CARBON BUDGET OF FOREST PRODUCTS HARVESTED FROM MOUNTAIN PINE BEETLE-ATTACKED FORESTS IN THE PRINCE GEORGE REGION

by

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LIST OF ACRONYMS & ABBREVIATIONS

AAC	Allowable Annual Cut
AD	Air-Dry
AFOLU	Agriculture, Forests and Other Land Uses
BSKP	Bleached Softwood Kraft Pulp
C	Carbon
CO_2e	Carbon Dioxide Equivalent
CBM-CFS3	Canadian Budget Model of the Canadian Forest Service
GHG	Greenhouse Gas Emission
HWP	Harvested Wood Products
IPCC	Intergovernmental Panel on Climate Change
LCA	Life Cycle Assessment
MBFM	Thousand Board-Feet
MC	Moisture Content
MPB	Mountain Pine Beetle
OD	Oven-Dry
SFM	Sustainable Forest Management
THLB	Timber Harvesting Land Base
TSA	Timber Supply Area
UNFCCC	United Nations Framework on Climate Change

GLOSSARY

Attributional approach:

"Attributional studies attempt to calculate the impacts of the system (i.e., its attributes) looking at the system as it actually exists, normally without regard to other systems or alternative courses of action" (Miner and Gaudreault 2013).

Attributional carbon footprint:

"(An attributional) carbon footprint study is simply a life cycle assessment (LCA) study which is limited to carbon and greenhouse gases [...] Attributional studies are those intended to characterize the system as it actually exists, without consideration of how it may affect other systems" (Miner and Gaudreault 2013).

Biogenic GHG emissions:

"Biogenic CO_2 emissions are defined as CO_2 emissions related to the natural carbon cycle, as well as those resulting from the combustion, harvest, combustion, digestion, fermentation, decomposition, or processing of biologically based materials" (US EPA 2014).

Carbon accounting:

a) "Carbon accounting comprises the recognition, the non-monetary and monetary evaluation and the monitoring of greenhouse gas emissions on all levels of the value chain and the recognition, evaluation and monitoring of the effects of these emissions on the carbon cycle of ecosystems" (Stechemesser and Guenther 2012).

b) “The rules for comparing emissions and removals, as reported, with commitments assumed by Annex I Parties under the Kyoto Protocol. Thus accounting means calculating ‘debits’ and ‘credits’ with reference to the agreed target” (Cowie et al. 2006).

Carbon accounting approach:

“The conceptual framework for estimating emissions and removals of greenhouse gases in inventories. Approach refers to the system boundary, defining which emissions and removals are to be reported or accounted by each Party” (Cowie et al. 2006).

Carbon accounting method:

“The calculation framework within an approach for estimating emissions and removals [...] of greenhouse gases in inventories [...] In practice, (carbon accounting) method refers to the measurement and estimation of GHG emissions [and removals]. That is, the approach defines WHAT is being estimated and reported in an inventory (determined from the system boundary) while the method describes HOW the reported values are derived, that is, the techniques used in estimation. An approach can make use of any method, and within each approach, there may be more than one method” (Cowie et al. 2006).

Carbon neutral (-ity):

a) **Inherent carbon neutrality** – “Biomass was only recently removed from the atmosphere; returning it to the atmosphere merely closes the cycle” (Malmsheimer et al. 2011).

b) **Carbon-cycle neutrality** – "If uptake of carbon (in CO_2) by plants over a given area and time is equal to emissions of biogenic carbon attributable to that area, biomass removed from that area is carbon-cycle neutral" (Malmsheimer et al. 2011).

c) **Accounting neutrality** – "If emissions of biogenic CO_2 are assigned an emissions factor of zero because net emissions of biogenic carbon are determined by calculating changes in stocks of stored carbon, that biogenic CO_2 is accounting neutral" (Malmsheimer et al. 2011).

Carbon sink:

"Any process or mechanism which removes a greenhouse gas (GHG), an aerosol, or a precursor of a GHG from the atmosphere. A given pool (reservoir) can be a sink for atmospheric carbon if, during a given period, more carbon is moving into it than is flowing out" (Cowie et al. 2006).

Carbon source:

"Any process, activity or mechanism that releases a greenhouse gas (GHG), an aerosol or a precursor to a GHG into the atmosphere" (Cowie et al. 2006).

Climate change mitigation:

a) "The Intergovernmental Panel on Climate Change (IPCC) defines mitigation as the implementation of policies to reduce GHG emissions and increase sinks" (Lemprière et al. 2013).

b) “…the amount of reduction in GHG emissions or increase in removals that can be achieved by a mitigation activity relative to a baseline or reference case in a given time period at a given cost per tonne” (Lemprière et al. 2013).

Consequential approach:

“Consequential studies attempt to calculate the impacts resulting as a consequence of using the system (often compared to pursuing a “business as usual” course of action)” (Miner and Gaudreault 2013).

Displacement factors:

See *Substitution Factor.*

Greenhouse gas (GHG) emission:

“Greenhouse gases are those gaseous constituents of the atmosphere, both natural and anthropogenic, which absorb and emit radiation at specific wavelengths within the spectrum of thermal infrared radiation emitted by the Earth’s surface, by the atmosphere itself, and by clouds. This property causes the greenhouse effect. Water vapour (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), and ozone (O_3) are the primary greenhouse gases in the Earth’s atmosphere” (Stocker et al. 2013).

Harvested wood products:

“Wood products are defined as all wood-based material transported from the forest at harvest” (Cowie et al. 2006).

Life cycle assessment (LCA):

a) Life cycle assessment is a "… compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006).

b) "Life cycle assessments (LCA) reduce the many LCI measures into risk indexes affecting human or ecosystem health with the objective of making comparisons between alternatives that reveal opportunities for improvement" (Lippke et al. 2011).

Life Cycle Inventory (LCI):

"Life cycle inventories (LCI) measure every input (energy, materials etc.) and every output(emissions, waste, product and co-products) for every stage of processing from extraction or regeneration through processing, ultimate use, maintenance and disposal" (Lippke et al. 2011).

Lumber Recovery Factor (LRF):

Lumber recovery factor (LRF) is a metric used to indicate a mill's efficiency at producing lumber. In British Columbia, lumber recovery is defined as the amount of lumber recovered in thousand board feet from a specific amount of roundwood in cubic meters (MFLNRO 2015c).

Merchantable timber:

"A tree or stand that has attained sufficient size, quality and (or) volume to make it suitable for harvesting. Timber that (a) was older than 75 years on January 1, 1975, and (b) is on an area of Crown land in sufficient quantities (as determined by the regional manager) to be commercially valuable when the timber cruise is submitted" (MFR 2008).

Merchantable volume:

"The amount of sound wood in a single tree or stand that is suitable for marketing under given economic conditions" (MFR 2008).

Moisture content on an oven-dry basis (MC_{OD})

MC_{OD} = 100 x weight of water / oven-dry weight (Briggs 1994)

Moisture content on a wet or original basis (MC_W)

MC_W = 100 x weight of water / original weight (Briggs 1994)

National greenhouse gas inventory:

"...accounting conventions for preparing greenhouse gas inventories submitted by nations under the United Nations Framework Convention on Climate Change (UNFCCC). The guidelines for developing these greenhouse gas inventories are issued by the Intergovernmental Panel on Climate Change (IPCC) and were most recently updated in 2006" (Miner and Gaudreault 2013).

Non-merchantable forest types:

"Stands that are accessible and otherwise available for harvesting, but are assumed to be non-merchantable because of stand characteristics (e.g., small piece size, incidence of decay, species composition, and low stocking)" (MFR 2008).

Oven-dry weight:

"The only case when wood contains no moisture is when it is kept in an oven above 100°C. In this environment all water is eliminated and the wood is referred to as *oven dried.*" (Briggs 1994).

Roundwood:

"Any section of the stem, or of the thicker branches, of a tree of commercial value that has been felled or cut but has not been processed beyond removing the limbs or bark, or both, or splitting the section (for fuelwood)" (MFR 2008).

Substitution (emission) factors:

"Substitution refers to replacing product A with product B, such as substituting wood for cement or biofuel for fossil fuel. Substitution replaces the LCI footprint of A for B and may cause additional indirect impacts as output volumes adjust. Each LCI has a system boundary (Lippke et al. 2011).

Sustainable forest management:

"Management that maintains and enhances the long-term health of forest ecosystems for the benefit of all living things while providing environmental, economic, social, and cultural opportunities for present and future generations" (CCFM 2015).

Timber harvested landbase (THLB):

"The Crown forest land base consists of Crown land with forest cover within the timber supply area (TSA), excluding tree farm licences (TFL), community forests, woodlots and private lands. The timber harvesting land base (THLB) is that portion of the Crown forest land base that does not include: protected areas; areas deemed uneconomic for the protection and conservation of other forest values, such as wildlife, habitat, biodiversity, recreation, etc.; and areas with unstable terrain, roads, etc." (MFR 2008).

Timber supply area (TSA):

"A geographically based administrative area designated under the Forest Act (Section 7). Timber supply areas have an allowable annual cut as set by the Chief Forester, and are used to provide a sustainable flow of timber to both replaceable and non-replaceable forms of volume-based tenures" (MFR 2008).

Secondary structure:

Secondary structure consists of "tree seedlings, saplings, sub-canopy and canopy trees that will likely survive a pine beetle attack" (Coates et al. 2006).

Shelf life:

Shelf life refers to the window of opportunity the wood products industry has in utilizing a MPB-attacked stand given their "changes in wood properties and vertical stand structure over time" (Lewis and Hartley 2006). It encompasses "the rate and process of wood degrade, decay, and fall-down, and (biological) variables that influence these (vertical stand structure) changes with time" (Lewis and Hartley 2006). But it also addresses the changes in wood properties as they relate to the numerous non-biological variables (e.g. technology, market price, cost) that influence the wood products industry (Lewis and Hartley 2006).

Shrinkage:

"When wood dries below a certain moisture content (MC), referred to as the fiber saturation point (fsp), it begins to shrink and continues to do so until it is oven-dry. Conversely, wood that is below fsp will swell as it takes on moisture and this will continue until fsp is reached. Changes in moisture content above fsp have no effect on shrinkage and swelling. Fsp varies among species, but a value of 30% MC_{OD} (23% MC_{W}) is commonly assumed" (Briggs 1994).

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Chapter 1.

INTRODUCTION & LITERATURE REVIEW

1.0. INTRODUCTION

There is widespread agreement on the need to address human-induced climate change (Steffen et al. 2011, Barnosky et al. 2012). Anthropogenic greenhouse gas (GHG) emissions have increased rapidly since the pre-industrial era at a rate unprecedented in Earth's geological history (Steffen et al. 2011). These GHG emissions are one of the predominant anthropogenic drivers in human-induced climate change, and left unabated, they risk further warming and long-lasting changes to the climate system (Stocker et al. 2013). Recent international negotiations have focused on curtailing these emissions and have devised a series of reporting schemes and policy mechanisms for countries willing to commit themselves to these efforts (UNFCCC 2016).

Forests (and forest-related) activities have been proposed as an effective area within which to enact climate change policy reform (Nabuurs et al. 2007). While many of these proposals seek to develop policies that prevent deforestation in tropical regions, there is growing interest globally to decrease GHG emissions through more intensive forest and forest product management. This notion has led into an exploration of potential activities that could aid in these efforts of climate change abatement (Dymond 2012, Chen et al. 2014, Smyth et al. 2014). In many countries, such as Canada, these discussions have also included additional goals of climate change adaptation and mitigation (Kurz et al. 2013, Lemprière et al. 2013, Gauthier et al. 2014).

1.1. LITERATURE REVIEW

1.1.1. Contribution of Forests in Climate Change Mitigation Efforts

Forests play an important role in regulating the global climate system (Bonan 2008). They influence this system through various processes that affect the planet's energetics, hydrological cycle, and biogeochemical cycles. These processes have complex interactions, which ultimately can have either beneficial or adverse effects on the climate system. This concept is exemplified in Bonan (2008), whereby global forest reforestation and afforestation efforts benefit climate change by sequestering carbon dioxide (CO_2) and by providing evaporative cooling in tropical forests, but adversely affect the system by lowering the albedo of boreal forests. These phenomena represent known processes affecting the climate system but their net climate forcing has been difficult to quantify and include in climate change policy (Bonan 2008).

Instead, mitigation efforts have largely focused on the forest carbon (C) cycle and its contribution to global C cycling (Nabuurs et al. 2007). The forest C cycle continuously sequesters (via photosynthesis) and respires (via auto- and heterotrophic respiration) vast amounts of carbon dioxide (CO_2), a known greenhouse gas (GHG) agent. The net result of these CO_2 fluctuations has offset approximately one-third of annual global GHG emissions since 1990 (Pan et al. 2011). Yet direct and indirect forestry activities (e.g. agricultural expansion, deforestation) are also responsible for emitting large quantities of GHG emissions annually (Smith et al. 2014). Therefore, the management of this natural resource is an opportunity for simultaneously increasing C sequestration while decreasing C emissions in forests and in forest-related activities.

1.1.2. The Mountain Pine Beetle (MPB) Outbreak in British Columbia

The mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins) epidemic marks one of the largest natural disturbances ever recorded in the province of British Columbia (BC), Canada (Safranyik and Carroll 2006). This native insect reached its epidemic phase by surpassing a series of ecological thresholds (Raffa et al. 2008, Bentz et al. 2010). The successful bark beetle kills mature pine trees directly by overcoming a tree's chemical defenses and indirectly by inoculating the host with a blue stain fungus (Safranyik and Carroll 2006). The preferred tree host for the beetle is mature lodgepole pine (*Pinus contorta* Dougl. ex Loud. var. *latifolia* Engelm.). In the years following attack, the infected tree host dies and quickly loses its moisture content, resulting in checking or cracking of the wood (Lewis and Hartley 2006). This damage to the timber, in addition to the unaesthetic blue stain in the sapwood, decreases the product value recovery in BC's forest products industry (Bogdanski et al. 2011).

Currently, the MPB outbreak has attacked over 723 million m^3 of trees and 18.3 million hectares of BC forests (MFLNRO 2013). Susceptible lodgepole pine trees are not, however, distributed evenly across the forested landscape. As a result, British Columbia's timber harvesting land base (THLB) has an uneven distribution in the severity of beetle-attacked trees. The THLB is divided into individual timber supply areas (TSA) and covers an area of 22 million hectares (MFML 2010). In response to the MPB epidemic, the TSAs hardest hit by the MPB have had uplifts in their allowable annual cuts (AAC) (MFLNRO 2012a). The purpose of these uplifts is meant to harvest wood from these attacked stands before they become uneconomically feasible to salvage and restock.

In 2001, it was estimated that there was approximately 2.2 billion m^3 of timber on THLB areas, with pine consisting of approximately half of this volume (FPB 2014). Since then, industry has harvested over 500 million m^3 – 60% of which has being dedicated to the salvaging of pine. Meanwhile, various researchers have advocated for better retention of well-stocked MPB-attacked stands during these salvaging efforts – rather than simply harvesting pine and/or pine-leading stands (Coates et al. 2006, Dhar and Hawkins 2011). It is thought that these well-stocked forests, if protected, would alleviate some of the predicted slumps in the upcoming midterm timber supply gap (Coates et al. 2006, Burton 2010, MFLNRO 2012b). Whether or not these recommendations were fully taken into consideration has yet to be seen. Forestry audits by the Forest Practices Board (FPB) (2007) found general adherence to provincial recommendations (Eng 2004, Snetsinger 2005) in earlier years, but has since found some neglect (MFLNRO 2011, FPB 2014).

The purpose of previous and current AAC uplifts is meant to encourage the explicit salvaging and restocking of public forests attacked by the MPB outbreak (MFLNRO 2012a). This has meant consciously overharvesting the long-run sustained yield of timber in heavily impacted TSAs (Burton 2010), with the intention of mitigating future economic and social losses with current economic and social prosperity.

In recent discussions focused on addressing the issue of a timber supply shortage, new areas of previously unmanaged forested areas are being considered for harvest to mitigate the drop in timber supply (MFLNRO 2012b). These areas included previously protected areas (e.g. old-growth forest management areas, forest retained to meet visual quality objectives, ungulate winter range), which were set in place to ensure a diverse range of values from forests. Equitable forest governance thus will be a challenge moving forward as conflicts arise from various forest stakeholders as the timber supply becomes constrained in the coming years (Burton 2010). Moreover, the concept of sustainable forest management and its certification faces adversity during these times, as the sustainability of forestry in the province has not been demonstrated.

With that in mind, a prominent alternative to increasing the THLB is increasing the utilization of BC's forest resource (Special Committee on Timber Supply 2012). Proponents of this concept advocate for more intensive management of harvest residues (i.e. tree tops and slash, broken and unmerchantable logs) for bioenergy purposes. It is thought that this could help salvage additional revenue, which could help leverage the overall feasibility in salvage logging campaigns. Such notions are often predicated on supplying a low GHG emission fuel (i.e. harvest residue) to emerging bioenergy industries in domestic and international markets. To date, however, the emergence of these industries has been supported by an abundance of mill residues – which were formerly considered wood waste in the province. The extraction of harvest residues, on the other hand, requires increased expenses in obtaining wood fibre from the forest, and they are not considered to be economically feasible (Abbas et al. 2011). [With that said, the provincial government has recently created special forest tenures for parties interested in pursuing such goals (MFLNRO 2012c, 2015b).]

A more controversial forestry practice meant to intensify forest management is the explicit harvesting of forests solely for bioenergy (Greenpeace 2011, McKechnie et al. 2011, RSPB 2012). In the post-epidemic phase of the MPB outbreak, it is expected that there will be an abundance of MPB-attacked forests that are not economically feasible to harvest for traditional forest product industries, but will slowly release C as the dead trees decompose. Under this paradigm, the use of bioenergy markets to pay for the salvage and restocking of these forests is a convincing argument. However, this conviction largely depends on whether or not there are any perceived climate change mitigation benefits in such activities (Lamers et al. 2014).

In summary, the MPB outbreak has shone light on a difficult question for forestry policymakers: whether or not to promote bioenergy (and thereby climate change mitigation objectives) in their continued salvage logging campaign. Such questions are complex, and difficult to answer given the current disparity between environmental science research and government policy on this issue. For example, the provincial government currently reports the GHG emissions from forests and forest product industries separately in its GHG inventory report. This decision in turn, makes it difficult for environmental science research to inform the current situation of the MPB outbreak and to promote the bioenergy industry in a clear and timely manner.

1.1.3. Estimation of Forest Greenhouse Gas (GHG) Emissions

Since 1992, multiple nations have joined an international treaty, the United Nations Framework Convention on Climate Change (UNFCCC), which recognizes the importance of international collaboration in reducing global greenhouse gas (GHG) emissions (UNFCCC 2016). This convention specifies a universal framework for nations to collaborate in climate change mitigation and adaption activities. These international negotiations coincide with another treaty, the Kyoto Protocol, which is an internationally binding commitment to emission reduction targets. Canada withdrew from this latter commitment in 2011, but continues to estimate forest GHG emissions at national and provincial levels in accordance with the UNFCCC (MOE 2012, EC 2015).

In this context, Canada's managed forest has intermittently acted as both a C sink (contributing to net C storage) and as a C source (resulting in net C emissions) over the past few years (EC 2015). Canada does not, however, report these emissions in its National Inventory Report, which is annually submitted to the UNFCCC. The threat of natural disturbances (e.g. fire, insect outbreaks) puts the nation's forest C balance at risk and creates uncertainty in their estimation and reporting (Kurz et al. 2008a, 2008b, Metsaranta et al. 2010, EC 2015). Additionally, these natural disturbances are also expected to increase in frequency and severity as global temperatures increase in the northern hemisphere (Price et al. 2013). As a consequence, there is interest in reducing this uncertainty by adapting to climate-induced changes in forests and by mitigating their overall impact (Lemprière et al. 2013, Gauthier et al. 2014). Current disturbances most affecting Canada's forest C balance are wildfires, the MPB outbreak in British Columbia, and annual timber harvesting (EC 2015).

1.1.3.1. GHG Emissions from MPB-Attacked Forests

Historically (1990-2002), BC's managed forest has been estimated as a C sink (net C storage) in its provincial GHG inventory, using the Carbon Budget Model of the Canadian Forest Service (CBM-CFS3) (MOE 2012). Since 2002, however, provincial forests have transitioned into a C source (with net C emissions). This transition from sink to source has been attributed to the MPB outbreak, wildfires and increases in annual harvesting. In 2012, forest emissions were estimated as: -49.2 Mt CO_2 being sequestered by net primary productivity and decay of organic matter, 63.1 Mt CO_2 being emitted by harvesting, 17.1 Mt CO_2 being emitted by wildfires, and 8.0 Mt CO_2 being emitted through slash burning (MOE 2012)[1]. This C accounting approach presents a "snapshot" of the

[1] Negative values here connote GHG removals, while positive values connote GHG emissions.

current C balance of BC forests, but it does not inform policy makers of the total C emissions attributable to the MPB outbreak, nor any proactive forest management activities that could help alleviate the damages caused by the beetle outbreak.

Kurz et al. (2008a) predict that the MPB-attacked region will be a C source both during, and immediately after, the beetle outbreak. The net outcome of this outbreak, therefore, is predicted to be a large source of GHG emissions to the atmosphere (~900 $MtCO_2e$ over 21 years). The paper suggests that disturbances of this magnitude create a possible feedback loop, whereby this disturbance accentuates climate change by increasing the likelihood of subsequent natural disturbances (Kurz et al. 2008a). This perspective of MPB-attacked forests being a large C source is a predominant notion in the grey and scientific literature. [In retrospect however, the current provincial projections of 55% pine mortality are far less than the Province's initial predictions in 2006 of 80% mortality, which was the provincial MPB projection used by Kurz et al. (2008a) in their C modeling (MFLNRO 2015e).]

In Metsaranta et al. (2011), BC's forested land base (~67 million ha) was predicted to act as a C source for most of the 21st century. This C source reflects a history of natural and anthropogenic disturbances on the land base, as well as expected future disturbances. The duration of BC's forest C source is expected to be highly variable and uncertain given Metsaranta et al.'s (2011) most optimistic and pessimistic scenarios, which tested the effects of future changes in growth rates, decay rates, and area burned by wildfire on the cumulative impact and recovery of provincial forests.

Coinciding with these modeling predictions by Kurz et al. (2008a) and Metsaranta et al. (2011) are field measurements using the eddy covariance technique. Studies using this technique have found sub-boreal spruce stands in BC to act intermittently as a slight C sink or a slight C source following MPB attack (Brown et al. 2010, 2011). These findings seemingly contradict the widespread notion that these stands are an immediate and large source of C emissions in need of immediate attention (Kurz et al. 2008a). Other studies corroborate Brown et al.'s (2010) findings by demonstrating that stand-level C fluxes can be minimal post-disturbance and can return to a C sink status shortly after these types of events – albeit at a lower C sink strength than pre-disturbance (Edburg et al. 2011, Pfeifer et al. 2011, Goetz et al. 2012). Forests are thought to follow many different C trajectories depending on the nature of the natural disturbance regime (e.g. severity, frequency), as well as other variations in climate and environmental conditions (Goetz et al. 2012). The presence of secondary structure and residual vegetation in Brown et al.'s (2010) MPB-attacked stands were found to mitigate the loss in net primary productivity (from dead trees) and increase in heterotrophic respiration (i.e. decomposition, respiration) (Bowler et al. 2012, Emmel et al. 2014). [Kurz et al. (2008a) recognized the beetle outbreak as a non-stand replacing disturbance event but it was parameterized based on the percentage of tree crowns killed rather than the secondary structure and residual vegetation present.]

In unmanaged (or conserved) forests, the long-term recovery back to pre-disturbance C status is hindered as dead trees begin to fall and decay (Edburg et al. 2011, Harmon et al. 2011, 2013). While many of these snags are expected to fall relatively quickly (Angers et al. 2010, Edburg et al. 2011), others are expected to remain standing for some time (Axelson et al. 2010, Amoroso et al. 2013). The duration of this dead (and potentially decaying) wood is thought to be a complex and important temporal aspect of forest C dynamics, and yet, it is too often simplified in C modeling by assigning default decay functions to snags as downed wood (Boudewyn et al. 2007), and by omitting any secondary structure and residual vegetation photosynthetic response.

In managed forests, the C budget basis for deciding to harvest MPB-attacked forests is not apparent. This is a result of the magnitude of this disturbance and its unprecedented impact on BC's timber supply. By harvesting a stand, a large and persistent (8-10 years) C source is created on-site (Fredeen et al. 2007). This C source, in turn, has traditionally been offset by forest regrowth under the presumption of a sustainable rotation at the stand-level and/or sustainable forest management at the landscape-level. Due to the extensive damage caused by the MPB, however, these presumptions are now less evident, and likely debatable. [To date, forest C modeling has not explicitly mentioned nor adequately addressed the issue of whether or not MPB-attacked stands should be harvested above sustainable harvesting rates (i.e. > long-run sustained yield) (Kurz et al. 2008a, Metsaranta et al. 2011, Lamers et al. 2014).]

From a pragmatic standpoint, the MPB-attacked stands that can be harvested profitably are done so regardless of these C considerations. The decision on whether or not to harvest MPB-attacked stands from a C budget basis however, has been largely deferred until more is known about the C budgets of forest products harvested from these forests. One of the general observations in forestry has been that forest products offer temporary C storage. Traditionally, forest C budgets were ecological concepts typically discussed without accounting for the C in forest products, or treated as if the C from these products were emitted immediately (Kurz et al. 2008a). Therefore, better C accounting of the C storage held in forest products has resulted in reduced estimates of forest C emissions by refining their forest C budgets (Metsaranta et al. 2011, MOE 2012).

Proponents of bioenergy claim overall reductions in GHG emissions by replacing fossil fuel derived GHG emissions with renewable, biomass-derived GHG emissions. This claim is typically argued on the basis of contributing to GHG mitigation. Mitigation is defined here as the amount of GHG emissions avoided by either reducing emissions or increasing removals (i.e. C storage) that can be achieved by promoting a mitigation activity relative to a baseline (or business as usual) condition (Lemprière et al. 2013). The concept of GHG mitigation is pervasive in environmental policy, and yet, its validity in environmental science remains unclear. While it is generally accepted that bioenergy's use of wood waste materials reduces GHG emissions, there are also those who advocate bioenergy generation through the harvesting of healthy (non-merchantable) and/or damaged forests.

The addition of the mitigation concept to both forestry and bioenergy industries has led to confusion in the decision-making process of whether or not to harvest MPB-attacked stands from a C budget standpoint. The versatility of the mitigation concept in forest and forest product systems has led to a profusion of perspectives in the literature. Some of these perspectives relevant to MPB-attacked forests include: (1) accelerating their recovery by restocking the forest, (2) storing and using what would otherwise be "dead and decaying" timber in forest products and (3) substituting more fossil-fuel dependent products and processes. These are certainly apparent and convincing arguments; however, there is little support for any of them in environmental science and policy.

Traditionally, studies have been built on an attributional approach that describes a system as it exists (Miner and Gaudreault 2013), and it relies on direct measurements built on a strong environment science foundation. The contemporary concept of mitigation however, uses a consequential approach (Miner and Gaudreault 2013). This approach describes how the system changes in response to decisions and is more commonly associated with environmental policy than environmental science. The advantage of this approach is that it can be a strong and effective tool in promoting climate change mitigation efforts.

However, it requires that the modeling accurately portray business-as-usual and alternative scenarios. This is difficult to do in BC given the uncertainty in C cycling of MPB-attacked forests and lack of established (or verified) parameters for the forest product industry.

In summary, the decision of whether MPB-attacked forests should be harvested (above sustainable levels) is obscured by the diversity of perspectives on how the Province can mitigate the C budget impact of the outbreak. This is problematic given the rudimentary knowledge and history of forest C science and management in the province. The MPB outbreak and salvage logging have had an unprecedented impact on the forest C cycle and there remains a great deal of uncertainty in the long-term modeling of post-disturbance C cycling, making it difficult to establish accurate and credible baselines for MPB-attacked forests and their potential forest product streams.

1.1.3.2 Carbon Neutral Forestry

Forestry results in C emissions from the biological (i.e. forest) and industrial (i.e. forest product) systems (Gower 2003). It is generally assumed that these C emissions are offset by C sequestration in a sustainably managed forest landbase. C neutral forestry (or C-cycle neutrality) is when C emissions are balanced by C sequestration at the landscape-level (Malmsheimer et al. 2011).

One of the issues with the C neutral concept being applied in BC is the ambiguity between environmental science and policy defined forest and forestry C neutrality (Klopp and Fredeen 2014). The province's timber supply is unlikely to be sustainable given the unprecedented impact of the MPB outbreak and excessive salvage logging above the AAC (Burton 2010). Sustainably managed forests are one of the prerequisites for assuming C neutrality. As a result, it is possible that not all biogenic GHG emissions can be considered C neutral and may need to be included in the C accounting of the forest products industry.

1.1.4. Estimation of Forest Product GHG Emissions

Traditionally, C accounting has assumed forest products' GHG emissions to follow a default trajectory equivalent to their immediate combustion on site (UNFCCC 2003, Eggleston et al. 2006). These forest product emissions fell under the Agriculture, Forestry, and Other Land Use (AFOLU) section according to the UNFCCC. In light of recent amendments, forest products can now be estimated and/or reported as part of a nation's submission to the UNFCCC. Nations are still required to report forest products under the default assumption in the AFOLU section but they can now offset these emissions by claiming any C storage held in forest products. Dymond (2012) recently found that BC could benefit from this new amendment in the UNFCCC

There are numerous C accounting approaches that are used to inform environmental science and policy on forest products, each having their own strengths and weaknesses. A C accounting approach is defined here as the "conceptual framework for estimating emissions and removals of greenhouse gases in inventories" (Cowie et al. 2006). In other words, the "approach refers to the system boundary, defining which emissions and removals are to be reported or accounted by each (entity)" (Cowie et al. 2006). The establishment of system boundaries is important due to the complexity of the forest products industrial system, and as a result, various entities have created different C accounting approaches to distinguish the GHG emissions for which they are accountable.

1.1.4.1. National GHG Inventory

The C accounting approach in National GHG Inventories is set up to estimate and report a Party's forest products GHG emissions according to the conventions required in submitting a national inventory to the UNFCCC (Eggleston et al. 2006). The intent of this inventory is to provide an international basis for quantifying and comparing a Party's efforts towards climate change mitigation (UNFCCC 2016). One of its limitations is that it does not quantify all of the GHG emissions attributable to an activity or product, but instead distributes them amongst various sectors (e.g. industrial, waste, forestry, etc.) (MOE 2012). This makes it difficult for a Party to equitably inform decision-making in forestry activities.

1.1.4.2. Carbon (C) Footprints and Life Cycle Assessments (LCA)

Carbon footprint studies all share a similar goal in their desire to quantify GHG emissions attributable to an entity or product (Miner and Gaudreault 2013). The most common and standardized approach for forest products is the life cycle assessment (LCA) (ISO 2013). This approach is an analytical tool used to help industries define the environmental impact of their products (ASMI 2012, Puettmann et al. 2010). The strength (or weakness) in this approach centers on its definition of scope, which delimits the environmental impacts to those solely attributable to the reporting entity. These impacts typically include upstream GHG emissions, but rarely downstream GHG emissions. One of the limitations of this approach for my purposes is its scope definition. The environmental impact assessment in LCAs is only relevant to its defined boundary, rather than the overall environmental impacts these products have on their forest and forest product systems.

1.1.4.2.1. Consequential LCA

Another strength of the LCA is that it enables comparisons between alternative and competing products and processes. This type of LCA approach is known as a consequential LCA (CLCA). A large component of CLCAs is substitution, which "...refers to replacing product A with product B, such as substituting wood for cement or biofuel for fossil fuel." In forest product C accounting, the avoided emissions generated in forming CLCAs (i.e. substitution emission factors) have been shown to represent significant climate benefits (Damen and Faaij 2006, Sathre and O'Connor 2010). Sathre and O'Connor (2010) performed a meta-analysis on substitution emission factors in the forest products industry and found an average substitution emission factor of 2.1 tC removed per tC of wood product. The limitation of using this approach for my purposes is that CLCAs are built on the concept of functional units. [A functional unit is the "quantified performance of a product system for use as a reference unit" (ISO 2006).] That is, by calculating a single substitution factor for a product, only GHG emissions attributable to the product are included, rather than all the GHG emissions attributable to the material or activity (Sathre and O'Connor 2010). This potentially allows a product with inefficient material use to compete with a product with a more efficient material use. In this manner, Sathre and O'Connor (2010) hypothesize that a single substitution factor may not necessarily inform the most efficient use of the biomass material in addressing climate change concerns.

1.1.4.2.2. Environmental LCA

Environmental LCAs (or industrial forest C budgets) are differentiated from traditional LCAs by the fact that they are material-oriented rather than product-oriented (like traditional LCAs). While they are far less common and/or standardized, they are known to offer unique insight into the environmental impacts of the forest product industry system. White et al. (2005) found that their industrial forest C budget could help inform management with different levels of forest governance (e.g. state, national, and private non-industrial). The authors' C budget for forest products (7 gC m^{-2} yr^{-1}) was not seen as making a significant contribution to the overall forest C budget (-904 to 341 gC m^{-2} yr^{-1}). Similarly, Ingerson (2011) tracked the material flow of forest products and found that very little of the original timber remains in use at 100 years (~1%). The limitation of this approach is that there is no precedent in BC for an environmental LCA and it would require extensive amounts of time and capacity to begin building this type of inventory for the forest products industry.

1.1.4.3. Forest Product C Models

1.1.4.3.1. BC-HWPv.1

Dymond (2012) created a spreadsheet model (BC-HWPv.1) for forest products harvested in British Columbia. The intent of this model was to inform the provincial government on the GHG mitigation potential of including forest products and their activities in BC's GHG inventory. Specifically, this model estimated the C storage in forest products by establishing a series of C stocks and flows of products throughout its industrial system. The limitation of the BC-HWPv.1 model for my purposes is that it only measures the biogenic C stored in forest products and not the GHG emissions associated with their industrial systems.

1.1.4.3.2. CBM-FHWP

Smyth et al. (2014) created a forest products model (CBM-FHWP) meant to inform the government of the greatest mitigation potential of its forest products. The design of the model is unique in that it applies a "systems perspective" to Canada's forest sector. This "systems perspective" includes the GHG emissions avoided by adopting a forest product system over more industrial systems (i.e. cement, steel or coal systems). The limitation of using the CBM-FHWP model for my purposes is a lack in transparency in how the authors derived the substitution emission factors used in their "systems perspective". An understanding of the derivation of substitution is paramount in discussing the GHG mitigation benefits of forest products, as it is known to significantly alter the outcomes of scientific research.

1.1.4.3.3. HWP-CASE

Chen et al. (2014) created a model (HWP-CASE) meant to provide a comprehensive assessment of C stocks and emissions for Canada's forest sector. This model's intent is very similar to that proposed by Lemprière et al. (2013) and Smyth et al. (2014) but its C accounting is set up differently. Chen et al. (2014) measured the C storage, GHG emissions, and avoided GHG emissions (when substituting forest products for fossil fuel intensive products). One of the limitations of the HWP-CASE model for my purposes is its scale, as it is currently configured to inform the country's forestry sector rather than stand- and/or regional scaled studies.

1.1.4.4. Forest-Forest Product C Models

1.1.4.4.1. GORCAM

Schlamadinger and Marland (1996) created a forest-forest product C model (GORCAM) meant to inform policy-makers of the merits of adopting forest and bioenergy strategies in reducing CO_2 emissions into the atmosphere. The forest product model component of the GORCAM model is confined to four "displacement factors" (i.e. substitution emission factors) for biofuel (0.60-1 MgC MgC^{-1}), long-lived products (0.5-1 MgC MgC^{-1}), short-lived products (0.25-0.5 MgC MgC^{-1}), and very short-lived products (0.25-0.5 MgC MgC^{-1}). The derivation of these factors is unknown. The limitation of GORCAM model for my purposes is then the applicability of these displacement (i.e. substitution) emission factors in North America. Their factors are derived from observed trends in Europe and its applicability remains unclear due to its differences in various industrial sectors (e.g. forestry, construction, energy).

1.1.4.4.2. FICAT

The Forest Products Association of Canada (FPAC) used the FICAT model in its assessment of the environmental performance of Canadian forest products (FPAC 2011). This model creates C footprints of forest-based manufacturing activities according to UNFCCC (2003) and WRI/WBCSD GHG Protocol (2011) based methods. FPAC (2011) used this model to outline the C footprints of a variety of forest product activities, including 12 different forest products and 16 different pathways. These activities were then compared based on their C footprints and their socio-economic potential. The strength of this model is its ability to inform industry of its own operational GHG emissions in addition to its upstream impacts on the forest system. The limitation of this model for my purposes is its scope definition as it only accounts for the GHG emissions for which it is responsible, rather than the total GHG emissions.

1.1.4.4.3. Perez-Garcia et al. (2005)

Perez-Garcia et al. (2005) created a C model with the purpose of informing others of the ability of forest products to reduce atmospheric GHG emissions through their displacement of functionally equivalent products in housing construction. They modeled the forest system, the forest product system, and avoided emissions displaced by forest product system. Their results demonstrate reduced GHG emissions in constructing houses with forest products and suggest more intensive forest management. This modeling framework and discussion of its results has continued in more recent publications (such as Lippke et al. 2011) and has been expanded to incorporated bioenergy products in their assessments (Lippke et al. 2012). The limitation of this C model for my purposes is its applicability to BC forest products. The intent of the model was specific to engineered forest products (e.g. I-joists) in housing construction and is not necessarily specific in its handling of traditional forest products (e.g. lumber).

1.1.4.4.4. Lamers et al. (2014)

Lamers et al. (2014) created a C model specific to forestry in MPB-attacked forests. The authors used the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) for the forest component and their own calculations for the forest product component of their model. In their model they calculated the forest and forest products systems using a consequential approach. In doing so, they found overall reductions in forest C emissions when harvesting MPB-attacked stands for lumber and for pellets. The limitation of their model for my purposes is a lack of transparency and validity in their C modeling of the forest industrial system.

1.1.5. Conclusion

In summary, my literature review found that there is a precedent for C accounting and modeling of forest products harvested from MPB-attacked forests in British Columbia. Lamers et al. (2014) demonstrated reductions in overall C emissions by harvesting MPB forests for lumber and for bioenergy (i.e. pellets). In their study, they found lumber as the preferable forest product over pellets, but also found that harvesting heavily damaged stands (> 85% MPB attack) for pellets was acceptable, assuming lumber's ineligibility. This implies that the explicit harvesting of MPB-attacked stands for bioenergy would be preferable over inaction (from a C budget perspective) during BC's salvage logging campaign. With that said, there are some concerns I have in their modeling of the forest product system.

In their assessment, Lamers et al. (2014) chose to include substitution emission factors for their forest products. The benefits of avoided emissions in substitution are well known but poorly established. The authors borrowed a substitution emission factor of 1.7 tC tC^{-1} for lumber products from Sathre and O'Connor (2010). Sathre and O'Connor (2010) derived this factor based on a meta-analysis of reported displacement factors in the literature, which came from a broad array of C budgets performed in different countries, under different scopes, and for different wood products.

Generally, a substitution emission factor is the difference between the fossil fuel emissions (i.e. non-biogenic GHG emissions) attributable to a forest product with those emissions attributable to a functionally equivalent product (such as steel or cement). These factors are product specific, meaning they are distinct from all other products in a forest product system. Lamers et al.'s (2014) forest product modeling component aggregates the substitution emission factors for lumber and for pellets (derived from sawdust and shavings) in their modeling of timber harvested for lumber production. They did not include a separate factor for pulp or paper, but instead assumed that these mill residues are short-lived and accounted for them as biogenic GHG emissions. This is somewhat problematic given the C accounting of biogenic GHG emissions in their model.

Lamers et al.'s (2014) C accounting has biogenic GHG emissions occurring as a default function of forest's GHG emissions. This obscures the biogenic GHG emissions specific to forest products and clouds them with those biogenic GHG emissions specific to the forest. While it was the intent of the authors to determine whether or not MPB-attacked forests should be harvested for forest products, it remains unclear what exact contribution the forest products made to the overall C budget of forestry. Moreover, their modeling of forest products is fairly abstract, rather than established and supported by data.

In this thesis, I create a C budget model for two forest products, lumber and pellets, harvested from MPB-attacked forests in the Prince George region. The purpose of my model and the selection of these forest products was to establish a context for each of these products in BC by tracking their material and product flows, biogenic C storage and the GHG emissions of their industrial systems. My primary objective was to estimate and to compare forest products based on their forest product C budgets. Dymond (2012) set precedent for the tracking of biogenic C stored in BC forest products, but otherwise there has not been a study detailing the forest products harvested from MPB-attacked forests. My secondary objective was to explore the influence of biogenic GHG emissions on forest product C budgets. This was chosen to specifically address the C accounting issues and concerns in assuming C-neutrality of biogenic GHG emissions.

1.2. RESEARCH QUESTIONS

1.) How does the choice in forest product (e.g. lumber vs. pellets) affect the C budget of the forest product industrial system when processing roundwood harvested from forests attacked by mountain pine beetle?

2.) How does the assumption of C neutrality influence the overall C budget of the forest product industrial system?

Chapter 2.

DEVELOPMENT OF A C BUDGET MODEL FOR FOREST PRODUCTS HARVESTED FROM MPB-ATTACKED FORESTS IN THE PRINCE GEORGE REGION

2.0. INTRODUCTION

In the post-epidemic phase of the mountain pine beetle (MPB) outbreak, there currently lacks an equitable carbon (C) budget model capable of considering different forest products harvested from provincial forests. The precedent set by contemporary C modelers has revealed the potential for forest products to reduce atmospheric GHG emissions and aid in climate change mitigation. These benefits, however, are highly context specific, and oftentimes obscure. This chapter outlines the development of a C budget model with the intent of establishing current knowledge of BC's forest product industrial system.

2.1. METHODS & METHODOLOGY

2.1.1. Carbon Budget Model for Forest Products

I developed a C budget model for forest products harvested from MPB-attacked forests. This model estimated: (a) the material and product flows; (b) biogenic C storage; and (c) greenhouse gas (GHG) emissions of the forest product industrial system. The material and product flows were based on prevalent trends in the BC Interior (MFLNRO 2015c). Biogenic C storage was modeled using various flows and stocks of the forest products in use and in disposal, according to Dymond (2012). This is important because each forest product has a different product recovery, use or uses, half life or half lives, and disposal methods. Greenhouse gas emissions were separated based on whether or not they were derived from forest biomass (i.e. biogenic). This is due to the fact that biogenic GHG emissions are oftentimes thought of as C-neutral emissions (i.e. zero emissions), and therefore are not always accounted for in C budgets. However, mounting evidence suggests that forest product C budgets should, at the very least, be estimated in the absence of forest C budget accounting (Searchinger et al. 2009, Helin et al. 2013).

My C accounting approach quantified the direct GHG emissions at each of the life stages of forest products. Each forest product system (or life cycle) was broken down into its individual life stage components, which were: harvest, transportation, primary processing, secondary processing, distribution, use and disposal. My approach characterized the system as it exists, rather than how it may have affected other industrial systems. This meant that we did not do a consequential study and compare my forest product industrial systems with other industrial systems (i.e. considering substitution effects). The purpose of my approach was to build a strong foundation of environmental science before addressing the persistent and complex issues of additionality and leakage in consequential (or political) studies.

The C accounting method in my C model aggregated individual data sets at each of a forest product's life stages. This was chosen over more extensive sampling of the entire industrial system due to the limited time and capacity for meaningful and complete stakeholder involvement in the development of a complete GHG inventory.

2.1.2. Experimental Design

The intent of my experimental design was to determine the C budgets of forest products harvested from MPB-attacked forests. While seemingly apparent, the experimental design required rigorous examination of experimental units, treatments, and the various contexts of forest product C budgets.

2.1.2.1. Experimental Unit & Treatment

My experimental unit was based on the oven-dry (OD) mass of 1 m^3 of dead merchantable lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.) roundwood. This material unit (m^3) was selected for its ability to equitably inform environmental science and policy on the most efficient use of this biomass material in reducing forest product C emissions. This unit did not account for timber utilization (i.e. harvest residues) or the composition of harvest (i.e. % green versus dead trees) prior to harvest. These issues are no doubt important and will need to be addressed in subsequent studies that incorporate a forest C modeling component.

The selection of pine's OD weight, as opposed to green or air-dry weight, was chosen to avoid complex wood-water interactions (i.e. shrinkage) and facilitate the C accounting between the inter-industry trading of mill residues (Shmulsky and Jones 2011). I assumed that there was not a significant change in OD weight of MPB-attacked roundwood from time since death.

My experiment was set up as a comparison between two very different forest products: lumber and pellets. These products were chosen to act as current and proposed uses of roundwood during BC's salvage logging campaign. Both products are thought to be beneficial in the Province's plans for reducing GHG emissions, but are thought to contribute to this initiative in two very different ways. Lumber is a long-lived forest product and temporarily stores C held in its wood fibre, whereas pellets are a short-lived forest product with low GHG emissions (i.e. kg $CO_2e\ kJ^{-1}$) compared to fossil fuels (assuming pellets are C neutral).

In my first scenario (scenario 1), roundwood was harvested for lumber production. The function of this scenario was to establish a baseline for the predominant use of roundwood in the BC Interior. This scenario was selected due to a general lack of studies on the C balance of lumber in the province, as well as concerns over the beetle's impact on lumber's recovery and how this might impact its C budget.

In my second scenario (scenario 2), roundwood was harvested for pellets. This proposed wood product was chosen as an attractive alternative to lumber due to its ability to use MPB-attacked timber as its wood quality declines. This scenario reflects growing scientific curiosity (Lamers et al. 2014, Lloyd et al. 2014), claims by government and industry (EMPR 2007, 2008; BC Hydro 2012), concerns of non-governmental organizations (Nikiforuk 2011, Greenpeace 2011, RSPB 2012), and in some cases, actual practice.

2.1.2.2. Case Studies

Case studies were used in my project to describe the C budget in two different contexts. These contexts represent two extremes in the treatment of biogenic GHG emissions. The intent of these case studies was to explore and demonstrate the sensitivity of forest product C budgets in the absence of the forest C modeling. This evaluation was prompted by the extensive damage to BC forests in the wake of the MPB outbreak and subsequent salvaging efforts.

My first case study (case 1) accepted the C-neutral forestry assumption, which meant excluding all biogenic GHG emissions from the C budget. This is a prominent assumption in forest product C accounting and modeling. It is predicated on the assumption that "(the) uptake of C (in CO_2) by plants over a given area and time is equal to emissions of biogenic C attributable to that area" (Malmsheimer et al. 2011).

My second case study (case 2) rejected the C-neutral forestry assumption, which meant including all biogenic GHG emissions in the C budget. I rejected this assumption based on the unlikely event that forest management is a completely C-neutral activity and that a failure to acknowledge these biogenic GHG emissions may lead to erroneous conclusions on the most appropriate use of this wood material. Lamers et al. (2014) found that harvesting stands exclusively for bioenergy (i.e. pellets) did not meet a C-neutral definition unless the harvested stands contained a high proportion of dead pine (>85%). Other researchers testing the validity of C-neutral claims in healthy forests have found mixed results

(McKechnie et al. 2011, Holtsmark 2012, Mitchell et al. 2012, Schulze et al. 2012).

Cases 1 and 2 in my study, therefore, represent the extremes of 100% and 0% C neutrality, respectively. However, I acknowledge that the true effect of forestry on the C balance of the MPB-attacked forests likely lies somewhere in between these two extremes. It is difficult to know exactly where BC forests are on this spectrum of C neutrality (0-100%) due to a lack of studies examining the landscape-level impacts on the forest C balance.

2.1.2.3. Study Boundaries

The establishment of study boundaries is necessary in the construction of a C budget model for forest products. The forest product system can be a fairly complex and extensive system, so the establishment of boundaries can be a fairly subjective procedure. To be as transparent and honest about my study boundary as possible, I have defined the physical, spatial, and temporal aspects of my C budget model. [Once this boundary is set, the GHG emissions reported in the C budget then reflect the boundaries imposed on the industrial system.]

2.1.2.3.1. Physical

The physical (or procedural) boundary determines which activities and processes of the forest product's industrial system are included in its C budget (Sathre and O'Connor 2010, Miner and Gaudreault 2013).

My forest product C budget includes the material and product flows of forest products (Figure 2.1.). This unusual in that primary processing residues (i.e. by-products) and their GHG emissions are accounted for in my model. Typically, modeling efforts have either focused entirely on the main product or assumed that the residues are immediately combusted. My model does not, however, include all of the upstream and downstream uses of the various non-forest product materials and products used in this system (e.g. bleach from chemical pulp plants).

My forest product C model includes the biogenic C storage of forest products in use and in disposal. It is well known that long-lived forest products can temporarily store C while in use, but they can also store C while in disposal. This storage changes over time – as these products are eventually disposed of in landfills. As a result, my model was designed to look at biogenic C storage over time.

My forest C budget accounts for all of the direct GHG emissions attributable to a forest product's industrial system. These emissions are categorized here as being either biogenic or non-biogenic GHG emissions. Biogenic GHG emissions are biomass-derived GHG emissions that are not typically included in forest product C budgets under a C-neutral assumption. Cases have been made, however, that advocate for including biogenic GHG emissions in forest product C budgets when the forest system is not considered (Searchinger et al. 2009, Helin et al. 2013). Non-biogenic GHG emissions are the remaining GHG emissions that are not biomass derived and/or eligible for C-neutral status. [Note: Methane (CH_4) and nitrous oxide (N_2O) derived from biomass are not considered C neutral.]

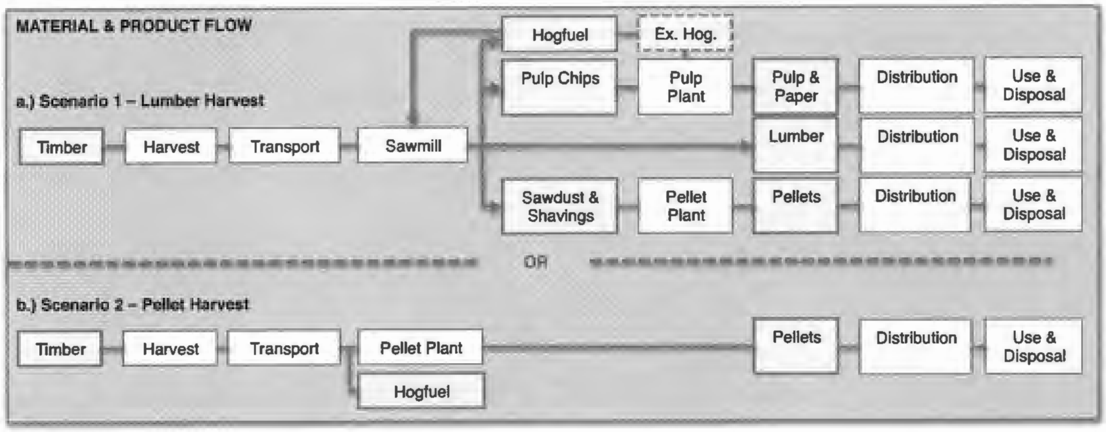

Figure 2.1. The material and product flow of MPB-attacked lodgepole pine (*Pinus contorta* var. *latifolia*) roundwood harvested for lumber (a; scenario 1) or for pellet (b; scenario 2) production (adapted from Klopp and Fredeen 2014).

2.1.2.3.2. Spatial

The spatial boundary defines the physical (or geographical) area of a forest product C budget (Sathre and O'Connor 2010, Miner and Gaudreault 2013). The forest product industrial system includes regional and industrial processes (such as harvesting, transportation, primary and secondary processing), but it can also include international distribution, secondary processing, use, and disposal. In my study, I chose to include both regional and international aspects of the forest product industrial system. It was assumed that some of the largest GHG emissions occur abroad. Moreover, the establishment of spatial (or political) boundaries has been known to skew the results of forest product C budgets (Nabuurs and Sikkema 2001).

Another aspect of establishing a spatial boundary is that it determines how biological processes are perceived and/or treated (Miner and Gaudreault 2013). In particular, the spatial boundary representing forestry and photosynthesis (as biological processes) at the stand- and landscape-level are important in discussing biogenic GHG emissions attributable to the forest product industrial system. In my study, I chose to discuss the spatial boundary of my C budget in a landscape-level setting. This was chosen based on landscape-level concerns, (such as overharvesting during the MPB outbreak) and the potential shortsightedness of a stand-level spatial boundary.

2.1.2.3.3. Temporal

The establishment of a temporal boundary is problematic in forest product C budgets, as it is known to significantly affect the outcomes of their budgets (Miner and Perez-Garcia 2007). In short, various temporal boundaries exist and have been used in a variety of applications. Short-term studies are promoted on the importance of short-term mitigation efforts and their ability to limit the uncertainty of C modeling. These studies include a focus on quick and tangible improvements during the rapid transitions of a forest product's early life stages. In contrast, long-term studies demonstrate the actual net impacts of forest products over the course of their lifetime. The accuracy of their portrayal is reliant on: current patterns of wood use and methods of disposal (Smith et al. 2006); the mathematical formulation of use and disposal functions (Marland et al. 2011); and the disposal technology available.

In my study, the temporal boundary was set at 100 years. With that said, the establishment of a timeline is fairly arbitrary. Generally, timelines are synchronized with the forest management rotation or set to an arbitrary, round-numbered standard (for example, 100 years). In my case, a 100-year timeframe is the most pragmatic as it aligns with current conventions in the Province's GHG inventory.

2.1.3. Data Collection

My forest product C budget model relied on a variety of resources, which help construct the material and product flow, biogenic C storage, and the GHG emission profile of my forest product industrial systems. A detailed overview of the current and most representative data available in BC is presented in the following sections.

2.1.3.1. Lumber (Scenario 1)

2.1.3.1.1. Material & Product Flow

2.1.3.1.1.1. Lumber Recovery

An exact estimate of the lumber recovery of MPB-attacked roundwood is uncertain for a variety of reasons. Lumber recovery is naturally variable between sawmills due to differences in the log size and quality (e.g. defects, grade) and a sawmill's machinery (Shmulsky and Jones 2011); but there is also a general lack of information on the lumber recovery of MPB-attacked roundwood.

At the peak of the MPB outbreak, the shelf life and lumber recovery of MPB-attacked timber was a large concern held by many researchers (Lewis and

Hartley 2006, Orbay and Goudie 2006, Byrne et al. 2006). At this time it was found that grey-stage (>five years since attack) roundwood had a lumber recovery factor of 0.241 mbfm m^{-3} (mbfm: thousand board feet) with current technology (Orbay and Goudie 2006) (Table 2.1.). Forestry Innovation Investment Ltd. (2016) has since continued the examination of lumber recovery of MPB-attacked roundwood but it is now considered market research and is not publically available.

British Columbia's Ministry of Forests, Lands and Natural Resource Operations (MFLNRO) Competitiveness and Innovation Branch reported an average lumber yield of 44.5% (on a volume basis) for Interior sawmills in 2012 (MFLNRO 2015c). This yield is calculated by dividing total lumber output by total roundwood input in the Interior. This value represents a fair estimate of the average lumber recovery in the Interior. One of its limitations is that it averages multiple tree species and sizes, and thus limits its usefulness in determining MPB lumber recovery. For example, the allowable annual cut (AAC) uplifts increased pine utilization in heavily impacted TSAs, and yet, the average lumber recovery in the BC Interior remains unchanged over this period of time (MFLNRO 2015c)

British Columbia's Ministry of Forests, Lands and Natural Resource Operations Timber Pricing Branch estimates specific lumber recovery yields by species, condition and size (MFLNRO 2015a). Their recoveries are calculated using a set of "log rules," which the Branch uses to estimate the expected recovery of lumber (Shmulsky and Jones 2011). Lumber recovery for healthy lodgepole pine ranges from 0.091 to 0.254 mbfm m^{-3} based on their log top diameter class (4.5-5.49 cm and +100 cm) (MFLNRO 2015a). [Due to the bucking of timber at the landing, some logs will have a large top diameter.] Lumber recovery from dead lodgepole pine can also be estimated with net adjustments (MFLNRO 2015a). However, these yields are suspect in being more conservative (i.e. lower) than their actual recoveries when compared to those described by the Competitiveness and Innovation Branch.

Life cycle assessments (LCA) performed in Canada and in the US Inland Northwest had lumber yields of 43 and 56% (on an oven-dry weight basis), respectively (Puettmann et al. 2010, ASMI 2012). This recovery value applies conversion factors (ODt m^{-3}) to relate the material and products into common units (i.e. ODt). The limitation of their lumber yields is a lack of geographical and temporal representativeness of the BC forest products industry, let alone any technological refinements during the MPB outbreak.

Carbon models typically assume lumber yields between 45-50% (on an oven-dry weight basis) (Schlamadinger et al. 1996, Lippke et al. 2012, Chen et al. 2014, Smyth et al. 2014). The origins of these estimates are not always apparent and many simply rely on that suggested by the IPCC and UNFCCC (UNFCCC 2003, Eggleston et al. 2006). Lamers et al.'s (2014) model of forest products used a lumber yield of 50% for MPB-attacked roundwood.

In my study, the lumber recovery value was taken from Orbay and Goudie (2006) (Table 2.1.). This was taken to be a fair estimate given the low estimates by MFLNRO's Timber Pricing Branch and the higher recoveries reported in the MFLNRO's Competitiveness and Innovation Branch, LCAs, and C modeling.

Table 2.1. Average lumber recovery values in sawmills.

Lumber Recovery			Reference
mbfm m^{-3}	% m^3	% ODt	
NA	NA	50	Milota et al. 2005
NA	NA	42	Milota et al. 2005
0.24*	(38)*	NA	Orbay and Goudie 2006
NA	NA	50	Chen et al. 2008
NA	NA	52	Puettmann et al. 2010
NA	NA	43	ASMI 2012
NA	NA	50*	Lamers et al. 2014
0.09 – 0.25	(14 – 40)	NA	MFLNRO 2015a
0.28	45	NA	MFLNRO 2015c

mbfm m^{-3}: Thousand board feet (mbfm) per cubic meter (m^3) of roundwood.
% m^3: Percent of m^3 lumber recovered from m^3 roundwood.
% ODt: Percent of oven-dry tonne (ODt) lumber recovered from ODt roundwood.
( ): Applied Nielson et al.'s (1985) conversion factor of 1.594 m^3 $mbfm^{-1}$.
*: Specific lumber recovery yields for MPB-attacked roundwood.
NA: Not available

2.1.3.1.1.2. Mill Residue Recovery

Mill residues originate from the various processing phases of lumber sawmills (i.e. wood handling, sawing, planing) (Shmulsky and Jones 2011). They typically include: pulp chips, sawdust, shavings, and hogfuel (i.e. bark and other mixed wood waste) (MFLNRO 2015c). While mill residues often form the majority of the original roundwood harvested, there is little governance in their use. Their volume, composition, and use depend on the sawmill and its prerogatives, as well as the surrounding forest product industries (e.g. pulp and paper plants, pellet plants) (Nielson et al. 1985). The wood fibre arrangements in this manuscript follow those described in MLFNRO (2015) (Figure 2.1.).

Pulp chips are created using a chipping headrig in lumber sawmills (Shmulsky and Jones 2011). Pulp chips produced in sawmills are valued in the BC Interior as an important feedstock for the pulp and paper industry (Canfor Pulp Inc. 2012ab, MLFNRO 2015). In the Prince George region, there exist two chemical pulp plants (Intercontinental Pulp Mill, Northwood Pulp Mill) and one chemical pulp and paper mill (Prince George Pulp and Paper Mill) (MFLNRO 2015c). It was assumed that the chemical pulp produced from the region was exported (based on pulp production capacities) to a paper mill outside of British Columbia (MFLNRO 2015c).

Sawdust and shavings are created as a result of the sawing and planing processes in lumber sawmills (Shmulsky and Jones 2011). Both mill residues are valued in the province as important feedstocks for the pellet industry (MLFNRO 2015c). In the Prince George area, there are two pellet plants: Pacific Bioenergy and Pinnacle Pellet. [Alternative markets for these residues in the province include: medium-density fibreboard and particleboard. These alternative wood product industries are not, however, currently present in the Northern Interior (MFLNRO 2015c).]

Hogfuel is created as a mixture of mill residues collected throughout all the processes in lumber sawmills (Shmulsky and Jones 2011). This mill residue can include a mix of sawdust (green and/or dry), shavings (green and/or dry), bark, and trim ends. The end result creates a residue with variable moisture content. As such, it is valued as a source of cheap and C-neutral fuel for bioenergy

(Nyboer 2015ab). The largest quantities of hogfuel are typically consumed in BC's pulp and paper industry (MFLNRO 2015c).

In my study, I used the mill residue composition and recoveries cited in ASMI (2012) and the wood fibre arrangements in MFLNRO (2015c) to delineate the material flow of mill residues from the lumber sawmill to their respective processing industries (Table 2.2.). To date, there is little information specifically on mill residue composition and recoveries while processing MPB-attacked roundwood. My chosen lumber recovery factor (from Orbay and Goudie 2006) did not give a breakdown of the mill residues for MPB-attacked roundwood. Therefore, I adapted ASMI's (2012) residue breakdown by distributing total residue volume evenly across the mill residues.

Table 2.2. Lumber and mill residue recovery values.

Lumber	Pulp Chips	Sawdust & Shavings	Hogfuel	Shrinkage	Reference
		% of roundwood			
50	29	12	10	NA	Milota et al. 2005
42	32	9	17	NA	Milota et al. 2005
51	25	NA	25¶	NA	Chen et al. 2008
52	26	11	11	NA	Puettmann et al. 2010
43	35	12	11	NA	ASMI 2012
NA	58	25	17	NA	Canfor Corp. 2013
50	NA	15	35¶¶	NA	Lamers et al. 2014
45	41¶¶¶	12	NA	2	MFLNRO 2015c

¶ "Processing residues"
¶¶ "Short-lived wood products"
¶¶¶ "By-product chips"
NA: not available

2.1.3.1.2. Biogenic C Storage

2.1.3.1.2.1. Lumber

Lumber has biogenic C storage in its use and disposal life stages. Lumber has many different uses in North America (Table 2.3.). These uses of lumber in turn determine the half-life ($t^{1/2}$) of the product, which dictates how much lumber (i.e. biogenic C) remains in use (Smith et al. 2006). Conversely, half-life also dictates how much lumber is sent to disposal. Lumber stores biogenic C in landfills via its degradable and non-degradable components (Table 2.4.). The degradable component succumbs to decay over time, whereas the non-degradable is thought of as permanent.

In my study, lumber's use and disposal was modeled after Dymond's (2012) BC-HWPv.1 model (Table 2.3., 2.4.). This model was created to monitor the biogenic C storage of BC forest products in North America. Alternatives to this C model include Chen et al. (2014), which examined the biogenic C storage of Canadian forest products (in general) in North America.

Table 2.3. Parameters governing biogenic C storage of lumber in use.

Use of Lumber	Dymond 2012 % ($t^{1/2}$)	Chen et al. 2014 % ($t^{1/2}$)
Single family homes	25 (90)	24 (85)
Multi-family homes	2 (75)	3 (50)
Commercial buildings	7 (30)	NA
Residential upkeep and moveable homes	25 (38)	27 (35)
Furniture & other manufacturing products	10 (38)	NA
Shipping	10 (2)	NA
Other	12 (38)	46 (20)
Landfill	8 (38)	NA
Recycle	2 (38)	NA

$t^{1/2}$: half-life

Table 2.4. Parameters governing biogenic C storage of lumber in disposal.

Disposal of Lumber	Dymond 2012, Chen et al. 2014
Disposal site	14% burned, 9% recycled, 8% composted, 67% landfilled, and 2% dumped¶
Degradable portion	23%
Half-life	29 years

¶Solid-waste disposal sites, landfill and open dumps, are differentiated from one another based on the presence of oxygen. Open dumps refers to sites "where oxygen is available to decompose all wood and paper over time, and landfills, where a covering is placed over waste periodically and oxygen is sealed out" (Skog 2008).

2.1.3.1.2.2. Paper

Paper has biogenic C storage in its use and disposal life stages. In contrast to lumber, paper is not generally given a specific use, but is instead given a generic half-life of 2.5 years (Table 2.5.). The brevity of paper's half-life results in very little biogenic C storage while in use. The bulk of paper's biogenic C storage is found as non-degradable portions in disposal (Table 2.6.).

In my study, paper's use and disposal was modeled after Dymond's (2012) BC-HWPv.1 model (Table 2.5., 2.6.).

Table 2.5. Parameters governing biogenic C storage of paper in use.

Use of Paper	Dymond 2012, Chen et al. 2014 % ($t^{1/2}$)
Paper	100 (2.5)

Table 2.6. Parameters governing biogenic C storage of paper in disposal.

Disposal of Paper	Dymond 2012	Chen et al. 2014
Disposal Site	14% burned, 50% recycled, 5% composted, 31% landfilled, and 1% dumped¶	16% burned, 66% recycled, 5% composted, 12% landfilled, and 1% dumped¶
Degradable portion	56%	56%
Half life	14.5 years	14.5 years

¶Solid-waste disposal sites, landfill and open dumps, are differentiated from one another based on the presence of oxygen. Open dumps refers to sites "where oxygen is available to decompose all wood and paper over time", and landfills, where a covering is placed over waste periodically and oxygen is sealed out" (Skog 2008).

2.1.3.1.3. GHG Profile of Lumber (Scenario 1)

2.1.3.1.3.1. Harvesting

British Columbia's Interior forests are predominantly managed under a clearcut-with-reserves silvicultural system (MFLNRO 2015d). The harvesting event creates a large and persistent C source on site (Fredeen et al. 2007). These emissions are not, however, attributed to the forest products industrial system, but rather, to the forest system. Instead, the harvesting emissions discussed here refer to the emissions surrounding the industrial harvesting system (e.g. harvesting, skidding, delimbing) – the most common system in the Interior being ground-based (MFLNRO 2015d).

Data collection on the harvesting system is primarily derived from energy consumption surveys by industry and academia (Karjailainen et al. 1996, Sambo 2002, Oneil et al. 2010). These surveys often collect energy consumption for: preharvest activities (e.g. right-of-way logging, road construction), camp, harvest, and silviculture (e.g. site preparation, replanting) unit processes. The primary method of calculating their harvesting efficiency is by dividing total energy consumption of the entire forest operation by the total merchantable volume harvested (Oneil et al. 2010). As a result, much of the information regarding harvesting-specific parameters is lost.

For my study, I used the energy consumption values reported in Sambo (2002). This study examined forest operations (i.e. harvesting, hauling) of roundwood in Western Canada, including a few sites in the Prince George region. The study's forest conditions had an average cutblock area of 24 ha, merchantable timber volume of 244 m^3 ha^{-1}, and a volume of 5,856 m^3 (Sambo 2002). This study was found to be the most comprehensive and representative set of data of current forest operations in the region. Other sources of data reviewed include: harvest data from industry (NRCan 2010, ASMI 2012), research (MacDonald 2006), and academia (Berg and Karkalainen 2003) (Table 2.7.). But overall, there has been very little (public) research conducted on the current GHG emission efficiencies of salvage logging operations in the BC Interior. Instead, much of the research during these campaigns has focused on the economic feasibility of such operations (MacDonald 2006).

Table 2.7. GHG emission factors in harvesting forests.

Non-Biogenic		Reference
kg CO_2e $m^{-3}_{roundwood}$	kg CO_2e m^{-3}_{lumber}	
10	NA	Sambo 2002
5	NA	Berg and Karkalainen 2003
6	NA	Berg and Karkalainen 2003
9	22	NRCan 2010
NA	10	ASMI 2012

NA: not available

2.1.3.1.3.2. Transportation

The transportation system tracks roundwood loaded at a harvest landing site and hauled to its designated mill (BCIT 1996, MacDonald 1999). It is often suggested that transportation is a limiting factor in MPB salvage logging operations due to its high cost. The damage caused by the beetle has decreased lumber value recovery ($ m^{-3}) of roundwood, making the supply chain less efficient. Known parameters impacting the transportation system's cost efficiency include: the diversity in roadways, speed, weight, distance, cycle-time, and fuel consumption (MacDonald 2006, MFLRNO 2008). These are discussed in detail in MacDonald (2006) and are the focus of ongoing research by the Forest Engineering Research Institute of Canada (FERIC) (Jokai 2006ab, MacDonald 2006). In contrast, little research has been conducted explicitly on the energy consumption and/or efficiency of salvage logging operations in the BC Interior (Sambo 2002, Lindroos et al. 2011).

Data collection on the transportation system's energy consumption is typically collected via surveys (Sambo 2002, Oneil et al. 2010, ASMI 2012). Hauling distances are cited on a one-way basis, although most surveys include backhaul in their overall energy consumption reporting (Sambo 2002, Puettmann et al. 2010, ASMI 2012). Similar to harvesting, the transportation system's energy efficiency is calculated by dividing the total energy consumption by the total merchantable timber volume (Oneil et al. 2010). As a result, much of the information regarding transportation-specific parameters is lost.

In my study, the transportation of roundwood is derived from Sambo (2002) (Table 2.8.). The energy consumption value here reflects an average value for the entire transportation system, meaning it covers a round trip and its various road conditions (e.g. highway, off-road). The average one-way haul distance for my study was 106 km (Sambo 2002). This distance is thought to be fairly representative of the region and falls within the distances discussed elsewhere in the literature, such as: 50-200 km (MacDonald 2006), 102 km (ASMI 2012) and <300 km (Lamers et al. 2014).

Table 2.8. GHG emission factors in transporting roundwood.

Non-Biogenic		Reference
kg CO_2e $m^{-3}_{roundwood}$	kg CO_2e m^{-3}_{lumber}	
10 (106 km)	NA	Sambo 2002
6 (NA)	NA	Berg and Karkalainen 2003
3 (NA)	NA	Berg and Karkalainen 2003
10 (88 km)	NA	NRCan 2010
NA	7 (NA)	ASMI 2012

NA: not available

2.1.3.1.3.3. Primary Processing Industry: Lumber Sawmill

The primary roundwood processing industry operating in the Interior is the large dimensional lumber sawmill (MFLNRO 2015c). Each sawmill is specialized with specific headrigs (i.e. saws, chippers) and kilns to match selected tree species, lengths, and moisture contents of incoming roundwood from the surrounding timber supply, while at the same time, meeting international market demands (Bogdanski et al. 2011). Dimensional lumber is valued as a low input, low cost structural product used primarily as timber framing in new residential housing (Shmulsky and Jones 2011). The MPB's impact on lumber quality threatens the industry's share in these markets because of its decreased product value recovery ($ m^{-3}; Bogdanski et al. 2011).

Data collection on sawmill energy consumption is collected via survey. These surveys can include intensive life cycle assessments (LCA) of forest products (ASMI 2012, Puettmann et al. 2013ab) or extensive industrial surveys (Nyboer 2015b). The trade-off between these surveys is a more detailed material flow and energy consumption in LCA, over better representation (e.g. geographical, temporal) of the industry in broader industrial surveys. In both cases, energy consumption is calculated by dividing annual energy use over annual product output, on a product volume basis (Nyboer 2015b).

In my study, I used the primary processing data from an LCA survey of Canadian softwood lumber (ASMI 2012) (Table 2.9.). This inventory presented the most comprehensive view of the typical Canadian sawmill. Granted, further refinement is still needed for processing MPB-attacked roundwood explicitly. [However, the feasibility of doing so remains unlikely.] Anecdotal evidence of the beetle's impact on the energy efficiency of sawmills has suggested that wood handling and drying would increase the overall energy consumption (Lewis and

Hartley 2006). However, this is difficult to quantify given the mix of tree species and overall improvements in energy efficiency experienced by most sawmills in recent years (Nyboer 2015b).

Table 2.9. GHG emission factors of a large sawmill.

Non-Biogenic		Biogenic	Reference
kg CO_2e m^{-3}	kg CO_2e tC^{-1}	kg CO_2e m^{-3}	
44	NA	97	NRCan 2010
32	NA	0*	ASMI 2012
113	NA	102	Puettmann and Oneil 2013
103	NA	135	Puettmann et al. 2013a
49	NA	198	Puettmann et al. 2013b
NA	297	0*	Chen et al. 2014
170	NA	0*	Nyboer 2015b

m^{-3}: cubic meter of lumber
* biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).
NA: not available.

2.1.3.1.3.4. Secondary Processing Industries

2.1.3.1.3.4.1. Chemical Pulp Plant

Chemical pulp plants are the predominant recipients of pulp chips in the British Columbia (Nyboer 2015a). This industry is characterized by its distinctive use of chemicals in the pulping process of wood fibre (i.e. pulp chips) (Shmulsky and Jones 2011). The chemical pulping process removes lignin from the wood fibre, decreasing overall pulp yield to approximately 45-47% (Shmulsky and Jones 2011, Canfor Pulp 2012ab). The main pulp product from this process is bleached Kraft market pulp (BKMP). This pulp is typically exported and processed into paper and paper products via paper mills in other countries (IC 2015).

Chemical pulp plants are very energy intensive (Browne and Williamson 1999, Francis et al. 2004, NRCan 2008, US DOE 2005). In British Columbia, much of the energy consumed is generated in-house using spent cooking chemicals (i.e. black liquor) and hogfuel sourced from neighboring sawmills (Canfor Corp. 2013, Nyboer 2015a). A pulp plant's energy mix (i.e. % coal, natural gas, biomass) is plant-specific, and thus, the GHG emissions from this industry can vary quite substantially among plants. Energy audits typically focus on energy consumption specific to a pulp plant's unit processes (e.g. chip conveying, digester, washing and screening, etc.) rather than its energy mix (NRCan 2008). As a result, the GHG emissions from this industry are often obscure.

In my study, energy consumption data of a chemical pulp plant were taken from an environmental product declaration of a chemical pulp plant located in Prince George (Canfor Pulp Inc. 2012a) (Table 2.10.). This was found to be the most geographically representative documentation of the pulp industry in the region. The plant's energy efficiency in 2012 was 38.3 GJ ADt^{-1}, which was composed of: 1.3% hydroelectric, 90.2% biomass, and 8.5% fossil fuels. This agrees well with similar energy consumption values of 36-37 GJ ADt^{-1} reported elsewhere in the literature (NRCan 2008, Nyboer 2015a).

Table 2.10. GHG emission factors of a chemical pulp plant.

Non-Biogenic		Biogenic	Reference
kg CO_2e ADt^{-1}	kg CO_2e tC^{-1}	kg CO_2e ADt^{-1}	
171	NA	2951	Canfor Pulp Inc. 2012a
240	NA	3110	Canfor Pulp Inc. 2012b
NA	578¶	0*	Chen et al. 2014
NA	1098¶	0*	Chen et al. 2014
325	NA	0*	Nyboer 2015a

ADt: air-dry tonne (ADt) of pulp.
0* biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).
¶ "cradle to gate" refers to allocating upstream GHG emissions into an industry's GHG total.
NA: not available

2.1.3.1.3.4.1.1. Paper Plant

Paper plants receive market pulp from chemical pulp plants and process it into paper. They do this by preparing, blending, pressing, and drying pulp into a desired paper product (US DOE 2005). The two largest paper and writing products from market pulp are: coated and uncoated freesheet paper (AFPA 2012). Coated paper can be used in glossy magazines and catalogs, while uncoated paper is commonly used as freesheet paper. The two largest importers and producers of these paper products are United States and China (IC 2015).

Paper plants are energy intensive (US DOE 2005, NRCan 2008). Their energy consumption can range anywhere between 11.1 to16.1 GJ ADt^{-1} (US DOE 2005, Nyboer 2015a). Similar to chemical pulp plants, the energy mix of paper plants is known to vary among individual plants. For example, paper plants in China are mainly based on coal (Newell and Vos 2012), whereas plants in the US and Canada have a more diverse mix of fossil and renewable fuels (US DOE 2005, Newell and Vos 2012, Nyboer 2015a).

In my study, I used the energy consumption values of an average Canadian paper plant (Nyboer 2015a) (Table 2.11.). The rationale behind this decision was based on insufficient technical coverage of paper plants in the US and China. There are a few studies examining GHG emissions of paper plants and those that do are obscured in scoping issues (i.e. cradle to gate) and/or C-neutral reporting of biogenic GHG emissions.

Table 2.11. GHG emission factors of a paper plant.

Non-Biogenic		Biogenic	Reference
kg CO_2e ADt^{-1}	kg CO_2e tC^{-1}	kg CO_2e ADt^{-1}	
1397¶	NA	0*	Newell and Vos 2012
2175¶	NA	0*	Newell and Vos 2012
975¶	NA	0*	AFPA 2012
1199¶	NA	0*	AFPA 2012
NA	1628¶	0*	Chen et al. 2014
175	NA	0*	Nyboer 2015a

ADt: air-dry tonne of paper.
0* biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).
¶ "cradle to gate" refers to allocating upstream GHG emissions into an industry's GHG totals.
NA: not available

2.1.3.1.3.4.2. Pellet Plant

Pellet plants are the primary beneficiaries of the sawdust and shavings from lumber sawmills in the BC Interior (MFLNRO 2015c). These plants grind and pelletize these mill residues into pellets, which can then be exported to their primary markets in the European Union.

The energy consumption in pellet plants is relatively minor 1.6 to 3.8 GJ t^{-1} (Magelli et al. 2009, Pa et al. 2012). The dryers are the largest consumer of energy in the pellet plant. Consequently, sawdust and shavings are typically separated into two distinct lines in BC pellet mills, each with their own dryer to increase efficiency (Envirochem Inc. 2008). Moreover, the energy demands of these dryers can then be met by burning oversized wood particles in a boiler or by burning natural gas in their boilers (Magelli et al. 2009).

In my study, the pellet plant energy consumption data were taken from Pa et al. (2012) (Table 2.12.). This study surveyed various pellet plants in the BC Interior. Due to the extensive use of electricity and biomass in their surveyed pellet plants, their energy consumption and GHG emissions are quite low in comparison to other pellet plant studies cited in the literature.

Table 2.12. GHG emission factors of a pellet plant.

Non-Biogenic kg CO_2e ODt^{-1}	**Biogenic** kg CO_2e ODt^{-1}	**Reference**
28	0*	Magelli et al. 2009
193¶	0	Magelli et al. 2009
15	0*	Sikkema et al. 2010
167¶	0	Sikkema et al. 2010
8	105	Pa et al. 2012
29	0*	Lamers et al. 2014

ODt: oven-dry tonne pellet
¶natural gas-fired boiler
0*: biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).

2.1.3.1.3.4.3. Bioenergy: Chemical Pulp Plant

Chemical pulp plants are the largest consumers of hogfuel in the Interior (Dymond and Kamp 2014, MFLNRO 2015c, Nyboer 2015a). A recent survey of the bioenergy industry in BC found a wide range in net calorific values for wood fibre in different energy applications (3-16.8 GJ t^{-1}) (Dymond and Kamp 2014). The biogenic GHG emission factor of 1.622 kg CO_2e kg^{-1} for hogfuel (i.e. wood waste) combustion (MOE 2012).

In my study, I chose not to report hogfuel combustion explicitly, but instead, assumed that it was consumed as part of my chemical pulp plants' requirements for purchased hogfuel.

2.1.3.1.3.5. Distribution

The distribution of wood products encompasses the transportation of forest products from their processing facility to supplier distribution centers; distribution centers to retailer distribution centers; and retailer distribution centers to their final use (Gower et al. 2006). The distribution model from processing facilities to supplier distribution centers was found to differ among BC forest companies (IBI 2006). The first model is the "produce to order" distribution system whereby production is matched with specific orders and vessel deliveries. The second model is the "shipped to order" systems whereby forest product industries have a set production and ship from their distribution center. With that said, there is a general lack of data collected on the distribution of forest products (i.e. supply chains). The few studies that have examined distribution have shown that it is relatively small component of a product's total GHG emissions in comparison to manufacturing and disposal GHG emissions (Winistofer 2005, Gower et al. 2006, AFPA 2012, Newell and Vos 2012).

In my study, the general GHG emission factors I used for product distribution were taken from CN (2015), as seen in Sikkema et al. (2013). These factors were: 0.114 kg CO_2e $t^{-1}km^{-1}$ by truck, 0.01785 kg CO_2e $t^{-1}km^{-1}$ by train, and 0.010732 kg CO_2e $t^{-1}km^{-1}$ by boat.

2.1.3.1.3.5.1. Lumber

In 2014, British Columbia exported 5,755 million dollars' worth of softwood lumber out of the province (IC 2015). The destinations of these exports were: United States (53%), China (25%), Japan (13%), Taiwan (2%), Philippines (2%), and others (5%) (IC 2015). The United States has traditionally been the leading importer of BC lumber, however exports to Asia are on the rise. The distribution systems can be very different between markets, which affects their overall GHG emissions. Meil et al. (2004) performed a study of the distance travelled by their building materials. In one of their case studies, wood was transported a very short distance (120 km) by truck to the construction site. In their other case study, wood was shipped very long distances by train (2,538 km) and truck (60 km).

In my study, I assumed that lumber was distributed to the US Midwest (Meil et al. 2004) (Table 2.13.). That study estimated the distribution emissions of lumber from British Columbia to Minnesota, and it is thought that this study is fairly representative of the distribution network established for lumber in North America (Upton et al. 2008).

Table 2.13. GHG emission factors in distributing lumber.

Non-Biogenic		Reference
kg CO_2e m^{-3}	kg CO_2e t^{-1}	
18 (120 km)	NA	Meil et al. 2004
32 (2,598 km)	NA	Meil et al. 2004
NA	NA	Gower et al. 2006
NA	22 (NA)	US EPA 2015
NA	109 (8,805 km)	Sikkema et al. 2013

Meil et al. (2004): 120 km by truck
Meil et al. (2004): 2538 km by train and 60 km by truck.
Sikkema et al. (2013): 780 km by train, 7295 km by boat, and 100 km by truck.
NA: not available

2.1.3.1.3.5.2. Pulp & Paper

In 2014, British Columbia exported 3 billion dollars worth of pulp out of the province (IC 2015). The destinations of these exports were: China (56%), United States (18%), Japan (7%), Italy (5%), South Korea (3%), and others (11%). Studies examining pulp's distribution have assumed long transportation distances to China (e.g. 8,050 km) or to the US Midwest (e.g. 2,414) (Gower 2006, Newell and Vos 2012). Studies examining paper's distribution, on the other hand, have assumed very short transportation distances (AFPA 2012). But overall, the GHG emission factor for pulp and paper distribution is thought to be relatively low in comparison to their other life stages (e.g. chemical pulp plant) (Gower 2006, Newell and Vos 2012).

In my study, I assumed that pulp was distributed to the US Midwest (Newell and Vos 2012) and I assumed the average GHG emission factor for paper distribution in the United States (AFPA 2012) (Table 2.14.). Pulp was distributed to the United States as opposed to China (i.e. the leading importer of Canadian chemical pulp) in my C model because of limited information on Chinese paper plants.

Table 2.14. GHG emission factors in distributing pulp and paper.

Non-Biogenic kg CO_2e t^{-1}	Reference
187 (8,050 km)	Newell and Vos 2012
23 (2,414 km)	Newell and Vos 2012
28* (NA)	AFPA 2012
29 (1,100 km)	Sikkema et al. 2013

*4.25 kg CO_2e $ream^{-1}$ ÷ 0.00215 ADt $ream^{-1}$ × 1.4% = 28 kg CO_2e t^{-1}
Newell and Vos (2012): 8050 km by boat.
Newell and Vos (2012): 2414 km by train.
Sikkema et al. (2013): 50 km by truck, 1000 km by train, and 50 km by truck.
NA: not available

2.1.3.1.3.5.3. Pellets

In 2014, BC exported 200 million dollars' worth of pellets (IC 2015). The destinations of these pellets were: United Kingdom (67%), Italy (14%), South Korea (11%), Japan (6%), and United States (2%). In contrast, much of the literature on BC pellets has focused on the export of pellets to Sweden (Magelli et al. 2009), Netherlands (Pa et al. 2012) or EU-27 (Sikkema et al. 2013).

In my study, I assumed the distribution of pellets described by Pa et al. (2012) (Table 2.15.). This study includes the pellets' travel by train from Prince George to the port of Vancouver and by ship to the Port of Rotterdam.

Table 2.15. GHG emission factors in distributing pellets.

Non-Biogenic kg CO_2e t^{-1}	Reference
221 (16,300 km)	Magelli et al. 2009
96 (17,400 km)	Sikkema et al. 2010
206 (17,500 km)	Pa et al. 2012
135 (NA)	Sikkema et al. 2013

Magelli et al. (2009): 137 km by truck, 750 km by train, and 15500 km by boat.
Sikkema et et al. (2010): 207 km by truck, 781 km by train, and 16500 km by boat.
Pa et al. (2012): 125 km by truck, 840 km by train, and 16668 km by boat.
NA: not available

2.1.3.1.3.6. Use & Disposal

The use and disposal of forest products is often thought to have the greatest contribution to forest product C budgets (Miner and Perez-Garcia 2007, Heath et al. 2010). This contribution, however, is also met with high degree of uncertainty. Due to the longevity of some forest products, modeling is required to estimate their impact over time. This requires intensive use of assumptions with regards to forest product use (e.g. single-family home, multi-family home, commercial) and its longevity (i.e. half-life); methods of forest product disposal (e.g. recycling, landfilling, combustion); and disposal sites and their decomposition rates, conditions and technologies (i.e. methane capture).

2.1.3.1.3.6.1. Lumber & Paper

Lumber and paper's disposal GHG emissions are comprised of biogenic (i.e. CO_2) and non-biogenic GHG emissions (i.e. CH_4) arising from their decomposition in landfills. When these products decompose, they generate CO_2 and CH_4 emissions. The CH_4 generated emissions can be converted into CO_2 using CH_4 capture technology, or it can be oxidized by soils in the landfill. The remaining CH_4 emitted is a potent GHG emission not eligible for a C-neutral status even though it is derived from biomass (MOE 2012).

In my study, I used the disposal parameters provided in Dymond (2012) (Table 2.16.). These parameters had a higher percentage of landfills with CH_4 capture technology and efficiency and a lower CH_4 oxidation rate than similar studies performed elsewhere in Canada (Chen et al. 2014).

Table 2.16. Parameters governing GHG emissions of lumber and paper products while in disposal.

Disposal of Lumber & Paper	Dymond 2012	Chen et al. 2014
Decomposition	50% CH_4	50% CH_4
Percentage of landfills with CH_4 capture technology	82%	29.2%
CH_4 capture efficiency	78%	51%
CH_4 oxidation in soils	22%	36%

2.1.3.1.3.6.2. Pellets

In my study, pellets' use and disposal was modeled after pellet stove in Pa et al. (2013) (Table 2.17.). This study estimated the GHG emission factor for burning pellets in a pellet stove. Biogenic GHG emission factors for pellet combustion were sparse in the scientific literature due to the assumption that these emissions are C neutral.

Table 2.17. GHG emission factors for pellets in use and disposal.

Non-Biogenic kg $CO_2e\ t^{-1}$	Biogenic kg $CO_2e\ t^{-1}$	Reference
NA	0*	Magelli et al. 2009
NA	0*	Sikkema et al. 2010
45¶	1731	Pa et al. 2013
19	1731	Pa et al. 2013
NA	0*	Sikkema et al. 2013

* biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).
t: tonne pellet
¶ pellet stove
NA: not available

2.1.3.2. Pellets (Scenario 2)

2.1.3.3. Material and Product Flow

2.1.3.3.1. Pellet Recovery

There is not a whole lot of information on pellet recovery specific to pellet plants sourcing roundwood. Most of the work on pellet recovery has come from pellet plants sourcing sawmill residues. Zhang et al. (2010) assumed a recovery of 85% (Zhang et al. 2010) for roundwood processing at a pellet plant. The remaining 15% of the harvested roundwood is burned in the pellet plant's dryers.

In my study, I used Zhang et al.'s (2010) pellet recovery of 85% (on an oven-dry weight basis). This recovery was consistent with that reported by Katers et al. (2012) for a pellet plant in the US Inland Northwest sourcing roundwood (Table 2.18.).

Table 2.18. Pellet recovery for pellet plants sourcing roundwood.

Lumber Recovery $\%_{OD}$	Reference
85	Zhang et al. 2010
85	Katers et al. 2012

2.1.3.2.2. GHG Emission Profile

2.1.3.2.2.1. Harvesting

The predominant silvicultural system in the BC Interior is the clearcut-with reserves-system (MFLNRO 2015d). The transition from traditional forest operations (for lumber, plywood, etc.) to proposed operations for pellets depends largely on the material input demands (e.g. whole logs, chips) of the receiving, primary processing facility (i.e. pellet plant) (Lindroos et al. 2011). As a result, the silvicultural and harvesting systems vary according to these mill demands. The notion of harvesting MPB-attacked forests for bioenergy also coincides with the notion of a more intensive harvest. For example, studies examining salvage operations have compared chipping whole trees and solely using harvest residues. However, the consensus on this matter has been that it is not economically feasible to chip these materials on site (MacDonald 2006, Abbas et al. 2011). The alternative then has had the pellet industry adopting more traditional harvesting systems in collecting roundwood as whole logs. Zhang et al. (2010) assumed traditional harvesting systems in their modeling of a pellet plant sourcing roundwood.

In my study, harvesting GHG emissions were taken from an energy consumption survey reported in Sambo (2002) (Table 2.19.). This was the same survey that was used to construct an emission factor for harvesting in scenario 1. One of the advantages of this overlap is that the assumptions regarding the forest conditions are the same. Sambo's (2002) forest conditions had an average cutblock area of 24 ha, merchantable timber volume of 244 m^3 ha^{-1}, and a volume of 5,856 m^3. A contemporary study by Zhang et al. (2010) did an LCI on whole-tree harvesting for pellets in northern Ontario that had a stocking of 150 m^3 ha^{-1} (or 66 ODt ha^{-1}).

Table 2.19. GHG emission factors in harvesting forests.

Non-Biogenic			Reference
kg CO_2e m^{-3}_{rw}	kg CO_2e ODt^{-1}_{rw}	kg CO_2e ODt^{-1}_{pellet}	
10	NA	NA	Sambo 2002
NA	36	NA	Zhang et al. 2010
NA	NA	60.6	Lamers et al. 2014

m^3_{rw}: cubic meter (m^3) of roundwood.
ODt_{rw}: oven-dry tonne (ODt) of roundwood.
ODt_{pellet}: ODT of pellet.
NA: not available

2.1.3.2.2.2. Transportation

The transportation system and equipment used to support the pellet industry can accommodate a wide range of material inputs (MacDonald 1999, 2006). These inputs include whole logs, short logs, residues, and wood chips (Lindroos et al. 2011). Due to economic feasibility, some studies have opted to transport whole logs directly to the pellet plants (Zhang et al. 2010, Lamers et al. 2014).

In my study, the emission factor for transportation is taken from Sambo (2002) (Table 2.20.). This inventory models the transport of logs from the harvest area to the primary processing facility (i.e. pellet plant). The GHG emission factor reported by Sambo (2002) is similar to those reported elsewhere in the literature.

Table 2.20. GHG emission factors for transporting roundwood.

Non-Biogenic			Reference
kg CO_2e m^{-3}_{rw}	kg CO_2e ODt^{-1}_{rw}	kg CO_2e ODt^{-1}_{pellet}	
10	NA	NA	Sambo 2002
NA	27	NA	Zhang et al. 2010
NA	NA	16	Lamers et al. 2014

m^3_{rw}: cubic meter (m^3) of roundwood.
ODt_{rw}: oven-dry tonne (ODt) of roundwood.
ODt_{pellet}: ODt of pellet.
NA: not available

2.1.3.2.2.3. Primary Processing Industry: Pellet Plant

Pellet plants sourcing whole logs would require an additional processing line to that commonplace in the BC Interior. Pellet plants sort wood fibre based on the wood particle size and moisture content of their input materials. The process line for roundwood would require it to be debarked, chipped, ground, and dried before becoming integrated into the main processing line (Zhang et al. 2010). Roundwood's drying requirements result in greater energy consumption and GHG emissions compared to pellet plants sourcing sawdust and shavings (Katers et al. 2012).

In my study, my pellet plant GHG emission data were based on Zhang et al. (2010) (Table 2.1.). A detailed GHG inventory of BC pellet plants processing whole logs was not available, so Zhang et al. (2010) was used as it modeled a whole-tree pellet processing facility in Ontario.

Table 2.21. GHG emission factors for processing roundwood at a pellet plant.

Non-Biogenic kg CO_2e ODt^{-1}	Biogenic kg CO_2e ODt^{-1}	Reference
34	0*	Zhang et al. 2010
59	0*	Sikkema et al. 2010
8	0*	Pa et al. 2012
190¶		Lamers et al. 2014

ODt: oven-dry tonne (ODt) of pellet.
0* biogenic CO_2 emissions are confidential and/or C neutral (i.e. zero GHG emissions).
¶Assumed Lamers et al. (2014) combined biogenic and non-biogenic GHG emission factors.

2.1.3.2.2.3. Distribution

Refer back to the section on pellet distribution described in lumber's industrial system (scenario 1) (§ 2.1.3.1.3.5.3.).

2.1.3.2.2.4. Use & Disposal

Refer back to the section on pellet use and disposal described in lumber's industrial system (scenario 1) (§ 2.3.1.3.6.2.).

2.1.4. Calculations

Various calculations and conversions were required in compiling all of the data collected for each of my industrial forest product systems (Table 2.22., Appendix I-III). They were compiled according to a reference unit. This reference unit was an oven-dry tonne (ODt) of wood fibre (Table 2.23., 2.24.).

I calculated my C budgets based off of this *reference unit* framework using my *experimental unit* of 1 m^3 roundwood (or 0.409 ODt m^{-3}).

2.1.4.1. Material & Product Flow

My lumber recovery yield was measured on a thousand-board-foot basis (mbfm) at ~15% MC_{OD} (Orbay and Goudie 2006). This yield assumes that the moisture content (MC) is above the fibre saturation point (FSP) of wood (~30% MC_{OD}). However, current findings by FII (2009) suggest that roundwood is below the FSP point (at 20.4% MC_{OD}) at which point MC adversely affects volume. My study converted "green" volume of roundwood (i.e. MC present in the field) to an oven-dry (OD) volume to facilitate C accounting amongst material and product flows (Table 2.22a.) (Briggs 1994 p.5).

2.1.4.2. Biogenic C Storage

The amount of forest products in use was calculated using a first order decay function (Table 2.22b.) (Eggleston et al. 2006 p.12.11). This function was prescribed to each of a forest product's uses and half-lives (Tables 2.3. to 2.5.). The amount of forest product still in use at year 100 was then converted into biogenic C storage (Table 2.22c.) (MOE 2011 p.87).

The amount of forest products in landfill disposal was calculated by summing up the amount of products left in its degradable and non-degradable pools. The non-degradable portion of forest products goes unchanged, whereas the degradable portion of forest products decomposes. Its decomposition over time is calculated using a first order decay function (Table 2.22b.) (Eggleston et al. 2006 p.12.11). Biogenic C storage is then calculated by converting the total amount of forest products remaining in disposal at Year 100 into units of carbon

(Table 2.22e) (MOE 2011 p.87).

2.1.4.3. GHG Emissions Factors

The non-biogenic GHG emission factors for most of forest product's life stages were calculated by multiplying energy consumed per unit product by energy-specific GHG emission factors (Table 2.22f., Appendix I) (MOE 2011 p.89). The biogenic GHG emission factors for each of these life stages were calculated in a similar fashion (Table 2.22f., Appendix II) (MOE 2011 p.98). These product-based GHG emission factors are then converted to material-based factors using various product-material conversion factors (Appendix III).

The non-biogenic GHG emission factors for distribution were based on CAS's (2011) amount and distribution approach for product distribution (Table 2.22h.) (MOE 2011 p.95).

The non-biogenic GHG emission factors for landfill disposal were calculated by first deriving the methane (CH_4) generated from the forest product under conditions of landfill disposal. The amount of CH_4 generated is calculated by multiplying the amount of material decomposed with the fraction that emits as CH_4 (Table 2.22i.). The actual CH_4 emitted is then calculated by multiplying the amount of CH_4 generated by the percentage of CH_4 recovery (and its efficiency), as well as the CH_4 that is oxidized by soil (Table 2.22j.).

The biogenic GHG emission factors for landfill disposal were calculated by deriving the amount of CO_2 generated and emitted in the landfill (Table 2.22k).

Table 2.22. List of equations used in calculating the C budgets of forest products harvested from MPB-attacked forests.

Material Flow
a) $Shv_x = Shv_t \times (fsp - MC_{OD,x}) \div fsp$
Biogenic C Storage
b) $C(t+1) = e^{-k} \times C(t) + (1-e^{-k}) \div k \times I(t)$
c) $GHG_{CO2,HWP_{in-use},t} = \sum_k\left[\sum_{x=0}^{p}(m_{k,t,x} \times f_{C,in-use,k,(100-x)}) \times (1 - f_{Production\ loss,k})\right] \times f_{C,wood} \times \frac{MW_{CO2}}{MW_C}$
d) $C = [C_{hwp} \times (1-D_f)] + [C_{hwp} \times D_f \times (e^{-k})^t]$
e) $GHG_{CO2,HWP_{in\ landfill},t} = \sum_k\left[\sum_{x=0}^{p}(m_{k,t,x} \times f_{C,in\ landfill,k,(100-x)} \times (1 - f_{Production\ loss,k})\right] \times f_{C,wood} \times \frac{MW_{CO2}}{MW_C}$
GHG Emission Factors
f) $GHG_{j,Emission\ Source_i,t} = EF_{i,j} \times AL_i \times CF$
g) $GHG_{j,PE8/BE8,t} = \sum_b EF_{b,j} \times AL_{b,t} \times CF_b$
h) $GHG_{j,PE6/BE6,t} = \sum_m\left[EF_{m,j} \times \sum_g(D_{m,g} \times C_{m,g,t}) \times CF_m\right]$
i) $M_{gen} = C_{hwp} \times D_f \times (1-(e^{-k})^t) \times M_f$
j) $M_{emit} = (1-R_{col}) \times M_{gen} \times (1-R_{oxi})$
k) $GHG_{CO2,HWP_{in\ landfill},t} = \sum_k\left\{\sum_{x=0}^{p}\left[(m_{k,t,x} \times (1 - f_{C,in-use,k,(100-x)} - f_{C,non\ landfill,k,(100-x)} - f_{C,in\ landfill,k,(100-x)}) \times (1 - f_{Production\ loss,k})\right] \times f_{C,wood} \times CH_{4,LFG} \times \frac{MW_{CH4}}{MW_C}\right\}$

Glossary:

AL_i or $AL_{b,t}$	The quantity of input/output or "activity level" for emission source *I* (e.g. volume of fuel combusted, amount of fertilizer applied, etc.).
CF or *CF_b* or *CF_m*	The conversion factor to be used when the units of the activity level do not match those of the emission factor. Where both the activity level and emission factor are expressed in the same units, CF would be set to 1.
$CH_{4,LFG}$	Molar % CH_4 in landfill gas.
C_{hwp}	C in harvested wood products
$C_{m,g,t}$	Total quantity of material, equipment, input, or personnel g transported the same distance using transport mode *m* during reporting period *t*. Where the same type of good is transported different distances to arrive at the project or baseline site, they should be treated as separate goods for the purposes of this calculation.
D_f	fraction of C_{hwp} that decomposes
$D_{m,g}$	transport distance for material, equipment, input, or personnel *g* using transport mode *m*.
$EF_{i,j}$ or $EF_{b,j}$ or $EF_{m,j}$	the emission factor for GHG *j,b,m* and emission source *I* [e.g. tonne CO_2/(activity or input/output)]
$f_{C,\ wood}$	the fraction of the dry mass of wood, excluding bark, that is carbon.
$f_{C,in-use,k,100-x}$	the fraction of C in HWPs of type *k* that remain in-use after (100–*x*) years.
$f_{C,in-landfill,k,(100-x)}$	The fraction of C in HWPs of type *k* that remains in landfill after (100-x) years.
$f_{C,non\ landfill,k,(100-x)}$	The fraction of carbon in HWPs of type *k* that have been discarded but not sent to landfill after 100 – x years.

$f_{production\ loss,k}$	The fraction of wood mass lost as residuals during production of HWP *k*
fsp	fibre saturation point (~30% MC_{OD})
$GHG_{CO2,HWP,in\ landfill,\ t}$	Mass of carbon dioxide, in tonnes, that remains stored in in landfill project or baseline HWPs harvested in reporting period *t*, 100 years after initial sequestration in the tree from which it is derived or after the start of the project, whichever is later.
$GHG_{CO2,\ HWP\ in\text{-}use,\ t}$	Mass of carbon dioxide, in tonnes, that remains stored in in-use project or baseline HWPs harvested in reporting period *t*, 100 years after initial sequestration in the tree from which it is derived or after the start of the project, whichever is later.
$GHG_{j,Emission\ Source,l,t}$	emissions of GHG *j* from emission source *l* during reporting period *t*.
$GHG_{j,PE8/BE8,t}$	Emissions of GHG_j , in tonnes, from on-site vehicle and equipment fuel combustion during reporting period *t*. Note that for this SSP, only CH_4 and N_2O are to be reported, as CO_2 is tracked as part of forest C pools.
$GHG_{j,PE6/BE6,t}$	emissions of GHG_j , in tonnes, from transportation of materials, equipment, inputs, and personnel to the project / baseline site during reporting period *t*.
GHG_f	GHG emission factor per unit energy (or fuel) (see Appendix I)
HF	half life
I(t)	inflow of HWP in a particular year
k	constant annual decomposition rate ($k = \ln(2) \div t_{1/2}$)
k	relevant HWP types.
M_f	methane fraction in landfill emissions
M_{gen}	methane generated by decomposition
$m_{k,\ t,\ x}$	dry mass, in tonnes, of harvested wood, minus bark, harvested in reporting period *t*, that grew *x* years prior to harvest, and that will be processed into HWP *k*.
$MC_{OD,x}$	intermediate moisture content
MW_C	molecular weight of carbon
MW_{CO2}	molecular weight of CO_2
MW_{CH4}	molecular weight of CH_4
p	A number of years prior to the harvest. *x* ranges from 0 (i.e. the year of harvest) to *p* , where *p* represents the lesser of the age in years of the oldest tree that is harvested in a given reporting period; and the number of years from project start to the end of reporting period.
SHv_x	volumetric shrinkage
Shv_t	volumetric shrinkage constant from 0-30% MC_{OD}
R_{col}	landfill CH_4 collection rate
R_{oxi}	the oxidation rate for CH_4 reaching the top layer of waste covering soil.
t	year
$t_{1/2}$	half life
x	A number of years prior to the harvest. *x* ranges from 0 (i.e. the year of harvest) to *p* , where *p* represents the lesser of the age in years of the oldest tree that is harvested in a given reporting period; and the number of years from project start to the end of reporting period.

Table 2.23. Individual components in lumber's C budget model based on a reference unit of oven-dry tonne (ODt) of wood fibre.

a.) Material and Product Flow			
	$\%_{OD}$		**Reference**
Lumber Recovery	40		Orbay and Goudie 2006
Pulp Chip Recovery	35		ASMI 2012
Sawdust & Shavings Recovery	9		ASMI 2012
Hogfuel Recovery	16		ASMI 2012
Chemical Pulp Recovery	46		Canfor Pulp Inc. 2012a
Pellet Recovery	89		Pa et al. 2012
b.) Biogenic C Storage			
	kg CO_2e ODt^{-1}		**Reference**
Lumber Use	-403		Dymond 2012
Lumber Disposal	-1132		Dymond 2012
Paper Use	0		Dymond 2012
Paper Disposal	-829		Dymond 2012
c.) GHG Emission Factors			
	Biogenic kg CO_2e ODt^{-1}	**Non-Biogenic kg CO_2e ODt^{-1}**	**Reference**
Harvest	0	24	Sambo 2002
Transportation	0	27	Sambo 2002
Primary Processing	67	21	ASMI 2012
Secondary Processing: Chemical Pulp Plant	1532	89	Canfor Pulp Inc. 2012a
Secondary Processing: Paper Plant	615	199	Nyboer et al. 2015
Secondary Processing: Pellet Plant	94	7.4	Pa et al. 2012
Distribution: Lumber	64	64	Meil et al. 2004
Distribution: Pulp & Paper	51	51	Newell and Vos 2012, AFPA 2012
Distribution: Pellets	183	28	Pa et al. 2012
Use & Disposal: Lumber	262	377	Dymond 2012
Use & Disposal: Paper	892	1286	Dymond 2012
Use & Disposal: Pellets	1541	40	Pa et al. 2013

Table 2.24. Individual components in pellets' C budget model based on a reference unit of oven-dry tonne (ODt) of wood fibre.

a.) Material Flow			
	$\%_{OD}$		**Reference**
Pellet Recovery	85		Zhang et al. 2010
b.) GHG Emission Factors			
	Biogenic kg CO_2e ODt^{-1}	**Non-Biogenic kg CO_2e ODt^{-1}**	**Reference**
Harvest	0	24	Sambo 2002
Transportation	0	26	Sambo 2002
Primary Processing: Pellet Plant	235	2	Zhang et al. 2010
Distribution	0	183	Pa et al. 2012
Use & Disposal: Pellets	1419	37	Pa et al. 2013

Sensitivity Analysis

A sensitivity analysis was performed on the C budgets of my two scenarios and their respective case studies. This analysis was performed by increasing individual parameter values by 10% and by observing their overall impact on the net C balance. The purpose of running a sensitivity analysis is to determine the most sensitive parameters in a model. My intent was to know where the greatest point of emphasis should be placed in refining parameters in the forest product industrial system.

Chapter 3.

C BUDGETS OF TWO DIFFERENT FOREST PRODUCTS (LUMBER, PELLETS) HARVESTED FROM BEETLE-ATTACKED FORESTS IN THE PRINCE GEORGE REGION.

3.0. RESULTS

3.0.1. C Fluxes Associated with the Industrial Systems of Lumber and Pellet Production

3.0.1.1. Lumber (Scenario 1)

3.0.1.1.1. Material & Product Flows

In an industrial system defined by lumber production (scenario 1), roundwood harvested from MPB-attacked forests is first processed at a lumber sawmill (Figure 3.1.). The lumber recovery for MPB-attacked roundwood at this sawmill was modeled with a 40% recovery yield of the original roundwood oven-dry (OD) weight. Meanwhile, the mill residue recovery at the sawmill was 60% and it was composed of 35% pulp chips, 16% hogfuel, and 9% sawdust and shavings.

Sawmill residues were processed in secondary processing industries (Figure 3.1.). These industries included chemical pulp, paper, and pellet plants. The chemical pulp plant recovered 16% of original roundwood volume as a chemical pulp product. It was assumed that the paper plant maintained this recovery during its processing of chemical pulp into paper (i.e. ~100% recovery). The pellet plant recovered 8% of the original roundwood volume as pellets.

Collectively, the secondary processing industries equated to a 24% recovery of the original roundwood volume as secondary wood products (i.e. paper and pellets). The remaining primary and secondary mill residues (i.e. black liquor and hogfuels) amounted to 36% of original roundwood volume and were used as a source of biomass fuel in bioenergy processes.

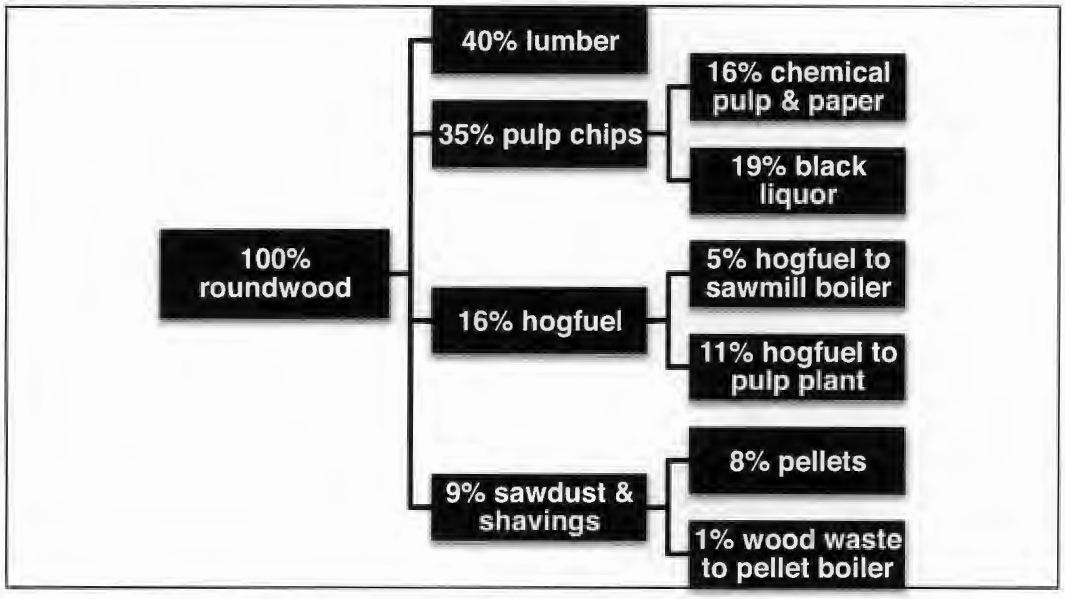

Figure 3.1. Material and product flow of 1 m^3 of dead merchantable lodgepole pine (*Pinus contorta* var. *latifolia*) roundwood in an industrial system defined by lumber production (scenario 1). For details, see Table 2.23.

3.0.1.1.2. Biogenic C Storage

Lumber and paper were the only forest products capable of physically storing biogenic carbon (C) in lumber production (Figure 3.2., 3.3.). Their storage was confined to the use and disposal stages of this industrial system.

The biogenic C stored in forest products while in use decreased over time (Figure 3.2.). In the beginning (Year 0), forest products are in use and amount to 56% of the original roundwood volume being stored (Table 3.3). At Year 100, forest product remaining in use amounted to 9% of the original roundwood volume and its composition at this time was predominantly lumber (Figure 3.3.).

The biogenic C stored in forest products while in disposal increased over time (Figure 3.2.). In the beginning (Year 0), there were no forest products in disposal, and so, there was no biogenic C storage. At Year 100, products in disposal amounted to 26% of the original roundwood volume being stored (Figure 3.3.). The composition of this storage at Year 100 was 19% lumber and 7% paper.

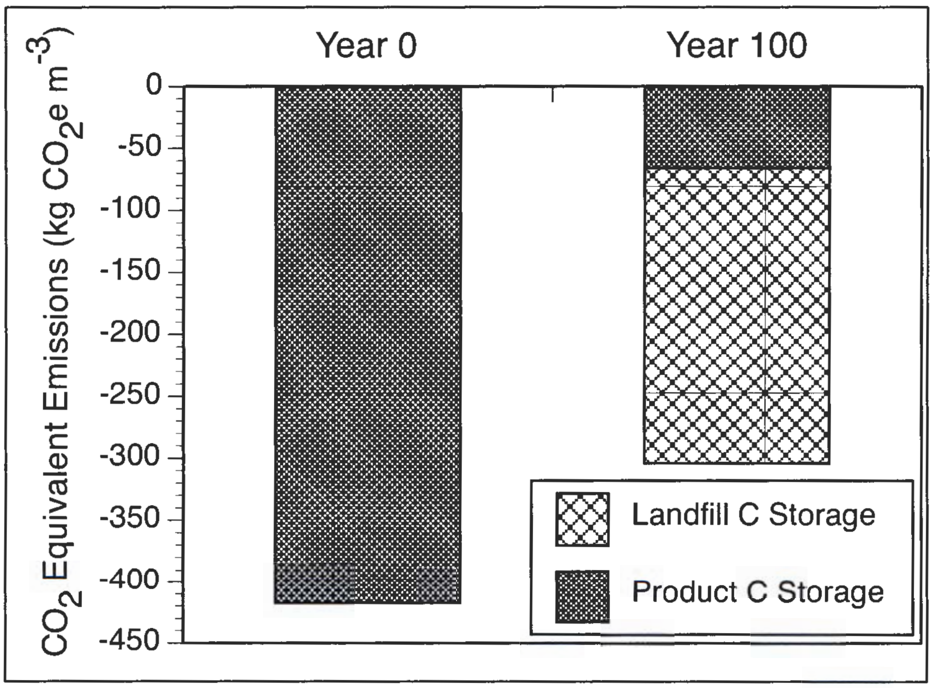

Figure 3.2. Change in biogenic C storage while in use (i.e. product C storage) and in disposal (i.e. landfill C storage) over time.

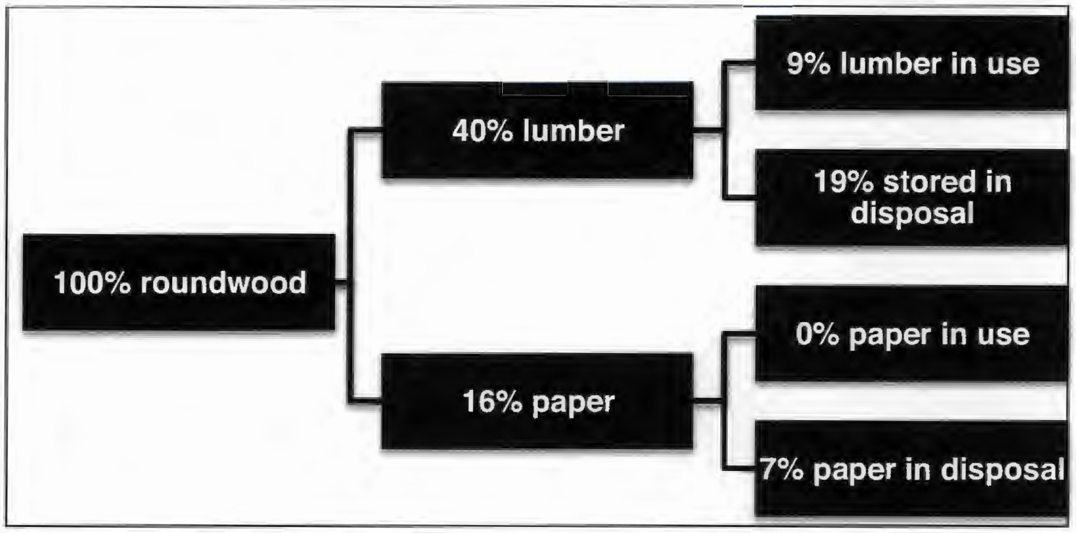

Figure 3.3. The distribution of biogenic C storage amongst forest products in lumber's industrial forest product system (scenario 1).

3.0.1.1.3. GHG Emission Profile

In my first scenario, the GHG emissions attributable to lumber production were complex and extensive (Figure 3.4.). Its GHG emission profile is described here briefly based on its (1) geographical, (2) temporal, and (3) physical aspects. (1) The geographical aspect of lumber production spanned many different geographical areas. The GHG emissions that occurred inside the province were harvest, transportation, primary processing and most secondary processing. The GHG emissions that occurred outside of the province were secondary processing (i.e. paper plant), distribution, and the use and disposal of wood products. (2) The temporal aspect of GHG emissions in lumber production is long-lasting due to the longevity of some forest products in use and in disposal. (3) The physical (or procedural) aspect of GHG emissions in lumber production encompasses multiple forest products and their life cycles.

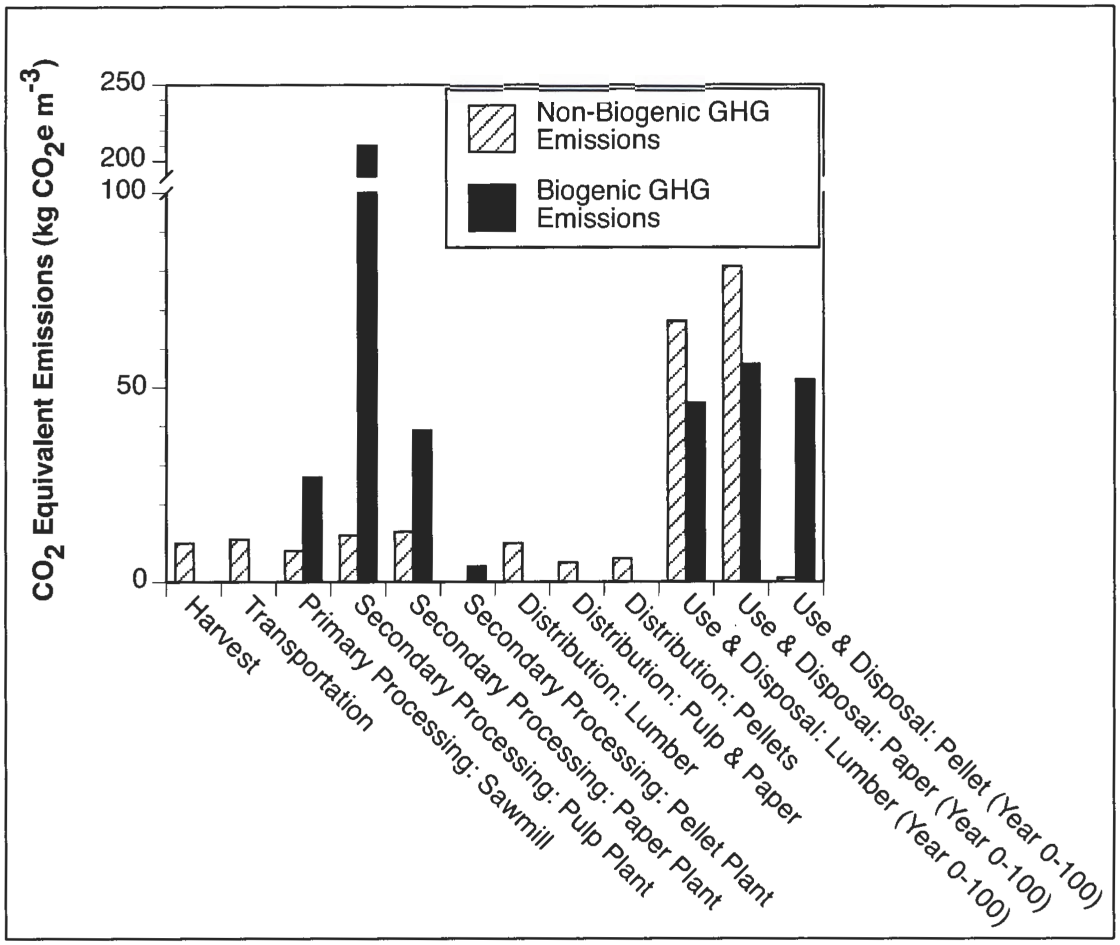

Figure 3.4. GHG emission profile in lumber production (scenario 1) based on 1 m^3 of dead merchantable lodgepole pine (*P. contorta*) roundwood. The GHG emissions from Use & Disposal are cumulative over time. Reference: m^3 = 0.409 ODt = 0.2045 tC = 750 kg CO_2e

3.0.1.1.3.1. Biogenic GHG Emissions

The greatest amount of biogenic GHG emissions in my C budget was released during the secondary processing of pulp chips at chemical pulp plants (Figure 3.5.). These emissions were derived from the use of black liquor and purchased hogfuel. From an accounting perspective, these emissions occurred in BC; they were emitted immediately, and they pertain to the mill residue stream of lumber production.

My forest C budget calculated biogenic GHG emissions (a) by life stage and forest product (Figure 3.5a.). Alternatively, the biogenic GHG emissions in my C budget could be broken down (b) by life stage, (c) by life stage and time, and (3) by parent wood product. (b) By breaking down biogenic GHG emissions by life stage, I found that the secondary processing of mill residues was the largest source of its emissions (Figure 3.5b.). (c) By breaking down these emissions by life stage and time, I found that secondary processing's emissions are immediate, in contrast to the prolonged decomposition emissions via forest product use and disposal (Figure 3.5b,c.). (d) By breaking down biogenic GHG emissions by their parent wood product I found that paper was the largest emitter in this system (Figure 3.5d.). Moreover, sawmill residues (i.e. pulp chips, sawdust and shavings, hogfuel), collectively, were responsible for 60% of the total biogenic GHG emissions in lumber's industrial system (i.e. 433 kg CO_2e m^{-3}).

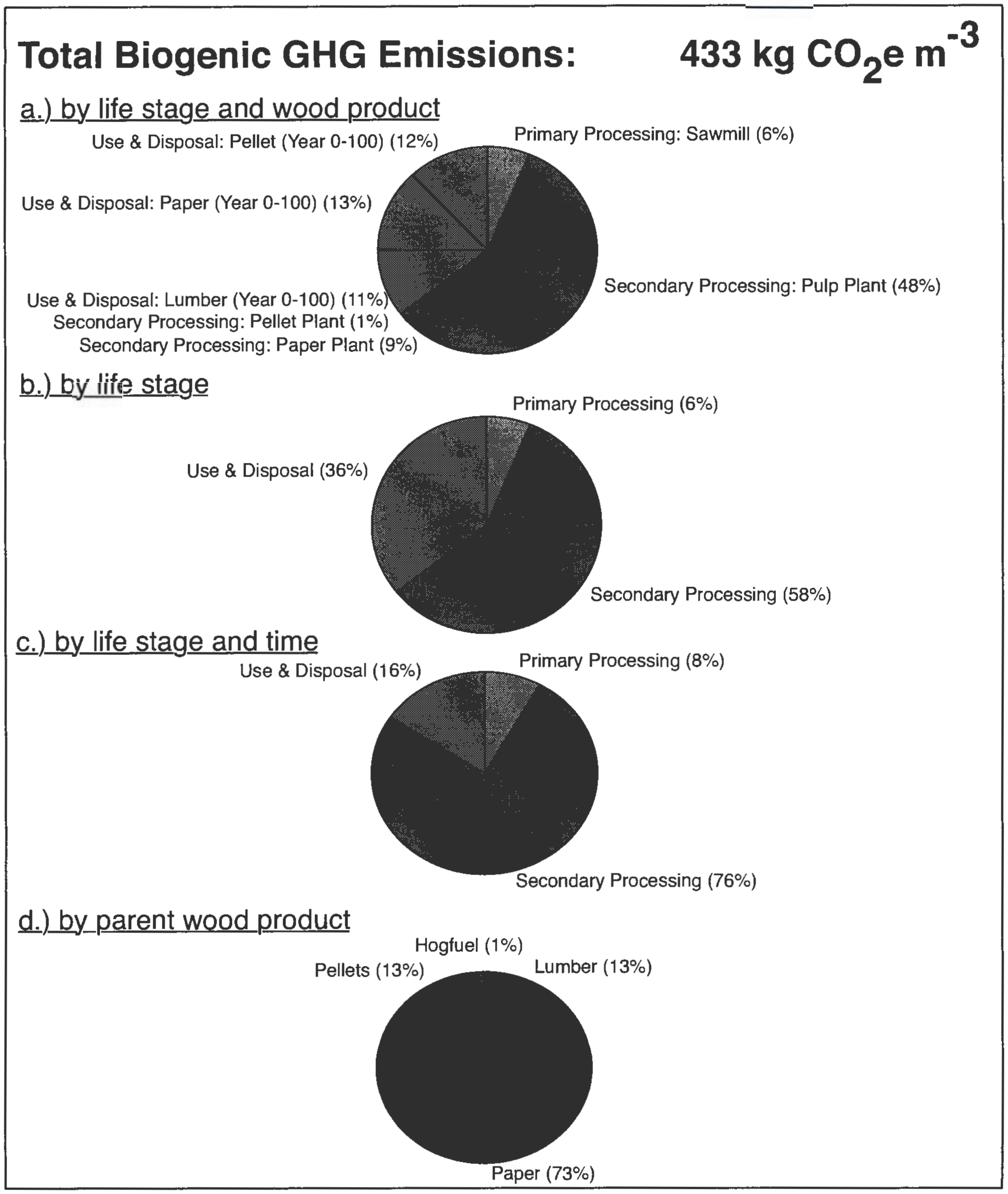

Figure 3.5. Alternative ways of interpreting the biogenic GHG emissions in lumber production (scenario 1).

3.0.1.1.3.2. Non-Biogenic GHG Emissions

The greatest amount of non-biogenic GHG emissions was released in the use and disposal of paper (Figure 3.6.). These emissions are released during the decomposition of paper in landfills. From an accounting perspective, their emissions occur outside of BC; they are emitted over time; and they pertain to the mill residue (or by-product) stream of lumber's industrial system.

My forest C budget calculated non-biogenic GHG emissions (a) by life stage and by wood product (Figure 3.6a.). Alternatively, the non-biogenic GHG emissions can be broken down (b) by life stage, (c) by life stage and time, and (d) by parent wood product. (b) By breaking down the non-biogenic GHG emissions by life stage, I found that the use and disposal of forest products was the greatest GHG emitter (Figure 3.6b.). (c) By breaking down these same emissions by life stage and time, I found that forest product's distribution and secondary processing GHG emissions are immediate, whereas their use and disposal GHG emissions occur over the long-term (Figure 3.6b, c). (d) By breaking down the non-biogenic GHG emissions by its parent wood product I found that paper was the largest source of non-biogenic GHG emissions (Figure 3.6d.).

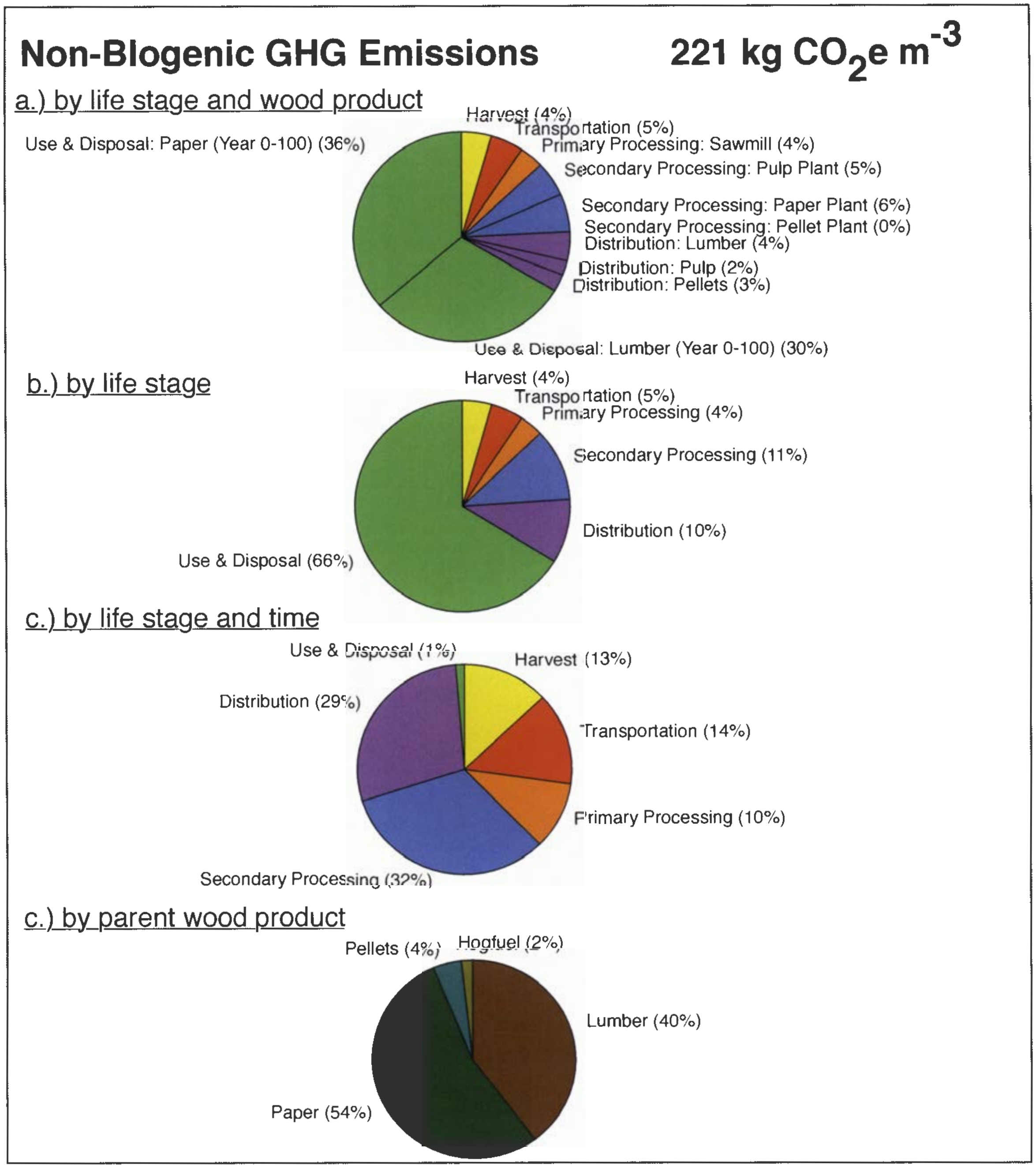

Figure 3.6. Alternative ways of interpreting the non-biogenic GHG emissions in lumber production (scenario 1).

3.0.1.2. Bioenergy: Pellets (Scenario 2)

3.0.1.2.1. GHG Emission Profile

In my second scenario, the GHG emissions attributable to the pellets were straightforward and simplistic (Figure 3.7.). Its GHG emission profile is discussed here briefly based on its (1) geographical, (2) temporal, and (3) physical aspects. (1) The geographical aspect of GHG emissions in pellet production spanned different countries. The emissions that occurred in BC were: harvest, transportation, and primary processing. The emissions that occurred outside of BC were: distribution, use and disposal. (2) The temporal aspect of GHG emissions in pellet production was short-lived. (3) The physical (or procedural) aspect was well defined.

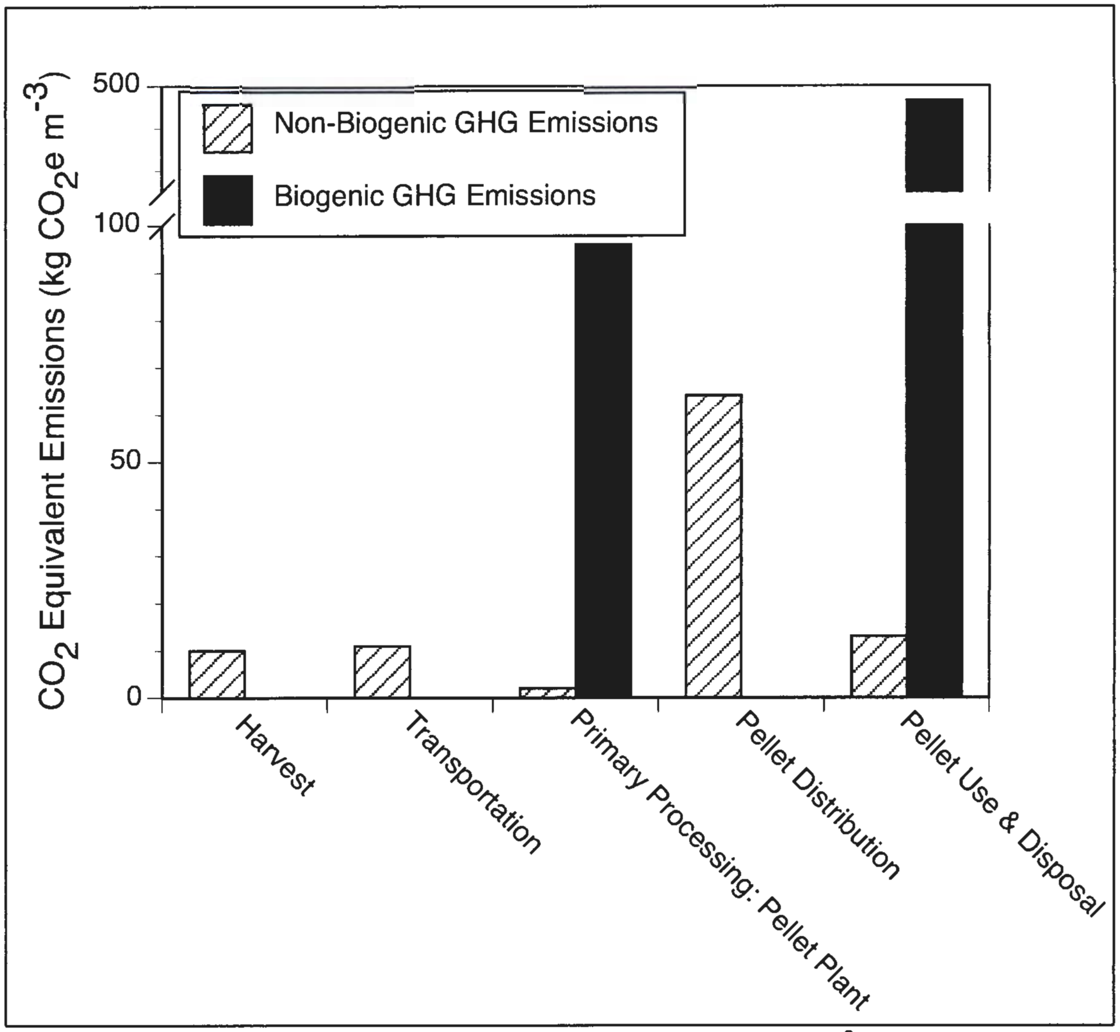

Figure 3.7. GHG profile of pellet production based on 1 m^3 of dead merchantable lodgepole pine (*P.* contorta) roundwood. The biogenic GHG emission factors for wood waste and pellet combustion varied in the literature (Appendix II). <u>Reference:</u> 1 m^3 = 0.409 ODt = 0.2045 tC = 750 kg CO_2e.

3.0.1.2.1.1. Biogenic GHG Emissions

The greatest amount of biogenic GHG emissions were emitted during the use and disposal of pellets (Figure 3.8.). Pellets are burned immediately in pellet stoves. These emissions occur in Europe, in the short term, and are typically not reported in GHG inventories because they are assumed to be C neutral and are accounted for as zero GHG emissions.

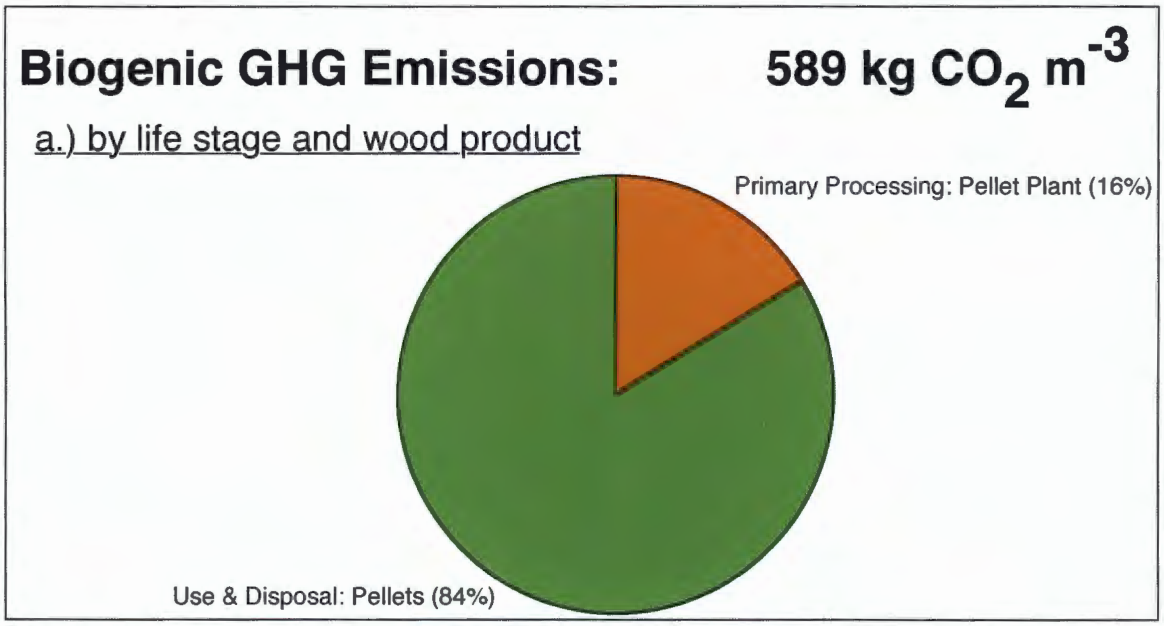

Figure 3.8. Breakdown of the biogenic GHG emissions in the pellet production (scenario 2).

3.0.1.2.1.2. Non-Biogenic GHG Emissions

The greatest amount of non-biogenic GHG emissions was emitted during the distribution of pellets (Figure 3.9.). Pellets are shipped overseas by boat to international markets. These emissions occur between BC and Europe, in the short term, and are typically included in pellet C budgets.

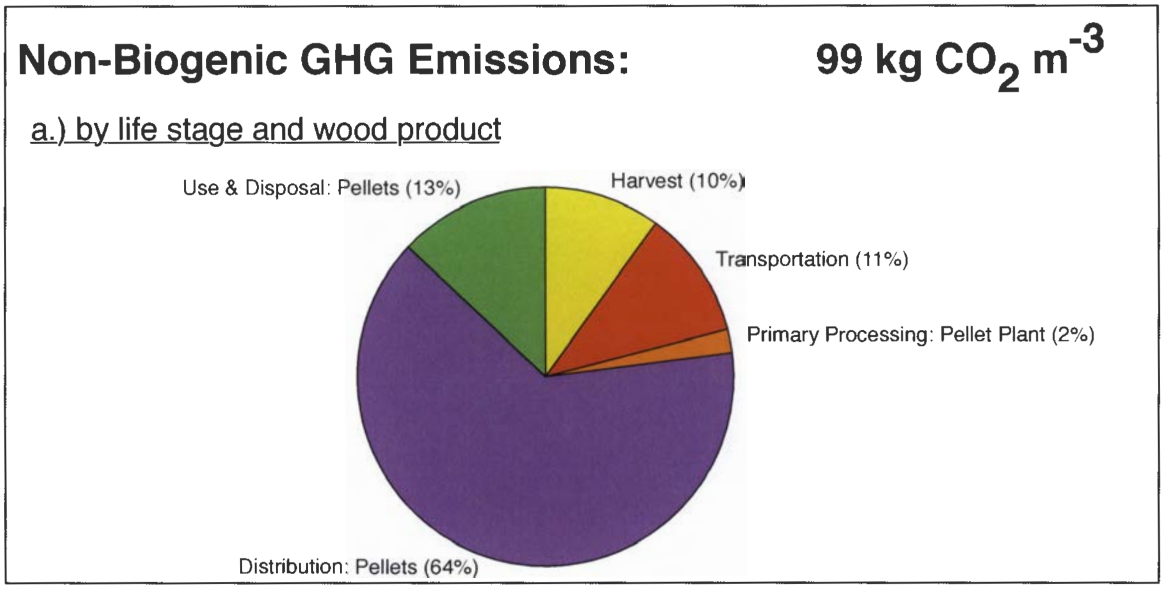

Figure 3.9. Breakdown of the non-biogenic GHG emissions in pellet production (scenario 2).

3.0.2. C Budget of Forest Products Harvested from MPB-Attacked Forests

3.0.2.1. Lumber

If I accepted the assumption that biogenic GHG emissions are C neutral (Case 1), roundwood harvested for lumber had net C storage (-82 kg CO_2e m^{-3}; Table 3.1a.). In this context, biogenic C storage outweighed the non-biogenic GHG emissions in my C budget. The largest source of biogenic C storage and non-biogenic GHG emissions occur at the same life stage. This life stage was the use and disposal phase of forest products. It was a dynamic life stage that had both biogenic C storage and non-biogenic GHG emissions change over time. In the end, the use and disposal phase accounted for 86% of the total C fluxes (i.e. C storage and GHG emissions) in my C budget. Alternatively, the breakdown of the total C fluxes by parent wood product was 54% lumber and 36% mill residues.

If I rejected the assumption that biogenic GHG emissions are C neutral (Case 2), roundwood harvested for lumber had net C emissions (350 kg CO_2e m^{-3}; Table 3.1b.). In this context, the amount of biogenic and non-biogenic GHG emissions was greater than the biogenic C storage in my C budget. The largest source of these GHG emissions was in the secondary processing and the use and disposal of forest products. These emissions differed in that the secondary processing's emissions occur in the near term, whereas use and disposal's emissions occur over the long term. Alternatively, it can be said that the use and disposal phase accounted for 61% of the total C fluxes in my C budget.

Table 3.1. The net C balance of lumber production a.) assuming biogenic GHG emissions are C neutral and b.) assuming biogenic GHG emissions are not C neutral.

a.) C Neutral

	Non-Biogenic GHG Emissions kg CO_2e m^{-3} (%)	Biogenic GHG Emissions kg CO_2e m^{-3} (%)	Biogenic C Storage kg CO_2e m^{-3} (%)
Harvest	10 (4)		
Transportation	11 (5)		
Primary Processing: Sawmill	8 (4)	0 (0)	
Secondary Processing: Pulp Plant	12 (6)	0 (0)	
Secondary Processing: Paper Plant	13 (6)	0 (0)	
Secondary Processing: Pellet Plant	0 (0)	0 (0)	
Distribution: Lumber	10 (5)		
Distribution: Pulp & Paper	5 (2)		
Distribution: Pellets	6 (3)		
Use & Disposal: Lumber (Year 0-100)	62 (28)	0 (0)	-251 (83)
Use & Disposal: Paper (Year 0-100)	82 (37)	0 (0)	-52 (17)
Use & Disposal: Pellet (Year 0-100)	1 (1)	0 (0)	
Subtotal	221(100)	0 (0)	-303 (100)
		NET C BALANCE	**-82**

b.) Not C Neutral

	Non-Biogenic GHG Emissions kg CO_2e m^{-3} (%)	Biogenic GHG Emissions kg CO_2e m^{-3}(%)	Biogenic C Storage kg CO_2e m^{-3} (%)
Harvest	10(4)		
Transportation	11(5)		
Primary Processing: Sawmill	8(4)	27(6)	
Secondary Processing: Pulp Plant	12(6)	210(49)	
Secondary Processing: Paper Plant	13(6)	39(9)	
Secondary Processing: Pellet Plant	0(0)	4(1)	
Distribution: Lumber	10(5)		
Distribution: Pulp & Paper	5(2)		
Distribution: Pellets	6(3)		
Use & Disposal: Lumber (Year 0-100)	62(28)	43(10)	-251(83)
Use & Disposal: Paper (Year 0-100)	82(37)	57(13)	-52(17)
Use & Disposal: Pellet (Year 0-100)	1(1)	52(12)	
Subtotal	221 (100)	433 (100)	-303 (100)
		NET C BALANCE	**350**

3.0.2.1.1. Sensitivity Analyses on Lumber's Case Studies

If I accepted the assumption that biogenic GHG emissions are C neutral (case 1), the most sensitive parameter in my C budget model was the biogenic C storage factor applied to lumber in a landfill (Figure 3.10.). This was closely followed by the lumber recovery factor. Other sensitive parameters in my model were: a.) material flow's pulp chip recovery and chemical pulp recovery; b.) biogenic C storage's lumber use, lumber disposal and paper disposal; and d.) non-biogenic GHG factors paper disposal and lumber disposal (Figure 3.10.). The C budget was not, however, sensitive to changes in my biogenic GHG emission factors nor was it sensitive to small changes in the non-biogenic GHG emissions factors of life stages (excluding use and disposal life stages).

If I rejected the assumption that biogenic GHG emissions are C neutral (case 2), the most sensitive parameter was pulp chip recovery (Figure 3.10.). Other sensitive parameters included: a.) material flow's lumber recovery and chemical pulp recovery; b.) biogenic C storage's lumber use, lumber disposal, and paper use; c.) biogenic GHG emission factor's pulp plant; and d.) non-biogenic GHG emission factor's lumber disposal and paper disposal (Figure 3.10.). My C budget was sensitive to changes in the biogenic GHG emission factors but it was not sensitive to small changes (i.e. 10%) in the non-biogenic GHG emission factors (excluding use and disposal life stages).

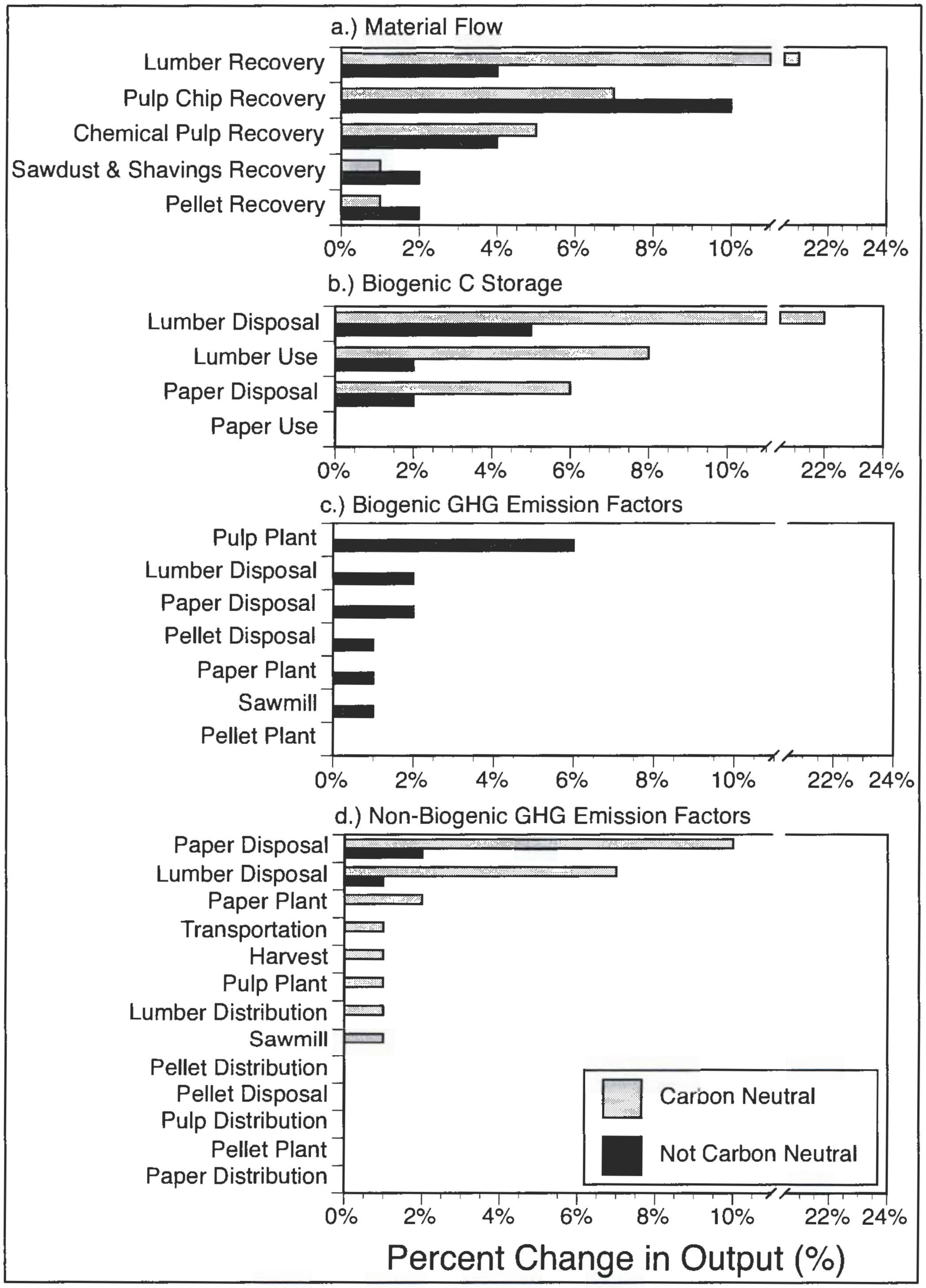

Figure 3.10. Results from the sensitivity analyses run on the parameters of my C budget model for lumber production. Individual parameters were increased by 10% and the percentage change reflects the percent change in the C budget's output.

3.0.2.2. Bioenergy: Pellets

If I accepted the assumption that biogenic GHG emissions are C neutral (Case 1), roundwood harvested for pellets emitted very little GHG emissions (99 kg CO_2e m^{-3}; Table 3.2a.). In this context, the largest amount of GHG emissions are emitted during the distribution of pellets to European markets (64% of total).

If I rejected the assumption that biogenic GHG emissions are C neutral (Case 2), roundwood harvested for pellets emitted large amounts of GHG emissions (688 kg CO_2e m^{-3}; Table 3.2b.). In this context, the largest GHG emissions are the biogenic GHG emissions emitted during the use and disposal of pellets (72% of total).

Table 3.2. The net C balance of pellet production a.) assuming biogenic GHG emissions are C neutral and b.) assuming biogenic GHG emissions are not C neutral.

a.) C Neutral

	Non-Biogenic GHG Emissions kg CO_2e m^{-3} (%)	Biogenic GHG Emissions kg CO_2e m^{-3} (%)
Harvest	10 (10)	
Transportation	11 (11)	
Pellet Plant	2 (2)	0*
Pellet Distribution	64 (64)	
Pellet Use & Disposal	13 (13)	0*
Subtotal	99 (100)	0
NET C BALANCE		99

b.) Not C Neutral

	Non-Biogenic GHG Emissions kg CO_2e m^{-3} (%)	Biogenic GHG Emissions kg CO_2e m^{-3} (%)
Harvest	10 (10)	
Transportation	11 (11)	
Pellet Plant	2 (2)	96 (16)
Pellet Distribution	64 (64)	
Pellet Use & Disposal	13 (13)	493 (84)
Subtotal	99 (100)	589 (100)
NET C BALANCE		688

3.0.2.2.1. Sensitivity Analyses on Pellets' Case Studies

In the absence of biogenic GHG emissions (case 1), the most sensitive parameter in my C budget model is pellet recovery (Figure 3.11.). This was followed closely by the non-biogenic GHG emission factor for the distribution of pellets. My C budget model was not sensitive to any changes in the biogenic GHG emission factors (Figure 3.11b.) nor was it really affected by small changes to the non-biogenic GHG emission factors (excluding distribution) (Figure 3.11c.).

In the presence of biogenic GHG emissions (case 2), the most sensitive parameter in my forest product model is also pellet recovery (Figure 3.11a). The runner-up was the biogenic GHG emission factor applied to use and disposal of pellets (Figure 3.11b.). Consequently, my C budget model was sensitive to changes in the biogenic GHG emissions factor but it was not sensitive to any small changes in the non-biogenic GHG emission factors (Figure 3.11c.).

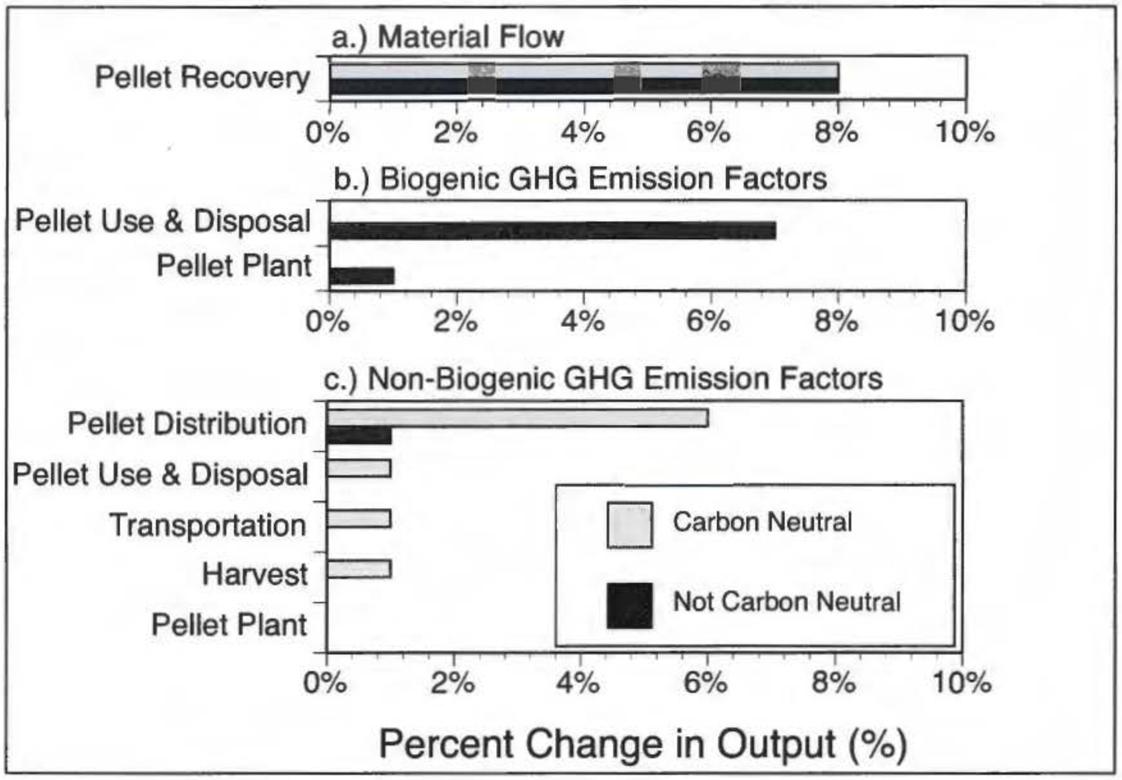

Figure 3.11. Results from the sensitivity analyses run on the parameters of my C budget model for pellet production (scenario 2). Individual parameters were increased by 10% and the percentage change reflects the percent change in the C model's output.

3.1. DISCUSSION

3.1.1. C Budget of Forest Products Harvested from MPB-Attacked Forests

The C budgets of my chosen forest products (lumber, pellets) had very different C balances. Their budgets differed with respect to their industrial (i.e. forest product) system, as well as their sensitivity to the state of the biological (i.e. forest) system. Roundwood harvested for lumber (scenario 1) had net C storage (-82 kg CO_2e m^{-3}) and net C emissions (350 kg CO_2e m^{-3}) with and without the C-neutral assumption, respectively. Meanwhile, roundwood harvested for pellets (scenario 2) had net C emissions (99 kg CO_2e m^{-3}, 688 kg CO_2e m^{-3}) regardless of the C-neutral assumption. These different C balances demonstrate the relative variability between two forest products in an industrial forest product C budget. Taken in this context, lumber was seen as contributing less GHG emissions to the atmosphere than pellets.

My findings on lumber and pellet C budgets support previous studies examining these forest products in British Columbia (ASMI 2012, Dymond 2012, Pa et al. 2012, Lamers et al. 2014). However, in our studies, lumber production was found to have a lower lumber recovery (due to the MPB attack) and higher GHG emissions attributed to its mill residues than previous studies (ASMI 2012, Dymond 2012, Lamers et al. 2014). Pellet production on the other hand, differed with respect to previous studies by including a biogenic GHG emission factor for pellet combustion (Pa et al. 2012). Finally, we excluded substitution related emissions reductions for both lumber and pellets.

The C budget of lumber production had large and opposing C fluxes of biological C storage and GHG emissions. Its biogenic C storage helped offset GHG emissions regardless of the C neutral assumption. In contrast, the C budget of pellet production did not have any biogenic C storage to counteract its GHG emissions. Moreover, its non-biogenic GHG emissions were relatively low in comparison to its biogenic GHG emissions. Its C budget was thus susceptible to assumptions regarding the C neutrality. Pellet's C budgets could emit very little or very large amounts of biogenic GHG emissions based simply on the C-neutrality of its biogenic GHG emissions.

That said, total GHG emissions (i.e. biogenic and non-biogenic) were comparable between lumber and pellet production. The mill residues in lumber production formed the majority of the roundwood harvested from forests and they were large biogenic GHG emitters. As a result, lumber production was not impervious to assumptions regarding C neutrality altogether.

Lumber and pellet production differed with respect to the timing of their C fluxes. Lumber's C storage and GHG emissions (from disposal) are protracted. Its budget begins as a strong C sink (i.e. net C storage), but it ends as a slight C source (i.e. net C emissions). Pellets' GHG emissions on the other hand, are immediate. Its C budget begins as a strong C source from the start. This observation echoes a concern held by many stakeholders in allowing whole-tree pellet harvesting (i.e. forest harvesting for bioenergy). The concern is that forest bioenergy releases more biogenic GHG emissions than forests are capable of sequestering via photosynthesis. In essence, these concerns of "slow in fast out" ultimately question the validity of any C-neutrality assumptions.

In closing, my forest C budgets were compared based on an experimental unit of 1 m^3 merchantable roundwood. This was chosen based on my desire to demonstrate the differences between lumber and pellet industrial systems. One logical alternative to my experimental unit is 1 m^3 timber. This could have examined the differing capabilities of lumber and pellet products in utilizing MPB-attacked stands and timber that were deemed "non-merchantable". Lumber is unable to use non-merchantable timber and it would otherwise decay in the forest, whereas pellets could utilize this timber for bioenergy. The problem with this experimental unit is that it requires a forest C modeling component and there is some uncertainty in the snagfall and decay rates of MPB-attacked forests.

3.1.1.1. Direct Influences of the Industrial System on Forest Product C Budgets

The industrial system for lumber production was complex and extensive. It had difficult geographical, temporal, and physical aspects to account for fully in a C budget. Consequently, my study focused on establishing actual material and product flows, biogenic C storage and GHG emission factors for each of the life stages of lumber and its by-products. The precedent for forest product C accounting in BC was set by two contemporary studies (Dymond 2012, Lamers et al. 2014). Dymond (2012) focused the biogenic C storage of BC forest products, while Lamers et al. (2014) focused on the possible avoided emissions in harvesting MPB-attacked forests and in substituting forest products.

In contrast, the industrial system for pellet production was simplistic and well documented. It had fairly straightforward geographical, temporal and physical aspects that could easily be accounted for in a C budget. My study focused on establishing an emission factor for whole-tree pellet harvesting of MPB-attacked forests. To date, much of the knowledge on pellets is based on those pellet industries sourcing sawdust and shavings. Lamers et al. (2014) used an emission factor for whole-tree pellet processing in BC, but this was taken from a confidential source and some of their parameters are unclear.

3.1.1.1.1. Material & Product Flow

In lumber production, the material and product flows of its forest products were important parameters governing lumber's overall C balance. The allocation of lumber and its mill residues is fairly well represented in the literature (MFLNRO 2015c), whereas their composition and/or recovery are not. This is somewhat problematic given the sensitivity of my C model to lumber and pulp chip recovery. The chosen lumber recovery factor for MPB-attacked roundwood in my study was much lower than the recovery used in Lamers et al. (2014), and slightly lower than the average recovery in the BC Interior (MFLNRO 2015c). Interestingly, lumber recovery became less important when C neutrality was not assumed. Up until this point, much of the narrative in MPB's salvage logging campaign has been focused on the lumber recovery factor and its ability to store biogenic C. My study expands this narrative to begin discussions surrounding the potential ramifications of higher mill residue recoveries and variation in mill residue composition, without the assurance that C neutrality is being met.

In pellet production, the material and product flow was simple but pivotal in determining its C balance. The pellet recovery was the most sensitive parameter in my C budget regardless of the biological context (i.e. C neutral vs. not C neutral). The pellet recovery factor used in my study was consistent with the recovery cited in whole-tree pellet plants in the US Midwest (Katers et al. 2012) and lower than the pellet recoveries cited in pellet plants sourcing sawdust and shavings in the BC Interior (Pa et al. 2012, Sikkema et al. 2013).

3.1.1.1.2. Biogenic C Storage

Biogenic C storage was a large C flux unique to lumber production. Both lumber and paper products in this industrial system contributed to its storage. These products stored C while they were in use and while they were in disposal. In the end, it was found that only 9% of the original roundwood harvested remains in use. The bulk of biogenic C storage lay in disposal C pools.

Biogenic C storage was found to play an important role in offsetting total GHG emissions. My C budget was sensitive to the biogenic C storage factors for lumber and paper, in use and in disposal. This finding is consistent with that reported in the literature -- that a forest product's use and disposal is the most important and uncertain life stage in an industrial forest product system (Miner and Perez-Garcia 2007, Heath et al. 2010). My study borrowed Dymond's (2012) parameters for biogenic C storage of BC forest products in North America. This study adapted the standard use and disposal parameters of North American forest products to those products harvested from BC forests (Smith et al. 2006, MOE 2011, Dymond 2012). While many of these parameters are the standard, they are by no means faultless or certain. Many of the parameters in use and disposal are the subject ongoing areas of research (Marland et al. 2010).

3.1.1.1.3. GHG Emission Profile

3.1.1.1.3.1. Non-Biogenic GHG Emissions

The non-biogenic GHG emissions in lumber production were relatively small and uncertain in comparison with its other C fluxes (i.e. biogenic C storage and GHG emissions). Yet these non-biogenic GHG emissions are juxtaposed here with the large GHG savings and simplifying assumptions that occur when substitution emission factors are used in other studies in place of direct, non-biogenic GHG emissions (e.g. Lamers et al. 2014). My study found the largest amount of non-biogenic GHG emissions occurred in the disposal of forest products. These emissions are included in Sathre and O'Connor's (2010) substitution emission factor, but there exists considerable variation in the disposal parameters among the studies used in their meta-analysis.

Meanwhile, the non-biogenic GHG emissions in pellet production were relatively small in comparison with its biogenic GHG emissions. Contrary to lumber production, the non-biogenic emissions in pellet production are fairly well established in the literature. My study found the distribution of pellets to be the largest source of non-biogenic GHG emissions and this was found to be a consistent finding among similar studies (Magelli et al. 2009, Sikkema et al. 2010, Pa et al. 2012).

3.1.1.1.3.2. Biogenic GHG Emissions

The biogenic GHG emissions in lumber and paper C budgets are large in comparison to its other C fluxes (i.e. biogenic C storage and non-biogenic GHG emissions). Yet little attention has been brought to the biogenic GHG emissions that occur at each of the life stages of a forest product's industrial system. They are typically excluded from this system based on the presumption that it is the responsibility of the biological (i.e. forest) system to account for forest product biogenic GHG emissions.

In lumber's C budget, it was found that the biogenic GHG emissions occurred at various life stages in the short term and in the long term. In the short term, the largest source of biogenic GHG emissions was in the secondary pulp processing of pulp chips at chemical pulp plants. In the long term, the largest biogenic GHG emission was in the use and disposal of forest products. In contrast, Lamers et al. (2014) did not examine biogenic GHG emissions directly. Instead, the authors accounted for short-lived forest products (i.e. pulp chips, hogfuel, sawdust and shavings) as immediate biogenic GHG emissions. Long-lived forest products accrue biogenic GHG emissions over time, as products are no longer being stored in use.

In the pellets C budget, biogenic GHG emissions were considerably larger than its non-biogenic GHG emissions. The largest source of biogenic GHG emissions was in its use and disposal. While these findings may seem apparent, most studies treat pellet emissions as C neutral or as if the C is immediately emitted upon harvest. Lamers et al. (2014) did the latter and treated roundwood harvested from MPB-attacked forests for pellets as an instantaneous loss to the system.

3.1.1.2. Indirect Influences of the Biological System on Forest Product C Budgets

The treatment of the biological system in forest product C accounting is problematic given its role in determining whether or not biogenic GHG emissions are included in the industrial system (Seachinger et al. 2009, Helin et al. 2013). This C accounting dilemma is best exemplified by the concept of C neutrality. The term, C neutrality, covers a suite of differing assumptions with regards to the accounting of biogenic GHG emissions in industrial forest product systems. These C-neutral definitions are discussed here as: C-cycle neutrality, C-accounting neutrality, and inherent C neutrality (Malmsheimer et al. 2011).

Carbon-cycle neutrality is “if (the) uptake of C (in CO_2) by plants over a given area and time is equal to emissions of biogenic C attributable to that area, (the) biomass removed from that area is (considered) C-cycle neutral” (Malmsheimer et al. 2011). It is typically predicated on the concept of sustainable forest management (SFM) as a means of bypassing any forest C modeling. The designation of SFM is thought to be legitimatized through its SFM certification process. This negates the fact, however, that SFM certification programs have failed to incorporate global C cycles and climate change as part of their criteria and indicators for SFM designation (Tittler et al. 2001, Lloyd et al. 2014). Moreover, few SFM certification processes can convincingly ensure SFM in BC given the recent MPB outbreak and upcoming mid-term timber supply gap. In fact, many of the SFM certification companies in MPB-attacked forests defer their definition of "sustainability" to that defined by BC's Chief Forester (Lloyd et al. 2014).

Carbon accounting neutrality, at least from a biogenic perspective, is met "if (the) emissions of biogenic CO_2 are assigned an emissions factor of zero because net emissions of biogenic C are determined by calculating changes in stocks of stored C" (Malmsheimer et al. 2011). Conventionally, nations did not have to report biogenic GHG emissions in their industrial sectors, provided that they reported these emissions as part of the forest's GHG emissions. This is problematic for the provincial government, as it does not currently report forest GHG emissions in its GHG inventory. This lack of reporting forfeits its rights to claim biogenic GHG emissions as C neutral, under a C accounting neutrality definition (Malmsheimer et al. 2011). By not reporting forest GHG emissions, these biogenic GHG emissions would then need to be included in their respective industrial sectors. [Instead, the Province considers biogenic GHG emissions as C neutral under inherent C neutral and/or C-cycle neutral definitions, and excludes them from being reported in its GHG inventory.]

Inherent C neutrality assumes that "biomass was only recently removed from the atmosphere; (and that by) returning it to the atmosphere (it) merely closes the cycle" (Malmsheimer et al. 2011). It is a simple and obvious definition that blindly promotes the renewable nature of biological C (i.e. bioenergy) over the non-renewable nature of fossil fuel C (i.e. natural gas, coal). Its limitation is that it disregards any incremental steps in forestry's climate change mitigation efforts, and instead promotes broad and abstract goals of reducing global fossil fuel consumption. While the latter is the ultimate goal of climate change mitigation, it may in fact be misguided given the lack of coverage of the biological system. Biomass is a low-energy fuel and releases more GHG emissions per unit of energy harnessed than most fossil fuels in the short-term. It remains unclear what the immediate ramifications will be of increasing the atmospheric CO_2 concentration by promoting large quantities of low-energy biomass in exchange for fossil fuels.

In summary, my study found a lack of convincing evidence to suggest that forest product biogenic GHG emissions are being managed appropriately in British Columbia. This is troublesome given the importance of biogenic GHG emissions on the outcome of forest product C budgets, and the broader implications of promoting the role of forests, forestry and forest products in climate change mitigation efforts.

3.1.2. Providing Context for Environmental Science and Policy

The intent of this manuscript was meant to equitably inform environmental science and policy on the decision of whether or not to harvest MPB-attacked stands for forest products, and particularly, bioenergy products. As previously mentioned, my chosen forest products (lumber, pellets) involved very different industrial processes and systems. Depending on the C accounting approach and its conventions, the C budget for these forest products was found to vary widely in the literature. Moreover, additional discrepancies were found in their definitions (and/or treatment) of the biological system. My study deconstructed many of the established conventions for this reason, which meant reexamining their scope and their selection of cut-off criteria and assumptions. My efforts here strive to provide clarity for how different C accounting contexts and conventions differ from one another with respect to their perspectives in harvesting MPB-attacked forests for forest products.

3.1.2.1. National GHG Inventories

British Columbia currently reports its annual GHG emissions in a provincial GHG inventory report (MOE 2012). The intent of this inventory is to quantify, monitor, and inform others of the Province's commitments to addressing climate change. While this inventory allows BC to clearly communicate its GHG emissions to a national and international audience, the Province's current C accounting framework and reporting on forest (and forest product) GHG emissions is not conducive to activity-related climate change mitigation efforts.

The Province does not currently account for forest GHG emissions in its GHG inventory report, so any forest- (or forest product)- related mitigation activity results in no overall reductions in provincial GHG emissions. Dymond (2012) found that the Province could reduce forest GHG emissions by claiming the biogenic C storage of its forest products, but this would require the Province to begin accounting forest GHG emissions in its GHG totals as well. British Columbia's forests are currently estimated as a large C source and outweigh Dymond's (2012) expected benefits, so it is unlikely these will be pursued anytime in the near future.

That said, recent studies by Lemprière et al. (2013) and Smyth et al. (2014) discuss the possibility of including a "systems perspective" to bolster Canada's forest products industry and its efforts in climate change mitigation. This "systems perspective" is reminiscent of substitution emission factors and the faults therein. It remains unclear how these researchers plan on incorporating these derived benefits into current national and provincial GHG inventories.

3.1.2.2. Carbon Footprint and LCA Studies

Carbon footprint and LCA studies provide the most standardized and detailed account of the environmental impacts specific to a forest product industry and its products. The advantage of this framework is that it provides tangible environmental impact assessments for the industry and for forest product consumers. Moreover, they lend themselves to consequential LCA studies (i.e. substitution emissions factors), which are very popular with tall wooden building and bioenergy advocates because they have shown significant reductions in global GHG emissions (Sathre and O'Connor 2010).

The limitation of C footprint and LCA studies is that they rely on simplifying assumptions with regards to their C accounting of industrial and biological systems. Historically, these studies have been able to exclude the life cycle of any by-products, under the assumption that they are included in separate LCAs (where they form the material input for another industrial system). This is currently not the case. In my study, for example, I found a lack of LCAs specific to the pulp and paper industry. This is concerning given the fact that secondary processing of residues is among the largest sources of biogenic and non-biogenic GHG emissions. Secondly, C footprint and LCA studies rely on the assumption that the biological system (i.e. biogenic GHG emissions) is C-cycle neutral. This is an assumption that lacks support and is likely disputable considering the unprecedented impact of the MPB outbreak on the sustainability of forestry in the province.

3.1.2.3 Forest-Forest Product C Models

Forest-forest product C models are great analytical tools used to explore the various aspects of forest and forest product systems. They provide insight into current and alternative ways of managing and manipulating these systems over space and time, without always performing physical measurements and/or treatments. Traditionally, these top-down modeling efforts reflected physical C fluxes that could then be followed up through validation with actual measurements. In recent years however, these modeling efforts have begun incorporating indirect (or hypothetical) C fluxes that cannot be validated directly.

Some of the concerns and limitations in modeling forest and forest product systems are its lack of standardization, transparency, validity and accountability. Without standardization, the C accounting can be a fairly subjective procedure. This is problematic considering the different C accounting contexts that are possible and the different narratives that can be communicated. Without transparency, the goals and intent of these models are obscured and their communication is likely to transcend their own context. Lastly, without validity and accountability, the models are capable of making assumptions that may not in fact be representative of the actual forest and forest product systems. Instead, many of these C models keep expanding their forest-forest product systems to include more consequential impacts associated with the forest product industry (e.g. Ter-Mikaelian et al. 2015).

3.1.3. Importance of Forest Products in Forestry's Efforts to Mitigate Climate Change

The contribution of forest products to the broader impacts of forestry remains ambiguous in British Columbia. As previously discussed, there are many different industrial and biological contexts for forest product C accounting. These different contexts add to the illusion of importance when it comes to forestry's efforts in mitigating climate change. My discussions of forest products so far have been largely confined to the direct impacts influencing the forest products industrial system. The focus of the discussion of these forest products will now shift and include the possible indirect impacts that influence the industrial system.

3.1.3.1. Indirect Influences of the Industrial System on Forest Product C Budgets

One of the largest indirect influences on the industrial forest system is the avoided emissions in substituting (or displacing) other industrial systems. Taken at face value, the substitution emission factors are capable of making substantial reductions in global GHG emissions (Sathre and O'Connor 2010). As a result, they have been heralded as the frontrunner in bolstering forestry's efforts and ambitions in mitigating climate change (Schlamadinger and Marland 1996, Lippke et al. 2011, Lamers et al. 2014, Smyth et al. 2014).

However, Sathre and O'Connor (2010) discuss the possibility that product-based substitution emission factors may not necessarily inform the most efficient use of biomass from a climate change mitigation perspective. They offer a singular perspective of a product's ability to reduce global GHG emissions. The most commonly used substitution emission factor for long-lived wood products (i.e. lumber) is currently borrowed from a meta-analysis performed by Sathre and O'Connor (2010). My problems with their meta-analysis are its selection bias, and the wide range of forest products and uses, country-specific data, and C accounting methodology and assumptions. For these reasons, my study chose not to blindly use substitution emission factors and sought to first establish a baseline for the BC forest product industry. Moreover, Lamers et al. (2014)

already used Sathre and O'Connor's (2010) substitution emission factor in their analysis and established a range of possible scenarios in harvesting MPB-attacked forests for forest products.

The biological system (and its biogenic GHG emissions) is not accounted for in substitution emission factors (Sathre and O'Connor 2010). The concern has been that by segregating the industrial systems from the biological system, forestry runs the potential risk of inadvertently deteriorating the biological system to a point at which forestry is no longer sustainable and/or C neutral. This concern is most prevalent and contentious in discussions surrounding the C-neutrality of forest bioenergy (McKechnie et al. 2011, Holtsmark 2012, Mitchell et al. 2012, Schulze et al. 2012).

Recent efforts to validate the C-neutral assumption in forest bioenergy have consequently included industrial systems' avoided emissions in the modeling of the biological system. This marks a radical shift in the way C neutrality is discussed and defined in the literature. These new and emerging C-neutral concepts include: "C offset point", "break-even period", "C parity period", "time to C neutrality", and "C sequestration parity" (Ter-Mikaelian et al. 2015). These concepts differ from traditional definitions of C neutrality in that they incorporate various avoided emissions in their C-neutrality validation process. They differ from one another in that they have different break-even points and different degrees of avoided emissions. The result of these differences has broadened our understanding of possible outcomes of harvesting forests for bioenergy, but it has also created confusion in the word "carbon neutral".

In the midst of these shifting perspectives on C neutrality and forest bioenergy, my thesis asks the general question of whether or not harvesting MPB-attacked forests for different forest products is beneficial or detrimental to forest GHG emissions. This remains a difficult question to answer conclusively due to the lack of environmental science research on the direct and indirect impacts of industrial and biological systems. Lamers et al. (2014) did examine the direct and indirect influences of industrial and biological systems in MPB-attacked forests and they found that lumber and pellet products could reach C sequestration parity in heavily damaged forests when including the avoided emissions of forests and forest products. However, the validity of the avoided emissions used in this study remains unclear.

3.1.4. Role of Forests in Climate Change Mitigation Efforts

It is widely recognized that forests can help mitigate the effects of climate change. As stated in early versions of the IPCC: "in the long term, a sustainable forest management strategy aimed at maintaining or increasing forest C stocks, while producing an annual sustained yield of timber, fibre, or energy from the forest, will generate the largest sustained mitigation benefit" (Nabuurs et al. 2007). Yet for a province currently in the midst of dealing with the unprecedented impacts of an MPB outbreak, it has been difficult to establish a credible sustainable forest management strategy incorporating global C cycles and climate change mitigation aspects in a timely manner.

The environmental science research supporting the role of BC forests in mitigating climate change is unclear. Kurz et al. (2008) and Metsaranta et al. (2011) provide the cornerstones of BC's understanding of the regional- and landscape-level C impacts of the MPB outbreak. Yet these studies are currently at odds with stand-level studies by Brown et al. (2010) and do not address the appropriateness of timber harvesting during BC's salvaging logging efforts, directly. Instead, they demonstrate the impacts of additional harvests indirectly, as increased C emissions following increasing levels of harvesting in the allowable annual cut (AAC).

An environmental policy supporting the role of BC forests in mitigating climate change is not present. The Province has yet to establish any environmental policies focused directly at forestry's role in mitigating climate change. This is particularly glaring given the fact that the Province does not currently include BC's managed forests in its provincial GHG inventory report.

In closing, it is my belief that if the provincial government is truly serious about curtailing global GHG emissions and mitigating climate change, it will need to begin reporting forest GHG emissions, in addition to many other industrial sectors currently not included in the Province's GHG inventory report.

Chapter 4.

CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

4.0. CONCLUSIONS

In the post-epidemic phase of the mountain pine beetle (MPB) outbreak, there still remains a degree of uncertainty in quantifying carbon (C) budgets for industrial forest product systems. Consequently, there is also uncertainty in promoting the broader notion of any mitigation benefits by including global carbon (C) cycles and climate change objectives into forest management planning and policy in British Columbia (BC), Canada.

In my study, I looked at the C budgets of forest products harvested from MPB-attacked forests and found lumber contributed lower GHG emissions than pellets (in an industrial system). While seemingly modest, my findings contribute unique insights into the role forest products may play in forestry's climate change mitigation efforts. For one, the Province does not account for forest nor forest product GHG emissions in its provincial GHG inventory. Secondly, it lacks support in its claims that biogenic GHG emissions in the forest products industry are C neutral. Lastly and most importantly, any discussion surrounding avoided emissions (or substitution emission factors) of MPB-attacked forests and forest products is conjecture. Environmental science research is ongoing in its quantification of the MPB's impact on forests and the forest products industry.

4.1. RECOMMENDATIONS FOR FUTURE RESEARCH

My recommendations for future research begin with the targeting of specific parameters that were found to be problematic in synthesizing a forest product C budget for MPB-attacked forests, and it ends with broader and more ambitious research that needs to be performed before these forests are managed for global C and climate change mitigation efforts.

4.1.1. Direct Influences of the Industrial System on Forest Product C Budgets

One of the major limitations in my approach was the limited data available in constructing forest industry C budgets in BC. In particular, C budgets for dimensional lumber had many unknowns and uncertainties at each of its life stages. The C budget for pellets (bioenergy), on the other hand, had fewer unknowns and uncertainties, with the exception being the lack of data on whole-tree pellet processing. Future research needs to perform an environmental LCA (i.e. similar to White et al. 2005) on roundwood utilized for lumber versus pellets. To improve these LCAs, greater access to forest industry parameters is needed. In dimensional lumber C budgets there remain some uncertainties in the product recoveries at sawmills processing MPB-affected roundwood. While the attention has been on lumber recovery, mill residue recovery and composition were found to be of some significance. Future research needs to establish the current lumber recovery level of MPB-affected roundwood, and to more fully document the subsequent composition and use of mill residues.

Modeling parameters governing forest product use and disposal were important in deciding the amount of biogenic C stored. My C budget borrowed use and disposal parameters from Dymond (2012). Future research could examine specific aspects of these parameters, such as: the allocation of lumber use and its half-lives, mathematical functions representing the disposal of lumber by type and use (e.g. first-order decay versus distributed approach) (Marland et al. 2010), and method of disposal (e.g. open dumps, landfills, combustion).

I found that the greatest uncertainties in GHG emissions from lumber lies in the secondary processing of its chip byproducts (e.g. chemical pulp plant, paper plant) and in its disposal stages (e.g. lumber and paper decay). Future research should examine emissions data for pulp and paper manufacturing and the rates and efficiency of methane capture and efficiency in landfills.

The timing of GHG emissions between forest products differed. Lumber emitted emissions over time, whereas pellet emissions were generated immediately. Levasseur et al. (2010) discuss the possibility of current and future atmospheric CO_2 emissions (i.e. cumulative radiative forcings) changing how we view the impacts of the timing of GHG emissions on the atmosphere.

4.1.2. Indirect Influences of the Industrial System on Forest Product C Budgets

Substitution emission factors allegedly offset large amounts of GHG emissions from other industrial sectors. These emissions factors are not typically screened using C offsetting criteria (MOE 2011). Future research could examine the actual GHG emission offset by the construction sector using C offsetting criteria, such as additionality (e.g. ability to prove an action is more than business as usual) and leakage (e.g. inability to prove an action actually curtailed fossil fuel dependency).

Similarly, more precise end-use modeling of forest products could yield substantial GHG emission savings. This could include options for their re-use or emissions-avoiding options (e.g. bioenergy) (Sikkema et al. 2013).

4.1.3. Indirect Influences of the Biological System on Forest Product C Budgets

Many forest C budgets from the scientific literature were found to use C-neutral assumptions to exclude the biogenic GHG emissions from their reporting. My study found that there is a lack of support for many of the definitions currently in use. Specifically, C-cycle neutrality was found to be questionable given the uncertainties of sustainable forest management (e.g. growing inventory of not satisfactory restocked (NSR) forest lands, uncertainty about future growth and yield of plantations in the face of climate change, midterm timber supply gaps). Meanwhile, inherent C neutrality lacked any scientific basis and C accounting neutrality is not possible given the Province's policy on forest GHG emissions. Future research could look into stakeholders' perceptions of C neutrality and gauge their support for, or lack thereof, in the Province adopting C-accounting neutrality as its default assumption.

4.1.4. Role of Forest Products in Mitigating GHG Emissions

The logical next step in broadening my research would be to validate the C-cycle neutral assumption. Validation of this assumption would require the integration of the direct C fluxes associated with forest and forest product systems when harvesting MPB-attacked forests (e.g. actual C balances of harvested stands using eddy covariance and process-based ecosystem models such as 3PG) (Landsberg and Waring 1997). The outcome of this research would shed light on whether or not biogenic GHG emissions are fully accounted for by the forest. Moreover, the findings from this research could be used to justify whether or not MPB-attacked forests should be harvested for forest products, and in particular, whether forest bioenergy should be promoted in BC's salvage logging campaign and climate change mitigation efforts.

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Appendix I. GHG emission factors for non-biogenic materials used in this study.

	MOE 2012
BC electricity (kg CO_2e kWh^{-1})	0.0091
heavy fuel oil (kg CO_2e L^{-1})	3.146
diesel (kg CO_2e L^{-1})	3.007
gasoline (kg CO_2e L^{-1})	2.299
propane (kg CO_2e L^{-1})	1.529
natural gas (kg CO_2e m^{-3})	1.927

Appendix II. C storage and GHG emission factors for biogenic material and products used in this study

	Dymond 2012	MOE 2012	Pa et al. 2013
product storage	1.833*	n/a	n/a
wood waste (15% MC)	n/a	1.621^	n/a
pellets	n/a	n/a	1.731`

* C content: 0.5 tC t^{-1} wood.

^ C content: 0.434 tC t^{-1} wood.

` density: 840 kg m^{-3} @ 50%MC_{OD}

Appendix III. Conversion factors used in this study.

Conversion Factors	Description
0.409 ODT m^{-3}	basic density of lodgepole pine (*Pinus contorta* var. *latifolia*) wood (Nielson et al. 1985)
1.594 mbfm m^{-3}	thousand board feet per merchantable cubic meter of roundwood (Nielson et al. 1985, ASMI 2012)
11.4%	volumetric shrinkage of lodgepole pine wood from green to dry (30-0% MC_{OD}) (Nielson et al. 1985).