of the data, the rankings of the species and provinces and the overall net trends of the updated data is still similar to that of the published data. Most of the interquartile ranges (representing the middle 50% of values) for all species but Engelmann spruce and all regions are within the net negative range of the x-axis, thus has net sequestration. The main change from the published data is in the larger positive whiskers that show high maximum values.

Table A.3 – Updated results from article 1 (Table 1). Ecosystem carbon costs at year 100 of simulation, in $tC{\cdot}m^{-3}$ wood harvested

Category	mean	std dev	wgt mean by harvest volume	minimum	Q1	median	Q3	maximum	C in wood
Western larch (Larix occidentalis)	-0.233	0.036	-0.237	-0.286	-0.241	-0.221	-0.212	-0.205	0.300
Eastern white cedar (Thuja occidentalis)	-0.127	0.054	-0.116	-0.212	-0.148	-0.137	-0.093	-0.037	0.175
Red pine (Pinus resinosa)	-0.144	0.078	-0.121	-0.288	-0.178	-0.125	-0.099	-0.048	0.201
Eastern white pine (Pinus strobus)	-0.119	0.099	-0.111	-0.288	-0.170	-0.114	-0.082	0.054	0.200
Douglas fir (Pseudotsuga menziesii)	-0.107	0.116	-0.023	-0.175	-0.174	-0.172	-0.072	0.027	0.244
Engelmann spruce (Picea engelmannii)	0.283	0.504	0.094	-0.040	-0.031	0.073	0.388	1.025	0.195
Red spruce (Picea rubens)	-0.156	0.082	-0.094	-0.249	-0.227	-0.144	-0.101	-0.005	0.218
Lodgepole pine (Pinus contorta)	-0.179	0.042	-0.171	-0.257	-0.192	-0.173	-0.146	-0.136	0.215
Jack pine (Pinus banksiana)	-0.198	0.034	-0.183	-0.270	-0.216	-0.197	-0.179	-0.142	0.222
Balsam fir (Abies balsamea)	-0.116	0.067	-0.064	-0.183	-0.173	-0.142	-0.056	-0.027	0.175
Black spruce (Picea mariana)	-0.085	0.115	-0.102	-0.232	-0.134	-0.107	-0.058	0.178	0.220
White spruce (Picea glauca)	-0.128	0.077	-0.124	-0.230	-0.178	-0.128	-0.092	0.132	0.195
Prince Edward Island	-0.206	0.020	-0.202	-0.227	-0.217	-0.210	-0.199	-0.178	
Newfoundland	-0.156	0.039	-0.159	-0.230	-0.171	-0.142	-0.133	-0.111	
Manitoba	-0.100	0.038	-0.094	-0.199	-0.096	-0.086	-0.082	-0.075	
Saskatchewan	-0.119	0.054	-0.091	-0.196	-0.176	-0.102	-0.062	-0.055	
Nova Scotia	-0.213	0.026	-0.199	-0.245	-0.233	-0.215	-0.196	-0.172	
New Brunswick	-0.236	0.039	-0.196	-0.288	-0.259	-0.229	-0.216	-0.173	
Ontario	-0.157	0.045	-0.122	-0.227	-0.177	-0.167	-0.129	-0.057	
Alberta	-0.176	0.061	-0.142	-0.286	-0.208	-0.177	-0.142	-0.040	
Quebec	-0.066	0.097	-0.100	-0.218	-0.140	-0.090	-0.027	0.178	
British Columbia	-0.039	0.291	-0.076	-0.226	-0.161	-0.136	-0.060	1.025	
Eastern Canada	-0.144	0.091	-0.110	-0.288	-0.212	-0.167	-0.111	0.178	
Western Canada	-0.114	0.176	-0.104	-0.286	-0.181	-0.138	-0.082	1.025	

APPENDIX B MODEL INPUT PARAMETERS (ARTICLE 1)

Table B.1 provides key climatic and forest management characteristics of each of the 117 landscapes.

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
Balsam fir	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Balsam fir	Quebec	Boreal Shield East	0.0047	0.0036	1.0	1002
Balsam fir	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1156
Balsam fir	Ontario	Boreal Shield East	0.0047	0.0031	2.2	801
Balsam fir	Ontario	Mixedwood Plains	0.0006	0.0006	6.9	938
Balsam fir	Newfoundland	Boreal Shield East	0.0047	0.0032	3.8	1316
Balsam fir	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.4	1382
Black spruce	Manitoba	Boreal Plains	0.0126	0.0007	1.0	513
Black spruce	Manitoba	Boreal Shield West	0.0306	0.0006	-0.3	511
Black spruce	Ontario	Mixedwood Plains	0.0006	0.0006	7.0	951
Black spruce	Quebec	Mixedwood Plains	0.0006	0.0029	5.0	1118
Black spruce	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1156
Black spruce	Saskatchewan	Boreal Shield West	0.0306	0.0012	-0.9	499
Black spruce	Saskatchewan	Boreal Plains	0.0126	0.0017	0.7	452
Black spruce	Quebec	Boreal Shield East	0.0047	0.0036	1.0	1001
Black spruce	Newfoundland	Boreal Shield East	0.0047	0.0032	3.8	1316
Black spruce	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Black spruce	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.4	1382

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape (continued)

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
Douglas fir	Alberta	Montane Cordillera	0.0021	0.0059	0.5	735
Douglas fir	Alberta	Boreal Plains	0.0126	0.0031	2.6	569
Douglas fir	British Columbia	Montane Cordillera	0.0021	0.0034	2.5	752
Engelmann spruce	British Columbia	Montane Cordillera	0.0021	0.0034	1.8	781
Engelmann spruce	Alberta	Boreal Plains	0.0126	0.0031	2.0	578
Engelmann spruce	British Columbia	Boreal Plains	0.0126	0.0014	1.5	632
Engelmann spruce	British Columbia	Pacific Maritime	0.0011	0.002	3.0	1155
Eastern white cedar	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Eastern white cedar	Quebec	Boreal Shield East	0.0047	0.0036	1.8	1007
Eastern white cedar	Newfoundland	Boreal Shield East	0.0047	0.0032	3.2	1176
Eastern white cedar	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1156
Eastern white cedar	Quebec	Mixedwood Plains	0.0006	0.0029	5.0	1122
Eastern white cedar	Ontario	Boreal Shield East	0.0047	0.0031	2.2	802
Eastern white cedar	Manitoba	Boreal Plains	0.0126	0.0007	1.9	580
Eastern white cedar	Manitoba	Boreal Shield West	0.0306	0.0006	1.9	601
Eastern white cedar	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.0	1342
Eastern white pine	Quebec	Boreal Shield East	0.0047	0.0036	1.8	1002
Eastern white pine	Newfoundland	Boreal Shield East	0.0047	0.0032	3.5	1137
Eastern white pine	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1156
Eastern white pine	Ontario	Boreal Shield East	0.0047	0.0031	2.7	824
Eastern white pine	Manitoba	Boreal Shield West	0.0306	0.0006	2.3	619

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape (continued)

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
Eastern white pine	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.5	1370
Eastern white pine	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Eastern white pine	Quebec	Mixedwood Plains	0.0006	0.0029	5.0	1122
Eastern white pine	Ontario	Mixedwood Plains	0.0006	0.0006	7.0	951
Jack pine	Alberta	Taiga Shield West	0.037	0.0016	-1.8	347
Jack pine	Manitoba	Boreal Shield West	0.0306	0.0006	-0.3	511
Jack pine	Manitoba	Boreal Plains	0.0126	0.0007	0.9	510
Jack pine	Ontario	Mixedwood Plains	0.0006	0.0006	6.5	927
Jack pine	Ontario	Boreal Shield East	0.0047	0.0031	2.2	802
Jack pine	Quebec	Mixedwood Plains	0.0006	0.0029	5.4	1099
Jack pine	Quebec	Atlantic Maritime	0.001	0.0023	3.0	1152
Jack pine	Saskatchewan	Boreal Shield West	0.0306	0.0012	-1.0	499
Jack pine	Saskatchewan	Boreal Plains	0.0126	0.0017	0.6	452
Jack pine	Saskatchewan	Taiga Shield West	0.037	0.0016	-2.3	354
Jack pine	Quebec	Boreal Shield East	0.0047	0.0036	1.0	1000
Jack pine	Alberta	Boreal Plains	0.0126	0.0031	1.1	473
Jack pine	Alberta	Montane Cordillera	0.0021	0.0059	1.4	603
Jack pine	Newfoundland	Boreal Shield East	0.0047	0.0032	3.3	1149
Jack pine	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Jack pine	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.1	1241
Jack pine	Alberta	Taiga Plains	0.0175	0.0016	-2.1	442

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape (continued)

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
Jack pine	Alberta	Boreal Shield West	0.0306	0.0001	-0.3	394
Jack pine	Prince Edward Island	Atlantic Maritime	0.001	0.0001	5.9	1124
Lodgepole pine	Alberta	Boreal Plains	0.0126	0.0031	1.2	491
Lodgepole pine	British Columbia	Taiga Plains	0.0175	0.0012	-0.7	522
Lodgepole pine	Saskatchewan	Boreal Shield West	0.0306	0.0012	-0.9	386
Lodgepole pine	Alberta	Montane Cordillera	0.0021	0.0059	0.2	708
Lodgepole pine	Alberta	Taiga Shield West	0.037	0.0016	-1.9	349
Lodgepole pine	Alberta	Taiga Plains	0.0175	0.0016	-2.2	452
Lodgepole pine	British Columbia	Montane Cordillera	0.0021	0.0034	2.1	728
Lodgepole pine	British Columbia	Boreal Cordillera	0.0138	0.0027	-1.5	586
Lodgepole pine	British Columbia	Boreal Plains	0.0126	0.0014	1.1	506
Lodgepole pine	Alberta	Boreal Shield West	0.0306	0.0001	-0.3	393
Lodgepole pine	British Columbia	Pacific Maritime	0.0011	0.002	3.5	1099
Red pine	Quebec	Mixedwood Plains	0.0006	0.0029	5.5	1089
Red pine	Quebec	Boreal Shield East	0.0047	0.0036	2.2	1016
Red pine	Ontario	Boreal Shield East	0.0047	0.0031	3.0	837
Red pine	Ontario	Mixedwood Plains	0.0006	0.0006	6.8	956
Red pine	Prince Edward Island	Atlantic Maritime	0.001	0.0001	5.9	1124
Red pine	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Red pine	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.4	1362
Red pine	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1156

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape (continued)

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
Red pine	Manitoba	Boreal Shield West	0.0306	0.0006	2.5	621
Red spruce	Ontario	Mixedwood Plains	0.0006	0.0006	6.9	950
Red spruce	Newfoundland	Boreal Shield East	0.0047	0.0032	4.1	1276
Red spruce	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1158
Red spruce	Quebec	Boreal Shield East	0.0047	0.0036	1.8	1017
Red spruce	Ontario	Boreal Shield East	0.0047	0.0031	3.1	908
Red spruce	Prince Edward Island	Atlantic Maritime	0.001	0.0001	5.9	1124
Red spruce	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.4	1382
Red spruce	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
Red spruce	Quebec	Mixedwood Plains	0.0006	0.0029	5.2	1107
Western larch	Alberta	Montane Cordillera	0.0021	0.0059	1.9	741
Western larch	British Columbia	Pacific Maritime	0.0011	0.002	3.7	1176
Western larch	British Columbia	Montane Cordillera	0.0021	0.0034	3.4	766
Western larch	Alberta	Boreal Plains	0.0126	0.0031	1.8	621
White spruce	Newfoundland	Boreal Shield East	0.0047	0.0032	3.5	1190
White spruce	Saskatchewan	Boreal Shield West	0.0306	0.0012	-0.9	500
White spruce	Saskatchewan	Boreal Plains	0.0126	0.0017	0.8	449
White spruce	Saskatchewan	Taiga Shield West	0.037	0.0016	-2.3	355
White spruce	Ontario	Boreal Shield East	0.0047	0.0031	2.2	801
White spruce	British Columbia	Boreal Cordillera	0.0138	0.0027	-1.6	592
White spruce	Alberta	Taiga Plains	0.0175	0.0016	-2.0	428

Table B.1 – Fire and harvest disturbance rates (%), Mean annual temperature (MAT, °C) and precipitation (P, mm) for each landscape (continued and end)

Common name	Province/territory	Terrestrial ecozones	Fire intensity (% burned annually)	Harvest intensity (% harvested annually)	MAT (°C)	P (mm)
White spruce	Quebec	Mixedwood Plains	0.0006	0.0029	5.2	1129
White spruce	Quebec	Atlantic Maritime	0.001	0.0023	3.1	1160
White spruce	British Columbia	Montane Cordillera	0.0021	0.0034	2.1	678
White spruce	Alberta	Boreal Shield West	0.0306	0.0001	-0.3	394
White spruce	British Columbia	Pacific Maritime	0.0011	0.002	2.7	883
White spruce	British Columbia	Boreal Plains	0.0126	0.0014	1.2	502
White spruce	British Columbia	Taiga Plains	0.0175	0.0012	-0.6	513
White spruce	Prince Edward Island	Atlantic Maritime	0.001	0.0001	5.9	1124
White spruce	Nova Scotia	Atlantic Maritime	0.001	0.0001	6.4	1382
White spruce	New Brunswick	Atlantic Maritime	0.001	0.0001	4.5	1146
White spruce	Alberta	Montane Cordillera	0.0021	0.0059	0.3	704
White spruce	Quebec	Boreal Shield East	0.0047	0.0036	1.0	1001
White spruce	Ontario	Mixedwood Plains	0.0006	0.0006	7.0	950
White spruce	Manitoba	Boreal Plains	0.0126	0.0007	1.1	515
White spruce	Manitoba	Boreal Shield West	0.0306	0.0006	-0.3	513
White spruce	Alberta	Boreal Plains	0.0126	0.0031	1.2	466
White spruce	Alberta	Taiga Shield West	0.037	0.0016	-1.8	347

APPENDIX C MODEL INPUT PARAMETERS (ARTICLE 2)

C.1 End-of-life fates of wood across Canada

Table C.1 provides the proportions of wood recycled vs. landfilled for at both construction sites and demolition sites (building end-of-life) for different regions across Canada.

Table C.1 – End-of-life fate of clean wood (lumber) and composite/engineered wood (CLT, glulam, I-joist, LVL, OSB, plywood), across Canada. "Construction" refers to waste occurring at the construction site at the beginning of a building's life, whereas "demolition" is waste occurring at the end of a building life. Rec = recycled, Ldf = Landfilled.

Jurisdiction	Solid wood				Composite/engineered wood			
	Construction		Demolition		Construction		Demolition	
	Rec	Ldf	Rec	Ldf	Rec	Ldf	Rec	Ldf
Canada	18%	82%	21%	79%	26%	74%	23%	77%
British Columbia	30%	70%	42%	58%	41%	59%	44%	56%
Alberta	8%	92%	9%	91%	13%	87%	10%	90%
Saskatchewan	1%	99%	1%	99%	2%	98%	1%	99%
Manitoba	4%	96%	4%	96%	6%	94%	5%	95%
Ontario	16%	84%	17%	83%	24%	76%	19%	81%
Quebec	21%	79%	27%	73%	30%	70%	29%	71%
New Brunswick	2%	98%	2%	98%	4%	96%	2%	98%
Nova Scotia	40%	60%	47%	53%	51%	49%	49%	51%
Prince Edward Island	0%	100%	0%	100%	0%	100%	0%	100%
Newfoundland	0%	100%	0%	100%	0%	100%	0%	100%
Northwest Territories	0%	100%	0%	100%	0%	100%	0%	100%
Nunavut	0%	100%	0%	100%	0%	100%	0%	100%
Yukon	0%	100%	0%	100%	0%	100%	0%	100%

Source: Kelleher Environmental, Guy Perry and Associates (2015) Characterization and Management of Construction, Renovation and Demolition Waste in Canada.

C.2 Wood coproduct fates for each wood product

Tables C.2 through C.8 show the end-of-life fates of the coproducts in the wood processing stage of each wood product, in percentages.

Table C.2 – Wood coproduct fates, lumber

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	43.1%				
Bark	8.9%		85%	3%	12%
Planer shavings	6.3%		72%		28%
Sawdust	5.6%		79%	21%	
Pulp chips	34.5%		100%		
Trim ends	0.6%		100%		
Chipper fines	0.2%				100%
Wood waste	0.7%		42%	58%	

Sources: ASMI (2009, 2012d, 2018d)

Table C.3 – Wood coproduct fates, CLT

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	54.0%	87.3%		12.7%	
Bark	9.0%		97.0%	3.0%	
Sawdust	4.4%		99.2%		0.8%
Pulp chips	32.5%		100%		
Chipper fines	0.2%		100%		

Source: ASMI (2013c)

Table C.4 – Wood coproduct fates, glulam

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	50.3%	86.9%	10.9%		2.2%
Bark	8.7%		85.0%	3.0%	12.0%
Planer shavings	2.9%		72.0%		28.0%
Sawdust	4.8%		79.0%	21.0%	
Pulp chips	32.5%		100%		
Trim ends	0.3%		100%		
Chipper fines	0.2%				100%
Wood waste	0.003319		0.42	0.58	

Sources: ASMI (2012a, 2018a)

Table C.5 – Wood coproduct fates, I-joist

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	55.0%				
Bark	6.7%		71%	1.3%	27.7%
Planer shavings	2.1%		72%		28%
Sawdust	1.9%		79%	21%	
Pulp chips	21.1%		100%		
Trim ends	0.2%		100%		
Chipper fines	0.1%		100%		
Wood waste	0.3%		29%	71%	0
Off-spec	2.4%		100%		
Peeler cores	3%		100%		
Wood for hog fuel	6%				100%
Byproducts	1%		100%		

Source: ASMI (2013b)

Table C.6 – Wood coproduct fates, LVL

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	47.3%				
Bark	11.3%		60%		40%
Pulp chips	28.7%		100%		
Off-spec	2.6%		100%		
Peeler cores	10.1%		100%		

Source: ASMI (2013a)

Table C.7 – Wood coproduct fates, OSB

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	79%				
Wood waste	0.3%			100%	
wood for hog fuel	17%				100%
Byproducts	3%		100%		

Source: ASMI (2012b)

Table C.8 – Wood coproduct fates, plywood

Coproducts	% from log	Treatment			
		Main product	Sold	Landfill	Bioenergy
Main product	50%				
Pulp chips	19%		100%		
Peeler cores	9%		100%		
Wood, hog fuel, internal	14%				100%
Wood, hog fuel, external	8%		100%		
Veneer	0%		100%		

Sources: ASMI (2012c, 2018b)

APPENDIX D ADDITIONAL INFORMATION (ARTICLE 3)

D.1 Additional information on sensitivity analyses

D.1.1 Using different ecosystem carbon cost (ECC) values

The default calculations are done using ecosystem carbon cost (ECC) emissions after 100 years of constant sustainable harvest. Although this generally represents the historical forest management situation in Canada (Head et al., 2019a), a more recent initial exploitation of the landscape would attribute higher ECC to the harvest of a given m3 as it would be further up the curve, and thus have a less sequestration as a result. Likewise, if the historical harvest modelled continues beyond 100 years, the ECC would be expected to attain more sequestration and to eventually reach a sequestration steady state.

D.1.1 All wood waste from building is incinerated

Details on the processes used in this sensitivity analysis case are shown in Table D.1.

Table D.1 – Inventory processes for electricity substitution for incineration (grid mix and marginal processes)

Wood type	E SPF	W SPF	DF-L	Cedar
	Lumber	CLT	LVL	Lumber
grid mix	Electricity, high voltage (CA-QC)| market for | Cut-off, S	Electricity, high voltage (CA-BC)| market for | Cut-off, S	Electricity, high voltage (CA-AB)| market for | Cut-off, S	Electricity, high voltage (CA-NS)| market for | Cut-off, S
marginal	Electricity, high voltage (CA-QC)| electricity production, natural gas, conventional power plant | Cut-off, S	Electricity, high voltage (CA-BC)| electricity production, natural gas, conventional power plant | Cut-off, S	Electricity, high voltage (CA-AB)| electricity production, hard coal | Cut-off, S	1 MJ Electricity, high voltage (CA-NS)| electricity production, natural gas, conventional power plant | Cut-off, S

D.2 Common species combinations for softwood products

The four wood mix types used as case studies were composed based on definition of softwood trees species in Canada as defined by the Canadian Wood Council (2019). An overview of these wood species combination is given in Table D.2.

Table D.2 – Common species combinations for softwood products

Wood product	Species Combination	Species included in Combination
Lumber	Douglas Fir-Larch (D.Fir-L or DF-L)	Douglas fir, Western larch
	Hem-Fir or H-F	Pacific Coast Hemlock, Amabilis Fir
	Spruce-Pine-Fir (S-P-F)	White spruce, Engelmann Spruce, Red Spruce, Black Spruce, Jack Pine, Lodgepole Pine, Balsam Fir, Alpine Fir
	Northern Species (North or Nor)	Western red cedar, Red pine, Ponderosa pine, Western white pine, Eastern white pine, Trembling aspen, Largetooth aspen, Balsam poplar
Cross-laminated timber (CLT)	S-P-F	White spruce, Engelmann Spruce, Red Spruce, Black Spruce, Jack Pine, Lodgepole Pine, Balsam Fir, Alpine Fir
Oriented strand board (OSB)	Aspen or Poplar	Trembling aspen, balsam poplar
Plywood	Douglas fir	Douglas fir
	Softwood	Unspecified
	Poplar	Balsam poplar
I-joist	See species for lumber, LVL, OSB	
Laminated veneer lumber (LVL)	Douglas fir, larch, southern yellow pine and poplar.	Douglas fir, Western larch, Southern yellow pine, Balsam poplar
Glulam	Douglas Fir-Larch (D.Fir-L or DF-L)	Douglas fir, Western larch
	Hem-Fir	Western hemlock, Amabilis fir, Douglas fir
	Spruce-Pine	Spruce (all species except coast Sitka spruce), Lodgepole pine, Jack pine

D.3 Net life cycle climate change impacts of biogenic carbon

Figures D.1 through D.7 show the net life cycle climate change impacts of biogenic carbon for all seven wood products. The box and whiskers for each BL (for each wood product) samples 1404 complete life cycle results at a time horizon of 100 years.

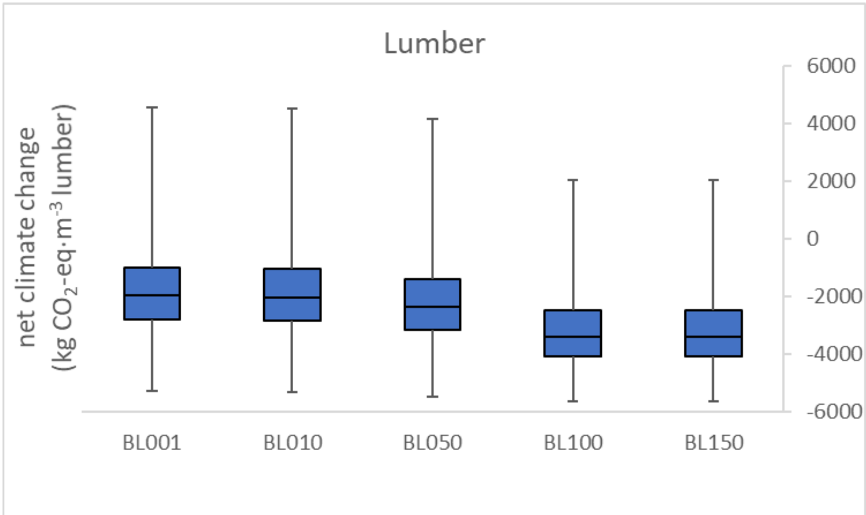

Figure D.1 – Net life cycle climate impacts of biogenic C - lumber

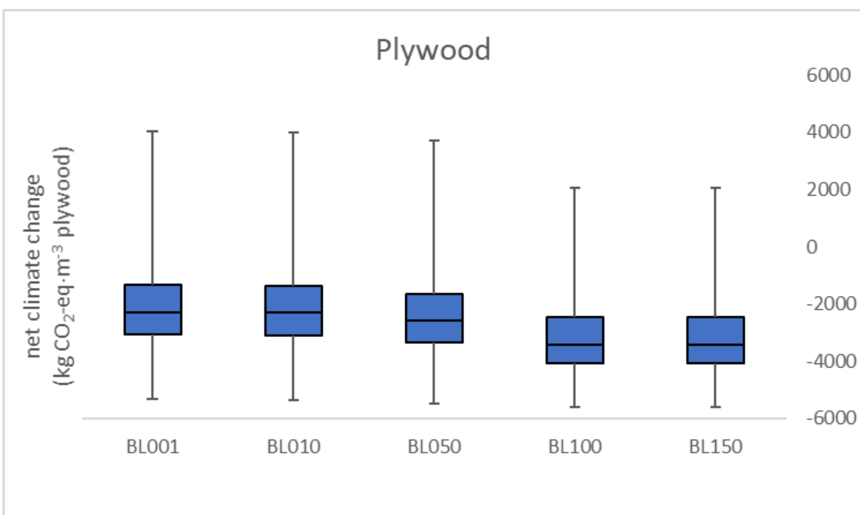

Figure D.2 – Net life cycle climate impacts of biogenic C - plywood

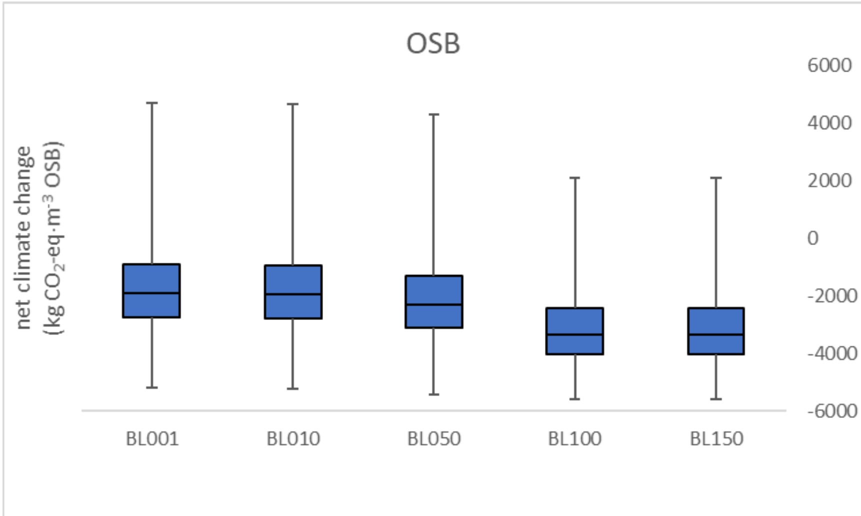

Figure D.3 – Net life cycle climate impacts of biogenic C - OSB

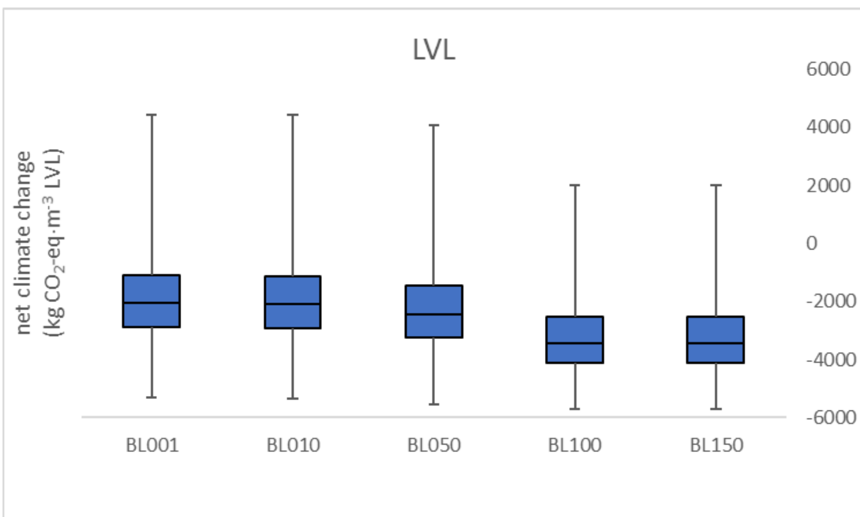

Figure D.4 – Net life cycle climate impacts of biogenic C - LVL

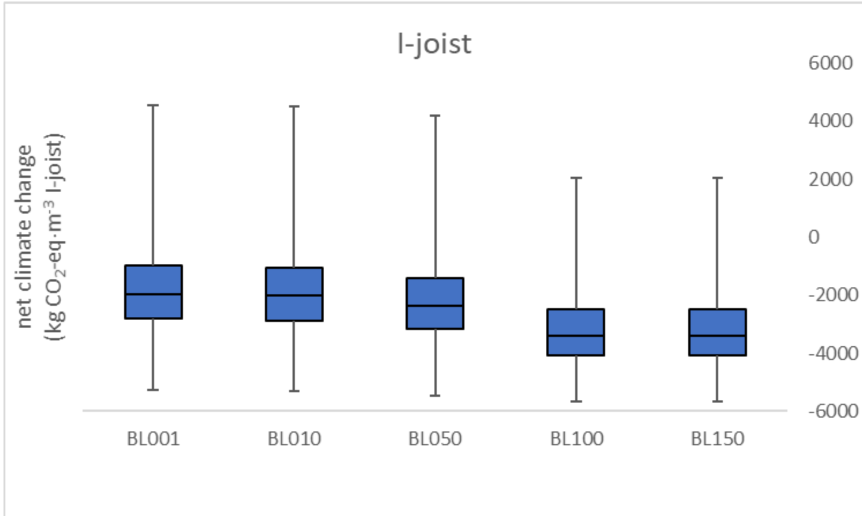

Figure D.5 – Net life cycle climate impacts of biogenic C - I-joist

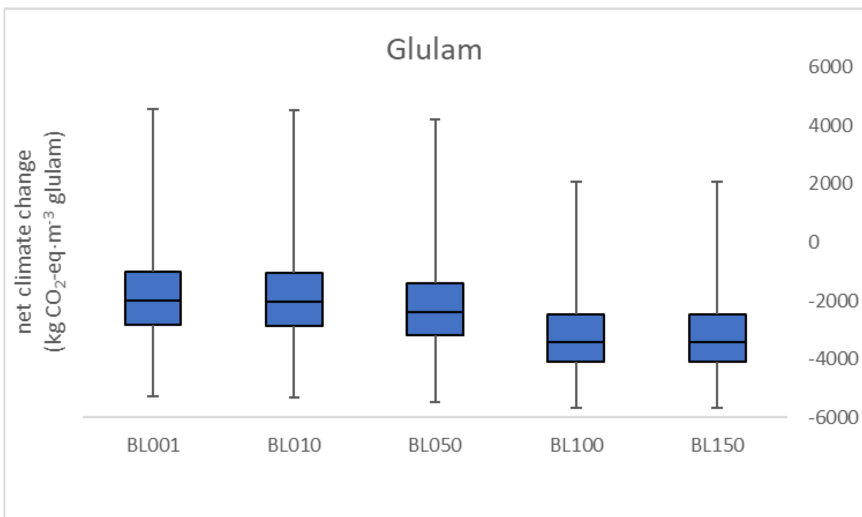

Figure D.6 – Net life cycle climate impacts of biogenic C - glulam

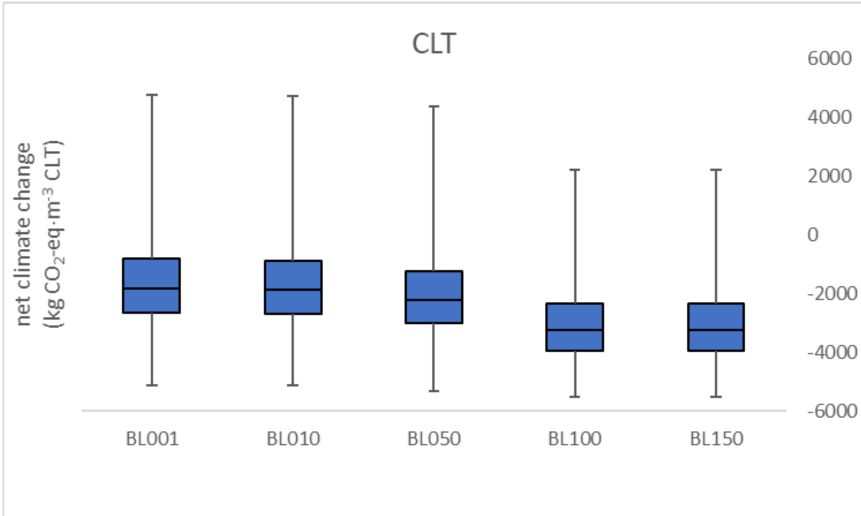

Figure D.7 – Net life cycle climate impacts of biogenic C - CLT