

# **The potential of the aviation sector to reduce greenhouse gas emissions by using biojet fuels**

by

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## ABSTRACT

Aviation is the fastest growing industry in the transport sector and its GHG emissions are expected to increase 7-fold over the next 35 years. To achieve the industry's goals of a 50% emission reduction by 2050, groups such as the International Civil Aviation Organisation (ICAO) have stated that biofuels will play an essential role. Although various methods of producing biojet fuel have been proposed, the specific GHG emission reductions that might be achieved have yet to be fully elucidated. This thesis explores the Lifecycle Assessment (LCA) of biojet fuel production via thermochemical and oleo-chemical means through the review of biojet LCA literature. By comparing the assumptions used within, it became apparent that the nature of the LCA model had a significant impact on the carbon intensity results. Results using the GHGenius model were found to be significantly different than results from GREET or SimaPro, likely due to the inclusion of land use change and the use of the displacement allocation method in the GHGenius model. Although these two variables influenced the results more than any other variable, the location of production also had a significant impact on the oleo-chemical and pyrolysis methods, as did the source of hydrogen. Even with these differences, all models agreed that biojet fuel produced by gasification provided the lowest greenhouse gas emissions.

In the second part of this thesis, the LCA of B.C. forest biomass-to-biojet pyrolysis scenarios was modeled, assessing three possible biomass supply chains: (Vancouver Mainland (forest residue), Vancouver Island (forest residue) and Prince George (wood pellets)). The GHG emission reductions of each supply chain scenario compared to petroleum jet fuel were 71.1%, 70.6%, and 68.2%, respectively. A sensitivity analysis of the Prince George scenario indicated that the results were most sensitive to the type of feedstock used for pellet production, the allocation method used, the moisture content of the feedstock and the source of hydrogen. It was shown that, independently, these variables can change the GHG emission results by 10% - 60% or, combined, could reduce the overall GHG emissions to - 22.13 gCO<sub>2</sub>eq/MJ biojet fuel (125% reduction).

## LAY SUMMARY

The International Civil Aviation Organization has set a target to reduce greenhouse gas emissions by 50% from 2005 levels by 2050. To reach this goal, biofuels will be required. This thesis reviews the current body of literature on biojet fuels to determine which production methods have the greatest potential to reduce greenhouse gas emissions compared to petroleum jet fuel. This thesis found that the differences between some studies are significant, making it difficult to compare the results from these studies, but that biojet via gasification tends to provide the greatest reductions. In the second part of the thesis, the greenhouse gas emission reductions from a potential biojet fuel refinery in British Columbia was found to be 68% - 71% lower than petroleum jet fuel when produced via pyrolysis from forest residuals and wood pellets. Under optimal conditions, this reduction increased to around 125%.

## PREFACE

This thesis is the original, unpublished, and independent work of the author, A. Ringsred Beavers.

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## LIST OF ABBREVIATIONS

ASTM: American Society of Testing and Materials  
ATJ: alcohol-to-jet  
CS: corn stover  
DSHC: direct-sugar-to-hydrocarbon  
EA: energy allocation  
EPA: Environmental Protection Agency (U.S.)  
FU: functional unit  
FR: forest residue  
FT: Fischer-Tropsch  
GHG: greenhouse gas  
GWP: global warming potential  
GREET: 'Greenhouse gases, Regulated Emissions, and Energy use in Transportation' model  
HTL: hydrothermal liquefaction  
HEFA: hydrogenated esters and fatty acids  
HRJ: hydro-treated renewable jet  
ICAO: International Civil Aviation Organization  
IPCC: International Panel on Climate Change  
iLUC: indirect land use change  
ISO: International Standards Organization  
IEA: International Energy Agency  
LCA: life-cycle assessment or analysis  
LUC: land use change  
LCFS: low carbon fuel standard  
MA: mass allocation  
MSW: municipal solid waste  
MLPY: million liters per year  
NREL: National Renewable Energy Laboratory  
ODT: oven dry tonne  
PNNL: Pacific Northwest National Laboratory  
RED: Renewable Energy Directive (E.U.)  
RFS: Renewable Fuel Standard (U.S.)  
SE: system expansion (displacement)  
SMR: steam methane reforming  
SOC: soil organic carbon  
SG: switchgrass  
SRF: short rotation forestry  
WTW: Well-to-wheel  
WTP: Well-to-pump

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## DEDICATION

To my parents, who encouraged me to follow my dreams and have been with me every step along the way.

# CHAPTER 1: INTRODUCTION AND BACKGROUND

## 1.1 Introduction

According to groups such as the International Energy Agency (IEA), aviation is one of the fastest growing modes of transportation [1]. Consequently, aviation related emissions are expected to increase 5-fold over the next 30 years [2]. For this reason, the aviation sector has committed to reducing their emissions by 50% from 2005 levels by 2050 [3]. Biojet fuel is expected to be an essential part of the solution and its development is currently a strong focus of the industry. As will be described in more detail within this thesis, there are many different technologies to produce biojet fuel utilizing a variety of feedstocks. Although several airlines have invested in biojet production facilities, e.g. United Airlines and Cathay Pacific, it is still not clear which biojet technologies and feedstocks will give the greatest greenhouse gas (GHG) reductions. This thesis tries to clarify this point using a tool commonly used by governments, companies and researchers to determine the GHG reductions of fuels: Lifecycle Assessment (LCA).

This introductory chapter provides a background on the aviation industry's sustainability initiative, the various technologies used to produce biojet fuel (currently and under development), what LCA is and how it works. This chapter will also introduce the different variables used in LCA, as well as common LCA models.

In Chapter 2, the question of whether published LCA studies can be compared to determine which types of biojet fuel have the greatest GHG reduction potential will be explored. To date, many authors have published LCA studies that give the GHG emission reductions of various pathways for making biojet fuel. However, most of these studies only included one or two pathways [4] [5] and different models [6] [7] [8] were often used, making it more of an "apples-to-oranges" comparison. Attempts to determine which biojet pathway is "better" do so by comparing the results from various studies using different models [9] [10] [11] [12]. This chapter also aims to determine whether this approach is valid by conducting a systematic review of the biojet literature and determining how model variability might affect the GHG emission results.

Chapter 3 uses the insights gained from the systematic review to perform an LCA case study of the pyrolysis pathway to producing biojet fuel in British Columbia. Pyrolysis has shown superior economic potential to many of the alternatives with good environmental benefits [13]. Three potential sites in BC were modeled and the greenhouse gas emissions compared. A sensitivity analysis was conducted on

several parameters to determine their influence on the results. The merits of this pathway are discussed as well as the optimal conditions for GHG emission reduction. In the final chapter (Chapter 4), all major contributions and recommendations for future work, LCA methodological improvements, and policy are presented.

## 1.2 Background

### 1.2.1 History of Greenhouse Gas Reduction in Aviation

Though aviation only produces approximately 2% of the anthropogenic emissions worldwide, the sector has been growing steadily in recent years. For this reason, the International Civil Aviation Organization published a set of goals in 2009 to reduce their net CO<sub>2</sub> emissions in 2050 by 50% from 2005 levels [3]. Airlines, airports, and airplane manufacturers worldwide have been seeking ways to reduce emissions to reach ICAO's target (see Figure 1). To date, most of the reductions seen in aviation originate from new energy efficiency technologies and infrastructure, as well as improved operations of the aircraft [3]. Due to these efforts, a flight today emits only half as much CO<sub>2</sub> as in 1990 [14]. Most of this reduction is due to the decreased weight of the aircraft, improved engine efficiency and aerodynamics.

Increased operation efficiency also reduces emissions, so significant efforts have been made to develop technology and methods that minimize delays on the runway, unnecessary taxi and circling time and determine the most direct flight paths [14]. Aircraft materials and engines also continue to be optimized but will soon plateau as the minimum weight requirements are reached [15]. With a goal of a 50% reduction by 2050 over 2005 levels, it is apparent that increasing efficiency alone will not be sufficient to meet the greenhouse gas reduction goals set by the ICAO.

Thus, as Figure 1 indicates, the use of some type of alternative fuel will be critical for the industry to reach their reduction goals.

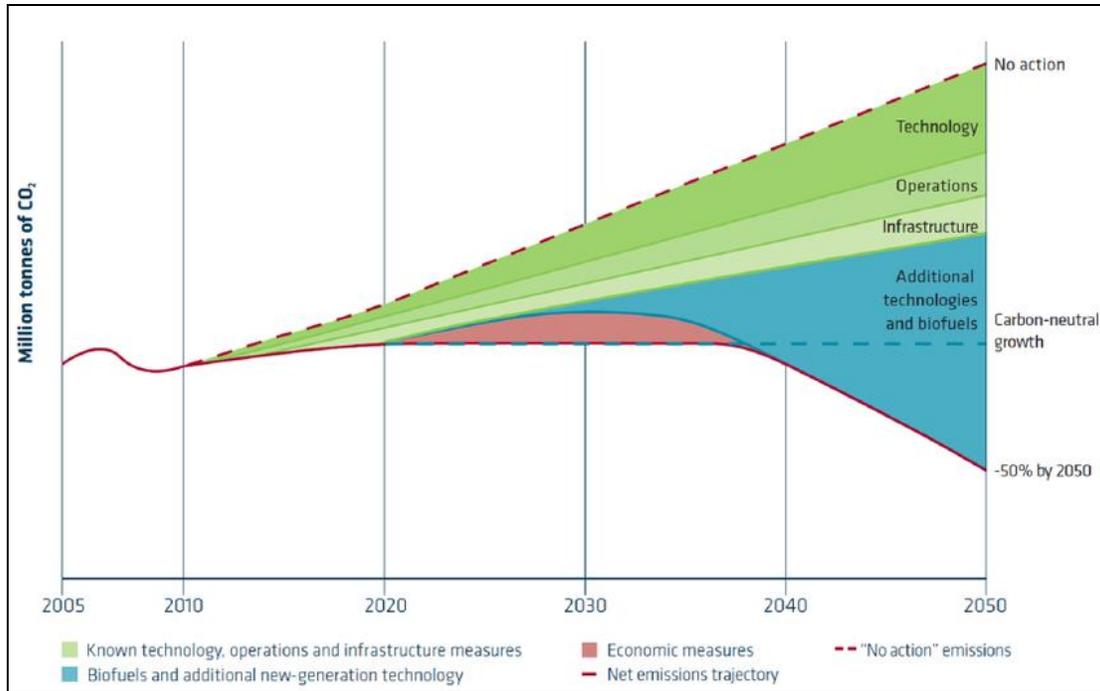


Figure 1: ICAO's Greenhouse Gas Reduction Plan [16]

As both hydrogen and electric planes would require new aircraft designs and technology that is not yet developed (see Appendix VI for more information), the aviation sector will need biofuels compatible with existing planes and infrastructure if they are to decarbonise over the next 20-30 years.

### 1.2.2 Biojet Fuel Production

Aviation jet fuel is a petroleum-based fuel made of kerosene (C8 to C16) range hydrocarbons. The most commonly used fuels for commercial aviation are jet A and jet A-1. As is the case for all petroleum fuels, jet fuel is a mixture of alkanes and aromatics. Jet fuels are designed to operate a turbine engine and must adhere to stricter requirements than other transport fuels due to safety considerations, including operability at very low temperatures (-43°C) [17]. Additionally, planes require a fuel with a high energy density for long intercontinental flights [18]. Traditional biofuels like biodiesel and ethanol contain a large amount of oxygen, which lowers the energy density of the fuel. Furthermore, these fuels have poor cold flow properties, which causes problems at flight temperatures. For these reasons, neither biodiesel nor ethanol can be used as an aviation fuel. Alternative, “advanced” biofuels are under development that can

be used in aviation. These fuels, often called “drop-in fuels”, are functionally equivalent to petroleum hydrocarbons and do not contain oxygen. Over the past decade, multiple pathways to producing biojet fuel have been developed. These pathways can be separated into three main categories: oleo-chemical, thermochemical and biochemical (see Figure 2).

The oleo-chemical pathways utilize oil feedstocks such as oilseeds, animal fats or lipids from algae. Most of the biojet fuel in production today is produced via this pathway and it is certified by ASTM for use in blends up to 50% [9]. However, concerns about the high cost of the feedstock, land use and food displacement issues have encouraged research into utilizing lignocellulosic feedstocks. Lignocellulosic feedstocks can be converted to biojet fuel via a thermochemical pathway, which utilize high temperatures and pressures to convert the biomass into fuel. Lignocellulosic materials can also be used as the feedstock for biochemical pathways, though the biomass must first undergo hydrolysis to C5 and C6 sugars.

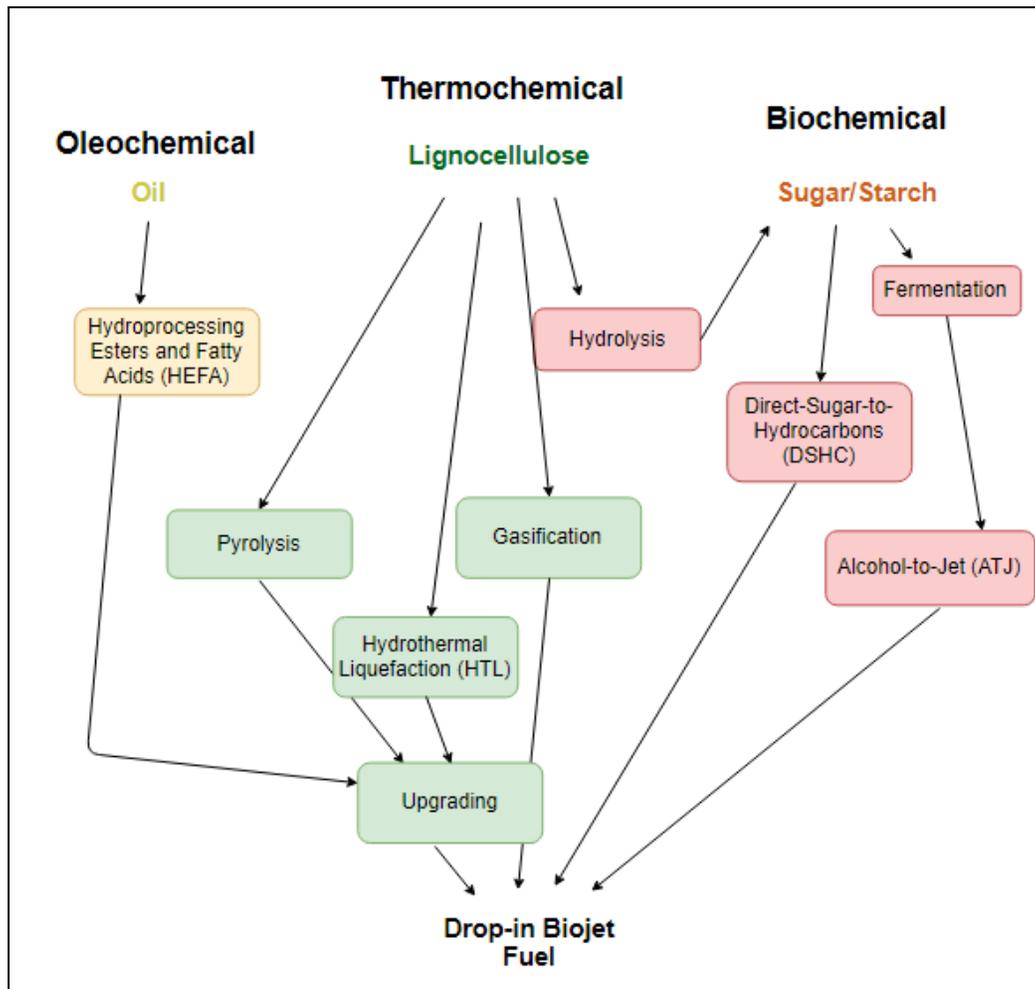


Figure 2: Biojet Fuel Production Pathways

Although other chemical and hybrid pathways have been proposed for the conversion of lignocellulose into biojet, such as syngas fermentation or the REACH process by Mercurius, they were outside the scope of this thesis [19].

Each of the main technologies considered for biojet production are described in more detail below.

#### 1.2.2.1 Hydrogenated Esters and Fatty Acids (HEFA)

Hydrogenated esters and fatty acids (HEFA), also known as hydrotreated renewable jet (HRJ), is a biojet fuel made from vegetable or animal oils. It is equivalent to conventional petroleum in properties but with a much higher cetane number, lower aromatic content and a negligible sulfur content [9]. Vegetable oils are comprised of a mixture of triglycerides of various types of fatty acids, usually between C12 and C22 in length [20]. To convert these oils into HEFA jet fuel, hydrogen is added to break the triglycerides and hydrogenate the fatty acid, producing a propane co-product [9].

The hydrogenated fatty acids must also be deoxygenated, which is accomplished through either decarboxylation or hydro-deoxygenation. Hydro-deoxygenation requires more hydrogen but retains all of the carbon of the original fatty acid. Decarboxylation, on the other hand, does not require the addition of hydrogen at all, but rather removes oxygen in the form of CO<sub>2</sub> (see Figure 3).

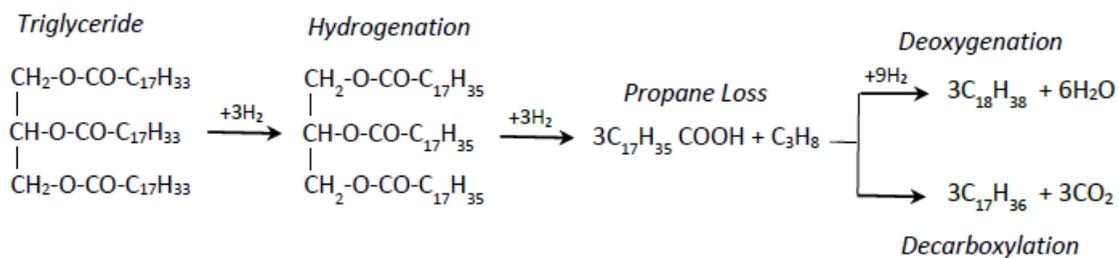


Figure 3: Deoxygenation pathways of triglyceride to hydrocarbon [21]

In most industrial processes, a combination of the two mechanisms occur simultaneously, though ratios can be adjusted to favour one mechanism over the other [22]. The yield of HEFA liquids via this process is typically around 80%. This includes all jet, diesel and gasoline, balanced by light gases, such as methane, propane, carbon dioxide and carbon monoxide [22]. Diesel is typically the predominant product and jet fuel yields are typically around 13% when a decarboxylation process is used [22]. Although it is possible to increase the jet fuel fraction of fuel to 50% through further hydrocracking and isomerization, this would reduce the overall yield by 10% and increase the needed hydrogen by 30% [21].

Biojet fuels made from plant oils (HEFA) are certified by ASTM and have been demonstrated in over 2000 flights worldwide [23]. Boeing has also tested the use of green diesel (15% blends) in a demonstration flight and is now awaiting ASTM certification [24].

#### *1.2.2.2 Pyrolysis*

Pyrolysis is the thermal decomposition of biomass that occurs in the absence of oxygen or air. Pyrolysis converts lignocellulosic materials into gas, liquids, and solid fractions. The relative amount of product depends upon the temperature and reaction time. Fast pyrolysis, where the lignocellulose particle is rapidly heated to 500 °C in one or two seconds, produces the greatest amount of liquids and is therefore typically employed for biofuel synthesis [22]. In fast pyrolysis, the yield of liquid, often called bio-oil, can be upwards of 75% of the mass of the initial material [25]. This rapid heat transfer is challenging and typically requires a heat transfer medium such as sand in a fluidized bed reactor [22]. While slow pyrolysis has been commercial for decades (the process that produces charcoal or briquettes), fast pyrolysis is not as mature and is only employed by a few companies commercially [22].

Bio-oils are relatively unstable due to their high oxygen content (15-30%), so must be upgraded to produce a fuel suitable for transportation. [9]. Hydrotreatment subjects the bio-oil to hydrogen at high pressure over a catalyst to de-oxygenate the liquid, much like the treatment used for HEFA. However, the reactions are not highly selective, and a significant amount of side polymerization and condensation reactions also occur, reducing the overall yield [22]. Significant amounts of hydrogen are generally consumed in this process.

#### *1.2.2.3 Hydrothermal Liquefaction*

Hydrothermal liquefaction (HTL) is a similar process to pyrolysis but operates under very high pressures (2611 psia) [22]. Under these conditions, water in the mixture acts as a subcritical fluid and favours the production of liquid products. Unlike pyrolysis, the biomass feedstock need not be dried, but can enter the process at its full moisture content, up to 50%.

The HTL reactor yields an oil phase, an aqueous phase, and gases. The oil phase is the main bio-oil product and has a yield of 45-50% on a dry basis. It has an oxygen content of 10-20%, which is significantly lower than that of pyrolysis bio-oils (35-40%). The aqueous phase contains water and dissolved organics and is easily separated. The gases are mostly carbon dioxide and carbon monoxide, with trace amounts of methane and hydrogen [26]. Most of these streams are used internally to pre-heat the incoming wood feedstock, while the remainder is sent to a wastewater facility and treated using anaerobic digestion.

After HTL treatment, the bio-oil must be further upgraded before it can be utilized as a transport fuel. This upgrading process is similar to that of pyrolysis, except that the hydrogen requirements are lower, due to the lower content of oxygen in the bio-oil.

#### *1.2.2.4 Gasification*

Gasification is the thermal conversion of solid biomass to mostly gas, with some liquid and solid co-product. Called “syngas,” the resultant gas is mainly comprised of hydrogen and carbon monoxide, with small amounts of methane, water vapour, and carbon dioxide [22]. Gasification of biomass occurs when biomass particles (10% moisture content and 2mm size) are subjected to high temperatures (800-1000 °C) and pressures (1-50 bar) in the presence of stoichiometric amounts of air, oxygen, or steam [22].

Unlike combustion, where carbon reacts with oxygen to form CO<sub>2</sub> in an oxygen rich environment, gasification takes place in a limited oxygen environment, resulting in a product that is only partially oxidized [22]. If the syngas is to be upgraded into a liquid transportation fuel, oxygen and steam are preferred over air, which has high amounts of nitrogen and produces a low energy product [27].

After gasification, the gas stream passes through a direct-quench cooling system where the syngas is cleaned by removing NO<sub>x</sub>, SO<sub>x</sub>, coke, and ash [22]. Syngas clean-up is still an evolving area of research and is one of the most challenging and expensive steps of liquid fuel production via gasification. Utilizing high temperatures (>200 °C) during gasification can reduce the formation of char and coke, however this is rarely used due to the high capital and operating costs [22]. Though challenging, syngas cleanup is essential prior to the conversion to liquids, as particulates, NO<sub>x</sub> and SO<sub>x</sub> will poison catalysts used in Fischer-Tropsch synthesis.

Fischer-Tropsch (FT) is a catalytic process for converting syngas into liquid transportation fuel. In this reaction, carbon monoxide and hydrogen react upon a catalyst surface to form a long-chain alkane. The Fischer-Tropsch reaction is conducted at 10-40 bar, while the temperature is dependent upon the desired hydrocarbon product. For example, 300-350 °C favors gasoline formation, while 200-240 °C will favor diesel and other heavy hydrocarbons [22]. If a large jet fraction is preferred, a fraction of the longer diesel-range molecules can also be cracked, isomerized, and distilled to adhere to jet fuel specifications.

#### *1.2.2.5 Alcohol-to-jet (ATJ) and Direct-Sugar-to-Hydrocarbon (DSHC)*

ATJ and DSHC are processes that employ both thermochemical and biochemical processes. In ATJ, biologically produced ethanol (produced from starch or sugar via enzymatic hydrolysis and anaerobic fermentation) is upgraded to jet range hydrocarbons through dehydration, oligomerization and

hydrogenation [28]. DSHC uses a similar upgrading technique but with a different biologically produced intermediate: isoprenoids and terpenes [22]. One example is farnesene, which after upgrading is certified for blending with fossil jet fuel in blends up to 10% [28]. However, the value of the biochemical intermediates (\$1.75/L farnesene intermediate [29]) is often much higher than the jet fuel itself (\$1.61/gallon jet A-1 [30]), which discourages the conversion of these intermediates to biojet fuel [31]. As a result, these biochemical pathways will not be discussed further in this review.

### 1.2.3 Life Cycle Assessment

Life cycle Assessment (LCA) is a tool that calculates the environmental impact of a product or process, taking into account the impacts made over the entire lifecycle [32]. LCA is capable of measuring impacts such as eutrophication, acidification, and toxicity, but the most commonly measured metric is the global warming potential (GWP), also called the cumulative greenhouse gas (GHG) emissions. LCA of this metric includes the GHG emissions that occur at every stage of a product's life, which for biofuels includes feedstock cultivation, transportation, conversion and end use (see Figure 4). The combination of all these emissions gives the overall lifecycle GHG emissions of a product that can then be compared against other fuels or alternatives.

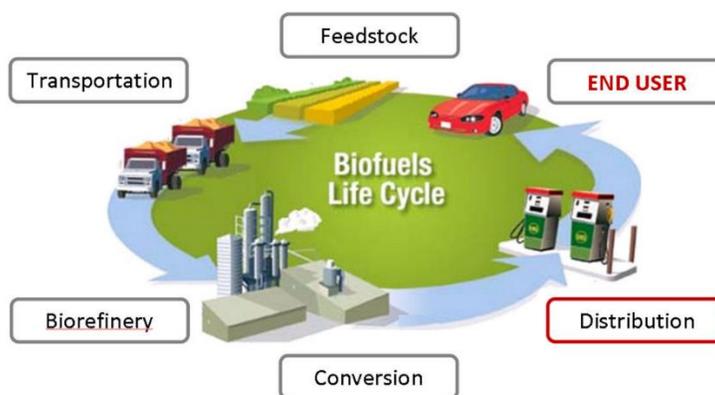


Figure 4: Stages of a biofuel lifecycle that are accounted for in biofuel LCA [33]

Lifecycle assessment was first used in industry by companies such as Coca-Cola to compare the environmental impact of alternative products, but today is commonly employed by governments to track the carbon emissions of jurisdictions and inform climate change policy [34]. In the U.S., LCA is an essential part of the Renewable Fuel Standard (RFS) legislation, which requires all transportation fuels to contain a minimum percentage of biofuel [35]. In British Columbia, the Renewable and Low Carbon Fuel Requirements Regulation also requires that fuels sold within the province contain a minimum amount of

biofuel, but also requires that fuel suppliers gradually reduce the carbon intensity of their fuels so that by 2020 there is a 10% GHG emission reduction relative to the fuel mix in 2010 [36]. Lifecycle assessment is used to determine the carbon intensity of these fuels and whether fuel suppliers are complying with the regulation.

Depending on the goal and scope of the LCA, the lifecycle system boundary can be changed to only include the elements of interest. Within climate change policy, well-to-wheel (WTW) is the most common system boundary, which incorporates all of the upstream emissions of feedstock production (including land use change) through to the combustion of the fuel in an engine or boiler, as shown in Figure 4 [35]. However, when comparing and contrasting various technologies, gate-to-pump (GTP) may be preferred. In this type of analysis, the system boundary begins at the cultivation and procurement stage of the feedstock and continues through to the point at which the fuel reaches the pump (i.e. distribution). Other combinations of these two system boundaries are also possible, such as well-to-pump and/or gate-to-wheel. Furthermore, lifecycle assessments may be attributional, which means that they are a snapshot of the lifecycle of a product at a specific point in time; consequential, where the lifecycle and its inputs are dynamic and indirect impacts on the technosphere are also considered; or a hybrid of both.

Lifecycle Assessment is guided by the International Standards Organization under ISO 14041-14044, which defines the various steps and methodology required to perform a lifecycle assessment and gives suggestions for certain variables [37]. However, the user is left to make many assumptions, which can impact results and comparability between studies. A few of the more important variables will be defined and discussed in the following section.

#### *1.2.3.1 Lifecycle Analysis Variables*

Lifecycle Analysis (LCA) variables can be split into two categories: process variables and methodological variables. Process variables include decisions made about the process, such as whether to recycle heat, use co-products for heat generation, or whether to include methane capture on wastewater anaerobic digesters. These process decisions are made by the fuel and feedstock producers during the process design. Others are the result of those process decisions, such as process yield. In this analysis, these variables are defined as process conditions and system choices. The location of the process and year the LCA is performed are also considered process variables.

Methodological variables are defined by the modeller during the goal and scope of a lifecycle assessment. Unlike process variables, these choices are qualitative and difficult to measure. Though ISO 14040 gives suggestions for some methodological variables, most of the decisions are left to the user's discretion, which can introduce a large amount of variance to identical processes. Several of the most important methodological variables are described in detail below.

#### *1.2.3.1.1. Functional Unit*

The functional unit is the basis of all calculations within an LCA. The functional unit for biofuel LCA is often gCO<sub>2</sub>eq/MJ of fuel, but the emissions are also sometimes expressed per volume or mass of input biomass feedstock, volume or mass of biofuel, transport distance (km or mile), or feedstock land area (hectare) [38] [39]. Wiloso et al. [40] ascertain that the most appropriate functional unit choice for biofuel is gCO<sub>2</sub>eq/MJ of fuel, since energy is the primary use of the fuel.

#### *1.2.3.1.2 Land Use Change (LUC) and Indirect Land Use Change (iLUC)*

According to work done by the IPCC, 14% of total global emissions are due to agriculture cultivation, while 17% of all world-wide greenhouse gas emissions originate from change in land use [41]. Land use change (LUC) emissions originate when land is converted from a type that stores a large amount of carbon (such as forests and peatland) to one that stores minimal carbon (such as pasture land). Carbon is also stored in the soil, so when the soil is tilled or disturbed, a large amount of carbon dioxide and nitrogen oxide is emitted to the atmosphere [42]. Many studies now include emissions from land use change and soil disturbance in their calculations in order to give a more accurate estimate of agricultural emissions.

However, land use change is not always defined in the same way. For example, while the IPCC defines land use change as only the emissions associated with a change in land use, the U.S. LCA Model GREET (The Greenhouse gas, Regulated Emissions, and Transportation Model) includes cultivation emissions from N<sub>2</sub>O in its definition of LUC [43]. Some policies, such as the European Renewable Energy Directive, separates Land Use Change into direct land use change and agricultural improvement. In the work reported in this thesis, the IPCC definition of LUC is used throughout.

Indirect Land Use Change (iLUC) is another form of land use change that some believe should also be included in LCA. Indirect Land Use Change is the carbon loss attributed to a process that displaces another product, thereby requiring land conversion elsewhere in the world to provide pastureland for the displaced product. If included in LCA, iLUC has the potential to add a significant amount of emissions. In

2015, the E.U. required all LCA for the E.U. Renewable Energy Directive (RED) fuel compliance to include an estimate for this variable, though many advocate that iLUC is too imprecise for inclusion [44] [45]. In 2018, Argonne National Laboratories released a comprehensive study on iLUC inclusion for U.S. biodiesel and found that it did not have as large an impact as previously thought, due to increased crop intensities rather than increased land clearing [46]. Due to the relative uncertainty of this variable, this variable will not be included or considered in any of the studies within this work.

### 1.2.3.1.3 N<sub>2</sub>O Emissions

Nitrous oxide (N<sub>2</sub>O) occurs throughout most biojet production processes, but the majority of these emissions occur during feedstock cultivation. Nitrogen is abundant in both soil and organic matter, as it is an essential building block of plant cells and protein. Plants require nitrogen from the soil to grow, so fertilizer is generally applied to soils to maintain nitrogen levels after crop harvest. Synthetic fertilizers or manure are the most common type of fertilizer, though crop residues left on the field also assist in the maintenance of soil nitrogen and organic carbon [47]. N<sub>2</sub>O emissions from volatilization and the nitrification/denitrification processes of fertilizers, crop residue, and below ground biomass (roots) are considered direct emissions [42]. While most of the nitrogen enters the soil and eventually the crop, some of the applied fertilizer and crop residue will leach or run off. These emissions are considered indirect emissions [42]. In 2006, the IPCC approximated that 1.325% of all applied fertilizer ends up in the atmosphere as N<sub>2</sub>O from both direct (1%) and indirect sources (0.325%).

For crop residues (above and below-ground biomass), the IPCC estimates that approximately 1.225% of all crop residue decay contributes to atmospheric nitrogen, both directly and indirectly. This therefore suggests the following formula for use in LCA to determine the nitrogen emissions from crop residues with fertilizer use and crop residues:

$$N_2O \text{ Emissions} \left( \frac{g_{N_2O}}{bu} \right) = \left( \left( \frac{g_{\text{nitrogen fertilizer}}}{bu} \right) \cdot 0.01325 + \left( 200.7 \frac{g_{N\text{-crop residue}}}{bu} \right) \cdot 0.01225 \right) \cdot \left( \frac{44g_{N_2O}}{28g_N} \right)$$

Estimates of N<sub>2</sub>O are subject to large uncertainties, as they are affected by factors such as soil type, climate and tillage method [48]. As a result, IPCC has also developed more detailed and regional-specific equations for N<sub>2</sub>O calculations which increase the accuracy of the calculation. These are called Tier 2 and Tier 3 equations and require the use of soil and climate data [42]. These methods give differing results that could provide significant differences in the LCA of cultivated crops.

#### *1.2.3.1.4 Co-Product Allocation*

One of the major inconsistencies between LCA studies results from co-product allocation assumptions. The ISO 14040 guidelines allow the allocation of co-products among any of the functional bases (energy, mass, value, function) though prefer the use of system displacement, also known as displacement.

#### *System Expansion: Displacement method*

In this method, co-product allocation is avoided by expanding the system boundaries to include the product that the co-product displaces. The emissions associated with the “displaced” product are subtracted from the total emissions of the pathway, acting as a credit. In this method, all of the emissions produced during the production of the product and co-products are retained in the analysis and do not leave the system. For example, with the production of canola biodiesel, canola meal is produced as a co-product. Using this method, the emissions from soymeal production, which is what would be used if the canola meal hadn’t been produced, are subtracted from the total emissions of canola biodiesel production [7]. The use of this co-product method requires information about each co-product to be included in the database, as well as a suitable basis for the substitution. This becomes challenging when considering novel products, as the substitution must be assumed.

#### *Energy Allocation*

In this method, emissions associated with a production process are allocated among co-products based on their energy content (lower (LHV) or higher heating value (HHV)). This is a useful unit of comparison when the products are all fuels, as they are produced for their energy content [49]. Most fossil fuel LCAs use the energy allocation method for distributing emissions among fuel co-products [50]. The disadvantage of this method becomes apparent when the co-products are not all fuels. For example, glycerin is a co-product of biodiesel production and is used as a cosmetic product, so using energy allocation would ignore its value in this context.

#### *Mass Allocation*

Mass allocation is a common basis of allocation among co-products of non-fuel LCAs. Most manufacturing models, such as SimaPro and Gabi, allocate emissions from a production process based on co-product mass [39]. Although simple, this method does not take the value of a product into account. If the co-products all have a similar market and use, this method may be representative, but this is rarely the case. It has been suggested that this method will alter the results in a way that is not representative of reality [7].

### *Economic Allocation*

Economic allocation has been referred to by ISO 14040 as a “means of last resort”, to be used only when no other satisfactory method can be found [49]. In this approach, the product selling prices are used as the basis for emission allocation. The advantage of this method is that it recognizes the value of each product and distributes emissions based on the perceived value. When co-products enter different markets, such as is the case of glycerin and biodiesel, this method can accurately distribute the emissions. By using this allocation method, the value of the glycerin co-product is accounted for. Another advantage of this method is that it discourages the allocation of emissions to manufacturing wastes, which could be misleading.

#### *1.2.3.1.5 Emission Potentials (GWP)*

Greenhouse gases (GHGs) are attributed to global warming due to the “greenhouse effect” they cause by their absorption and re-emittance of infrared energy. Greenhouse gases are not affected by the short-wave radiation from the sun but block the infrared (long-wave) energy radiated from the earth from leaving the atmosphere, thereby acting as a blanket insulating the earth. Greenhouse gases that exhibit this quality, called “radiative forcing,” are CO<sub>2</sub>, methane, N<sub>2</sub>O, and fluorocarbons. Each of these gases have a different radiative forcing effect, as well as a different lifetime in the atmosphere [51]. The global warming potential (GWP) was developed to allow comparisons of the different global warming impacts of GHGs. The GWP of a greenhouse gas is determined by its warming effect relative to CO<sub>2</sub>: it is the amount of energy 1 tonne of the greenhouse gas will absorb over a time period relative to 1 tonne of CO<sub>2</sub> [51]. The period typically used for these calculations is 100 years, though 20 years is also used [51]. The IPCC publishes GWP values based on current scientific knowledge and periodically updates these values as findings change. The radiative forcing of a gas can also change over time, based on the concentration of the gas and other gases in the atmosphere [51]. Table 1 gives the GWP values that the IPCC has attributed to each GHG over time. As can be seen, the value for methane continues to increase, while the value for N<sub>2</sub>O has dropped with each update [51].

**Table 1: IPCC Global Warming Potentials [52]**

	<b>Second Assessment Report (SAR)</b>	<b>Fourth Assessment Report (AR4)</b>	<b>Fifth Assessment Report (AR5)</b>
	<b>1996</b>	<b>2007</b>	<b>2014</b>
<b>Carbon Dioxide (CO<sub>2</sub>)</b>	1	1	1
<b>Methane (CH<sub>4</sub>)</b>	21	25	28
<b>Nitrous Oxide (N<sub>2</sub>O)</b>	310	298	265

As these variations can have a significant impact on overall LCA results, it is important to understand which IPCC report a study has utilized.

#### **1.2.3.1.6 Data Source**

Lifecycle Analysis (LCA) has shown to be heavily influenced by the amount and accuracy of input elements and these elements are often based on data of varying source and quality [40]. This data is generally a combination of two types: primary and secondary data. Primary data is a quantified value of a unit process obtained from a direct measurement or a calculation based on direct measurement. This type of data is generally situation-specific and varies for each LCA. Secondary data, on the other hand is not obtained directly from the source, but rather from a database, report, or literature study. It is often used for auxiliary processes such as fertilizer manufacture and electricity production [32]. While some central databases exist (e.g. Ecoinvent), most do not include the data required for advanced biofuel processes. Instead, many authors utilize the data within a biofuel-specific LCA model or from literature, and the inconsistencies between these sources can add variation into LCA studies.

#### 1.2.4 Lifecycle Assessment Models

Numerous models exist both in theory and as software, but only a few can be used for biofuel emissions evaluation. A non-exhaustive list of models that could be used for biofuel production is shown below [37].

##### Generic Models:

- **SimaPro**
- Umberto
- GEMIS
- Quantis
- BESS
- Gabi
- Aspen-Plus

##### Biofuel-specific Models:

- **REET**
- **GHGenius**
- LEM
- BioGrace

In literature, the most commonly used models are REET, GHGenius, and SimaPro. These models are computer-based and contain all the formulas and data required to perform LCA calculations. Each of the models considered in this work are described in detail below.

##### 1.2.4.1 REET

REET was developed by the Argonne National Laboratory in 1996, sponsored by the U.S. Department of Energy's Office of Energy Efficiency and Renewable Energy [53]. Its purpose is to allow researchers and analysts to fully evaluate the energy and emission impact of advanced vehicle technologies and new transportation fuels. It is also part of the development of the U.S. Renewable Fuel Standard and a California-specific version (CA-REET) was created for the California Low Carbon Fuel Standard (LCFS) legislation. Both versions are still used by the government to determine whether a specific biofuel reaches the required GHG reduction thresholds required to qualify under the Standard [53].

REET.net was released in 2013 and includes data from over 100 conventional and renewable fuel pathways, as well as 80 fuel/vehicle systems. The user may perform calculations for one of these pathways, modify them to include their own process data, or create their own. Either a well-to-wheel (WTW) or well-to-pump (WTP) analysis can be performed [54]. Data is stored internally and all greenhouse gas emissions are converted and presented in terms of CO<sub>2</sub> equivalence.

##### 1.2.4.2 GHGenius

GHGenius was developed by Mark Delucchi in 1999 as a Canadian version of his LEM life-cycle analysis model. LEM (Lifecycle Emissions Model) was developed between 1987 and 1993 and is an LCA model that determines the greenhouse gas and non-greenhouse gas emissions for various alternative fuels in the transport sector [55].

Unlike LEM, GHGenius is based on Canadian data from Natural Resources Canada, Statistics Canada, and other Canadian governmental agencies. It has been used directly by NRCan and industry as the basis of many environmental reports [49]. The GHGenius model is based in excel and does not have a visual user interface. It is a complex assortment of spreadsheets, which contain all of the source data and formulas needed to run the model.

The model default is well-to-wheel LCA, which includes the lifecycle of the vehicle as well as the fuel of over 200 vehicle/fuel combinations. However, it is possible to shrink the system boundary to a WTP analysis. GHGenius calculates the emissions of 10 different gases, which are then converted to a CO<sub>2</sub> equivalence based on the GWP factors from IPCC or its own developed list of standard values [49].

#### *1.2.4.3 SimaPro*

Unlike GHGenius and GREET, SimaPro is not a biofuel-specific model. SimaPro is a professional LCA package that is used in over 80 countries and has been considered the industry standard in LCA for over 30 years [56]. It contains many industrial processes and products but is mostly focused on material products and flows. The program relies heavily upon external databases, which determine the products and materials included in the model [57]. The most commonly used database is Ecoinvent, which is a European database. Other databases included in the model are Agri-footprint, ELCD, Swiss Input Output database, and USLCI. Of these databases, only the USLCI database contains non-European data.

Of all the models, SimaPro offers the most polished user interface and is designed for the industrial and business user [57]. However, SimaPro is not free to use as are GHGenius and GREET, which limits its use as a tool for the general public. While SimaPro contains significantly more data than either GHGenius or GREET, most of the data is not applicable to biofuel production. For example, the only biofuels contained within the database are ethanol, soy biodiesel, and biogas [57]. Any other fuels, such as renewable diesel or biojet, would need to be added to the model by the user.

#### *1.2.4.4 Model Comparison*

GHGenius, GREET and SimaPro all have the capability of conducting an LCA on biojet fuel, but have noticeable differences. Table 2 identifies several important differences between the models.

Table 2: Comparison of LCA model parameters and variables [58] [54] [59]

Parameters	GHGenius (4.03a)	GREET.net (2016)	SimaPro (8.3)
Type of LCA	Attributional or Hybrid	Attributional or Hybrid	Attributional, Hybrid, or Consequential
Biofuel-Specific	Yes	Yes	No
Location	Canada, (primary), U.S., Mexico, India (Secondary)	U.S.	Europe (primary), U.S. (Secondary)
System Boundary	Well-to-Wheel/Pump	Well-to-Wheel/Pump	Well-to-Wheel/Pump
Inventory Database	Internal	Internal	External: Ecoinvent (default) or 8 others
Functional Unit	Service (km), Energy (MJ)	Service (mile, km) Energy (Btu, MJ)	Defined by user
Co-product allocation method	Displacement + Energy (fuel products only) hybrid	Market + Energy (fuel products only) hybrid	Depends on database (generally mass or energy)
GWP	IPCC 1995, 2001, 2007, or internal	IPCC 2013	IPCC 2013 or 2007
Land use change	Direct included by default (specific to region and crop)	Indirect and Direct available as add-on (CCLUB) for some crop varieties	Direct can be included (specific to region, not crop)
Cultivation emissions	IPCC Tier 2 or 3	Altered IPCC Tier 1	IPCC Tier 1
Fossil jet fuel baseline Regional Jet (gCO <sub>2</sub> eq/MJ)	87.13 (Jet A1 – 0.06%S)	86.36 (Jet A)	85.9
Impact categories	GHG; Energy; Cost	GHG; Energy;	GHG; Energy; Eutrophication; Acidification; Human Toxicity; etc.

As the table indicates, each model handles methodological variables differently. For example, all of the models are capable of handling a variety of co-product allocation methods but employ different defaults. GHGenius defaults to the displacement allocation method in all areas except for energy and fuel products, while GREET uses market and energy allocation. The default in SimaPro depends on the inventory database selected by the user, and not all choices are generally available for each process.

Since each model utilizes different variable assumptions, this can have a profound impact on the overall LCA GHG emission results, even for the same technology and feedstock. This adds a degree of uncertainty when comparing published LCA literature, especially because these variables are rarely discussed. The following chapter aims to reduce this uncertainty by performing a systematic review of the biojet literature, taking into account the assumption variance inherent between models. The current state of the art of biojet literature is presented after some harmonization and theories, regarding the influence certain variables have on results, are explored.

## CHAPTER 2: AN ASSESSMENT OF HOW VARIOUS ASSUMPTIONS INFLUENCE THE OUTCOMES OF “ADVANCED BIOFUEL” LIFE-CYCLE ANALYSIS MODELS

### 2.1 Introduction

Many life-cycle analyses have been conducted on biojet fuel to compare GHG emissions, but generally only one or two pathways are included per study [4] [5]. Many use different models [6] [7] [8]. The use of different models in lifecycle analysis (LCA) literature is problematic because LCA models (such as GREET, GHGenius, and SimaPro) have been shown to give different results for the same process [60].

To date, most attempts at determining which biojet pathways provide the greatest GHG reducing benefits do so by comparing the results from LCA studies without accounting for model or variable differences [9] [10] [11] [12]. These differences can affect the overall GHG emission results, leading to erroneous conclusions. Therefore, this chapter performs of a systematic review of the biojet LCA literature in order to provide insight into which variables have the largest impact on results and reduce the uncertainty in result comparison. While performing the systematic review, the research questions that are considered are the following:

- 1) Are the variable assumptions the same for all studies utilizing the same model?
- 2) Do models agree on which biojet pathways offer the greatest GHG reductions?
- 3) Which variables/assumptions have the largest impact on LCA results?

To answer these, a methodology for literature comparison was determined. All literature studies were harmonized and separated according to this methodology and the raw data presented. Next, a qualitative analysis was conducted on all literature studies. This includes both an in-depth look into the variable choices used by the LCA literature and how these might cause variance within and between models. The insights from these analyses was then used to answer the research questions and provide recommendations for future research. This work provides the foundation for a meta-analysis to be conducted on the current body of literature and provide average GHG emission values for each pathway.

### 2.2 Methodology

#### 2.2.1 Experimental Design

In this chapter, a systematic review of biojet literature using oleo-chemical and thermochemical methods was performed to compare the GHG reduction potential of these fuels. As previously discussed,

comparing the results of literature can be challenging because studies often use different models, assumptions and data. According to ISO 14040, users can change inputs and make assumptions in order to best represent each particular case [32]. This makes it difficult to perform quantitative research, which requires that all variables be controlled to ascertain a primary variables' influence on the dependent variable. Therefore, this work aimed to rectify this inconsistency by normalizing the literature LCA GHG data and identifying the influence of study variables on results. Data was separated by model used, as this was identified as a possible source of variance, and a preliminary analysis of the data was performed to identify variables with high impact on results. These were then modified to reduce variance and improve comparability between studies. A qualitative analysis was then performed on all literature studies to determine the variability of all other variables between models and develop theories about how the variable selections of each model impact the overall GHG emission results. This raw data and variable influence insight could be used to perform a meta-analysis on the biojet literature, though this step is outside the scope of this thesis.

Prior to LCA GHG emission result comparison, all LCA literature was screened to eliminate any study that did not use either the GREET, GHGenius, or SimaPro model. These models were chosen for study based on their prevalence in the biojet literature (see Section 2.2.2). Any literature studies that did not contain information regarding their assumptions or variables were also discarded. Once filtered, the remaining studies were separated according to the model used, according to Figure 5. The following sections provide more details about the preliminary analysis and variable harmonization that the literature data underwent prior to display and comparison, as well as other details about the experimental procedure and data analysis strategy.

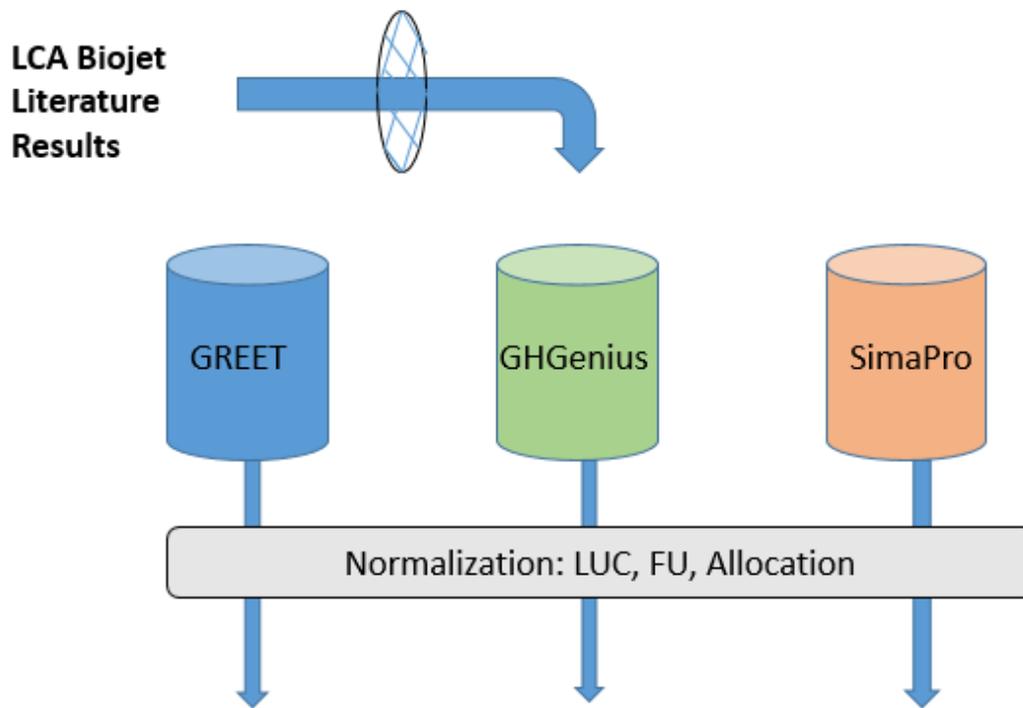


Figure 5: Methodology employed for preparing biojet literature

### 2.2.2 Preliminary Analysis and Variable Harmonization

The challenges associated with land use change inclusion and co-product allocation method are well discussed in literature. For this reason, many authors present their results in terms of several allocation methods to show the difference [48] [61] [28]. ISO 14040 recommends the use of the displacement co-product method whenever possible, but several authors argue that this should not always be the case. Stratton et al. [10] find that under some circumstances, using the displacement allocation method can greatly skew results. Figure 6, which is from their work, shows that the results given when using the displacement allocation method are strongly influenced by the amount of co-product being produced. As the amount of co-product produced increases relative to that of the product, the GHG LCA emissions approach negative infinity. With large amounts of co-products, the pathway becomes carbon negative, as shown in the figure when product yields fall below 20%. This phenomenon is problematic, as it can result in negative emissions without any type of carbon sequestration taking place. Furthermore, utilizing displacement adds a consequential aspect to an otherwise attributional lifecycle, especially when you cannot ensure that the emission savings credited to the fuel lifecycle are not being credited by the co-

product elsewhere. For these reasons, the displacement allocation method can skew results under high co-product circumstances.

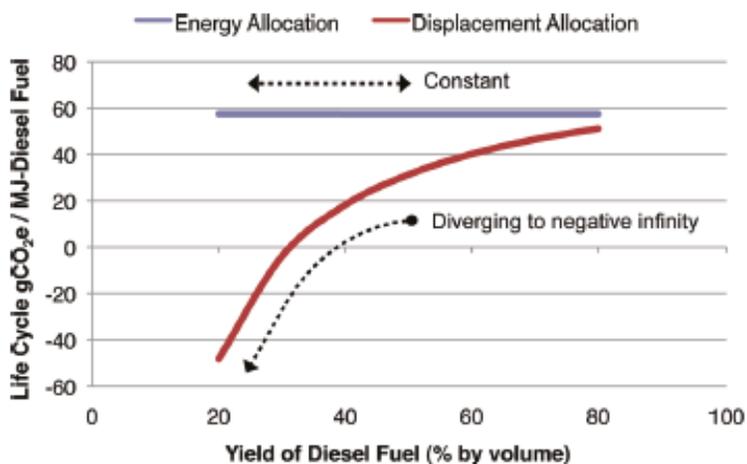


Figure 6: Effect of the Displacement allocation method with increasing co-product volume, reproduced with permission [10]

Additionally, it was found that utilising the displacement allocation method often provides lower GHG emission results on average. As Table 3 indicates, this was the case among the biojet literature within this study that utilized several allocation methods. A proof by de Jong et al. [28] showed that this occurs anytime that the product being displaced in the market has a higher carbon intensity than the co-product (see Appendix III).

Table 3: GHG emission results from biojet LCA literature that utilize multiple allocation methods (gCO<sub>2</sub>eq/MJ)

Biojet fuels from various sources	Displacement	Energy Allocation	Market Allocation	Mass Allocation	Displacement vs. Allocation (%)
Soy [63]	37.3	37.5	36.2	21.0	+19
Soy [60]	40.2	44.9	--	--	-11
Soy [89] gCO <sub>2</sub> eq/MMBTU	-30,000	35,000	40,000	--	-85
Palm [87]	24.7	26.2	26.2	25.8	-6
Palm [60]	30.3	39.2	--	--	-23
Camelina [60]	48.9	47.6	--	--	+3
Camelina [28]	44	47	--	--	-6
Jatropha [60]	1.69	59	--	--	-98
Jatropha [28]	22	55	--	--	-60
Pennycress [60]	-18.3	32.7	--	--	-145
Tallow [60]	1.3	31.42	--	--	-96
Used cooking oil [28]	27	27	--	--	0

Forest residue <sup>2</sup> [85] gCO <sub>2</sub> eq/MMBTU	78,000	156,000	147,000	--	-48
Forest residue <sup>1</sup> [28]	-3	6	--	--	-150
Forest residue <sup>2</sup> [28]	37	40	--	--	-8.5
Corn Stover <sup>1</sup> [28]	4	13	--	--	-70

<sup>1</sup>Gasification

<sup>2</sup>Pyrolysis

For these reasons, literature data that utilized the displacement method was discarded in this work whenever an alternative was possible, such as with the studies in Table 3. The default allocation method of both SimaPro and GREET is energy or an energy hybrid (see Table 2), so data utilizing these methods was preferred for these models. However, some studies utilizing the GREET and SimaPro models utilized displacement without alternative, as did all studies using the GHGenius model. In these cases, the data was marked to indicate these differences in all figures. Details about the data presentation when displacement was used is given in Section 2.2.4.

The inclusion of land use change (LUC) can also significantly affect overall GHG results, especially for cultivated oleo-chemical pathways. Though important, many authors are hesitant to include these emissions, as they can easily dominate the GHG emission results. Stratton et al. [62] and Wong [63] calculated the GHG emissions for soy and palm for different direct land use changes, which are presented in Table 4 [63] [7]. As can be seen, the type of land use change makes a significant difference and easily overwhelms the baseline results.

**Table 4: Additional GHG emissions from direct land use change (LUC) [7]**

	Soy (gCO <sub>2</sub> eq/MJ)	Palm (gCO <sub>2</sub> eq/MJ)
<b>Baseline (No land use change)</b>	<b>37.0</b>	<b>30.1</b>
Grassland to Cropland	+60.8	
Rainforest to Cropland/Plantation	+527.2	+135.8
Logged forest to plantation		+9.6
Peatland to plantation		+667.9

Furthermore, these emissions come with a great deal of uncertainty, as direct land use change calculations are generally site specific and inconsistent methodologies have been used. For example, Stratton et al. [62] calculated direct land use change emissions between 60 and 527 gCO<sub>2</sub>eq/MJ for soy, Yan et al. [64]

estimated them to be between 169 and 845 gCO<sub>2</sub>eq/MJ and the U.S. Environmental Protection Agency estimated them at 32.3 gCO<sub>2</sub>eq/MJ [20]. Similar variances exist for other oilseed crops, as well [20].

Finally, not all land use change increases emissions. There are some instances where the land use change consists of agricultural improvements on degraded land. In such instances, studies have shown that soil carbon concentrations may increase, thereby reducing overall GHG emissions (see Figure 7). This is often the case with switchgrass and with canola in certain regions [7] [65]. Furthermore, the type of tilling practices employed on the field can also significantly impact GHG emissions. The use of no-till, or reduced-till, agricultural management, which is becoming prevalent in western Canada, reduces the soil disturbance, thereby reducing the soil carbon lost as CO<sub>2</sub> and increases soil organic carbon (SOC) over time [66]. A reduction in summer fallow, which is the practice of resting land in-between crop cycles, was also shown to improve SOC and reduce the breakdown of crop residue as emissions [67].

A preliminary analysis of the procured literature oleo-chemical data indicated that the inclusion of land use change for this platform would have a significant impact. Figure 7 shows the GHG emission data of the oleo-chemical pathways included in this analysis with land use change and displacement included and Figure 8 shows the results when these variables were removed. As can be seen, the variance between results was significantly reduced when land use change and displacement were removed.

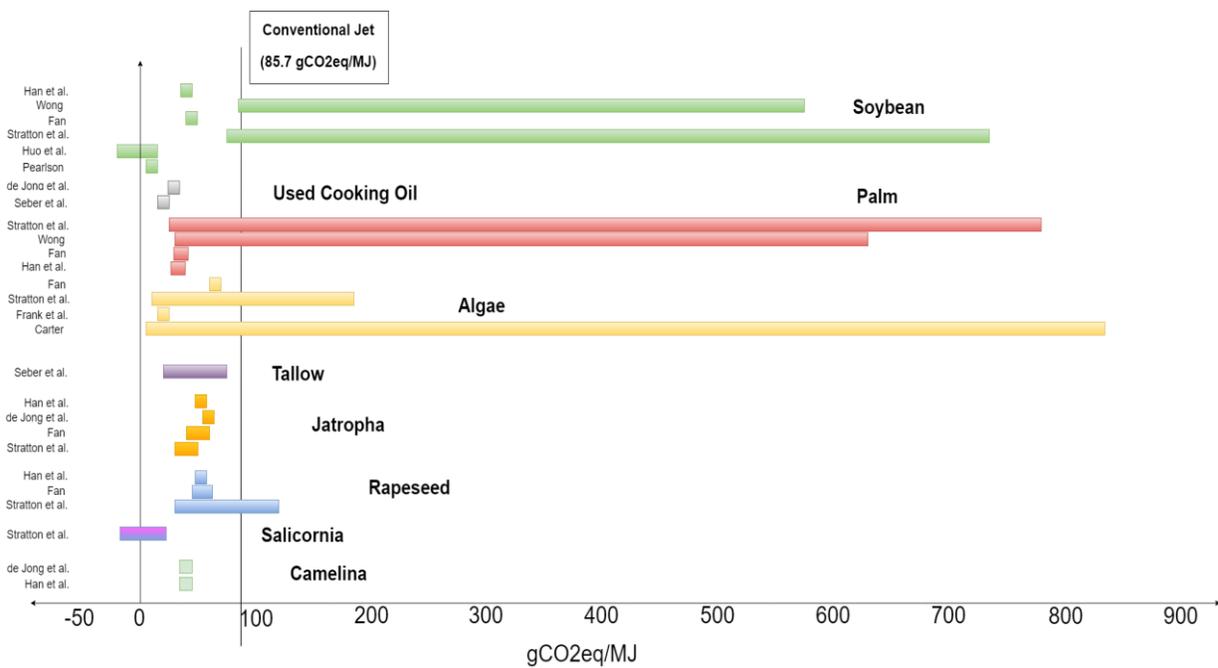
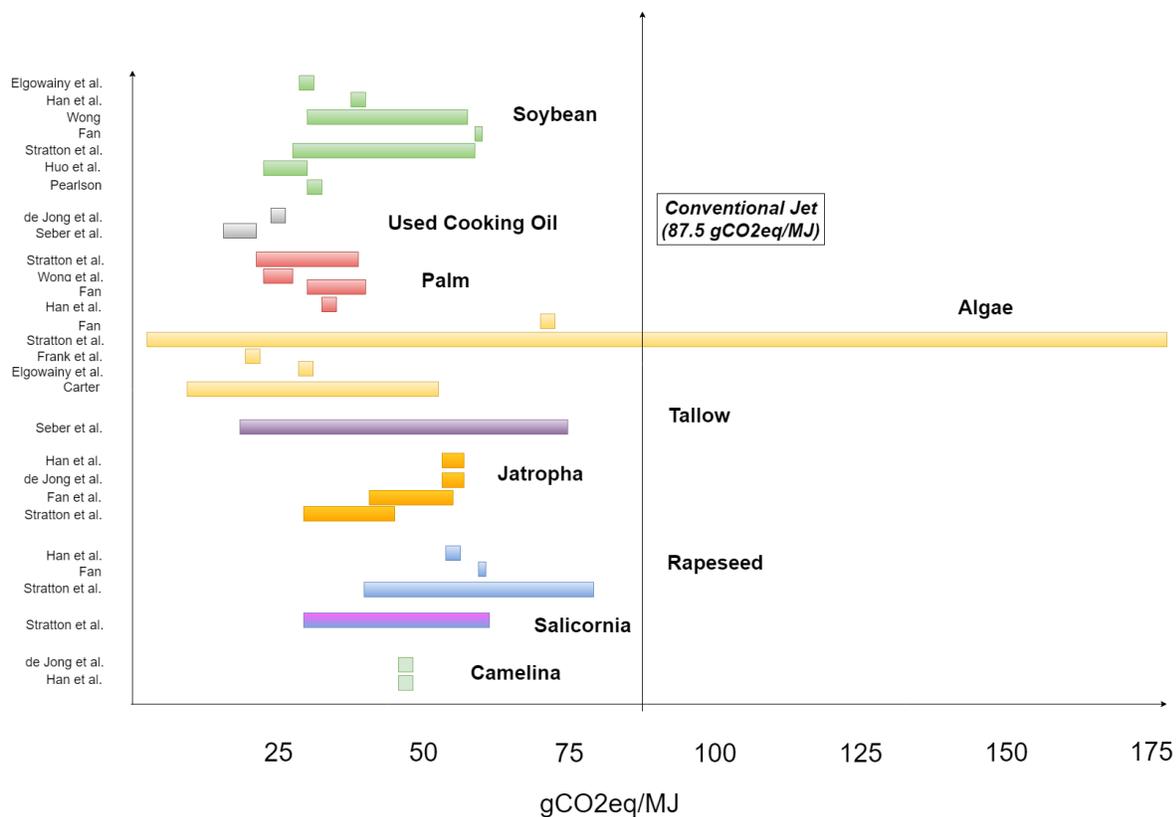


Figure 7: GHG emissions of oleo-chemical pathways from GREET literature



**Figure 8: GHG emissions of oleo-chemical pathways from GREET literature, excluding emissions from land use change and displacement co-product allocation**

Land use change also has the potential to impact lignocellulosic pathways such as gasification and pyrolysis. This impact can be seen in literature gasification studies using switchgrass or corn stover feedstock. Switchgrass is a perennial grass native to North America that can be found naturally along roadsides and in remnant prairies. As a replacement for annual crops, switchgrass has been shown to improve wildlife habitat and improve the organic carbon of soils [7]. The soil carbon sequestration is especially beneficial for the soils of degraded cropland, which is where the planting of this crop has its greatest potential [62]. When this grass is grown on previous cropland that has been cultivated for over 215 years, the sequestering potential was found to be approximately 0.17 Mg of C/acre/year over the course of 15 years in a study by Adler et al [68]. Other authors estimated an even higher SOC sequestration of 0.21 Mg/acre/year [69]. Stratton et al. [62] found that the inclusion of the soil carbon improvements of Adler et al. can lower the lifecycle GHG emissions of switchgrass by 19.8 gCO<sub>2</sub>eq/MJ (272%), making the pathway carbon negative (- 9.8 gCO<sub>2</sub>eq/MJ total). This inclusion far outweighs all other emissions, which is common for land use change impacts.

Corn stover is generally left on the field as a soil nutrient, so when it is removed, nitrogen and soil organic carbon levels drop significantly. A study by Sheehan et al. [70] determined that 2.2 tons/acre/yr of corn stover must remain on the ground to maintain soil carbon levels, while 1.1 tons/acre/yr is required when no-till agricultural management is employed. Under no-till practices (which are the norm for corn cultivation in the United States), the allowed removal is 1.65 dry tons/acre/year (based on average corn yields of 4.2 tons/acre/year) to ensure no loss in soil carbon levels [7]. If the soil carbon balance is not considered and all the available corn stover is removed (approximately 4.2 tons/acre/year), then soil carbon is reduced and the overall GHG emissions for corn stover gasification will increase from 7.8 gCO<sub>2</sub>eq/MJ to 12.4 gCO<sub>2</sub>eq/MJ [7].

The research reported in this thesis has focused on comparing the gate-to-wheel GHG emissions associated with various biojet manufacturing technologies, excluding the land use change element due to the large uncertainties and the significant impacts arising from its inclusion. Land use change was therefore removed from all studies whenever possible. This was the case for all data utilizing SimaPro and GREET, since land use change is not included by default within the model. However, all studies that utilize the GHGenius model include land use change since this is the model default, making it difficult to extract this variable from the results. Table 6 summarizes how the variables were harmonized for each model. Due to the difference in system boundary between models, it is important to take this variable into consideration whenever comparing data across models. Table 5 indicates the range of emissions that could be expected from land use change, though it is only provided as a rough estimate to facilitate model comparisons.

**Table 5: Additional GHG emissions from land use change impact**

<b>Pathway</b>	<b>LUC emissions (gCO<sub>2</sub>eq/MJ)</b>	<b>Low-range feedstocks</b>	<b>High-range feedstocks</b>
<b>Oleochemical</b>	-9 to +667	Canola [68]	Palm, Soy [7]
<b>Thermochemical</b>	-20 to +5	Switchgrass [65]	Corn Stover [7]

Functional units were also varied among literature, though the most common was gCO<sub>2</sub>eq/MJ. Because it is difficult to compare the results of studies utilizing different functional units, all results were standardized to this functional unit using standard or provided unit conversions.

**Table 6: Harmonization parameters for variables by Model**

<b>Variables</b>	<b>REET</b>	<b>SimaPro</b>	<b>GHGenius</b>
<b>Land Use Change</b>	Not Included	Not Included	Included
<b>Co-Product Allocation Method</b>	Energy/Market Allocation	Energy/Market/Mass Allocation	Displacement
<b>Functional Unit</b>	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ
<b>System Boundary</b>	Gate-to-wheel	Gate-to-wheel	Well-to-wheel

### 2.2.3 Experimental Procedure

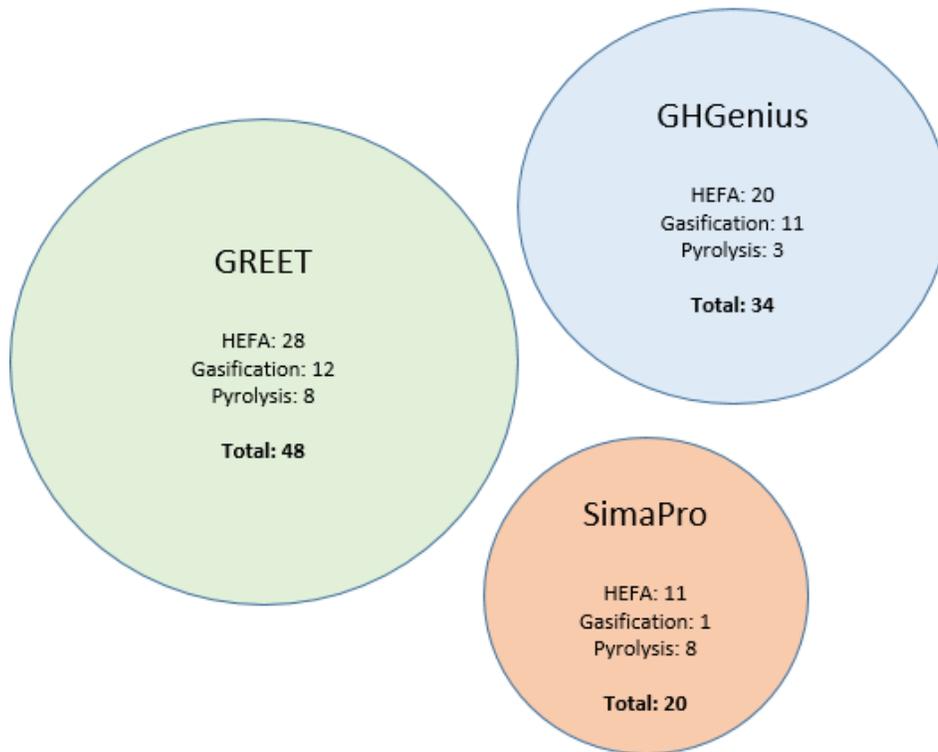
For this review, 34 papers and reports were included and compared. These papers were chosen from an initial pool of 468 LCA papers and reports obtained from a Boolean search of literature in Google Scholar, the UBC library, and model databases using the keywords “lifecycle analysis”, “lifecycle assessment,” “biojet or aviation fuel.” These were narrowed down to 72 papers after reviewing the content (only those containing significant LCA data were included) and then to 34 after the exclusion of studies based on internally produced models or models other than REET, GHGenius, and SimaPro, which were the most prevalent models used in the literature. The remaining 34 papers consisted of work from peer-reviewed journals, theses, and governmental reports and were sorted by model used. 102 studies were available within these 34 papers, since many authors analyzed more than one pathway. Figure 9 shows the distribution of pathways by model.

After sorting, study assumptions were tabulated for each variable and the study results recorded. The differences in study assumptions are discussed in section 2.4 and all variables were given an impact rating (low, medium, and high) based on historical evidence of impact on the LCA GHG emission results. This was done for each technological pathway of studies using the REET model, since this was the largest sample. Table 7 shows how much each variable must impact the overall GHG emission results to obtain a certain rating.

**Table 7: Lifecycle GHG emission change for each variable impact rating**

<b>Impact Rating</b>	<b>GHG Emission Change</b>
<b>Low</b>	<b>&lt; 10%</b>
<b>Medium</b>	<b>10-30%</b>
<b>High</b>	<b>&gt; 30%</b>

Appendix II contains details about the assumptions used within each literature study, while Appendix I provides the corresponding GHG emission reductions of each study. Some studies provided aggregate data for various processes or results for the same pathway when different assumptions were employed. This allowed adjustments to be made to some of the literature results, in order to harmonize the data within each model. Details of the nature of these adjustments is described in section 2.2. Any data which was altered is highlighted in Appendix I.



**Figure 9: LCA literature pathways sorted by Model**

#### 2.2.4 Data Analysis Strategy

Each study in the analysis provided either one data point or a range of data points that corresponded to the LCA GHG emission results of the associated pathway. A range of results occurred when the author considered multiple scenarios. These scenarios incorporated a range of possibilities for each variable, often labeled as a “worst case” or “best case”. The worst case typically used the lower published values for process data, such as process efficiency and product yield, while the best case included the higher values of the range. Sometimes these ranges represented different design scenarios: for example,

different sources of hydrogen or different heat recycle options. These data ranges could be caused by any variance in process variables analyzed by the study.

The results from literature studies were presented graphically for each model and pathway, grouped by feedstock (see Figures 10-12). In this thesis, a bar was used to represent the range of GHG emission results provided by an author whenever multiple scenarios were modeled. Some studies only considered one scenario and provided one data point: this data was represented by a very short bar. As previously discussed, any results that were obtained through the displacement allocation method using GREET or SimaPro were differentiated. This data was presented with an arrow pointing to the right to help compare it to other results, since using displacement tends to reduce the emission value (see section 2.2.2). GHGenius uses displacement by default, so all data from this model (Figure 12) was assumed to utilize this co-product methodology unless otherwise noted. Arrows were not included in Figure 12 to avoid redundancy but should be considered when comparing data between models.

#### 2.2.5 Limitations and Uncertainties

The use of LCA literature data rather than LCA data modelled specifically for this analysis creates some limitations. While using literature allowed a larger scope of technologies to be compared, it also prevented the absolute standardization of all variables. This made it difficult to be definitive about the exact cause of variation seen between the studies, as well as their precise impact on results. This limitation was addressed to some degree by the harmonization of high impact variables and the identification of the impact of other variables used within each study. By knowing the variables used by each study, most points of inconsistency could be identified. While this did not eliminate these variable inconsistencies from affecting the results, it provided insight into the possible causes of external variance inherent to the system. The preferred method of assessing these differences would be to perform LCAs for all oleochemical and thermochemical pathways using identical assumptions and variables, but this is extremely time-intensive and is therefore outside the scope of this thesis.

Another limitation in this work arose from the limited amount of biojet LCA studies in literature. For some feedstock pathways, only one or two studies could be found. This small sample size increased the uncertainty of these results and made it difficult for solid conclusions to be drawn. Furthermore, many of the studies utilized data from the same reports, so even the different data points available were not representative of all possible operating conditions. To eliminate this uncertainty, more studies need to be conducted in this area utilizing the largest spread of possible operating conditions.

Another issue with the LCA studies included in this report was that most were based on process modeling data rather than real industrial data. This is often necessary, since many of the advanced biojet technologies represented here, especially the thermochemical technologies, are not yet at the commercial stage. Many of the process models for pyrolysis and gasification were therefore based on data from pilot-scale studies, which may not consider scale-up and logistical challenges. This affects the confidence of the LCA results and should be updated as soon as demonstration or commercial-scale production data becomes available for these pathways.

Finally, the results of an LCA study using a model is not always indicative of the model itself. Many modelers changed the model defaults to their own values, so only some of the original and default assumptions and data were retained. While enough model data was typically retained to determine trends, the exact emission values should be used for comparison purposes only. It would be advantageous to compare all these studies to one using strictly model defaults, but this is outside the scope of this thesis.

### **2.3 Biojet Literature Results**

Of the 51 studies included in the review, 24 used the GREET model. Most of the literature focused on oleo-chemical pathways, but there were several papers that also modeled pyrolysis, gasification and hydrothermal liquefaction. The papers by Stratton et al. [7], Wong [7], and Fan [60] were extensive and included many studies of various feedstocks and pathways. Seber et al. [71] and Frank et al. [72] were more focused and considered fewer pathways. Figure 10 presents the GHG emission results from the literature lifecycle analysis studies that used the GREET model.

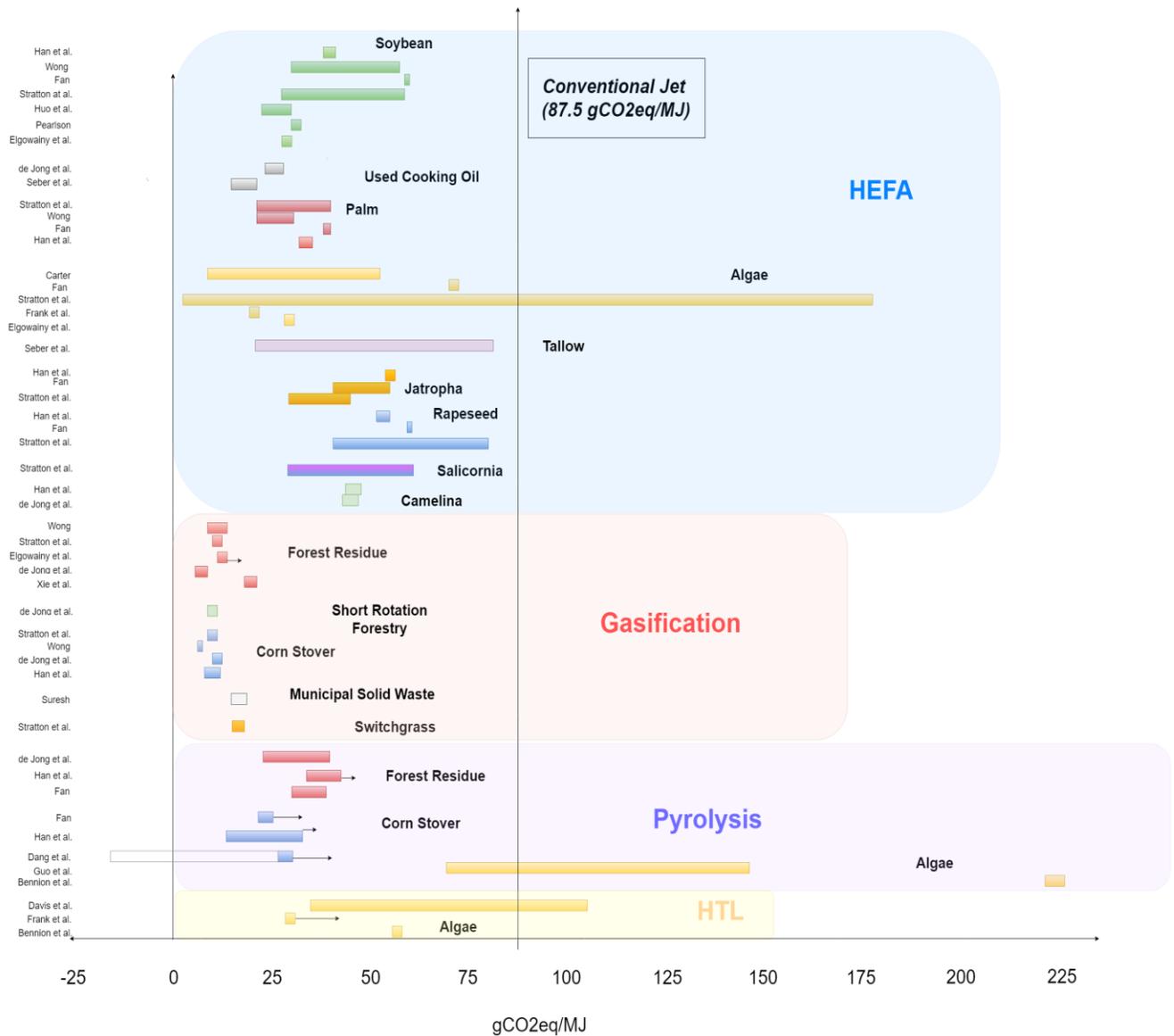


Figure 10: GHG literature results with GREET [16] [12] [7] [19] [20] [17] [18] [21] [22] [23]

The GHG emissions of conventional jet kerosene was included in all figures for comparison. The GHG emissions of this fuel range from 80.7 – 109.3 gCO<sub>2</sub>eq/MJ [62] depending on the energy efficiency of the aircraft, the weight of passengers and freight, the crude oil source and the refinery location and process. Most LCA literature considered in this study utilized a baseline value of around 87.5 gCO<sub>2</sub>eq/MJ, which is similar to the model default GHG emission values (see Table 2), so this value was used as the fossil fuel baseline for this review.

The LCA literature results that utilized the SimaPro model are presented in Figure 11. There were 13 LCA publications that analyzed the GHG emissions of renewable diesel and biojet using SimaPro, most of which analyzed the HEFA and pyrolysis pathways. Unlike the studies using GREET, most of these papers only analyzed one feedstock, with the exception of Fan [60] and Kalnes et al. [73]. Some platforms, such as gasification or HTL, have only one study and one feedstock. This made comparison between SimaPro and GREET results challenging, since it is hard to make solid conclusions based on one data point. The gasification comparison was further weakened because the feedstocks modeled were different than any of those analyzed with GREET. Other platforms, such as HEFA and pyrolysis, also included different feedstocks than those presented in Figure 10, such as corn oil and Eucalyptus.

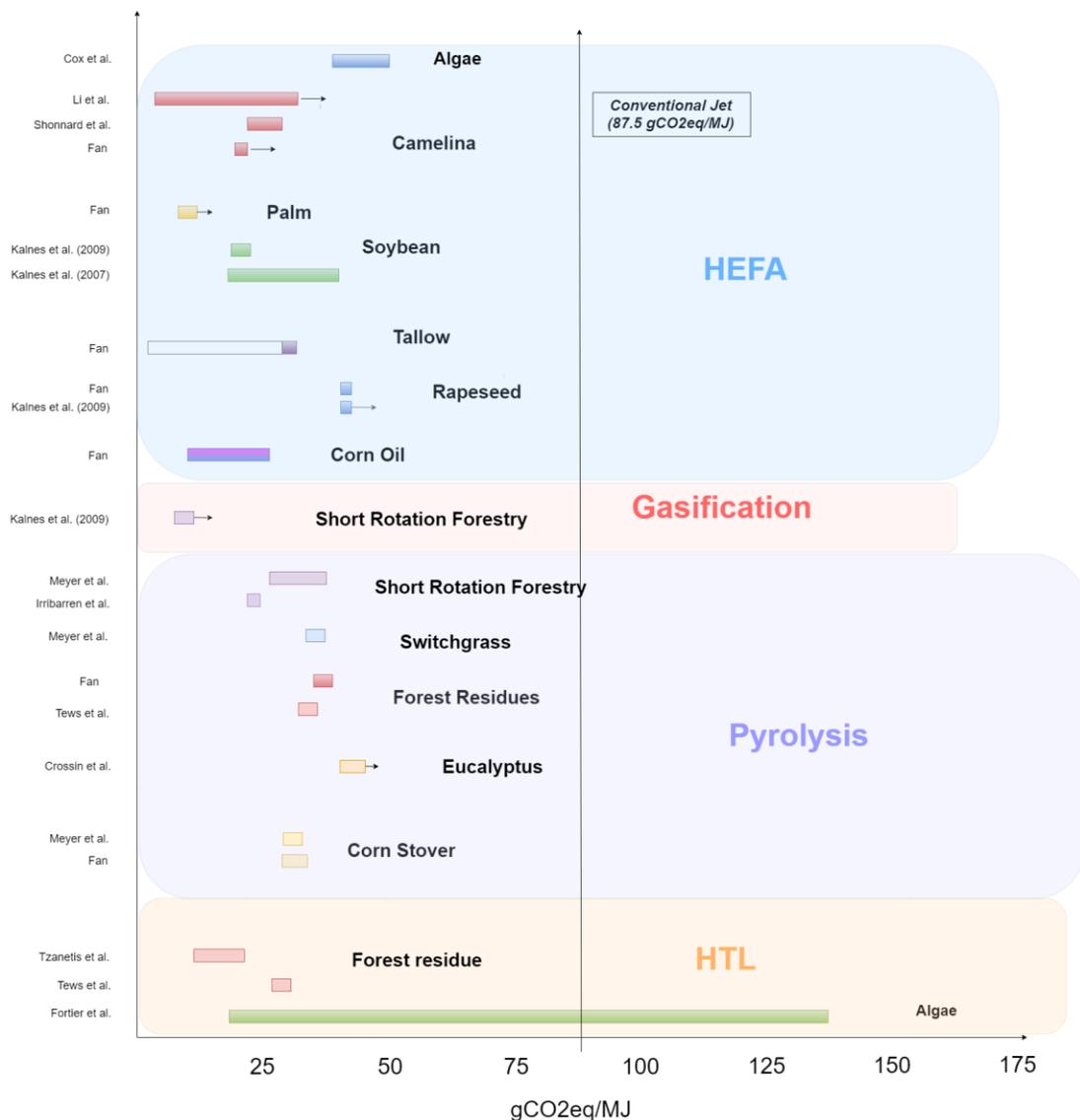


Figure 11: GHG Emissions of Studies using SimaPro [4] [8] [74] [73] [75] [76] [77]

Though there were an equivalent number of pathways modeled by GHGenius, there were considerably fewer authors. Other than a study by Fan [60], all of these studies were performed by (S&T)<sup>2</sup> Consultants for Natural Resources Canada, Novo Group, or the IEA (see Figure 12). Though most of these reports have not been peer-reviewed, they are very thorough and are clear regarding the assumptions and data used to obtain the results, so were included in this review. Most of the studies used Canadian data, except for the study by O'Connor [78] (in bold) which took place in the USA. All GHGenius studies used the displacement co-product allocation method, which is the default allocation method of the model. Land use change was also included to some extent in all of the studies, as this is also a GHGenius default and cannot be extracted. These two factors strongly affected the comparison of GHGenius results to those using other models and is discussed in further detail in section 2.5.

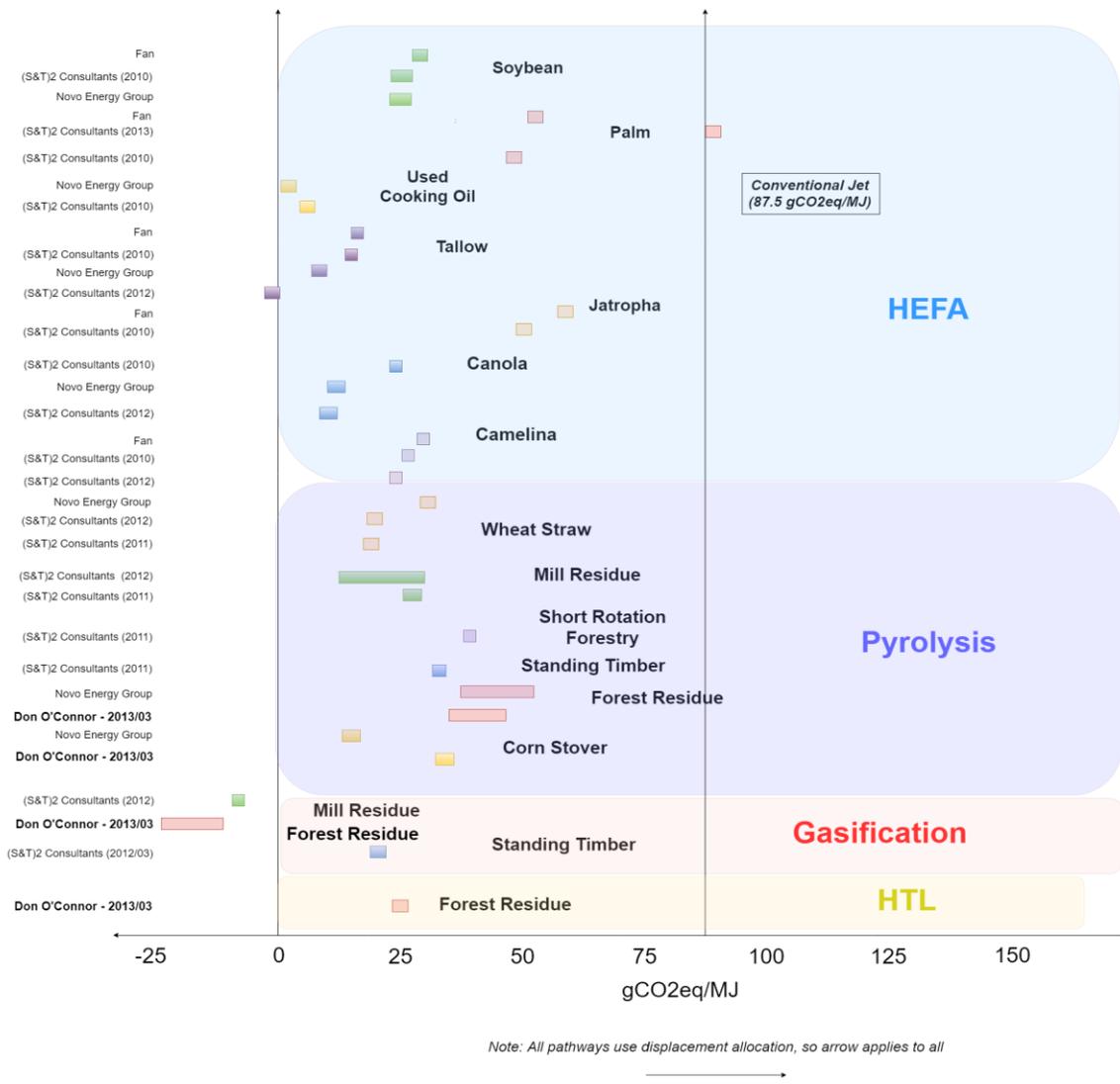


Figure 12: GHG Emissions of studies using GHGenius [60] [79] [80] [81] [82] [66] [83]

Upon first glance, the data presented in Figures 10-12 show a wide range of GHG emissions for each technology. As expected, the absolute GHG emissions are different between models, but some trends are apparent regarding technology reductions. However, before making any conclusions about the trends, the variables utilized by each study must be investigated to determine if there is any variance that could be contributing to the variance seen within the results. Since there is variability within the data presented for the same pathway within a model (e.g. GHGenius shows two different GHG values for pyrolysis of corn stover), it is likely that this is the case.

In Section 2.4, the variables from each study are quantified and their influence on LCA results investigated. Later, the GHG emissions of different biojet fuel pathways from all the GREET studies are compared utilizing these variable insights. Finally, results from different models are compared in Section 2.5 and rationale for these differences presented.

## 2.4 Literature Variable Analysis

In every LCA, an enormous amount of data is required. Information about the energy and mass flows of each process must be included, along with assumptions regarding how this data is to be handled and converted into overall GHG emissions. Figure 13 shows the relationships between typical variables and assumptions in LCA.

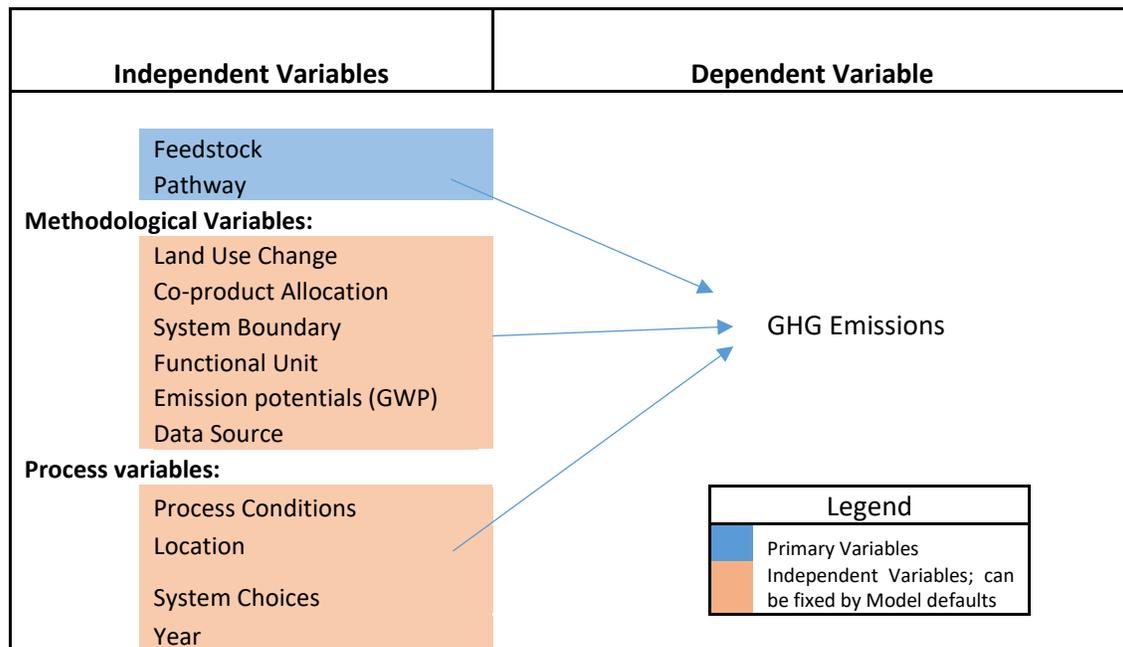


Figure 13: Relationship between process and methodological variables used in LCA

To provide insight into whether the studies in this review used the same variable inputs, the inputs to all the variables in Figure 13 were tabulated for each study and attached in Appendix II (Table 24 contains the inputs for all GREET studies, Table 25 contains the inputs for SimaPro studies, and Table 26 for GHGenius). By looking at these tables, it becomes apparent that the variable inputs are not identical between studies, even between those using the same model. This is likely because modelers altered some of the variables within their individual studies from the model defaults. Modelers will often add more specific primary data for the process variables, since this allows the modeler to model a specific plant or production facility rather than the national averages within the defaults of the model. However, as Tables 24-26 demonstrate, some modelers also change the methodological variables away from the model defaults (Table 2). This introduces an element of uncertainty when comparing the results computed by each model, as these results may not always be representative of how the model typically behaves.

To determine what effect these variable differences had upon the LCA results, the variables and their effect on GHG emission data of GREET studies are analyzed in detail below. GREET studies was chosen because GREET was the most commonly used model in literature, so had the most data points to facilitate the comparison. Table 8 shows the input ranges of each variable, separated by technological platform, and Table 9 shows the input modes, based on the information in Appendix II. After the analysis, variables are ranked from low to high based on their anticipated impact on the results.

Table 8: Input range of variables from GREET studies

Variables	Oleochemical	Gasification	Pyrolysis	HTL
<b>Model Version/Year</b>	GREET 1.8a (2008) – GREET.net v1.3.0 (2016)	GREET 1.8b (2008) – GREET.net v1.3.0 (2016)	GREET 2012 (2010) – GREET.net v1.3.0 (2016)	GREET 2011 – GREET 2013
<b>Cultivation N<sub>2</sub>O Emissions</b>	IPCC Tier 1 – GREET defaults	IPCC Tier 1 – GREET defaults; corn stover removal N <sub>2</sub> O credit >= additional fertilizer	GREET defaults; Corn stover removal N <sub>2</sub> O credit <= additional fertilizer	N/A
<b>Co-Product Allocation</b>	Energy, Market, Mass, Displacement hybrid	Energy, Market, Displacement hybrid	Energy, Energy/Displacement hybrid, Displacement	Energy
<b>System Boundary</b>	Renewable Diesel and Jet; most exclude wastewater treatment	Renewable Diesel and Jet	Renewable Diesel, Renewable Jet, Diesel/Jet mix	Renewable Diesel
<b>Functional Unit</b>	gCO <sub>2</sub> eq/MJ; gCO <sub>2</sub> eq/MMBTU; gCO <sub>2</sub> eq/kg-km; gCO <sub>2</sub> eq/MJ/passenger	gCO <sub>2</sub> eq/MJ; gCO <sub>2</sub> eq/kg-km; gCO <sub>2</sub> eq/passenger; gCO <sub>2</sub> eq/mmBTU	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ; gCO <sub>2</sub> eq/MMBTU
<b>Global Warming Potential</b>	IPCC 4 <sup>th</sup> assessment (1, 25, 298); GREET defaults	IPCC 4 <sup>th</sup> Assessment (1, 25, 298); GREET defaults	IPCC 4 <sup>th</sup> Assessment; GREET defaults	IPCC 4 <sup>th</sup> assessment
<b>Data Sources</b>	Various	Various	Various	Various
<b>Location</b>	USA average	USA	Michigan (USA), USA average, China	USA average
<b>Process Conditions</b>	Various	Various	Various	Various
<b>System Choices</b>	Various	Various	H <sub>2</sub> from multiple sources, Char used as either soil amendment, internally, or for electricity production	Continuous HTL, Energy Recycle, CHP

**Table 9: Input mode for variables in GREET studies**

<b>Variables</b>	<b>Oleochemical</b>	<b>Gasification</b>	<b>Pyrolysis</b>	<b>HTL</b>
<b>Model Version/Year</b>	2008/2011 (GREET 1.8/GREET 2012)	2008/2011 (GREET 1.8/GREET 2012)	2013 (GREET 2013)	2013 (GREET 2013)
<b>Cultivation N<sub>2</sub>O Emissions</b>	GREET defaults	GREET defaults; corn stover removal N <sub>2</sub> O credit > additional fertilizer	No agreement	N/A
<b>Co-Product Allocation</b>	Energy (fuels) + Market (non-fuel) hybrid	Energy; Energy/Displacement Hybrid	Energy	Energy
<b>System Boundary</b>	Renewable Jet fuel	Renewable Jet fuel	Renewable Jet fuel	Renewable Diesel
<b>Functional Unit</b>	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ	gCO <sub>2</sub> eq/MJ
<b>Global Warming Potential</b>	IPCC 4 <sup>th</sup> assessment (1, 25, 298)	IPCC 4 <sup>th</sup> Assessment (1, 25, 298)	IPCC 4 <sup>th</sup> Assessment	IPCC 4 <sup>th</sup> assessment
<b>Data Sources</b>	GREET database, literature, UOP operating data	GREET database, literature	GREET database, literature, PNNL and NREL reports	GREET database, literature, PNNL reports
<b>Location</b>	USA average	USA	USA average	USA average
<b>Process Conditions</b>	UOP operating conditions	Process efficiency of 45%	Various	Various
<b>System Choices</b>	Hydro-deoxygenation pathway maximized	Modeled with and without carbon capture and storage	H <sub>2</sub> from multiple sources; char as soil amendment	No agreement

#### 2.4.1 Model Version/ Year

Temporally, the studies ranged from 2008 to 2016 for all pathways, though some had a smaller range (e.g. HTL). Different models of GREET were used in these analyses, all containing different levels of detail, since data is updated with each model revision as new information becomes available. These revisions mean that even studies that rely solely on GREET defaults could be different if performed in different years. Process and fuel efficiencies are also expected to increase yearly, so studies performed in future years should provide lower GHG emission results. However, since the period is only 8 years, these effects are

likely minimal between the literature studies assessed here, so the effect of this variable on results is expected to be low.

#### 2.4.2 Co-Product Allocation

According to Table 9, most of the oleochemical studies utilized energy and market allocation for co-products. One study utilized mass allocation for one co-product (rapeseed seedmeal), which often provides a slightly lower result than other allocation methods for oleochemical pathways (see Table 3) [48]. Some studies compared these allocation techniques to those using displacement, but only the results using non-displacement techniques were included in the results of this study, since this is a high-impact assumption (see Section 2.2.2). The only exception is the result by Elgowainy [84] in which the non-energy co-products were modeled using the displacement method. The GHG emission result datapoint from this study was marked as all others using displacement with an arrow pointing to the right in Section 2.3.

With gasification, the co-product allocation method for most studies was energy allocation. Xie et al. [85] and de Jong et al. [28] modeled pathways utilizing multiple allocation methods, but only the energy allocation methods were used in this study. Displacement was partially used by Elgowainy et al. [84] as well but could not be removed from the analysis as no other allocation methods were modeled. In this study, displacement was used for the electricity co-product sold to the grid, which all other studies assumed was consumed internally (see System Choices for details) [84]. This assumption slightly reduces the overall emissions of the pathway by this author, so is represented with an arrow in Figure 10 as is customary.

For pyrolysis and HTL, energy allocation was used by all authors (when specified) except for Dang et al. [86], who used displacement [86]. For this pathway, the main co-products were char, electricity and steam. Most authors assumed that char would be used as a soil amendment, where 80% of the carbon is sequestered [60] [7]. Additionally, char can be burned to produce steam for electricity, which is the assumption used by Dang et al. [86] and Han et al. [20]. The other co-produced electricity and steam are used internally for process heat and power, so the co-product allocation method is not significant with these products. On the other hand, when the aqueous portion of the bio-oil was used to produce hydrogen in excess of what is required by the process, this hydrogen obtained a very large credit when the displacement allocation method was used. However, this impact is considered a system choice, so the impact of this assumption was accounted for under the system choice variable. Displacement was treated as customary and some information regarding its impact was gleaned from the results of Dang et al. who used displacement, which was  $-18.1 \text{ gCO}_2\text{eq/MJ}$ . Since only displacement can produce a negative result,

the impact of displacement in this case on the overall GHG emissions is at least 64%, making this variable high impact for these studies.

#### 2.4.3 Cultivation N<sub>2</sub>O Emissions

Not all LCA studies or models used the proposed formula by IPCC. In fact, most of the studies in this analysis that used GREET utilized GREET's defaults, which considers an N<sub>2</sub>O emission factor of 1.325% for both fertilizer and crop residue/below-ground biomass [61]. This differs from the IPCC Tier 1 equation, which assumes 1.325% of fertilizer volatilizes into N<sub>2</sub>O emissions, but only 1.225% of crop residue. This inconsistency in methodology can make an impact on overall GHG emission results, since the GWP of N<sub>2</sub>O is so large. This means that the N<sub>2</sub>O calculation for studies utilizing the GREET model default will be slightly higher than those that use the IPCC Tier 1 equation, especially for feedstocks that produce a large amount of crop residue (e.g. rapeseed, canola, soy and camelina) [10]. Though several of the studies did not indicate the N<sub>2</sub>O equation employed, it is expected that GREET defaults were used unless specified.

The only oleochemical studies that utilized other methods of calculation were those by Stratton et al. [7] and Wong [87]. The study by Stratton et al. [7] considered the full IPCC Tier 1 equation, but only for certain scenarios and crops. For example, the full IPCC equation was considered for rapeseed, but for soy it was not used at all. With soy, no nitrogen fertilizer was assumed in the low case, the GREET defaults were utilized for the baseline, and a significantly higher N<sub>2</sub>O estimate from literature was used for the high case [7]. For palm, however, the IPCC Tier 1 equation was considered for all cases, though crop residue was not included due to lack of data.

Wong [48] also used different N<sub>2</sub>O accounting methodologies in his analysis. For example, Wong used the GREET defaults when calculating emissions from soy, but only used a factor of 1% for palm multiplied by the applied fertilizer (like Stratton et al. [62], crop residues were excluded). However, palm oil comes from perennial tree plantations, where very little crop residue is produced other than co-product residues from the fruits. As such, this exclusion would likely not have a large effect on this pathway.

For gasification and pyrolysis pathways, cultivation N<sub>2</sub>O emissions are only relevant for switchgrass, short rotation forestry, and corn stover. Forest residue and municipal solid waste (MSW) are considered wastes and thereby do not assume any N<sub>2</sub>O emissions during cultivation. Furthermore, forest residues generally come from previously logged forest stands where fertilizer has not been applied. All nitrogen fertilizers added to the algae water system were assumed to assimilate into the algae [88]. Corn stover is also considered a waste but is generally left on the field after corn harvest to prevent erosion and replenish soil nutrients. When corn stover is removed for fuel purposes, the soil not only loses organic carbon, but

nitrogen as well. Therefore, most studies assumed that nitrogen fertilizer was applied to make up for this loss, based on how much corn stover was removed. The GREET model also uses this assumption as default. As corn stover decay on the field is also a source of N<sub>2</sub>O emissions, the GREET model gives a credit for the N<sub>2</sub>O emissions avoided due to corn stover removal. According to the model defaults, the N<sub>2</sub>O emissions credit from corn stover removal is slightly larger than that of the additional fertilizer [7]. This gives an overall N<sub>2</sub>O cultivation emission of -0.3 gCO<sub>2</sub>eq/MJ for corn stover cultivation [63]. This assumption was utilized by most authors in this review, but some authors such as Stratton et al. [7] and Han et al. [20] utilized different N<sub>2</sub>O emission values in their calculations (+1 gCO<sub>2</sub>eq/MJ and 0 gCO<sub>2</sub>eq/MJ, respectively). Since these calculation differences added 5-11% to the total GHG emissions, this variable was considered low-medium impact for the gasification and pyrolysis pathways. Since all HTL pathways modeled algae, this variable was not considered, so its impact on GHG emissions could not be determined for this platform.

While the ramifications of corn stover removal on soil health is important, it is unclear whether using fertilizer to compensate for these losses is practical. This assumption has been incorporated into all LCA studies, but it is unclear whether this practice has been justified in the field. This practice would likely be dependent upon the relative price of fertilizer and corn stover residue and may require the development of new harvesting methods. The likelihood of the adoption of this practice should therefore be determined before inclusion as a general LCA assumption, but this is outside the scope of this thesis.

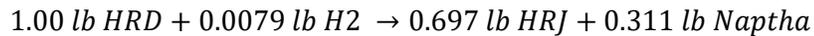
#### 2.4.4 System Boundary

Most of the papers included in this review modeled the full lifecycle of biojet fuel, but some papers only modeled the LCA of renewable diesel. For oleochemical pathways, this means that they do not include the extra step necessary to produce biojet from renewable diesel. Most vegetable oils produce a predominately diesel cut after undergoing hydrogenation because they are comprised of mainly C18 length fatty acids, which are in diesel range (C16-C20). These hydrocarbons must be hydrocracked to qualify as jet fuel (C8-C16). However, fatty acids from some oils such as palm kernel oil are already in the jet fuel range, so may not require this step (see Table 10).

**Table 10: Fatty acid profiles of vegetable oils (%) [20] [7]**

	C8:0	C10:0	C12:0	C14:0	C16:0	C18:0	C18:1	C18:2	C18:3	C20:1	C22:1
<b>Soy</b>					11.0	4.0	22.0	53.0	8.0		
<b>Palm</b>	0.3	0.6	4.3	1.3	40.8	3.7	37.2	10.1			
<b>Rapeseed</b>					3.0	1.0	17.0	14.0	9.0	11.0	45.0
<b>Jatropha</b>					13.0	8.0	45.0	34.0			
<b>Camelina</b>					7.8	3.0	16.8	23.0	31.2	12.0	2.8
<b>Palm Kernel</b>	3.0	7.0	47.0	14.0	9.0	1.0	19.0	1.0			

Kalnes et al. [73] indicated that the additional hydrocracking required to split the C18 to C12 and C13 jet range molecules adds about 10% natural gas and 30% electricity per pound of renewable product. They determined that the hydrogen requirement also increased according to the following equation [73]:



Without this step, 65 wt% of the incoming vegetable oil was converted to diesel-range molecules and only 13 wt% was converted into jet [7]. Increasing the jet fraction to 50 wt% required about 30% more hydrogen and the overall yield dropped from 84% to less than 60% [10]. Though this extra step requires substantially more hydrogen, the results from the LCA studies on biojet are not significantly different than those of renewable diesel. According to Stratton et al. [7], the additional GHG emissions added by the inclusion of this step is only about 1.5 gCO<sub>2</sub>eq/MJ. For this reason, most authors did not include these additional emissions for biojet fuel production within the LCA system boundary. Some assumed that renewable diesel and renewable jet had the same emission profile [63] [84], while others included the additional emissions in their analysis [7]. The study by Pearlson [21] calculated scenarios where each product was maximized, realizing that there will always be a mix. As the difference between GHG emission results of renewable diesel and renewable jet appears to be minimal, studies that model renewable diesel are also included in the analysis and this difference is assumed to be low [72] [89].

It is also important to note that most of the authors did not calculate the different hydrogen requirements for using unsaturated oils (such as soy and rapeseed) from saturated oils (palm and tallow) Table 10 provides the level of saturation for each of the modeled triglycerides. All processes in this study were modeled after the UOP or Neste process given by Pearlson [21], which determined the hydrogen requirements for HEFA using unsaturated soy oil [21]. GHG emissions are therefore overestimated for saturated oils such as tallow and palm. The only authors that took this effect into account were Han et al. [20]. Han et al. [20] found that by considering the level of unsaturation in the triglyceride, the process

emissions due to hydrogen consumption were reduced by 2.2 gCO<sub>2</sub>eq/MJ for palm. Since this increase could have an impact on results, it was suggested that this impact be included in all LCA studies that model oleochemical pathways. However, for this review this difference will only be relevant for comparisons including Han et al. [20].

The inclusion of wastewater treatment is another boundary decision that can affect GHG results. Most of the data used in the LCA for the soybean and rapeseed pathways came from industry energy use reports. These reports generally include wastewater treatment energy and emissions because it is treated onsite, as most commercial plants are remote [90]. However, if the facility were close to a municipal wastewater treatment facility, it is likely that the water would be sent away for treatment and these emissions would not be captured in the report. Since most studies used the energy and emission use of soy to represent all HEFA production [7] [21], the wastewater treatment emissions from production should be captured for all oleochemical pathways. Furthermore, industrial reports indicate that wastewater treatment from fuel production only contributes about 1.5% to 1.7% of total energy use, so the contributions of these emissions is small [90]. However, upstream emissions can be quite substantial whenever water or steam is used during oilseed pressing. For example, palm oil is extracted from the seeds of fresh fruit bunches using steam [66]. The waste water contains residual amounts of organic matter, so must be treated in an anaerobic digester before release. While methane is generally captured from digesters in North America, this practice has only been adopted at newer, more advanced facilities in Indonesia and Malaysia where most of the world's palm oil is produced [66]. While these emissions are included within the defaults of the GHGenius model, they only appear in a few GREET studies. Stratton et al. [7] included between 3.3 and 5.0 kg methane/lb of oil, which is equivalent to about 5.2 to 7.4 gCO<sub>2</sub>eq/MJ (17 – 25%). According to the EPA, these emissions can add as much as 41.71 CO<sub>2</sub>eq/MJ to the overall emissions, so can substantially increase emissions [60]. While the GREET studies handled this variable similarly, more research must be conducted to determine which values are more accurate. As this variable contributes over 10% to total lifecycle emissions, it was considered a medium impact variable in this analysis.

The inclusion of liming emissions in the lifecycle analysis also has not reached a consensus. Both Wong [48] and Stratton et al. [62] included liming emissions in their high emissions scenarios, but the GREET software does not include any emissions from the liming process as default [87]. Although the additional emissions seem high (+2,347 gCO<sub>2</sub>eq/bushel of soybean), it only adds about 5 gCO<sub>2</sub>eq/MJ (~6%) to the overall LCA emissions of the fuel, so was considered low impact [63].

Unlike the HEFA process, the Fischer-Tropsch upgrading process of syngas creates a wide range of hydrocarbon molecules. Depending upon the process conditions used, the Fischer Tropsch process can be adjusted to target either longer-chain diesel or the slightly shorter jet. Regardless of which is targeted, a distribution of longer and shorter molecules will also be created, according to the Anderson-Schulz-Flory distribution model. The maximum volume of middle distillates that can be produced (jet and diesel) is about 40% [91]. Additional hydrocracking of the longer hydrocarbons is generally employed to increase the diesel and jet fraction. To improve the jet fuel fraction over diesel, a greater amount of hydrocracking is generally required, as well as increased syngas recycling. This slightly increases the hydrogen and power requirements of the plant [84].

In this analysis, about half of the studies modeled the LCA of FT-diesel, while the other half modeled FT-jet. However, upon closer inspection, there was no difference between the two. Those that modeled FT-Jet (Elgowainy et al. [84], Stratton et al. [7], and Wong [87]) argued that although there was an increase in the hydrogen and power requirements of the plant when jet was targeted over diesel, there was also a reduction in CO<sub>2</sub>, since more of the CO<sub>2</sub> was allocated to the diesel. These studies assumed that these trade-offs would negate each other and make any differences between FT-diesel and FT-jet CO<sub>2</sub> emissions negligible. In 2011, Argonne National Laboratories included a new jet module in GREET which was based on the work by Elgowainy et al. [84], which means that the FT-jet in the model is mostly synonymous with FT-diesel. de Jong et al. [28] also modeled the LCA of FT-jet, but rather than assume the negation strategy of the other authors, he assumed a diesel fraction that was 25% jet. How these different process yields (50% vs. 25%) might affect the lifecycle GHG emissions falls under the process conditions. While system boundaries indirectly affect this variable, for the sake of simplicity no indirect effects were considered in this analysis, making the impact of the system boundary variable zero.

For pyrolysis, only three of the studies modeled the emissions for jet fuel. All the other studies either modeled the emissions for renewable diesel or a combination of diesel and gasoline (generally 50/50).

As previously discussed, most diesel can be catalytically cracked to obtain hydrocarbon molecules in the jet range. However, this is rarely done on all the diesel fraction, as the selectivity is poor and cracked products range in size from C<sub>3</sub> to C<sub>15</sub> rather than just around C<sub>12</sub> [85]. So, while a process can be developed to maximize jet, this would be unusual as long-chain hydrocarbons are generally more valuable. de Jong et al. [28] modeled jet fuel from pyrolysis but assumed the jet fraction was only 25% of the diesel cut, giving a product slate of 7% jet, 21% diesel, 56% gasoline, and 16% heavy oil. This is substantially different than the 50/50 gasoline to diesel assumption of Dang et al. [86] and others. While this would be

expected to make a big difference on overall GHG emissions, very little difference is made when energy allocation is used. This is because the emissions are divided between the fuel products (jet, diesel, gasoline) based on their energy contents, which are very similar (see Appendix III). Even studies that use the displacement method generally use energy allocation to divide emissions between the fuel products [86]. As such, there is not much difference between studies that model renewable jet or renewable diesel/gasoline for pyrolysis processes, so this variable has minimal impact on results.

#### 2.4.5 Functional Unit

The functional unit used by most papers is gCO<sub>2</sub>eq/MJ, so all variables were converted to this functional unit using conversion information from within the source when possible. This conversion is not optional, since it is not possible to compare LCA results using different functional unit metrics. The study by Frank et al. [72] published the results in gCO<sub>2</sub>eq/MMBTU, which were converted into gCO<sub>2</sub>eq/MJ for presentation in this thesis. Other studies publish the gCO<sub>2</sub>eq in terms of kg-km, passenger-km and MJ, but only the results in terms of MJ are used in this study. All variance was removed for this variable for these studies, thereby reducing the variable impact to zero.

#### 2.4.6 Global Warming Potential (GWP)

The Global Warming Potential used by most papers is that of the IPCC 4<sup>th</sup> assessment report, though some used GREET defaults. In the IPCC report, the GWP values for CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O were 1, 25, and 298, respectively [42]. These values were published in 2007, so it is likely that the GREET model was updated with these values when it was revised in 2007 and most definitely by 2011 [84]. All the studies that utilized GREET defaults or did not specify any GWP accounting method were conducted after 2011, so it is likely that they utilized the IPCC 4<sup>th</sup> assessment values. Though a new set of IPCC values was published in 2014, these values were not used by any of the studies after 2014 included in this analysis. Therefore, all studies likely utilized the same GWP, eliminating variance within this variable.

#### 2.4.7 Data Source

All the studies included in this analysis draw upon both literature and GREET defaults for data. Many utilized different pieces of literature, though there is some overlap. Some even cited each other as references. For example, the report by Stratton [62] used the thesis by Wong [48] as the source for much of its data [10]. The thesis by Pearlson [21] was often used by other authors to obtain information about the UOP process [21]. For gasification, Han et al. [20] use Stratton et al. [62] as a source, and both Han et al. [20] and Elgowainy et al. [84] use Huo et al. [61]. Due to the wide range of literature data seen within the oleochemical studies, it is difficult to determine the impact of this variable on results, so it was deemed indeterminate. For gasification, many of the studies utilize the same data, since data is only available from

a few pilot and demonstration projects [20]. For this reason, the impact of this variable on the GHG emissions was considered low for gasification pathways. This is similar for pyrolysis studies, which only have a few government pilot scale studies to draw from [92] [93]. While some studies rely on other literature, most pyrolysis studies obtained their process data from these governmental reports. As a result, the impact from this variable was also considered low.

#### 2.4.8 Process Conditions

Though all the oleochemical studies modeled the same technology, they drew from different data sources for process inputs. Data from these sources were used to determine the input ranges for process variables such as process yield, electricity requirements, natural gas use, water requirement, solvent requirements, oil pressing yield, fertilizer requirements, cultivated yield, co-product yield, etc. As was discussed above, many of the studies used similar data. Most drew primarily from UOP data, either provided by UOP or Pearlson [21]. They also drew heavily upon the data stored within GREET. Similar to the data source variable, this variable has some variance between studies that could influence GHG emission results, though the exact effect is indeterminate for oleochemical studies.

Within the gasification studies, Wong, Stratton, and Elgowainy all modeled the process used by CHOREN. At the time of these reports, CHOREN was the only promising Biomass-to-liquids (BTL) facility under production [94]. In 2008, it was operating a 300 bbl/day facility and had announced plans for a larger facility to be built in Finland [94]. The proprietary CHOREN technology favors the production of long-chain waxy hydrocarbons to reduce the gaseous fraction. These waxes are then subject to hydro-isomerization and hydrocracking in order to produce middle distillates. However, Elgowainy et al. [84] also used information from the plans of Rentech and Solena of a 1800 bbl/day BTL plant in California and Xie et al. [85] used data from a study by Kreutz et al. [95] that obtained data by computer modeling. Since it is unclear how these facilities differ, the effect of this variable on these studies was indeterminate.

For gasification, the process yield of a pathway has some impact on overall GHG results as well. Elgowainy et al. [84] assumed a process efficiency of 50% (GREET default) to FT products and Stratton and Wong assumed an efficiency of 45%. In the case of gasification from forest residue, Wong [87] demonstrated that a 5% decrease in process efficiency resulted in an 8% increase in lifecycle GHG emissions (see Table 11). Since this increase is below 10%, the impact of this variable was considered low.

**Table 11: Variation of Life-cycle GHG Emissions with Fischer-Tropsch Process Efficiency (forest residue) [82]**

Process efficiency (%)	Mass ratio of feedstock to F-T Fuel	Life-cycle GHG emissions (gCO <sub>2</sub> eq/MJ)
40	6.8:1	13
45	6.8:1	12
50	6.8:1	11
55	6.8:1	10
60	6.8:1	9

For pyrolysis studies, most of the process conditions were similar because they all modeled the pyrolysis processes of PNNL, NREL and/or the UOP process for upgrading. There were generally two yields given: that of the bio-oil from feedstock and that of bio-oil to fuel. Dang et al. [86] gave a 71.6% yield for bio-oil from the feedstock and an overall yield of 43-16%, the lower end occurring when the aqueous bio-oil layer was used to produce hydrogen, which lowered the overall yield as it consumed bio-oil that would otherwise go toward the fuel product. Other authors calculated yields around 35%, which is within this range [20]. While decreasing the overall yield does increase overall GHG emissions, these additions were low. O'Connor [78] found that decreasing the overall yield by 50% only increased emissions by 1.5 gCO<sub>2</sub>eq/MJ (~3%). However, increasing the hydrogen requirements by 50% increased the overall emissions by about 10 gCO<sub>2</sub>eq/MJ (~24%) [78]. The insensitivity of the overall GHG emissions to yield is common whenever the feedstock emissions are low, which is the case with most pyrolysis and gasification feedstocks. Therefore, the impact of process yield on GHG emissions for pyrolysis pathways was considered low, but the impact of hydrogen requirements on this pathway was considered medium.

#### 2.4.9 System Choices

Most of the gasification studies modeled the UOP process, which favors the decarboxylation mechanism to biojet [21]. However, Huo [89] modeled both the UOP process and Canmet's "High Cetane" process for making HEFA fuel. These two processes differ mostly in their co-product output and overall yields. Since co-product output has a large impact when displacement is used, the GHG process emissions of these two processes would likely differ greatly when the displacement co-product allocation is used. Since only the energy and market allocation results were presented for Huo [89], the differences between these processes was considered to likely be medium to low.

Within the gasification studies, all modeled the recycle design for Fischer-Tropsch production. The RD design recycles the unconverted syngas, which increases the product yield but reduces the co-produced

electricity [85]. Stratton et al. [7], Wong [87], and Elgowainy et al. [84] assumed that the recycle is such that there is no excess electricity, while Xie et al. [85] assumed that 11% of the product-share is electricity. This means that Xie et al. [85] obtained a credit for this co-product, which is assumed to displace the grid electricity, while the other studies did not. While this could have a significant impact if displacement allocation were used, the energy allocation used by this study lowers the impact of this difference. However, due to lack of detailed information in the Xie et al. [85] study, its exact impact was indeterminate.

The main difference between the pyrolysis studies using GREET is the source of hydrogen. Han et al. [96] and Guo et al. [88] assumed that all the hydrogen was obtained via the steam methane reforming (SMR) of methane. This process is the most traditional and opens up the co-produced biogas to dry the feedstock and produce electricity, obtaining a co-product credit. All char co-product is generally used to dry the feedstock but is considered a soil amendment or electricity feedstock if some remains. Han et al. [20] found that utilizing the biochar product as a soil amendment, which is a form of carbon sequestration, gave greater GHG emission reductions than when it was used to produce electricity, even while using energy allocation (76% reduction rather than 68%) [20]. In this review, most authors (J. Fan, F. Guo and J. Han) assumed that excess char was used as a soil additive [60] [88] [20]. Only de Jong et al. [28] and one scenario from Han et al. [20] assumed that the excess char was combusted to produce steam-driven electricity.

While SMR is the most common method of producing hydrogen, some authors considered alternate methods. For example, Fan [60] and de Jong et al. [28] also considered scenarios where the bio-gas produced during pyrolysis was used to offset some of the methane required for hydrogen production. This is the method followed by PNNL and NREL [92] [93] and reduces the amount of external methane required. According to these authors, using biogas for SMR lowered overall GHG emissions by about 2 gCO<sub>2</sub>eq/MJ (10%).

In addition to modeling a scenario where all the hydrogen is obtained from biogas, Dang et al. [86] also modeled scenarios where a portion or all of the aqueous bio-oil was used to make hydrogen. The situation where only a portion of the aqueous bio-oil (roughly half) was used to produce hydrogen satisfied all of the hydrogen requirements of the process so that no additional hydrogen was needed and reduced GHG emissions by about 13% [86]. This was based off of a design by Virent Energy Systems called Aqueous Phase Reforming (AQR) and was employed by UOP [97] [98]. When all of the aqueous bio-oil is used to produce hydrogen, excess hydrogen is produced and can be sold to the grid. This gave substantial GHG

improvements when modeled by Dang et al. [86](-162%), especially since the study also used displacement. It is unclear how the use of all the aqueous bio-oil for hydrogen production would compare to the other scenarios if a different type of allocation method were used, but it would likely be greater than 30%. The impact of hydrogen source on the overall GHG emission results was therefore considered high.

While most studies considered that the hydrotreatment upgrading took place on site with pyrolysis, some studies assumed that the bio-oil upgrading occurred at a separate facility. While the latter scenario requires an additional hydrotreating step to stabilize the pyrolysis oil for transport, Fan [60] found that the additional emissions this added to the overall LCA are minimal. For forest residue feedstock, this extra step added negligible emissions (less than 1 gCO<sub>2</sub>eq/MJ), while it added about 3 – 4 gCO<sub>2</sub>eq/MJ (11%) to the overall GHG emissions of corn stover feedstocks [60]. This variable was thus considered to have a minimal impact for forest residue, but a medium impact for corn stover.

#### 2.4.5 Conclusions

A comprehensive, systematic review of the variables and assumptions used within GREET biojet fuel literature indicated that while all LCA variables affect the overall result, there are several that make the largest difference. Furthermore, it appears that these variances affect each technological platform differently (see Table 12). For example, the preliminary analysis highlighted the strong influence that land use change and the co-product allocation method, especially displacement, had on results. While these variables affected all pathways, they had the strongest impact on cultivated pathways that produced a large amount of co-product. The oleo-chemical pathways in this review utilized mainly cultivated crops as feedstock, which in turn produced a large amount of seedmeal co-product. As a result, land use change and displacement assumptions strongly affected the results from these pathways. Land use change generally added a significant amount of GHG emissions to cultivated feedstocks as well, except for a few crops that improved the carbon content of soil (canola, switchgrass, salicornia). The GHG emissions of these SOC enhancing crops and grasses were found to reduce with the inclusion of land use change. Displacement had a tendency to lower the GHG emissions of all pathways, though it had the greatest impact on oleochemical and pyrolysis pathways due to the large number of co-products produced in these pathways. Table 12 provides a summary of the impact LCA variables had on the literature studies of each pathway.

**Table 12: Summary of variable impacts for most pathways included in review (GREET literature)**

<b>Variables</b>	<b>Oleochemical Impact</b>	<b>Gasification Impact</b>	<b>Pyrolysis Impact</b>
<b>Model Version/Year</b>	Low	Low	Low
<b>Land Use Change<sup>1</sup></b>	High	High (switchgrass)	NA
<b>Cultivation N<sub>2</sub>O Emissions</b>	Medium	Low- Medium	Low - Medium
<b>Co-Product Allocation<sup>1</sup></b>	High (displacement); Low (others)	Low – Medium (displacement)	High (displacement)
<b>System Boundary</b>	Low (Diesel/Jet; liming); Medium (palm wastewater)	NA	Low
<b>Functional Unit</b>	NA	NA	NA
<b>Global Warming Potential</b>	NA	NA	NA
<b>Data Sources</b>	Indeterminate	Low	Low
<b>Location</b>	NA	NA	NA
<b>Process Conditions</b>	Indeterminate	Low (process efficiency); Indeterminate (technology)	Low (process yield); Medium (hydrogen yield)
<b>System Choices</b>	Medium - Low	Indeterminate	High (hydrogen source); Low-Medium (facility integration)

<sup>1</sup>Determined during preliminary analysis

Unlike gasification, pyrolysis requires a large amount of hydrogen, as pyrolysis oil is very high in oxygen. This technological pathway requires more hydrogen than any other pathway, which made the source of hydrogen a highly impactful variable for the pyrolysis pathways. Hydrogen is typically produced from methane via steam methane reforming, but it is also possible to produce from electrolysis, biogas, or the bio-oil itself. While none of the studies here considered the use of hydrogen from electrolysis, using the aqueous bio-oil through Aqueous Phase Reforming (APR) for hydrogen production significantly reduced overall GHG emissions, especially when excess hydrogen was produced. For thermochemical processes, the yield of co-products had a significant effect on overall GHG emissions, even when displacement was not used.

Variables that had a medium to low impact included N<sub>2</sub>O emissions for all pathways, co-product allocation for gasification, the system boundary for oleochemical pathways, and system choices for all but gasification. These variables and those with high impact were given special consideration when comparing the GHG emission results from literature, since variance between results will likely stem from these variables. All variables designated as high impact (co-product allocation, LUC, and hydrogen source for

pyrolysis) were either removed or the affected data marked. As literature results are compared in the following two sections, any data affected by medium to high impact variables is examined to determine whether these variables could be affecting the differences seen between feedstocks, technological platforms, and model results.

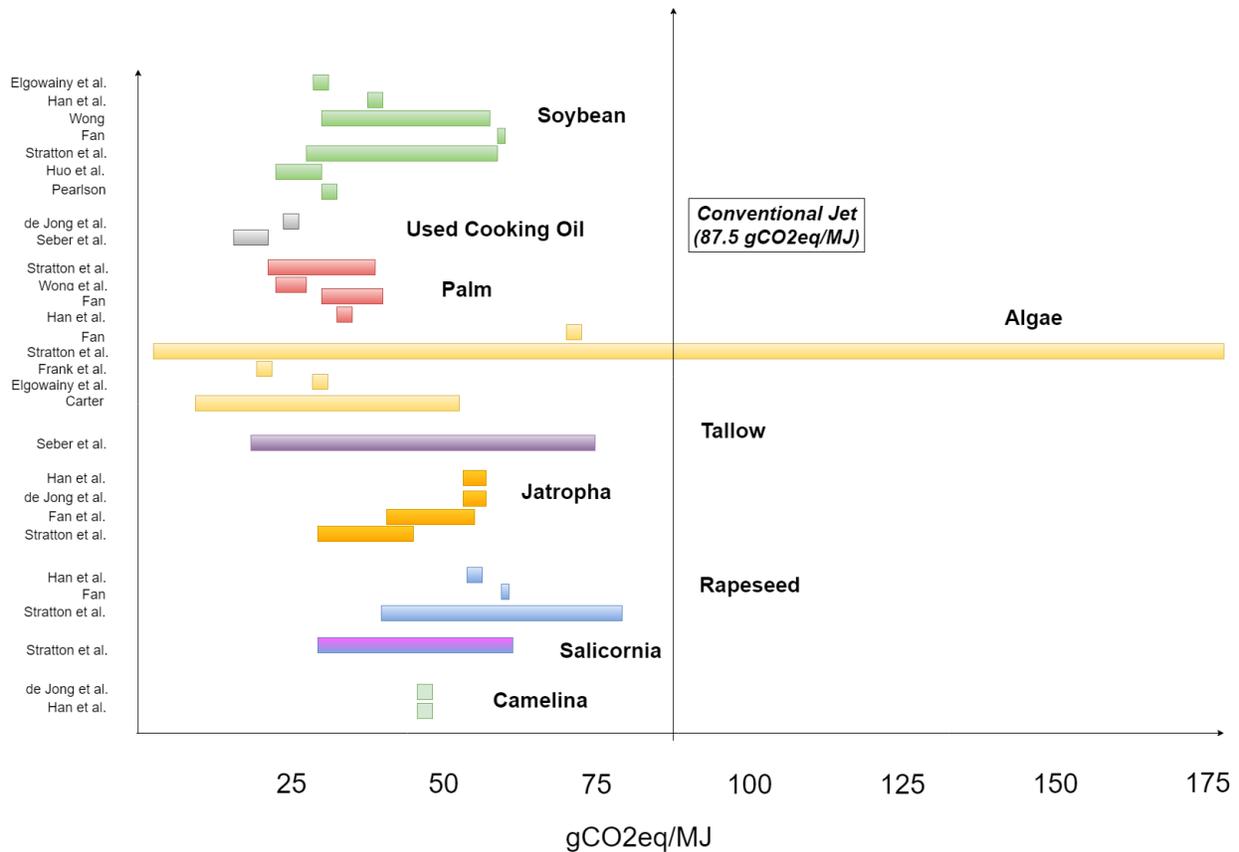
## 2.5 Technology and Feedstock Comparison

In this section, the GHG emission results of the GREET literature are analyzed to determine which feedstocks provided the greatest GHG emission reductions according to literature. Each piece of data was scrutinized to determine whether it was comparable to the other studies, based on the variable impacts discussed in Section 2.4. Differences between results of similar pathways are discussed and the rationale for these differences explained based on the variables used in these studies.

### 2.5.1 Oleo-chemical

The oleo-chemical pathway is the most widely studied biojet pathway in literature. Hydrogenated Esters and Fatty Acids (HEFA), which are produced via the oleo-chemical pathway, are one of two biojet fuels currently certified by ASTM. In 2011, Lufthansa became the first airline to use biojet for commercial flights using HEFA fuel (also known as Hydrotreated Renewable Jet (HRJ)) produced by Neste Oil [12]. A leading licensor of the HRJ technology, Honeywell's UOP, has produced over 590,000 gallons of HRJ fuel for the U.S. Navy and Air Force since 2009.

The HEFA technology platform can utilize any lipid feedstock to produce biojet HEFA fuel, though the following are the most commonly studied: soybean, palm, rapeseed, used cooking oil, tallow, jatropha, and algae. According to Figure 14, there is a wide range of GHG emissions for oleo-chemical pathways published in literature that uses the GREET model. As discussed in Section 2.2.4, the bars in the figure represent the span of possible scenarios analyzed by each study (e.g. range of process variables and or system choices).



**Figure 14: GHG Emissions of oleo-chemical pathways from literature using GREET, excluding emissions from land use change and studies employing the displacement co-product allocation technique.**

Without the land use change and displacement impacts, HEFA from used cooking oil, algae, and palm oil showed the lowest overall emissions. The technologies for HEFA production from used cooking oil, tallow, and palm oil are commercially ready and employed by companies such as Neste and Green Diamond Diesel [99] [100]. However, the technology to produce HEFA from algae is currently under development [101].

The low GHG emissions associated with used cooking oil (UCO) are mainly because it is considered a waste, thereby not accruing any of the emissions from oilseed cultivation or pressing (N<sub>2</sub>O emissions, LUC). Due to this significant advantage, some feel that allowing wastes to ignore all upstream impacts is incorrect [7]. Used cooking oil and municipal solid waste (MSW) are generally agreed to be wastes, since they have already been used and are therefore recycled products. Tallow is considered a waste under most regulations (RFS and RED), which is often more controversial than the UCO designation since it has not previously been used. For this reason, Seber [71] calculated the GHG emissions for two scenarios: one in

which tallow was considered a waste of the meat industry and one in which it was considered a co-product. Depending on which viewpoint was chosen, the emissions for HEFA from tallow changed by 50 gCO<sub>2</sub>eq/MJ [71]. The case where tallow was considered a waste had a GHG emission of 21.7 gCO<sub>2</sub>eq/MJ, while considering tallow a co-product of production increased the emissions to 80.5 gCO<sub>2</sub>eq/MJ [71]. This is what gives such a large range of results for the tallow bar shown in Figure 14. If the viewpoint used by the RFS and RED were adopted, then only the lower end of the results should be considered.

However, even when the RFS and RED viewpoint was adopted and tallow was considered a waste, Seber [71] indicated that UCO still provided greater emission reductions. Both are considered wastes, but tallow has higher energy processing requirements due to the rendering process. The emissions from this extra processing is what gives used cooking oil the advantage.

Of all the cultivated feedstocks, Figure 14 illustrates that HEFA from palm oil can provide the greatest reductions. When compared with Jet A kerosene (87.5 gCO<sub>2</sub>eq/MJ [7]), palm oil HEFA provided a 63% - 75% reduction in GHG emissions. Soybean HEFA only provided a 40% - 65% reduction. These differences are likely because palm plantations have higher yields and reduced fertilizer input, farming energy, and extraction energy [87]. Additionally, because of higher productivity per hectare, the use of palm feedstock requires less total land than soybeans. To produce the amount of biofuel necessary to fuel a large airport (roughly 25,000 bpd), 640,000 acres of palm cropland would be required, compared to 6.7 million acres of soybean cropland [63]. Using this example, approximately 10% of the current U.S. soybean production would be required to fuel this one small airport [63]. However, it should be noted that wastewater treatment was not included in these studies, which is controversial since a large amount of wastewater is produced [69].

Salicornia is an oily halophyte that can grow on saline soils. In fact, it is one of the few feedstocks where the inclusion of land use change improves the overall LCA. Like switchgrass, this feedstock has the potential to improve the carbon retention of the soil, thereby acting as a carbon sink if maintained for at least 30 years [7]. Stratton et al. [7] demonstrated that the inclusion of land use change for this feedstock could reduce the emissions of production by 30 gCO<sub>2</sub>eq/MJ, making the pathway carbon negative. However, as land use change was not included in this review, this benefit was not realized in the results presented in Figure 14.

Jatropha is another unusual feedstock that has been considered for HEFA renewable diesel production, namely because it can be grown in semi-arid regions and can improve the soil carbon in these regions

[63]. Since only the oily seeds of the jatropha fruit tree are required for harvest, the orchard has the potential to be a carbon sink for degraded lands. The GHG emissions for this pathway were given by Wong [48] and Stratton et al. [7], but did not include the carbon sequestration. If this land use change were included, it is likely that this pathway would be carbon negative. Unfortunately, the seeds contain highly toxic and carcinogenic substances that make its domestication ethically questionable, though that has not stopped it from being used for biodiesel production internationally [63]. Consequently, the GHG emissions in the figure should be higher, since the process energy for the detoxification of the oil and seed cake was not included in the results.

Algae shows the potential to achieve the largest GHG reductions, but there is still much uncertainty surrounding the technology, as it is not yet commercial. There are currently many technologies within this pathway being researched, whose combination provide the wide range of GHG emissions shown by Stratton et al. [7]. The technology with the largest effect on GHG emissions is the CO<sub>2</sub> injection method. If the algae pond is not located next to a CO<sub>2</sub> flue gas source, the gas must be scrubbed with monoethanolamine (MEA), compressed, and transported. This process added upwards of 200 gCO<sub>2</sub>eq/MJ to the final LCA emission results of Stratton et al. [7], represented in the high emissions case. The type of algae growth system has the second largest impact on emissions. Only by using open pond or flat panel technology could emissions be kept under 87.5 gCO<sub>2</sub>eq/MJ [72]. The open pond provided the best GHG reductions, especially when paired with de-watering by settling and centrifuge and wet extraction. Frank et al. [72] and Stratton et al. [7] found that this method gave the lowest results (20.2 and 14.1 gCO<sub>2</sub>eq/MJ, respectively). The efficiency of anaerobic digestion, nutrient recycling, and the de-watering technique also significantly impacts results [72]. Land use change was rarely included with algae, since it was assumed to be produced on marginal land. Since none of these methods are commercial, all methods are still on the table and therefore considered by the authors.

The main takeaways from oleo-chemical studies using GREET are:

- HEFA biojet from used cooking oil, tallow and palm oil provided the greatest emission reductions when land use change and auxiliary processing was not included.
- Fuels from waste feedstocks will always have the lowest GHG emissions because they include no N<sub>2</sub>O emissions from cultivation. However, this practice is controversial and there is a very limited supply of these feedstocks.

- HEFA from palm oil had the highest potential to lower GHG emissions than any other vegetable oil when land use change and wastewater treatment were not considered. Including either of these could substantially increase emissions.

### 2.5.2 Gasification

There were 12 studies from 7 papers and reports that calculated the GHG emissions of biojet or renewable diesel by gasification using GREET. Two of these papers (de Jong et al. [28] and Xie et al. [85]) were from peer-reviewed journals. Two were theses from MIT (Wong [87] and Suresh [102]) and the others were funded by government agencies (Stratton et al. [62], Elgowainy et al. [84]). The report by Elgowainy et al. [84] was funded by the Argonne National laboratories, the developers of the GREET model.

The results of the gasification studies in this review are displayed in Figure 15 below. The feedstocks considered are all cellulosic biomass, except for municipal solid waste modeled by Suresh [102]. All are presented with the use of energy allocation, apart from Elgowainy et al. [84], who used displacement for the electricity co-product.

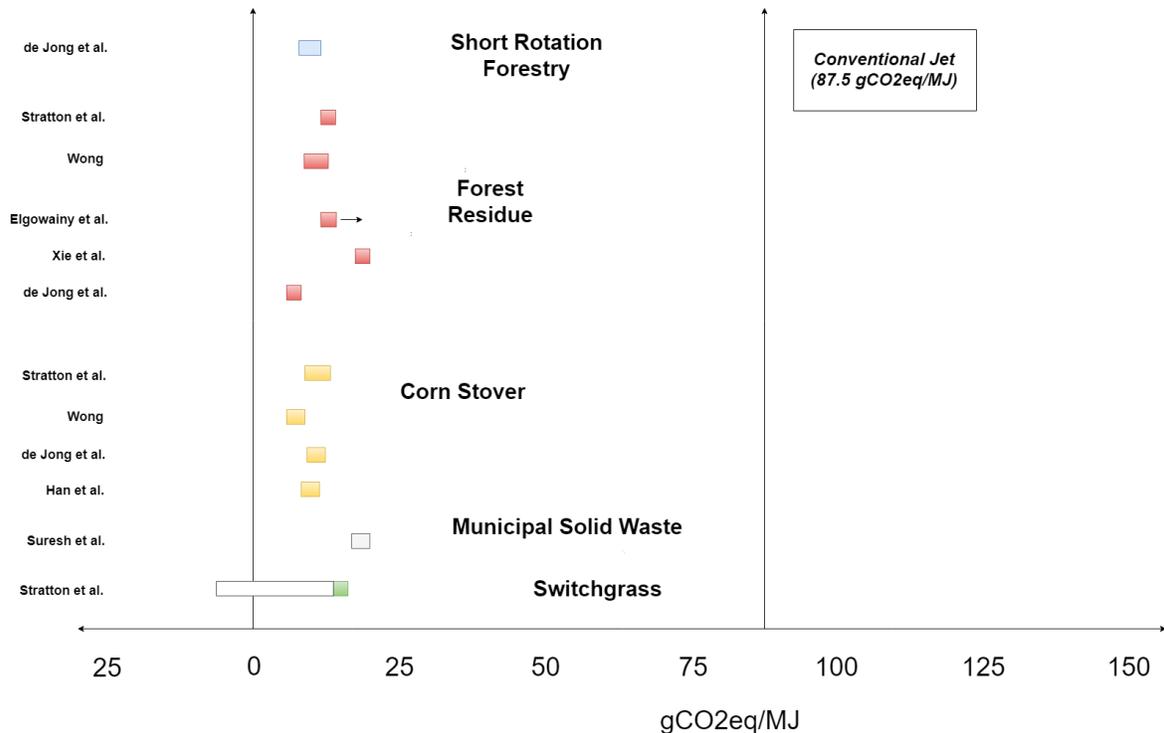


Figure 15: GHG emissions of Gasification pathways using GREET [7] [11] [30] [22] [12] [38]

According to Figure 15, the corn stover and short rotation forestry (SRF) pathways show the lowest GHG emissions, followed by forest residue, switchgrass and municipal solid waste. However, all pathways are very close and have some overlap. All feedstocks exhibit emissions that are among the lowest of all biojet pathways.

When land use change was not included, switchgrass had emissions slightly higher than those of corn stover at 17.7 gCO<sub>2</sub>eq/MJ [7]. This makes sense, since switchgrass is cultivated and requires more fertilizer than any of the other feedstocks, thereby increasing the N<sub>2</sub>O emissions which has a medium impact for this platform. Municipal solid waste, while a waste, is comprised of a large amount (37-47%) of non-biogenic material [102]. This means that a large portion of the fuel emissions were not negated by the biogenic nature of the feedstock, even when the loss in landfill methane was considered. While only one study was conducted on this feedstock, the results are as expected.

Most of the studies indicated that the gasification emissions from using forest residue were slightly higher than that of corn stover. This is most likely because of the longer transportation distances for forest residue as well as its higher moisture content. Since the impact of many of the methodological variables are low for gasification, process variables and system choices like transportation distance are likely to have a stronger effect. Gasification feedstock must be dried to 5-35% moisture content and ground to a particle size of less than 1 mm before entering the gasifier [103]. Forest residue generally has a higher moisture content (30-70%) than corn stover (15-30%) after gathering and therefore requires more energy to reduce the moisture content to acceptable levels [103]. Forest residue must also be transported farther distances for processing, as forest residue supply is not as dense as those of corn stover: Pierobon et al. [104] estimated the travel distance for forest residue to be about 75 miles, given the annual requirement of feedstock for the processing facility, while Ebadian et al. [105] estimated the supply radius of corn stover to be around 30 miles.

The importance of collection energy on forest residue emission results is exemplified by de Jong et al. [28]. Unlike other authors, de Jong et al. considered forest residue collection energy to be 54% lower than corn stover. On the contrary, Stratton et al. [7] considered forest residue to have collection emissions that were 80% higher than corn stover, which is more in line with the results of other authors. Figure 15 shows the impact that these different collection energies can have on results. While it is unlikely that forest residue collection energies would be less than those of corn stover, as assumed by de Jong et al. [28] due to the greater travel distances to acquire logging forest residue (70 km vs. 30 km on average [7]), the results by de Jong et al. [28] may be exemplary of situations where mill residues or municipal woody

feedstocks are utilized. Furthermore, de Jong et al. [28] also considered a lower moisture content of woody feedstocks (as seen by the high amount of electricity co-product from pyrolysis char, which is entirely consumed by drying for forest residues for other studies) [28]. This shows that the moisture content of starting feedstock can also change the LCA results significantly.

Only one study [28] was conducted on the use of woody biomass from short rotation forestry (SRF) as a feedstock for biojet production via gasification. However, compared to other feedstocks by this author, the GHG emissions from SRF had similar emissions to corn stover, while higher than those of forest residue. This is likely because of the low diesel and fertilizer requirements of the managed tree stands. Within SRF, de Jong et al. [28] considered two hardwood species: poplar and willow. Willow requires very little fertilizer (only 2.85 g/kg product) and 0.18 MJ/kg diesel fuel for collection [28]. Poplar requires a little potassium (2.02 g/kg K<sub>2</sub>O) and phosphate (1.01 g/kg P<sub>2</sub>O<sub>5</sub>) fertilizer, but only 3.02 g/kg N fertilizer and 0.25 MJ/kg diesel [28]. This is in contrast to the 43 g/kg of N-fertilizer required by Camelina, or the 8.77 g/kg required by corn stover [28]. These levels are slightly lower than the amounts calculated for corn stover, so give a similar GHG emission after the higher moisture content is considered.

Takeaways from this section are as follows:

- Gasification offers significant GHG reductions for most feedstocks.
- Biojet fuel made via gasification appears to have the best performance when made from corn stover over forest residue or switchgrass, due to collection and fertilizer emissions.
- The inclusion of land use change can improve GHG emission results if cultivation shows an improvement in the soil organic carbon (SOC) levels of an area. This is the case when switchgrass is grown on degraded land.
- The transportation distance, moisture content, and collection energy have a significant impact on the emissions of waste feedstock pathways.

### 2.5.3 Pyrolysis

Eight studies from 5 papers were available in literature that considered the LCA of biofuel via pyrolysis of biomass using GREET. All of the reports that were referenced modeled fast pyrolysis coupled with hydrotreatment upgrading from either corn stover, forest residues, or algae. All the authors used energy allocation to distribute emissions between co-products, except for Q. Dang, who used displacement.

Pyrolysis is generally performed in two stages to produce a transportation fuel: fast pyrolysis to produce bio-oil, and then hydrotreatment to stabilize and upgrade the bio-oil to a hydrocarbon fuel. None of the LCA studies included here considered a co-processing strategy with petroleum fuel, but rather modeled the hydrotreatment process at a standalone bio-refinery.

Feedstocks modeled for the pyrolysis pathway utilizing the GREET model are limited to corn stover, forest residue, and algae. Figure 16 displays the GHG emission results from these works. As is customary, any GHG emissions obtained using displacement are displayed with an arrow, as it is considered a high impact variable. Since the hydrogen source is also a high impact variable for this pathway, the GHG emission result for scenarios using other sources of hydrogen are displayed by an empty bar. The higher end of the bar (28.8 gCO<sub>2</sub>eq/MJ) indicates the more common scenario where all of the hydrogen is produced externally through steam methane reforming (SMR). In this scenario, no co-products are produced, so the allocation method does not matter and the arrow should only correspond to the bottom of the bar.

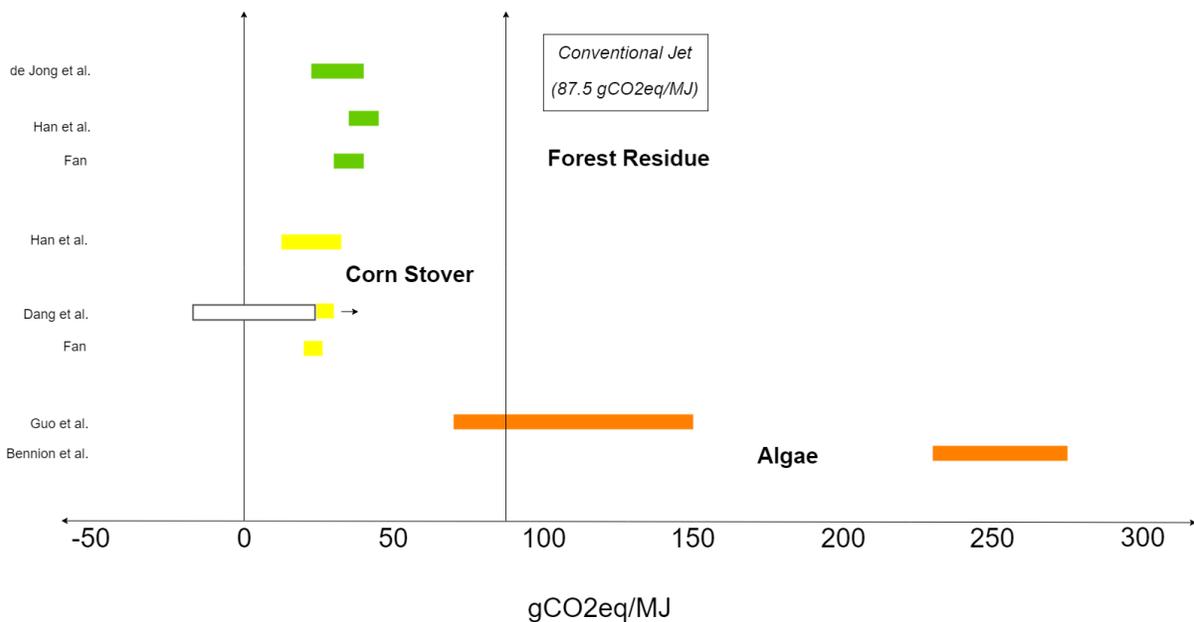


Figure 16: LCA Emissions of the Pyrolysis Pathway [28] [20] [60] [86] [88] [106]

Figure 16 shows that pyrolysis from corn stover had lower GHG emissions than forest residue or algae. When the high impact of the hydrogen source variable is removed (in the case of the black bar), feedstock procurement and process variables have the highest impact. Therefore, the feedstock differences likely

occur for the same reasons they do with gasification, namely because of the moisture content and residue transportation differences.

However, with pyrolysis, the impacts of these differences are even larger, since the co-product yield is affected. When high moisture forest residues are used as the pyrolysis feedstock, all of the produced char is generally burnt to provide the heat necessary for drying. On the other hand, corn stover generally requires less energy for drying, so some bio-char remains after drying and able to obtain a co-product credit (11.47 MJ char/kg fuel) [60]. Most studies considered this extra char to be used as a soil amendment, thereby sequestering the carbon in the soil. This also explains why the results from de Jong et al. [28] were generally lower than the other results for forest residue, as this study considered a lower moisture content of forest residue, thereby including a char co-product credit where other studies of forest residue did not [10].

Between corn stover studies, the GHG results from Fan [60] (20.3 – 22.9 gCO<sub>2</sub>eq/MJ) and the lower end of Han et al. [20] were lower than the results from Dang et al. [86] with equivalent hydrogen source (SMR). This is likely because Fan [60] and Han et al. [96] considered a lower moisture content (15%) than Dang et al. (25%) [86]. As a result, both Fan and Han et al. had char leftover that could obtain co-product credit, thereby lowering the GHG emissions. Dang et al. [86] showed that the GHG emissions of corn stover could be negative (-18.14 gCO<sub>2</sub>eq/MJ) when all of the aqueous bio-oil was used to produce excess hydrogen, which would then obtain a credit if the displacement method were used.

The highest GHG emissions were found when algae was used as a feedstock, which were 5 to 6 times higher than either forest residue or corn stover [106]. These extremely high values were caused by the high energy requirement of the dewatering stage. Since pyrolysis feed must be very dry, the dewatering required to take a 95% aqueous algae solution to a 7% aqueous solution made this pathway very GHG intensive [76] [92]. The resultant GHG emissions were significantly higher than the petroleum kerosene A-1 alternative, making certain algae scenarios undesirable for lowering GHG emissions.

Based on the studies in this review, the following trends for pyrolysis LCA were observed:

- Diesel and gasoline from corn stover bio-oil had lower GHG emissions than from forest residue. This is likely because of the higher energy requirements of drying forest residue.
- The displacement allocation method gave lower GHG emission results. This effect is more pronounced when there are high amounts of co-products produced in proportion to the main product.

- Using hydrogen produced from aqueous bio-oil or bio-gas lowers emissions by about 12% to 100% depending on the amounts used.

#### 2.5.4 Hydrothermal Liquefaction

Only three HTL literature studies were found using the GREET model and all these studies modeled diesel production from algae (see Figure 17). Davis et al. [107] provided a range based on various process conditions and scenarios, while the others calculated average values based on their chosen process and the available data.

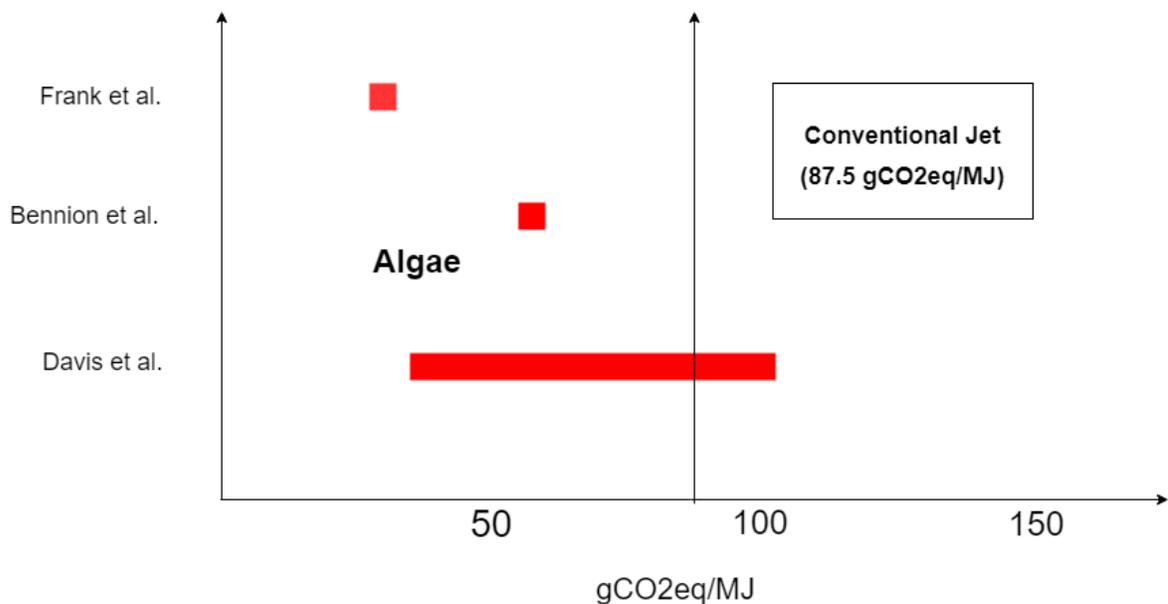


Figure 17: HTL GHG Emissions [72] [106] [108]

According to Figure 17, Frank et al. [72] had lower GHG emissions than Bennion et al. [106], which is most likely due to a combination of several things. The process by Frank et al. [72] assumed that all of the hydrogen requirements were satisfied by the conversion of the aqueous bio-oil through aqueous phase reforming, which has been previously shown to lower overall GHG emissions during pyrolysis [86]. The process also used catalytic hydrothermal gasification rather than the conventional anaerobic digestion to produce biogas from algae wastewater, which reduced fugitive methane emissions by over 30% [72]. Davis et al. [108] considered a continuous HTL process where the aqueous bio-oil layer was catalytically gasified to produce higher yields. Davis et al [107] also modeled the seasonal effect of algae HTL

production, showing that working within three seasons (spring, summer, and fall) provides much higher GHG reduction benefits compared to working all year.

Algae biofuel has not yet been commercialized, so most of the data used for these studies came from a patchwork of experiments and pilot-plant data. The production of biofuel from this feedstock requires much more processing than any of the other feedstocks, so varying conditions and yields at each stage creates a larger uncertainty in overall algae-biofuel LCA results. When using algae to make HEFA fuel through the oleo-chemical pathway, lifecycle GHG emissions were generally quite high. The fact that both papers provided much lower GHG emissions for algae-based biofuel than via pyrolysis or oleo-chemical (see Figures 14 and 16) may indicate that hydrothermal liquefaction has significant GHG reducing potential, especially for feedstocks with high water content. These results indicate that for algae pathways, HTL should probably be the process of focus over pyrolysis or HEFA.

#### 2.5.5 Conclusions

Based on the included GREET LCA literature, the gasification platform displayed the lowest GHG emissions of any of the other platforms analyzed in this review. This observation is reasonable because the processes studied used mainly waste feedstocks and did not require much external energy – most of the process heat and electricity was provided by the biomass itself. Pyrolysis and HTL displayed potential for some feedstocks, particularly corn stover, but these studies had a high level of uncertainty as most were based heavily on a single project by PNNL and therefore only representative of a small sample size. The HEFA pathways were the only pathways that were fully based on commercial processes and were specific to biojet production rather than diesel or gasoline. These pathways had much larger ranges of GHG emissions because there was more data available, giving a wider range than the other pathways. HEFA pathways showed significant promise and the best reductions were seen by used cooking oil, tallow, palm, and soybean feedstocks.

Though many of the differences seen between studies could not be confidently attributed to just one variable, the variable analysis highlighted that many of the variables used between studies were similar. With the elimination or marking of the most highly impactful variables (co-product allocation, land use change, and hydrogen source) the difference between studies was minimized and allowed feedstock pathways to be compared with higher accuracy. However, the literature data displayed in this chapter can still only be considered estimates of pathway potential. Production data is still scarce and many of these studies rely on models such as Aspen or bench and pilot-scale data. Scale-up of production often produces

unforeseen challenges that these studies may not consider. Furthermore, the impact of several variables was indeterminate due to lack of information but could prove to have a large impact. These LCA studies should therefore be considered a starting point only and should be updated as new information becomes available.

## 2.6 Model Differences

If all models used identical data and all authors used model defaults, LCA results would be the same regardless of the model used. However, as Figures 10, 11, and 12 demonstrate, this is not generally the case. Results are often different between studies for the same pathway because users use their own inputs and models use different assumptions and data. The difference a user makes on LCA data was explored in Section 2.4. In this section, the difference that models and their assumptions have on results is explored. This is accomplished by comparing the trends seen between LCA results that used different models. The theories and conclusions developed within the GREET variable and technology analysis is applied to the GHGenius and SimaPro sub-studies and their relationships explored. The combination of these two analyses allow broader theories and conclusions about the field to be made, in particular regarding how they pertain to the research questions of this study.

Though the actual GHG emission values are different between studies of different models since the models use different process data and carbon intensities, the relative ranking of the pathways are expected to be similar. When Figures 10, 11, and 12 in Section 2.3 are compared, it is apparent that there are trends between the models when it comes to which pathways provide the greatest GHG reductions. For example, all figures indicate that the gasification platform provides the lowest emissions of any other technological platform. This is followed by oleo-chemical pathways, where waste feedstocks provide the greatest emission reductions. HTL is next, excluding algae feedstocks, followed by pyrolysis. GREET and SimaPro studies considered different feedstocks, but the feedstocks in common had similar rankings. GHGenius studies, on the other hand, gave quite different results, especially for oleo-chemical pathways.

These differences raise several questions: Why does palm appear so much worse with GHGenius and canola so much better than with the other two models? Why are the LCA results from GREET studies so much higher than those of SimaPro and GHGenius? Are different variable choices behind these differences? Can anything meaningful be drawn from these results? By answering these questions and others, differences and trends between models are explored.

### 2.5.1 Why are the HEFA GHG emission results from GHGenius and SimaPro lower than those of GREET?

For oleo-chemical pathways, GREET results were much higher than those from GHGenius and SimaPro on average. As can be seen by Figures 10, 11, and 12 in Section 2.3, most HEFA results for GREET studies were between 25 and 50 gCO<sub>2</sub>eq/MJ, around 30 gCO<sub>2</sub>eq/MJ for SimaPro, and 20 gCO<sub>2</sub>eq/MJ for GHGenius. A large source of this variance is likely due to the different use of co-product allocation method, which was found to have a high impact on oleochemical pathways. All of the studies using GHGenius used the displacement allocation method, which was shown to give lower emission values (see Appendix III). Some of the studies using SimaPro also used the displacement allocation method.

However, this is not the only reason for lower results. A study by Fan [60] indicated that even when GREET and SimaPro used the same allocation method, SimaPro still gave lower results on average (see Table 13).

**Table 13: GREET and SimaPro LCA result comparison from oleo-chemical feedstocks using similar allocation methods [60]**

	GREET (gCO <sub>2</sub> eq/MJ)	SimaPro (gCO <sub>2</sub> eq/MJ)	
<b>Rapeseed:</b>	57.7	41.0	(Energy Allocation)
<b>Palm:</b>	39.2	10.0	(Energy Allocation)
<b>Camelina:</b>	48.9	21.37	(Displacement)

Another likely cause of the GHG emission depression seen by SimaPro is the N<sub>2</sub>O emission calculation method. While GREET studies multiplied all fertilizer and crop residue nitrogen by 1.325% to obtain the emission factor, most SimaPro studies utilized the IPCC Tier 1 equation (1.325% of fertilizer + 1.225% of crop residue) or did not include these emissions at all. GHGenius also used the IPCC Tier 1 and 2 equations, which generally give lower emissions. GHGenius studies gave the lowest HEFA emissions of all, likely because of their lower N<sub>2</sub>O emission factors, the use of displacement throughout, and the consideration of only one scenario.

SimaPro is a European model that relies on data from databases. Most of the SimaPro studies in this analysis utilized the European EcolInvent database, along with other literature sources. This regional difference likely adds some variation to the results. The system boundary of SimaPro also affects the results. Unlike GREET and GHGenius, treatment of palm oil mill effluent (POME) was not included. The emissions from this process were significant (41.7 gO<sub>2</sub>eq/MJ), so its exclusion significantly dropped the overall GHG emission value calculated by SimaPro for this pathway [60].

2.5.2 Why is the oleo-chemical ranking from GHGenius so different than that of GREET and SimaPro? Why is Palm so much worse in GHGenius and Canola so much better?

Using GHGenius, used cooking oil (1.7-6.6 gCO<sub>2</sub>eq/MJ), tallow (9.9 – 16.4 gCO<sub>2</sub>eq/MJ), and canola (10.1-24.1 gCO<sub>2</sub>eq/MJ) provided the lowest GHG emissions of all the oleo-chemical feedstocks presented. The performance of UCO is in concert with the results of previous models, which also showed it to have the lowest emissions of any pathway (see Figure 18), but the low emissions of canola was different. Though the exact GHG emissions of pathways should not be compared between models due to the model differences previously discussed, the relative rankings of the pathways can still provide information about which feedstocks and technologies provide the greatest GHG emission reductions. Both the GREET and SimaPro studies indicated that the rapeseed pathway had one of the highest relative emissions of any biojet pathway, with GHG emissions higher than that from palm and soybean (see Figures 10 and 11). Canola is a cultivar of rapeseed that contains lower erucic acid, so the two are very similar in terms of agricultural practices and use as a fuel feedstock [109]. Palm HEFA provided some of the lowest emissions according to the GREET and SimaPro studies, where it offered lower emissions than either soy or rapeseed. In GHGenius, this is quite the opposite: rapeseed gave lower emissions than soy and palm had some of the highest GHG emissions (51.1 – 90.0 gCO<sub>2</sub>eq/MJ). Figure 18 below illustrates this difference.

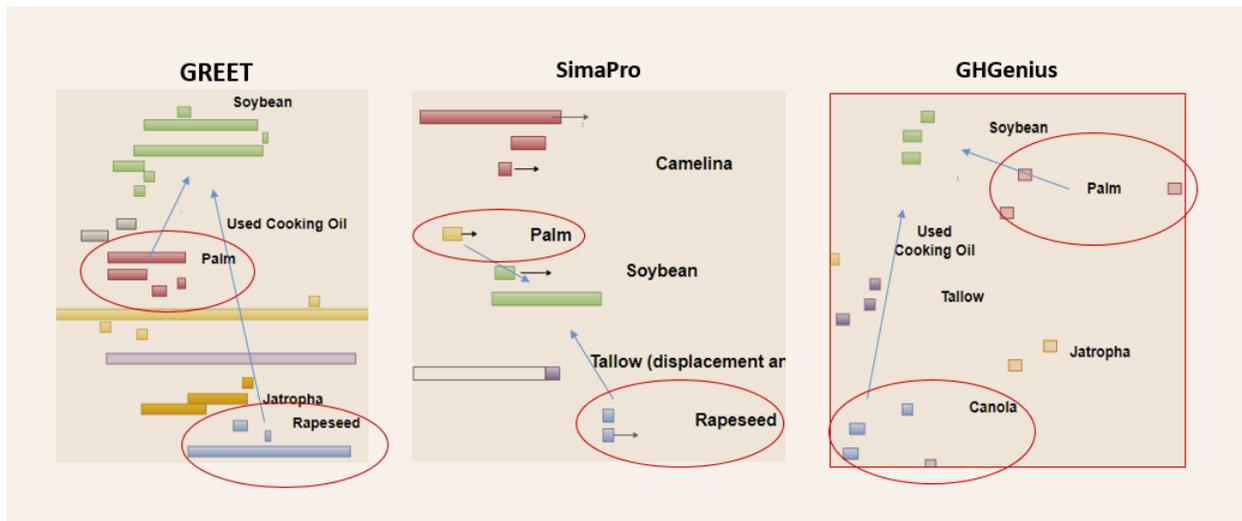


Figure 18: Palm and Canola relative to Soy for GREET, SimaPro, and GHGenius

These two differences, the substantially higher emissions of palm and the low emissions of canola with GHGenius, are likely due to the inclusion of land use change in the GHGenius model. GHGenius has always allowed the user to incorporate soil organic carbon (SOC) changes into the emission calculations, but since the release of version 3.16 in 2008, the inclusion of soil carbon changes due to agricultural improvements are included by default for several feedstocks, namely corn, wheat, soy and canola [110]. In 2013, GHGenius updated the SOC improvements of canola according to a large study by the Canola Growers Association and aggregated the data to the provincial level. The inclusion of palm oil emissions from land conversion (see Section 2.5.1) was also added to the model defaults during this revision [66]. These emissions are embedded into the cultivation module and cannot be removed post-study. This is different from the GREET and SimaPro studies, where these emissions are calculated separately and can easily be separated from the final results.

The increased GHG performance of the soy pathways using GHGenius is also most likely due to the inclusion of land use change in the model. GHGenius relies on Canadian data and Canada has been moving toward no-till agricultural management over the past decade [66]. While this practice improves the soil carbon retention of arid dry soils like the Canadian Prairies, where canola is grown, it actually increases the soil GHG emissions from wet soils, where soy is typically grown in Canada [80]. As a result, canola receives a carbon credit for soil improvement through no-till management, while soy receives a debit [111]. This is likely the main factor to why soy has a lower ranking in GHGenius than in the other two models. Furthermore, soy is a nitrogen fixing crop, which means that it will increase the nitrogen in the

soil, thereby increasing the N<sub>2</sub>O emissions upon disturbance [111]. GREET and SimaPro do not automatically include these emissions, so their calculations of soy will be lower.

Another reason why canola may give greater GHG reductions in GHGenius is because the model defaults are from specific industrial data. Most of the canola data was obtained from a thorough report by the Canadian Canola Growers Association, which was based on a survey of over 1000 farms across Canada in 2010 [66]. As this type of study has not been done for the other feedstocks, the data from other feedstocks is obtained through a combination of Environment Canada reports (fertilizers usage) and IPCC emission factors (SOC and N<sub>2</sub>O emissions) [65]. Some authors believe that the IPCC factors may overestimate the emissions from land use and SOC changes, which may contribute to higher emissions for models that use IPCC factors [80].

The poor ranking of palm biojet fuel in GHGenius is also likely attributed to the inclusion of land use change. Figure 18 shows three studies for palm HEFA, two by (S&T)<sup>2</sup> consultants, all of which show much higher GHG emissions than the other feedstocks. The highest GHG emissions are from a 2013 study by (S&T)<sup>2</sup> consultants. This study utilized GHGenius 4.03, which includes 11.4% of the land use change emissions associated with peatland drainage for plantation cultivation in its GHG emission calculations [66]. This default value represents the historical probability of peat drainage for palm plantation development occurring in Indonesia and Malaysia, where 90% of palm plantations are located [66]. The emissions from this type of land use change are extremely high (up to 900 gCO<sub>2</sub>eq/MJ), so any reflection of this practice in the calculations will increase them substantially [7].

The other palm studies presented in Figure 18 used previous models of GHGenius, so did not yet include the land use change addition. The relatively high GHG emissions from these pathways are therefore likely due to the methane emission factor from wastewater treatment. Palm oil mills use water and steam during palm oil extraction, resulting in a large amount of water that must be treated prior to disposal. Wastewater treatment for this effluent, traditionally referred to as Palm Oil Mill Effluent (POME), generally consists of anaerobic digestion in open tanks [66]. Methane is produced during digestion and is typically released to the atmosphere, except in more modern mills which capture this gas [7]. As a result, significant emissions can occur during this step. Though POME emissions are included in the GREET studies, the emission values differ. For example, Stratton et. al [7] (GREET) assumed that between 2.3 and 3.6 m<sup>3</sup> of methane were emitted per ton of palm fresh fruit (FFB), while (S&T)<sup>2</sup> consultants used the GHGenius defaults, which assumed methane emissions of 55.4 m<sup>3</sup>/FFB [7] [80]. These different values are the result of drawing input data from different literature. Stratton et al. [7] obtained their information

from a paper by Reijnders [112] and GHGenius obtained its value from Subramium et al. [113]. As methane has a very large global warming potential, this difference has a profound impact on the final GHG emission results. For the studies before 2013, this is likely the main reason why the GHG emissions of palm HEFA are so much higher in GHGenius than the other two models.

Other differences that may cause the higher emissions of palm biojet in GHGenius is the inclusion of nitrogen cover crops on palm plantations and the low value co-product. Palm oil has the highest yield of any of the oils, but the palm kernel meal is produced in low quantities (unlike soymeal) and has very poor protein content (18%) [80]. Since the co-product is allocated a credit in proportion to its protein content in GHGenius, the co-product credit allocated to palm kernel meal is only about 2% of that of soy. In GREET, the palm kernel meal was allocated emissions based on its energy or market value, which is larger than 2%, so more emissions were allocated away from the palm oil product, thereby lowering the overall GHG emission results.

Tallow gave lower GHG emissions with GHGenius than some of the values expressed by GREET or SimaPro, but this is most likely due to the displacement allocation method. As the study by Fan [60] shows (see Figure 11), the GHG emissions for tallow are reduced considerably when displacement allocation is employed. This is due to the large credit given for the bone meal co-product for its high protein content (50%) [60] [80]. GREET and SimaPro do not generally use displacement, so the emissions tend to be higher for this feedstock.

Jatropha also shows higher emissions in GHGenius for the same reason. Though Jatropha can grow well in arid soils, the seeds are toxic. This means that the seed meal is generally not used as animal feed, but rather for electricity production [80]. This significantly reduces the credit generally given to the seedmeal co-product, raising the emissions of this pathway relative to the other pathways in GHGenius.

### 2.5.3 Why are short rotation forestry (SRF) emissions higher than corn stover emissions in GHGenius, but not in SimaPro or GREET?

This issue will first be explored within GHGenius. Though the locations are different for all three studies, this is not the reason that SRF shows higher emissions than corn stover with this model. The corn stover study by O'Connor [78] actually used a more carbon intensive electricity mix than that used by (S&T)<sup>2</sup> Consultants [114] (U.S. vs Can). If the study by O'Connor [78] were to adopt the same electricity mix used by (S&T)<sup>2</sup> Consultants [114], the corn stover GHG emissions would likely drop, widening the gap between corn stover and SRF. As there are no other internal differences between the GHGenius studies, the differences between the assumptions used within GHGenius and GREET/SimaPro will be explored.

Upon comparing the assumptions used between the studies of GREET and GHGenius, it became apparent that there is a difference in the N<sub>2</sub>O emission data used. The GREET model studies considered much lower fertilizer requirements for the SRF stands than those using GHGenius (SRF assumes about 50% of the fertilizer used by corn stover in GHGenius, while only 33% in the GREET study). This would significantly reduce the GHG emissions of GREET, which may explain why the results of SRF are higher within GHGenius than GREET [82] [28].

With SimaPro, not only are the emissions from corn stover very close to those of SRF, but one study even considers the emissions of SRF to be lower than that of corn stover [76]. While this result may be reasonable in certain situations (i.e. when only willow is included, low fertilizer requirements, sun-dried wood, minimal transport distance), the main reason that this result is lower than the other corn stover and SRF results in SimaPro is because of the location of the study. The study by Irribarren et al. [76] took place in Spain, whereas the other SimaPro studies took place in the USA (see Appendix II). In Spain, the electricity mix is predominantly from low-carbon sources (69% from renewables or nuclear in 2015) [115]. On the other hand, the U.S. electricity mix used for the SimaPro corn stover study was comprised of 67% fossil fuel in 2015 [116]. As O'Connor [78] demonstrated, this will greatly impact results and is the most likely reason for the low GHG emission value displayed for short rotation poplar by Irribarren et al. [76].

It is unclear which of these studies is most accurate and how the two pathways would compare in each model if they were in the same region and used similar assumptions. It is likely that modeling the pathways in a similar region using GREET and SimaPro fertilizer requirements would give similar results with all models, where corn stover has a slightly greater GHG emission reduction than SRF.

2.5.4 Why does GHGenius show pyrolysis-fuel from standing timber to have lower emissions than from short rotation poplar? Why are emissions higher when the fuel is produced from forest residue than mill residues?

Figure 12 in Section 2.3 shows that corn stover, mill residues, and wheat straw had the lowest GHG emissions for the thermochemical pathways modeled by GHGenius. This can be expected because they are wastes, so do not have the high emissions associated with cultivation and land use change. However, the emissions from forest residues were surprisingly high. Unlike mill residue (sawdust and shavings), the forest residues must be gathered from the forest and transported to site. In order to obtain enough material to supply the input requirements, a large area must be swept (estimated to be around 75 miles [104]). These branches and logs must also be chipped and ground to be ready for pyrolysis. The energy emissions for the pre-treatment are likely what cause the overall GHG emissions of forest residue-based fuels to be higher than that from mill residues.

Since GHGenius does not account for the land use change and SOC change associated with tree harvesting and managed forest development, the emissions associated with using standing timber (natural stands) and short rotation forestry (managed poplar and willow stands) were low [82]. In fact, they were lower than that of the forest residues because forest residue must be gathered from a much larger radius, adding significant emissions to the feedstock collection stage. Managed forests have only slightly higher emissions than standing timber because of the fertilizer that is used during cultivation. Fertilizer use for willow cultivation is exceptionally low [117]. If this research were considered for the willow species in GHGenius, this value would likely drop closer to that of standing timber.

#### 2.5.5 Why are the corn stover pyrolysis values in GHGenius so different?

The result variance for corn stover within GHGenius (Figure 12) is most likely due to the regional variance between the studies. As seen in Appendix II, both studies utilized similar process assumptions, system choices and methodological variables. The only variables that were noticeably different were the system boundary and location. The study by Novo Energy Group [83] considered biojet to be the final product, whereas O'Connor's study [78] was for a diesel product. However, as discussed in Section 2.4, the expansion of the system boundary to biojet rather than diesel is expected to have little to no impact on the overall GHG emissions. Therefore, other factors must be at play in this situation. As both studies take place in different locations, the impact of this variable on the results is likely, though its impact could not be ascertained from the variable analysis using GREET literature.

The study by Novo Group [83] indicated that the electricity mix of a region could have profound impacts on the overall GHG emission results, especially for energy-intensive processes like pyrolysis. These effects were also seen for the HEFA pathways, but not nearly to the same extent. Figure 19 shows the emissions of biojet fuel via pyrolysis from various feedstocks in Alberta (AB) or Ontario (ON). The feedstocks considered for AB included forest residues and wheat straw, while forest residues and corn stover were considered for ON.

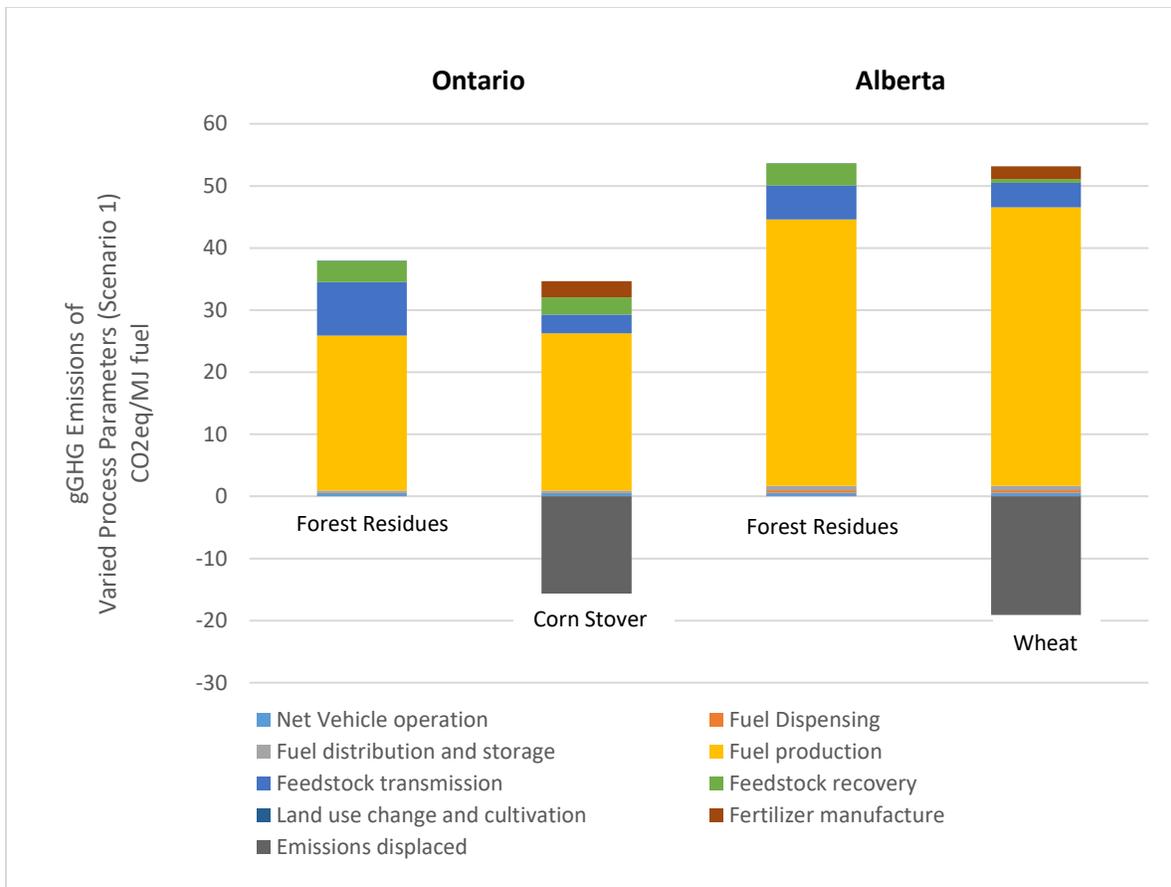


Figure 19: GHG Emissions for Pyrolysis Pathways in Canada [83]

As Figure 19 shows, fuel production generated the largest amount of emissions (other than vehicle operation, which was negated by the biogenic nature of the feedstock). Interestingly, the fuel production emissions in AB were almost double of that of ON. This is likely due to the different electricity mix carbon intensities of the two provinces. Alberta’s electricity mix had a carbon intensity of 227 gCO<sub>2</sub>eq/MJ of electricity delivered, while Ontario’s electricity mix had only a carbon intensity of 39 gCO<sub>2</sub>eq/MJ [58]. While only 2 kWh of electricity/L of biojet fuel was used during fuel production, this alone was enough to increase the fuel production emissions of Alberta by 71% (17.8 gCO<sub>2</sub>eq/MJ). These additional emissions are the main reason that the overall GHG emissions for forest residues in Alberta were 15.7 gCO<sub>2</sub>eq/MJ higher (practically double) those of Ontario for the same feedstock.

Since the corn stover study by O’Connor [78] in Figure 12 took place in the USA and the study by Novo Group [83] took place in Ontario, a similar effect is taking place. The carbon intensity of the electricity mix in the USA is similar to that of Alberta, so the corn stover results from O’Connor [78] would be expected

to be much higher than those of the Novo Group [83]. Most of the other studies that utilized the GHGenius model used an average Canadian mix, which is somewhere in the middle (see Table 14). As a result, the GHG emissions for corn stover would likely be somewhere in the middle of the two studies published here if the average Canadian mix were used. This should be kept in mind when comparing corn stover to other studies presented for GHGenius.

**Table 14: Mix of Electricity used within GHGenius for various regions (GHGenius Model 4.03)**

<b>U.S. Average</b>	<b>Canada average</b>	<b>Alberta</b>	<b>Ontario</b>
Coal (45%)	Hydro (58%)	Coal (57%)	Nuclear (46%)
NG Turbine/Boiler (23%)	Nuclear (13%)	NG Boiler (32%)	Hydro (27%)
Nuclear (17%)	NG Boiler (11%)	Wind (4%)	NG Boiler (18%)
Hydro (8%)	Coal (10%)	Biomass (4%)	Wind (4%)
Wind (4%)	Wind (4%)	Hydro (3%)	Biomass (4%)
Biomass (2%)	Biomass (3%)		
Oil (1%)	Oil (1%)		

### 2.5.6 Why does Forest Residue show improved emissions over Mill Residue for gasification in GHGenius, but not for pyrolysis?

According to the GHGenius studies in Figure 12, the GHG emissions for biojet via the pyrolysis of forest residue are slightly higher than those of mill residue. In Section 2.5.4, it was explained that this was most likely due to the additional energy required for collection. Forest residues must be collected from multiple logging sites which are usually far from processing facilities, whereas mill residues are collected at sawmills. Mill residues also require minimal grinding to reach the 3-5 mm particle size required for pyrolysis, whereas forest residues must still be chipped and ground, requiring extra energy. All this extra energy translates into higher GHG emissions.

The extra emissions for forest residue occur prior to conversion, so these emissions will be present regardless of the technology. This suggests that the improvement of forest residue over mill residue for gasification exhibited by the GHGenius studies (see Figure 12, Section 2.3) must be due to other variables. A look at the variables of the two residue gasification studies indicate that both the location and system

choices for these studies are different. The study by (S&T)<sup>2</sup> Consultants [79] took place in Canada (Canadian average electricity mix) and modeled a high temperature process (HT), while the study by O'Connor [78] took place in the USA (average US electricity mix) and modeled both a HT and a LT (low temperature) process. The LT gasification process, while less efficient, gave lower GHG emissions [78]. Therefore, including this case with mill residue would likely lower GHG emissions substantially. However, even with these improvements, there would still be a gap between the results of the two feedstocks.

As seen in Section 2.5.5, the location of a process can have a large impact on the results of processes where electricity is a significant percentage of total energy use. This is the case with both gasification and pyrolysis. Both processes consume electricity to grind the feedstock prior to processing and to power auxiliary processes. For this reason, any pyrolysis or gasification fuel would generally have higher overall emissions when produced in a location with a carbon intensive electricity mix. As Table 13 shows, the average electricity mix of the USA is much more carbon intensive than that of Canada due to the high percentage of coal use. Therefore, any thermochemical process that occurred in the USA would be expected to have higher emissions than one in Canada.

However, when a process produces a significant amount of electricity, which occurs with gasification and some cases of pyrolysis, the opposite trend can occur when the displacement allocation method is used. With displacement, the extra electricity produced by the process displaces electricity from the grid. When the grid electricity is very carbon intensive, the process obtains a substantial credit for displacing it with its renewable electricity. In these cases, this credit is greater in regions where the electricity is very carbon intensive, giving a large “negative” GHG emission result for the pathway. Since the U.S. electricity mix is much more carbon-intensive than that of Canada, the forest residue pathway modeled by O'Connor [78], which took place in the USA, obtained a much larger credit than that of the Canadian mill residue pathway. This explains the lower GHG emissions seen by forest residue compared to mill residue. If another allocation method were employed, such as energy or market allocation, neither of these pathways would exhibit negative emissions (a trend only seen when displacement is used) and mill residue would likely exhibit lower overall GHG emissions than that of forest residue.

## 2.7 Conclusions

A comprehensive, systematic review of the LCA biojet fuel literature indicated that the relative GHG emissions of the various biojet fuel pathways were similar for GREET and SimaPro, but different for GHGenius. This difference was shown to be largely due to the inclusion of land use change and the use of displacement as GHGenius model defaults. The inclusion of land use change had a very strong impact on

all cultivated feedstocks, (especially palm, canola, and switchgrass), while the use of displacement had a strong impact on any process that produced a substantial amount of co-product, sometimes making the pathways carbon negative. However, these variables did not affect each pathway equally, which caused the observed difference in ranking between GHGenius and the other two models. These two variables, along with the location and N<sub>2</sub>O emission factor, accounted for the majority of the differences seen between the models.

However, even with these differences, the literature from all the models indicated that the gasification route to biojet could provide the greatest GHG emission reductions. This was largely because gasification uses very little external energy or inputs once it is running. To the contrary, pyrolysis and HEFA fuels required significant amounts of energy-intensive hydrogen. As well, the feedstocks for gasification are generally wastes, which have a substantially lower carbon footprint than cultivated feedstocks. The use of corn stover, mill residues and wheat straw tend to have the greatest GHG emission reductions for thermochemical processes, though the other feedstocks are not significantly higher and overlap under certain conditions.

All models also confirmed that used cooking oil provided the greatest GHG emission reduction of any of the oleo-chemical feedstocks. Wastes will likely always provide lower GHG emissions than non-wastes because they do not include any emissions for their cultivation/production. Exceptions such as tallow and forest residues require substantial amounts of pre-processing or are energy-intensive to collect.

When harvested sustainably, palm oil can provide the greatest emission reductions of any cultivated oil feedstock due to its high yields per acre. However, when not produced sustainably, this pathway can be much more carbon-intensive than petroleum diesel, as seen by the different results of GHGenius and GREET/SimaPro. Due to the high occurrence of peatland and rainforest conversion in some regions, promotion of this feedstock for biojet production is very debatable. Voluntary schemes such as the Roundtable of Sustainable Biomaterials require the reporting of all feedstock sources in order to calculate GHG reduction credits in Europe. However, some have suggested that this could easily be circumvented unless a thorough history of the land and accurate record-keeping is conducted [118].

Through the variable analysis of the GREET literature, it was discovered that there is also variation between studies that utilize the same model. Although all the studies were drawn from different data sources, most of the variation was found to be caused by just a few key variables. It was apparent that the oleo-chemical HEFA pathway was most affected by the inclusion of land use change, allocation

method, N<sub>2</sub>O emission factor and whether the extra conversion process to biojet fuel from renewable diesel was included. These variables also appeared to be the largest sources of variance between models and why there were often large variations in the results of these pathways between models. For gasification, the moisture content and collection energy of the feedstock had the largest impact for feedstocks that were not cultivated (and after LUC and displacement impacts were removed), while the source of hydrogen and allocation method were most impactful for the pyrolysis and HTL pathways. The location of the process and feedstock production also had a substantial impact, especially for energy-intensive processes like pyrolysis or whenever the displacement allocation method was used for thermochemical pathways. The location also had a large impact on N<sub>2</sub>O emissions for models that used IPCC Tier 2 or 3 calculations (i.e. GHGenius) or whenever land use change was included (SOC impacts).

From this work, it has become apparent that there is a real need to establish consistent guidelines when conducting a life-cycle assessment. Without them, modelers can modify their GHG emission results just by choosing the assumptions that offers the greatest GHG reductions. This reduces the legitimacy of LCA as a tool for policy compliance and may be part of the reason that RED compliance has not produced the desired results.

One assumption that would benefit greatly from being standardised is the co-product allocation method. The ISO 14040 guidelines are followed by almost every model and policy, so updating the current guidelines to incorporate these details would most likely have the greatest influence on international policy. It is recognized that while the displacement allocation method is recommended by the ISO, it has the potential to skew results when a pathway produces a significant amount of co-product. In these situations, a market or energy-based allocation method may be preferred. Furthermore, displacement is a consequential (holistic) technique used in an attributional (systematic) model. While the benefits of displacement are representative of reality, its use in an attributional model increase the possibility of double-counting and unequal treatment. It therefore might be worthwhile to limit using displacement in all attributional LCA and that ISO reconsider its suggestion.

The inclusion or exclusion of land use change and agricultural improvement in LCA should also be standardized, as this would decrease a lot of the variance seen between models and studies. Another source of variability that could be eliminated is the use of different GWP and N<sub>2</sub>O emission factors. If ISO was updated to recommend a certain source for these factors, one that was updated regularly by a large scientific body, these variances would likely decrease substantially. One example that would have a profound impact in eliminating variance between models would be to develop a world-specific biofuel

database that contained all the latest data required for biofuel LCA with any LCA model could draw upon. Such a resource would not only aid in biofuel policy compliance legitimacy, but also improve the comparison of biofuel pathways, thereby decreasing the uncertainty in result comparison between studies.

It is also recommended that the concept of “waste” be reconsidered. ISO 14040, RED, and RFS consider “waste” to be any material that has already been used or is of low value [32]. According to these standards, all wastes are considered to have no GHG emissions associated with their production. This definition allows feedstocks such as tallow and used cooking oil to avoid all cultivation and processing emissions, giving them a significant advantage over other feedstocks. In reality, used cooking oil and tallow are both used in animal feed and have a recognized value. In fact, tallow (\$1.38/gallon) [119] and used cooking oil (aka yellow grease at \$1.82/gallon) [119] are considered commodities by the USDA, with prices greater than that of jet fuel (\$1.61/gallon) in the case of used cooking oil [30]. While the use of these feedstocks should be encouraged, it should be determined whether their designation as a “waste” product is the best way to do so.

While the systematic review performed in this chapter gave significant insight into the relative effects of the variance between studies on the overall GHG emission results, this is only the first step in a full literature analysis. It is suggested that a meta-analysis be performed on the data presented in this chapter as a continuation of the work presented within. A meta-analysis is a type of literature analysis that quantifies the data of multiple studies, providing a weighted average for the study sample. This chapter provides the raw data as well as variable impact information that could be used to determine the weights of each study. These average pathway GHG emissions would allow policy-makers to compare technologies statistically and provide solid numbers that could support policy decisions.

While it was outside the scope of this work, it would also be beneficial to perform full lifecycle assessments on each pathway using the same model, assumptions, location and database. This would eliminate most high impact sources of variance and allow very clear differences between models for each pathway to be identified. It would also be useful to perform a sensitivity analysis on each of the variables previously identified to have a medium to high impact on results. While some authors have already performed analyses on allocation method, yields, and hydrogen source on some pathways, the exact impact of these assumptions is not available for all pathways.

In order to apply some of the insights from this chapter, an LCA case-study was performed on a fast pyrolysis drop-in biojet fuel process developed in British Columbia. British Columbia has a large forest sector, making the region a prime location for a lignocellulosic conversion technology. British Columbia is also home to the Low Carbon Fuel Standard, a provincial regulation that encourages biofuel production. Assumptions used within this pyrolysis LCA were informed by the variable impacts and LCA norms discovered in the literature review. A sensitivity analysis was later performed on all variables that were identified in the variable analysis to have a medium to high impact on results for the pyrolysis process, and their impact on this case study is presented in the discussion.

## CHAPTER 3: THE IMPACT ON LCA VALUES OF THE POSSIBLE FAST PYROLYSIS PRODUCTION OF BIOJET FUEL IN THREE BRITISH COLUMBIA LOCATIONS

### 3.1 Introduction

In Western Canada, forest resources are the most prevalent source of biomass. Most of British Columbia's northern regions are covered in forests, which account for 18% (55 million hectares) of Canada's total forested area [120]. B.C. also has the highest proportion of sustainably certified forests in the world, which has facilitated a robust forest products industry and a record of forest product innovation [103]. These include traditional lumber and pulp and paper, as well as a new pellet industry and innovative products such as Parallam and Oriented Strand Board (OSB) [103]. In B.C., most of the recent innovations have evolved around finding a use for low quality wood residues, such as wood chips, shavings and sawdust. The prevalence of forest biomass industries and proximity to existing infrastructure and equipment makes B.C. a prime location for the development of a biojet refinery. Furthermore, the regulatory climate in the province also provides an incentive to develop a bio-economy in this region. Under the B.C. Low Carbon Fuel Standard, fuel producers are obligated to meet GHG emission reduction targets on their fuels or purchase credits from other low carbon fuel producers [36]. While low jet fuel prices currently hinder biojet fuel development worldwide, local policies such as the Low Carbon Fuel Standard help bridge the gap between jet fuel and biojet pricing and make B.C. an attractive location for biojet development.

To convert B.C.'s biomass resources to biojet fuel, either biochemical or thermochemical means can be employed. However, as described in Chapter 2, biojet production via biochemical means are unlikely since the value of the intermediate products are higher than biojet fuel, making the conversion uneconomical. It is therefore much more likely that thermochemical methods will be employed to convert the biomass in B.C. to biojet fuel. These routes include gasification, pyrolysis and hydrothermal liquefaction (HTL).

Gasification, Pyrolysis and HTL all have the potential to reduce greenhouse gases significantly, as presented in Chapter 2, but economic and supply logistics must also be considered when determining the feasibility of implementation. For example, while gasification generally provides the lowest GHG emission profile of any other technology, it also has a high capital cost of production [121]. This increases the risk of construction and obtaining the required financing to build gasification projects is difficult. For gasification to become economical, it must be built at a very large scale [87]. This creates feedstock supply challenges, especially when forest and mill residues are utilized, since the required supply for this size is often not available.

Pyrolysis and hydrothermal liquefaction, on the other hand, have a much lower capital expenditure for small to medium-sized projects. For example, the capital investment of a 200 MLPY pyrolysis facility is only about \$200-280 Million, compared to \$500-610 Million of a gasification facility of the same size [121]. The optimal facility size for pyrolysis and HTL technologies is also more appropriate for the biomass supply in these regions, which is often spread out over large areas. Commercial gasification facilities, such as the Sasol Oryx GTL facility in South Africa, produce around 34,000 barrels of liquids per day (1,970 MLPY) [122]. A facility this size would require about 15.8 Million ODT of biomass per year, which is more than the forest and mill residue supply of the entire province (4.2 Million ODT forest residues/year, 9.7 Million ODT mill residues) [103] [123]. For these reasons, further advancements in the technology will likely be necessary before it can be utilized economically at the small-scale required by the region, so this technology will not be considered in this study.

Hydrothermal liquefaction shows significant potential, as it can utilize wet biomass, thereby eliminating the need for feedstock drying, but the information about this technology is limited. HTL is nascent and still under development, so the required data to perform a lifecycle analysis is not yet available. Furthermore, the high pressures required with this technology make scale-up challenging, adding technical challenges to this technology [103]. The lifecycle analyses in this chapter therefore analyzed the GHG emission reduction potential of pyrolysis to produce biojet fuel within B.C.

In B.C., there are three potential sources of biomass that could be used for biojet fuel production: forest residues, standing timber, and wood pellets. Since fuels generally have a lower value than lumber, the bioenergy industry will likely need to use the low-quality sources of biomass not utilized by the industry already for its biomass supply. The pellet industry and sawmills themselves consume most of the sawmill residue (sawdust and shavings) [124]. Surplus mill residues still exist in the region, but tend to be widely dispersed at low volumes, making acquisition expensive. Forest residues, on the other hand, are a plentiful and currently under-utilized resource. In B.C., about 6 Million ODT are available annually [103]. Forest residues consist of the residues left over by logging operations, such as branches, treetops, and broken trees. Twenty-five percent of these residues must remain on the forest floor to ensure soil health, but the rest is often collected in piles known as “slash.” While some of these residues are utilized in the pellet and bioenergy sector (0.6 M ODT), the majority remain on the forest floor or are burned in the forest to eliminate fuel for forest fires [103]. The primary barrier for their use is the lack of logistical systems to collect and deliver these feedstocks at a reasonable cost. The fluctuation in supply, moisture content, and high bark content also makes their use as a feedstock challenging.

In contrast, wood pellets contain low levels of bark and have consistent quality and composition. Pellets are a densified form of biomass that can easily be shipped for use in bioenergy applications. While some pellets are used within Canada, 95% of the pellets are exported for bioenergy applications in the U.S., Europe or Asia [103]. Wood fibres, usually a type of sawdust or other sawmill residue, are dried, ground and compressed to reduce shipping costs and facilitate handling and storage. While more expensive than forest or mill residues, pellets provide a consistent, low moisture feedstock without the need to establish new supply chains or compete for raw material. The current annual production of wood pellets in B.C. is 1.7 million tonnes, with the potential to produce 400,000 tonnes more. However, the cost for these pellets is more than forest residues at about \$100-\$117/ODT [103].

Standing timber is the component of the Annual Allowable Cut that is deemed commercially unviable for lumber or pulp production. These timbers generally remain unharvested due to factors such as the long distance from the marketplace, small size and/or low quality (e.g. rot or damage), and difficulty in harvesting. While significant supplies of standing timber exist in B.C. (2.9 M ODT), acquiring these resources would require a new supply chain and system, increasing the cost of acquisition to about \$138-\$199/ ODT [103].

The purpose of this case study is to determine the ideal location and conditions for a 100 MLPY pyrolysis facility and biorefinery in B.C. based on GHG emission reductions. This size was chosen because it is large enough to provide some economy of scale while small enough to minimize feedstock collection costs. For a 100 MLPY facility, approximately 400,000 ODT/yr of biomass is required [103]. This amount of forest residue is available in several regions (~750,000 ODT/yr) in the province (Vancouver Island, Vancouver Mainland, and Cariboo) at a collection cost of between \$66 and \$80 [103]. Other areas, such as Prince George, have approximately this supply in pellets (~350,000 ODT/year). Based on feedstock supply and proximity to existing refinery and logistical infrastructure, Vancouver Mainland, Vancouver Island, and Prince George are considered possible sites for production. This chapter analyzes the LCA of biojet fuel produced from these three possible pyrolysis sites to determine the optimal location for construction in terms of the carbon intensity of the fuel. Each site utilized the prevalent biomass resource of the region for production, which was forest residue for the Vancouver area and wood pellets for Prince George. The GHG emissions for these pathways were optimized by performing a sensitivity analysis on the most influential variables for pyrolysis discovered in Chapter 2. Through this analysis, “Greenhouse gas hotspots” and potential areas for GHG reduction improvements were determined to maximize the GHG reduction potential of the bio-refinery design.

## 3.2 Methodology

In this work, GHGenius 4.03 was used to perform the lifecycle analysis of biojet fuel for the scenarios considered. While this model has some challenges, it is the only model that is populated with regional B.C. data. GHGenius is also required by the Government of British Columbia for all carbon intensity calculations under the Low Carbon Fuel Standard and is likely to be adopted by the federal Clean Fuel Standard, as well [125]. As a result, utilizing GHGenius makes the most sense for a B.C. case study. The main challenges with this model, as discussed in Chapter 2, are the default use of the displacement co-product method, the inclusion of land use change, and the difficulty in use. However, all these challenges were mitigated within this study. With forest and mill residue feedstocks, land use change was not considered, so this drawback was avoided. Furthermore, it is possible to change the default co-product allocation method in the model from displacement. All scenarios utilized the energy allocation method as default, since this was determined to be the preferred method after economic allocation (see Chapter 2). Finally, while GHGenius is challenging to use and the data not always transparent, access to the model developer, (S&T)2 Consultants, for this study helped mediate this challenge.

All scenarios in this study utilized specific B.C. data whenever possible. The majority of the data used was obtained from the GHGenius model database but was complimented with additional data from literature for processes not included in the model. The ISO 14040 guidelines for LCA were followed in this work and details from each stage are given below.

### 3.2.1 LCA goal and scope definition

The goal of the lifecycle assessments performed in this chapter was to compare the greenhouse gas reductions from three scenarios in British Columbia and to identify options for even greater GHG emission reductions. In lifecycle analysis, the emissions from the entire lifecycle of a fuel are considered. The final value represents not just the emissions associated with the fuel combustion, but all of the emissions that result from the production of the product. All scenarios used lignocellulose biomass as the feedstock, either in the form of forest or pellets (mill residue). Forest residues were considered to be any wood leftover in the forest after harvesting operations, typically comprised of branches, tree tops, and bark and sorted into slashpiles. These slashpiles are often burned to eliminate fuel that could contribute to forest fires, though credit for this loss in emissions was not included in this LCA. As forest residue was considered a waste of logging activities, no up-stream emissions associated with primary logging activities were considered. Only emissions from the collection and transport of these forest residues to the pyrolysis facility were included in this study.

Mill residues were considered to be any residues from sawmill activities, such as sawdust and shavings. These residues are considered a waste of the lumber industry, so no upstream emissions from logging or sawmill activities were included. While counting mill residue as a waste is common practice among LCA practitioners and LCA models, this practice is controversial due its common use as wood pellet feedstock. Wastes are defined as any material with practically no economic value, but this changes as soon as a new use is found [40]. While this study assumed the traditional definition of forest and mill residues as waste, the GHG emission difference if mill residue were considered a by-product of sawmill production rather than a waste was considered as well.

Since both feedstocks in this study are considered wastes, no land use change or N<sub>2</sub>O emissions were included in these scenarios. Based on the arguments presented in Chapters 1 and 2 about the shortcomings of the displacement allocation method, this co-product allocation method was not used even though it is the default for GHGenius. Instead, energy allocation was used for all pathways. Though economic allocation is generally preferred, energy allocation is appropriate in this situation as all co-products produced via pyrolysis are energy products. To determine how this change affected the results, a sensitivity analysis was performed on each pathway using displacement and mass allocation. Economic allocation, which was the preferred method according to Chapter 2, is not available in GHGenius so was not considered.

Chapter 2 identified that methodological variables can have a significant impact on LCA results, so all methodological assumptions used in this analysis are presented in Table 15. Details about process variables and assumptions are provided in Sections 3.2.2 and 3.2.3.

**Table 15: LCA methodological assumptions**

Analysis Year:	2017
Location:	B.C.
Allocation Method:	Energy
Land Use Change:	Not included
Cultivation N <sub>2</sub> O:	Not included
Functional Unit:	gCO <sub>2</sub> eq/MJ refined bio-oil
Emissions included in CO <sub>2</sub> eq:	CH <sub>4</sub> , N <sub>2</sub> O, CO <sub>2</sub>
Global Warming Potential (GWP):	IPCC 2007
Atmospheric Lifetime:	100 years
Heating Value of fuels:	HHV

All scenarios in this study were based on the year 2017 and included emissions from CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O as greenhouse gases. The greenhouse gas emissions were presented in terms of gCO<sub>2</sub> equivalent per MJ of bio-oil based fuel (functional unit), which is a mix of all gasoline, jet, and diesel fractions produced from pyrolysis upgrading, using IPCC 2007 Global Warming Potentials (1, 25, and 298 for each GHG, respectively). The HHV of fuels was used for all calculations and all the weights in the LCA inventories are on a dry basis unless otherwise stated. These assumptions are consistent with most LCA literature.

### 3.2.2 System Boundary and Scenarios

In order to supply a 100 MLPY pyrolysis facility, a consistent supply of approximately 400,000 ODT/year of biomass is needed [103]. Furthermore, it is likely that the use of existing infrastructure will be needed for the process to be economical. Proximity to wood processing facilities, petrochemical plants, oil refineries, transportation infrastructure and airports are all important considerations. Oil refineries and petrochemical plants have hydrogen resources, natural gas pipelines, and fuel storage and handling equipment that could be shared if the bio-oil upgrader were built nearby. Furthermore, it may become possible at a future date to co-process the stabilized bio-oil with petroleum in the Fluid Catalytic Cracker or Hydrocracker, which would further reduce the capital expenditure of bio-oil upgrading [126]. While bio-oil upgrading via co-processing with petroleum products was not considered in this study, this synergy has significant potential to reduce capital costs. While there is little data available currently to determine the GHG emission benefits of co-processing compared to upgrading the bio-oil separately, this upgrading scenario should be modeled in future work once this information becomes available. For the scenarios in this study, the bio-oil upgrader was assumed to be built nearby existing refineries whenever possible in order to utilize some of the hydrogen and logistical infrastructure, but was considered a stand-alone facility that only refined bio-oil product.

Based on the proximity to biomass resources and airports, three pyrolysis locations were chosen for this analysis: Coastal Mainland, Vancouver Island, and Prince George. The bio-oil was assumed to be upgraded separately near the Parkland oil refinery in Burnaby for Scenarios 1 and 2, whereas the pyrolysis and upgrader were assumed to be co-located near the Husky oil refinery in Prince George for Scenario 3. Details about each of the three scenarios are given below.

#### 3.2.2.1 Scenario 1: Coastal Mainland

The coastal mainland surrounding Vancouver, B.C. offers significant biomass resources, as well as access to the Burnaby refinery and the Vancouver airport. Aldergrove, B.C. was selected as the location for the pyrolysis plant in this analysis due to its proximity to inexpensive forest residues. Over 758,000 ODT/year

of forest residues are available in the area within a supply radius of 155 km [103]. When delivered to Aldergrove, B.C., the cost of collection and delivery of these feedstocks is only about \$66/ODT. Furthermore, about 190,676 ODT of excess mill residues are also available in the Vancouver Island and Coast region [124]. These volumes exceed the required volume of 400,000 ODT/year, so a combination of feedstocks would likely be used, based on market price. The base case of this scenario was modeled using a supply of 100% forest residue but is compared to ratios of mill residue in the sensitivity analysis. The volume of biomass required for the plant (400,000 ODT) is available within 113 km of the facility, so an average acquisition distance of 79.6 km was assumed.

Forest residue is considered a logging waste, so emissions from upstream logging activities were not included in the lifecycle analysis. However, all the emissions from processing post-slashpile were included. At the harvest site, this includes all emissions from loading, chipping, and transport of the forest residue chips to the pyrolysis facility. All processes included in the lifecycle assessment of Scenario 1 are included in Figure 20.

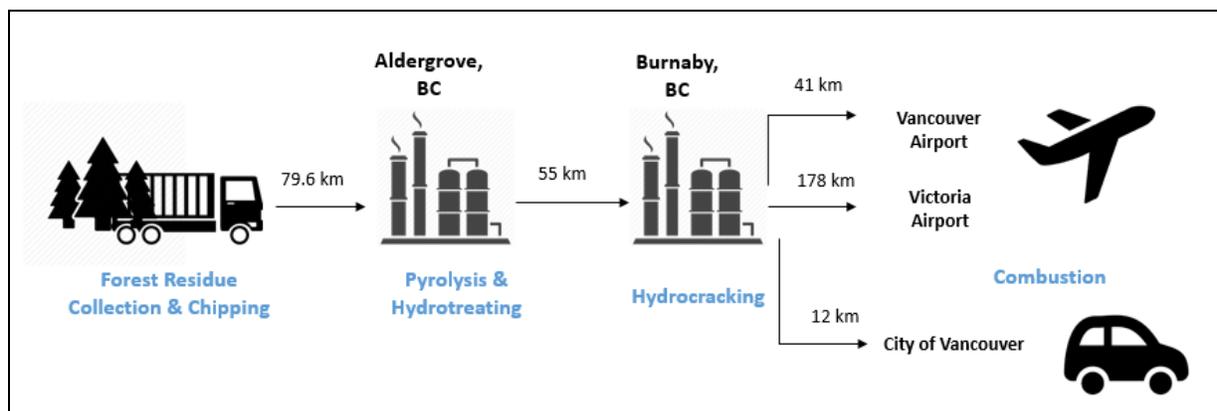


Figure 20: Biofuel lifecycle for Coastal Mainland, B.C.

Once at the pyrolysis facility, the forest residue was dried to 10% moisture content and ground to 2mm [114]. This powder was then introduced into the pyrolyzer, where it was heated in the absence of oxygen to produce pyrolysis oil, biogas, and char. This study modeled the Envergent/Ensyn RTP technology for the pyrolysis process, since they are one of the few commercial pyrolysis plants in operation [127]. Envergent utilizes a bubbling fluidized bed reactor for pyrolysis production, which operates at 500 °F [127]. The RTP technology is conventional, un-catalyzed fast pyrolysis, so is thermal only and uses hot sand as the heat transfer medium. After spending 1-2 seconds in the reactor, the wood particles are pyrolyzed

and the condensable vapors are collected and condensed into bio-oil. All of the biogas and char produced are burned internally to dry and grind the feedstock.

The bio-oil is unstable due to its high oxygen content, so must be further upgraded before it can be used as a transport fuel. The oil must first be hydrotreated to reduce the acidity and stabilize the oil and then hydrocracked to produce hydrocarbons in the diesel, gasoline, and jet fuel range. In this scenario, the hydrocracking and refining occurred at a facility built near the Chevron oil refinery in Burnaby, B.C. Building the bio-refinery next to the oil refinery allows the producers to take advantage of existing infrastructure and utilize available hydrogen sources. It may be possible to perform the hydrocracking with petroleum crude in the existing petroleum hydrocracker, but this strategy is still under development and was not analyzed here. The hydrogen in this study was assumed to come from steam methane reforming [114].

Once upgraded, the fuel was transported by truck to the City of Vancouver, the Vancouver airport, and the Victoria airport for use in airplanes and trucks. As jet fuel was the main product being assessed, the final emissions from combustion in a jet turbine engine were included. Figure 20 indicates the distances that each feedstock/fuel must travel between each process.

Most of the data for the pyrolysis and upgrading process was obtained from the GHGenius.net databank. This data originated from papers by PNNL and operating data from Ensyn [25]. Whenever information was unavailable in this databank, original data from S. Jones (PNNL) was utilized [25]. All input data used for this scenario is included in Appendix IV.

#### *3.2.2.2 Scenario 2: Vancouver Island*

Vancouver Island is predominately forested and is home to the largest privately managed forests in B.C (850,000 hectares [128]). While the island is only a small portion of B.C.'s 159 M hectares of forest land, the proximity of the island to infrastructure and the airport in Vancouver make it a prime location for biomass supply. The closest refinery to the island is the Parkland refinery in Burnaby, B.C., so the bio-oil was sent to Burnaby for refining after stabilization at a pyrolysis facility in Parksville, B.C. Forest residue availability around Parksville is over 758,000 ODT/year within a 155 km supply radius, averaging approximately \$67/ODT [103]. Parksville was chosen as the location of the pyrolysis facility in this study due to the proximity to inexpensive forest residues, proximity to the port of Nanaimo, and local lumber

facilities. Like Scenario 1, the volume of biomass required for the plant (400,000 ODT) was available within 100 km of the facility, so an average acquisition distance of 79.6 km was assumed.

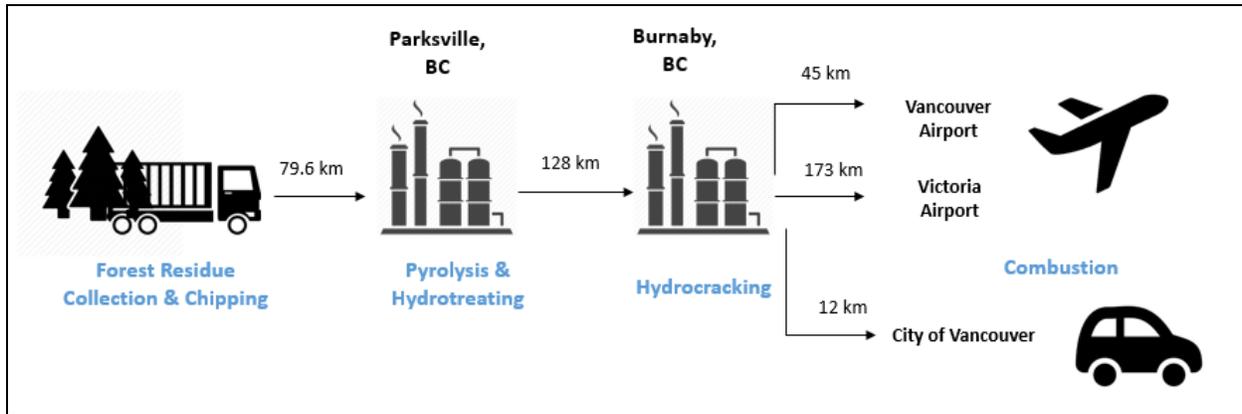


Figure 21: Biofuel lifecycle for Vancouver Island, B.C.

All pyrolysis process and hydrotreating/cracking conditions were the same as Scenario 1, apart from transport distances of the stabilized bio-oil. Figure 21 shows the processes and transport distances assumed for this scenario. All specific input data for this scenario is available in Appendix IV.

### 3.2.2.3 Scenario 3: Prince George, B.C.

Prince George, considered the forestry capital of B.C, is located about 780 km north of Vancouver [129]. It is surrounded by forests, which makes it an ideal location for lumber and other wood-based industries. Over 25% of the city's income comes from forestry related activities, such as the 10 lumber and 3 pulp mills in the area [129]. In the past decade, pellet production has developed from the sawmill residues from these industries. Pacific Bioenergy operates a pellet plant in Prince George with a capacity of 350,000 tonnes of pellets/year and other pellet plants are located within a 120 km radius that would increase the area's biomass availability to 740,000 tonnes of pellets/year [103]. These plants are the Pinnacle Pellet Meadowbank plant in Strathnaver (220,000 tonnes pellet/year), the Premium Pellet plant in Vanderhoof (140,000 tonnes pellet/year) and the Vanderhoof Specialty Wood Products plant in Vanderhoof (30,000 tonnes pellet/year) [103]. Prince George is also home to the Husky oil refinery and an airport, making Prince George an ideal location for a bio-refinery.

Scenario 3 calculates the Greenhouse Gas emissions of a bio-refinery co-located at the Husky refinery at Prince George utilizing pellet feedstock. While forest residues were also available in the area (over

299,000 ODT/year within 195 km of Fort St. James), pellets can be stored easier than forest residues and are already dried and ground [103]. Most pellet companies utilize mill residue as their feedstock, but some utilize a small portion of forest residue. This competition further reduces the attractiveness of using forest residues for biofuel production, as supply of economical forest residue is difficult to obtain. This study considered pellet production from 100% mill residue as the pyrolysis feedstock for the base case but considers other ratios with forest residues in the sensitivity analysis. Figure 22 shows the various processing that will be considered under this scenario and the transportation distances of each intermediate.

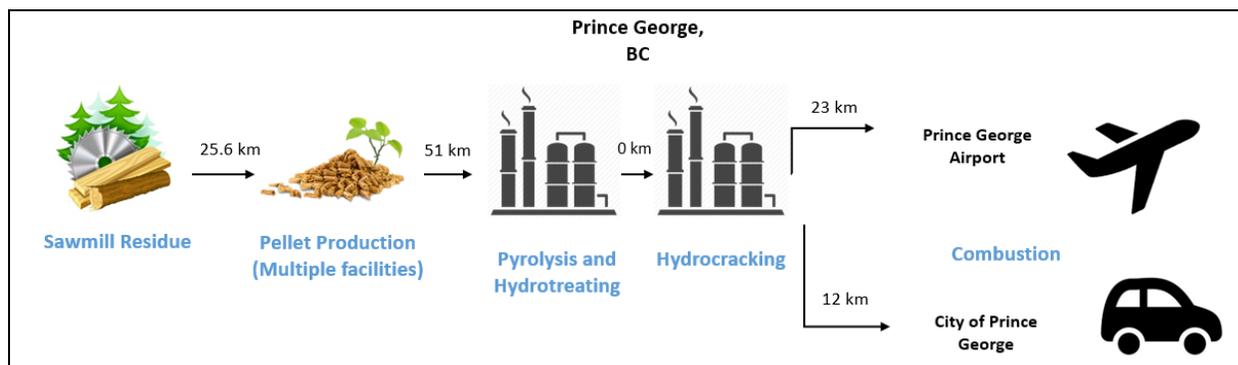


Figure 22: Biofuel Lifecycle assessment for Prince George, B.C.

While most pellets are produced for export overseas, this scenario analyzed the potential for this feedstock to be used locally for biojet production. The pellet production data used in this study is based on the operating data from B.C. pellet plants [130]. Since sawmill residue is considered a waste of lumber production, only emissions from the transportation of the sawdust to the pellet plant and subsequent processing were included in the analysis. No emissions from sawmill or logging activities were included in the base-case, though the impact of this assumption is assessed in the sensitivity analysis. This study also assumed that pellet wood is similar in composition to that of forest residue and mill residue.

In this scenario, the feedstock was dried at the pellet plant rather than at the pyrolysis facility. This leaves pyrolysis biogas available for other uses, such as hydrogen production or combustion. In this scenario, the extra biogas was sold and assumed to displace Liquefied Petroleum Gas (LPG) in the marketplace, based on the assumption in GHGenius and GREET [58]. The energy used for drying in the pellet facility was assumed to be mostly from biomass combustion, according to B.C. pellet facility reports [130].

The pyrolysis and upgrading processes for this scenario are identical to that of the previous two scenarios, but this time the pyrolysis and upgrading facilities are at the same location. This eliminates the transportation of the bio-oil between the two facilities and provides greater potential for synergy, such as waste heat recycling and shared hydrogen and logistical infrastructure. These synergies increase the attractiveness of co-locating the two facilities, but previous studies have shown that it may not have a large impact on overall GHG emissions [60]. This assumption is tested in the sensitivity analysis (section 3.4.1.4) to determine the impact of co-locating. Since the GHG emission difference between integrated and distributed facilities is likely minimal when heat recycle synergies are not included, this study considered similar energy requirements between the two configurations.

After pyrolysis and upgrading, the fuel was transported via truck to the City of Prince George and the Prince George Airport for combustion. Figure 22 displays the average distances that the feedstock/fuel must be transported by truck along the entire supply chain. All other process parameters and material characteristics assumed in this analysis can be found in Tables 27 and 28 in Appendix IV.

### 3.2.3 Inventory Analysis

#### 3.2.3.1 Biomass Production

This process includes the energy use and emissions associated with collecting forest residues or producing wood pellets. Forest residues were assumed to consist of tree tops, branches and bark located in slash piles after logging activities. Emissions from both a wood chipper and skid loader were included in this process to prepare the forest residues for transport from site. The wood chipper was assumed to chip 35 tonnes of wood per hour and to consume 15 gallons of diesel fuel per hour. The skid loader could load 50 tonnes of wood chips per hour into the truck while consuming of 2.72 gallons of diesel an hour [131]. This equated to a total of 1.84 L diesel per tonne of wood consumed during the preparation of forest residue. Forest residue was assumed to have 50% moisture content, which is the GHGenius model default and common in most studies (see Table 22 in Section 3.4.5) [114]. All moisture contents in this study are reported on a dry basis.

Pellets were assumed to be produced from mill residue, which consists of sawdust and shavings. The average moisture content of the mill residue when it arrived at the pellet facility was assumed to be 47% (wet basis), which was based on industrial surveys [130]. The mill residue was then ground, dried and densified into uniform wood pellets. Based on industrial surveys of B.C. pellet producers, 490 MJ of electricity, 23.5 MJ of diesel, 6.16 MJ of propane and 1,059 MJ of wood waste were used to produce 1 tonne of wood pellets [130]. These surveys were conducted by the Wood Pellet Association of Canada in

2008 and 2009. The input data for wood pellet production included in this study are included in Table 27 in Appendix IV. The moisture content of the finished pellets was assumed to be 9% (wet basis), based on shipping port data [130].

#### *3.2.3.2 Biomass Transportation*

Biomass was assumed to be transported by Heavy Duty Vehicle (HDV) diesel trucks from the logging site or pellet plant to the pyrolysis facility according to GHGenius [58]. Forest residue was gathered from within a 113 km radius of the pyrolysis facility for both Scenarios 1 and 2 (based on 400,000 ODT residue/year requirement), with an average travel distance of 79.6 km. These distances were based on the annual available forest residue supply according to the Industrial Forestry Service Ltd. and FP Innovations [132] [133]. Trucks were assumed to return empty and have a fuel efficiency of 34.86 L/ 100 km or 0.0056 L/tonne-km [58].

For pellet production, mill residue was assumed to be transported 25.6 km by truck from the sawmill to the pellet plant [134]. This value was based on the industrial survey data from the Wood Pellet Association of Canada. The travel distance for the wood pellets to the pyrolysis facility was based on the average weighted distance from the 4 pellet facilities in the Prince George area to the Husky refinery. These values were 20 km for the Pacific Bioenergy refinery, 65 km for Pinnacle Pellet, 110 km for Premium Pellet, and 115 km for Vanderhoof Specialty Wood Products [103]. Each pellet plant was assumed to supply 50% of their total annual pellet production, except Pacific Bioenergy (59%), providing 400,000 tonnes pellets/yr total (Pacific Bioenergy (205,000), Pinnacle Pellet Meadowbank (110,000) Premium Pellet (70,000) and Vanderhoof (15,000)). The weighted average distance of these facilities to the pyrolysis facility based on supply was 50.9 km. This value was further adjusted in GHGenius to account for the higher energy density (due to lower moisture content) of the wood pellet over forest residue (which is assumed by the model), according to the mass ratio of 0.62, respectively [130].

#### *3.2.3.3 Biomass Pre-treatment*

Biomass must be ground to 2mm and dried to 10% moisture content (wet basis) before it can be used in pyrolysis, so pre-treatment of the forest residue chips was required. The theoretical energy required for drying is 2.44 MJ/kg H<sub>2</sub>O evaporated, but typically more is needed. O'Connor suggests that 3 MJ/kg H<sub>2</sub>O is required, while a report by M. Wright suggests 3.6 MJ/kg H<sub>2</sub>O [90] [92]. With these energy requirements, approximately 1.2 – 1.44 MJ per kg of wood (dry) is needed. This heat is generally supplied by the combustion of the bio-gas and/or char produced by the pyrolysis process. The energy for drying and grinding were included in the pyrolysis process within GHGenius. The grinding energy was included in this

study but unknown, as it was aggregated into the total energy required for pyrolysis in GHGenius. However, data by P. Meyer et al. indicated that the energy required for grinding was approximately 0.25 MJ per kg of wood [135].

Energy for biomass pre-treatment was included in the biomass production stage for pellets, as both grinding and drying are required prior to pellet production. However, additional grinding was still required at the pyrolysis facility prior to pyrolysis to reduce the pellets to 2mm. Therefore, only the drying energy was removed from the GHGenius assumptions for this feedstock, to eliminate double counting. The energy required to dry mill residue and forest residue was assumed to be the same (1.98 – 2.38 MJ/L bio-oil).

#### *3.2.3.4 Bio-oil production*

Bio-oil was produced via fast pyrolysis from wood feedstock in a 214 Million Liter/year facility (based on a 100 MLPY of refined bio-oil fuel and a yield of bio-oil to refined bio-oil of 2.14:1) [58]. All data used in the model was based off ENSYN's Rapid Thermal Processing (RTP) technology, as it was the only commercial process that has operated on a long-term basis [114]. In this process, woody biomass (<10% moisture content and <2mm) enters the reactor where it is rapidly heated to 500 °C at ambient pressure through contact with a cyclone of hot sand. All vapours are rapidly condensed within 2 seconds to produce bio-oil, while the char is separated and combusted to heat the sand, and the bio-gas is combusted to pre-treat (dry) the biomass feed. When the moisture content of the biomass feed is 50% (i.e. forest residue), all bio-gas is combusted internally and there are no co-products from the process. When the moisture content is lower than 50%, as in the case of pellets, not all of the bio-gas is consumed and some is left as co-product. In GHGenius 4.03, this bio-gas was assumed to displace Liquefied Petroleum Gas (LPG), which is what was assumed in this study. In both scenarios, all char was consumed for process heat [114].

The yield of bio-oil assumed in this work was 73% (mass basis), with biogas and char yields of 7% and 20%, respectively. These yields were based off ENSYN's process and are included as the default values in GHGenius [114]. While the bio-oil yield is only slightly lower than that reported by S. Jones (75%), who also considered woody biomass feedstock, the char value is closer to that reported by M. Wright from NREL when using corn stover (19.4-19.5%) [25] [92]. The bio-oil composition assumed in this study is given in Table 28 in Appendix IV and is very comparable to those reported by S. Jones [25]. The HHV of bio-oil assumed in this work was 21 MJ/L and the specific gravity was 1.2, as per GHGenius defaults [58]. The bio-oil characteristics and yield were assumed to be similar for both forest residue and pellets. As discussed earlier, there are compositional differences between forest residues and pellets that will likely affect bio-

oil yields and composition (i.e. percentage of bark, dirt, glues, and additives). However, information about how the feedstock compositional variations affect bio-oil production is not yet available, so is outside the scope of this work.

Energy requirements for this process were based on the ENSYN process and included as GHGenius defaults. This inventory data is provided in Table 27 in Appendix IV and was based off 95% of operating energy expenditure and 5% of start-up energy requirements.

#### *3.2.3.5 Bio-oil Upgrading*

Bio-oil contains a large amount of water and oxygen, which must be removed before it is suitable for use as a transportation fuel. This is usually performed in two steps: the first to reduce the acidity and stabilize the oil (hydrotreating) and the second to remove all remaining oxygen and crack the molecules to diesel, jet and gasoline-range products (hydrocracking) [86]. This secondary stage could either be performed at a standalone refinery or co-refined with petroleum crude in a hydrocracker. Interest around the latter case has increased recently, as this would reduce capital costs surrounding this process and utilize existing infrastructure. Co-processing trials have been performed successfully in trials at up to 10% stabilized bio-oil, but how much of the initial bio-oil ends up in the fuel is unclear [126]. While this strategy holds significant potential for reducing GHG emissions and costs in the industry, it was not considered here.

In this study, the two upgrading stages were performed in separate locations for Scenarios 1 and 2, and in the same location for Scenario 3. While this makes a difference in capital cost and logistics, energy recycling was not included between processing stages as it is outside the scope of this study. As a result, the energy and hydrogen requirements for both scenarios were considered the same, though it would be interesting to determine how steam recycle between plants would affect the overall results in future work.

After pyrolysis, the bio-oil was filtered, pressurized, and pre-heated before entering the hydrotreater [25]. Hydrotreatment is typically performed in a hydrogen-rich environment (5 wt% H<sub>2</sub>) in two stages at increasing temperature to protect the unstable bio-oil from degradation. The first stage occurs at a low temperature (240 °C) and high pressure (2500 psig) over a cobalt molybdenum catalyst to stabilize the oil and reduce acidity [25]. The second phase occurs at a higher temperature (370 °C) and high pressure (2015 psig), but a lower space velocity over the same CoMo catalyst. The schematic for this process is presented in Figure 23. The process yields assumed in this study were 46.7% hydrotreated bio-oil, 40.2% wastewater,

and 13% off-gas [25] [114] (see Table 22 of Section 3.4.1.5 for comparison to other literature yields). Other inventory data for this process is included in Table 27 of Appendix IV. The specific gravity of the hydro-processed bio-oil was 0.87 and included about 1.5% oxygen.

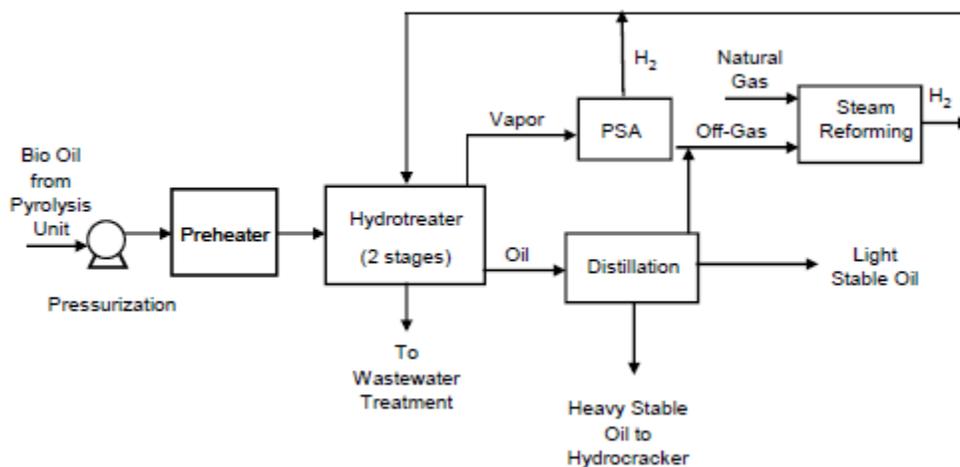


Figure 23: Bio-oil hydro-treating process [25]

The off-gas of the hydrotreatment was sent through Pressure Swing Absorption (PSA) to recover the unused hydrogen gas. The low-pressure PSA stream was rich in hydrocarbon gases (methane), so was combined with the natural gas stream prior to Steam Methane Reforming (SMR) [25]. When co-located at the refinery with the hydrocracker and distillation column, the light gases from distillation and cracking were also recycled with the off-gas to replace natural gas in the SMR. When the hydrotreatment and hydrocracking occurred separately in different locations, two SMR facilities were required. Figure 24 shows the hydrocracking process as a separate facility, though this could be integrated into the hydrotreatment process in Figure 23 when co-located. The energy and emissions associated with wastewater treatment were not included in this analysis as the process is outside the traditional LCA system boundary.

After hydrotreatment, the stable bio-oil was distilled into heavy, light, and gaseous fractions. The heavy fraction was hydrocracked to diesel, jet and gasoline-ranged hydrocarbons and then mixed with the light fraction. The conditions of hydrocracking are generally more severe than hydro-processing, with temperatures of 400-450 °C (427 °C used in study) and pressures of 1200 – 2000 psig (1280 psig used in

study) over a nickel-molybdenum catalyst [25] [92]. Off-gas from the preliminary distillation and hydrocracker was recycled to offset some of the natural gas used to produce hydrogen for the hydrocracker. The ratios of each fraction (by mass) were assumed to be 41.25% diesel, 13.75% jet, and 45% gasoline [114] [28]. Though the GHG emissions in GHGenius were displayed in terms of a gasoline/diesel blend, the change was negligible when converted to jet fuel. This similarity occurs whenever the emissions are allocated by energy and the HHV of the co-product fuels are similar, which was the case here (see Appendix III for proof). As a result, the emissions associated with the refined bio-oil and biojet were assumed to be the same.

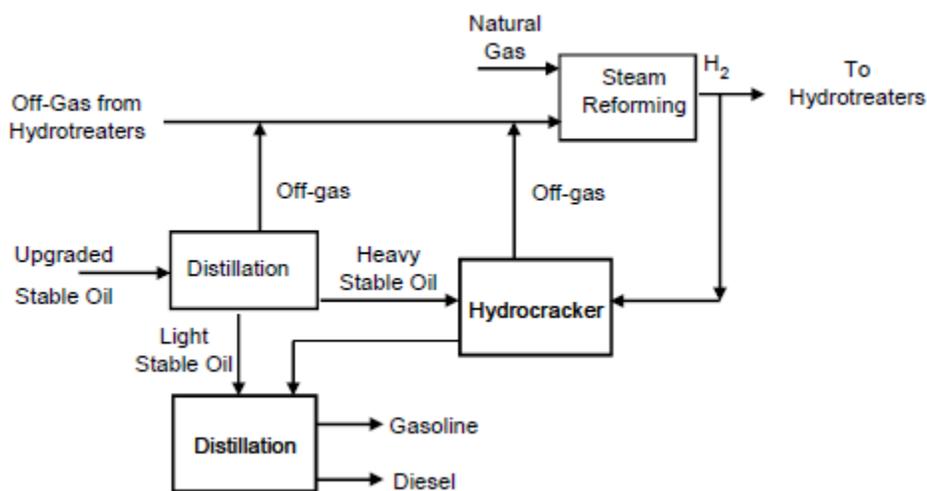


Figure 24: Hydrocracking of hydro-treated bio-oil

Energy and mass balance data for this study was based off the PNNL study by S. Jones, which performed lab-scale testing of bio-oil upgrading from similar woody feedstocks and had a more complete mass and energy balance data than other studies (e.g. NREL, UOP) [25]. All inventory data used in this study is presented in Table 27 of Appendix IV. All hydrogen was assumed to come from steam methane reforming (SMR) of natural gas and off-gas unless otherwise noted. Hydrogen can also be produced from the biogas produced during pyrolysis through SMR, through Aqueous Phase Reforming (AQR) of the aqueous bio-oil phase, or electrolysis. A few of these other sources of hydrogen are explored in the sensitivity analysis.

### 3.2.3.6 Product Transportation and Distribution

After upgrading, the refined bio-oil was distributed to local airports and municipalities for distribution. In Scenarios 1 and 2, the jet fraction was distributed to the Vancouver and Victoria airports by truck and by

barge. The Vancouver airport is approximately 41 km from the Parkland refinery, and the jet fuel is assumed to travel this distance by pipeline [136]. Jet fuel was assumed to travel from the refinery to the Victoria international airports 150 km by barge and 28 km by truck. The gasoline and diesel fractions were assumed to be transported to the city of Vancouver, but as these are co-products, their transportation emissions were not included in the LCA. It was assumed that 75% of the bio-jet fuel was consumed by the Vancouver Airport and 25% was shipped to the Victoria airport. Distribution distances once at the airport were assumed to be 4 km by truck for all scenarios, based on the size of the Vancouver airport.

In Scenarios 1 and 2, the hydrotreated bio-oil must also be transported from the pyrolysis facility to the Parkland oil refinery for hydrocracking. In the first scenario, the bio-oil is assumed to travel by barge to the refinery (39 km by truck to Nanaimo port, then 90 km by barge to Burnaby), while the bio-oil is assumed to travel 55 km by truck from Aldergrove in the second scenario.

In Scenario 3, the pyrolysis and hydrocracker are in the same location, so there is no transportation of the intermediate bio-oil. Once fully refined, the jet fraction of bio-oil is assumed to be transported to the Prince George Airport by truck (23 km) and all other liquids to the city of Prince George (20 km). As in the other scenarios, only the emissions associated with the biojet transportation are included.

### 3.3 Results

The lifecycle results using GHGenius for Scenarios 1, 2, and 3 are presented in Tables 16, 17, and 18, respectively. The total GHG emissions are given for each pathway, along with the emissions from each individual process. Land use change was not included in this analysis, so there are no emissions associated with fertilizer manufacture, land use change, or feedstock upgrading. Due to the high moisture content of the forest residue, Scenarios 1 and 2 did not produce any biogas or char co-product, so there are no emission credits given in Tables 16 and 17 under “Emissions displaced.” Scenario 3 did produce a biogas co-product, which is represented by the negative value in Table 18. This value represents the emissions allocated to the biogas co-product, so is removed from the system which represents the emissions from producing biojet fuel. This negative value is mostly a result of accounting – the forest residue still needs to be dried just as much as in Scenario 1, but in this case the drying occurred at the pellet mill. However, rather than using biogas to dry the pellet as in Scenario 1, wood residue biomass was used for drying [134]. The emissions associated with drying were therefore included under “Fuel production” in Table 18.

The final GHG emission values of each table are presented in terms of gCO<sub>2</sub>eq/GJ and include all greenhouse gas emissions (CH<sub>4</sub>, CO<sub>2</sub>, N<sub>2</sub>O) converted to carbon dioxide equivalent according to their global warming potentials. These values are most commonly presented in terms of MJ, so were therefore 25.7 gCO<sub>2</sub>eq/MJ and 26.1 gCO<sub>2</sub>eq/MJ for Scenarios 1 and 2, accordingly. Scenario 3 gave GHG emissions between 28.3 and 28.4 gCO<sub>2</sub>eq/MJ depending on the amount of co-product biogas available from pyrolysis (1.98-2.38 MJ). If compared against the B.C. average for jet fuel (0.06%), which is 88.9 gCO<sub>2</sub>eq/MJ according to GHGenius default for regional jet (0.015 GJ/tonne-km) [58] this translates into reductions of 68.2% to 71.1% [137].

**Table 16: Greenhouse Gas emissions (gCO<sub>2</sub>eq/GJ) from LCA of Scenario 1: Coastal Mainland, B.C. Pathway**

Source of Emissions	Residue procurement, Pyrolysis, Upgrading,		Fuel Use	Total
	Distribution			
Region	B.C.		B.C.	B.C.
Fuel	Refined bio-oil		Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue		Forest Residue	Forest Residue
Fuel dispensing	38			38
Fuel distribution and storage	296			223
Fuel production	22,369			22,369
Feedstock transmission	4,075			2,265
Feedstock recovery	653			653
Feedstock upgrading	0			0
Land-use changes, cultivation	0			0
Fertilizer manufacture	0			0
Gas leaks and flares	0			0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0			0
Emissions displaced	0			0
Emissions from fuel use			195	195
<b>Total</b>				<b>25,743</b>

**Table 17: Greenhouse Gas emissions (gCO<sub>2</sub>eq/GJ) from LCA of Scenario 2: Vancouver Island, B.C. pathway**

<b>Source of Emissions</b>	<b>Residue procurement, Pyrolysis, Upgrading, Distribution</b>	<b>Fuel Use</b>	<b>Total</b>
Region	B.C.	B.C.	B.C.
Fuel	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue	Forest Residue	Forest Residue
Fuel dispensing	38		38
Fuel distribution and storage	296		223
Fuel production	22,369		22,369
Feedstock transmission	4,408		2,599
Feedstock recovery	653		653
Feedstock upgrading	0		0
Land-use changes, cultivation	0		0
Fertilizer manufacture	0		0
Gas leaks and flares	0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0		0
Emissions displaced	0		0
Emissions from fuel use		195	195
<b>Total</b>			<b>26,076</b>

**Table 18: Greenhouse Gas emissions (gCO<sub>2</sub>eq/GJ) from LCA of Scenario 3: Prince George, B.C. pathway**

Source of Emissions Region	Pellet production B.C.	Pyrolysis, Upgrading and Distribution B.C.	Fuel Use B.C. Refined bio-oil	Total B.C. Refined bio-oil
Fuel	Pellets Sawmill residue	Refined bio-oil	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		234		215
Fuel production		22,398		22,398
Feedstock transmission	2,599	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		-687 to -812		-687 to- -812
Emissions from fuel use			195	195
<b>Total</b>				<b>28,385 to 28,260</b>

The lifecycle analysis results presented in Table 18 assumed that the sawdust (mill residue) was a waste product from the sawmill. Therefore, this feedstock was allocated zero emissions. However, if this assumption were changed and the mill residue were considered a co-product of production, then the pellet production GHG emissions would increase to 14,245 gCO<sub>2</sub>eq/GJ, which would bring the total LCA GHG emission value from 28.3 gCO<sub>2</sub>eq/MJ to 37.6 gCO<sub>2</sub>eq/MJ. Inventory data for these processes were obtained from work by A. Pa and are presented in the Appendix [134].

### 3.4 Discussion

As Tables 14 through 16 indicate, all scenarios provide significant GHG reductions for the aviation sector. The Vancouver mainland, Vancouver Island, and Prince George pathways produced GHG emissions of 25.7, 26.1, 28.3 gCO<sub>2</sub>eq/J respectively, or reductions of 68.2% to 71.1% over the Jet-A1 Kerosene used in B.C. (88.9 gCO<sub>2</sub>eq/MJ fuel). These results are similar to the results of other pyrolysis studies utilizing GHGenius (see Table 15 in Section 2.3), though lower than those that utilize forest residue feedstock.

According to Tables 16-18, the pyrolysis process and upgrading (fuel production) are the most emission intensive processes for these pathways, contributing to 78% to 87% of all emissions. Bio-oils contain a significant amount of oxygen which must be removed during hydro-processing before it can be used as a drop-in fuel. The removal of this oxygen requires a significant amount of hydrogen. 96% of hydrogen worldwide is produced via steam methane reforming, which was the method assumed in these analyses. This process is very emission intensive, generating around 13.7 gCO<sub>2</sub>eq per gram of hydrogen produced [138]. If the hydrogen were produced differently, such as through electrolysis or reforming of the aqueous bio-oil phase, the overall GHG emissions would likely be substantially reduced. This hypothesis will be tested by performing a sensitivity analysis on the hydrogen source used in these scenarios.

After fuel production, feedstock transmission and recovery are the most emission intensive processes. The transportation of feedstock contributed to 9-10% of all emissions for Scenarios 1 and 2, while the recovery of feedstock contributed to approximately 17% of the emissions for Scenario 3. The high emission levels of feedstock transportation in the first two scenarios is because of the relatively large distances (79.6 km average) required to obtain enough forest residue to maintain a 100 MLPY facility. If the production facility were 50 MLPY, the transport distances required would shrink by roughly 29.3% (assuming equally scattered residue due to lack of aggregate data), reducing the GHG emissions of the feedstock transport equally. Alternatively, if the size of the production facility were increased, the overall GHG emissions would increase. This emission increase would be about 22.5% if the facility were increased in size by 50% (150 MLPY) if only the transport radius needed to be enlarged to account for the increased requirement for biomass feedstock. This is the case for Scenarios 1 and 2, as the total available biomass in each region was 758, 880 ODT/year, which is enough to support a 190 MLPY facility [103]. If a larger facility were desired (e.g. 200 MLPY), additional biomass feedstock would have to be transported from other regions, greatly increasing the overall GHG emissions of the pathway.

Unlike Scenarios 1 and 2, the feedstock recovery was the second highest source of GHG emissions for Scenario 3, contributing to 17% of overall emissions. This is because the emissions from the conversion of mill residue into pellets is included in this process, a process that is not included in Scenarios 1 and 2. While the transportation of mill residue to the pellet plants and pellets to the pyrolysis facility were included under feedstock transmission, the GHG emissions were much lower (approximately 50 %) in Scenarios 1 and 2. This is likely due to the shorter transportation distances of both materials (25.3 km between mills and pellet plants, 50.9 km average from pellet plant to pyrolysis facility) as well as the increased density and lower moisture content of the pellets [130] [103]. If the pyrolysis facility were

reduced in size to 50 MLPY in Prince George, the feedstock transmission emissions would reduce by 70% since only pellets from the closest pellet plant, Pacific Bioenergy at 15km from the pyrolysis facility, would be required. However, since the emissions from feedstock transmission are low to begin with (4.6% of total emissions), the overall emissions would only decrease by 3.2%. If the capacity of the pyrolysis facility were increased, overall emissions would not change unless the relative percentage of pellets from each plant changed. However, if the pellet plants were not able to provide more than 70% of their pellets to biojet production, pellets from Quesnel or Strathnaver would be required, which would increase the average travel distance by 48% (if received 59% production from Pacific Bioenergy, 55.6% from others) [103]. It is unlikely that all pellet plants would be able to provide exactly the same amount of pellets due to contracts with other clients, so future work should be done to determine the actual amounts of pellets available for purchase from these pellet plants and incorporate this into the results.

It should be noted that all of the scenarios studied assumed that all of the available forest residue (excluding 25% required to remain on the ground for soil health [103]) on Vancouver Island, around Aldergrove, and 50% + of regional pellets were available exclusively for biojet production. This assumption is more likely for forest residue, as only about 10% of forest residue is currently used in the market (mostly by the pellet industry) [103]. Competition for the closest forest residue, which is the most economical to retrieve (estimated at \$67/tonne around Vancouver and the Island, \$75/tonne around Prince George), would result in either increased prices and/or increased transportation distances for biojet feedstock [103]. In turn, this would increase the overall GHG emissions of pathways using forest residue. For pellets, on the other hand, the assumption that 50% of the pellets from each pellet plant around Prince George were available for biojet production is rather unlikely. The actual availability will depend upon the existing long-term contracts of each pellet producer with other customers. Currently, 95% of pellets produced in B.C. are exported to Europe, Asia and the USA. Long-term contracts would have to be negotiated with these pellet producers to ensure availability at likely higher rates than current market rates (\$110 - \$117/tonne, currently) [103]. If a pyrolysis plant were built in Prince George as outlined in Scenario 3, pellets from Pacific Bioenergy should be maximized. This is because they provide the lowest GHG emissions due to the proximity to the proposed pyrolysis facility in Prince George. However, this analysis did not include energy and emission differences between the facilities, so these should be compared and included in future work to confirm this conclusion.

While pellets are a more expensive feedstock than forest residue (\$115/tonne compared to \$67/tonne), they do offer some technical advantages [103]. The primary barrier to the collection and use of forest

residue is the lack of logistical systems to retrieve the residues and transport to a processing facility economically. Current logging practices would need to be adjusted to incorporate the collection of the residues and piling at the roadside. While this is sometimes done, it is not common practice, and becomes much more expensive to do later. The inconsistency of forest residues physical and chemical properties is also problematic during pyrolysis. Forest residues often contain a large amount of bark and dirt, which can affect pyrolysis yields and require solvents, and the inconsistent species and moisture content can make process optimization difficult. Pellets, on the other hand, are uniform and consistent, making them an ideal feedstock for thermochemical processing. Pre-conditioning of the woody biomass (drying and grinding) occurs upstream at the pellet plant, reducing the operating and capital expenditure of the pyrolysis facility. Furthermore, this densified form of biomass is less costly and carbon intensive to transport and logistical systems are already in place. This study determined that the carbon intensity of using pellets in place of forest residue was similar, but work still needs to be done to compare the economics of the two scenarios. While pellets are more expensive than forest residue, the reduction in pre-processing steps at the pyrolysis facility and improved yields/reduced solvents needed for pyrolysis due to low bark concentration may negate some of the material expense. Furthermore, the additional solvents required and lower yields that result from high levels of bark in forest residues were not included in this LCA analysis. Due to limited data, this analysis considered that the chemical composition of both forest residue and pellets was the same (clean wood). However, it is likely that the inclusion of these details would increase the overall GHG emissions for pathways using the forest residue feedstock. Once this data becomes available, the impact of the wood feedstock quality on GHG emissions should be determined.

While the GHG emission reduction potential of pyrolysis-based jet fuel is substantial, the purpose of producing biojet fuel is to reduce worldwide GHG emissions and reach ICAO's reduction goal. If the biofuel requirement were to come only from pyrolysis, numerous facilities would be required to reach the aviation sector's goal of 50% reduction in greenhouse gases by 2050. Roughly 2/5 of this reduction is expected to come from energy efficiency improvements in airplanes and logistics and the remainder from renewable fuels (see Figure 1). In Canada, approximately 6,036 MLPY of aviation fuel was consumed in 2005, which is expected to quadruple by 2050, even with aircraft fuel efficiency improvements [139]. At this rate, 1,945 of the 100 MLPY pyrolysis-based biojet facilities would be required across Canada to provide the anticipated demand and airports would have to use 100% biojet blends to reduce their emissions by 50% of 2005 levels. If the Vancouver airport were to strive for the same GHG emission reduction goal as ICAO, it would have to switch to 100% pyrolysis-based jet fuel and over 117 100 MLPY

facilities would have to be built in B.C. by 2050 to supply this demand (see Appendix III). Not only is this not likely to occur due to financial constraints, but there is not enough mill and forest residue feedstock to supply even a fraction of this demand (total residue supply in B.C. is 6,554,000 ODT [123]). Improvements in the GHG emission reduction potential of biojet will reduce the amount of biojet required by each airport to reach their ICAO goals, so the maximum GHG emission reduction is assessed in the following sensitivity analysis by optimising influential methodological and process variables.

In Chapter 2, pyrolysis was found to be affected significantly by the allocation method and source of hydrogen. These variables are tested in a sensitivity analysis to determine their effect on overall GHG emissions of the pyrolysis pathway in B.C. Other variables that are tested are pellet feedstock, pyrolysis and upgrader integration, feedstock moisture content, and bio-oil yields. These variables all have the potential to affect emissions, but the extent of this effect is unknown. The results of these sensitivity analyses will provide insight into which variables have the greatest impact on the pyrolysis process as well as how the operating parameters for a pyrolysis facility built in B.C. can be optimized for carbon intensity reduction.

### 3.4.1 Sensitivity Analysis

#### 3.4.1.1 *Effect of coproduct allocation method*

In pyrolysis, the only co-products that are typically produced are biogas and biochar. Char is a coal-like solid fuel that can be used as a coal replacement or as a soil amendment, while biogas is a fuel gas that can replace natural gas or liquefied petroleum gas (LPG). The char is typically entrained with the heat transfer sand and is burned during sand recycle to heat the sand and provide process heat for pyrolysis [25]. Biogas is often burned internally as well to provide process heat and to dry the biomass feedstock. When the biomass feedstock has a moisture content lower than 50%, not all the biogas is required for drying [114]. Some biogas will be left over, which can then be sold as a co-product.

This is the case for Scenario 3, as the drying occurs at the pellet stage, leaving a significant portion of the biogas produced during pyrolysis unused. If the char were separated and collected separately, it could also be sold as either a coal substitute for electricity generation or as a soil amendment. Han et al. [96] and Fan [60] considered char as a co-product rather than biogas and found that the results when char was used to produce electricity were similar to when biogas was the co-product, though were slightly improved when char was used as a soil amendment. In this analysis, the char was assumed to be burned internally to provide process heat and the biogas was used for process heat and to dry the biomass

feedstock, leaving some available as co-product depending on the feedstock's moisture content. This was based on data within GHGenius, provided by ENSYN [82].

Whenever co-products are present, the GHG emissions of the whole process must be allocated between the products and co-products. In Chapter 2, it was shown that the choice of allocation method can have a profound impact on the overall GHG emission results. To investigate the impact of the allocation method on the GHG emissions of a pyrolysis pathway in B.C., three coproduct allocation methods were applied to Scenario 3 in a sensitivity analysis: energy, mass, and displacement. In British Columbia, the Low Carbon Fuel Standard calculates the carbon intensity of fuels using the displacement method, so it is important to determine how using this allocation method will compare to the energy allocation method used in this study.

While the economic allocation method was suggested as one of the more realistic methodologies in Chapter 2, this allocation method is not available in GHGenius, so was not included in this analysis. No other variables were altered other than the allocation method in this section. Scenarios 1 and 2 were not analyzed as they do not produce co-products, therefore would not be affected by the coproduct allocation method.

For the mass allocation, an allocation ratio of 5.36% was used to allocate the emissions to the biogas (the remainder is allocated to the bio-oil) [58]. This ratio was calculated based on the relative masses of biogas to bio-oil. This is similar to the 5.82% energy allocation ratio, since the relative mass and energy of biogas to bio-oil is similar. When the displacement method is used, it is assumed that the biogas is replacing liquefied petroleum gas (LPG) according to its energy content, with an emission intensity of 1,743 gCO<sub>2</sub>eq/L LPG [58]. LPG was chosen over natural gas by GHGenius, as most of the pyrolysis facilities are in remote locations without access to natural gas pipelines [140]. This displacement assumption is also used by the GREET model, whenever excess biogas, or “fuel gas”, is produced [96]. Since the scenarios modeled in this chapter are in less remote areas that likely have access to natural gas, the displacement of natural gas was also modeled to determine the impact of this assumption.

Figure 25 shows the overall GHG emission results of Scenario 3 when different allocation methods are used. When displacement was used, the overall GHG emission of the pathway dropped from 28.28 -28.38 gCO<sub>2</sub>eq/MJ (energy allocation) to 20.9-22.3 gCO<sub>2</sub>eq/MJ, a roughly 23.7% reduction. When the mass allocation is used, the change in GHG emission is low (28.8 gCO<sub>2</sub>eq/MJ). This is expected, due to the similarity in energy and mass allocation ratios. The decrease in emissions when displacement is utilized is

also expected, since this will occur anytime that the lifecycle carbon intensity of the co-product is lower than the product it is displacing (see Appendix III). As a result, the emission intensity of the product the co-product is displacing will have a significant impact on the results when using displacement allocation.

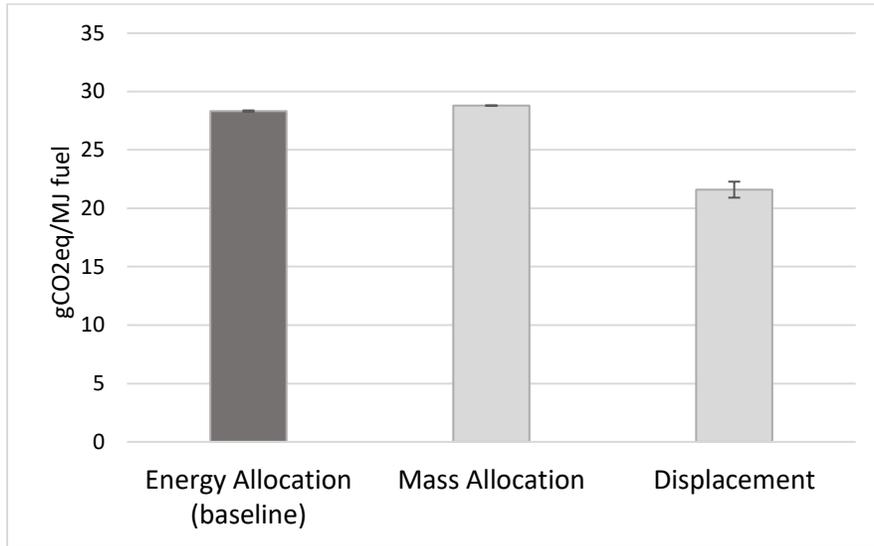


Figure 25: Effect of changing allocation method on GHG Emissions (Scenario 3)

For this study, the displaced product for the biogas was LPG. In determining the emission intensity of LPG, the average carbon intensity for the LPG industry in Canada was used. However, if the carbon intensity of the specific LPG being displaced were lower than average, which could occur with improved efficiencies, renewable energy usage, or by using a renewable feedstock for production such as biogas, than the reduction caused by using the displacement allocation method would diminish. On the other hand, if the carbon intensity of the product being displaced were higher than that for the average liter of LPG (1,743 gCO<sub>2</sub>eq/L), the difference between energy allocation and displacement allocation results would increase. This may occur if natural gas was displaced by the co-product rather than LPG, since natural gas has a carbon intensity of 2,212 gCO<sub>2</sub>eq/L [58]. If the co-produced biogas displaced natural gas rather than LPG, the overall GHG emissions of Scenario 3 using displacement would decrease by 5.2% (20.46 gCO<sub>2</sub>eq/MJ). This is important to note, since the displacement method is utilized when determining GHG emissions under the B.C. Low Carbon Fuel Standard. Whenever the displacement co-product allocation method is used, care must be taken to accurately track and identify which product is being displaced by a fuel's co-products, since these choices can have an impact on the final GHG emission results.

### 3.4.1.2 Effect of Hydrogen Source

In chapter 2, the source used for hydrogen production (methane, biogas, aqueous phase reforming of bio-oil, etc.) was found to have a substantial impact on pyrolysis emissions. Steam methane reforming (SMR) is the most common method for producing hydrogen, accounting for 50% of production worldwide [141]. Oil and naphtha reforming account for another 30%, with 18% of the remaining balance produced from coal reforming [141]. However, it is possible to utilize biogas to produce hydrogen via SMR and new pyrolysis and upgrader designs often incorporate the hydrocarbon-rich off-gas into the reformer, offsetting some of the natural gas intake [25]. As a result, this sensitivity analysis compares the GHG emissions of these two scenarios, as well as looked at what the GHG emissions would be if electrolysis were used rather than SMR for hydrogen production.

Electrolysis is the extraction of hydrogen from water, attained by passing an electric current through water. The most common types of electrolyzers currently on the market are Alkaline Reactors and Proton Exchange Membranes (PEM). While the latter has a higher capital cost, it allows higher throughput and operates at higher efficiencies, so is more practical for large-scale application. For these reasons, this was the type of electrolyser assumed in this study. According to the Gibbs free energy of water, 237 kJ of energy is the minimum amount of input energy required for the electrolysis reaction to occur. However, a total of 268 kJ is required to account for the additional change in entropy [142]. This energy can be provided by both heat and electricity, so a combination is generally used. For all the scenarios in this study, 5.15 MJ of hydrogen was considered to produce each liter of refined bio-oil [25]. This is within the range published within literature (see Table 19). In the base-cases, this hydrogen was produced from natural gas and any residual off-gases from upgrading. If electrolysis were used to produce the hydrogen instead, 2.0 kWh of electricity would be required per L of refined fuel. This is based off the efficiency of the electrolyser, assumed to be 56 kWh/kg of hydrogen produced (GHGenius default), which is within the range seen for commercial electrolyzers (53.4 – 70.1 kWh/kg H<sub>2</sub>) [142].

**Table 19: Hydrogen consumption during biogas upgrading (MJ/L refined bio-oil)**

5.15	S. Jones [25]
4.39-5.46	M. Wright [92]
3.45	Beckman et al. [143]
5.48	Elliot et al. [144]

All of the three hydrogen production methods (electrolysis, SMR, and SMR with pyrolysis biogas) are compared for Scenario 3, where excess biogas is available. A comparison between SMR and electrolysis is performed for Scenario 1 as well, to determine whether the findings from Scenario 3 transfer between pathways.

Figure 26 indicates that substantial reductions in GHG emissions are possible when hydrogen is produced by an alternative to SMR in B.C. As is shown, using excess pyrolysis biogas to offset some of the natural gas needed for hydrogen production improved the carbon intensity of the pathway by about 24% (21.56 gCO<sub>2</sub>eq/MJ vs. 28.32 gCO<sub>2</sub>eq/MJ). When only natural gas was used, as was the case for the base-case scenario, the biogas produced via pyrolysis was considered a co-product that offset LPG in the market [58]. While this product obtains an emission credit based on its energy content, this credit is not as large as the emissions associated with producing an equivalent amount of natural gas. This is demonstrated by the results, displayed in Table 20, which shows that the credit obtained for the biogas when it is sold as a coproduct (emissions displaced) is far less than the credit obtained when used in place of natural gas for hydrogen production (fuel production). This is typically the case whenever energy or mass coproduct allocation methods are used, since the product that the biogas is displacing does not get considered. Displacement always takes this “displacement ratio” into account, so there was very little difference between these two scenarios when this allocation method was used (using biogas internally only reduces GHG emissions by about 1 gCO<sub>2</sub>eq/MJ). Therefore, when energy coproduct allocation was used for these scenarios, using biogas internally was encouraged, while with displacement it did not matter much whether it is used internally or externally.

**Table 20: GHG Emission comparison between pyrolysis biogas uses**

<b>GHG Emissions (gCO<sub>2</sub>eq/GJ)</b>	<b>Scenario 3 Base-case (SMR using natural gas and off-gas; pyrolysis biogas is sold)</b>	<b>Scenario 3 (SMR using natural gas, off-gas, and pyrolysis biogas)</b>
Fuel Dispensing	39	39
Fuel distribution and storage	215	215
Fuel Production	22398	15,362 – 14,420
Feedstock Transmission	1300	1,300
Feedstock Recovery	4,926	4,926

Emissions Displaced	-687 to -812	0
Emissions from Fuel Use	195	195
<b>Total</b>	<b>28,260-28,385</b>	<b>21,036 – 22,036</b>

While the use of biogas for hydrogen production lowered the overall emissions, the use of electrolysis in B.C. provided the greatest GHG reduction, especially when the displacement allocation was used (all others utilized energy allocation). When electrolysis was used in B.C., the GHG emissions were reduced by 62% (10.66 gCO<sub>2</sub>eq/MJ) with energy allocation or by 117% (-4.97 gCO<sub>2</sub>eq/MJ) when displacement was used. The effect is similar for Scenario 1, which dropped by 66% (8.58 gCO<sub>2</sub>eq/MJ overall) when electrolysis was used (see Appendix IV for details).

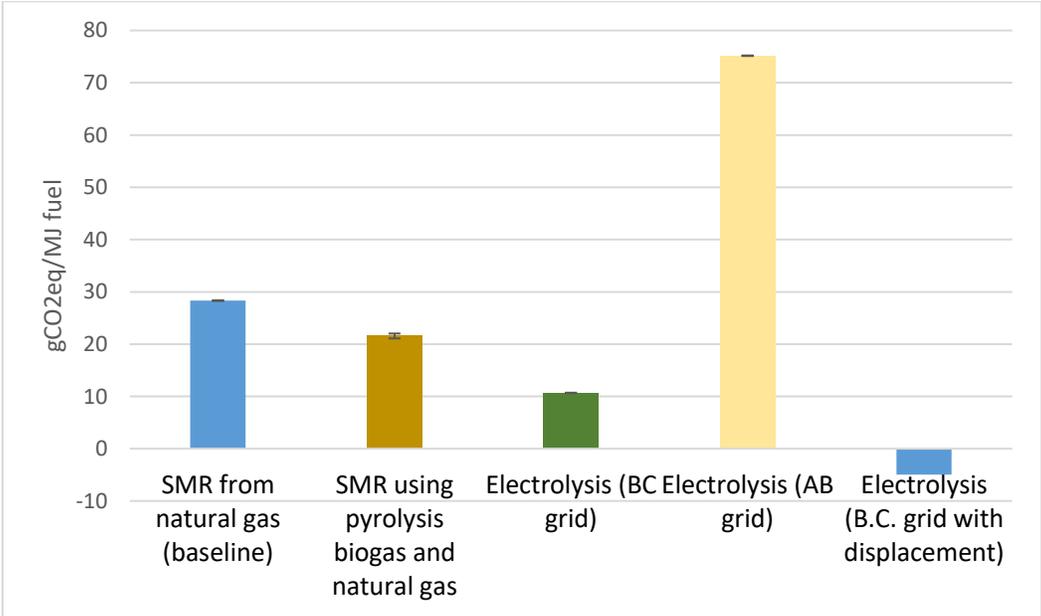


Figure 26: Effect of changing hydrogen source on GHG Emissions (Scenario 3)

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he reduction in GHG emissions when electrolysis was used for the B.C. scenarios was likely caused by three factors: natural gas elimination, biogas co-product increase, and B.C.’s low carbon electricity mix. To produce the hydrogen necessary for bio-oil upgrading through SMR, 315L of natural gas is required for every liter of upgraded pyrolysis fuel [58]. When the hydrogen is produced via electrolysis, this natural gas is no longer required. This is reflected by the significant decrease in GHG emissions for the fuel

production process in Table 21. Furthermore, the electricity produced in B.C. is primarily renewable (hydro), giving the mix a very low carbon intensity (15,404 gCO<sub>2</sub>eq/GJ electricity delivered) [58]. The electricity used by electrolysis (2.0 kWh/L upgraded bio-oil) adds only minimal GHG emissions to the overall process, as seen by the very low emissions for fuel production in Table 21 (5,728 gCO<sub>2</sub>eq/GJ fuel).

However, if this facility were in Alberta, this would be very different. In Alberta, the electricity mix is 236,639 gCO<sub>2</sub>eq/GJ electricity delivered, so utilizing electrolysis increases overall emissions rather than lower it (see Figure 26). The fact that B.C. has a highly renewable electricity grid makes this technology very attractive for use in the Province from a GHG point of view, but this is highly dependent upon the electricity mix, so should not be considered in areas with high carbon intensity mixes. This agrees with the finding in Chapter 2 that the location of a process has a large effect on the overall GHG emissions. While the GHG emissions did not change substantially within a province, according to the comparability of results between Scenarios 1-3, the change was significant when between provinces.

**Table 21: Comparison of GHG emissions from electrolysis and SMR hydrogen production methods**

<b>GHG Emissions (gCO<sub>2</sub>eq/GJ)</b>	<b>Scenario 3 Base-case (SMR using natural gas (and some off-gas) used for H<sub>2</sub> production)</b>	<b>Scenario 3 (Electrolysis is used for H<sub>2</sub> production)</b>
Fuel Dispensing	39	39
Fuel distribution and storage	215	234
Fuel Production	22398	5728
Feedstock Transmission	1300	1300
Feedstock Recovery	4,926	4,925
Emissions Displaced	-687 to -812	-1,498 to -1,597
Emissions from Fuel Use	195	195
<b>Total</b>	<b>28,260-28,385</b>	<b>10,709 – 10,610</b>

The increase in biogas co-product available when it was no longer needed for hydrogen production, which is taken care of by electrolysis, also reduced the overall GHG emissions. As seen in Table 21, the co-product credit increased by 206% when electrolysis was used. This is because the light hydrocarbon off-gas

produced during upgrading, due to hydrocarbon cracking, could no longer be used for hydrogen production as was usually the case. Instead, this off-gas was sold as a co-product (2.88 MJ/L refined bio-oil), which increased the co-product credit for the pathway and reduced the GHG emission reduction of the pathway even further. When the displacement allocation method was used, the GHG emissions of this pathway decreased further still, making the pathway carbon negative (-5.33 to -4.43 gCO<sub>2</sub>eq/MJ fuel).

This phenomenon is common when the co-product-to-product ratio is high (as seen here, where 4.86 to 5.26 MJ of biogas is produced per liter of bio-oil product) and the displacement co-product allocation method is used. According to work by Stratton et al. [62], the GHG emissions using displacement becomes increasingly skewed the larger the ratio of co-product to product becomes. This is the primary reason that energy allocation was chosen as the primary allocation method for the scenarios in this chapter. However, displacement is used in most calculations of fuel carbon intensities under the B.C. Low Carbon Fuel Standard (since it is the default method of GHGenius), giving way to carbon negative pathways when high amounts of co-products are produced. This ends up favoring pathways that produce more co-product than product, a side-affect of this method, so its use as default in the B.C. Low Carbon Fuel Standard should be carefully considered.

Additionally, it should be noted that all the scenarios considered that the electricity used for electrolysis was obtained from the provincial electrical grid. When using electrolysis, each 100 MLPY biojet facility consumes roughly 200 GWh per year. This is equivalent to the average total electricity exported from B.C. every year [145]. Currently, the Province of British Columbia has approximately 3,000 GWh/yr excess capacity, but electricity demand is forecast to surpass this supply by 2021. This supply will therefore likely not be available for pyrolysis biorefinery projects, which are still in the pilot-scale and demonstration stages. B.C. Hydro has scheduled to open the reserved turbine at the Mica Dam when needed and expand the Revelstoke Dam by 2026 to add an additional 4,634 GWh and 2,317 GWh, respectively [145]. If the construction of the Site C Hydroelectric Dam is accomplished by 2024 as planned, an additional 5,100 GWh of low-carbon base load electricity will also be added to the grid [146]. If all the additional electricity, including the 200 GWh of current exports, were utilized strictly for pyrolysis biojet production expansion, the grid could supply enough electricity for roughly 61 100 MLPY pyrolysis plants utilizing electrolysis. While the required amount of fuel needed by airports would be reduced due to the lower carbon intensity of fuel produced with electrolysis (see Figure 26), still roughly 91 (energy allocation) or 68 (displacement allocation) pyrolysis facilities would be required. Of course, this does not consider the 40% increase in

electricity demand during this time or the increase in electrical car usage or liquefied natural gas (LNG) industry expansion, which may consume a majority of the electrical supply increase itself [145].

Therefore, if more than a handful of 100 MLPY pyrolysis facilities were to be built, the grid will likely not be able to provide the required load and marginal power will have to be supplied. Most marginal power in B.C. is provided by diesel generators (218.4 gCO<sub>2</sub>eq/GJ electricity), which would greatly influence any pathways that utilized electrolysis [58]. Under these conditions, an nth generation facility utilizing marginal power would likely see a GHG emission increase of 121% (66,746 gCO<sub>2</sub>eq/GJ) when using electrolysis. Therefore, if more than a couple 100 MLPY pyrolysis facilities are to be built, the benefits of using electrolysis are lost and using SMR should be used for the additional facilities.

### *3.4.1.3 Effect of feedstock*

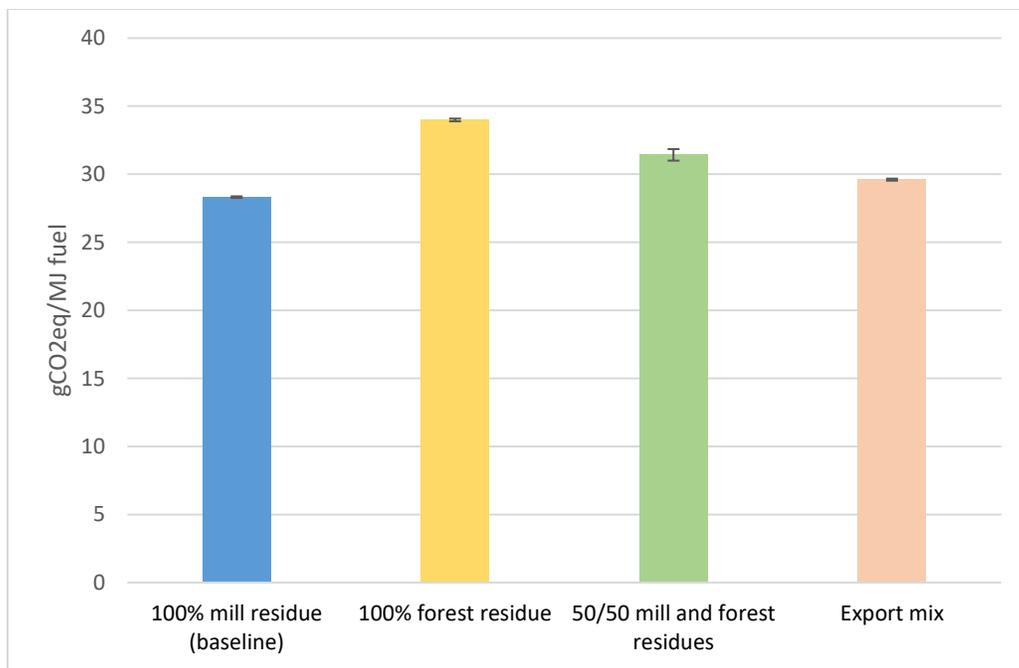
#### **3.4.1.3.1 Pellet Feedstock (Scenario 3)**

Mill residue was the pellet feedstock considered in the base-case of this analysis, as this is the most common biomass source for pellet production. According to DRAX, a Canadian pellet importer in the UK, the pellets imported from Canada consist of 79.1% mill residue, 13.2% forest residues, and 7.7% standing timber (thinnings and low-grade roundwood) [147]. However, as pellet production increases, producers will likely be required to utilize greater amounts of forest residue as mill residue supply diminishes. In 2013, Ghafghazi et al. reported a sawmill residue surplus of only 18,508 ODT within the region of Prince George, a 0.5% increase over the current residue demand (3,410,500 ODT) [124]. Large amounts of mill residues still exist in other regions of B.C. (190,600 ODT on Vancouver Island and the Vancouver area, 170,769 ODT in the Kootenays), but importation of these residues to the current pellet mills will add expense. Once the surplus of mill residue in the Prince George area is consumed, pellet producers will likely have to utilize a combination of imported mill residue and local forest residues to increase production. Figure 27 shows the affect on the overall GHG emissions that utilizing increasing amounts of forest residue would have on overall GHG emissions compared to the base-case of using 100% mill residue and the current Canadian average (as per DRAX) of biomass sources for pellets.

As the graph indicates, utilizing mill residue as the biomass feedstock for pellet production provides lower overall GHG emissions when utilized for biojet production. When the pellet biomass source changed from 100% mill residue to 100% forest residue, the overall GHG emissions of the pathway increased by 20% (34.0 gCO<sub>2</sub>eq/MJ biojet). The average Canadian pellet utilizes 13% forest residue and 8% roundwood, which is responsible for the slight increase seen in GHG emissions of the DRAX mix compared the using

full mill residue (29.6 gCO<sub>2</sub>eq/MJ vs 28.3 gCO<sub>2</sub>eq/MJ), but as these percentages increase over the next decade, it is likely that the GHG emissions will increase closer to that of 50/50 forest residue and 100% forest residue.

If mill residue were imported from other regions, such as the Vancouver area, the overall GHG emissions would increase to account for the additional transportation emissions. For example, if 40% of the biomass feedstock were imported from the Vancouver region, the GHG emissions would increase by about 6 +/- 1.1 gCO<sub>2</sub>eq/MJ depending on the exact location of the Vancouver mills (see Appendix III for details). At this level, forest residues would be preferred in terms of GHG emissions (see Figure 27), however it is likely that the economics of the two supply options will influence this decision.



**Figure 27: Effect of changing pellet feedstock source on GHG Emissions (Scenario 3)**

Due to the biomass supply shortage for additional pellet production in Prince George, new pellet production may develop in areas of high mill residue supply, such as the Vancouver or Kootenay regions (190,676 BDT and 170,769 BDT of mill residue surplus, respectively) [124]. Most of these available mill residues are spread out over large regions, making their collection more expensive than the mill residues currently available in the Prince George region. However, once the surplus of mill residue in the Prince George area is consumed, locating a pellet plant within these secondary sources may be more economical

than transporting mill and/or forest residues to the Prince George area. Likely a combination of these supply scenarios will be required for expansion, and a techno-economic analysis should be performed to determine the economics of each option.

It should be noted that all of the scenarios utilizing mill and forest residue in this chapter assumed that the residues were wastes. This is the protocol adopted by most LCA practitioners and regulatory bodies. ISO 14040 defines a waste as any output with little to no economical value that needs to be disposed of [32]. However, this is somewhat vague and provides room for interpretation regarding what is considered of “little economical value.” Most jurisdictions attempt to further define this definition, since the classification as a waste can have a large impact on total product GHG emissions. Under the UK Renewable Transport Fuel Obligation (RTFO), wastes are defined as any by-product that has an economical value of less than 15% of the main product [148]. The California Low Carbon Fuel Standard, on the other hand, considers a specific value arbitrary and subject to the market dynamics, so chooses to differentiate between a waste and a co-product by its economical value as well as its supply elasticity. This is similar to how the U.S. EPA determines a waste, which is determined by satisfying the following requirements [148]:

- Its production cannot be the primary reason for producing the feedstock
- It cannot have a well-established market and/or market prices
- It cannot have a significant impact on the market if it is removed
- If the primary alternative use is to burn for heat, etc. in stationary combustors

According to these definitions and others, forest and mill residues are considered a waste under the B.C. LCFS, California LCFS, UK RTFO and the EU RED. While mill residues are currently considered wastes under the U.S. RFS as well, this may change as the market develops for these residues in the pellet industry. The removal of mill residues from the market would have a significant impact on this industry, so its exclusion from the classification as a waste could be argued based on the above requirements.

For this reason, it is important to determine how the overall GHG emissions would change for this pathway if the mill residue classification changed and it were no longer considered a waste. Pa et al. determined that including the additional upstream emissions from harvesting and sawmill activities in pellet production (which would be required if the residue were no longer considered a waste) would add an additional 9.32 gCO<sub>2</sub>eq/MJ fuel to the overall pathway [134]. This would increase the baseline emissions in Figure 27 to 37.66 gCO<sub>2</sub>eq/MJ, which is 3.67 gCO<sub>2</sub>eq/MJ higher than when the pellets are produced from 100% forest residue. As a result, the classification of mill residue as a waste is essential in providing

the reduced emissions seen in Figure 27 and any change in its classification will eliminate the benefits of using this feedstock over forest residues.

#### 3.4.1.3.2 Raw feedstock (Scenarios 1 and 2)

The base-case of Scenarios 1 and 2 assumed that forest residues would be utilized as feedstock if a pyrolysis facility were built in the Vancouver area. Forest residues are the predominant feedstock in this region (758,000 ODT), but there are some mill residues available as well that are not currently used for other processes (190,600 ODT) [124]. These mill residues are widely dispersed and not available in large enough volumes to supply a pyrolysis facility entirely, but they could offset a portion of the feed. However, the exact location of the available mill residues is not available, so the exact GHG emissions of the process utilizing the available residues cannot be calculated. Nevertheless, if mill residues were introduced to the feed, they would likely reduce the GHG emissions of the process as seen above in Figure 27. However, this would only be the case if the mill residues were not located far from the pyrolysis facility. If mill residues were acquired from areas greater than 26 km from the facility, the reduction would minimize. If the mill residues were located farther than 113 km on average from the pyrolysis facility, then they probably should not be considered, as the GHG emissions of the pathway would increase over that of using forest residues (see Appendix III for details).

Of course, all of this assumes that the mill residues are considered wastes. As described in section 3.4.1.3.1, this assumption is what makes mill residues a low carbon feedstock. If this classification were removed, as some LCA practitioners advocate, then the GHG emission savings from utilizing this feedstock would disappear and the two feedstocks (mill and forest residue) would become equal in terms of GHG savings.

#### 3.4.1.4 Effect of pyrolysis facility location

Locating the pyrolysis facility in the same area as the upgrading facility can have multiple advantages. Firstly, utility and hydrogen production infrastructure can be shared, lowering the capital cost of the project. Secondly, locating the pyrolysis facility in the same location as the hydrogen production unit facilitates the use of pyrolysis biogas directly for hydrogen production, offsetting some of the natural gas. This reduces overall costs as well as GHG emissions, as shown in Figure 28. Thirdly, co-locating the pyrolysis plant and upgrader reduce the distance that the stabilized bio-oil must be transported. Because of these advantages, the GHG emissions of the first two scenarios (Vancouver Mainland and Vancouver Island) were calculated when the pyrolysis and upgrading facility were co-located. Scenario 3 was not

included, since this scenario already assumes the co-location of the pyrolysis and upgrading plant. In the co-located scenarios, the pyrolysis plant is assumed to move from Aldergrove and Parksville, B.C., respectively, to the refinery in Burnaby.

Figure 28 shows the GHG emission impact of co-locating the two facilities in Scenarios 1 and 2. While economic benefits may exist, these results indicate that there is a GHG emission cost to co-location. When co-located, the GHG emissions of both scenarios increased by 3.1% and 3.7%, respectively. This is largely due to the increased transportation of the low density and high moisture forest residue (increase of 55 km (S1) or 128 km (S2)). This analysis considered that forest residue was chipped at the forest site and transported directly to the pyrolysis facility. If the wood were ground and dried prior to transportation, this would remove a large amount of the extra emissions and financial costs associated with transport and also increase the amount of pyrolysis biogas available to offset natural gas as hydrogen feed. In the base-case, Scenarios 1 and 2 did not produce any excess biogas to use for hydrogen production. If the forest residues were air-dried onsite (in the forest stand) to 20% moisture content (dry basis), then about 1.5-1.75 MJ/L bio-oil of biogas co-product becomes available and the transportation emissions are reduced by approximately 20% due to the lighter load (see Appendix III for details). Under these conditions, the GHG emissions of the integrated process would begin to reach that of the distributed process (25.94 gCO<sub>2</sub>eq/MJ for Scenario 1).

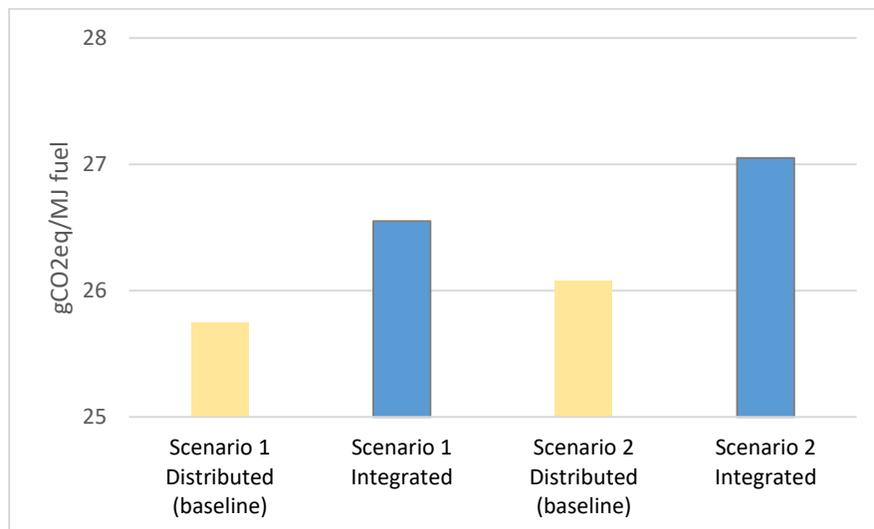


Figure 28: Effect of pyrolysis location on GHG emissions (Scenarios 1 and 2)

However, even if the forest residues are not dried on site, the reduction in capital expenditure may be large enough to accept the small GHG increase. The financial benefits of co-locating, as well as the break-even forest residue moisture content required to drop the GHG emissions to that of the distributed baseline, are outside the scope of this work but an interesting point for further study. In future work, it would also be interesting to determine at which distance (between logging site and refinery) it becomes advantageous to separate the pyrolysis and refining facilities, taking into account all possible synergies.

#### *3.4.1.5 Effect of process parameters (moisture content and bio-oil yield)*

The operating conditions of a process can also have an impact on the overall GHG emissions of a pathway. The results of chapter 2 indicated that while the operating conditions can affect the LCA results, the impact is not usually as large as the source of hydrogen, location, and allocation method. To test this and to determine what the impact of varying the process variables had on the case study pathways, two operating conditions were varied: moisture content of the forest residue feedstock and the bio-oil yield. These operating conditions were chosen as they were expected to have the largest overall impact and are often varied between studies. The moisture content of forest residue tends to fall between 20% and 70%, depending on the weather and season, according to a study by Boeing [103]. This agrees with other LCA studies utilizing forest residue, where forest residue moisture content fell between 20% and 50% (see Table 22). The base-case studies of each scenario in this chapter assumed that the moisture content of the original forest or mill residue was 50%, so the sensitivity of this variable was tested to see how the LCA results changed over the moisture content range of 20% -70%.

The yield of bio-oil and refined bio-oil are two of the most significant parameters affecting process economics [135]. The yield will also impact the GHG emission results, since a larger yield will reduce the amount of feedstock required to produce the product. Generally improved yields indicate improved efficiency, translating to lower overall GHG emissions. However, higher product yields often result in lower co-product yields. Since co-products tend to absorb a large portion of the GHG emission burden, this may negate some of the efficiency benefits. This relationship was explored by utilizing the high and low end of the bio-oil yields seen within literature and their equivalent co-product yields. As seen in Table 22, the bio-oil yields seen in literature ranged from 53.1% to 78%, while the refined bio-oil yields ranged from 36% to 47%. However, these ranges included yields calculated on a dry basis as well as data for corn stover (highlighted in grey), so the yields that were considered relevant and included in the sensitivity analysis were 70.3%-78% and 36.1%-44%, for bio-oil and refined bio-oil, respectively. The GHG emissions

of the pathway using these yields were compared against the baseline yields assumed in this study, which were 72.5% for bio-oil and 36.1% for refined bio-oil.

**Table 22: Operating Parameters for Pyrolysis and Upgrading**

Moisture Content (wet wt%)	Bio-oil Yield (wet wt%)	Refined Bio-oil yield (wet wt%)	Total yield (wt%)	Source
50	70.3-78	44	31-34	[25]
50	72.7-75	36.1	26.2 – 27.1	[114]
30	53.1 - 60.3 <sup>1</sup>	46 <sup>1</sup>	24.2-27.4 <sup>1</sup>	[135]
-----	71	49	35	[20]
25	71.6	43.5	31.2	[86]
20	-----	-----	31.7	[60]
25	61.6-62.9	42	25.8 – 26.4	[92]
50	72	44	31.8	[76]

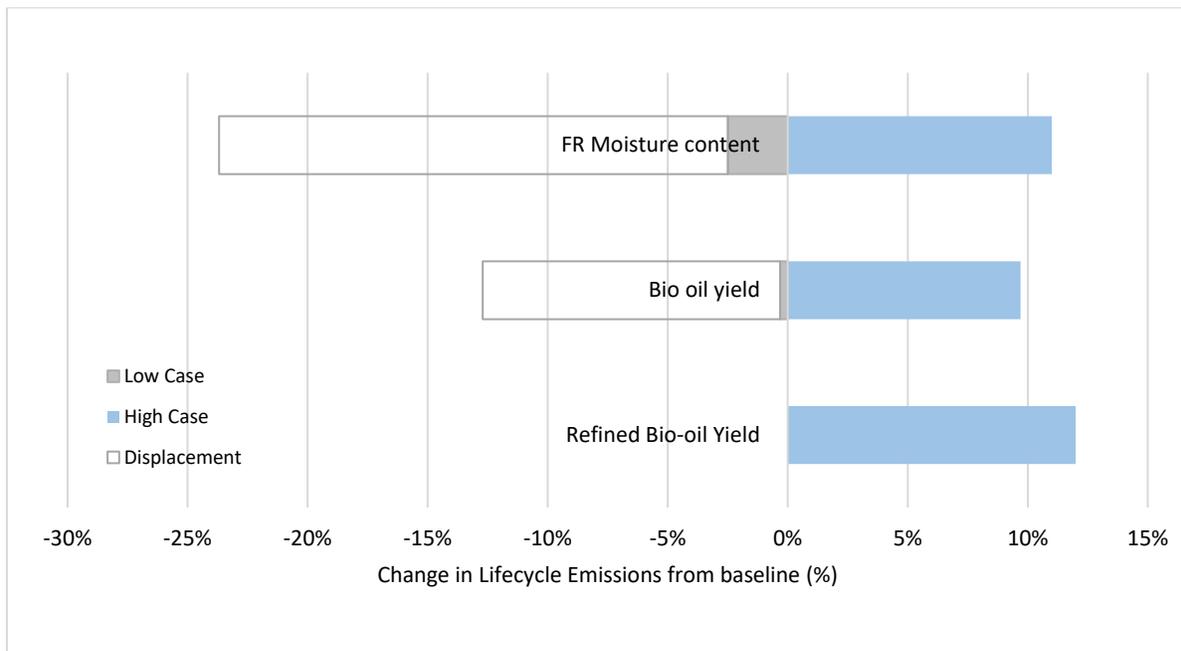
<sup>1</sup> Dry basis

The results of the sensitivity analysis on moisture content, bio-oil yield and refined bio-oil yield are presented in Figure 29. According to the figure, the alteration of these variables within the published ranges altered the GHG emissions from baseline by -2.4% to +12%. When the displacement allocation method was used rather than energy allocation, the GHG emissions seen for the low scenarios were reduced. These impacts (especially for the high end or when displacement is used) were higher than expected, making these process parameters significant.

Of these changes, the moisture content and refined bio-oil yield have the greatest impact. When the moisture content of the collected forest residue was around 20% rather than 50%, which could be the case if the slash piles were collected at the end of a hot and dry summer, the GHG emissions of the pathway reduced by about 0.64 gCO<sub>2</sub>eq/MJ (2.4%) when energy allocation was used, or by 6.1 gCO<sub>2</sub>eq/MJ (21%) when displacement was used. On the other hand, if the residues were collected after a lot of rain and were saturated (e.g. 70% mc), the GHG emissions of the process increased by 2.8 gCO<sub>2</sub>eq/MJ (11%). It is interesting to note that when energy allocation is used, the GHG emissions are impacted more by the increase in natural gas required to dry the wet (70% mc) forest residues than by the increase in co-product produced when the forest residues are dry (20% mc). This is because with energy allocation, the

emissions allocated to the co-product are lower than the carbon intensity of natural gas. However, when displacement was used, the carbon intensity of the produced biogas and natural gas were similar, causing the much larger reductions seen in Figure 29.

Therefore, when displacement is used for modelling, the weather has a significant impact on the GHG emissions (both dry weather and rain), while only the rain before residue pickup is significant when energy allocation is used.



**Figure 29: Effect of moisture content and bio-oil yield on GHG Emissions (Scenario 1) from baseline (50% mc, 72.5% bio-oil yield, 36.1% refined bio-oil yield)**

When it comes to bio-oil yield, the lowest GHG emissions occurred when the bio-oil yield was low (70.3% yield). This is because of the increased biogas that is usually produced when the yields are low. As with moisture content, the GHG emission impact of the increased biogas production was much greater when displacement was used. For example, when the bio-oil yield was 70.3%, the co-produced biogas yield increased by 1.1% (0.98 MJ biogas/L bio-oil), which negated the reduction in yield and reduced the overall GHG emissions by 12.4% (0.3% with energy allocation) [25]. When bio-oil yields improve, this generally results in a lower amount of produced biogas, which results in higher overall GHG emissions. This trend shows that improving the bio-oil yield may not be beneficial if it results in a lower biogas yield.

A similar trend occurred when the refined bio-oil yield was increased. Even though the base-case utilized one of the lowest refined bio-oil yields published, utilizing this lower yield decreased emissions by about 12%. This is because when the refined bio-oil yield was increased, a larger amount of hydrogen was consumed to increase the yield (~1.63 MJ H<sub>2</sub>/L refined bio-oil). Due to this hydrogen consumption increase, lower yields tended to provide improved GHG emissions overall.

### 3.5 Conclusions

The lifecycle assessment results for renewable aviation fuel in B.C. indicated that substantial GHG reductions can be made when refined pyrolysis fuel replaces Jet A kerosene. British Columbia is one of Canada's largest producers of forest residue and pellets, which could be used as feedstocks for this process. Using pyrolysis bio-oils and upgrading them to biojet fuel was shown to reduce emissions by around 68-71% compared to petroleum jet fuel. Gasification and HTL can also reduce emissions to this level but encounter significant economic and technological barriers which make large-scale production difficult. Though B.C. has a significant amount of resources to produce this low carbon fuel, the cost of production is still higher than that of petroleum fuel. It has been estimated the Minimum Fuel Selling Price (MFSP) of pyrolysis-based fuels is currently \$2.04/gallon [25]. While this price might have been competitive 10 years ago, it is now 30 to 40 cents higher than 2017 diesel (\$1.72/gallon), gasoline (\$1.71/gallon) and Jet A-1 Kerosene (\$1.61/gallon) refiner prices [30]. Although BC's low carbon fuel standard might provide the impetus for some low-CI (carbon intensity) fuel production, additional policies will likely be needed to bridge the current price gap. This could include policies that encourage the collection of forest residues at the roadside, shift economic subsidies from petroleum to renewable fuels, or including forest residue removal as part of the forest fire reduction strategy and budget. While the exact impact of these policies on the economics of aviation fuel production in B.C. is outside the scope of this thesis, policies like these could help reduce the price disparity between pyrolysis and petroleum-based fuels.

While three pyrolysis facility locations (Vancouver Mainland, Vancouver Island, and Prince George) and two biomass sources (forest residue and pellets from mill residue) were investigated, it was apparent that the geographical location had a negligible impact in terms of GHG emissions as the overall GHG emissions of each pathway were 25.7, 26.1, and 28.3 gCO<sub>2</sub>eq/MJ fuel, respectively. Chapter 2 identified that the geographical location made a difference between provinces and countries, but the results from this chapter indicate that the geographical location within a province makes little difference. A sensitivity

analysis found that the GHG emissions were more affected by the type of pellet feedstock (forest or mill residue), the source of hydrogen (methane, biogas, or electrolysis), moisture content, yield and the type of allocation method used rather than the location of the plant. The lowest GHG emissions occurred when the pyrolysis facility was located at the same site as the upgrader, when mill residue was used as feedstock, when electrolysis was used to produce hydrogen (using the B.C. grid, and when the displacement allocation was used. Under these conditions, the pathway became carbon negative, with overall GHG emissions of  $-5.65 \text{ gCO}_2\text{eq/MJ}$  fuel. When the biogas displaced natural gas rather than LPG, the moisture content was low (20%) and the bio-oil yield was low (70.3%), the GHG emissions were reduced even further to  $-22.13 \text{ gCO}_2\text{eq/MJ}$ .

In B.C., the Low Carbon Fuel Standard encourages the use of displacement for carbon intensity modeling, which is to the advantage of fuel producers in the region [149]. Using this methodology, the maximization of co-produced biogas will decrease the carbon intensity of the pyrolysis fuel. While this is practice under the current regulation, using displacement can skew results whenever large amounts of co-products are produced (see Appendix III). Thus, this methodology should be used with care under these circumstances since it could have unintended consequences on the economy, such as encouraging the industry to develop more co-products than product. Thus, the effects of this practice should be thoroughly assessed and understood before implementing this co-product allocation method into the Federal Clean Fuel Standard.

When it comes to impacting the ICAO Greenhouse Gas reduction vision, pyrolysis will likely only play a small part. In order to reduce the overall GHG emissions of the Canadian aviation sector 50% by 2050 (assuming 30% of this reduction will come from infrastructure energy efficiency reductions), it would require over 1,945 100 MLPY pyrolysis facilities across Canada. Approximately 117 100 MLPY pyrolysis facilities would be needed to support the Vancouver airport alone (see Appendix III). Under the optimum GHG-reducing conditions above, this number is reduced to 63. However, under these electricity-demanding conditions, almost all of the electricity produced from the proposed Site C dam and the Revelstoke and Mica Dam expansion would be required. For this reason, though a few facilities could utilize electrolysis, most of the facilities will need to employ SMR. When utilizing SMR, the maximum reduction possible for pyrolysis fuel is 92% over petroleum fuel ( $7.03 \text{ gCO}_2\text{eq/MJ}$ ), and would require 87 100 MLPY facilities and 34.8 Million ODT of biomass per year to reach ICAO's goal within the province. However, in B.C., the maximum available biomass residue is 6.16 Million ODT (4.24 Million ODT forest residue, 1.92 Million ODT pellets), which could supply a maximum of 15 100 MLPY facilities. Furthermore,

this does not consider the current pellet contracts or the high prices of some feedstocks, so this number would likely be even lower. As a result, while B.C. is well situated for the construction of a pyrolysis-based jet fuel facility due to the abundance of biomass and its renewable electricity mix, biojet fuel made via this technology will not be sufficient to supply all the biojet fuel required for B.C. to reach ICAO's GHG reduction goals within the province. Instead, these pyrolysis bio-refineries will need to be complimented with bio-refineries utilizing other technologies and feedstocks. Other possible feedstocks for a pyrolysis-based refinery include urban wood waste, grass clippings, and food waste. Once developed, HTL could produce biojet fuel from algae or municipal solid waste. And while minimal, several HEFA facilities using used cooking oil and other rendering fats could be developed to tap into this local resource. If ICAO's goal is going to be met, multiple technologies and processes will have to be developed simultaneously and their regional locations optimized to ensure the maximum GHG reductions are obtained.

## CHAPTER 4: CONTRIBUTIONS AND FUTURE WORK

### 4.1 Main Contributions and Conclusions

One of the main contributions of this thesis is the review of LCA biojet fuel literature and the LCA methodology used within. The results were harmonized and presented to facilitate comparison of the different feedstock and technology pathways between models. The variables used within LCA were also identified and their inputs tabulated and compared between models and studies. This provided insight into the differences seen between LCA literature results and allowed the variable impacts on results to be quantified. The second part of the thesis conducts a lifecycle assessment of the pyrolysis pathway to producing biojet using local feedstocks in British Columbia. Several locations and feedstocks (forest residue and wood pellets) were modeled and compared using the same technology to determine if there was a more suitable location for the development of this technology in the province. In both parts, sensitivity analyses were performed to determine the impact that LCA assumptions and variables have on results. Based on the analyses and review, this work identified several key findings:

- 1. Gasification generally provides the lowest GHG emissions of all thermochemical and oleochemical pathways.** Even with differences between models, each dataset showed that this technological pathway gave the greatest reductions on average for all pathways studied. This is likely because of the low energy and hydrogen requirements of this pathway compared to others, as well as the ability to use mainly waste feedstocks (e.g. forest residue and corn stover).
- 2. Land use change, the displacement allocation method, source of hydrogen, and regional electricity mix have the largest impact on results.** The analysis of literature variables used in the review indicated that these variables and assumptions had the largest impact on the results of the lifecycle analyses in literature. The degree of impact on results varied for each variable by pathway but could be as large as 700% in the case of land use change, though no less than 30% for all others. As a result, transparency regarding these variables when publishing LCA results is critical and caution should be taken when comparing studies where these variables vary.
- 3. Vancouver, Vancouver Island, and Prince George are all suitable locations for the development of a pyrolysis bio-jet fuel facility.** All three locations have suitable access to either forest residue or wood pellets in sufficient quantities to support a 100 MLPY pyrolysis facility. Furthermore, the lifecycle analysis of a pyrolysis facility built in each location indicated that there is little difference

in terms of GHG emissions for each scenario, even when different feedstocks are used. Instead, the source of hydrogen used during pyrolysis upgrading and the source of biomass used to produce the wood pellets should be considered, as these have a larger impact on GHG reduction.

- 4. For ICAO to reach their GHG reduction goal in B.C., approximately 117 facilities would be required to support the Vancouver airport (assuming the scenarios studied) or 63 under optimum GHG-reducing conditions.** However, there is only enough forest residue and pellets available to supply approximately 15 facilities. As a result, other technologies and feedstocks will also have to be employed if ICAO wants to achieve their goal in B.C.

This work should not only help LCA practitioners but also provide policy and decision-makers with insights when using LCA within a regulatory context. When striving to maximize GHG reductions, this thesis identifies which aspects of biojet production should be of focus. It also identifies which variables can be manipulated to artificially reduce GHG emissions. As a result, the findings of this work can be utilized by either fuel blenders seeking how to maximize their low carbon fuel credits under the B.C. Renewable and Low Carbon Fuel Requirements Regulation or the California Air Resource Board's Low Carbon Fuel Standard or by the regulatory bodies when ensuring compliance. This work also identifies which variables must be identified and similar to accurately compare LCA results from different studies.

This work also identifies the magnitude of new facilities that are required to reach ICAO's goal in the province, which provides insight into the amount of infrastructure and feedstock that will be required to significantly reduce GHG emissions across the globe. The large increase in infrastructure required as well as the insufficient availability of a single type of feedstock highlights how essential it will be that all technologies and feedstocks are utilized to their fullest potential. Generally only food-based feedstocks such as vegetable oils are considered limited, but this work shows that advanced feedstocks are also limited. For this reason, it will be important to maximize the use of technologies with the lowest reductions, such as gasification, with these limited resources to reach ICAO's goal.

## 4.2 Recommendations and Future Work

**Perform a Meta-Analysis on the biojet literature data presented in Chapter 2.** The data presented in Chapter 2 was displayed as a compilation of LCA literature results, with some harmonization to enhance comparability. While the systematic review of the variability between studies provided an insight into their independent effects on results, a meta-analysis would provide GHG emission averages for each

technology, based on the weighting of literature data according to their variable choices. Hopefully, rather than a range of emissions, a meta-analysis would provide policy-makers with one value that might be used to try rank technologies. The systematic review performed in this thesis was the first step in preparing the literature data for meta-analysis.

**Perform a compilation of Life Cycle Analyses and Techno-economic analyses including each biojet technological and feedstock pathway utilizing identical assumptions, model, and databases.** While this thesis attempted to harmonize and present the current body of LCA literature studies to facilitate comparison, methodology limitations restricted concrete conclusions being derived regarding the merits of each technology. By performing an LCA with identical assumptions and databases, these limitations would be removed, and the conclusions drawn wherein confirmed. Combining the LCA studies with techno-economic studies should provide information about the relative merits of each technology and help decision-makers determine which areas should be a focus.

**Produce a Technological Map of Canada highlighting the optimal locations for crop/technology development to reach ICAO's GHG reduction goals.** The production of Biojet fuel can result in different carbon intensities based on the plants location. It was apparent that some areas are better suited for certain technological development than others. A map of the most optimal locations for each technology/crop across Canada, along with the amount of facilities and sizes required to reach ICAO's GHG reduction goals, would help producers determine which technology to develop in their own region.

**Develop and maintain one world database with country-specific data relevant to biofuel production that all LCA models can access.** This would reduce the significant amount of differentiation between models and provide the most relevant and up-to-date data. This is typically a limitation for most models. This would also remove the barriers which prevent groups from using models produced in other countries. It could be the first step towards normalizing models that can be used to facilitate GHG reduction efforts in multi-national industries such as aviation.

**Further assess the use of the displacement allocation method within the B.C. Low Carbon Fuel Standard and the federal Clean Fuel Standard.** This allocation method combines a holistic aspect with an otherwise process-based methodology. It has the potential to skew results under certain circumstances. The LCA system boundaries when using this methodology should be assessed to determine inconsistencies. These should be addressed to ensure fair representation amongst fuels.

**Update the B.C. case study to include more specific information when it becomes available.** Specific additions that were previously mentioned include:

- Determine how compositional differences in biomass affect the pyrolysis operating conditions and overall cost/GHG emissions.
- Determine the GHG and cost benefits of co-processing bio-oil with petroleum at a refinery.
- Determine how steam and heat recycle between the pyrolysis plant and upgrader affects the co-location energy balance and LCA. Determine the breaking point of biomass moisture content and distance at which it becomes optimal to co-locate incorporating these synergies.
- Determine how access to pellets around Prince George based on existing contracts will affect the LCA and include actual energy/mass balance data from each supplying plant.
- Determine whether the wastewater produced during the pyrolysis and hydrotreatment processes requires additional treatment before entering the Parkland/Husky refinery or Metro Vancouver/Prince George municipal wastewater treatment facilities. If so, the extra emissions from this processing should be quantified and included in the LCA system boundary.

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APPENDICES

Appendix I: Literature LCA Data

Table 23: GHG Emission LCA Data used in Literature Comparison Figures (gCO<sub>2</sub>eq/MJ)

	Petrol	Oleo-chemical (HEFA)									Gasification					Pyrolysis						HTL					
	Jet	Soy	UCO	Tallow	Palm	Jatropha	Rapeseed/Canola	Camelina	Corn	Algae	FR	CS	SRF	SG	MSW	FR	MiII	CS	SG	SRF	WS	Algae	Timb	FR	CS	Algae	
				GREET																							
Han et al. [96]	N/A	39				55	53	46				9.9				35.83		13.26									
Wong [87]	80.3-92 (85.0)	90.4			24.8-26.2						9-13	6.9															
Fan [60]	88.05	40.2-44.9			30.2-39.2	40-59	57.7-61.4			71.4						30.69		20.33									
Stratton et al. [7]	80.7-109.3 (87.5)	81.7			22.5-38.1	31.8-45.1	39.8-75.9			14.1-193.2	12.2	9		17.7													
Pearlson [21]	N/A	10.1-13																									

	Jet	Soy	UC O	Tall ow	Palm	Jatr oph a	Rape /Can ola	Cam elina	Co rn	Alga e	FR	CS	SRF	SG	MS W	FR	Mi ll	CS	SG	SRF	WS	Alg ae	Tim b	FR	CS	Alga e	
Huo et al. [61]	N/A	- 23.69 -28.4																									
de Jong et al. [28]	87.5		27			55		47			6	13	9			22									18		- 20
Seber et al. [71]	87.5		16.8 - 21.4	21.3 - 80.5																							
Frank et al. [72] <sup>1</sup>	NA									20.3																	
Carter [150]	87.5									8.9- 53.5																	
Elgowainy et al. [84]	80- 109	30								30	12. 75																
Frank et al. [72] <sup>1</sup>	NA																									29.3	
Xie et al. [85] <sup>1</sup>	NA										19. 5- 22																
Suresh [102]	NA														32.9 - 62.3												
Dang et al. [86]	NA																									- 18.1 4- 28.8 3	

		Soy	UC O	Tall ow	Palm	Jatr oph a	Rape /Can ola	Cam elina	Co rn	Alga e	FR	CS	SRF	SG	MS W	FR	Mi ll	CS	SG	SRF	Euc alyp tus	Alg ae	Tim b	FR	CS	Alga e
Guo et al. [88]	90																					68.3 - 146. 7				57
Bennion et al. [106]	NA																					234 - 278				
Davis et al. [108]	95 (Diese l)																									32- 105
SimaPro																										
Cox et al. [151]	82									38																
Li et al. [4]	90 (D)							3.1- 31																		
Shonnard et al. [74]	88							22- 29																		
Fan [60]	88.05 /90.0( D)			1.3- 31.4 2	7.84- 10.03		41.3	21.3 7	9.7 -26							34. 4- 37. 2		28.6 - 33.4								
Kalnes et al. (2009) [73]	85.9 (D)	17.3					41						8													
Kalnes et al. (2007) [75] <sup>2</sup>	85.6 (D)	17- 38.6																								

	Jet	Soy	UC O	Tall ow	Palm	Jatr oph a	Rape /Can ola	Cam elina	Co rn	Alga e	FR	Mi ll	SRF	Tim b	MS W	FR	Mi ll	CS	SG	SRF	WS	Euc alyp tus	Tim b	FR	CS	Alga e
<b>Meyer et al.</b> [135]	93.1 (Gas)																	31.9	36 .5	27.8 - 37.1						
<b>Iribarren et al.</b> [76]	NA																			23.8						
<b>Tews et al.</b> [152]	91.94 (D)/9 3.1(G)															34									27. 3	
<b>Crossin</b> [153]	NA																					42.4				
<b>Tzanetis et al.</b> [154]	87.5																								13. 1- 14. 4	
<b>Fortier et al.</b> [77]	88.1																									19.7- 132
<b>GHGenius</b>																										
<b>Fan</b> [60]	88.05 /90.0 (D)	34.6		16.4 3	54.7	59. 9		31.9																		
<b>(S&amp;T)2 Consultan 2010<sup>3</sup></b> [80]	89.1- 94.8 (D)	31.9	6.6	16.2	51.1	48. 3	24.1																			
<b>Novo Energy Group</b> [83] <sup>3</sup>	86.93- 89.84	28.8	1.7	9.95			10.1									37. 3- 53		15.1			33.4					

		Soy	UC O	Tall ow	Palm	Jatr oph a	Rape /Can ola	Cam elina	Co rn	Alga e	FR	Mi ll	SRF	Tim b	MS W	FR	Mi ll	CS	SG	SRF	Tim b	WS	Tim b	FR	CS	Alga e
<b>(S&amp;T)2 Consultants- 2013 [66]<sup>3</sup></b>	NA				90																					
<b>(S&amp;T)2 Consultants - 2012/03 [79]<sup>3</sup></b>	84.94			-3.6			12.1	25						20. 1												
<b>(S&amp;T)2 Consultants - 2012/08 [81]<sup>3</sup></b>	90.69 (D)										- 9.1						12. 9- 30. 2				17.6					
<b>(S&amp;T)2 Consultants - 2011 [114]</b>	89.2 (D)																28. 1		25 .3	39.6	34.7	18.6				
<b>Don O'Connor - 2013/03 [78]</b>	92.1 (D)										- 22. 2- 11. 9					35. 5- 49. 4		36.3						27. 37		

<sup>1</sup>Functional unit converted from gCO<sub>2</sub>eq/MMBtu to gCO<sub>2</sub>eq/MJ

<sup>2</sup>Cultivation N<sub>2</sub>O emissions added (26.3 gCO<sub>2</sub>eq/MJ from Stratton [7]) to make more comparable

<sup>3</sup>Functional Unit converted from gCO<sub>2</sub>eq/tonne-km to gCO<sub>2</sub>eq/MJ by applying a payload conversion factor of 0.015 MJ/tonne-km [83]

## Appendix II: Literature Variables and Assumptions

Table 24: Variable inputs for GREET literature

	Model Version	Land Use Change	Cultivation N <sub>2</sub> O Emissions	Co-Product Allocation	System Boundary	Functional Unit	Global Warming Potential (CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O)	Data Source	Process Conditions	Location	Year	System Choices
<b>Oleo-chemical Pathways (HEFA)</b>												
<b>Wong [87]</b>	GREET 1.8b	No iLUC. Direct LUC optional.	GREET defaults (crop residue not included)	Energy Allocation (fuels), Market Allocation (rest)	Renewable Jet Product (Not same as HRD)	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	GREET defaults, UOP, Huo et al, Wicke et al, Fargione et al	UOP HEFA Process	USA average	2008	UOP HEFA Process (hydro-deoxygenation)
<b>Stratton et al. [7]</b>	GREET 1.8a	No iLUC. Direct LUC optional.	IPCC Tier 1	Energy Allocation (fuels), Market Allocation (rest)	Renewable Jet Product (Not same as HRD)	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	GREET defaults and H. Wong	UOP HEFA Process	USA average	2008	UOP HEFA Process; Jet fraction maximized (25% of product slate). Jet fuel assumed to have same emissions as diesel.
<b>Fan [60]</b>	GREET 2012	No LUC	GREET defaults (crop residue not included)	Energy Allocation (fuels), Market/mass Allocation (rest)	Includes minimal wastewater treatment (CH <sub>4</sub> from POME)	gCO <sub>2</sub> e q/MJ	GREET defaults	GREET, GHGenius, R. Stratton, B. Bailis	UOP HEFA Process	USA average	2012	UOP HEFA Process
<b>Carter [150]</b>	GREET 2011	N/A	N/A	Energy allocation (fuel), Market allocation (rest)	Renewable Jet Product	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	GREET, Sierra et al, Liu et al	Various	USA average	2011	Outgas and CO <sub>2</sub> recycle; Open Pond, Tubular and Flat Panel
<b>Huo et al. [89]</b>	GREET 1.8	No LUC	GREET defaults (1.325%)	Various Scenarios: Energy, Market, and Hybrid	Renewable Diesel Product	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	UOP (Green Diesel), Canmet (SuperCetane), NREL	Not given	USA average	2008	Diesel fuel only: Diesel 1 - "SuperCetane", Diesel 2 - "Green Diesel"
<b>Frank et al. [72]</b>	GREET 2011	N/A	N/A	Energy allocation	Renewable Diesel Product	gCO <sub>2</sub> e q/MMB TU	IPCC 4th assessment( 1, 25, 298)	GREET, Williams and Lauren 2010, Lundquist et al 2010	Various: based on literature	USA average	2011	Considers CHP instead of Anaerobic digestion (fewer fugitive emissions); Open pond, dewatered by settling and centrifuge

<b>Han [20]</b>	REET 2012	No LUC	Not provided	Energy allocation	Renewable Jet Product	gCO <sub>2</sub> e q/MJ	Not provided	REET; Pearlson et al; Huo et al; Stratton et al; Shonnard et al; IPCC 2007	REET defaults; UOP; Pearlson	USA average	2012	SMR for H <sub>2</sub> production; Hydro-deoxygenatoin
<b>Pearlson [21]</b>	REET 1.8D	No LUC	Not provided	Energy allocation	Renewable Jet and Diesel, Process utilities included	gCO <sub>2</sub> e q/MJ	Not specified	UOP; REET defaults; Stratton et al.; H. Hong	Details given	USA average	2010	Hydro-deoxygenation, selective isomerization and catalytic cracking
<b>de Jong et al. [28]</b>	REET. net v1.3.0	No LUC	Not provided	Energy allocation (full); displacement /energy hybrid (non-fuel/fuel)	Renewable Jet	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	REET defaults	Some details provided in supplementary materials	USA average	2016	H <sub>2</sub> from SMR
<b>Elgowainy et al. [84]</b>	REET 2011	No LUC	1.325% of N in fertilizer and crop residue	Energy/displacement hybrid (though unclear if this was used for modeled pathway)	Renewable Diesel (assumed same as Renewable Jet)	gCO <sub>2</sub> e q/MJ; gCO <sub>2</sub> e q/kg-km; gCO <sub>2</sub> e q/passenger-km	REET defaults	REET defaults, Huo et al; Pradhan et al; Omni Tech; National Agricultural statistics; Frank et al; Pearlson	From given sources	USA average	2011	Modeled with and without CCS; UOP Hydro-deoxygenation process
<b>Gasification Pathways</b>												
<b>Wong [87]</b>	REET 1.8b	No iLUC. Direct LUC optional.	REET defaults ; N <sub>2</sub> O credit from removed crop residue > N <sub>2</sub> O from fertilizer input	Energy Allocation (fuels), Market Allocation (rest)	Renewable Jet	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment( 1, 25, 298)	REET defaults; Baitz et al;	Process efficiency of 45%; Biomass to fuel yield of 6:1	USA average	2008	CHOREN process (REET default)

<b>Stratton et al. [7]</b>	REET 1.8b	No iLUC. Direct LUC optional (SOC change). Agr. Management included	IPCC Tier 1; N <sub>2</sub> O credit from removed crop residue > N <sub>2</sub> O from fertilizer input	Energy Allocation (fuels), Market Allocation (rest)	Renewable Jet	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment (1, 25, 298)	REET defaults; Vardas et al; McLaughlin et al; Schmer et al; Whilmhelm et al; National Academies; Wilhelm et al;	Details given. Process efficiency of 45%	USA average	2008	Provided with or without CCS
<b>Elgowainy et al. [84]</b>	REET 2011	No LUC	N/A - No N <sub>2</sub> O for Forest Residue	Energy/displacement hybrid (though unclear if this was used for modeled pathway)	Renewable Diesel (assumed same as Renewable Jet)	gCO <sub>2</sub> e q/MJ; gCO <sub>2</sub> e q/kg-km; gCO <sub>2</sub> e q/passenger-km	REET defaults - likely 4th assessment	REET defaults, based on: Huo et al; Pradhan et al; Omni Tech International; National Agricultural statistics; Frank et al; Pearson, M	From given sources (some data specified)	USA average	2011	Modeled with and without CCS; Recycle gasifier
<b>de Jong et al. [28]</b>	REET. net (v1.3.0)	No LUC	REET defaults (1.325%) for N <sub>2</sub> O from N Fertilizer. Additional Fertilizer for corn stover, SRF	Energy allocation (full); displacement /energy hybrid (non-fuel/fuel)	Renewable Jet: 25% of Renewable Diesel. Biogas and electricity also co-products	gCO <sub>2</sub> e q/MJ	IPCC 4th assessment (1, 25, 298)	REET defaults; Swanson et al;	Some details provided in supplementary materials	USA average	2016	NA
<b>Xie et al. [85]</b>	REET 1.8d	No LUC	No N <sub>2</sub> O for Forest Residue	4 methods compared: Energy, market, and 2 displacement hybrids (displacement used for electricity co-product)	Renewably Diesel; WTW and WTP	gCO <sub>2</sub> e q/mmBtu	Not provided (likely REET defaults)	REET defaults; Kreutz et al;	Details given	USA average	2010	Forest residue includes mill residue and urban forest residue; Recycle design
<b>Han et al. [20]</b>	REET 2012	No LUC	For corn stover: N <sub>2</sub> O from fertilizer addition = crop	Energy allocation	Renewable Jet Product	gCO <sub>2</sub> e q/MJ	Not provided	REET; Pearson et al; Huo et al; Stratton et al; Shonnard et al; IPCC 2007	Various: based on literature and REET defaults	USA average	2012	SMR for H <sub>2</sub> production

			residue removal									
Pyrolysis Pathways												
<b>de Jong et al. [28]</b>	GREET. net (v1.3.0)	No LUC	Not included for Forest residue	Energy allocation (full); displacement /energy hybrid (non-fuel/fuel)	Renewable Jet (25% of Renewable Diesel)	gCO <sub>2</sub> eq/MJ	IPCC 4th assessm ent( 1, 25, 298)	GREET defaults; Tews et al; Wright et al	Tews process	USA average	2016	H2 from SMR or biogas/SMR
<b>Han et al. [20]</b>	GREET 2012	No LUC	For corn stover: N <sub>2</sub> O from fertilizer addition = crop residue removal	Energy allocation	Renewable Jet	gCO <sub>2</sub> eq/MJ	Not provided	GREET; Pearlson et al; Han et al, Burnham et al, Xie et al	Distributed and Integrated refinery (Wright)	USA average	2012	H2 from SMR; Biochar to electricity (rest used to dry feedstock with biogas)
<b>Fan [60]</b>	GREET 2012	No LUC	GREET defaults; Urea as fertilizer (higher emissions)	Energy Allocation	Ren. Diesel/gasoline mix product;	gCO <sub>2</sub> eq/MJ	GREET defaults - 4th assessm ent	GREET defaults; Wright et al; PNNL; Morey et al; UOP proprietary data; S. Jones (NREL)	Most details provided; corn stover 15% mc	Michigan, USA	2012	H2 from SMR or biogas/SMR; Integrated or Distributed; Char used as soil amendment (80%C)
<b>Dang et al. [86]</b>	GREET 2013	No LUC	Includes additional fertilizer needed for corn stover removal (8.49 g/kg corn stover); 2.21 g/kg P2O5 and 13.23 g/kg K2O	Displacement	Fuel is 50/50 Ren diesel and gasoline	gCO <sub>2</sub> eq/MJ fuel	Not provided	GREET defaults, Aspen Model; Wright et al; PNNL; Hsu (NREL); T. Marker (UOP)	Lots of detail in paper: 50% corn stover transported 61 km at 25% mc; dried to 7% mc; 71.6% yield for bio-oil; 43-16% yield	USA average	2010	Look at 3 scenarios: 1) All H2 from SMR; 2) 35% of aqueous bio-oil to H2 (all needed); 3) All aqueous bio-oil (62%) to H2 (leftover is co-product)

<b>Bennion et al.</b> [106]	GREET 2013	N/A	N/A	Not given	Renewable Diesel Product	gCO <sub>2</sub> eq/MJ	IPCC 4th assessment( 1, 25, 298)	GREET defaults; Chudnovsky et al; Shelef et al; Internal photo-bioreactor model and test plot data; NREL; PNNL	29.3% Bio-oil yield; 51% efficient; All char and biogas burned internally for energy	USA average	2013	Open Pond, Catalytic Pyrolysis, Rotary Kiln drying,
<b>Guo et al.</b> [88]	GREET 1 (2015)	N/A	N/A	Energy allocation	Renewable jet product	gCO <sub>2</sub> eq/MJ	Not provided	GREET model; Yang et al	3 technologies assessed: lipid extraction; pyrolysis and hydrotreatin; pyrolysis and co-processing	China	2015	Char used as soil amendment
<b>Hydrothermal Liquefaction Pathways</b>												
<b>Davis et al.</b> [108]	GREET 1.5 (2013)	N/A	N/A	Not given	Renewable Diesel Product	gCO <sub>2</sub> eq/MJ	Not given	GREET defaults; PNNL	Some details in Supplementary materials	USA average	2013	Continuous HTL process; Aqueous HTL layer fed to cat. Gasifier
<b>Frank et al.</b> [72]	GREET 2011	N/A	N/A	Energy allocation	Renewable Diesel Product	gCO <sub>2</sub> eq/MMBTU	IPCC 4th assessment( 1, 25, 298)	GREET, Williams and Lauren 2010, Lundquist et al 2010; Minowa et al, Jena et al, Yu et al	Various: based on literature; 33% yield	USA average	2011	Considers CHP instead of Anaerobic digestion (fewer fugitive emissions); Open pond, dewatered by settling and centrifuge
<b>Bennion et al.</b> [106]	GREET 2013	N/A	N/A	Not given	Renewable Diesel Product	gCO <sub>2</sub> eq/MJ	IPCC 4th assessment( 1, 25, 298)	Vardon et al; Jones, et al (PNNL); Lopez et al; Biller et al.	55% efficient (feed to bio-oil)	USA average	2013	Energy Recycle

Table 25: Variable inputs for SimaPro Literature

	Model Version	Land Use Change	Cultivation N <sub>2</sub> O Emissions	Co-Product Allocation	System Boundary	Functional Unit	GWP (CO <sub>2</sub> , CH <sub>4</sub> , N <sub>2</sub> O)	Data Source	Process Conditions	Location	Year	System Choices
<b>Oleo-chemical (HEFA) Pathways</b>												
<b>Cox et al. [151]</b>	SimaPro 7.3.3	N/A	N/A	Market allocation; displacement (separate)	Renewable Jet fuel (diesel and naphtha co-products)	gCO <sub>2</sub> e q/MJ	Not given	Agusdinata DB et al; Australian Unit Processes LCI Database	Honeywell UOP Process	Australia	2013	Electricity from biogas and grid; solvent extraction
<b>Li et al. [4]</b>	SimaPro 7.2	No LUC included.	Uses full IPCC formula, specific to Canadian Praries (SK)	Displacement	Renewable Jet fuel	gCO <sub>2</sub> e q/MJ (LHV)	Impact 2002+ Method (Long time horizon)	Ecolinvent; Vakulabharanam et al; Saskatchewan growers; Shonnard et al; Miller and Kumar; Kalnes et al; EPA; Stratton et al	NA	Canadian Praries (SK)	2013	NA
<b>Shonnard et al. [8]</b>	SimaPro 7.1	No LUC included.	IPCC for fertilizer (1.325%). None for crop residue	Energy Allocation (LHV)	Renewable Jet fuel	gCO <sub>2</sub> e q/MJ	IPCC 2007 100 year (1, 25, 298)	Ecolinvent Database, US Electricity data; Farmer data (WA)	UOP Process	USA (WA area)	2010	H <sub>2</sub> : 60% from naphtha/gasoline reforming; 40% from SMR
<b>Fan [60]</b>	SimaPro 7.2	dLUC included, but separable	IPCC Tier 1 (Full equation)	Camelina: EA/displacement hybrid; rapeseed: EA; Palm EA/SE; tallow: EA or SE	Renewable Jet Fuel; HRJ and HRD assumed to be same emissions	gCO <sub>2</sub> e q/MJ	IPCC 2007 (1, 25, 298)	Ecolinvent Database, Ecoindicator 99, EPA, CARB, Fan et al; Pippin et al; UOP; Mueller	Generally UOP with some variations	USA	2012	Does not include wastewater treatment (eg methane from POME effluent); bone meal not considered co-product for tallow (soap instead)
<b>Kalnes et al. (2009) [73]</b>	SimaPro 7.0	LUC not included	Soybean: some included; Rapeseed: included	Soybean: Mass allocation; Rapeseed: Displacement	Green Diesel product	gCO <sub>2</sub> e q/MJ	Ecolindicator 95	UOP; Ecolinvent Database; DOE (Soy); CONCAWE (Rapeseed); National Academy of Science (Soy)	UOP Process	USA	2009	H <sub>2</sub> from bio-naphtha (60%) and SMR (40%)

<b>Kalnes et al. (2007)</b> [75]	SimaPro 7.0	LUC not included	N <sub>2</sub> O emissions NOT included	Mass allocation	Green Diesel product	gCO <sub>2</sub> e q/MJ	EcoIndicator 95	UOP (Marker et al.); EcoInvent Database	UOP Process	Western EU	2007	Two cases: All H <sub>2</sub> from bio-naptha (60%) and biogas (40%), or H <sub>2</sub> from bio- naptha(60%) and SMR(40%)
<b>Seber et al.</b> [71]	SimaPro 7.3.3	No LUC included.	N/A	Market allocation (non-fuel); Energy allocation (fuel)	Diesel and Jet modeled	gCO <sub>2</sub> e q/MJ	IPCC 2007 100 year (1, 25, 298)	REET, SimaPro, M. Pearlson	Maximum Jet and Maximum Diesel cases, based from Pearlson (UOP HEFA)	USA	2013	Rendering included; Tallow both waste (low) and co-product (high);
<b>Gasification Pathways</b>												
<b>Kalnes et al. (2009)</b> [73]	SimaPro 7.0	LUC not included	N/A	Displacement	No info given	gCO <sub>2</sub> e q/MJ	EcoIndicator 95	EcoInvent Database; CONCAWE;	Not provided	Europe	2009	Not given
<b>Pyrolysis Pathways</b>												
<b>Meyer et al. (PNNL)</b> [135]	SimaPro 8	LUC Not included	REET defaults: 1.325% of fertilizer application/crop residue. Except pine, crop res. Not incl.Corn stover: Fertilizer add = crop residue	Not provided	Product is gasoline/diesel blend	gCO <sub>2</sub> e q/MJ	IPCC 4th amendment 100 year	Howe et al.; EcoInvent database; REET database; INL	All based on experimental data (Howe et al.) Transport radius of pulpwood (poplar) larger than other feedstocks	USA	2016	Fast pyrolysis and hydrodeoxygenation. Wood and stover assumed to dry to 30% mc in field (switchgrass 20%), dried to 10% mc and then converted into pellets prior to pyrolysis
<b>Iribarren et al.</b> [76]	SimaPro 7	Not included	Details not provided	Economic allocation	Product is diesel	gCO <sub>2</sub> e q/MJ	Not provided	Jones et al (NREL); Peacocke et al; J. Fan et al; EcoInvent database	Fast Pyrolysis (72% bio oil yield) and hydrotreating/hydrocracking	Spain	2011	H <sub>2</sub> from SMR and biogas
<b>Fan</b> [60]	SimaPro 7.2	N/A	N <sub>2</sub> O emissions from corn stover (fertilizer application)	Energy allocation	Gasoline and Diesel mix; Char used for energy	gCO <sub>2</sub> e q/MJ	IPCC 2007 (1, 25, 298)	EcoInvent Database, T. Kalnes, Envergent, Mihalek et al,	RTP Technology	Quebec (FR); Iowa (CS)	2012	Scenario 1: All H <sub>2</sub> from SMR; Scenario 2: H <sub>2</sub> from biogas and SMR

			included, w/ ammonium)									
<b>Tews et al. (PNNL) [152]</b>	SimaPro 8	N/A	N/A	Energy allocation	Renewable Jet fuel product	gCO <sub>2</sub> e q/MJ	IPCC 100 year	Johnson et al; Ecoinvent; US LCI database; Jones et al; GREET	50/50 forest residue and tree thinings;	USA	2014	H <sub>2</sub> from biogas and SMR; Wastewater treatment included
<b>Crossin [153]</b>	SimaPro 8.0.4.6	dLUC included (displaces sheep farming)	IPCC defaults	Displacement (non-fuel) and Energy Allocation (fuel)	Renewable Jet fuel product	gCO <sub>2</sub> e q/MJ	IPCC 4th amendment ( CH <sub>4</sub> adjusted from 25 to 22.25)	IFP Energy Nouvelle; FFCRC (K. Goss et al); Dynamotive Energy systems; Jungbluth et al;	Most given, some proprietary data	Western Australia	2015	Char used as soil amendment (credit); H <sub>2</sub> from SMR; Wastewater treatment included (at municipal facility)
<b>Tzanetis et al. [154]</b>	SimaPro 7.3	Not included	Not given	Economic allocation	Renewable Jet fuel product	gCO <sub>2</sub> e q/GJ	Not given	Ecoinvent 2.2; Aspen Plus simulation	Different catalysts assessed	EU (also Sweden)	2017	Catalytic Pyrolysis; H <sub>2</sub> from biogas (after recycle) and SMR
<b>Hydrothermal Liquefaction</b>												
<b>Tzanetis et al. [154]</b>	SimaPro 7.3	Not included	Not given	Economic allocation	Renewable Jet fuel product	gCO <sub>2</sub> e q/GJ	Not given	Ecoinvent 2.2; Aspen Plus simulation; Sun et al; Mortenson et al; Goudriaan et al; Veses et al; Panisko et al; Furimsky et al	Different catalysts and temperatures assessed; Most important inputs provided	EU (also Sweden)	2017	H <sub>2</sub> from biogas (after recycle) and SMR;
<b>Tews et al. (PNNL) [152]</b>	SimaPro 8	N/A	N/A	Energy allocation	Renewable Jet fuel product	gCO <sub>2</sub> e q/MJ	IPCC 100 year	Johnson et al; Ecoinvent; US LCI database; Jones et al; GREET	50/50 forest residue/ tree thinings; 14% higher bio-oil yield than pyrolysis	USA	2014	H <sub>2</sub> from biogas and methane (from wastewater treatment digesters); Wastewater treatment included
<b>Fortier et al. [77]</b>	SimaPro 7.3.3	N/A	N/A	Not provided	Renewable Jet fuel product	kgCO <sub>2</sub> e q/GJ	IPCC as per TRACI	Mostly lab and pilot-scale primary data; Literature for data provided	Wastewater effluent is water and nutrient source for algae	USA	2014	HTL at refinery and alone modeled; Wastewater treatment not included;

Table 26: Variable inputs for GHGenius Literature

	Model Version	Land Use Change	Cultivation N <sub>2</sub> O Emission	Co-Product Allocation	System Boundary	Functional Unit	Emission Potential (GWP)	Data Source	Process Conditions	Location	Year	System Choices
<b>Oleo-chemical (HEFA) Pathways</b>												
<b>Fan [60]</b>	GHGenius 4.01	Agricultural Management and SOC changes included (dLUC)	IPCC Tier 2: N <sub>2</sub> O of both fertilizer and crop residue (includes regionalization and crop-specific values)	Displacement	Renewable Jet Fuel; Includes aviation emissions	gCO <sub>2</sub> e q/MJ	Model defaults	Model defaults	Model defaults	Canada	2012	Model defaults; Includes wastewater treatment; Methane from POME = 44.5 gCO <sub>2</sub> eq/MJ
<b>(S&amp;T)2 Consultants (2010) [80]</b>	GHGenius 3.19	Agricultural Management and SOC changes included (dLUC)land to cropland	Uses Environment Canada factors; does not appear to include crop residue; IPCC Tier 1	Displacement	Renewable Jet	gCO <sub>2</sub> e q/GJ	IPCC 2007 100 year	Environment Canada; EPA; GHGenius defaults	Model defaults	Canada avg	2010	Model defaults; Includes wastewater treatment (eg methane from POME = 22.5 gCO <sub>2</sub> eq/MJ)
<b>Novo Energy Group [83]</b>	GHGenius 4.03	Agr. Management + % of deforestation/peatland conversion for palm (dLUC)	IPCC Tier 3: Very regionalized/crop specific (USA); IPCC Tier 2: Some regionalization and crop specific (Canada)	Displacement	Renewable Jet; Canola and Tallow for W Canada, UCO and Soy for E. Canada	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; Canada Canola Council; Agri Food Canada; GHGenius defaults; Pearlson et al	UOP Process; 50% of fuel yield is biojet fuel	Eastern Canada (ON) and Western Canada (AB)	2020 and 2025	Uses all system defaults except for transport distance, which is modeled based on real locations in E and W Canada; Process is optimized for maximum biojet yield

<b>(S&amp;T)2 Consultants (2013)</b> [66]	GHGenius 4.03	dLUC included as before, but now also includes % of deforestation/peatland conversion for palm	IPCC Tier 3: Very regionalized/crop specific (USA); IPCC Tier 2: Some regionalization and crop specific (Canada)	Displacement	Renewable Jet; Includes aviation emissions	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; Canada Canola Council; Agri Food Canada; GHGenius defaults	Model defaults	Canada avg	2013	Model defaults
<b>(S&amp;T)2 Consultants (2012/03)</b> [79]	GHGenius 4.01	Agricultural Management and SOC changes included (dLUC)	IPCC Tier 2: N <sub>2</sub> O of both fertilizer and crop residue (includes regionalization and crop-specific values)	Displacement	Renewable Jet; Includes aviation emissions	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	GHGenius defaults	Model defaults	Canada avg	2012	Model defaults; Includes wastewater treatment (eg methane from POME = 41.7 gCO <sub>2</sub> eq/MJ);
<b>Gasification</b>												
<b>(S&amp;T)2 Consultants (2012/03)</b> [79]	GHGenius 4.01	Not specified	Not specified	Displacement	Not specified. Fuel is Diesel and gasoline mixture	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	GHGenius defaults	Model defaults	Canada avg	2012	Model defaults
<b>(S&amp;T)2 Consultants – (2012)</b> [81]	GHGenius 4.02	dLUC included; none for corn stover, mill residue, or wheat straw	None assumed for corn stover, mill residue, or wheat straw	Displacement	Fuel is Diesel and Gasoline mixture	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; GHGenius defaults; Jones et al; NREL; ENSYN	Corn stover/wood same efficiency; MC of 25% to 7%; Pyrolyzer at 480C and atmP;	Canada avg	2012	Fast Pyrolysis; hydrotreating and hydrocracking (Same as NOVO Group); Char and bio-gas used to heat feedstock(all for wood, most for corn stover)

<b>Don O'Connor (2013/03)</b> [78]	GHGenius 4.00c	Not included	Not included (Fertilizer assumed to negate lost N <sub>2</sub> O emissions for corn stover)	Displacement	Diesel/gasoline mix (ratio not specified)	gCO <sub>2</sub> e q/GJ	IPCC 2007 100 year	GHGenius defaults; ENSYN company data; Jones et al. for upgrading	ENSYN process; Corn stover/wood same efficiency; MC of 25% to 7%; Pyrolyzer at 480C and atmP;	USA avg	2012	Fast Pyrolysis; hydrotreating and hydrocracking; Char and bio-gas used to heat feedstock(all for wood, most for corn stover(rest displaces NG); H <sub>2</sub> from SMR and bio-oil (for forest res)
<b>Pyrolysis Pathways</b>												
<b>(S&amp;T)2 Consultants – (2012)</b> [81]	GHGenius 4.02	None assumed for corn stover, mill residue, or wheat straw	None assumed for corn stover, mill residue, or wheat straw	Displacement	Fuel is Diesel and Gasoline mixture	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; GHGenius defaults; Jones et al; NREL	Corn stover and wood assumed to have same efficiency (53%)	Canada avg	2012	Uses high temperature scenario of NREL (GHGenius default)
<b>Novo Energy Group</b> [83]	GHGenius 4.03	dLUC included; none for corn stover, mill residue, or wheat straw	None assumed for corn stover, mill residue, or wheat straw	Displacement	Fuel is biojet	gCO <sub>2</sub> e q/tonne-km	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; GHGenius defaults; ENSYN; Jones et al; NREL	Model defaults: MC of 7% before pyrolysis from 25%;	Eastern Canada (ON) and Western Canada (AB)	2020 and 2025	Uses all system defaults except for transport distance, which is modeled based on real locations in E and W Canada; Char is not given a displacement credit (used internally to dry the feedstock)
<b>(S&amp;T)2 Consultants – (2011)</b> [114]	GHGenius 3.20	None assumed for corn stover, mill residue, standing timber, SRF or wheat straw. dLUC included for switchgrass	Fertilizer included for SRF, switchgrass, wheat straw and corn stover (to replace the removed crop residue)	Displacement	Fuel is Diesel and Gasoline mixture; Road infrastructure included for standing timber	gCO <sub>2</sub> e q/GJ	IPCC 2007 100 year	Environment Canada; EPA; US National GHG Inventory; GHGenius defaults; Jones et al; NREL; ENSYN	ENSYN process; Corn stover/wood same efficiency; MC of 25% to 7%; Pyrolyzer at 480C and atmP;	Canada avg	2011	Fast Pyrolysis; hydrotreating and hydrocracking (Same as NOVO Group); Char and bio-gas used to heat feedstock(all for wood, most for corn stover); H <sub>2</sub> from SMR
<b>Don O'Connor – (2013/03)</b> [78]	GHGenius 4.00c	Not included	Not included	Displacement	Diesel/gasoline mix (ratio not specified)	gCO <sub>2</sub> e q/GJ	IPCC 2007 100 year	GHGenius defaults; NREL (based on corn stover data)	HT efficiency = 53%, LT efficiency = 43%	USA avg	2012	Models both a Low temperature ( and High Temperature option; Excess electricity produced offsets grid electricity

Hydrothermal Liquefaction Pathways												
Don O'Connor – (2013/03) [78]	GHGenius 4.00c	Not included	Not included	Displacement	Diesel	gCO <sub>2</sub> e q/GJ	IPCC 2007 100 year	GHGenius defaults; Faber and Gielen (says PNNL data insufficient)	50% mc (fr); biocrude with 10% o <sub>2</sub> ; 40% yield (optimistic)	USA avg	2012	Not much info provided; H <sub>2</sub> from SMR

## Appendix III: Proofs and Detailed Calculations

### 1. Displacement vs. Energy Allocation Method Proof, excerpt from de Jong et al. [28]

For a hypothetical system with one main product and one co-product the GHG emission intensity of the main product can be calculated as follows:

$$\text{Energy allocation: } \varepsilon_{\text{main product,EA}} = \varepsilon_{\text{system}} \quad (1)$$

$$\text{Displacement: } \varepsilon_{\text{main product,DM}} = \frac{\varepsilon_{\text{system}} - (1 - \eta_p) * r_D * \varepsilon_{\text{displaced product}}}{\eta_p} \quad (2)$$

In which  $\varepsilon$  is the specific GHG emission intensity for the total system (including the product and co-products) and the displaced product. The product share ( $\eta_p$ ), co-product allocation ratio ( $r_A$ ) and displacement ratio ( $r_D$ ) are defined here as:

$$\eta_p = \frac{1}{1 + r_A} = \frac{E_p}{E_p + E_{cp}} \text{ in which } r_A \equiv \frac{E_{cp}}{E_p} \quad (3)$$

$$r_D \equiv \frac{E_{cp}}{E_{dp}} \quad (4)$$

In which  $E$  is the energy content of the product ( $p$ ), co-product ( $cp$ ) and displaced product ( $dp$ ). The parameters  $\eta_p$  and  $r_D$  can be defined in terms of units (e.g. kg/kg, MJ/MJ). To avoid conversion, we use MJ/MJ in this example. It should be noted that  $\eta_p$  is inversely proportional to  $r_A$ . Furthermore, equation 1 and 2 show that while energy allocation yields strictly positive emission intensities (excluding the possibility that the total system is carbon negative due to the application of carbon capture and storage), the displacement method may give negative emission intensities.

The difference ( $\Delta$ ) between the GHG emission intensity of the main product using energy allocation and the displacement method is expressed as:

$$\begin{aligned} \Delta = \varepsilon_{EA} - \varepsilon_{DM} &= \left( \frac{1}{\eta_p} - 1 \right) * (r_D * \varepsilon_{\text{displaced product}} - \varepsilon_{\text{system}}) \\ &= r_A * (r_D * \varepsilon_{\text{displaced product}} - \varepsilon_{\text{system}}) \quad (5) \end{aligned}$$

The sign of the difference is determined by the term  $r_D * \varepsilon_{\text{displaced product}} - \varepsilon_{\text{system}}$ . Hence, energy allocation yields higher GHG emissions when:

$$r_D * \varepsilon_{\text{displaced product}} > \varepsilon_{\text{system}} \quad (6)$$

The displacement method yields higher GHG emissions for systems in which:

$$r_D * \epsilon_{displaced\ product} < \epsilon_{system} \quad (7)$$

As an illustration we take a system in which the co-product is green electricity. As electricity is often a direct substitute for grid electricity  $r_d$  is likely equal to 1. As such, inequalities 6 and 7 reduce to a comparison between the emission intensity of the displaced product (grid electricity) and the energy intensity of the system. For systems having lower specific emission intensity than grid electricity, the displacement method will always yield a lower GHG emission intensity for the main product. For systems having higher emission intensity than grid electricity, the energy allocation method will always yield lower results than the displacement method.

The size of  $\Delta$  grows with higher co-product allocation ratios and increasing divergence between the terms  $r_D * \epsilon_{displaced\ product}$  and  $\epsilon_{system}$ . In other words, larger differences between the results of both methods may be expected for systems producing high amounts of co-products. Furthermore, for systems which are much less emission intensive than the system producing the displaced product (i.e.  $\epsilon_{displaced\ product} \gg \epsilon_{system}$ ) larger differences between both methods occur for systems producing co-products that effectively displace emission intensive products (i.e. high  $r_D$  and  $\epsilon_{displaced\ product}$ ).

For systems with more co-products the general dynamics still hold, i.e. higher allocation ratios will lead to higher differences. Calculation of the sign of the difference requires, however, more parameters than postulated for the example system above.

## 2. Proof of refined bio-oil and pyrolysis jet fuel functional equivalence

In literature, there is a discrepancy in the functional units used between studies for refined pyrolysis oil. Some studies present their lifecycle GHG emissions in terms of MJ of renewable jet, others in terms of renewable diesel, and still others in terms of refined bio-oil (gasoline, diesel, and jet combined). The authors that utilize the latter two functional units argue that when energy allocation is employed (which is traditionally done when all products are fuels), the diesel and refined bio-oil functional units are practically equivalent to renewable jet fuel, since the HHV of the fuels is very similar. If this were the case, the GHG emission results would be the same regardless of the functional unit employed. To test this assumption, a simplified lifecycle assessment is presented and represented mathematically to assess their equivalence.

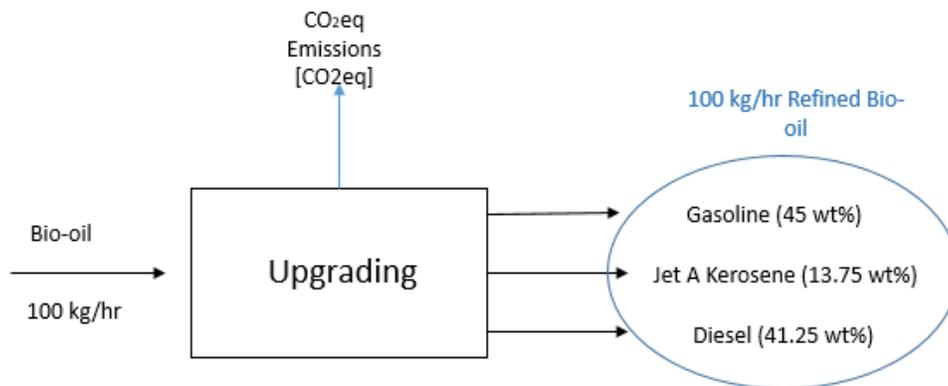


Figure 30: Simplified LCA of Bio-oil Upgrading

### Refined bio-oil Functional Unit

When the functional unit is refined bio-oil, the types of upgraded fuels are not broken down, but is rather a mixture of all fuels. According to Figure 1, if the HHV of the the refined bio-oil is considered to be  $x$  MJ/kg, then the CO<sub>2</sub>eq of upgrading is as follows when allocated by energy content:

$$\frac{CO_2eq \text{ Emissions/hr}}{Energy \text{ of Refined Bio - oil/hr}} = \frac{[CO_2eq]gCO_2eq/hr}{100 \text{ kg/hr} * xMJ/kg} = \frac{[CO_2eq] gCO_2eq}{100x \text{ MJ}} \text{ Refined Bio - oil} \quad (1)$$

### Jet A Kerosene Functional Unit

When the functional unit is Jet A Kerosene, the emissions must be allocated to each individual product and divided by the chosen unit of Jet A Kerosene. In this example, the functional unit is MJ Jet A Kerosene and

allocation is done by energy. Since the Jet A stream is our product, the energy allocation factor for this stream is calculated:

$$\begin{aligned} \frac{MJ \text{ Jet A}}{MJ \text{ Total products}} &= \frac{12.5 \frac{kg}{hr} * x(1+a) \frac{MJ}{kg}}{12.5 \frac{kg}{hr} x(1+a) \frac{MJ}{kg} + 37.5 \frac{kg}{hr} x(1+b) \frac{MJ}{kg} + 50 \frac{kg}{hr} x(1+c) \frac{MJ}{kg}} \\ &= \frac{12.5x(1+a)}{100x + (12.5a + 37.5b + 50c)} \end{aligned} \quad (2)$$

Where  $x$  = HHV refined bio-oil (MJ/kg)

$a$  = fraction difference of jet HHV and refined bio-oil HHV

$b$  = fraction difference of diesel HHV and refined bio-oil HHV

$c$  = fraction difference of gasoline HHV and refined bio-oil HHV

This allocation factor for the jet A stream is then multiplied by the CO<sub>2</sub> emissions for the upgrading process to determine the amount of emissions that should be allocated to this stream. This is then divided by the functional unit previously described, MJ jet A (or  $12.5*x(1+a)$ ):

$$\begin{aligned} \frac{gCO_2eq}{MJ \text{ Jet A}} &= \frac{12.5x(1+a)}{100x + (12.5a + 37.5b + 50c)} * \frac{[CO_2]}{12.5x(1+a)} \\ &= \frac{[CO_2]}{100x + (12.5a + 37.5b + 50c)} \frac{gCO_2eq}{MJ \text{ Jet A}} \end{aligned} \quad (3)$$

When the HHV of all the fuels are similar  $a, b,$  and  $c$  are approximately 0. This simplifies equation (3) to Eq. (4). In this case, the result is the same as that of using the refined bio-oil functional unit, proving their equivalence.

$$\begin{aligned} &\frac{[CO_2] gCO_2eq}{100x MJ \text{ Jet A}} \quad (4) \\ \frac{[CO_2] gCO_2eq}{100x MJ \text{ Jet A}} &= \frac{[CO_2] gCO_2eq}{100x MJ \text{ Refined Bio - oil}} \quad (4) = (1) \end{aligned}$$

When  $a, b, c \sim 0$  (i.e.. HHV approximately the same for all fuels)

In GHGenius, the HHV's of gasoline, diesel, jet and refined bio-oil are 46.9, 45.8, 46.3, and 46.1 MJ/kg respectively. For this case,  $x = 46.1$ ,  $a = (46.3-46.1)/46.1 = 0.004$ ,  $b = (45.8-46.1)/46.1 = -0.0054$ , and  $c = (46.9-46.1)/46.1 = 0.0173$ . When introduced into equation (3), it becomes:

$$\frac{[CO_2]}{100(46.1) + 0.613 \text{ MJ Jet A}} \frac{gCO_2eq}{A}$$

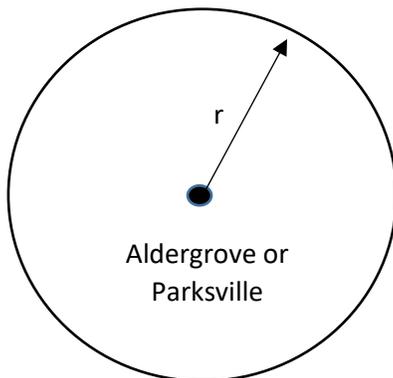
The difference between this equation and equation 1 are negligible, therefore the assumptions made by the authors regarding the equivalence of using the various functional units (MJ of refined bio-oil, MJ diesel, or MJ jet) is valid and will be adopted by this study.

### 3. Detailed Calculations (Chapter 3)

#### A) Transportation Distances

To calculate the distances the biomass must travel to reach the pyrolysis facility, the biomass required by the pyrolysis facility must be taken into account. The baseline biorefinery of this study is 100 MLPY of upgraded fuel (gasoline/diesel/jet blend). For this size, 400,000 ODT of biomass per year is required, based on estimates by Boeing [103].

For scenarios 1 and 2, the forest residue was assumed to be distributed evenly across an area of radius 155 km [103]. 758,880 ODT of forest residues were available within this radius [103], however, only the closest forest residues are used due to economic restraints. The minimum supply radius required to obtain the 400,000 ODT (52.8% of supply) was calculated as follows:



$$A = \pi r^2 \quad (\text{Eq. 1})$$

$$A = \pi (155)^2 = 75,477 \text{ km}^2$$

$$(0.528)A = \pi r_{S1}^2$$

$$r = 112.6 \text{ km}$$

$$\frac{A}{2} = \pi r_{avg}^2 \quad (\text{Eq. 2})$$

$$r_{avg} = 79.6 \text{ km}$$

The supply radius required to obtain the 400,000 ODT is 112.6 km for scenarios 1 and 2, but the average supply distance for the trucks to travel is 79.6 km.

For a 50 MLPY facility, the required forest residue supply is only 200,000 ODT/yr, so the supply area is reduced by half. The radius is determined therefore by dividing the left side of Eq 2 by half and solving for r. In this case, r becomes 56.3 km. When the facility is increased to 150 MLPY, 600,000 ODT/yr of forest residue is required and Eq.2 is multiplied by 3/2 to obtain a radius of 97.3 km.

For scenario 3, pellets were obtained from pellet plants surrounding the Prince George area. It was assumed that each facility was able to supply about 50% of their production to the biorefinery. Details of how the transportation distance was calculated for the 100 MLPY basecase is supplied in section 3.2.3. When the facility decreases to 50 MLPY, only 200,000 ODT/yr is required, which can be supplied by just Pacific Bioenergy at 15 km. Under these conditions, 57% of Pacific Bioenergy's stock is required.

For a 150 MLPY biorefinery, it was anticipated that the supply would continue to come from all 4 pellet production plants in the area, but that each would have to supply a larger share of their pellet production. The weighted distance was not assumed to change from the 100 MLPY basecase, but this could change if some pellet plants were more willing to provide the biomass than others.

## **B) Biogas required for drying biomass feed prior to pyrolysis**

GHGenius includes the energy from drying the mill and/or forest residue biomass from 50% to 10% moisture content (dry basis) as default for all pyrolysis processes. The energy for this drying comes from the biogas and some char. Under these conditions, all the biogas and char are consumed in the pyrolysis process for pre-conditioning of the biomass (drying and grinding) and process heat. However, when drying takes place elsewhere or the biomass feed has a lower moisture content than 50%, some biogas produced from the pyrolysis process will not be consumed. Under these conditions, excess biogas co-product becomes available.

In Scenario 3, all drying takes place at the pellet mill and the energy required is included in the energy inventory for this process. All of the biogas normally used for biomass drying at the pyrolysis facility is therefore available to receive a co-product credit. The amount of biogas co-product is determined by calculating the amount of biogas typically used to dry biomass from 50% to 10% moisture content (dry basis).

$$50\% \text{ mc (dry basis)} = 1 \text{ kg dry wood} + 0.5 \text{ kg H}_2\text{O}$$

$$10\% \text{ mc (dry basis)} = 1 \text{ kg dry wood} + 0.1 \text{ kg H}_2\text{O}$$

While the specific enthalpy of vaporization of water is 2.26 MJ/kg H<sub>2</sub>O, more energy is required in practice. M. Wright and Don O'Connor suggest that the energy required in practice is 3.6 MJ/kg H<sub>2</sub>O and 3 MJ/kg H<sub>2</sub>O, respectively. The energy required for drying (and now as biogas co-product) is:

$$3 \text{ to } 3.6 \frac{\text{MJ}}{\text{kg H}_2\text{O}} * 0.4 \text{ kg H}_2\text{O} = 1.2 \text{ to } 1.44 \frac{\text{MJ}}{\text{kg dry wood}} * \frac{1.65 \text{ kg dry wood}}{1 \text{ L bio-oil}} = 1.98 \text{ to } 2.38 \frac{\text{MJ biogas}}{\text{L bio-oil}}$$

### C) Pyrolysis biorefineries required to reach ICAO's goal

ICAO's goal is to reduce their overall GHG emissions by 50% over 2005 levels by 2050. In 2005, Canada consumed 104,000 barrels per day (6,036 MLPY) [139]. With a carbon intensity of 85.2 gCO<sub>2</sub>eq/MJ (average Canadian CI of jet fuel in 2005), the Canadian GHG emissions from jet fuel in 2005 was around 18.29 million tonnes CO<sub>2</sub>eq/year. Therefore, ICAO's Canadian target was anticipated to be around 6,875 tonnes CO<sub>2</sub>eq/year. According to ICAO projections, global aviation emissions will increase from 591 million to 4,531 million tonnes CO<sub>2</sub>eq/yr (767% increase) from 2006 until 2050 if no improvements are made. If advanced technology and operational improvements are made, 2050 emissions can reduce to around 2,496 million tonnes of CO<sub>2</sub>eq (422% increase).

Assuming that this increase applies to Canada, the expected emissions in 2050 (even with aircraft improvements) are 77.18 million tonnes CO<sub>2</sub>eq/year, 8.44 times higher than the target.

The GHG emissions of the pyrolysis biojet calculated in this thesis have reductions of 63.2 - 60.6 gCO<sub>2</sub>eq/MJ of fuel. Within a 100 MLPY biorefinery, 13.75 MLPY of biojet is produced (the balance is gasoline and diesel, see Appendix x). Therefore, the 100 MLPY facility proposed in this thesis for development in B.C. has the potential to reduce GHG emissions by 13.75 Million L/yr \* 35.57 MJ/L \* 61.9 gCO<sub>2</sub>eq/MJ = 30,274 tonnes/year. The amount of 100 MLPY pyrolysis biorefineries that would be required to be built by 2050 to reach ICAO's GHG reduction goal within Canada would therefore be:

$$\frac{77.18 \times 10^6 \text{ tonnes} \frac{\text{CO}_2\text{eq}}{\text{yr}} - 18.29 \times 10^6 \text{ tonnes} \frac{\text{CO}_2\text{eq}}{\text{yr}}}{30,274 \text{ tonnes CO}_2\text{eq/yr}} = 1,945 \text{ facilities}$$

This would provide Canada with 1945 \* 13.75 Million L/year = 26,744 MLPY of biojet fuel, which is over 100% of the expected consumption in 2050 (6,036 MLPY x 433%) = 25,472 MLPY. Therefore, all of the jet fuel consumed in Canada would need to be comprised of 100% pyrolysis based biojet.

According to Boeing, the Vancouver International Airport consumes roughly 1600 million liters of jet fuel per year [103]. At this rate, 1600 MLPY/13.75 MLPY = 117 biorefinery facilities would have to be built to support this airport alone. To support these refineries, over 400,000 ODT \* 117 = 46.8 Million ODT would be required for the YVR airport alone.

If electrolysis were used rather than SMR to produce hydrogen, fewer biorefineries would be required since the fuel has a higher carbon intensity. Since 117 biorefineries are required when the carbon reduction is 68% (scenario 3), this means that only 91 would be required when the carbon reduction is 88% (electrolysis):

$$(0.68) * 117 = (0.88) * x$$
$$x = 91$$

When displacement is used, the GHG reduction for the fuel is 117%, so the facility number drops to 68.

## D) Hydrogen and Electricity requirements

The hydrogen requirements used in this analysis were obtained from the report by Jones et al. [25], where the H<sub>2</sub> consumption during bio-oil hydro-treatment was found to be 4.96 lb/100 lb refined oil. The units are converted to L of refined bio-oil for use in this study as follows:

If electrolysis is used for hydrogen production, electricity must be used to produce this hydrogen. The electricity required to produce a MJ of hydrogen is based on the efficiency of the electrolyser, which was assumed to be 56 kWh/kg of hydrogen [58]. The electricity requirements are calculated as follows:

$$56 \frac{kwh}{kg H_2} * \frac{1 kg}{1000 g} * \frac{0.089 g}{L} * \frac{L}{0.0128} * 5.15 \frac{MJ H_2}{L RB} = 2.0 \frac{kwh}{L refined bio - oil}$$

It was assumed that the bio-gas produced during upgrading would be available for sale as a co-product rather than used for hydrogen production, since hydrogen was not being produced via electrolysis. The off-gas produced was obtained from Jones et al. [25] and converted from 0.128 g/hr (per g of refined bio-oil) and converted to 2.88 MJ biogas/L bio-oil using physical properties from Table 28 and assuming that bio-gas has similar properties to natural gas.

When biogas from pyrolysis was used to produce hydrogen, it was assumed that the biogas replaced an equal amount of natural gas on a volume basis. Therefore, since 112 – 127L of biogas was produced during pyrolysis (taken from Jones et al. [25]), the amount of natural gas required was reduced from 315L to 188-203L.

## E) Sensitivity Analysis Details

### 3.4.1.3 Details

If the mill residue was not available in Prince George and some were taken from the Vancouver area instead, this would add emissions due to the extra transportation. The distances used in modelling these extra emissions were 753 +/- 121 km, which covers the forested area surrounding the region. Therefore, the transportation distance added to the model for mill residue when 40% is obtained from the Vancouver region is: (0.8)\*57.16 + (0.4)\*753 = 347 km (+/- 48.7 km).

To determine the distance at which using mill residue does not provide a benefit over forest residue for scenario 1, the GHG emissions are modelled in GHGenius by removing the diesel usage of forest residue procurement and increasing the transport distance. Once the GHG emissions of this scenario are equal to the GHG emissions of scenario 1, the breakeven distance is recorded. The distance recorded was 113 km.

#### 3.4.1.4 Details

The quantity of biogas co-product available if the forest residue were dried onsite and did not require biogas for drying is calculated similarly to above in part B. In this instance, the biogas saved from not drying the residue from 50 to 20% moisture content is determined. Since this amount is  $\frac{3}{4}$  of that calculated in part B (which is from 50 to 10%), the amount saved in this scenario is  $\frac{3}{4} * (1.98 - 2.38 \text{ MJ/L ref bio-oil}) = 1.5 \text{ to } 1.75 \text{ MJ/L ref bio-oil}$ . This is added to the integrated case for scenarios 1 and 2 to determine the reduction in GHG emissions for these cases. When 1.75 MJ/L is considered for the integrated scenario 1 case, the GHG emissions drop from 26.55 to 25.94 gCO<sub>2</sub>eq/MJ.

## Appendix IV: Inventory Data for Chapter 3

**Table 27: Scenario and Sensitivity Analysis Inventory Data**

		Base-cases			Sensitivity Analyses														
Process	Inventory Parameter		S1	S2	S3	A S1	A S2	Bi	Bii	Biii	Ci	Cii	Ciii	Civ	Cv	Di	Dii	Diii	
Feedstock Preparation	Fuel for forest residue chipping and skid loader (L or kWh/tonne wood) <sup>1</sup>	Diesel	1.84	1.84	NA	1.84	1.84	1.84	0.92	0.99	1.84	1.84	1.84	1.84	1.84	1.84	NA	1.84	
		Electricity								0.006									
	Estimated Drying Energy (MJ/L bio-oil)		-	-	1.98-2.38	-	-	1.98-2.38	1.98-2.38	1.98-2.38	-	-	-	-	-	-	1.98-2.38	-	
Pellet Plant Operation <sup>3</sup>	Diesel (L/tonne pellet)		NA	NA	0.61	NA	NA	0.61	0.61	0.61	NA	NA	NA	NA	NA	NA	0.61	NA	
	LPG (L/tonne pellet)		NA	NA	0.24	NA	NA	0.24	0.24	0.24	NA	NA	NA	NA	NA	NA	0.24	NA	
	Electricity (kWh/tonne pellet)		NA	NA	136.1	NA	NA	136.1	136.1	136.1	NA	NA	NA	NA	NA	NA	136.1	NA	
Transportation	Forest Residues/pellets to fuel production facility <sup>1</sup>	by truck	79.6	79.6	57.16	134.6	117.6	300	167.5	103.2	64		91	79.6	79.6	79.6	57.16	57.16	
		by barge	0	0	0	0	90	0	0	0	0	0	0	0	0	0	0	0	0
	Pyrolysis oil to upgrading facility <sup>1</sup>	by truck	55	38	0	0	0	0	0	0	55	55	55	55	55	55	0	0	
		by barge	0	90	0	90	0	0	0	0	0	0	0	0	0	0	0	0	0
	Pyrolysis jet to airport blending facility <sup>1</sup>	by truck	7	7	23	7	7	23	23	23	7	7	7	7	7	7	23	23	
		by barge	37.5	37.5	0	37.5	37.5	0	0	0	37.5	37.5	37.5	37.5	37.5	37.5	0	0	
	Pyrolysis jet to airport blending facility <sup>1</sup>	by pipeline	30.75	30.75	0	30.75	30.75	0	0	0	30.75	30.75	30.75	30.75	30.75	30.75	0	0	
		by truck	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	
Pyrolysis plant Operation Data <sup>2</sup>	Location		AG	PV	PG	Burnaby	Burnaby	PG	PG	PG	AG	AG	AG	AG	AG	AG	PG	PG	
	Target (Production) Year		2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017	2017
	Moisture content reduction (% dry basis)		40	40	0	40	40	0	0	0	40	40	40	40	40	40	0	40	
	Electricity consumption (kWh/L bio-oil)		0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24	0.24
	Diesel (L/L bio-oil)		0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	Natural gas (L/L bio-oil)	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	21.2 – 23.9	0.02	23.72	0.02	0.02	0.02	0.02
	Coal (kg/L bio-oil)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Yield ( kg forest residue/L bio-oil)	1.65	1.65	1.65	1.65	1.65	1.65	1.65	1.65	1.65	1.71	1.54	1.65	1.65	1.65	1.65	
Co-Products Produced <sup>2</sup>	Gaseous fuel (MJ/L final fuel)	0	0	1.98- 2.38	0	0	1.98 – 2.38	1.98 – 2.38	1.98- 2.38	1.48 – 1.75	0	0.98	0	0	2.88	4.86 – 5.26	0
	Liquid fuel (MJ/L final fuel)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Electricity (kWh/L final fuel)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Upgrading plant Operation Data <sup>2</sup>	Location	Burn aby	Burna by	PG	Burn aby	Burna by	PG	PG	PG	Burn aby	Burn aby	Burn aby	Burna by	Burna by	Burna by	PG	PG
	Target (Production) Year	201 7	2017	2017	201 7	2017	201 7	201 7	2017	201 7	201 7	201 7	2017	2017	2017	2017	201 7
	Electricity consumption (kWh/L final fuel)	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	0.56	2.56	2.56	0.56
	Diesel (L/L final fuel)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Natural gas (L/L final fuel)	315	315	315	315	315	315	315	315	315	315	315	315	377.4	0	0	203- 188
	Coal (kg/L final fuel)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Yield (kg bio-oil/L final fuel)	2.14	2.14	2.14	2.14	2.14	2.14	2.14	2.14	2.14	2.14	2.14	2.14	1.75	2.14	2.14	2.14

Table 28: Fuel Characteristics and Properties used in Analysis [152] [151] [146]

<b>Wood Residue</b>	Moisture content	50% (dry basis)
	Source	B.C. Sawmill sawdust (wet) and shavings (dry)
	Species	Hybrid Poplar (assumption for bio-oil upgrading)
	HHV at 10% MC (MJ/kg)	20
	Carbon Content	51%
	Oxygen Content	42%
	Other	Hydrogen (6%); Sulfur (0.09%)
<b>Pellets</b>	Moisture content	5.9% (dry basis)
	HHV	19.4
<b>Bio-oil</b>	HHV (MJ/L)	21
	Density (g/L)	1200
	Oxygen content	37.30%
	Hydrogen content	6.20%
	Carbon Content	56.40%
	Water Content	15-30%
	<b>Refined Bio-oil</b>	HHV(MJ/kg)
HHV (MJ/L)		35.57
Density (g/L)		772
Jet fraction (mass)		13.75%
Diesel fraction (mass)		41.25%
Gasoline fraction (mass)		45%
<b>Diesel</b>	HHV(MJ/kg)	45.8
	HHV (MJ/L)	38.65
	Density (g/L)	843.2
<b>Gasoline</b>	HHV(MJ/kg)	46.9
	HHV (MJ/L)	34.69
	Density (g/L)	739.2
<b>Jet A-1</b>	HHV(MJ/kg)	46.3
	HHV (MJ/L)	37.4
	Density (g/L)	808
<b>LPG</b>	HHV (MJ/L)	25.46
	Density (g/L)	506.5
<b>Natural Gas</b>	HHV (MJ/L)	0.0375
	Density (g/L)	0.721
<b>Hydrogen</b>	HHV (MJ/L)	0.0128
	Density (g/L)	0.089
<b>Electricity Mix</b>	B.C.	Hydro (84%)
		Biomass (7%)
		Nat Gas Boiler (6%)
		Wind (3%)

## Appendix V: Lifecycle Sensitivity Analysis Results

Table 29: Displacement Results (Scenario 3)

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		215		215
Fuel production		22,398		22,398
Feedstock transmission	1,300	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		-6794 to -8166		-6794 to -8166
Emissions from fuel use			195	195
<b>Total</b>				<b>20,906 gCO<sub>2</sub>eq/GJ to 22,278 gCO<sub>2</sub>eq/GJ</b>

Table 30: Mass Allocation (Scenario 3)

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		215		215
Fuel production		22,398		22,398
Feedstock transmission	1,300	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0

Gas leaks and flares	0	0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0	0
Emissions displaced	-253 to -303	-253 to -303
Emissions from fuel use		195
<b>Total</b>		<b>28,769 gCO<sub>2</sub>eq/GJ to 28,819 gCO<sub>2</sub>eq/GJ</b>

Table 31: Integrated pyrolysis facility, Scenario 1 (A S1) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.
Fuel	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue	Forest Residue	Forest Residue
Fuel dispensing	38		38
Fuel distribution and storage	223		223
Fuel production	22,379		22,379
Feedstock transmission	3,060		3,060
Feedstock recovery	653		653
Feedstock upgrading	0		0
Land-use changes, cultivation	0		0
Fertilizer manufacture	0		0
Gas leaks and flares	0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0		0
Emissions displaced	0		0
Emissions from fuel use		195	195
<b>Total</b>			<b>26,548</b>

Table 32: Integrated pyrolysis facility, Scenario 2 (A S2) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.
Fuel	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue	Forest Residue	Forest Residue
Fuel dispensing	39		39
Fuel distribution and storage	223		238

Fuel production	22,383		22,383
Feedstock transmission	3,539		3,539
Feedstock recovery	653		653
Feedstock upgrading	0		0
Land-use changes, cultivation	0		0
Fertilizer manufacture	0		0
Gas leaks and flares	0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0		0
Emissions displaced	0		0
Emissions from fuel use		195	195
<b>Total</b>			<b>27,047</b>

Table 33: Pellets from 100% forest residue, Scenario 3 (Bi) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel Feedstock	Pellets Sawmill residue	Refined bio-oil Pellets	Refined bio-oil Pellets	Refined bio-oil Pellets
Fuel dispensing		41		41
Fuel distribution and storage		215		215
Fuel production		22,444		22,444
Feedstock transmission	6,822	0		6,822
Feedstock recovery	5,596	0		5,596
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		-1224 to -1425		-1224 to -1425
Emissions from fuel use			195	195
<b>Total</b>				<b>34,090</b>
			to	<b>33,889</b>

Table 34: Pellets from 50/50 forest and mill residue, Scenario 3 (Bii) in gCO2eq/MJ

Source of Emissions Region	Pellet production B.C.	Pyrolysis, Upgrading and Distribution		Fuel Use B.C.	Total B.C.
Fuel	Pellets	Refined bio-oil		Refined bio-oil	Refined bio- oil
Feedstock	Sawmill residue	Pellets		Pellets	Pellets
Fuel dispensing			40		40
Fuel distribution and storage			215		215
Fuel production			22,419		22,419
Feedstock transmission	3,809		0		3,809
Feedstock recovery	5,260		0		5,260
Feedstock upgrading			0		0
Land-use changes, cultivation			0		0
Fertilizer manufacture			0		0
Gas leaks and flares			0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG			0		0
Emissions displaced			-934 to -1086		-934 to - 1086
Emissions from fuel use				195	195
<b>Total</b>					<b>31,004</b>

to **30,852**

Table 35: Pellets sourced from forest Residue, mill residue, and standing timber according to DRAX, Scenario 3 (Biii) in gCO2eq/MJ

Source of Emissions Region	Pellet production B.C.	Pyrolysis, Upgrading and Distribution		Fuel Use B.C.	Total B.C.
Fuel	Pellets	Refined bio-oil		Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets		Pellets	Pellets
Fuel dispensing			40		40
Fuel distribution and storage			215		215
Fuel production			22,408		22,408
Feedstock transmission	2,346		0		2,346
Feedstock recovery	5,281		0		5,281
Feedstock upgrading			0		0
Land-use changes, cultivation			0		0
Fertilizer manufacture			0		0
Gas leaks and flares			0		0

CO <sub>2</sub> , H <sub>2</sub> S removed from NG Emissions displaced	0	0
Emissions from fuel use	-808 to -955	-808 to -955
	195	195
<b>Total</b>		<b>29,677</b>
	to	<b>29,530</b>

Table 36: Moisture content of forest residue is 20% (lower bound, Scenario 1 – Ci) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution		Fuel Use	Total
	B.C.		B.C.	B.C.
Region	Refined bio-oil		Refined bio-oil	Refined bio-oil
Fuel	Forest Residue		Forest Residue	Forest Residue
Feedstock	Forest Residue		Forest Residue	Forest Residue
Fuel dispensing	38			38
Fuel distribution and storage	223			223
Fuel production	22,367			22,367
Feedstock transmission	1,911			1,911
Feedstock recovery	653			653
Feedstock upgrading	0			0
Land-use changes, cultivation	0			0
Fertilizer manufacture	0			0
Gas leaks and flares	0			0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG Emissions displaced	-294 to -252			-294 to -252
Emissions from fuel use				0
<b>Total</b>				<b>24,896</b>
			to	<b>24,938</b>

Table 37: Moisture content of forest residue is 70% (upper bound, Scenario 1 – Cii) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution		Fuel Use	Total
	B.C.		B.C.	B.C.
Region	Refined bio-oil		Refined bio-oil	Refined bio-oil
Fuel	Forest Residue		Forest Residue	Forest Residue
Feedstock	Forest Residue		Forest Residue	Forest Residue

Fuel Feedstock	Refined bio-oil Forest Residue	Refined bio-oil Forest Residue	Refined bio-oil Forest Residue
Fuel dispensing	38		38
Fuel distribution and storage	223		223
Fuel production	24744 to 25046		24744 to 25046
Feedstock transmission	2,518		2,518
Feedstock recovery	653		653
Feedstock upgrading	0		0
Land-use changes, cultivation	0		0
Fertilizer manufacture	0		0
Gas leaks and flares	0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0		0
Emissions displaced	0		0
Emissions from fuel use		195	195
<b>Total</b>			<b>28,672</b>
		to	<b>28,370</b>

Table 38: 70.3% Bio-oil Yield, lower bound (Scenario 1 - Ciii)

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution		Fuel Use	Total
	B.C.		B.C.	B.C.
Fuel Feedstock	Refined bio-oil Forest Residue	Refined bio-oil Forest Residue	Refined bio-oil Forest Residue	Refined bio-oil Forest Residue
Fuel dispensing	38			38
Fuel distribution and storage	223			238
Fuel production	22,380			22,380
Feedstock transmission	2,336			2,336
Feedstock recovery	678			678
Feedstock upgrading	0			0
Land-use changes, cultivation	0			0
Fertilizer manufacture	0			0
Gas leaks and flares	0			0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0			0
Emissions displaced	-191			-191
Emissions from fuel use			195	195
<b>Total</b>				<b>25,673</b>

Table 39: 78% Bio-oil yield, upper bound (Scenario 1 - Civ) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution		Fuel Use	Total
	B.C.		B.C.	B.C.
Region				
Fuel	Refined bio-oil		Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue		Forest Residue	Forest Residue
Fuel dispensing	38			38
Fuel distribution and storage	223			238
Fuel production	25,007			25,007
Feedstock transmission	2,149			2,149
Feedstock recovery	611			611
Feedstock upgrading	0			0
Land-use changes, cultivation	0			0
Fertilizer manufacture	0			0
Gas leaks and flares	0			0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0			0
Emissions displaced	0			0
Emissions from fuel use			195	195
<b>Total</b>				<b>28,237</b>

Table 40: 44% Refined bio-oil yield, upper bound (Scenario 1 - Cv) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution		Fuel Use	Total
	B.C.		B.C.	B.C.
Region				
Fuel	Refined bio-oil		Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue		Forest Residue	Forest Residue
Fuel dispensing	38			38
Fuel distribution and storage	223			238
Fuel production	25,979			25,979
Feedstock transmission	1,852			1,852
Feedstock recovery	534			534
Feedstock upgrading	0			0
Land-use changes, cultivation	0			0
Fertilizer manufacture	0			0
Gas leaks and flares	0			0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0			0
Emissions displaced	0			0
Emissions from fuel use			195	195
<b>Total</b>				<b>28,836</b>

Table 41: Hydrogen from Electrolysis, B.C. (Scenario 1 - Di) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Residue procurement, Pyrolysis, Upgrading, Distribution	Fuel Use	Total
	B.C.	B.C.	B.C.
Region			
Fuel	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Forest Residue	Forest Residue	Forest Residue
Fuel dispensing	38		38
Fuel distribution and storage	223		238
Fuel production	5,700		5,700
Feedstock transmission	2,265		2,265
Feedstock recovery	653		653
Feedstock upgrading	0		0
Land-use changes, cultivation	0		0
Fertilizer manufacture	0		0
Gas leaks and flares	0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0		0
Emissions displaced	-504		-504
Emissions from fuel use		195	195
<b>Total</b>			<b>8,585</b>

Table 42: Hydrogen from Electrolysis, B.C. (Scenario 3 - Dii) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Pellet production B.C.	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
		B.C.	B.C.	B.C.
Region				
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		215		215
Fuel production		5,728		5,728
Feedstock transmission	1,300	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		-1498 to -1597		-1498 to -1597
Emissions from fuel use			195	0

<b>Total</b>	<b>10,709</b>
	<b>to 10,610</b>

Table 43: Integrated pyrolysis facility, biogas used in hydrogen production (Scenario 3 - Diii) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		215		215
Fuel production		15362 to 14420		15362 to 14420
Feedstock transmission	1,300	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		0		0
Emissions from fuel use			195	195
<b>Total</b>				<b>22,036</b>
			to	<b>21,094</b>

Table 44: Hydrogen from Electrolysis, B.C. using displacement (Scenario 3) in gCO<sub>2</sub>eq/MJ

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		234		215
Fuel production		10,146		5,724
Feedstock transmission	2,599	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0

Gas leaks and flares	0	0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG	0	0
Emissions displaced	-23470 to -22578	-16676 to -18048
Emissions from fuel use		195
<b>Total</b>		<b>-5,331</b>
		to <b>-4,439</b>

Table 45: Hydrogen from Electrolysis, AB (Scenario 3)

Source of Emissions	Pellet production	Pyrolysis, Upgrading and Distribution	Fuel Use	Total
Region	B.C.	B.C.	B.C.	B.C.
Fuel	Pellets	Refined bio-oil	Refined bio-oil	Refined bio-oil
Feedstock	Sawmill residue	Pellets	Pellets	Pellets
Fuel dispensing		39		39
Fuel distribution and storage		215		215
Fuel production		70,044		70,044
Feedstock transmission	1,300	0		1,300
Feedstock recovery	4,925	0		4,925
Feedstock upgrading		0		0
Land-use changes, cultivation		0		0
Fertilizer manufacture		0		0
Gas leaks and flares		0		0
CO <sub>2</sub> , H <sub>2</sub> S removed from NG		0		0
Emissions displaced		-1597 to -1498		-1597 to -1498
Emissions from fuel use			195	195
<b>Total</b>				<b>75,121</b>
				to <b>75,220</b>

## Appendix VI: Additional Information

### Hydrogen and Electric Flight

Using renewable electricity, battery-powered road vehicles can see upwards of 60% greenhouse gas reductions [155]. This has inspired research into the development of electric aircraft, which is being led by NASA's Sceptor and LEAPTech programs. The main challenge these groups have faced is weight. Batteries are heavy, so the amount required for long distance transport often make the plane too heavy to fly. For example, in order to fly a plane for 8 hours, 2800 kg of battery would be required. Only 100 kg of jet fuel would be needed to power the same flight [156]. Furthermore, batteries are energy-intensive to produce, accounting for up to 50% of a craft's lifecycle emissions [157]. For these reasons, NASA and Airbus are focused on developing recreational electrical plane designs [158] [159]. Mark Moore, an aerodynamicist at NASA, expects that while recreational electric planes are feasible in the short term, the weight limitations of electric aircraft will be a barrier for transport aircraft for years to come [160].

Due to the limitations of electrifying flight, researchers are investigating hydrogen fuel as an alternative. In 2004, NASA published results from the "Quiet Green Transport" study [161] which analyzed the potential of using hydrogen fuel cells in aviation to reduce emissions and noise. This study found that, like electrically powered aircraft, the largest issue was the size and weight of the fuel cell-based propulsion system which was 10 to 20 times heavier than a traditional turbine engine. The report concluded that "significant advances in fuel cell propulsion system technologies are necessary just to make a fuel cell powered transport aircraft possible" [161]. It is anticipated that fuel cells will be reasonably light within 25 to 30 years, but this must be combined with airframe weight reductions in order to make fuel cell propulsion feasible [161].

Other companies, such as Boeing, are looking at combusting hydrogen on board rather than using it to power a fuel cell. Hydrogen gas has a large volume, so it must be liquefied and stored cryogenically on-board to be practical. However, even with this size reduction, cryogenic hydrogen fuel still takes up over 4 times the space of current jet fuel due to the heavy insulated tanks and heat exchangers required to keep the gas in liquid state [162]. This limits hydrogen fuel from being used in current models and would require new aircraft and engine design to accommodate the fuel [163].