Development of Methodological Framework and its Application for Life Cycle Assessment of Renewable Fuels for Transportation

by

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Abstract

There is scientific consensus on climate change and the adverse impacts of anthropogenic greenhouse gas (GHG) emissions. In order to reduce GHG emissions, all-encompassing transitions in energy, transportation, buildings, industrial systems, and land are required. The extraction, production, and combustion of fossil fuels used in transportation are responsible for GHG emissions. Blending liquid biofuels such as biodiesel with fossil fuels has the potential to reduce life cycle GHG emissions. There has been an increase in the production and use of renewable fuels, including new biofuel technologies such as oxymethylene ether (OME). However, a thorough quantitative assessment of the environmental, economic, and social implications of emerging biofuel technologies is crucial to understand both the short- and long-term consequences of their wide deployment. This can be achieved through life cycle assessment (LCA).

In this research, an environmental LCA model was developed to assess the environmental characteristics of OME production and combustion in vehicles as a diesel additive from two different types of woody biomass, such as whole tree and forest residue. The forest residue pathway was found to produce a lower environmental burden than the whole tree. The upstream GHG emissions from the forest residue pathway (18g CO_2eq/MJ) were significantly lower than those of the whole tree pathway (27 g CO_2eq/MJ) because the forest residue pathway did not consider emission-intensive road construction operation. The environmental LCA was broadened from its solely environmental scope to include the economic and social components by developing a life cycle sustainability assessment (LCSA) framework combining the three

sustainability dimensions namely environmental, social, and economic aspects. A multicriteria decision model was integrated into the LCSA framework to compare and identify the most sustainable pathway. The forest residue pathway outranked the whole tree pathway based on eight different sustainability indicators, namely GHG emissions, soot emissions, water depletion, capital cost, operating cost, overall cost, employment potential, and employee wage and benefits. For OME-diesel blend scenarios, higher OME ratios were found less preferred than lower ones considering all three sustainability dimensions.

In this thesis, we proposed a framework to determine the coproduct credits and the net environmental impacts from using coproducts; this framework can be used for any biofuel coproduct. This thesis discussed the environmental significance of applying credits to bio-ethanol and biodiesel coproducts based on their potential applications as energy substitutes, animal feed, and fertilizer. The largest coproduct credits were found from using DDGs pellets for heating (as an alternative to coal firing) in the bioethanol pathway. The coproduct credits ranged from 13.43-67.14g CO_2eq/MJ of ethanol based on the percentage of use. In the canola to biodiesel pathway, the highest coproduct credit was earned when the crude glycerine obtained from the biodiesel processing replaced synthetic glycerine. The coproduct credits ranged from 17.13-3.95 g CO_2eq/MJ when crude glycerine was processed into synthetic glycerine depending on its percentage use.

Earlier proposed LCA methodologies and frameworks looked at the environmental, economic, and social consequences of emerging and existing biofuel developments from a short-term perspective; this approach fails to reflect the long-term sustainability impacts of an energy

system. To assess credibility, market competitiveness, and financial and technological feasibility and to gain public and investor trust, a thorough quantitative environmental assessment of biomass energy technology is needed. This can be done through a consequential life cycle assessment (C-LCA). In this thesis, a thorough literature review of the C-LCA approach has been conducted, focusing on both the methodological aspects and the application areas for biofuels, along with their strengths and limitations. Key research gaps were identified and recommendations for further investigation were made. The thesis provides scientific contributions to decision makers and researchers working on biofuel technologies.

Preface

This thesis is an original intellectual product of the author, Nafisa Mahbub. Some parts of this work are published as follows:

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Nafisa Mahbub was responsible for defining the problem, developing the model, data interpretation, and manuscript preparation. Dr. Oyedun provided intellectual guidance and support with the manuscript composition. Dr. Oestreich shared experimental data. Dr. Kumar, Dr. Arnold, and Dr. Sauer provided supervisory oversight, intellectual guidance and support with the manuscript composition.

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This thesis is dedicated

to my parents Md. Shamsul Alam, Mrs. Zerina Begum,

my husband SM Hassan Shahrukh,

and my sister Nazia Mahbub

for their continuous support, encouragement, and faith

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ARP	Acid rain precursors
CNG	Compressed natural gas
CO	Carbon monoxide
CO_2eq	Carbon dioxide equivalent
FU	Functional unit
GHG	Greenhouse gas
GOP	Ground-level ozone precursors
HC	Hydrocarbon
LCA	Life cycle assessment
LNG	Liquefied natural gas
LPG	Liquefied petroleum gas
MJ	Mega joule
NO_x	Nitrogen oxide
OME	Oxymethylene ether
PM	Particulate matter

VOC	Volatile organic compounds
WTW	Well to wheel
LCSA	Life Cycle Sustainability Assessment
CLCA	Consequential Life Cycle Assessment
GAIA	Geometrical Analysis for Interactive Assistance
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization

Chapter 1: Introduction

1.1 Background

With the rapidly growing global economy and population, the demand for fossil-based energy is increasing. Combustion of fossil fuels results in emissions of greenhouse gases (GHG) which has impacts global warming. It is projected to increase further in the coming decades, posing a threat to the world's climate (Abas et al. 2015). Petroleum, coal, and natural gas together accounted for 86% of global energy supply, while the contribution from renewable and nuclear energy sources was 10% and 4.5%, respectively, in 2016. In 2017, global GHG emissions were reported to be around 53.5 $GtCO_2$-eq (UNEP 2018). Energy production and use were responsible for 78% of the emissions (NRCan 2019).

Canada, the world's 9th largest emitter, contributed around 1.6% of the global GHG emissions in 2017, around 716 $MtCO_2$-eq, an 8 $MtCO_2$-eq increase from 2016(Environment and Climate Change Canada 2018, 2019a, 2019b) . The transportation sector and the oil and gas sector were the major GHG emitters and together contributed more than 50% of the emissions (Environment and Climate Change Canada 2019a). Emissions have increased by 43% and 84% in the respective sectors between 1990 and 2017 (Environment and Climate Change Canada 2019a). Alberta, the fourth most populous province in Canada, produced 273 $MtCO_2$-eq in 2017 (Environment and Climate Change Canada 2019a), around 38% of the country's total. A large share of these emissions is from mining, crude petroleum extraction, upgrading, refining, and its use in vehicles. In the last few decades, road transport patterns have changed considerably. Road transportation emissions are growing because of the increased distances driven by both light and heavy duty vehicles (Environment and Climate Change Canada 2019a), which consume large amounts of fuel. Passenger vehicles and freight trucks contributed around 93 Mt CO_2-eq and 71 Mt CO_2-eq emissions, respectively, in 2016 (NRCan 2019). All of these factors highlight the crucial role the transportation sector could play in reducing the adverse impacts of climate change. Several GHG mitigation options have been proposed to make the sector greener. Alternative fuel production from renewable sources, blending renewable fuels in conventional fossil fuels and the use of biofuel coproduct credits are among the climate actions the sector can take.

There is an increasing trend towards the production and use of alternative fuels from renewable sources to replace conventional fossil sources (Nguyen et al. 2013, Kajaste 2014). These are, for example, biomass, hydro, solar, and wind.

1.1.1 Biofuel

Biomass from agricultural and forest residue, manure, and energy crops are the best candidates for the production of biofuels (i.e., biodiesel, bioethanol, renewable diesel, and jet fuel) (Bhawana Pathak 2013, Longhurst 2016, Mahbub et al. 2017). Canada has an estimated 347 million hectares of forest lands, or around 9% of the world's total forest area (NRCan 2018), a huge potential resource for biofuel production. In Alberta, forest biomass is considered a great source of biofuel production (Kabir et al. 2012, Mahbub et al. 2017). In Canada, wheat and corn are the most commonly used agricultural feedstocks for biofuel production. Around four million tonnes of wheat are potentially available for bioethanol production annually (Saha 2010, Mahbub et al. 2019). Agricultural residues are also available in abundance after taking into account mandatory soil conservation and animal feeding and bedding. In Western Canada, around 27 million tonnes of agricultural residues are potentially available for biofuel production (i.e., ethanol and biodiesel) (Sokhansanj et al. 2006, Xue Lia 2012).

Ethanol and biodiesel are the most produced biofuels of all the renewable fuels used in the transportation sector globally. They accounted for 4% of the liquids used in the sector in 2013 (REN21 2013). Biofuels are blended with conventional fossil fuels for use in vehicles. A number of biofuel additives are described in the literature; they reduce combustion GHG emissions when they are mixed with fossil fuels. Diesohol (15% ethanol and low sulfur diesel), hydrated/azeotropic ethanol, petrohol (10% ethanol and unleaded petrol), E95 (95% ethanol and 5% petrol), E10 blend (10% ethanol and 90% gasoline), BD20 blend (20% biodiesel and 80% diesel), and BD35 (35% biodiesel and 65% diesel) are some examples (Beer et al. 2002, Beer et al. 2003, Niven 2005, Beer and Grant 2007).

In addition to GHG emissions, soot or carbon black emissions from fossil fuel combustion in vehicles also significantly contribute to global warming, next to CO_2 emissions (Bond et al. 2013). To make conventional fuel combustion clean from carbon black or soot, oxygenated compounds such as methanol, dimethyl ether (DME) or dimethoxymethane (DMM) are added (Zhang et al. 2014, Zhang et al. 2016). An emerging oxygenated fuel, oxymethylene ether (OME), has the potential to be used as an alternative transportation fuel or as a diesel additive (Pellegrini et al. 2013, Mahbub et al. 2017). The distillation range of diesel and OME can be matched, providing smooth miscibility due to the adjustable composition of the oligomer molecules in OME (Pellegrini et al. 2013), and this could mean no additional cost as no design modification is required (Pellegrini et al. 2013, Zhang et al. 2016). Moreover, OME can be used in vehicles without requiring any diesel particulate filter (Pellegrini et al. 2013). OME can be produced from both fossil and renewable sources like biomass (Pellegrini et al. 2013, Mahbub et al. 2017). OME, as a diesel additive, has the potential to reduce the soot emissions from combustion significantly (Pellegrini et al. 2012, Pellegrini et al. 2013, Pellegrini et al. 2014, Zhang et al. 2014, Zhang et al. 2016). Pellegrini et al. showed that 10% OME blended in diesel can reduce the particulate matter (or soot) emissions from vehicles by up to 20% without any alteration of the engine (Pellegrini et al. 2012, Pellegrini et al. 2013); this has good potential to mitigate GHG emissions. Currently, OME is not produced commercially. Hence, it is important to quantitatively assess the potential environmental benefits from using this biofuel additive in vehicles and determine the sustainability, market competition, technical feasibility, social response, and financial viability before its large-scale application.

1.1.2 Biofuel Coproducts and their Potential Applications

The environmental and economic benefits of biofuel technologies are not limited to the main fuels but include the use of their coproducts (Zhang 2017, Mahbub et al. 2019). Biofuel coproducts have great potential to be used in a range of applications replacing fossil-based products, further reducing GHG emissions. Over the last few years, the production of ethanol and biodiesel has increased significantly (Sawin et al. 2017) and is expected to increase further in the coming decades through the deployment of local, regional, and global renewable energy and biofuel policies to meet energy demand. With increased biofuel production, large amounts of coproducts are simultaneously produced. Dry distiller grains (DDGs) are usually coproduced during ethanol processing from the dry milling of the wheat feedstock (Saha 2010, Moore 2011).

The annual production of ethanol in Canada is around 1.46 billion litres, which results in 1.16 million tonnes of DDGs (Saha 2010). Similarly, biodiesel from canola or soybean seeds unavoidably coproduces seed meal, crude glycerine, propylene glycol, etc., which can potentially replace other products in the market (Inc.(S & T)^2 Consultants 2013). Animal feed, energy source, and soil fertilizer are the three key uses of biofuel coproducts. DDGs, soybean meal, canola meal, etc., contain a high percentage of crude protein, which is a fundamental element in animal feed (Lory et al. 2008, Moore 2011, Inc.(S & T)^2 Consultants 2013).

Biofuel coproducts also have the potential to be used as an energy source. Several studies note that DDG's fuel characteristics are comparable with conventional fuels including bio-crude oil (Mansur et al. 2018) and coal used for space heating (Saha 2010, Eriksson et al. 2012).

Biofuel coproducts have a lower heating value than coal; hence, they can reduce combustion emissions drastically when cofired with coal as densified pellets like wood (Demirbas 2004, Saha 2010, Kabir et al. 2012, Mahbub et al. 2019). Furthermore, biofuel coproducts are rich in plant nutrients like calcium, magnesium, nitrogen, phosphorous, potassium, boron, iron, etc., and have a comparatively low cost (Moore 2011, Shroyer et al. 2011), which makes them a great source of organic fertilizer.

1.1.3 Renewable Fuel Initiatives: Global and Canadian Perspectives

A number of policies, mandates, targets, and regulations have been proposed and put into action by local, regional, and worldwide jurisdictions to support the commercialization and large-scale application of various GHG mitigation technologies, including biofuel production, coproduct applications, the blending of biofuels with fossil petroleum fuel, etc. The global concern and agreement to reduce anthropogenic GHG emissions has led to the establishment of regulations and policies such as the Intergovernmental Panel on Climate Change (IPCC) and the United Nations Framework Convention on Climate Change (UNFCCC) (United Nations Climate Change 2018, 2019a). The Kyoto Protocol and the Paris Agreement are examples of global climate change global initiatives corroborated by most of the country members of the United Nations (United Nations Climate Change 2018, 2019b). The Paris Agreement aims to limit the global average temperature rise to well below 2°C above pre-industrial levels (United Nations Climate Change 2018). Studies have shown that severe climate change actions are needed to achieve this target (Fuss et al. 2014, Millar et al. 2017). Canada has also undertaken several climate change mitigation strategies under the Pan-Canadian Framework on Clean Growth and Climate Change. As part of the framework's approach to carbon emission pricing, the Clean Fuel Standard encourages the use of low carbon fuels to reduce annual GHG emissions by 30 million tonnes by 2030 (Environment and Climate Change Canada 2018). Alberta, a high-emitting Canadian province, has implemented different environmental mandates and standards to reduce the GHG emissions. Examples include Alberta's Carbon Competitiveness Incentive Regulation (CCIR), which mandates submitting reports and compliance quarterly along with the annual forecasting for the facilities emitting GHG emissions more than 100,000 tonnes per year per facility (Government of Alberta 2019b), and Alberta's Renewable Fuel Standard (RFS), which mandates blending renewable biofuels in conventional transportation fuels like gasoline and diesel (Government of Alberta 2019a). In Canada, the blending ratio requirement varies by province; however, a minimum of 2% biofuel by weight is mandatory in conventional petroleum fuels. Alberta's RFS mandates blending (by weight) 5% renewable ethanol in gasoline and 2%

biodiesel in diesel used in vehicles (Charlotte Helston 2012, Government of Alberta 2019b). Canada has been producing more biofuels with the aim of reducing its annual GHG emissions by four million tonnes (Government of Alberta 2019b). A thorough quantitative, life cycle assessment of these climate change mitigation targets and environmental policies is necessary to determine the long-term environmental benefits, sustainable merits, future potential, market acceptance, and feasibility of large-scale application (Mahbub et al. 2018).

1.1.4 Life Cycle Assessment

Life cycle assessment (LCA) is an impact assessment tool used to determine the environmental performance of a product or product system throughout its life cycle. LCAs are usually used to obtain a thorough and holistic quantitative assessment of different sustainability aspects and to help with decision making (Ciroth et al. 2011). An environmental LCA evaluates environmental performance, while life cycle costing (LCC) and social LCAs address the economic viability and social relevance of a product, respectively. Life cycle sustainability assessment (LCSA) integrates environmental LCA, LCC, and social LCA, and has been refereed as a triple bottom line model (Kloepffer 2008, Ciroth et al. 2011), (Klöpffer and Renner 2007). LCSA is also known as a sustainability framework rather than a single model that combines different models and answers a particular sustainability question (Guinee et al. 2010, Guinee 2016). LCA, as defined by the International Organization for Standardization (ISO), is a tool made up of principles and framework (ISO 14040:2006) and requirements and guidelines (14044:2006). According to the ISO, LCA has four phases: goal and scope definitions, inventory analysis, life cycle impact assessment, and interpretation (ISO International Organization for Standardization 2006-07a, 2006-07b). In the goal and scope phase, the objective and the system boundary of the

LCA study is decided. Inventory analysis includes the collection and finalizing of data and assumptions to conduct the impact assessment. The impact assessment phase translates the life cycle inventory results to certain environmental impact categories (e.g., global warming potential, water footprint, land use change). The results are interpreted in the final stage.

Allocation is one of the crucial aspects of LCA of biofuel energy systems as the biofuel production processes are multifunctional by nature (they deliver more than one product) (Andrew and Forgie 2008). The environmental burden between the main product and its coproducts can be attributed through partitioning, economic and mass allocation, and system expansion. Which to use depends on the product being analyzed. ISO 14040/44 set the allocation procedures (ISO International Organization for Standardization 2006-07a, 2006-07b). According to ISO 14044, whenever possible, allocation through partitioning the unit processes or by expanding the system to include the additional functions similar to the coproduct should be avoided (ISO International Organization for Standardization 2006-07b). Mass or economic allocations can be implemented when allocation cannot be avoided.

There are two modelling choices depending on the questions to be addressed, namely attributional life cycle assessment (A-LCA) and consequential life cycle assessment (C-LCA) (Thomassen et al. 2008). A-LCA attributes inputs and outputs to a defined functional unit by linking each unit process through normative allocation. Further details on A-LCA are included in Chapter 2 of the thesis. C-LCA, on the other hand, evaluates the potential change in a product system following a change in demand in the functional unit. A-LCA uses market average data, while C-LCA uses marginal data and the system expansion approach to assess the long-term environmental consequences of changes in the product system (Reinhard and Zah 2009, Vázquez-Rowe et al. 2014). The role of C-LCA as a decision-making tool has been in use for a decade and is gradually becoming popular because of its ability to capture the long-term consequences of any large-scale policy implementation or technological change. C-LCA is particularly useful in the evaluation of possible environmental, social, economic, or technical consequences prior to the implementation of a new technology or policy, for example, regional or global GHG mitigation technologies or renewable policies, thus assisting in decision making (Sanchez et al. 2012, Earles et al. 2013, Vázquez-Rowe et al. 2014). Further details on C-LCA are included in Chapter 5.

1.1.5 Literature Gaps

This section highlights the key literature gaps the thesis attempts to address. Using biofuels as additives in transportation fuels can reduce GHG emissions significantly. OME, an emerging biofuel, is discussed in several studies (Pellegrini et al. 2012, Pellegrini et al. 2013, Pellegrini et al. 2014, Zhang et al. 2014, Zhang et al. 2016). However, these studies focus mostly on process modelling, key process parameters, and combustion characteristics (Pellegrini et al. 2013, Pellegrini et al. 2014, Zhang et al. 2014, Zhang et al. 2016). None of the studies discusses the life cycle energy and GHG emission impacts of OME production from biomass. This study aims to address the gap through an LCA in order to provide the environmental characteristics of OME production and combustion in vehicles as a diesel additive from two types of woody biomass. Performing an LCA of emerging energy technology can help identify hotspots and provide important insights (into design, for instance) so that changes can be made at a relatively lower cost, before a technology is fully operational, and thus avoid environmental impacts.

Applying LCA on bioenergy systems gives insights into environmental performance, which can be used to support policies. However, it is important to broaden LCA from a solely environmental or ecological assessment tool to include economic and social components; this can be done under the LCSA framework. There have been advancements in the development and application of LCSA frameworks; however, LCSA studies on bioenergy/ biofuels mostly focus on particular sustainability aspects, i.e., environmental, economic, technical, and social (Afgan and da Graça Carvalho 2000, Keesom et al. 2009, Budsberg et al. 2012, Thakur et al. 2014), or a combination of few aspects (Valente et al. 2011, Zhang and Haapala 2014). Furthermore, sustainability studies on bioenergy and biofuels have analyzed and compared different alternatives to different multidimensional aspects, i.e., environmental, social, economic, and technical (Luk et al. 2010, Sultana and Kumar 2012). However, none of these studies addressed the sustainability through a life cycle approach.

The literature on LCSA methods and applications is inconsistent and indefinite, which hinders LCSA's proper application in practical situations that combine all the aspects of sustainability. Moreover, the limited number of case studies on the application of an LCSA framework in biofuel energy systems makes the situation even more challenging (Ciroth et al. 2011, Guinee 2016). In addition, a comparative analysis of different biofuel pathways that considers all the dimensions of sustainability following a life cycle approach is not commonly done. This study, therefore, addresses these gaps by developing an LCSA framework that allows us to compare production of OME from two different biomass feedstocks namely whole tree and forest residue. A multicriteria decision model was integrated into the framework to identify the most sustainable pathway. Further details on LCSA are included in Chapter 3 of this thesis.

Another literature gap this thesis aims to address is the methodological approach to determine coproduct credits. Several studies have determined biofuel coproduct credits and assessed the impacts on total biofuel life cycle emissions (Henderson 2000, Bremer et al. 2010, Falano et al. 2014). However, to the best of the authors' knowledge, none of the studies proposed a standard framework on how to assign credits to biofuel coproducts. There is very limited standard procedure or framework on coproduct credit calculation that can be used as a reference for different biofuel coproducts. Moreover, most of the studies focus on a single use of biofuel coproducts, mainly as an alternative to animal feed (Bremer et al. 2010, Lardy and Anderson 2014). Biofuel coproducts have different potential applications, and it is important to evaluate how they can affect the environmental profile of the biofuel. A framework has been proposed to determine the coproduct credits and the net environmental impacts from using coproducts; the framework can be used as a reference for any biofuel coproduct. Further details on coproducts assessment is detailed in Chapter 4 of this thesis.

Most LCSA studies apply the A-LCA modelling approach and assess the short-term implications of coproduct uses (Afgan and da Graça Carvalho 2000, Afgan and Darwish 2011), which fails to reflect the long-term sustainability impacts of the energy systems. C–LCA, on the other hand, evaluates the possible long-term future consequences of any sustainable energy technology or policy implementation (Kloverpris et al. 2008, Sanchez et al. 2012, Earles et al. 2013). Here, the main challenge is the lack of a standardized framework to conduct a C-LCA, which could result in system delimitation, difficulties identifying the realistic marginal technology and competing products, lack of modelling tools, etc. (Mathiesen et al. 2009, Dandres et al. 2011). The C-LCA studies available in the literature explain different approaches to these issues, adding even more uncertainty to the model. Hence, this study has conducted a thorough literature review on the current state of the work on C-LCA of biofuels and considered both the methods and the application areas. The review highlights the existing C-LCA studies on biofuels and their strengths and limitations, identifies current research gaps, and makes recommendations for further investigation.

1.2 Objectives of the Research

The focus of this research is the LCA of renewable transportation fuels produced from woody biomass feedstock. The overall objective of the thesis is to propose methodological frameworks to assess the sustainability of different GHG mitigation technologies and discuss the long-term environmental consequences of the application of different climate change mitigation technologies and renewable policies. The specific objectives of this research are to:

1. Develop an A-LCA model to assess energy and environmental impacts, namely those of GHG emissions, acid rain precursors (ARPs), and ground-level ozone precursors for the whole chain of a newly emerging biofuel, oxymethylene ether (OME) production, and its use in vehicles as a diesel additive.
2. Propose an LCSA framework to evaluate the sustainability performances of liquid biofuels.
3. Demonstrate the developed LCSA framework by conducting a case study on OME production technology for two types of woody biomass, whole tree and forest residue.
4. Conduct a sensitivity analysis to study the impact of input parameters on the overall life cycle sustainability impacts.

5. Identify potential alternate applications of coproducts from two biofuel pathways, DDGs from the wheat to ethanol pathway and canola meal and crude glycerine from the canola to biodiesel pathway.
6. Investigate the environmental benefits from the use of DDGs, canola meal, and crude glycerine as alternative animal feed, fuel, and fertilizer.
7. Review the application of C-LCA for biofuels, identify the research gaps in methods and applications, and recommend areas for further investigation.
8. Investigate how the C-LCA methodologies can help determine the long-term consequences of different GHG mitigation technologies and renewable policies.

1.3 Scope, Challenges, and Limitation

1.3.1 Scope

This research conducts different types of sustainability assessments of renewable transportation fuels following a life cycle approach. The LCA system boundary includes all the unit operations in the biofuel supply chain from raw material extraction to transportation, plant conversion, combustion of fuel, and withdrawal/disposal. The life cycle energy and emission impacts from the material, fuel, and equipment used are also considered in the calculation. The life cycle sustainability indicators are selected based on the comprehensive review of published sustainability assessments and discussions with subject matter experts. The commonly available biofuel coproducts such as DDGs (dry distiller grains) from the wheat to ethanol pathway and canola meal and crude glycerine from the canola to biodiesel pathway are considered for investigation in the study. The displacement values that are calculated to determine the environmental benefits from the alternate coproduct applications are based on energy content.

1.3.2 Challenges

The challenges faced during the study include:

- The lack of standard guidelines and methods for the selection and evaluation of different sustainability criteria to assess the biofuel pathways.
- The lack of standard sensitivity and uncertainty assessment methods.
- The defining of different sustainability impact scenarios and finalizing the values for different input variables such as sustainability indicator weights and characteristic functions (i.e., preference function) type to develop the scenarios in the LCSA study.
- Limited work on multiple biofuel coproduct use and its impact on the overall biofuel life cycle emissions.
- The limited number of case studies on the LCSA and CLCA of different biofuel technologies and renewable policies.

1.3.3 Limitations

The life cycle data inventory (i.e., biomass feedstock yield, harvest area, moisture content, tortuosity factor for biomass transportation distance, types of biofuel coproducts obtained, and so on) and assumptions used in this research study are mostly taken from the literature. Alberta-specific assumptions and current sustainable practices such as sustainable forest management practices are used to assess the life cycle energy and emissions from different biofuels. However, the developed LCA frameworks can be applied to other jurisdictions and other biofuel pathways.

This study does not include the rebound effects of the biofuel technology discussed given the lack of adequate knowledge and evidence in the literature.

1.4 Organization of the Thesis

This thesis organized into six chapters: Chapter 1 is an introductory section that discusses the background, objectives, and thesis organization. Chapters 2 to 5 are a consolidation of individual papers and each is intended to be read independently. As a result, some concepts and data may be repeated. Chapter 6 is the conclusion section in which the key outcomes of the thesis are given, and the recommendations of future research are highlighted. A brief summary of each chapter is presented below.

- Chapter 1 introduces the research, that is, the fuels considered (renewable transportation fuels, biofuel as petroleum fuel additives, biofuel coproducts) and their potential applications, renewable fuel initiatives from both global and local perspectives, different types of LCA methodologies or frameworks, followed by the overall objectives of the research and scope, limitations, and challenges faced during the research.
- Chapter 2 investigates the life cycle energy consumption and GHG emission performances of a newly emerging biofuel, oxymethylene ether (OME), produced from two different forest biomass feedstocks, whole tree and forest residue, and used as a diesel additive. OME can be blended with conventional fuels like diesel without the need for engine modification. The use of OME as a fuel additive could significantly minimize combustion-related GHG emissions. An intensive process-based LCA model was developed to characterize the energy and GHG profiles of forest biomass-based OME considering the full life cycle stages (harvesting, transportation, chemical conversion, and combustion). The life cycle emission estimates include GHG emissions, acid rain

precursors, and ground-level ozone precursors (GOP). The comparative assessment results show that the forest residue pathway generates fewer GHG emissions per energy output than the whole tree pathway, mainly because of the latter's emission-intensive operations. The chapter also analyzes the environmental performances of different OME-to-diesel blending alternatives.

- Chapter 3 proposes a life cycle sustainability assessment (LCSA) framework to evaluate the environmental, economic, and social impacts of biofuels from different energy pathways. The framework compares, ranks, and selects the most sustainable pathway using multicriteria decision analysis. The framework was demonstrated through a case study on the LCSA of OME produced from whole tree and forest residue over the life cycle stages from biomass harvesting to the combustion of OME as a diesel additive. The research uses PROMITHEE (Preference Ranking Organization Method for Enrichment and Evaluation) to rank and select the most sustainable pathway. The forest residue pathway appears to be more sustainable than the whole tree pathway. The 10% OME to diesel blend offers the best sustainability performance when OME is used as fuel additive.
- Chapter 4 develops a framework to evaluate the environmental benefits from different biofuel coproducts of agricultural residues. The framework also identifies the potential alternate applications of the coproducts from two biofuel pathways, DDGs from the wheat to ethanol pathway and canola meal and crude glycerine from the canola to biodiesel pathway. The overall life cycle environmental emissions savings from using the two biofuels' coproducts in diverse applications, namely animal feed, fuel, and fertilizer, were investigated.

- Chapter 5 reviews in detail the available literature on the C-LCA of biofuels. The chapter highlights the current state of the C-LCA of biofuels, explores the existing research areas including the methods and the applications, identifies research gaps, and recommends areas for further investigation.
- Chapter 6 summarizes the key research findings and provides a list of recommendations for further research.

References

Abas, N., A. Kalair and N. Khan (2015). "Review of fossil fuels and future energy technologies." Futures **69**: 31-49.

Afgan, N. H. and M. da Graça Carvalho (2000). Energy system assessment with sustainability indicators. Sustainable Assessment Method for Energy Systems, Springer**:** 83-125.

Afgan, N. H. and M. Darwish (2011). "Multi-criteria sustainability assessment of water desalination and energy systems—Kuwait case." Desalination and Water Treatment **25**(1-3): 241-250.

Andrew, R. and V. Forgie (2008). "Life Cycle Assessment of Canola Biodiesel in New Zealand." Landcare Research and Costing of Canola Biodiesel in New Zealand. Centre for Ecological Economics.

Beer, T. and T. Grant (2007). "Life-cycle analysis of emissions from fuel ethanol and blends in Australian heavy and light vehicles." Journal of Cleaner Production **15**(8): 833-837.

Beer, T., T. Grant, D. Williams and H. Watson (2002). "Fuel-cycle greenhouse gas emissions from alternative fuels in Australian heavy vehicles." Atmospheric Environment **36**(4): 753-763.

Beer, T., D. Olaru, M. Van der Schoot, T. Grant, B. Keating, S. Hatfield Dodds, C. Smith, M. Azzi, P. Potterton and D. Mitchell (2003). Appropriateness of a 350 Million Litre Biofuels

Target: Report to the Australian Government Department of Industry, Tourism and Resources, CSIRO, ABARE, BTRE, Canberra.

Bhawana Pathak, S. C. a. M. H. F. (2013). "Biomass - Resource for Sustainable Development." International Journal of Advancements in Research & Technology **2**(6): 271-289.

Bond, T. C., S. J. Doherty, D. Fahey, P. Forster, T. Berntsen, B. DeAngelo, M. Flanner, S. Ghan, B. Kärcher and D. Koch (2013). "Bounding the role of black carbon in the climate system: A scientific assessment." Journal of Geophysical Research: Atmospheres **118**(11): 5380-5552.

Bremer, V. R., A. J. Liska, T. J. Klopfenstein, G. E. Erickson, H. S. Yang, D. T. Walters and K. G. Cassman (2010). "Emissions savings in the corn-ethanol life cycle from feeding coproducts to livestock." Journal of Environmental Quality **39**(2): 472-482.

Budsberg, E., M. Rastogi, M. E. Puettmann, J. Caputo, S. Balogh, T. A. Volk, R. Gustafson and L. Johnson (2012). "Life-cycle assessment for the production of bioethanol from willow biomass crops via biochemical conversion." Forest Products Journal **62**(4): 305.

Charlotte Helston. (2012, May 2012). "Energy Profiles: Biofuels " Retrieved November 17 2016, 2016, from http://www.energybc.ca/biofuels.html

Ciroth, A., M. Finkbeier, J. Hildenbrand, W. Klöpffer, B. Mazijn, S. Prakash, G. Sonnemann, M. Traverso, C. M. L. Ugaya, S. Valdivia and G. Vickery-Niederman (2011). Towards a live cycle sustainability assessment: making informed choices on products, United Nations Environment Programma (UNEP)https://www.lifecycleinitiative.org/wp-content/uploads/2012/12/2011%20-%20Towards%20LCSA.pdf Accessed 02/05/2016.

Dandres, T., C. Gaudreault, P. Tirado-Seco and R. Samson (2011). "Assessing non-marginal variations with consequential LCA: application to European energy sector." Renewable and Sustainable Energy Reviews **15**(6): 3121-3132.

Demirbas, A. (2004). "Combustion characteristics of different biomass fuels." Progress in Energy and Combustion Science **30**(2): 219-230.

Earles, J. M., A. Halog, P. Ince and K. Skog (2013). "Integrated economic equilibrium and life cycle assessment modeling for policy-based consequential LCA." Journal of Industrial Ecology **17**(3): 375-384.

Environment and Climate Change Canada (2018). Clean Fuel Standard, Regulatory Design Paper, Environment and Climate Change Canada,https://www.canada.ca/content/dam/eccc/documents/pdf/climate-change/clean-fuel-standard-regulatory-design-paper-2018-en-1.pdf Accessed 10/01/2019.

Environment and Climate Change Canada (2019a). National Inventory Report 1990-2017: Greenhouse Gas Sources and Sinks in Canada Environment and Climate Change Canada (ECCC),https://www.canada.ca/en/environment-climatechange/services/climate-change/greenhouse-gas-emissions.html Accessed 05/05/2019.

Environment and Climate Change Canada (2019b). Canadian Environmental Sustainability Indicators : Greenhouse gas emissions, Environment and Climate Change Canada (ECCC),https://www.canada.ca/en/environment-climate-change/services/environmental-indicators/greenhouse-gas-emissions.html Accessed 01/05/2019.

Eriksson, G., A. Grimm, N. Skoglund, D. Boström and M. Öhman (2012). "Combustion and fuel characterisation of wheat distillers dried grain with solubles (DDGS) and possible combustion applications." Fuel **102**: 208-220.

Falano, T., H. K. Jeswani and A. Azapagic (2014). "Assessing the environmental sustainability of ethanol from integrated biorefineries." Biotechnology Journal **9**(6): 753-765.

Fuss, S., J. G. Canadell, G. P. Peters, M. Tavoni, R. M. Andrew, P. Ciais, R. B. Jackson, C. D. Jones, F. Kraxner and N. Nakicenovic (2014). "Betting on negative emissions." Nature Climate Change **4**(10): 850.

Government of Alberta. (2019a). "Renewable Fuels Standard." Retrieved 01/05/2019, from https://www.alberta.ca/renewable-fuels-standard.aspx?utm_source=redirector

Government of Alberta (2019b). Carbon Competitiveness Incentive Regulation. Alberta, Canadahttps://www.alberta.ca/carbon-competitiveness-incentive-regulation.aspx Accessed 20/06/2019.

Guinee, J. (2016). Life cycle sustainability assessment: What is it and what are its challenges? Taking Stock of Industrial Ecology, Springer**:** 45-68.

Guinee, J. B., R. Heijungs, G. Huppes, A. Zamagni, P. Masoni, R. Buonamici, T. Ekvall and T. Rydberg (2010). "Life cycle assessment: past, present, and future." Environmental Science and Technology **45**(1): 90-96.

Henderson, S. (2000). Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario. Agriculture and Agri-Food Canadahttps://ghgenius.ca/reports/AAFCCornEthanol.pdf Accessed 05/02/2018.

Inc.(S & T)^2 Consultants (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions, Alberta Innovates - Energy and Environment Solutions, https://books.google.ca/books/about/Life_Cycle_Assessment_of_Selected_Biofue.html?id=tT1pngEACAAJ&redir_esc=y.

ISO International Organization for Standardization (2006-07a). ISO 14040:2006 Environmental Management–Life Cycle Assessment–Principles and Framework https://www.iso.org/standard/37456.html Accessed 05/01/2018.

ISO International Organization for Standardization (2006-07b). ISO 14044:2006 Environmental management -- Life cycle assessment -- Requirements and guidelineshttps://www.iso.org/standard/38498.html Accessed 05-09-2018.

Kabir, M. R., B. Rooke, G. Dassanayake and B. A. Fleck (2012). "Comparative life cycle energy, emission, and economic analysis of 100 kW nameplate wind power generation." Renewable Energy **37**(1): 133-141.

Kajaste, R. (2014). "Chemicals from biomass–managing greenhouse gas emissions in biorefinery production chains–a review." Journal of Cleaner Production **75**: 1-10.

Keesom, W., S. Unnasch, J. Moretta and J. Consultancy (2009). Life cycle assessment comparison of North American and imported crudes, Alberta Energy Research Institute.

Kloepffer, W. (2008). "Life cycle sustainability assessment of products." The International Journal of Life Cycle Assessment **13**(2): 89.

Klöpffer, W. and I. Renner (2007). "Lebenszyklusbasierte Nachhaltigkeitsbewertung von Produkten." Technikfolgenabschätzung–Theorie und Praxis (TATuP) **16**(3): 32-38.

Kloverpris, J., H. Wenzel and P. H. Nielsen (2008). "Life cycle inventory modelling of land use induced by crop consumption-Part 1: Conceptual analysis and methodological proposal." International Journal of Life Cycle Assessment **13**(1): 13-21.

Lardy, G. P. and V. L. Anderson (2014). Feeding coproducts of the ethanol industry to beef cattle, NDSU Extension Service https://www.ag.ndsu.edu/publications/livestock/feeding-coproducts-of-the-ethanol-industry-to-beef-cattle/as1242.pdf Accessed 07/08/2015.

Longhurst, P. J. B.-I., Juliet; Manovic, Vasilije (2016). "Biomass resources and biofuels potential for the production of transportation fuels in Nigeria." Renewable and Sustainable Energy Reviews **63**: 172-192

Lory, J. A., R. E. Massey, C. D. Fulhage, M. C. Shannon, R. L. Belyea and J. M. Zulovich (2008). "Comparing the Feed, Fertilizer, and Fuel Value of Distiller's Grains." Crop Management **7**(1): 0-0.

Luk, J., H. Fernandes and A. Kumar (2010). "A conceptual framework for siting biorefineries in the Canadian Prairies." Biofuels, Bioproducts and Biorefining **4**(4): 408-422.

Mahbub, N., E. Gemechu, H. Zhang and A. Kumar (2019). "The life cycle greenhouse gas emission benefits from alternative uses of biofuel coproducts." Sustainable Energy Technologies and Assessments **34**: 173-186.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**: 1249-1262.

Mahbub, N., A. O. Oyedun, H. Zhang, A. Kumar and W.-R. Poganietz (2018). "A life cycle sustainability assessment (LCSA) of oxymethylene ether as a diesel additive produced from forest biomass." The International Journal of Life Cycle Assessment: 1-19.

Mansur, D., T. Tago and T. Masuda (2018). "Utilization of DDGS using ethanol solution for biocrude oil production by hydrothermal liquefaction." Biofuels **9**(3): 325-330.

Mathiesen, B. V., M. Münster and T. Fruergaard (2009). "Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments." Journal of Cleaner Production **17**(15): 1331-1338.

Millar, R. J., J. S. Fuglestvedt, P. Friedlingstein, J. Rogelj, M. J. Grubb, H. D. Matthews, R. B. Skeie, P. M. Forster, D. J. Frame and M. R. Allen (2017). "Emission budgets and pathways consistent with limiting warming to 1.5 C." Nature Geoscience **10**(10): 741.

Moore, A. (2011). Fertilizer potential of biofuel byproducts, INTECH Open Access Publisher.

Nguyen, T. L. T., J. E. Hermansen and R. G. Nielsen (2013). "Environmental assessment of gasification technology for biomass conversion to energy in comparison with other alternatives: the case of wheat straw." Journal of Cleaner Production **53**: 138-148.

Niven, R. K. (2005). "Ethanol in gasoline: environmental impacts and sustainability review article." Renewable and Sustainable Energy Reviews **9**(6): 535-555.

NRCan. (2018, 26/09/2018). "How much forest does Canada have? ." Retrieved 06/05/2019, from https://www.nrcan.gc.ca/forests/report/area/17601

NRCan. (2019, 27/05/2019). "Energy and Greenhouse Gas Emissions (GHGs)." Retrieved 20/05/2019, 2019, from https://www.nrcan.gc.ca/energy/facts/energy-ghgs/20063

Pellegrini, L., M. Marchionna, R. Patrini, C. Beatrice, N. Del Giacomo and C. Guido (2012). Combustion Behaviour and Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in a Light-Duty Diesel Engine. SAE 2012 World Congress & Exhibition United States, SAE International https://saemobilus.sae.org/content/2012-01-1053.

Pellegrini, L., M. Marchionna, R. Patrini and S. Florio (2013). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car. SAE 2013 World Congress & Exhibition, United States, SAE International https://saemobilus.sae.org/content/2013-01-1035 Accessed 05/05/2015.

Pellegrini, L., R. Patrini and M. Marchionna (2014). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car. SAE 2013 World Congress & Exhibition, United States, SAE International.

Reinhard, J. and R. Zah (2009). "Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment." Journal of Cleaner Production **17**: S46-S56.

REN21 (2013). Renewables 2017 Global Status Report, Renewable Energy Policy Network for the 21st century (REN21).

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan

Sanchez, S. T., J. Woods, M. Akhurst, M. Brander, M. O'Hare, T. P. Dawson, R. Edwards, A. J. Liska and R. Malpas (2012). "Accounting for indirect land-use change in the life cycle assessment of biofuel supply chains." Journal of The Royal Society Interface **9**(71): 1105-1119.

Sawin, J. L., F. Sverrisson, K. Seyboth, R. Adib, H. E. Murdock, C. Lins, I. Edwards, M. Hullin, L. H. Nguyen and S. S. Prillianto (2017). "Renewables 2017 Global Status Report."

Shroyer, K. J., S. A. Staggenborg and J. Propheter (2011). "Utilization of dry distillers grains and charcoal as nitrogen fertilizer in corn." Agronomy Journal **103**(5): 1321-1328.

Sokhansanj, S., S. Mani, M. Stumborg, R. Samson and J. Fenton (2006). "Production and distribution of cereal straw on the Canadian prairies." Canadian Biosystems Engineering **48**: 3.

Sultana, A. and A. Kumar (2012). "Ranking of biomass pellets by integration of economic, environmental and technical factors." Biomass and Bioenergy **39**: 344-355.

Thakur, A., C. E. Canter and A. Kumar (2014). "Life-cycle energy and emission analysis of power generation from forest biomass." Applied Energy **128**: 246-253.

Thomassen, M. A., R. Dalgaard, R. Heijungs and I. de Boer (2008). "Attributional and consequential LCA of milk production." The International Journal of Life Cycle Assessment **13**(4): 339-349.

UNEP (2018). The Emissions Gap Report Nairobihttp://wedocs.unep.org/bitstream/handle/20.500.11822/26895/EGR2018_FullReport_EN.pdf?sequence=1&isAllowed=y Accessed 05/05/2019.

United Nation Climate Change. (2019a). "About the Secretariat." Retrieved 01/05/2019, from https://unfccc.int/about-us/about-the-secretariat.

United Nations Climate Change. (2019b). "Kyoto Protocol - Targets for the first commitment period." Retrieved 05/05/2019, from https://unfccc.int/process/the-kyoto-protocol.

United Nations Climate Change. (2018). "The Paris Agreement." Retrieved 05/05/2018, from https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement.

Valente, C., R. Spinelli and B. G. Hillring (2011). "LCA of environmental and socio-economic impacts related to wood energy production in alpine conditions: Valle di Fiemme (Italy)." Journal of Cleaner Production **19**(17): 1931-1938.

Vázquez-Rowe, I., A. Marvuglia, S. Rege and E. Benetto (2014). "Applying consequential LCA to support energy policy: land use change effects of bioenergy production." Science of the Total Environment **472**: 78-89.

Xue Lia, E. M., Satya Panigrahib, Lope Tabilb, Shahab Sokhansanj, Mark Stumborg (2012). "A review of agricultural crop residu esupply in Canada for cellulosic ethanol production." Renewable and Sustainable Energy Reviews **16**: 2954–2965.

Zhang, H. and K. R. Haapala (2014). "Integrating sustainable manufacturing assessment into decision making for a production work cell." Journal of Cleaner Production **105**: 52-63.

Zhang, X., A. Kumar, U. Arnold and J. Sauer (2014). Biomass-derived oxymethylene ethers (OMEs) as diesel additives: A thermodynamic analysis. The 6th International Conference on Applied Energy, Taipei, Taiwan.

Zhang, X., A. Kumar, U. Arnold and J. Sauer (2014). "Biomass-derived Oxymethylene Ethers as Diesel Additives: A Thermodynamic Analysis." Energy Procedia **61**: 1921-1924.

Zhang, X., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2016). "An optimized process design for oxymethylene ether production from woody-biomass-derived syngas." Biomass and Bioenergy **90**: 7-14.

Zhang, Y. (2017). Life Cycle Environmental Impacts of Biofuels:The Role of Co-products. Doctor of Philosophy University of California.

Chapter 2: A Life Cycle Assessment of Oxymethylene Ether Synthesis from Biomass-derived Syngas as a Diesel Additive[1]

2.1 Introduction

Around 46 billion metric tons of CO_2eq gases or GHGs were emitted worldwide in 2010, of which an estimated 71% came from energy production and use alone, including fuel combustion in vehicles (EPA 2014). Canada emitted 702 million tonnes of CO_2eq gases in 2011, and most of these emissions came from Alberta, in particular from the petroleum industry (Environment Canada 2013). In 2014, Alberta generated 250 Mt of GHGs. If Alberta continues to generate GHGs at this rate, it will produce more than 300 Mt CO_2eq gases per year by 2050, which is alarming (Mahbub and Kumar 2014, Row and Mohareb 2014). In 2011, Alberta generated around 239 Mt CO_2eq gases, of which almost 40% came from mining and oil gas extraction and the subsequent use of oil and gas in production, refining, and vehicle combustion (Mahbub and Kumar 2014, Row and Mohareb 2014).

Vehicle exhaust in the form of CO_2, soot, NO_x, HC, and CO creates environmental pollution. Carbon black, or soot, is considered to be the second largest emission-contributing global warming material after carbon dioxide. It is responsible for producing around 1.1 W/m^2 of warming effect in the atmosphere (Bond et al. 2013). The combustion of fossil fuels in vehicles is one of the highest potential sources of soot or black carbon.

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Oxygenated compounds are added to conventional fossil fuels as additives to reduce soot formation (Pellegrini et al. 2012) and make combustion cleaner (Zhang et al. 2014, Zhang et al. 2016). Oxymethylene ether (OME) is an emerging fuel that can be used as an alternative transportation fuel or a fuel additive. The composition of oligomer molecules in OMEs can be adjusted to match the distillation range of diesel, thus providing great miscibility with diesel. In addition, OME can be used in old vehicles without altering the engine or using any diesel particulate filter (DPF) or any other expensive maintenance device and can be produced from a range of feedstocks including both fossil sources and biomass (Pellegrini et al. 2013). To combat the environmental issues arising from fossil fuel combustion and fossil resource depletion, there is a move towards the production and use of alternative fuels (Kajaste 2014, Nguyen et al. 2013). GHGs can be reduced considerably by replacing fossil sources with bio-based energy sources such as whole tree biomass, forest residue, agricultural residue, etc. (Agbor et al. 2016; Thakur et al. 2014). In Alberta, forests are harvested mainly for pulp and lumber. Since the demand for paper is decreasing, forest biomass can be a potential source of energy that can replace fossil sources (Government of Canada 2016, Kabir and Kumar 2012). Beer and Grant discussed GHG emissions reduction from the production and use of several biomass-derived alternative fuel blends such as diesohol (15% ethanol blended with low sulfur diesel and an emulsifier), hydrated ethanol (azeotropic ethanol), petrohol (E10, a blend of 10% ethanol and premium unleaded petrol), and E85 (a blend of 85% ethanol, ignition improver, and a denaturant)(Beer and Grant 2007). Pre-combustion and combustion emissions from conventional fuels (i.e., diesel) and several alternative fuels such as CNG, LNG, LPG, ethanol blended with 5% petrol (E95), E10 blend (10% ethanol by volume mixed with gasoline), pure biodiesel

(BD100), biodiesel blended with 80% diesel (BD20), and 65% diesel (BD35) have been discussed in the literature (Beer et al. 2002, Beer et al. 2003).

Among the oxygenated compounds, methanol, dimethyl ether (DME), dimethoxymethane (DMM), and OME are the most prominent diesel additives discussed in the literature (Zhang et al. 2014, Zhang et al. 2016). Studies have discussed different processes of fuel grade methanol production such as direct conversion of conventional fossil fuels including NG, biomass gasification, CO_2 hydrogenation etc. and analyzed different process parameters on methanol yield (Liu et al. 2016, Riaz et al. 2013). Methanol and dimethyl ether produced from renewable sources like hydrogen from water or wind electrolysis and captured carbon dioxide (Matzen and Demirel 2016, Van-Dal and Bouallou 2013) can reduce greenhouse gas emissions by 82-86% compared to conventional fossil fuels (Matzen and Demirel 2016). However, due to their chemical properties, DMM and DME require engine modification prior to their use as diesel additives in vehicles whereas OMEs can be used as diesel additives without any engine modification (Pellegrini et al. 2013, Zhang et al. 2016). Burger et al. discussed the formation of OMEs from DMM and trioxane (TRI) and also investigated the physical and chemical properties of OMEs used as diesel additives (Burger et al. 2010).

Zhang et al. developed a detail process model to produce OMEs from biomass and investigated some of the key parameters affecting the process such as equivalence ratio, H_2/CO ratio, and water flow rates. The authors found that a blend of 20% OME and 80% diesel can reduce soot emissions by 50% (Zhang et al. 2014). Pellegrini et al. investigated the performance of neat OME (100%) and blended OME (10% OME blended with 90% diesel) in reducing the combustion emissions from old vehicles (Pellegrini et al. 2013). Usually, a 10% blend of any oxygenated component with any conventional transportation fuel is considered the maximum to be used in old cars (Löfvenberg 2010, Pellegrini et al. 2013). Because of its lower heating value (LHV) compared to diesel fuel, 100% OME as a transportation fuel is not considered to be strong enough for highway driving conditions (Pellegrini et al. 2013). Pellegrini et al. further investigated the polyaromatic hydrocarbon (PAH) emissions and particle number size distribution (PNSD) in an old vehicle fueled with 7.5% OME blended in diesel (Pellegrini et al. 2014). Zhang et al. designed an optimal process model for the production of high OMEs (such as OME_3, OME_4, and OME_5) from woody biomass (Zhang et al. 2016). A number of studies have been conducted on process modelling of OME synthesis from methanol (Zhang et al. 2014, Zhang et al. 2016), and there are a few studies on combustion emission performance of OME in vehicles (Pellegrini et al. 2012, Pellegrini et al. 2013). But there is almost no published literature on LCAs or life cycle emission performances of the whole supply chain of OME production from biomass to be used as a diesel additive. This study focuses on the life cycle environmental impacts of the production and combustion of OME from two different types of woody biomass in the western Canadian province of Alberta.

OME as a fuel or fuel additive has not been discussed widely in the literature. Nor is there an LCA of OME, which is essential to determine the environmental impacts of the technology. Therefore, the main objective of this study is to conduct an LCA of energy and emission performance of OMEs from whole tree and forest residue biomass in Western Canada. The specific objectives are to:

- Develop a system boundary diagram showing the production and use of OME from biomass;
- Develop energy consumption estimates of various unit operations for the whole chain of OME production from biomass and the use of OMEs;
- Estimate the life cycle GHG emissions for the whole chain of OME production and use;
- Estimate the life cycle acid rain precursors (ARPs) and the ground-level ozone precursors (GOPs) for the upstream operations;
- Conduct a sensitivity analysis to study the impact of variations in input parameters on overall life cycle GHG emissions.

2.2 Method

The goal of this study is to develop a data-intensive spreadsheet-based LCA model for OME synthesis from woodchips derived from two different types of forest biomass, whole tree and forest residue, and calculate the GHG emissions and net-energy-ratio (NER). The net energy ratio is the ratio of total energy output from the system to the total non-renewable energy input to the system (Shahrukh et al. 2015, Spitzley and Keoleian 2004). Information from the literature and Alberta-specific assumptions (such as biomass yield, biomass harvest area, moisture content, and tortuosity factor for biomass transportation distance) and current practices were used to evaluate energy consumption and GHG emissions.

In this study, a life cycle assessment of OME synthesis from two different types of forest biomass was carried out. The system boundary was made up of the following six unit operations for both pathways: biomass production, biomass transportation, chemical conversion, fuel mixing, fuel dispensing, and vehicle combustion. The unit operations were further divided into subunit operations for both biomass feedstock pathways (see Figure 2.1). Due to the lack of data and relatively less significance on overall life cycle emissions, the downstream operations such as fuel dispensing, blending, storage, etc., were not included. The results are given using a functional unit (FU) of 1 MJ of heat produced from OME so that the LCA results can be compared with the results of other LCA studies independent of any factors like geographic, demographic, road situation, etc. Hence, although OME is a transportation fuel, functional units like person-km or ton-km were not used. Rather, an FU of 1 MJ of heat produced from OME was used so that it can be compared with any forms of transportation energy produced and used in any country around the world.

Three gases – carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) – are considered to cause global warming. Their relative impact on global warming is assumed to be 1, 34, and 298 times, respectively (Myhre et al. 2013). A 100-year time horizon is assumed for this impact. The acid rain precursors (ARPs) are sulfur dioxide (SO_2) and nitrogen oxide (NO_x), which are considered mainly responsible for acidification. The weighting factor for ARPs (SO_2eq) is considered to be 1 for SO_2 and 0.7 for NO_x. The ground-level ozone precursor (GOP) was also calculated in this study. GOPs include NO_x and volatile organic compounds (VOC). Both of these compounds have a weighting factor of 1 as GOPs. In the presence of sunlight, NO_x and VOC react with each other chemically and create ground-level ozone (Kabir and Kumar 2011, Perera and Sanford 2011).

2.3 Life Cycle Inventory

Both pathways studied here involve biomass production, biomass transportation, OME synthesis from biomass-derived syngas, transportation of OME, blending, and fuel combustion in a vehicle. As mentioned above, downstream operations like fuel mixing, storage, distribution, distillation, etc., are not considered in this study. This study considers a plant (gasifier) capacity of 277 t/d for OME synthesis. The GHG emissions are calculated over 20 years of plant life and the results are given in g CO_2eq/MJ of OME. The results are also compared to conventional diesel emission numbers.

2.3.1 Biomass Production

For whole tree harvesting in Alberta, trees are cut in the stand, skidded to the roadside and delimbed, and, eventually, the trunk is used by pulp and paper industries (Pulkki 1997, Shahrukh et al. 2016b). The biomass harvesting unit operation includes the subunit operations felling, skidding, and chipping (Figure 2.1). The energy and emission impacts from manufacturing, operations, and disposal of the equipment used (feller, skidder, chipper) are also considered. Silviculture operations, which include fertilizing and pesticide spraying, are not considered in the base model because in Alberta it is assumed that first-generation trees are harvested. However, a case described later in the sensitivity analysis (section 4.4) was developed that includes the energy and emission impacts from silviculture operations.

In Alberta, the rotation of whole tree growth is assumed to be 100 years; this time frame was determined based on weather and soil conditions. Whole forest yield includes both hardwood and softwood. It is assumed in this study that 84 dry tonnes of forest biomass are harvested per hectare and that 20% of whole tree biomass is forest residues (Alberta Energy 1985, Kumar et al. 2003). Thus, the yield of forest harvest residue is 0.247 dry tonnes per hectare over a 100-year rotation (Alberta Energy 1985, Kumar et al. 2003). The current trend in Alberta is to burn the residues to prevent forest fires (Shahrukh et al. 2016a). The removal of forest residues removes nutrients required for forest growth that would otherwise be returned to the soil. It is assumed in this study that ash (from the bio-plant) would be returned to the forest floor after the biomass is used for fuel production and thus forest soil nutrients are balanced (Thakur et al. 2014, Wihersaari 2005). The forest harvest residues can be considered a good source for bioenergy production since the nutrient system can be balanced (through ash replacement) and the residues

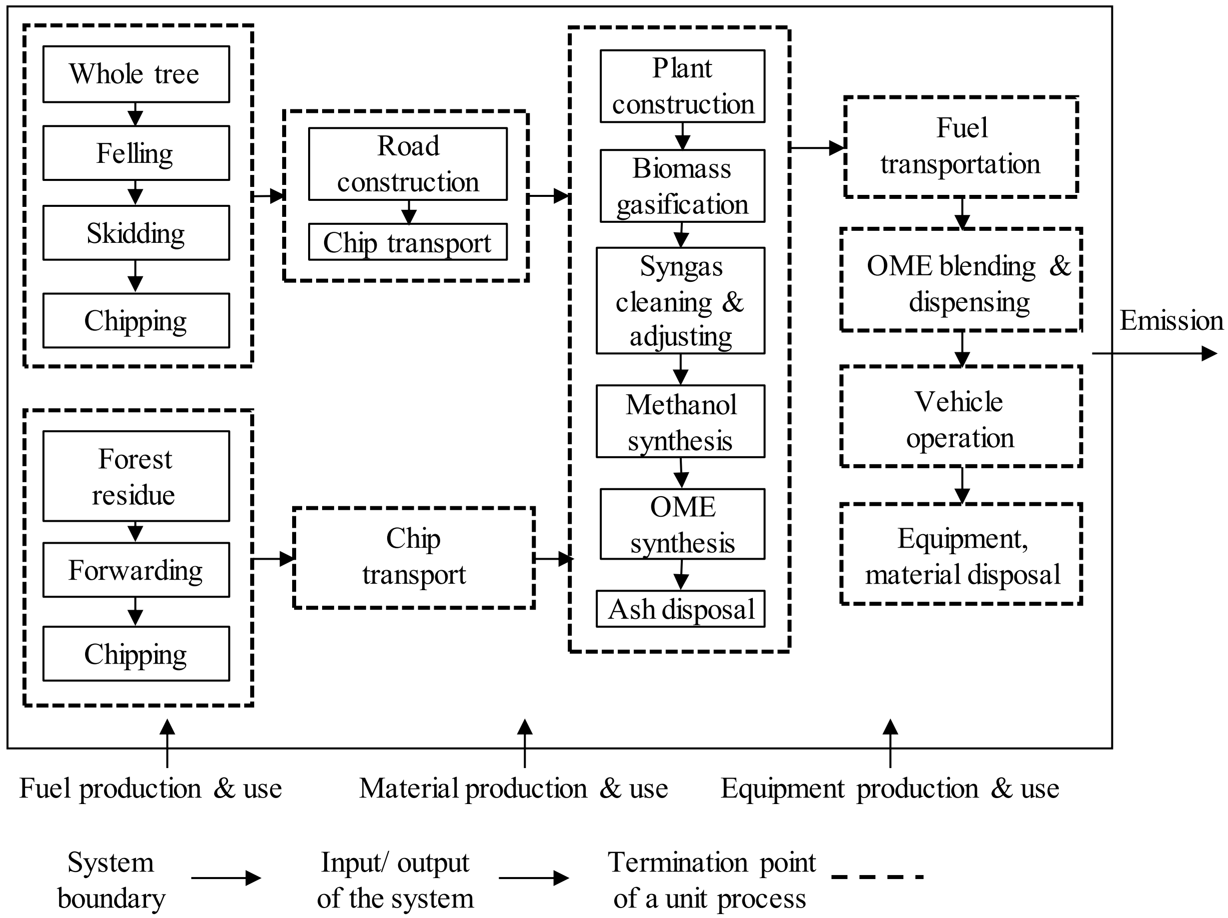


Figure 2. 1: System boundary for OME synthesis from whole tree and forest residue biomass

are otherwise considered waste (by their burning) (Thakur et al. 2014, Wihersaari, 2005). Table 2.1 shows the data and assumptions for biomass harvesting for both the whole tree and forest residue pathways.

Table 2. 1: Inventory data and assumptions for biomass harvesting, transportation, and chemical conversion

Assumptions/Properties	Units	Whole tree	Forest residue	Comments/ References
Biomass required over 20 years	t	776552	1009518	Dry basis. Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b)
Biomass production	t/ha	84	0.247	Dry basis (Kumar et al. 2003)
Higher heating value	GJ/t	20	20	Dry basis (Kumar et al. 2003)
Moisture content[a]	wt.%	50	45	(Kumar et al. 2003)
Annual biomass requirement	t/year	38828	50476	Dry basis. Calculated from (Kabir and Kumar 2011, Zhang et al. 2014, Shahrukh et al. 2016b)
Harvest area	ha	585	207158	Calculated from (Agbor et al. 2016)
Transportation distance	km	4.56	21.75	Calculated from (Agbor et al. 2016)
Ash content	wt.%	1	3	(Kumar et al. 2003)
Pesticide application	kg/ha	0.17	-	(Kabir and Kumar 2012)
Biomass flow to gasifier	t/d	277	277	Wet basis (Zhang et al.

Assumptions/Properties	Units	Whole tree	Forest residue	Comments/ References
				2014)
Plant life	years	20	20	(Kabir and Kumar 2011)
Capacity factor				
Year 1		0.7	0.7	(Shahrukh et al. 2016b)
Year 2		0.8	0.8	(Shahrukh et al. 2016b)
Year 3 & onwards		0.85	0.85	(Shahrukh et al. 2016b)
Volumetric truck capacity	m^3	70	70	(Mann and Spath 1997)
Lifetime of each truck	km	540715	540715	(Mann and Spath 2001)
Dedicated trucks required (WT)		1.56	7.82	Calculated from (Mann and Spath 2001, Zhang et al. 2014, Shahrukh et al. 2016b)
Bulk density of whole tree chip	kg/m^3	250	235	(Kabir and Kumar 2012)
Gross vehicle weight (GVW)	t	38	38	(Kabir and Kumar 2012)
Truck payload	t	23	23	(Kabir and Kumar 2012)
Truck fuel consumptions (empty/ full load)	L/km	0.24/0.33	0.24/0.33	(Sultana and Kumar, 2011)

Assumptions/Properties	Units	Whole tree	Forest residue	Comments/ References
Actual load carried by truck (WT)	t	17.5	16.5	(Kabir and Kumar 2012)
Road construction required in 20 yrs	km	36.5	N/A	Calculated from (Thakur et al. 2014, Winkler 1998)

The forest residue pathway includes forwarding and chipping (See Figure 2.1). The energy and emission impacts from production, operations, and disposal of the equipment used (forwarder and chipper) were also considered in this study.

Steel is used in the construction of all types of equipment and machines (e.g., fellers, skidders, forwarders, chippers, and transportation vehicles) considered in this study over the entire life cycle for both pathways.

Diesel is the fuel used to operate the machinery and equipment. Natural gas is used during the conversion of OME from biomass for syngas cleaning. The life cycle energy and combustion emission factors for different material and fuels considered in the system boundary for both pathways were taken from literature (Kabir and Kumar 2012, Pellegrini et al. 2013, Stripple 2001). The energy and emission factors are given in Table 2.2.

Table 2. 2: Energy and emission factors for fuel, materials, and road construction used in the system [derived from (Kabir and Kumar 2011, 2012) and (Stripple 2001)]

Diesel	HHV (MJ/L)	kg CO_2eq/GJ	kg SO_2eq/GJ	kg (NOx + VOC)/GJ	GJ/GJ
	35.97	100.30	0.39	0.63	1.29
Natural gas	HHV (MJ/kg)	kg CO_2eq/GJ	kg SO_2eq/GJ	kg (NOx +VOC)/GJ	GJ/GJ
	38.26	56.58	0.128	0.22	1.11
Steel	GJ/tonne	kg CO_2eq/GJ	kg SO_2eq/GJ	Kg (NOx +VOC)/GJ	-
	34.00	2494.86	21.15	9.66	
Road construction	GJ/km	kg CO_2eq/km	kg SO_2eq/km	Kg (NOx +VOC)/km	-
	1731	403,845	1015	1155	

The specifications of equipment used in biomass processing and harvesting for both pathways are given in Table 2.3. Energy and emission impacts from construction, operation, and disposal of equipment and machinery were considered in the system boundary.

Table 2. 3: Specifications of equipment used in whole tree and forest residue pathways for biomass harvesting, processing, and road construction.

Equipment specification	Value	Unit	Comments/References
Feller (whole tree pathway)			
John Deere 853J	205/274	kW/hp	(MacDonald 2006)
Feller lifetime productivity	95812.5	t WF[b]	Dry basis (MacDonald 2006)
Feller lifetime fuel consumption	514650	L diesel	(MacDonald 2006)
Dedicated feller required	18		Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b, Kumar et al. 2003)
Steel in each feller	28.84	t	(MacDonald 2006)
Skidder (whole tree pathway)			
John Deere 748 H	141/189	kW[b]/ hp[b]	(Han and Renzie 2001)
Skidder lifetime productivity	90000	t WF	Dry basis (Han and Renzie 2001)
Skidder lifetime fuel consumption	540000	L diesel	(Han and Renzie 2001)

Equipment specification	Value	Unit	Comments/References
Dedicated skidder required	19		Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b, Kumar et al. 2003)
Steel in each skidder	14.35	t	(Han and Renzie 2001)
Chipper (whole tree pathway)			
Morbark 50/48 chipper			(MacDonald 2006)
Chipper lifetime productivity	270000	t WF	Dry basis (MacDonald 2006)
Chipper lifetime fuel consumption	900000	L[b] diesel	(MacDonald 2006)
Steel in each chipper	28.16	t	(MacDonald 2006)
Dedicated chipper required	6		Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b, Kumar et al. 2003)
Forwarder (forest residue pathway)			
Wheel loader (Komatsu WA 250-6)	138	hp	(Mann and Spath 1997)
Forwarder lifetime productivity	101200	t FR[b]	Dry basis (MacDonald 2006)
Forwarder lifetime fuel	416000	L diesel	(MacDonald 2006)

Equipment specification	Value	Unit	Comments/References
consumption			
Steel in each forwarder	11.58	t	(Mann and Spath 1997)
Dedicated forwarder required	17		Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b, Kumar et al. 2003)
Chipper (forest residue pathway)			
Nicholson WFP 3A			(Desrochers et al. 1993)
Chipper lifetime productivity	252000	t FR	Dry basis (Desrochers et al. 1993)
Chipper lifetime fuel consumption	990000	L diesel	(Desrochers et al. 1993)
Steel in each chipper	57.82	t	(Desrochers et al. 1993)
Dedicated chipper required	7		Calculated from (Zhang et al. 2014, Shahrukh et al. 2016b, Kumar et al. 2003)
Crawler tractor (secondary and tertiary road construction)	140/105	hp /kW	(Winkler 1998)
Tractor lifetime productivity	8000	h	(Winkler 1998)

Equipment specification	Value	Unit	Comments/References
Tractor lifetime fuel consumption	184000	L diesel	(Winkler 1998)
Operating machine hours (secondary road)	70	h /km	(Winkler 1998)
Operating machine hours (tertiary road)	100	h /km	(Winkler 1998)
Dedicated tractor required (secondary and tertiary)	0.73		Calculated from (Thakur et al. 2014, Winkler 1998, Fulton Smyl, Business Analyst, Alberta Innovates-Technology Futures 2016 on June 28, 2016)

[b]WF= whole forest, FR= forest residue, kW=kilowatt, hp=horsepower, L= litre

2.3.2 Biomass Transportation

A circular biomass harvest area is assumed for both pathways. Biomass collection distance depends on two other aspects, tortuosity and geometric factors. The tortuosity factor is the ratio of the distance travelled for biomass collection divided by the visible biomass collection distance, and the geometric factor is used to measure the biomass distribution over the harvest area. A circular harvest area growing only a biomass feedstock has a geometric factor of one, and the tortuosity factor was assumed in this study (for practical transportation assumptions) to be 1.27 (Overend 1982). We assumed that the preprocessing plant was situated at the center of the harvest area. This method has been used in earlier studies (Shahrukh et al. 2016b). With these assumptions, the biomass collection distances used for the whole tree and forest residue pathways were 4.56 km and 21.75 km, respectively (Thakur et al. 2014).

Biomass is transported in heavy capacity trailer trucks. Fourteen tonnes of steel are used to manufacture a trailer truck. A trailer truck can carry 23 wet tonnes of biomass in a single trip and travel up to 2.55 km/L of fuel when empty and 2.12 km/L with a load (Kabir and Kumar 2012, Mann and Spath 1997). The energy and emission impacts of truck construction and operation are included in this study. For the whole tree pathway, road construction was considered as a subunit operation. Forest roads are classified as primary, secondary, and tertiary. Whole trees are usually slid to a primary roadside and chipped, and the chips are transported by truck on primary roads. Other harvesting machinery like fellers, skidders and chippers operate on secondary and tertiary roads on slow speed.

For an OME plant with a capacity of 277 dry tonnes of biomass per day, around 36.48 km of primary road, 42.98 km of secondary road, and 28.65 km of tertiary road construction were assumed for the whole tree pathway in this study. The road construction estimates are based on a discussion with Fulton Smyl (Business Analyst, Alberta Innovates-Technology Futures on June 28, 2016) on Alberta's forest management plans, roads classification, and design specifications. The energy and emission factors for primary road construction – 1731 GJ/km, 403,845 kg CO_2eq/km, 1015 kg SO_2eq/km, and 1155 kg (NOx + VOC)/km – are taken from previous studies (Kabir and Kumar 2011, 2012, Stripple 2001). Crawler tractors (140 horsepower /105 kilowatt) are assumed to be used for secondary and tertiary road construction (Winkler 1998). Primary roads are generally built as permanent roads, whereas secondary roads are built to be semi-permanent and tertiary roads are temporary trails mostly used for harvesting (Ontario Ministry of Natural Resources 1994). Hence, the tractor operating hours are considered to be 70 h/km for secondary road and 100 h/km for tertiary road construction in this study (Table 2.3) (Winkler 1998). For the forest residue pathway, no road construction is required. The chips are assumed to be transported using the existing road network used by regular logging companies (Kabir and Kumar 2011, 2012, Shahrukh et al. 2016b).

We calculated truck fuel consumption using a formula from (Sultana and Kumar 2011), given in Equation 1. We assumed that a truck carries less than its payload, or volumetric capacity. Truck fuel consumption is calculated as follows:

$$F_a = F_e + \{(F_f - F_e) \times (L_a / L_p)\} \tag{1}$$

where F_a = actual fuel consumed by a truck while carrying a load L_a (L/km), F_e = fuel consumed by an empty truck (L/km), F_f = fuel consumed by a fully loaded truck (L/km), L_a = actual load transported by a truck (t), and L_p= volumetric capacity of a truck (t). The inventory data for biomass transportation are given in Table 2.1.

2.3.3 OME Plant Construction

An OME plant is assumed to have 20 years of plant life. Due to similarities in the chemical conversion of OME with other fuels like biohydrogen production (such as biomass gasification, syngas cleaning, H_2/CO adjusting, etc.), the scale factor needed to determine the amount of material to construct an OME plant is taken from existing literature on other plants (Moore 1959, Spath and Mann 2000). The amount of construction material was determined through Equation 2 obtained from Sarkar and Kumar (2010a, b):

$$C_i/C_o = (S_i/S_o)^n \tag{2}$$

Here, S_i = the size of the OME plant, S_o = the size of reference plant, C_i = the amount of material required for an OME plant, S_o = the amount of material in the reference plant, and n= the scale factor. A scale factor of 0.76 was assumed in this study. The scale factor was taken from (Kabir and Kumar 2011) due to similarities in biohydrogen production operations and OME synthesis.

As an example, for a Battelle Columbus Laboratory (BCL) plant with a capacity of 250200 kg H_2/ day, (Kabir and Kumar 2011) estimated the amount of steel, concrete, and aluminium to be 5350, 16,535, and 44 t, respectively. We used these values as reference plant material amounts in Equation 2 and a scale factor of 0.76 and estimated the amount of material for an OME plant (capacity 24746 kg OME/day) to be 922 t steel, 2850 t concrete, and 7.58 t of aluminium. The energy and emission impacts of plant decommissioning and disposal of construction material are also included in the plant construction unit operation. The energy and emission impacts from plant decommissioning are assumed to be 3% of plant construction impacts (Elsayed and Mortimer 2001, Kabir and Kumar 2011). The construction materials are assumed to be disposed of in landfills 50 km from the plant (Kabir and Kumar 2011, 2012, Spath et al. 2005). Heavy capacity trucks used in biomass transportation are used for construction material disposal. All aluminium and concrete material are assumed to be landfilled, but 75% of the steel is recycled and 25% of it is landfilled (Spath et al. 2005, Spath and Mann 2000). The energy and emission factors for steel and aluminium landfilling are 0.01 tCO_2eq/tonne of material and, for concrete, 0.044 tCO_2eq/tonne of material (Spath et al. 2005, Spath and Mann 2000).

2.3.4 Chemical Conversion

Five subunit operations are included in the chemical conversion process: gasification, syngas cleaning and H_2/CO adjustment, methanol synthesis, OME synthesis, and ash disposal. Both feedstocks undergo the same process. Biomass conversion is considered to be carbon neutral as all carbon released during the combustion of the woodchips is compensated by the amount of carbon up taken during forest growth (Hartmann and Kaltschmitt 1999, Liu et al. 2013, Zhang et al. 2009). The input-output mass flow rates for chemical conversion unit operations are given in Table 2.4. The output includes OMEs 1 to 8 and some untreated gases such as N_2, O_2, water, etc. 6.80 MW of external heat energy, supplied by natural gas, are used in syngas cleaning and H_2/CO adjustment unit operations (Zhang et al. 2016). But this external energy is a small fraction (around 6.57%) of the energy consumed during the whole chemical conversion process. The remaining heat energy is supplied by the combustion of 13 – 17% of the input biomass depending on the biomass feedstocks as stated by Zhang et al. (Zhang et al. 2016).

Table 2. 4: Input-output data inventory for chemical conversion unit operations

Chemical conversion units	Inputs	Mass flow rate kg/s	Outputs	Mass flow rate kg/s
Gasification	Air	3.21	Raw syngas	5.35
	Woodchips	3.54		

Chemical conversion units	Inputs	Mass flow rate kg/s	Outputs	Mass flow rate kg/s
Syngas cleaning & adjusting	Raw syngas	5.35	Cleaned Syngas	4.09
Methanol synthesis	Cleaned syngas	4.09	Methanol	0.92
OME Synthesis	Methanol	0.92	Total OME	0.29

The ash contents in whole tree and forest residue biomass are assumed to be 1% and 3%, respectively (Van den Broek et al. 1995, Kumar et al. 2003). Over 20 years, 16,986 t and 50,957 t of ash are produced through the whole tree and forest residue pathways, respectively. The ash is assumed to be disposed of 50 km away from the plant in the forest area and is usually considered to replace the nutrients removed with the trees and residues (Kumar et al. 2003, Spath et al. 2005). The same heavy capacity trailer trucks used in biomass transportation are used to spread ash. The ash spreading rate is assumed to be 1 t/ha, and a 40' fertilizer spreader with a capacity of 4.41 ha/hour is used (Kabir and Kumar 2011, Spath et al. 2005). The life cycle energy and emission impacts of trucks and spreaders used for ash transportation and ash spreading are included in this study.

2.3.5 OME Transportation

In this study, we assumed a distance of 300 kilometers to transport OME from the chemical conversion plant to the blending plant. We also assumed that the high capacity trailer trucks used for biomass transportation and ash disposal would be used for OME transportation. The energy and emissions impacts of construction and operation of trucks are considered in this study.

2.3.6 Vehicle Combustion

Combustion emissions and fuel consumption numbers are taken from a study by Pellegrini et al. (Pellegrini et al. 2013). They tested an in-use diesel engine car with three different types of fuel, conventional diesel, 100% OME, and a 10% OME diesel blend. The particulate matter (PM) emissions were also calculated, and the PM composition determined the amount of soot emissions. Pellegrini et al. found that 77% of diesel PM emissions are black carbon/soot whereas in OME, only 33% of PM emissions are black carbon/soot and 50% of the PM emissions come from the volatile organic fractions in lube oil (Pellegrini et al. 2013). These figures are used in our model to calculate the soot emissions from vehicle combustion. The soot emissions for 100% OME and a 10% OME blend with diesel as calculated in this model are 0.0011 g/MJ of OME and 0.0071 g/MJ of OME, respectively. When a strong oxidation catalyst and a good synthetic lubricant in vehicles are used, PM emissions can be further reduced. According to the experimental results by Pellegrini et al., using 100% OME as a transportation fuel can reduce soot emissions significantly, although hydrocarbon (HC), carbon monoxide (CO), nitrogen oxides (NOx), and carbon dioxide (CO_2) emissions and fuel consumption increase (Pellegrini et al. 2013). The CO_2eq emissions from the combustion of 100% OME as a transportation fuel are considered to be zero in our model since we assume that the combustion emissions of biomass-derived fuels are compensated by the amount of CO_2 taken up by the tree during its growth. The combustion emissions from a 10% OME blend with diesel are around 0.060 CO_2eq/MJ of OME and come predominantly from the diesel fraction.

2.4 Results and Discussion

In this section, the results of the life cycle energy and emission impact assessments for both whole tree and forest residue pathways are presented, compared, and discussed. The sensitivity analyses for the different scenarios are also discussed.

2.4.1 Life Cycle Energy and Emission Impacts for the Whole Tree Pathway

Table 2.5 shows the energy consumptions and GHG emissions results for the upstream operations from whole tree biomass.

Table 2. 5: Life cycle energy use and emissions for different upstream unit operations of the whole tree pathway

Preliminary results	**Energy use**	**GHG emissions**	**ARP emissions**	**GOP emissions**
Units	GJ/MJ	g CO_2eq/MJ	g SO_2eq/MJ	g(NO_x+VOC)/MJ
Biomass production	0.18	14.25	0.057	0.088
Biomass transportation	0.03	5.41	0.014	0.017
Chemical conversion	1.24	5.61	0.017	0.020
OME transportation	0.01	0.50	0.002	0.003

Of the upstream unit operations (biomass production, biomass transportation, and chemical conversion), chemical conversion consumes the most energy, almost 85% of the energy consumed in the pathway. The primary energy input for chemical conversion is the heat from the wood chips, which is around 26.37 MW (8.215 MJ/kg). The output from one subunit operation is used as input for the next. About 6.80 MW of fossil heat energy (from natural gas) are used in the syngas cleaning and H_2/CO adjusting subunit operation; this is the only fossil energy considered in chemical conversion operation. Biomass harvesting is the second most energy-intensive unit operation, and biomass transportation consumes the least energy over the entire life cycle of whole tree pathway.

For chemical conversion, OME synthesis in the whole tree pathway consumes the most energy, around 28% of the energy used in the conversion (Figure 2.2). The energy required in gasification (around 23% of the energy consumed in chemical conversion) comes primarily from the biomass (wood chips). The fossil energy used in this pathway is around 6.57%, which is supplied by natural gas.

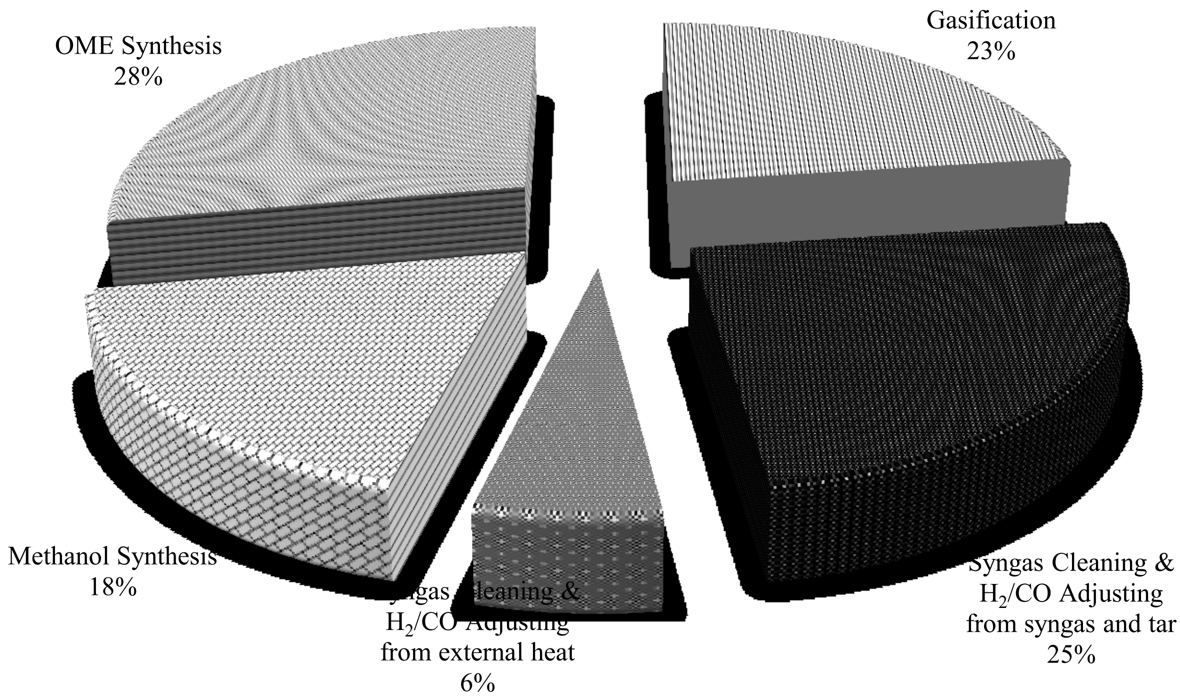


Figure 2. 2: Energy consumption by sub-unit operations in chemical conversion for the whole tree pathway.

Equipment construction energy is negligible compared to equipment operation energy for all the biomass harvesting equipment considered in this study. Around 40% of the energy used in the whole tree pathway is consumed in skidding operations. For this study, 19 skidders with a productivity of 7.5 dry tonnes whole trees per hour are required for a plant capacity of 277 t/d over 20 years (Table 2.3). A skidder's hourly productivity is comparatively much lower than that of a feller (8.75 dry tonnes whole trees per hour) or a chipper (30 dry tonnes whole trees per hour for a high-efficient chipper). But a skidder's life cycle fuel consumption is higher than that of a feller (Table 2.3), making it the highest energy-consuming unit in biomass production. Only six high-efficient chippers are required over 20 years, hence chipping is the least energy consuming unit (around 22%) throughout the life cycle (Table 2.3). Vehicle combustion is the most GHG emissions-intensive unit, contributing around 77% of life cycle GHG emissions in the whole tree pathway. However, this unit operation is considered to be carbon neutral, thereby nullifying the effect of GHG emissions. Biomass production produces the second highest GHG emissions and contributes 12% of emissions over the life cycle.

In this study, it was assumed that 36.5 km of primary roads were constructed in order to haul whole tree biomass from the forest. This is the third most emissions-intensive unit operation (5.35% of total life cycle GHG emissions) when using whole tree biomass as an energy source. The impact of this subunit operation on the entire life cycle emissions is discussed in the sensitivity analysis.

Biomass production contributes around 12% of the GHG emissions over the entire life cycle for the whole tree pathway. For biomass production, the skidder operation is the most emissions-intensive unit (Figure 2.3). Because of its relatively lower productivity and comparatively higher fuel consumption compared to the other unit operations, skidder operations contribute the most GHG emissions over the whole tree pathway life cycle, around 40%. The fellers contribute 36% of the life cycle GHG emissions followed by chipper operation emissions, which are around 22%. Equipment construction emissions are negligible compared to equipment operation emissions.

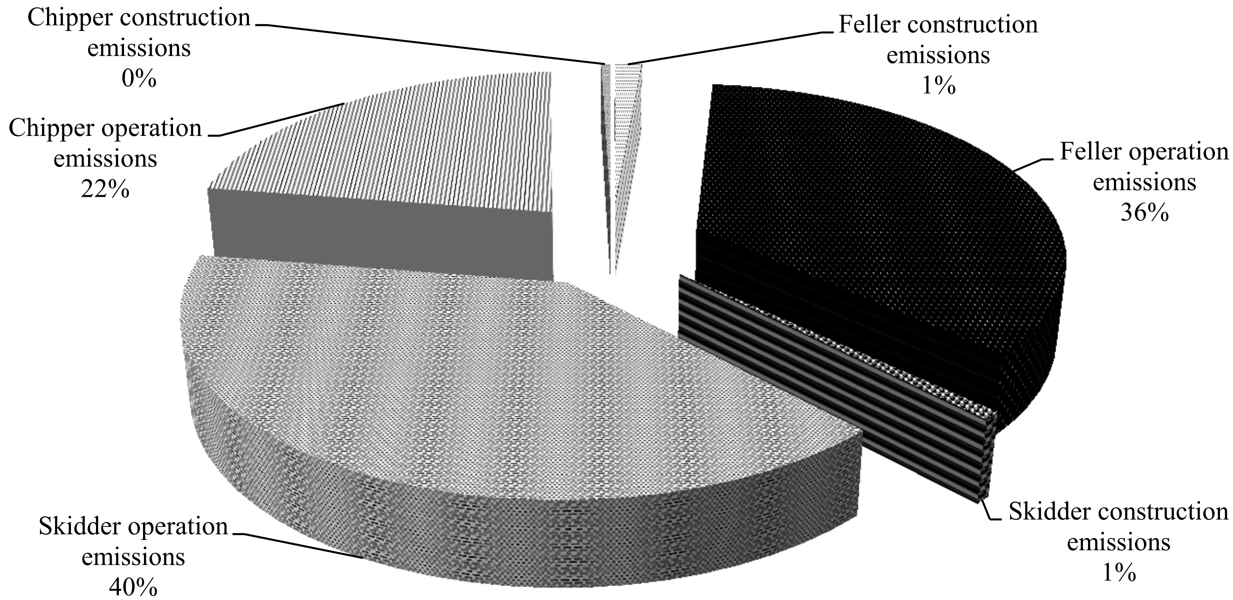


Figure 2. 3: GHG emissions from different sub-unit operations in biomass production (gCO_2eq/MJ) in the whole tree pathway.

Over the entire life cycle of the whole tree pathway, the chemical conversion unit process contributes very few GHG emissions (only 4.82%); this result is mainly based on the assumptions that the amount of CO_2 released during the gasification of the forest biomass is equal to the amount of CO_2 taken up by the tree during its growth and the amount of external fossil energy used during chemical conversion is negligible (only 6.57% of life cycle energy consumption). The GHG emissions from ash disposal, including ash transportation and ash spreading, are included in this unit operation. OME transportation emits the fewest GHGs over the entire life cycle, only 0.43 % of life cycle GHG emissions.

The biomass production unit operation contributes the highest ARP emissions, around 62% of the life cycle ARP emissions. The biomass transportation, chemical conversion, and OME transportation unit processes contribute around 17%, 18%, and 2% of the life cycle ARP emissions, respectively. Due to the lack of data, ARP and GOP emissions were not calculated for the downstream unit operation, combustion in vehicles, for either pathway.

GOP emissions for biomass production, biomass transportation, and chemical conversion are around 0.09 g (NO_x +VOC)/MJ, 0.018 g (NO_x +VOC)/MJ, and 0.02 g (NO_x +VOC)/MJ respectively. GOP emissions from the OME transportation unit operation are negligible, around 2% of the life cycle GOP emissions.

2.4.2 Life Cycle Energy and Emission Impacts for the Forest Residue Pathway

Table 2.6 shows the preliminary energy consumption and GHG emissions results for the upstream operations from forest residue biomass.

Table 2. 6: Life cycle energy use and emissions for different upstream operations in the forest residue pathway.

Preliminary results	Energy use	GHG emissions	ARP emissions	GOP emissions
Units	GJ/MJ	gCO_2eq/MJ	gSO_2eq/MJ	g(NO_x +VOC)/MJ
Biomass production	0.13	10.15	0.041	0.063
Biomass transportation	0.02	1.45	0.006	0.009

Preliminary results	Energy use	GHG emissions	ARP emissions	GOP emissions
Chemical conversion	1.24	5.67	0.020	0.021
OME transportation	0.01	0.50	0.002	0.003

Whole tree and forest residue biomass feedstocks use the same chemical conversion process. Thus, as for the whole tree pathway, chemical conversion is the highest energy-consuming upstream unit operation in the forest residue pathway, around 89% of the life cycle energy, followed by biomass production, biomass transportation, and OME transportation, which consume around 9%, 1.3%, and 0.47% of the energy, respectively. Among the four unit operations, vehicle combustion emits the most GHGs, around 83% of the life cycle GHG emissions.

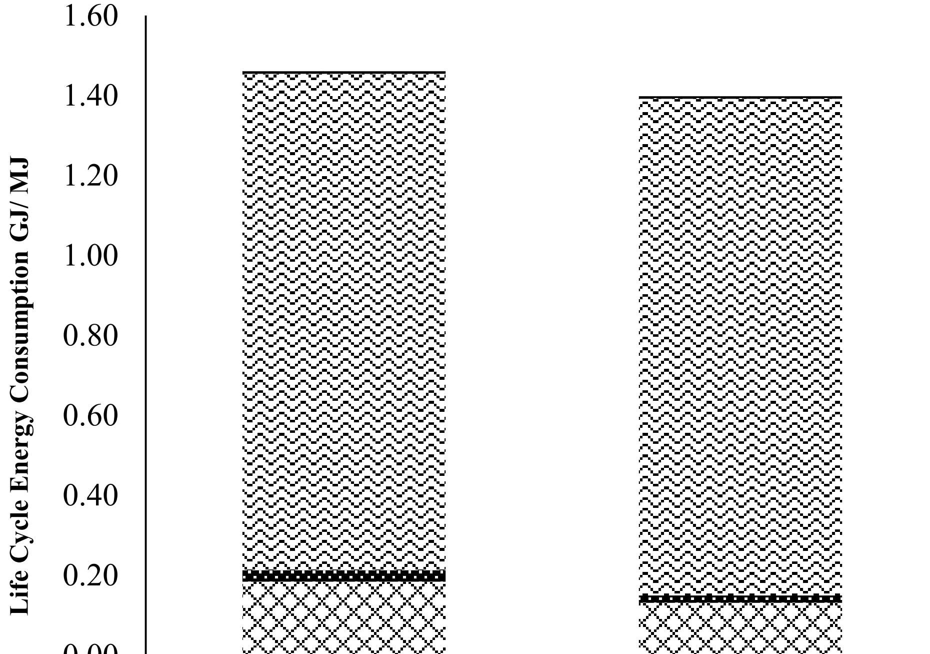
Biomass production produces 9.5% of the GHG emissions over the entire life cycle, followed by chemical conversion, biomass transportation, and OME transportation at around 5%, 1.35%, and 0.47% of life cycle GHG emissions, respectively. The transportation emissions are low mainly because no road construction is considered for the forest residue pathway (existing roads built for logging operations are used). Similar to the whole tree pathway, chemical conversion emissions are almost carbon neutral and hence contribute only 5% of the life cycle GHG emissions. Equipment construction emissions are also negligible compared to equipment operation emissions, as for the whole tree pathway. Around 50% of biomass production emissions are from forwarder operation emissions due to the forwarder's low productivity. For an OME plant with a capacity of 277 t/d, around 17 forwarders are required throughout the 20 years of plant life. Six highly productive (more than twice the productivity of a forwarder) chippers are used over 20 years of plant life, producing 47% of the biomass production emissions, almost the same as that from forwarders.

Similar to the whole tree pathway, ARP emissions are highest for the biomass production unit operation (around 61% of the life cycle ARP emissions), followed by the chemical conversion, biomass transportation, and OME transportation operations, which contribute around 26% , 9%, and 3% of the total life cycle ARP emissions, respectively.

In the forest residue pathway, biomass production GOP emissions are 66% of the total life cycle GOP emissions, and chemical conversion, biomass transportation, and OME transportation contribute around 22%, 9%, and 3% of the life cycle GOP emissions, respectively. OME transportation contributes the lowest GOP emissions, around 0.003 g (NO_x +VOC)/MJ of OME.

2.4.3 Comparison of Life Cycle Energy and Emission Impacts between the Two Pathways

Figure 2.4 shows the life cycle energy consumption of four unit operations – biomass production, biomass transportation, chemical conversion, and vehicle combustion – in the whole tree and forest residue pathways.

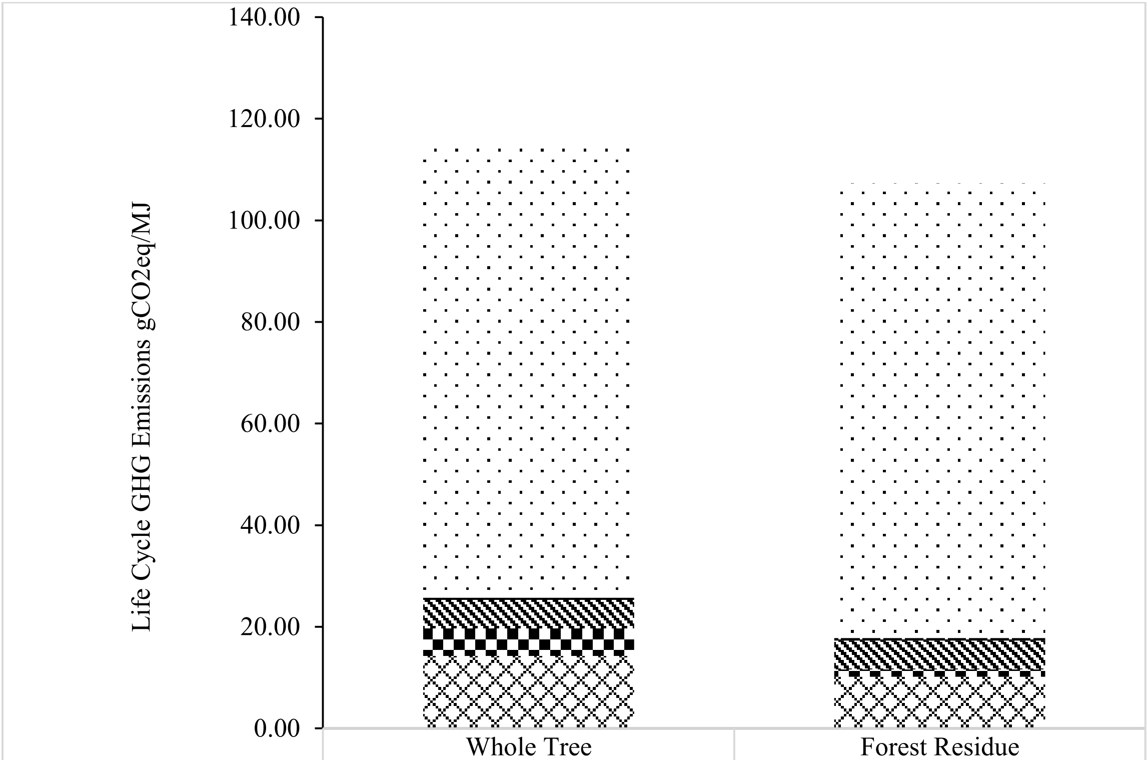


	Whole Tree	Forest Residue
OME transportation	0.01	0.01
Chemical conversion	1.24	1.24
Biomass transportation	0.03	0.02
Biomass production	0.18	0.13

Figure 2. 4: Whole tree and forest residue pathways' life cycle energy consumption comparison.

Both pathways use the same chemical conversion process, and chemical conversion is the most energy-intensive upstream operation for both (around 80-85%). Since road construction is considered in the whole tree and not the forest residue pathway, biomass transportation energy consumption in the whole tree pathway is twice as high as in the forest residue pathway (even though the transportation distance for biomass collection in the forest residue pathway [21.75 km] is almost 5 times higher than in the whole tree pathway [4.56 km]. However, biomass production energy in the whole tree pathway is higher than that of the forest residue pathway. This is due to the effects of the subunit operations involved in biomass production. In the whole tree pathway, biomass production has three subunit operations (skidding, felling, and chipping), and skidding consumes the most energy (almost 40% of the energy consumed in biomass production). In the forest residue pathway, the biomass production unit includes only forwarding and chipping, neither of which consumes large amounts of energy.

For both pathways, vehicle combustion produces the highest GHG emissions, around 89.55 g CO_2eq/MJ of OME. In the whole tree pathway, vehicle combustion contributes around 77% of the life cycle GHG emissions and in the forest residue pathway, vehicle combustion is responsible for 83% of the life cycle GHG emissions (see Figure 2.5). Since OME is produced from biomass, combustion emissions are considered to be carbon neutral. Hence 83% of life cycle GHG emissions in the forest residue pathway and 77% in the whole tree pathway are considered carbon neutral, and thus the forest residue pathway produces fewer GHGs than the whole tree pathway. In both pathways, the second highest GHG emissions come from biomass production (around 12% of the life cycle emissions from the whole tree and 9.5% from the forest residue pathway). Biomass transportation emissions in the whole tree pathway are almost four times higher than those of the forest residue pathway (Figure 2.5). This is mainly due to the emissions-intensive unit operation road construction. About 36.5 km of primary road construction is considered in the whole tree pathway.



	Whole Tree	Forest Residue
Vehicle combustion	89.55	89.55
OME transportation	0.49	0.49
Chemical conversion	5.61	5.67
Biomass transportation	5.41	1.45
Biomass production	14.25	10.15

Figure 2. 5: Whole tree and forest residue pathways' life cycle GHG emissions comparison.

Around 12% of life cycle emissions come from biomass production in the whole tree pathway and 9.5% the forest residue pathway. GHG emissions from whole tree and forest residue biomass production are 14.25 g CO_2eq/MJ and 10.15 g CO_2eq/MJ, respectively. GHG emissions from whole tree biomass production are higher because of the differences in biomass production energy consumption, as explained earlier.

GHG emissions from chemical conversion are around 5.67 g CO_2eq/MJ in the forest residue pathway and 5.61 g CO_2eq/MJ in the whole tree pathway. The difference is due to the higher ash content in forest residues. Because of the higher ash content, this pathway produces more ash than the whole tree pathway, thereby contributing slightly higher GHG emissions. With respect to ARP emissions, biomass production is the highest contributor in both pathways. Whole tree biomass production produces around 0.06 g SO_2eq/ MJ and forest residue biomass production contributes 0.04 g SO_2eq/MJ (see Tables 2.5 and 2.6). The highest GOP emissions come from whole tree biomass production and are around 0.09 g (NO_x +VOC)/MJ (see Table 2.5). The forest residue pathway also generates the highest GOP emissions from biomass production unit operations; these are around 0.06 g (NO_x +VOC)/MJ (see Table 2.6).

2.4.4 Comparison with Diesel Life Cycle Energy and Emission Impacts

We compared life cycle GHG emission numbers of OME derived from forest biomass to those of the conventional fossil fuel diesel. Several LCA studies have been published on diesel life cycle emissions (Garg et al. 2013, Gerdes and Skone 2009, Rahman et al. 2015). The diesel GHG emission numbers include emissions from crude recovery, crude transportation to the refinery, crude refining, transportation and distribution of finished fuels to the dispensing station, and combustion of fuels in vehicles. The upstream GHG emission numbers, from crude recovery to dispensing fuel, are taken from (Rahman et al. 2015), and the combustion emission numbers for diesel are taken from (Pellegrini et al. 2013).

The well-to-wheel (WTW) diesel life cycle GHG emissions calculated by Rahman et al. were 126.54 g CO_2eq/MJ (Rahman et al. 2015), whereas in this study the life cycle GHG emissions from 100% OME as a transportation fuel were found to be 27g CO_2eq/MJ when OME is produced from whole trees and 18g CO_2eq/MJ when OME is produced from forest residues (Figure 2.6). In the OME pathways, GHG emissions from vehicle combustion are assumed to be carbon neutral and the chemical conversion process is assumed to be almost carbon neutral since only 6.57% of life cycle energy consumption comes from a fossil source.

Table 2. 7: Upstream emissions, combustion emissions, total life cycle GHG emissions, total life cycle soot emissions, and reductions in GHG and soot emissions compared to diesel for OME and OME blends with diesel.

Fuels	Upstream emissions	Combustion emissions	Accountable combustion emissions	Total life cycle GHG emissions	Reductions compared to diesel (%)	Life cycle soot emission	Reductions compared to diesel (%)
	g CO_2eq/MJ	g CO_2eq/MJ	g CO_2eq/MJ	g CO_2eq/MJ		g/MJ	
Diesel	34.98	91.55	91.55	126.54	N/A	0.0101	N/A
100% OME (a)[c]	25.99	89.55	0	25.99	79.5	0.0011	89
10% OME blend (a)	33.65	91.44	86.56	120.21	5	0.0071	30
100%	17.76	89.55	0	17.76	86	0.0011	89

Fuels	Upstream emissions	Combustion emissions	Accountable combustion emissions	Total life cycle GHG emissions	Reductions compared to diesel (%)	Life cycle soot emission	Reductions compared to diesel (%)
OME (b)[c]							
10% OME blend (b)	33.21	91.44	86.56	119.77	5.35	0.0071	30

[c](a) denotes OME produced from whole tree biomass and (b) denotes OME produced from forest residues

Hence, total life cycle emissions from OME pathways are significantly lower than those of diesel. Total life cycle GHG emissions and percentage reductions in GHGs compared to conventional diesel for 100% OME and a 10% OME blend with diesel to be used as transportation fuels are given in Table 2.7. The upstream emissions from the forest residue pathway (18g CO_2eq/MJ) are significantly lower than those of the whole tree pathway (27 g CO_2eq/MJ).

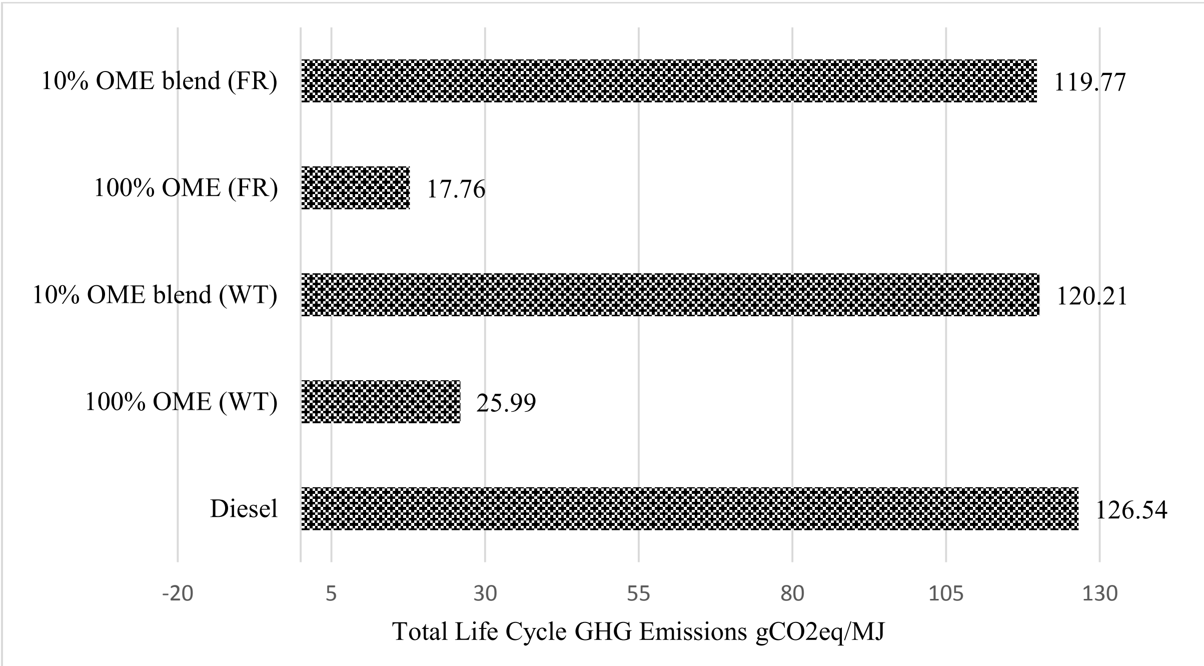


Figure 2. 6: OME and OME blends from whole tree and forest residue pathways' GHG emissions compared with conventional diesel.

Hence, 100% OME as a transportation fuel from the forest residue pathway contributes 86% fewer GHG emissions than diesel, whereas 100% OME from the whole tree pathway contributes 79% fewer GHG emissions than diesel. Similarly, when OME is used as a diesel additive, for the 10% OME blended with 90% diesel, the life cycle GHG emissions are reduced by 5% and 5.35% compared to that of diesel, when OME is produced from the whole tree and forest residue pathways, respectively. Upstream emissions are allocated to the OME blends depending on their mass in the finished fuel.

The soot emissions for 100% OME and a 10% OME blend with diesel as calculated in our model are 0.0011 g/MJ of OME and 0.0071 g/MJ of OME, whereas the soot emissions from diesel are 0.01 g/MJ of diesel (Pellegrini et al. 2013). We compared the soot emissions from a 10% OME blend and 100% OME to the soot emissions from diesel and found that soot emissions decrease by 30% and 89% compared to diesel for a 10% OME blend with diesel and 100% OME, respectively. The soot emissions for all three fuels are shown in Table 2.7.

2.4.5 Sensitivity Analysis

A number of scenarios were developed for both pathways by varying parameters and assumptions of upstream operations, and the impacts of these variations on life cycle energy and emissions are given in Table 2.8. The scenarios were developed independently of each other and compared with the base scenario. The downstream operation (vehicle combustion) is not included in this analysis. Four scenarios were developed for the forest residue pathway and six for the whole tree pathway.

In scenario 1, the change in capacity factors for both pathways was analyzed. The pathways were analyzed for two sets of capacity factors: set one at 0.7 for year 1, 0.8 for year 2, 0.95 from year 3 onwards and set two at 0.65 for year 1, 0.7 for year 2, 0.75 from year 3 onwards. Life cycle energy and emissions increased with the increased capacity factors for both pathways, and, in the forest residue pathway, both increase significantly. As an example, GHG emissions increased around 9% over the base scenario in the forest residue pathway with the increased capacity factors (see Table 2.8). Scenario 2 demonstrates the effects of a 10% increase and decrease in biomass yield. When the yield increases, life cycle energy consumption and emissions drop for both pathways, and when yield decreases, energy consumption and emissions increase. But the changes are insignificant and are within ±1%. Scenario 3 looks at the effects of a 10% increase and a 10% decrease in biomass moisture content for both pathways. The impact is small and is within ±1%. In scenario 4, we analyzed life cycle emission and energy consumption impacts by changing the capacity by ±10%. Overall energy consumption and emissions increase with increased capacity, but the energy consumption per unit output (per tonne of OME produced) decreases as the capacity increases. For the whole tree pathway, a fifth scenario was developed considering silviculture, which involves the application of fertilizer and pesticides and considers machinery fuel consumption. Energy consumption and emissions increases were negligible. Scenario 6 demonstrates the impact of excluding road construction operations in the whole tree pathway. Road construction is assumed to be an emissions-intensive operation in the whole tree pathway. We found that the energy consumption and life cycle emissions dropped significantly compared to the base scenario. The GHG emissions also dropped considerably, by around 33% compared to base scenario, and the other two emissions, ARP and GOP, dropped to 32% and 24% of the base scenario, respectively. Life cycle energy consumption was reduced by 4% from the base scenario (Table 2.8).

Table 2. 8: Sensitivity analysis and results

		Energy Use	GHG Emissions	ARP Emissions	GOP Emissions	% Change from Base Case			
	Scenario	GJ/MJ	g CO_2eq/MJ	g SO_2eq/MJ	g (NOx +VOC)/MJ	Energy Use	GHG Emission	ARP Emission	GOP Emission
FR[d]	1a[d]	1.39	24.52	0.09	0.14	-2.00	-9.37	-10.36	-10.18
	1b[d]	1.33	20.17	0.07	0.11	2.14	10.04	11.09	10.90
WT[d]	1a	1.76	89.92	0.27	0.37	-2.31	-3.54	-4.78	-5.50
	1b	1.68	83.51	0.25	0.33	2.50	3.84	5.17	5.95
FR	2a	1.36	22.31	0.08	0.13	0.11	0.52	0.59	0.56
	2b	1.36	22.56	0.08	0.13	-0.13	-0.61	-0.69	-0.65
WT	2a	1.72	86.83	0.26	0.35	0.00	0.01	0.01	0.01
	2b	0.00	86.85	0.26	0.35	-0.01	-0.01	-0.01	-0.01
FR	3a	1.36	22.32	0.08	0.13	0.10	0.47	0.53	0.50
	3b	1.36	22.52	0.08	0.01	-0.10	-0.45	-0.52	-0.49
WT	3a	1.72	86.83	0.26	0.35	0.01	0.01	0.01	0.01
	3b	1.72	86.85	0.26	0.35	0.00	-0.01	-0.01	-0.01
FR	4a	1.38	24.38	0.09	0.14	-1.87	-8.74	-9.66	-9.49
	4b	1.33	20.48	0.07	0.12	1.85	8.66	9.56	9.40
WT	4a	1.76	89.71	0.27	0.36	-2.15	-3.31	-4.46	-5.13
	4b	1.69	83.97	0.25	0.33	2.15	3.31	4.46	5.13
WT	5	1.72	86.87	0.26	0.35	-0.03	-0.04	-0.05	-0.06
WT	6	1.49	32.81	0.12	0.19	4	33	32	24

[d] *a* corresponds to a positive change of parameters, *b* corresponds to a negative change of parameters, FR = forest residue pathway and WT = whole tree pathway

The negative sign denotes an increase from the base case and the positive sign denotes a decrease from the base case.

2.5 Conclusion

This study determined the overall life cycle emissions of OME derived from two different types of forest biomass, whole tree and forest residue, and used as a diesel additive. The life cycle GHG emissions of OME from the whole tree and forest residue pathways are 27 g CO_2eq/MJ and 18 g CO_2eq/MJ, respectively. The results show that a 10% OME blend with diesel reduces GHG and soot emissions by 20-21% and 30%, respectively, compared to 100% diesel. Based on these results, it is obvious that OME, when used as a diesel additive, can decrease GHG emissions significantly compared to conventional diesel. This model can be used to design an optimal process for maximizing OME production and minimizing energy consumption and GHG emissions. The model can also be used to determine the optimum fuel mix (OME-diesel blend) contributing the lowest GHG emissions. We recommend for further studies that the model be extended to include other feedstocks such as agricultural residues, wood waste, or fossil fuels to produce OME and other modes of biomass transportation such as bales, pellets, etc. The results of this study will be of great interest to policy makers, petroleum-based fuel producers, and biofuel companies on the environmental impacts of blending OME with diesel fuels.

References

Agbor, E., Oyedun, A.O., Zhang, X., Kumar, A., (2016). "Integrated techno-economic and environmental assessments of sixty scenarios for co-firing biomass with coal and natural gas". Appl Energ 169, 433-449.

Alberta Energy 1985. "Alberta Phase 3 Forest Inventory - An Overview". Alberta Energy and Natural Resources. Available from: https://archive.org/details/albertaphase3for00albe (Accessed Date: October 20, 2016).

Beer, T., Grant, T., (2007). "Life-cycle analysis of emissions from fuel ethanol and blends in Australian heavy and light vehicles". J. Clean. Prod. 15, 833-837.

Beer, T., Grant, T., Williams, D., Watson, H., (2002). "Fuel-cycle greenhouse gas emissions from alternative fuels in Australian heavy vehicles". Atmos. Environ. 36, 753-763.

Beer, T., Olaru, D., Van der Schoot, M., Grant, T., Keating, B., Hatfield Dodds, S., Smith, C., Azzi, M., Potterton, P., Mitchell, D., (2003). "Appropriateness of a 350 Million Litre Biofuels Target: Report to the Australian Government Department of Industry, Tourism and Resources". CSIRO, ABARE, BTRE, Canberra.

Bond, T.C., Doherty, S.J., Fahey, D., Forster, P., Berntsen, T., DeAngelo, B., Flanner, M., Ghan, S., Kärcher, B., Koch, D., (2013). "Bounding the role of black carbon in the climate system: A scientific assessment". J. G. R. Atmospheres 118, 5380-5552.

Burger, J., Siegert, M., Ströfer, E., Hasse, H., (2010). "Poly (oxymethylene) dimethyl ethers as components of tailored diesel fuel: Properties, synthesis and purification concepts". Fuel 89, 3315-3319.

Desrochers, L., Puttock, D., Ryans, M., (1993). "The economics of chipping logging residues at roadside: a study of three systems". Biomass Bioenerg. 5(6), 401-411.

Elsayed, M., Mortimer, N., (2001). "Carbon and energy modelling of biomass systems: conversion plant and data updates". Harwell Laboratory, Energy Technology Support Unit. Report No: ESTU B/U1/00644/REP and DTI/Pub URN 01/1342.

Environment Canada, (2013). "In: National inventory report: greenhouse gas sources and sinks in Canada Part 1". Canada's 2013 UNFCCC Submission; p. 171–97.

EPA, (2014). "Climate Change Indicators in the United States: Global Greenhouse Gas Emissions". United States Environmental Protection Agency. Available from: https://www3.epa.gov/climatechange/pdfs/print_global-ghg-emissions-2014.pdf (Accessed Date: June 15, 2016).

Garg, A., Vishwanathan, S., Avashia, V., (2013). "Life cycle greenhouse gas emission assessment of major petroleum oil products for transport and household sectors in India". Energy Policy 58, 38-48.

Gerdes, K., Skone, T., (2009). An evaluation of the extraction, transport and refining of imported crude oils and the impact on life cycle greenhouse gas emissions. National Energy Technology Laboratory. Report No: DOE/NETL-2009/1362. Available from: http://www.netl.doe.gov/energy analyses/pubs/PetrRefGHGEmiss_ImportSourceSpecific1.pdf (Accessed Date: April 20, 2016).

Government of Canada, 2016. "Forestry: Region of Western Canada and the Territories: 2014-2016". Available from: http://www.edsc.gc.ca/img/edsc-esdc/jobbank/SectoralProfiles/WT/2A71-RPT-SectProf_WT_Forestry_EN--GEN-20150827-VF-MR.pdf (Accessed Date: October 21, 2016).

Han, H. S., Renzie, C., (2001). "Snip & skid: partial cut logging to control mountain pine beetle infestations in British Columbia". Prince George, BC: University of Northern British Columbia.

Available from: https://www.researchgate.net/profile/Han_Sup_Han/publication/228732211_Snip_skid_partial_cut_logging_to_control_mountain_pine_beetle_infestations_in_British_Columbia/links/568c091508ae71d5cd04a974.pdf (Accessed Date: April 05, 2017).

Hartmann, D., Kaltschmitt, M., (1999). "Electricity generation from solid biomass via co-combustion with coal: energy and emission balances from a German case study". Biomass Bioenerg. 16, 397-406.

Kabir, M.R., Kumar, A., (2011). "Development of net energy ratio and emission factor for biohydrogen production pathways". Bioresour. Technol. 102, 8972-8985.

Kabir, M.R., Kumar, A., (2012). "Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways". Bioresour. Technol. 124, 394-405.

Kajaste, R., (2014). "Chemicals from biomass–managing greenhouse gas emissions in biorefinery production chains–a review". J Clean Prod. 75, 1-10.

Kumar, A., Cameron, J.B., Flynn, P.C., (2003). "Biomass power cost and optimum plant size in western Canada". Biomass Bioenerg. 24, 445-464.

Liu, X., Saydah, B., Eranki, P., Colosi, L.M., Mitchell, B.G., Rhodes, J., Clarens, A.F., (2013). "Pilot-scale data provide enhanced estimates of the life cycle energy and emissions profile of algae biofuels produced via hydrothermal liquefaction". Bioresour. Technol 148, 163-171.

Liu, Z., Peng, W., Motahari-Nezhad, M., Shahraki, S., Beheshti, M., (2016). "Circulating fluidized bed gasification of biomass for flexible end-use of syngas: a micro and nano scale study for production of bio-methanol". J Clean Prod. 129, 249-255.

Löfvenberg, U., (2010). "The BEST experiences with low blends in diesel and petrol fuels. Integrated project (Project No: TREN/05/FP6EN/S07.53807/019854) for BEST (BioEthanol for

Sustainable Transport) Deliverable No: D3.15". Available from: http://www.stockholm.se/best-europe.

MacDonald, A.J., (2006). "Estimated costs for harvesting, comminuting, and transporting beetle-killed pine in the Quesnel/Nazko area of central British Columbia". FERIC advantage report, vol. 17, number 16, Vancouver, BC. Forest Engineering Research Institute of Canada (FERIC).

Mahbub, N., Kumar, A., (2014). "Co-product use of ethanol produced from wheat grain in Alberta", presented at the 64th Canadian Chemical Engineering Conference, Niagara Falls, Ontario, Canada http://abstracts.csche2015.ca/00000394.htm

Mann, M.K., Spath, P.L., (1997). "Life cycle assessment of a biomass gasification combined-cycle power system". National Renewable Energy Lab., Golden, CO (US). Report No: 23076. Available from: http://www.nrel.gov/docs/legosti/fy98/23076.pdf.

Mann, M., Spath, P., (2001). "A life cycle assessment of biomass cofiring in a coal-fired power plant". Clean Technol Envir, 3(2), 81-91.

Matzen, M., Demirel, Y., (2016). "Methanol and dimethyl ether from renewable hydrogen and carbon dioxide: Alternative fuels production and life-cycle assessment". J Clean Prod. 139, 1068-1077.

Moore, F.T., (1959). "Economies of scale: Some statistical evidence". The Quarterly Journal of Economics, 232-245.

Myhre, G., Shindell, D., Bréon, F. M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J. F., Lee, D., Mendoza, B., Nakajima T., Robock A., Stephens G., Takemura T., Zhang, H., (2013). "Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis". Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M.

Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Nguyen, T. L. T., Hermansen, J. E., Nielsen, R. G., (2013). "Environmental assessment of gasification technology for biomass conversion to energy in comparison with other alternatives: the case of wheat straw". J Clean Prod. 53, 138-148.

Ontario Ministry of Natural Resources, (1994). "Reasons for Decision and Decision: Class Environmental Assessment by the Ministry of Natural Resources for Timber Management on Crown Lands in Ontario". Environmental Assessment Board: Ontario Ministry of Natural Resources. Available from: http://www.web2.mnr.gov.on.ca/mnr/forests/timberea/decision_pdfs/intro.pdf. (Accessed Date: July 26, 2016).

Overend, R., (1982). "The average haul distance and transportation work factors for biomass delivered to a central plant". Biomass 2, 75-79.

Pellegrini, L., Marchionna, M., Patrini, R., Beatrice, C., Giacomo, N.D., Guido, C., (2012). "Combustion Behaviour and Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in a Light-Duty Diesel Engine". SAE Technical Paper No: 2012-01-1053.

Pellegrini, L., Marchionna, M., Patrini, R., Salvatore, F., (2013). "Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car". SAE Technical Paper No: 2013-01-1035.

Pellegrini, L., Patrini, R., Marchionna, M., (2014). "Effect of POMDME Blend on PAH Emissions and Particulate Size Distribution from an In-Use Light-Duty Diesel Engine". SAE Technical Paper No: 2014-01-1951.

Perera, E., Sanford, T., (2011). "Climate Change and Your Health. Rising Temperatures and Worsening Ozone Pollution". The Union of Concerned Scientists. Available from: http://www.ucsusa.org/sites/default/files/legacy/assets/documents/global_warming/climate-change-and-ozone-pollution.pdf. (Accessed Date: August 30, 2016).

Pulkki, R., (1997). "Cut-to-length, tree-length or full tree harvesting. Central Woodlands" 1(3), 22-27.

Rahman, M.M., Canter, C., Kumar, A., (2015). "Well-to-wheel life cycle assessment of transportation fuels derived from different North American conventional crudes". Appl. Energy 156, 159-173.

Riaz, A., Zahedi, G., Klemeš, J.J., (2013). "A review of cleaner production methods for the manufacture of methanol". J Clean Prod. 57, 19-37.

Row, J., Mohareb, E., (2014). "Energy efficiency potential in alberta. Technical report, Alberta Energy Efficiency Alliance". Available from: http://www.aeea.ca/pdf/ee-potential-in-ab.pdf (Accessed Date: October 20, 2016).

Sarkar, S., Kumar, A., (2010a). "Biohydrogen production from forest and agricultural residues for upgrading of bitumen from oil sands". Energy 35, 582-591.

Sarkar, S., Kumar, A., (2010b). "Large-scale biohydrogen production from bio-oil". Bioresour. Technol. 101, 7350-7361.

Shahrukh, H., Oyedun, A.O., Kumar, A., Ghiasi, B., Kumar, L., Sokhansanj, S., (2015). "Net energy ratio for the production of steam pretreated biomass-based pellets". Biomass Bioenerg. 80, 286-297.

Shahrukh, H., Oyedun, A.O., Kumar, A., Ghiasi, B., Kumar, L., Sokhansanj, S., (2016a). "Comparative net energy ratio analysis of pellet produced from steam pretreated biomass from agricultural residues and energy crops". Biomass Bioenerg. 90, 50-59.

Shahrukh, H., Oyedun, A.O., Kumar, A., Ghiasi, B., Kumar, L., Sokhansanj, S., (2016b). "Techno-economic assessment of pellets produced from steam pretreated biomass feedstock". Biomass Bioenerg. 87, 131-143.

Spath, P., Aden, A., Eggeman, T., Ringer, M., Wallace, B., Jechura, J., (2005). "Biomass to Hydrogen Production Detailed Design and Economics Utilizing the Battelle Columbus Laboratory Indirectly-Heated Gasifier". National Renewable Energy Laboratory Golden, CO. Report No: NREL/TP-510-37408. Available from: http://www.nrel.gov/docs/fy05osti/37408.pdf.

Spath, P.L., Mann, M.K., (2000). "Life cycle assessment of hydrogen production via natural gas steam reforming". National Renewable Energy Laboratory Golden, CO. Report No: NREL/TP-570-27637. Available from: http://pordlabs.ucsd.edu/sgille/mae124_s06/27637.pdf.

Spitzley, D.V., Keoleian, G.A., (2004). "Life cycle environmental and economic assessment of willow biomass electricity: A comparison with other renewable and non-renewable sources". Ann Arbor, MI: Center for Sustainable Systems, University of Michigan. Available from: http://css.snre.umich.edu/publication/life-cycle-environmental-and-economic-assessment-willow-biomass-electricity-comparison.

Stripple, H., 2001. Life Cycle Assesment of Road: A Pilot Study for Inventory Analysis; IVL Swedish Environmental Research Institute: Stockholm, Sweden, 2001.

Sultana, A., Kumar, A., (2011). "Development of energy and emission parameters for densified form of lignocellulosic biomass". Energy 36, 2716-2732.

Thakur, A., Canter, C.E., Kumar, A., (2014). "Life-cycle energy and emission analysis of power generation from forest biomass". Appl. Energ. 128, 246-253.

Van-Dal, É.S., Bouallou, C., (2013). "Design and simulation of a methanol production plant from CO2 hydrogenation". J Clean Prod. 57, 38-45.

Van den Broek, R., Faaij, A., Van Wijk, A., (1995). Biomass combustion power generation technologies. Biomass combustion power generation technologies: Background report 41 for the EU Joule 2+ project: Energy from biomass: An assessment of two promising systems for energy production (NWS-95029). Netherlands.

Wihersaari, M., (2005). "Greenhouse gas emissions from final harvest fuel chip production in Finland". Biomass Bioenerg. 28, 435-443.

Winkler, N., (1998). "A manual for the planning, design and construction of forest roads in steep terrain". Food and Agriculture Organization of the United Nations, Forest Harvesting Case Study 10. Available from: http://www.fao.org/3/a-w8297e/index.html. (Accessed Date: August 30, 2016).

Zhang, X., Kumar, A., Arnold, U., Sauer, J., (2014). "Biomass-derived Oxymethylene Ethers as Diesel Additives: A Thermodynamic Analysis". Energy Procedia 61, 1921-1924.

Zhang, X., Oyedun, A.O., Kumar, A., Oestreich, D., Arnold, U., Sauer, J., (2016). "An optimized process design for oxymethylene ether production from woody-biomass-derived syngas". Biomass Bioenerg. 90, 7-14.

Zhang, Y., McKechnie, J., Cormier, D., Lyng, R., Mabee, W., Ogino, A., Maclean, H.L., (2009). "Life cycle emissions and cost of producing electricity from coal, natural gas, and wood pellets in Ontario, Canada". Environ Sci Technol. 44, 538-544.

Chapter 3: A Life Cycle Sustainability Assessment (LCSA) of Oxymethylene Ether as a Diesel Additive Produced from Forest Biomass[2]

3.1 Introduction

There are increasing concerns with the use of fossil fuels in the transportation sector. In 2014, the energy industries accounted for around 60% of world's greenhouse gas (GHG) emissions, 24% of which were from transportation sector; production and use of fossil transportation fuels in vehicles (IEA 2016).

In Canada, transportation emissions are responsible for 25% of the country's total GHG emissions, and 76% of these are from road transportation (Mohamadabadi et al. 2009). Alberta, a province in western Canada, contributes a large share of these emissions. Alberta's conventional oil industries (including fuel refining and upgrading) emit 17% of the province's total GHGs (Kabir and Kumar 2011), a large share of which is from transportation fuels. To mitigate GHG emissions from the transportation sector, the commercialization of sustainable clean combustion fuels is a potential solution.

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Biomass is one of the best options to produce green liquid fuels. Several biomass-based fuels and fuel additives have been developed and their environmental impacts discussed in the literature, including diesohol (15% ethanol blended with diesel), E10 (10% ethanol blended with petrol), E85 (85% ethanol blended with 15% petrol), BD20 (biodiesel blended with 80% diesel, and pure biodiesel BD100 (Beer et al. 2001, Beer et al. 2002, Beer et al. 2003, Niven 2005, Beer and Grant 2007).

A renewable fuel solution, oxymethylene ether (OME) is an oxygenated fuel additive with a chemical formula of CH_3-O-(CH_2-O-) $_n$-CH_3. It is preferable over other alternatives because it can be produced from both fossils and renewables (i.e., biomass). It has similar chemical properties to diesel such as high viscosity, a large cetane number, and a high boiling point, which allow for great miscibility with conventional diesel (Pellegrini et al. 2013, Zhang et al. 2014). One of the most important benefits of OME is that it can reduce soot (black carbon) emissions significantly when used as a diesel additive (Pellegrini et al. 2013, Pellegrini et al. 2014). 100% OME as a transportation fuel can reduce black carbon emissions significantly (by up to 77%) from a diesel car engine without modifying the existing engine and without using any diesel particulate filter (Pellegrini et al. 2013).

Several sustainability assessment studies (i.e., environmental, economic, social and technical) have been done of renewable fuels. (Luk et al. 2010) conducted a comparative analysis to select a sustainable bioethanol refinery location for five different prairie sites in western Canada. The locations were analyzed and compared based on twelve criteria focusing on socioeconomic aspects, prairie resources, and support from policy-makers or government. (Sultana and Kumar

2012) developed a multi-criteria assessment model to compare five different biomass-based pellets to be used as an energy fuel in a power plant. The five alternatives were compared through thirteen qualitative and quantitative criteria covering environmental, economic, and technical aspects of sustainability. Mohamadabadi and colleagues compared transportation vehicles using conventional and biomass-derived fuels in terms of GHG emissions, fuel cost, vehicle cost, distance between fuel dispensing stations, and available number of vehicles (Mohamadabadi et al. 2009). (Kumar et al. 2006) compared sustainability impacts of different transportation vehicles that used gasoline, hybrid fuel gasoline-electric, E85 blend, fossil diesel, biodiesel and compressed natural gas. The most sustainable vehicle was selected based on the environmental, economic, and social impacts of the fuels. A multi-criteria decision model was also developed by (Kumar et al. 2006) to find the best agricultural biomass collection system among loafer/stacker, baling, and ensiling; and to rank the best biomass transportation system among rail, truck, and pipeline. In the studies cited above, different fuels, energy systems, and vehicles were analyzed based on multidimensional sustainability criteria. However, none of them addresses the sustainability of these fuels or energy systems throughout their life cycle.

A sustainability understanding of OME, an emerging alternative fuel technology, is needed but is limited, both in the literature and in industrial experience. Pellegrini et al. (2013) discussed environmental impacts from the combustion of different OME blends in diesel (such as 7.5%, 10%) and 100% OME and found that the particulate matter (soot) emissions can be reduced by 18% to 77% with different blends of OME with diesel (Pellegrini et al. 2013, Pellegrini et al. 2014). However, almost no studies were found in the literature on life cycle sustainability assessments of OME production and blending of OME with diesel. Before the technology can be commercialized, the environmental, economic, and social viability need to be evaluated, and this is a key challenge.

Different types of biomass are used to produce green energy, such as forest biomass, agricultural biomass, wood waste, energy crops, manures, etc. (McKendry 2002, Cherubini 2010, Thakur et al. 2014). Among them, the use of forest biomass to produce bioenergy is rapidly increasing due to the declining pulp and paper industry in Alberta (Kabir and Kumar 2012).

To achieve holistic and better decision making on sustainability, life cycle assessment (LCA) approaches are used (Ciroth et al. 2011). Though environmental LCAs have wide range of applications, life cycle costing (LCC) and social life cycle assessments (S-LCA) are not commonly used yet. However, because all sustainability assessments (environmental LCAs, LCCs, and S-LCAs) are built on the same ISO standard 14040 (2006), Walter Klöpffer suggested aggregating the three approaches into a single, holistic assessment, namely a life cycle sustainability assessment (LCSA) (Kloepffer 2008). Klöpffer and Renner referred to LCSA as a triple bottom line model, one in which the ISO environmental life cycle assessment is consolidated with economic and social assessments following a life cycle approach (Klöpffer and Renner 2007). The sustainability studies based on the energy sector mostly address a particular aspect of the energy system such as social or technical aspects (Afgan and da Graça Carvalho 2000, Carrera and Mack 2010) or focus on short-term impact assessments of energy systems (Afgan et al. 2000, Afgan and Darwish 2011). Afgan and Carvalho developed a multi-criteria sustainability assessment on energy systems (Afgan and da Graça Carvalho 2000). But their study was predominantly based on technical and social assessments, which lack the environmental and economic assessments. Dincer developed a LCSA model on hydrogen and fuel cell energy systems assessing the environmental, economic, social, and resource sustainability (Dincer 2007). However, the developed model was unable to reflect the social sustainability impacts of the considered energy systems. Elghali et al. proposed an LCSA framework for bioenergy production systems. The authors assessed the social indicators involving stakeholders from the relevant industries (Elghali et al. 2007). Similarly, Asseffa and Frostell (2007) used a community survey to assess the social indicators like knowledge, fear, and acceptance by society in the LCSA model (Assefa and Frostell 2007). The LCSA methodologies described in the literature are sometimes ambiguous and inconsistent, making it difficult to understand the practical implications of an LCSA, including all three dimensions of sustainability (Guinee 2016). The lack of case studies on the application of an LCSA framework is a great challenge in the field of sustainability assessments. In addition, there are few studies that compare the life cycle sustainability impacts of energy systems including all three dimensions of sustainability, namely environmental, economic, and social.

The objectives of this paper are to:

- investigate the life cycle environmental, economic, and social performance of OME production from two forest biomass feedstocks, whole tree and forest residue,
- propose a life cycle sustainability assessment framework to evaluate OME production sustainability based on nine criteria over the life cycle stages from biomass harvesting to combustion of the OME product;
- understand the sustainability of the OME production technology pathway; and
- conduct a case study for Alberta, a western province in Canada.

3.2 Method

This section presents a framework for a life cycle sustainability assessment (LCSA) of oxymethylene ether (OME) production and multi-criteria decision making in selecting the most sustainable pathway from different feedstocks (see Figure 3.1). Environmental, economic, and social assessments in this study are based on a functional unit of 1 MJ of produced OME Hence, the LCA results can be compared with the results of other LCA studies independent of any factors like geographic, demographic, road situation, etc. Although OME is a transportation fuel, functional units like person-km or ton-km were not used for OME in this study. Rather, an FU of 1 MJ of heat produced from OME was used so that it can be compared with any form of transportation energy produced and used in any country around the world.

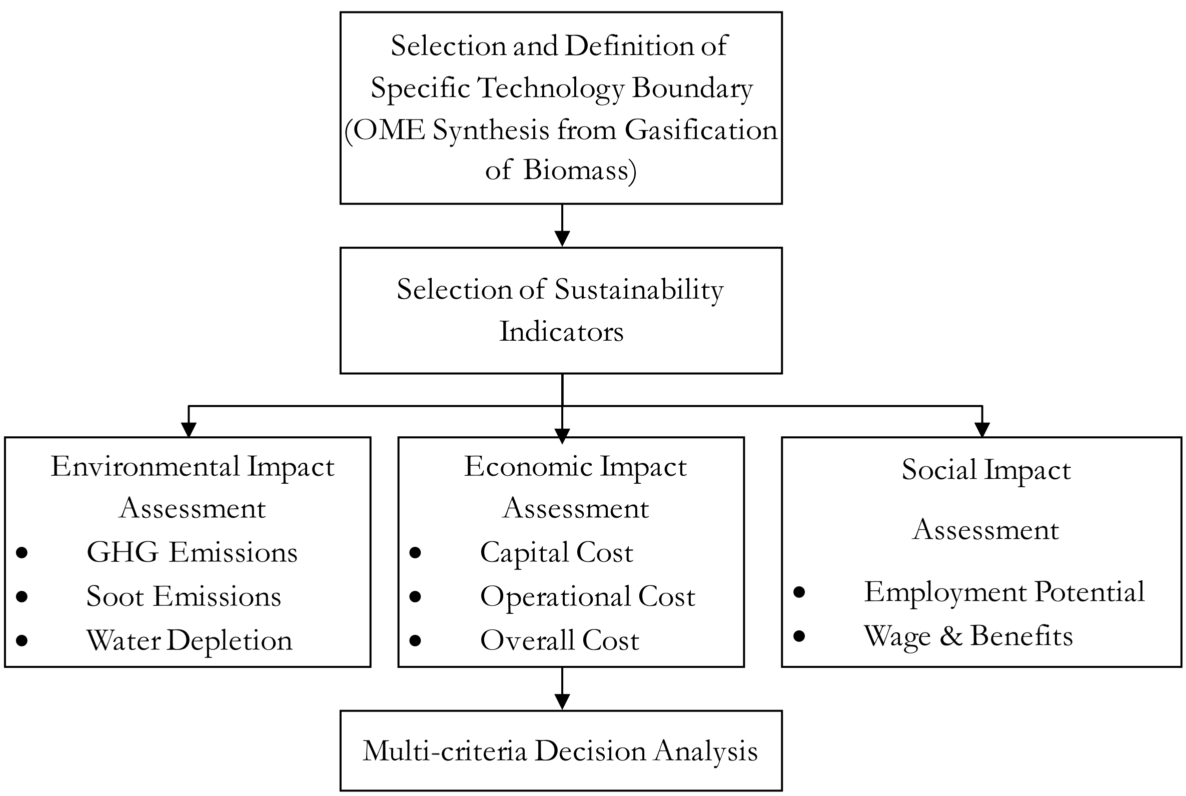


Figure 3. 1: Life cycle sustainability framework for OME production from forest biomass

3.2.1 System Boundary Selection and Definition of the Base Case

Defining the system boundary is the basis for conducting a life cycle sustainability assessment. The OME production life cycle system boundary includes unit operations such as forest biomass growth, harvesting, biomass transportation to the plant, chemical conversion within the plant, fuel transportation to blending, vehicle combustion, and disposal of material. These unit operations are identified based on current practices and existing literature (Mahbub et al. 2017). Figure 3.2a and b show the life cycle system boundary for OME production from whole tree and forest residue, respectively; the arrows represent inputs/outputs to the unit operations.

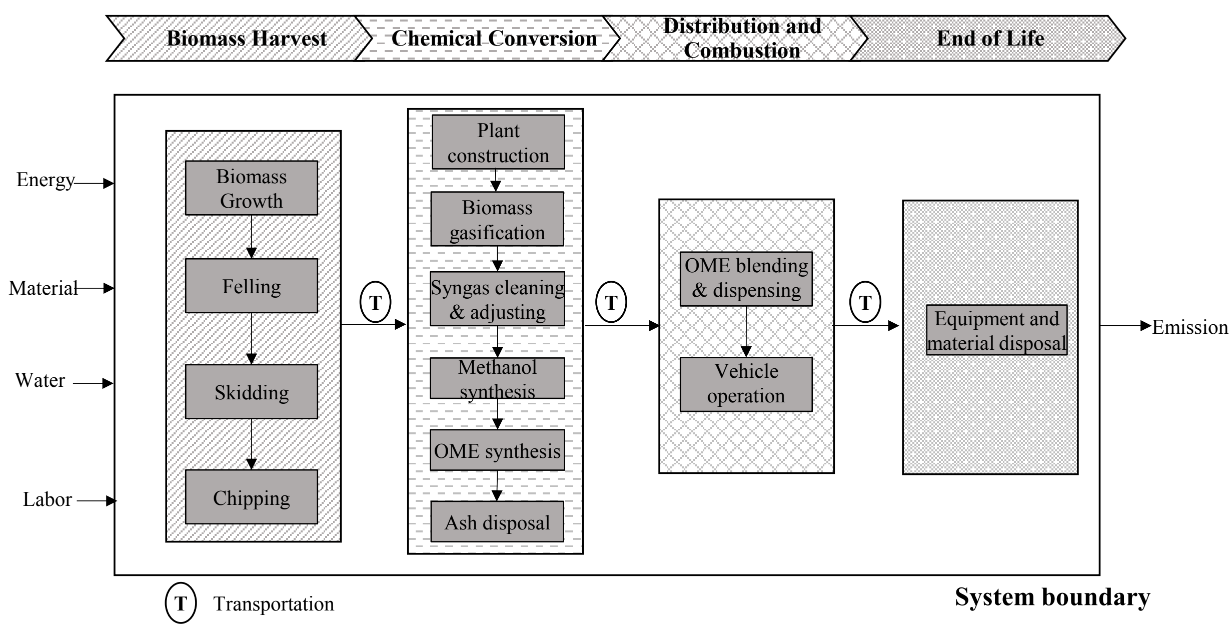


Figure 3. 2a: Life cycle system boundary for OME production from whole tree

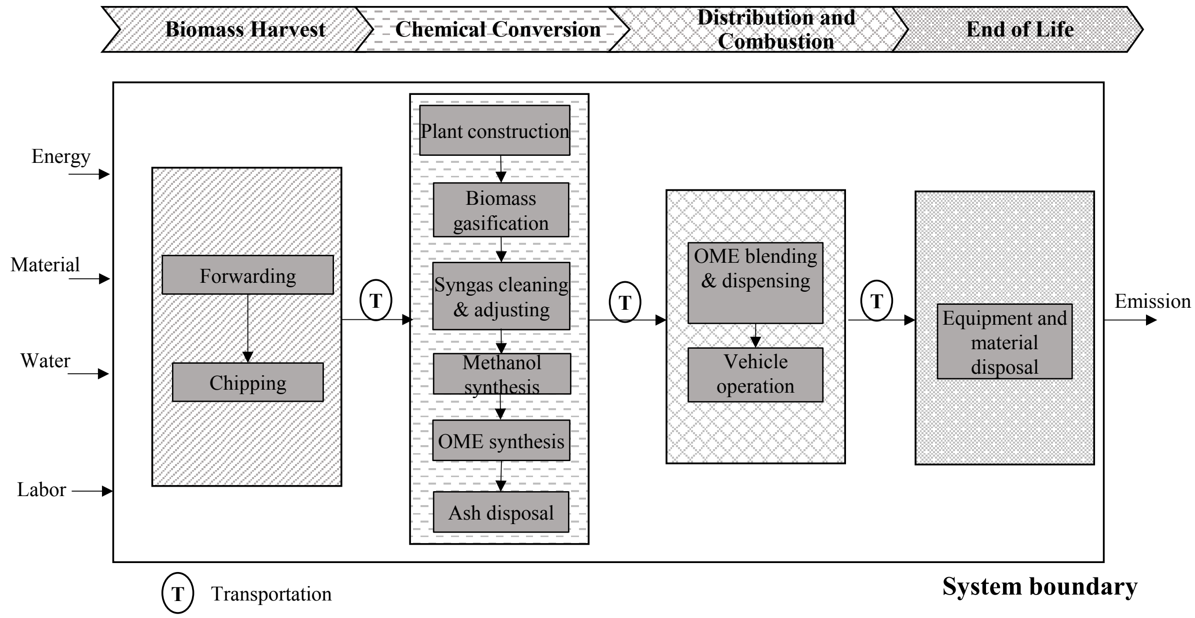


Figure 3. 2b: Life cycle system boundary for OME production from forest residue

Yield of 84 dry tonnes per hectare of whole tree (both hardwood and softwood) and 0.0247 dry tonne per hectare of forest residue harvested over a 100 year of rotation were considered in this study (Mahbub et al. 2017). The whole tree pathway includes silviculture. Silviculture involves the application of fertilizer and pesticides for biomass growth and nutrient replacement (Smith et al. 1997, Kumar et al. 2003, Kabir and Kumar 2011). In Alberta, to harvest whole trees, stands are cut and skidded to the roadside. Then trees are delimbed and the stems are chipped on the roadside into chips that is transported to the plant (Kabir and Kumar 2011, Kabir and Kumar 2012, Thakur et al. 2014). Thus, the whole tree harvesting operation includes feeling, skidding, and chipping. Forest residues, consisting of tops, limbs, and branches, which are generated from the logging operations, are forwarded to the road, chipped, and then transported to the plant (Kabir and Kumar 2012). Forest residues are assumed to be readily available and exclusive to whole tree production. It is assumed that forest residues are available in the forest even though whole trees are not used for energy production. Hence, the energy and emission impacts from the biomass growth stage are assumed to be zero in the forest residue pathway. In Alberta, if forest residues are not used they are usually burned in order to prevent forest fires (Shahrukh et al. 2016), and burning them leads to a significant amount of environmental pollution. in part to avoid the climate change impacts of forest fires and in part due to the declining pulp and paper industries in Alberta, there is a move toward alternative uses of forest biomass in the Western provinces in Canada, such as bioenergy production (Government-of-Canada 2016, Mahbub et al. 2017). It is assumed in this study that the ash produced during the chemical conversion process is disposed of in the forest and used as a fertilizer to increase the tree growth rate and carbon sequestration potential. The ash balances the soil organic carbon stock that is reduced through biomass removal and its impact on climate change (Kabir and Kumar 2011, Mahbub et al. 2017). The depletion of soil organic carbon (SOC) and the carbon stock reduction due to forest residue use are not considered in this study, since in both cases ash is returned to the forest (Kabir and Kumar 2012).

It is assumed that sustainable management practices are followed when whole trees are used to produce energy, just as when whole trees are used in the paper, pulp, and lumber industries (Kabir and Kumar 2012). This study also includes operations such as OME plant construction, road construction, equipment manufacturing, operating the equipment, natural resource extraction, and equipment fuel consumption and disposal (Figure 3.2). Table 3.1 shows the life cycle inventory of the operations included in the whole tree and forest residue pathways. Detailed specifications of the equipment used in biomass harvesting, data related to their construction, and energy consumptions can be found in Appendix A1. Energy and emissions impact factors for road construction, plant construction, raw materials, and fuels considered in this study are given in Appendix A2. The base case assumes diesel is used as fuel for the biomass transportation operation. A scenario using OME as a transportation fuel will be discussed in the sensitivity analysis section.

Table 3. 1: Data inventory and assumptions for the whole tree and forest residue pathways

Unit Operations	Data Inventory	Units	Whole Tree	Forest Residue	References
Biomass Harvesting	Biomass yield	dry tonnes / ha	84	0.247	(Kumar et al. 2003)
	Higher heating value	MJ/ dry kg	20	20	(Kumar et al. 2003)
	Moisture content	wt.%	50	35	(Kumar et al. 2003)
	Annual biomass requirement	dry tonnes /year	166,667	166,667	calculated
	Harvest area	ha	2106	572,976	calculated
Biomass Transportation	Transportation distance	km	8.65	36.17	calculated
	Truck capacity	dry tonnes	8.75	10.73	(Kabir and

Unit Operations	Data Inventory	Units	Whole Tree	Forest Residue	References
		biomass			Kumar 2012)
	Truck life time	years	7.5	7.5	(Mann and Spath 1997)
	Truck number	N/A	20	45	calculated
	Bulk density of wet biomass	kg/m³	250	230	(Kabir and Kumar 2011)
	Payload of truck	tonnes	23	23	(Kabir and Kumar 2012)
	Diesel consumption by truck (empty/fully loaded)	L/km	0.24 /0.33	0.24 /0.33	(Kabir and Kumar 2012)
	Actual load carried by truck (WT)	tonnes	17.5	16.5	(Kabir and Kumar 2012)
	Actual fuel consumption by	L	0.31	0.30	calculated

Unit Operations	Data Inventory	Units	Whole Tree	Forest Residue	References
	truck				
	Primary rad construction required in 20 years	km	69.2	N/A	calculated
	Secondary road construction required in 20 years	km	82	N/A	calculated
	Tertiary road construction required in 20 years	km	54	N/A	calculated
	Truck capacity (by volume)	m^3	70	70	(Kabir and Kumar 2012)
	Steel used in truck	tonnes /truck	14.7	14.7	(Kabir and Kumar 2012)
Chemical	Capacity of	dry tonnes	500	500	calculated

Unit Operations	Data Inventory	Units	Whole Tree	Forest Residue	References
Conversion	gasifier	/day			
	Plant life	years	20	20	
	Capacity factor				
	Year 1	N/A	0.7	0.7	(Kumar et al. 2003, Kabir and Kumar 2012)
	Year 2	N/A	0.8	0.8	
	Year 3 & onwards	N/A	0.85	0.85	
	Ash percentage	wt %	1	3	(Kumar et al. 2003)
	Ash spreader lifetime	hours	1200	1200	(Kabir and Kumar 2011)
	Ash spreader capacity	hectare /hour	4.41	4.41	

An OME plant is assumed to be located at the center of a circular biomass harvest area for which a geometric factor of 1 and a tortuosity factor of 1.27 are assumed to determine the average biomass collection distance (Sultana et al. 2010, Kabir and Kumar 2012, Mahbub et al. 2017).

High capacity trailer trucks (23 wet tonnes biomass) are used to transport biomass from the harvesting area to the chemical conversion plant and to transport ash from the plant to the landfill and OME from the plant to the retailer. Truck fuel consumption is calculated using equations from previous studies (Mann and Spath 1997, Kabir and Kumar 2012).

Based on a discussion on Alberta's forest road networks with subject matter expert Fulton Smyl (Business Analyst, Alberta Innovates-Technology Futures on June 28, 2016), there are three road types that are considered in this study for the whole tree pathway. These are: primary or permanent roads, secondary or semi-permanent, and tertiary or temporary roads (OMNR 1994). Around 69.2 km of primary roads, 82 km of secondary roads, and 54.3 km of tertiary roads are constructed prior to harvesting operations. The energy and emissions factors for primary road construction are taken from (Stripple 2001, Kabir and Kumar 2011, Kabir and Kumar 2012). It is assumed that crawler tractors are used for the construction of secondary and primary roads. The operating efficiency of crawler tractors during the construction of these roads is taken from (Winkler 1998). For the forest residue pathway, however, no road construction is required as forest residues are assumed to be readily available in forests (Kumar et al. 2003) and are harvested on existing logging roads.

The OME plant is assumed to have a 20-year production life with 8000 operating hours per year (Van Vliet et al. 2009, Kabir and Kumar 2011, Kabir and Kumar 2012). The chemical conversion includes plant construction, biomass gasification, syngas cleaning and adjusting, methanol production, OME production from syngas, and ash withdrawal. The energy and emissions impacts for plant construction, plant decommissioning, and construction material withdrawal are included in the system boundary. The method to estimate the amount of construction material, assumptions related to scale factors, plant decommissioning and construction material withdrawal are based on previous studies (Moore 1959, Sarkar and Kumar 2010, Sarkar and Kumar 2010, Kabir and Kumar 2011, Kabir and Kumar 2012, Mahbub et al. 2017). For an OME plant with a capacity of 500 dry tonnes of biomass /day (producing 97,701kg OME /day), around 2,618 tonnes of steel, 8,092 tonnes of concrete, and 22 tonnes of aluminum were estimated for construction. The method used to calculate the amount of plant construction material, plant decommissioning, and construction material withdrawal are directly taken from (Mahbub et al. 2017). It is assumed that the plant decommissioning impacts are 3% of plant construction impacts. Among the construction materials, 100% of the concrete and aluminum are landfilled, whereas 75% of the steel is assumed to be recycled and the rest landfilled (Spath and Mann 2000, Spath et al. 2005, Mahbub et al. 2017). At the plant, 500 dry tonnes of biomass /day are fed into the gasifier, where produced ash will be collected and dumped 50 km from the plant. The produced syngas is then cleaned and the tar reduced through thermal cracking and reforming (Li et al. 2004, Zhang et al. 2016). High hydrogen content in the syngas is required for high methanol yield and therefore the ratio of H_2 and CO in the syngas is adjusted using the water-gas shift (WGS) reaction and the conversion rate varied until the ratio is 2:1. Methanol is then synthesized from the adjusted syngas at a temperature of 300 ^{o}C and formaldehyde (FA) produced from methanol at a conversion rate of 60%. OMEs are then produced from methanol and FA using a continuous stirred-tank reactor (CSTR) reactor with a reactor volume of 1 L at a temperature of 60 ^{o}C and pressure of 1 bar through a series of reaction chains in the presence of the heterogeneous catalyst Dowex50Wx2, which is an acidic ion exchange resin (Zhang et al. 2016, Deutsch et al. 2017, Oestreich et al. 2017).

This study considers OME combustion to be carbon neutral, as is(Mahbub et al. 2017) commonly understood in biomass combustion assessments (Agbor et al. 2014, Shahrukh et al. 2015, Shahrukh et al. 2016). After OME is produced, it is assumed to be transported 300 km from the plant for blending.

3.2.2 Sustainability Indicators

Following a comprehensive review of published sustainability assessments and in discussion with the experts and decision makers, we selected eight indicators to assess environmental, economic, and social sustainability (Table 3.2).

Table 3. 2: Selected sustainability indicators

Environmental		**Economic**		**Social**	
Indicator	**Measurement**	**Indicator**	**Measurement**	**Indicator**	**Measurement**
GHG Emissions	Gram CO_2eq	Capital Cost	US dollar ($)	Employment Potential	Hours
Soot Emissions	Gram Soot	Operational Cost	US dollar ($)	Employee Wage & Benefit	US dollar ($)

Water Depletion	Liter H_2O	Overall Cost	US dollar ($)

Environmental indicators: Greenhouse gas (GHG) emissions, soot emissions, and water use (waster footprint) are used to assess environmental sustainability. GHG emissions are a universal environmental impact indicator used to assess the global warming potential of materials, processes, and systems (Sarkar and Kumar 2009). The measure of GHG emissions is carbon dioxide equivalent (CO_2eq) with a GWP conversion factor of 34 for methane and 298 for nitrous oxide (Myhre et al. 2013). In this study, GHG emission factors for energy and material use in unit operations were selected from several studies (Pellegrini et al. 2013); (Rahman et al. 2015). Soot (or black carbon) emissions are generated from transportation fuel combustion and are considered an air pollutant (Bond et al. 2013). The amount of soot in particulate matter (PM) emissions from OME combustion is estimated to be 33% and from conventional diesel 77% (Pellegrini et al. 2013). The combustion emissions (both the GHG and soot emissions) for 100% OME were taken from experimental results by (Pellegrini et al. 2013). Pellegrini et al. investigated the emission performance of an old light duty diesel engine Euro 2 car fueled with 100% OME and 100% diesel over the NEDC driving cycle and found that soot emissions from old vehicles can be reduced without any engine modification or using any diesel particulate filter. We have considered the average soot/GHG emissions for OME 1-8 in this analysis.

Water footprint as a measure of the total amount of freshwater consumed to produce a particular good or service is another important indicator in assessing life cycle sustainability (Dominguez-Faus et al. 2009, Hoekstra et al. 2011, Singh and Kumar 2011, Yang et al. 2011, Singh et al. 2014, Wong 2015) because water availability varies with region, weather, and plant location. In the OME production life cycle, water is consumed in processes such as biomass growing (Wong et al. 2016). The chemical conversion (OME synthesis) process, however, does not require any additional water because the steam used for syngas cleaning can be recovered from the moisture content of biomass during drying. In addition, water use in ash disposal and plant construction, subunit operations of chemical conversion, is so negligible water use in chemical conversion is considered to be zero (Singh and Kumar 2011, Singh et al. 2014).

Two aspects of water consumption are considered in the study, direct and indirect water use. Direct water use refers to the water required for biomass growth (Singh and Kumar 2011, Wong 2015) and indirect water use refers to water required in the production of energy inputs to the system such as diesel (Singh and Kumar 2011, Wong 2015). The average annual precipitation (rainfall) in the Western province of Alberta, Canada (480 mm/year), time required to harvest forest biomass (100 years of rotation is required for whole tree harvest whereas forest residues are harvested every year (Kumar et al. 2003)), and biomass yield extracted from (Wong et al. 2016) are used to calculate the water use factor for biomass growth and are 5714.3 L H_2O / kg dry wood for the whole tree and 3886.6 L H_2O / kg dry wood for the forest residue pathway (Wong et al. 2016). The equation to calculate the water use factor for biomass growth is illustrated in Appendix B. Water is also used in diesel production processes such as extraction and refining. A water use factor of 2.2 L H_2O / L diesel (King and Webber 2008) is considered in this study. Thus, the amount of water required in unit operations like biomass growth, biomass harvesting, biomass transportation, and road construction is estimated by using the water use factor and the amount of material (or biomass) used (or harvested) in the operations.

Economic indicators: Economical sustainability is measured through three cost indicators, capital cost, operational cost, and overall cost. In general, overall cost is the sum of capital cost and operational cost. The capital cost is an indicator of the competitiveness of a company or an investment on capital markets and is the base for calculating the equity yield rate. Potential investors use the information provided by the capital cost to determine if the technology yield rate compete with an alternative. The operating cost is an implicit indicator of short-term market risks. It comprises all costs that depend on short-term up-stream market developments, e.g., raw material costs. The overall cost is an indicator of the general competitiveness of a product as well as its long-term sustainability. Cost indicators are calculated based on available data and process modelling. The biomass delivery cost, which refers to the total cost of delivering biomass to the OME plant, is the sum of the biomass point of origin cost and transportation cost. The point of origin costs include biomass harvesting costs (i.e., costs of felling, chipping, forwarding, skidding, etc.), biomass field costs (royalties paid to the crown), nutrient replacement costs, silviculture costs, and road construction costs. The biomass transportation costs include the costs of loading and unloading the biomass feedstock and transporting the biomass from the forest/field to the OME plant (Kumar et al. 2003). Road construction cost is not considered for forest residues since they are transported on existing roads used for logging operation. Likewise, silviculture cost is not considered for forest residues since they are assumed to be available in the forests. Capital costs consist of costs for the construction and installation of the OME plant. The costs are estimated over a 20-year plant lifetime. Developed process models using Aspen software (Aspen-Icarus 2014) were used to estimate the capital costs. The equipment is mapped and sized before costs are estimated. Before the costs were analyzed in the process model, the mass and energy balance for each piece of equipment used in OME production in process model were calculated. An overall installation factor of 3.02 is used for all the purchased equipment, as suggested in the literature (Peters et al. 2003). The total purchased equipment costs (TPEC) are estimated from the process model and the total installed cost (TIC) calculated after factoring the installation factor of 3.02 (Peters et al. 2003, Swanson et al. 2010). The indirect costs (IC) are estimated as 89% of TPEC (construction expenses [34% of TPEC], engineering and supervision [32% of TPEC] and legal and contractors' fees [23% of TPEC] (Peters et al. 2003, Kumar et al. 2017). The total direct and indirect cost (TDIC) is the sum of TIC and IC. The project contingency is calculated as 20% of TDIC. The capital cost for the whole tree pathway includes an extra 5% of the other costs to account for camping costs (Kumar et al. 2003). All costs are given in US dollars ($) and based on the year 2016. The conversion rate of US$ to Canadian dollars (C$) is considered to be 0.7459 based on the Bank of Canada's rates on March 3, 2016.

Operational costs refer to raw material cost, maintenance cost, utilities (e.g., electricity) cost, plant overhead cost, operating charges, operating employee wage & benefit, and general and administrative (G&A) cost. Plant overhead is considered to be 50% of the total operating labor and maintenance costs and consists of costs during production for services, facilities, and payroll. Operating charges are 25% of the operating labor costs, and the general and administrative expenses (G&A) are specified as 8% of the total operating costs. The G&A costs are the costs incurred during production such as administrative salaries/expenses, research and development, product distribution, and sales costs. A discounted cash flow analysis model was developed to estimate the unit price of OME based on a 10% IRR on investment over 20 years of plant life. While 10% IRR was considered in this study, a sensitivity analysis (in a techno-economic study) was also done to see the impacts of IRR on the unit price of OME (Oyedun et al. 2016).

The life cycle costs of diesel include the costs of both oil extraction and refining diesel from fossil oil. The cost of refining diesel is assumed to be 30% more than the oil price (Van Vliet et al. 2009). King and Weber have assumed a price of $44.75/barrel for petroleum oil, thus the cost of refining fossil diesel from petroleum oil is $58.18/barrel (Van Vliet et al. 2009). That value was used in this study.

Social Indicators: Employment potential and employee wages and benefits are used to assess social sustainability. The social indicators in this study have been decided and developed according to the S-LCA guidelines and the methodological sheets established by the UNEP/SETAC Life Cycle Initiative (Andrews 2009, Benoît Norris et al. 2013). The methodological sheets are the latest version of detailed guidelines on how to design and apply the S-LCA in practical situations (Benoît Norris et al. 2013). The S-LCA guideline proposed by UNEP/ SETAC is in line with the ISO 14040 and 14044 (Andrews 2009). The UNEP/SETAC guidelines address the social impacts by using different steps, which includes defining the stakeholder category, impact category, sub-categories, and indicators, and collecting data. The sub-categories reflect the social or socio-economic attributes impacting a particular stakeholder category and needs to be addressed using different social indicators (Andrews 2009, Benoît Norris et al. 2013). In the developed LCSA framework, two social indicators have been used to calculate the social impacts, namely, employment potential or job creation and employee wages and benefits. Employment potential falls under the subcategory "local employment," which determines the potential for job creation as a result of introducing a new technology in the local community, and "local community" is the stakeholder that is socially impacted (Andrews 2009, Benoît Norris et al. 2013). Similarly, employee wages and benefits fall under the sub-category "fair salary," which impacts the stakeholder category "worker" according to the UNEP/SETAC guideline (Andrews 2009, Benoît Norris et al. 2013).

Employment potential is considered a relevant social impact assessment indicator because the newly emerging OME production technology can affect local employment both directly and indirectly (Benoît Norris et al. 2013). In this study, employment potential for a particular unit operation can be assessed by dividing a ratio of operation time by the biomass volume (m^3) involved in the operation (Valente et al. 2011). Employment potential is assessed for biomass harvesting, biomass transportation, chemical conversion, and OME transportation.

Wages and benefits are widely used in corporate social responsibility assessments because income is employees' primary concern and directly affects their well-being (Benoît Norris et al. 2013). In this study, employee wages and benefits for chemical conversion are determined based on the required working skill, plant scale, and typical employee wage in similar plants. For the harvesting and transportation operations, employee wages were calculated based on hours of operation and hourly labor rates (details in Appendix B).

3.2.3 Multi-Criteria Decision Analysis

PROMETHEE was used to compare the sustainability of the two OME production pathways (Brans and Vincke 1985, Brans et al. 1986, Brans and Mareschal 2005). PROMETHEE is one of the most commonly used alternative ranking methods for a wide range of applications, including energy systems (Kumar et al. 2006, Mohamadabadi et al. 2009, Behzadian et al. 2010, Luk et al. 2010, Sultana and Kumar 2012, Zhang and Haapala 2014). PROMETHEE compares different alternatives based on both quantitative and qualitative criteria, and its application and interpretation of results can be easily understood by decision-makers (Sultana and Kumar 2012).

In this analysis, alternatives are compared through several criteria and the alternative with the higher preference is selected as a preferred solution. This work studied two pathways of OME production, whole tree (WT) and forest residue (FR). The variable i denotes the criterion of the pathways, as in WT_i and FR_i. If the objective of a criterion is to maximize its value, the pathway with the higher criterion value is preferred over others, and vice versa. In this study, all the environmental, economic, and social indicators are minimized except employee wages and benefits and employment potential.

Step 1: Define Preference Function

The two pathways are first compared by criterion (indicator), and the difference between the estimates of the two pathways on a specific indicator is converted to a degree of preference quantified from 0 to 1 (0 being not preferred at all and 1 being strictly preferred) by using a preference function (Fülöp 2005, Mohamadabadi et al. 2009). For example, Equation 1 shows the preference function of the whole tree pathway (WT) over the forest residue pathway (FR) on a particular criterion *i* as

$$P_i(WT,FR)=p_i\,(WT_i\text{-}FR_i) \tag{1}$$

where p_i is a non-decreasing function, and $p_i\,(WT_i\text{-}FR_i) = 0$ when $(WT_i\text{-}FR_i)\leq 0$ and $0 \leq p_i\,(WT_i\text{-}FR_i)\leq 1$ when $(WT_i\text{-}FR_i)>0$

Usual and linear preference functions are used in this study. For usual preference functions, indifference occurs when the deviation between the evaluations of the two pathways on a specific indicator is 0 (the evaluations are equal). When the deviation is not 0, the pathway with a higher value is strictly preferred over the lower value one (Brans and Vincke 1985). No threshold is required for the usual preference function. Linear preference functions require two threshold types, indifference (Q) and preference thresholds (P), to make a preference decision. The indifference threshold (Q) for a specific indicator is determined by the largest difference between the estimates of the two pathways on that indicator. The pathways have no preference over one another below Q. The preference threshold (P) is determined by the smallest deviation between the estimates of the two pathways, above which the alternatives have strict preference over one another (Mohamadabadi et al. 2009, Sultana and Kumar 2012). In linear preference, indifference occurs when the deviation between evaluations exceeds the indifference threshold, and above this, the threshold preference increases progressively until the deviation equals the sum of the indifference and preference thresholds (Brans and Vincke 1985). Detailed mathematical equations of preference functions are given in Appendix B. The preference and indifference thresholds are usually determined based on the decision-maker's assumed choices. In this study, preference and indifference thresholds are assumed to be 10% and 5% of average estimates, respectively, based on literature reviews (Kumar et al. 2006, Mohamadabadi et al. 2009); (Sultana and Kumar 2012).

Step 2: Weighing the Indicators & Multi-criteria Preference Index

Weights are assigned to the criteria based on their relative importance in the decision-making process (Mohamadabadi et al. 2009, Luk et al. 2010, Sultana and Kumar 2012). Weight is usually decided by the decision-maker's preference for a criterion and the contribution of the criterion towards sustainability (Mohamadabadi et al. 2009, Luk et al. 2010, Sultana and Kumar 2012). Each alternative is compared pairwise with other alternatives and the weighted sum of the preference functions is calculated. This weighted sum is known as the multi-criteria preference index (Mohamadabadi et al. 2009) and is a value between 0 and 1, indicating the preference of one alternative over the others considering all the weighted criteria (indicators). For example, the multi-criteria preference index for WT over FR is defined in Equation 2 as:

$$\pi(\mathrm{WT,FR}) = \sum_{i=1}^{m} w_i\ P_i(\mathrm{WT,FR}) \tag{2}$$

where $w_i>0$ is a normalized weight assigned to criterion i and m is the number of indicators; m=9.

Step 3: Partial and Complete Ranking of Alternatives

Two types of outranking flows are calculated to rank the alternative pathways: positive outranking flow (leaving flow) and negative outranking flow (entering flow). For a particular pathway, these flows are calculated using the multi-criteria preference index (Luk et al. 2010). The positive outranking flow $\emptyset^{+}(WT)$ determines how much the WT pathway outranks or dominates the other pathway (FR). A higher $\emptyset^{+}(WT)$ value indicates that WT is more favorable than FR. The calculation for positive outranking flow is given by Equation 3 (Fülöp 2005):

$$\emptyset^{+}(\mathrm{WT})=\frac{1}{\mathrm{n}-1}\sum_{\mathrm{k}=1}^{\mathrm{n}} \pi\ (\mathrm{WT},\mathrm{FR}) \tag{3}$$

where n is the number of pathways and for this study n=2.

The negative outranking flow $\emptyset^{-}(WT)$ shows how much the WT pathway is outranked or dominated by the other pathway (FR). A lower $\emptyset^{-}(WT)$ value indicates a more favorable selection. Equation 4 shows the calculation for a negative outranking flow.

$$\emptyset^{-}(\mathrm{WT})=\frac{1}{\mathrm{n}-1}\sum_{\mathrm{k}=1}^{\mathrm{n}} \pi\ (\mathrm{WT},\mathrm{FR}) \tag{4}$$

Both the PROMETHEE I partial ranking and the PROMETHEE II complete ranking were conducted to rank the alternatives. In the PROMETHEE I partial ranking, the WT is preferred over the FR pathway if $\emptyset^{+}(WT) \geqslant \emptyset^{+}(FR)$, $\emptyset^{-}(WT) \leqslant \emptyset^{-}(FR)$, and one of them is a strict inequality. The WT and FR pathways are indifferent if $\emptyset^{+}(WT)=\emptyset^{+}(FR)$ and $\emptyset^{-}(WT)= \emptyset^{-}(FR)$. Otherwise the WT and FR pathways are incomparable (Fülöp 2005). In the PROMETHEE II complete ranking, the net outranking flows $\emptyset(WT)$ and $\emptyset(FR)$ are determined by adding the respective positive and negative outranking flows given by Equations 5 and 6. The net outranking flow determines the final preference of the two alternatives (Fülöp 2005).

$$\emptyset(\mathrm{WT})=\emptyset^{+}(\mathrm{WT})-\emptyset^{-}(\mathrm{WT}) \tag{5}$$

$$\emptyset(\mathrm{FR})=\emptyset^{+}(\mathrm{FR})-\emptyset^{-}(\mathrm{FR}) \tag{6}$$

If $\emptyset(WT) > \emptyset(FR)$, WT is preferred to the FR pathway and the pathways are indifferent if $\emptyset(WT) = \emptyset(FR)$. The pathway with the largest net outranking flow value *($\emptyset$)* is considered to be the best sustainable pathway.

3.3 Results & Discussion

The results are discussed in two sections. We developed a base case to select the most sustainable pathway of OME production from two types of biomass, and, with the base case results, we developed a scenario in order to select the most sustainable OME-diesel blend ratio from the preferred pathway. Sections 3.1-3.3 present the assessment results and section 3.4 presents the multi-criteria decision analysis for the two cases.

3.3.1 Environmental Impact Assessment

Table 3.3 presents the environmental impact assessment results for the two pathways. The emission values are given in the unit of per MJ heat produced from OME. Around 80% of total GHG emissions were found to come from vehicle operation (combustion of OME in vehicles) for both pathways (shown in parentheses in Table 3.3). However, biomass combustion in vehicles and in chemical conversion is considered to be carbon neutral because the amount of CO_2 released during combustion is compensated by the amount of CO_2 taken by the tree during its growth (Chum and Overend 2001, Sultana and Kumar 2011, Mahbub et al. 2017). Hence, the GHG emissions from vehicle operation (combustion of OME in vehicles) for both pathways are shown as 0 in Table 3.3. Here it is worth mentioning that the carbon and climate neutrality of bioenergy production from forest residue is beyond the above-mentioned simplified assumption. The carbon stock capacity of the residue and the temporal dynamics of the emissions and their consequent climate change effect are important aspects that need to be considered in the assumptions. Forest residues normally act as carbon stock, and harvesting them and using them as a source of energy release CO_2 emissions that would otherwise have been stored for a long time, depending on their decomposition rate. The potential climate change effect due to forest biomass removal should be compensated by increasing tree growth rate and carbon sequestration.

Table 3. 3: Environmental impacts of whole tree and forest residue pathways

Unit Operation	Pathway	Energy Consumption	GHG Emissions	Soot Emissions	Water Depletion
		GJ /MJ	gCO_2eq/MJ	gm/MJ	L H_2O/MJ
Biomass Growth	Whole Tree	0.001	0.22	0*	1238
	Forest Residue**	0**	0**	0**	842
Biomass Harvest	Whole Tree	0.17	13.01	0.001	0.008
	Forest Residue	0.09	7.13	0.001	0.004
Biomass	Whole Tree	0.017	2.96	0.0002	0.001

Unit Operation	Pathway	Energy Consumption	GHG Emissions	Soot Emissions	Water Depletion
		GJ /MJ	gCO_2eq/MJ	gm/MJ	L H_2O/MJ
Transportation	Forest Residue	0.022	1.69	0.0002	0.001
Chemical Conversion	Whole Tree	1.04	4.02	0*	0*
	Forest Residue	1.04	4.06	0*	0*
OME Transportation	Whole Tree	0.01	0.49	0*	0.0003
	Forest Residue	0.01	0.49	0*	0.0003
Vehicle Operation	Whole Tree	N/A	0(89.55)***	0.0011	N/A
	Forest Residue	N/A	0(89.55)***	0.0011	N/A

* Impact values from these unit operations were found to be negligible and so assigned a value of zero.

** Forest residues are assumed to be readily available in the forests and are harvested on the logging roads; hence, the impact values of energy consumption, GHG emissions, and soot emissions from the biomass growth operation in the forest residue pathway are considered to be zero.

***Combustion emissions from vehicle operations are considered to be carbon neutral or zero as the CO_2 emitted during combustion of OME is same as taken up by the plants during its growth.

Within the chemical conversion system, 4% (≈4 g CO_2eq/MJ) of total life cycle GHGs are emitted, and these mainly come from ash disposal and the use of a fossil source. A very small amount of natural gas (around 5.65% of total life cycle energy consumption) that is used during the chemical conversion process contributes to these emissions. Biomass transportation emissions are relatively low in the forest residue pathway (1.69 gCO_2eq/MJ) compared to the whole tree pathway (2.96 gCO_2eq/MJ) as there is no road construction involved in harvesting forest residues. The whole tree pathway has more energy-intensive harvesting unit operations, resulting in higher GHG emissions (13.01 gCO_2eq /MJ) than the forest residue pathway (7.13 gCO_2eq /MJ). Soot emissions from OME combustion in vehicles are the same for both pathways. However, total life cycle soot emissions are higher in the forest residue pathway (0.004 gm/ MJ) than the whole tree pathway (0.003 gm/MJ) due to the higher diesel requirement throughout in forest residue pathway. Soot emissions from OME transportation and chemical conversion for both pathways are negligible (Table 3.3). Water is primarily consumed in biomass growth (almost 99.99%) for both pathways. Water consumption is almost negligible in all other unit operations compared to water consumption in biomass growth (Table 3.3). Water consumption in biomass transportation for the forest residue pathway is 0.001 L H_2O/MJ, much higher than that of whole tree pathway, which uses only around 0.0003 L H_2O/MJ water for biomass collection and road construction. This is mainly due to the longer transportation distance for biomass collection in the forest residue pathway (36.17 km) compared to the whole tree pathway (8.65 km). Since the moisture content of biomass serves as a source of steam in the chemical conversion process, no extra water is needed (Zhang et al. 2016). Thus, water consumption in the chemical conversion process of OME from biomass mainly comes from water required for ash disposal and is almost negligible for both pathways.

3.3.2 Economic Impact Assessment

Table 3.4 lists the economic indicators along with all other cost components considered in this study for both pathways. The unit cost of producing OME from the whole tree pathway (1.92 $/Liter) is significantly higher than that of forest residue (1.71 $/Liter). The base year for all cost figures in this study is 2016 and the US$ is used. The initial moisture content of the biomass plays a significant role in the final unit cost of producing OME. Capital and operational costs are higher for the whole tree pathway than the forest residue pathway. Biomass yield has a large impact on biomass transportation cost since biomass yield is inversely proportional to harvest area, which is directly related to transportation distance. The higher the yield, the shorter the transportation distance, resulting in lower transportation cost. The higher yield of whole trees, around 84 dry tonnes/ hectare, compared to that of forest residue (0.247 dry tonne/ hectare), results in a biomass transportation cost of the forest residue pathway ($14.83 / dry tonne forest residue) that is significantly higher than that of the whole tree pathway ($11.10 / dry tonne whole tree). In spite of higher costs on the upstream side, due to the lower capital and operating costs, the forest residue pathway has a lower overall cost than the WT ($279.06 / dry tonne compared to around $310.25 / dry tonne).

Table 3. 4: Economic indicators for the whole tree and forest residue pathways

Cost Components	Units	Whole Tree	Forest Residue
Unit Cost of OME	$/ Liter	1.92	1.71
Capital Cost	$/dry tonne	55.30	43.65
Biomass Harvesting Cost	$/dry tonne	31.14	29.94
Biomass Transportation Cost	$/dry tonne	11.10	14.83
Silviculture Cost	$/dry tonne	1.75	N/A
Total Raw Materials Cost	$/dry tonne	45.16	47.26
Maintenance Cost	$/dry tonne	33.18	26.19
Utilities Cost	$/dry tonne	80.05	70.33
Plant Overhead	$/dry tonne	34.04	30.55
Operating Charges	$/dry tonne	8.73	8.73
Employee Wage & Benefit	$/dry tonne	62.58	67.26
G and A Cost	$/dry tonne	18.88	17.44
Total Operating Cost	$/dry tonne	254.95	235.41
Overall Cost	$/dry tonne	310.25	279.06

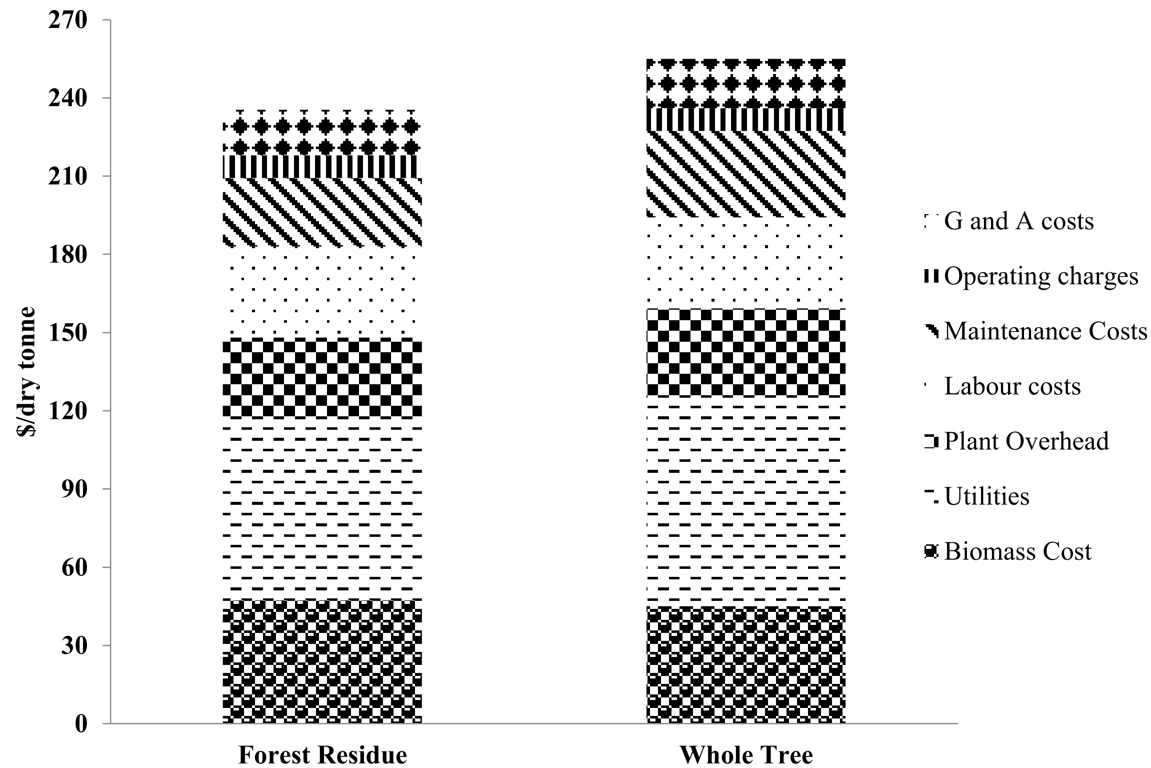


Figure 3. 3: Breakdown of operational costs for the whole tree and forest residue pathways

3.3.3 Social Impact Assessment

Table 3.5 shows the social impact assessment results of the two OME production pathways.

Table 3. 5: Social impact for whole tree and forest residue pathways

Unit Operations	Pathways	Employment Potential	Employee Wage & Benefit
		hours/m^3	$/ dry tonne
Biomass	Whole	0.04	8.17

Unit Operations	Pathways	Employment Potential	Employee Wage & Benefit
		hours/m^3	$/ dry tonne
Growth	Tree		
	Forest Residue	0.05	8.17
Biomass Harvest	Whole Tree	0.04	7.33
	Forest Residue	0.03	5.06
Biomass Transportation	Whole Tree	0.03	6.27
	Forest Residue	0.14	21.40
Chemical Conversion	Whole Tree	0.01	34.91
	Forest Residue	0.01	34.91

Unit Operations	Pathways	Employment Potential	Employee Wage & Benefit
		hours/m^3	$/ dry tonne
OME Transportation	Whole Tree	0.03	5.89
	Forest Residue	0.04	5.89
Vehicle Operation	Whole Tree	N/A	N/A
	Forest Residue	N/A	N/A

Employment potential is higher in the forest residue pathway (around 0.27 hours/ m^3 of woody biomass or 0.0004 hours/MJ of OME) than the whole tree (0.15 hours/ m^3 biomass or 0.0003 hours /MJ). Thus, the forest residue pathway leads to more jobs than the whole tree pathway. As for wages and benefits, from the employees' perspective, a higher number means a more secure life situation and a higher living standard. Employee wages and benefits overall are higher for the forest residue pathway ($67.26 / dry tonne) than the whole tree ($62.58 / dry tonne). Wages and benefits for each unit operation can be found in Table 3.5. The wages and benefits for the harvesting and transportation operations are estimated to be \$26.11/hour (Canada-Visa 2014), equivalent to the required skill level of the job.

3.3.4 Multi-criteria Decision Analysis for the Comparison of the Whole Tree and Forest Residue Pathways

The PROMETHEE outranking method is used to compare the alternatives and select the most sustainable pathway. A base case was developed as a starting point to compare the two pathways. Based on the selected pathway, a second case was developed to compare different ratios of OME as a diesel additive. Preference function selection is based on the decision-maker's judgement. This study uses a combination of linear and usual preference functions. For GHG emissions, water depletion, and soot emissions indicators, linear preference function are used because the deviations among these indicators can be sensitive to a decision-maker's judgment. For cost indicators and employment potential indicators, the usual preference function is used because lower cost and higher job creation potential are always preferred. In the base case, all indicators are given an equal weight. We conducted a separate sensitivity analysis to examine outcomes from different decision-making scenarios based on changes of weights. Table 3.6 shows the nine sustainability indicators, values, weights, and corresponding objectives to meet sustainability, preference functions, and respective threshold values for the chosen functions.

Table 3. 6: Preference functions, threshold values, objectives, and weights for selected sustainability indicators

Criteria	Unit	Obj *	Preference Function	WT*	FR*	Wt.	Pref. Threshold P	Indiff. Threshold Q
GHG Emissions	gCO_2eq/MJ	Min	Linear	20.70	13.38	0.125	1.70	0.85
Water Depletion	L H2O /MJ	Min	Linear	1238.05	842.06	0.125	104.01	52
Soot Emissions	gm/MJ	Min	Linear	3.72	4.21	0.125	0.25	0.13
Employment Potential	hours/m^3	Max	Usual	0.15	0.27	0.125	N/A	N/A
Employee Wage & Benefit	$/ dry tonne	Max	Usual	62.58	67.26	0.125	N/A	N/A
Capital	$/ dry	Min	Usual	55.30	43.65	0.125	N/A	N/A

Criteria	Unit	Obj*	Preference Function	WT*	FR*	Wt.	Pref. Threshold P	Indiff. Threshold Q
Cost	tonne							
Operating Cost	$/ dry tonne	Min	Usual	254.95	235.41	0.125	N/A	N/A
Overall Cost	$/ dry tonne	Min	Usual	310.25	279.06	0.125	N/A	N/A

*WT= Whole Tree, FR= Forest Residue, Obj= Objective, Min= Minimize, Max=Maximize

The alternative ranking is generated from Visual PROMETHEE software (Visual-PROMETHEE 2015). Figure 3.4 shows the base case ranking results in a GAIA (geometrical analysis for interactive assistance) plane where a preferred pathway is determined on the decision axis by using the position of the pathways and the orientation of the indicator axes towards the pathways. By comparing GHG emissions from both pathways, for example, we can see

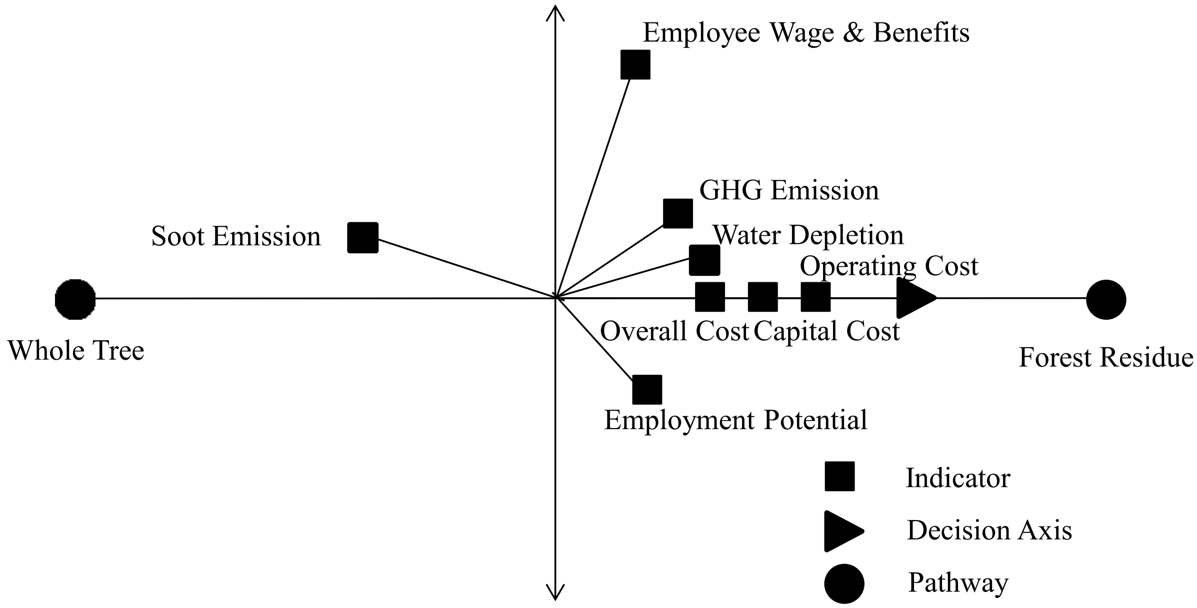


Figure 3. 4: PROMETHEE ranking results for the two pathways

that forest residue outranks whole tree, as the GHG emission indicator sign is located in the same direction as the decision axis. However, in figure 3.4, the soot emissions indicator axis is in the opposite direction of the decision axis (shown by the triangle bullet in Figure 3.4). This implies that all the indicators except soot emissions are in accordance with the obtained ranking (Mohamadabadi et al. 2009, Mareschal 2013). The position of the pathway relative to any indicator reflects the performance of the pathway on this particular indicator and the closer the distance, the better the performance (Mohamadabadi et al. 2009). Figure 3.4 shows that forest residue performs better than whole tree in all areas except soot emissions. Having seven indicators in its favor, the forest residue pathway outranks the whole tree pathway. It is worth noting that the decision axis does not represent the optimum solution; rather, it indicates the preferred compromise based on the assigned weights of the indicators (Mohamadabadi et al.

2009, Mareschal 2013). Thus, forest residue is the preferred pathway based on the current weights of the indicators.

Positive, negative, and net outranking flows for the two pathways are shown in Table 3.7. For the forest residue pathway, the positive outranking flow (0.88) is higher than that of the whole tree (0.13) and the negative outranking flow (0.13) is less than that of the whole tree (0.88). Hence, based on the PROMETHEE I partial ranking, the forest residue pathway outranks the whole tree pathway. Nevertheless, the net outranking flow is higher for the forest residue (0.75) than that of the whole tree (-0.75). Hence, the PROMETHEE II complete ranking also shows a preference for the forest residue pathway over the whole tree.

Table 3. 7: Ranking results for the whole tree and forest residue pathways

Pathways	$\emptyset^+$(positive flow)	$\emptyset^-$(negative flow)	$\emptyset$(net flow)
Whole Tree	0.13	0.88	-0.75
Forest Residue	0.88	0.13	0.75

3.3.5 Multi-criteria Decision Analysis for the Comparison of Different OME-Diesel Blending Ratios

Pellegrini et al. recommended that OME be used as a diesel additive in vehicles. They investigated the particulate matter (PM) emissions from different OME blends with fossil diesel and found that l8-77% emissions reduction is possible compared to neat diesel (Pellegrini et al. 2013, Pellegrini et al. 2014). Zhang et al. found that a blend of 20% forest biomass-based OME in 80% fossil diesel can decrease PM emissions by 50% from old used cars (Zhang et al. 2014). The life cycle GHG emissions and soot emissions of OME derived from forest biomass were investigated by Mahbub et al. (Mahbub et al. 2017). The authors compared the life cycle GHG and soot emissions of OME and diesel. Diesel life cycle emissions include emissions from crude extraction, crude refining, and the combustion of diesel in vehicles. Mahbub et al. found that 79-86% life cycle GHG emissions can be reduced using OME as a transportation fuel rather than diesel and life cycle soot emissions can be reduced by 89% compared to using 100% diesel. The authors also compared the performance of 10% OME blended with 90% diesel and found that life cycle GHG emissions can be reduced by up to 5.35% and the life cycle soot emissions can be reduced even more, 30% compared to neat diesel (Mahbub et al. 2017).

Therefore, in this study a second case was developed based on the selected forest residue pathway to examine the sustainability performance of different OME-diesel blend ratios. Ten different ratios of OME blends were compared in this study: 10% OME and 90% diesel, 20% OME and 80% diesel, 30% OME and 70% diesel, 40% OME and 60% diesel, 50% OME and 50% diesel, 60% OME and 40% diesel, 70% OME and 30% diesel, 80% OME and 20% diesel, 90% OME and 10% diesel, and 100% OME. The sustainability of each blend was examined with respect to four indicators: GHG emissions, soot emissions, water consumption, and overall cost. These indicators were selected because they are considered to be the main sustainability contributors. Table 3.8 shows the impact values of all the OME blends.

Table 3. 8: Sustainability impacts for different OME & diesel blends

Ratio of OME in Diesel	GHG Emissions	Water Consumption	Soot Emissions	Overall Cost
	gCO_2eq/MJ	L H_2O/MJ	gm/MJ	$/MJ
10%	120.36	1576.85	0.33	0.018
20%	113.53	3151.49	0.30	0.023
30%	105.92	4726.14	0.26	0.028
40%	97.42	6300.79	0.23	0.034
50%	87.84	7875.43	0.19	0.040
60%	76.96	9450.08	0.16	0.048
70%	64.51	11024.73	0.12	0.056
80%	50.13	12599.38	0.09	0.066
90%	33.31	14174.02	0.06	0.078
100%	13.38	15748.67	0.02	0.091

The environmental, economic, and social impacts for different OME blends are calculated on a volume basis. The GHG emissions from the two fuels, diesel and OME, are first converted to volume-based emissions (g CO_2eq/Liter of fuel). For example, GHG emissions from 90% diesel on a volume basis are added to the GHG emissions coming from 10% OME on a volume basis. Equal weights are assigned to each impact areas assessed: 50 % for environmental impact, and 50 % for economic impact. For the environmental impact assessment, the 50% weight is divided further: 16.67 % for GHG emissions, 16.67 % for soot emissions, and 16.67 % for water depletion. Table 3.9 shows the weights, preference functions, and thresholds for the indicators used to compare different OME blends.

Table 3. 9: Preference functions, objectives, weights, and thresholds used to compare OME blends

Criteria	Objective	Preference Function	Weights	Preference Threshold P	Indifference Threshold Q
GHG Emissions	Minimize	Linear	16.67	7.63	3.82
Water Consumption	Minimize	Linear	16.67	866.28	433.14
Soot Emissions	Minimize	Linear	16.67	0.02	0.01
Overall cost	Minimize	Usual	50	0.01	0.002

Table 3.10 shows the positive, negative, and net outranking flows for the ten OME-diesel blends. According to the PROMETHEE I partial ranking and the PROMETHEE II complete ranking, the preference is highest for a 10% OME blended with 90% diesel fuel. For GHG and soot emissions, the preference increases with an increase of OME in diesel.

Table 3. 10: Alternative ranking of different OME blends with diesel

Ranking of OME blends with diesel	$\emptyset^{+}$(positive flow)	$\emptyset^{-}$(negative flow)	$\emptyset$(net flow)
10%	0.67	0.33	0.34
20%	0.63	0.37	0.26
30%	0.59	0.41	0.18
40%	0.56	0.44	0.11
50%	0.52	0.48	0.04
60%	0.48	0.52	-0.04
70%	0.44	0.56	-0.11
80%	0.41	0.59	-0.19
90%	0.37	0.63	-0.26
100%	0.33	0.67	-0.33

However, for overall cost (economic impact indicator) and water depletion, the preference decreases with an increase of OME in diesel. Hence, with the indicator weights included, fuel blends with higher OME ratios are always less preferred over the lower ones, which also comply with the experimental results. Experimental results recommend that a maximum 10% of any oxygenated compound be added with petroleum-based fuels - in old vehicles with little or no engine alteration (Löfvenberg 2010, Pellegrini et al. 2013).

3.3.6 Sensitivity Analysis

There are a number of variables with uncertainties in the model. These include but are not limited to indicator weights (which are based on decision-maker's preferences), threshold values, and the calculated indicator's impact values. Therefore, we conducted a sensitivity analysis to examine the impact of these variables and to represent different scenarios of decision-makers' preferences.

3.3.6.1 Weight Sensitivity

A weight sensitivity analysis was conducted using the stability interval in Visual PROMETHEE. A stability interval determines the weight intervals for all the indicators, across which the ranking is not altered or the ranking remains stable (Genc 2014). If the stability interval is small on a particular indicator, the ranking becomes sensitive to the indicator's weight (Luk et al. 2010, Sultana and Kumar 2012) and even a small change beyond the interval can impact the ranking significantly. On the other hand, a large stability interval implies that the ranking is not affected by the change in weights within the interval (Safari et al. 2012, Sultana and Kumar 2012). For the base case, all the indicators except soot emissions have large stability intervals ranging from 0 to 100%. Soot emissions have a stability interval of 0 to 50%, which means that keeping all the other criteria equally weighted, the ranking will be altered (the whole tree pathway will be preferred) when the soot emission weight is assigned a value higher than 50%. Table 3.11 shows the weight stability intervals for the second case. The result shows that GHG and soot emissions have smaller sensitivity intervals. As an example, for GHG emissions, the preference rank will reverse (preference will increase with the rise of the OME ratio in the diesel blend) when the GHG emissions' weight is increased to over 36%. That means that if the decision-maker considers GHG emissions a key factor (with a value above 36%), the final rank will reverse.

Table 3. 11: Sensitivity analysis of weights for the OME-diesel blend case

Indicators	**Weights Assigned**	**Stability Interval**
GHG Emissions	16.67%	[0%, 36%]
Soot Emissions	16.67%	[0%, 37%]
Water Consumption	16.67%	[0%, 100%]
Overall Cost	50%	[26%, 100%]

3.3.6.2 Sensitivity of Environmental, Economic and Social Impacts

In order to study preference ranking for the sustainability factors (environmental, economic, and social), we developed three scenarios: environmental, economic, and social. In the environmental scenario, environmental impact is given a weight of 60% and the other two impacts 20% each. The other two scenarios are developed in the same way. Tables 3.12 and 3.13 show the sensitivity analysis results for the base case and the OME-diesel blend case, respectively. The ranking remains the same for the base case for all three scenarios. For the OME-diesel blend case, 80% of the weights are assigned to the preference scenario and the remaining 20% are assigned for the other scenario. As Table 3.13 shows, the ranking remains the same in the economic scenario. In the environmental scenario, however, the ranking pattern changes drastically, resulting in the 100% OME blend being the most preferred and the 10% OME the least. Hence, the environmental impact is sensitive to overall ranking in the OME-diesel blend case.

Table 3. 12: Sensitivity analysis of sustainability impacts for the base case

	Rank	Pathways	$\emptyset$(net flow)	$\emptyset^+$(positive flow)	$\emptyset^-$(negative flow)
Environmental Scenario	1	Forest Residue	0.60	0.80	0.20
	2	Whole Tree	-0.60	0.20	0.80

	Rank	Pathways	∅(net flow)	$\emptyset^{+}$(positive flow)	$\emptyset^{-}$(negative flow)
Social Scenario	1	Forest residue	0.87	0.93	0.07
	2	Whole tree	-0.87	0.07	0.93
Economic Scenario	1	Forest residue	0.87	0.93	0.07
	2	Whole tree	-0.87	0.07	0.93

Table 3. 13: Sensitivity analysis of sustainability impacts for the OME-diesel blend case

	Rank	OME Blends	∅(net flow)	$\emptyset^{+}$(positive flow)	$\emptyset^{-}$(negative flow)
Environmental Scenario	1	100% OME	0.07	0.53	0.47
	2	90% OME	0.05	0.53	0.47
	3	80% OME	0.04	0.52	0.48
	4	70% OME	0.02	0.51	0.49
	5	60% OME	0.01	0.50	0.50
	6	50% OME	-0.01	0.50	0.50

	Rank	OME Blends	Ø(net flow)	Ø⁺(positive flow)	Ø⁻(negative flow)
	7	40% OME	-0.02	0.49	0.51
	8	30% OME	-0.04	0.48	0.52
	9	20% OME	-0.05	0.47	0.52
	10	10% OME	-0.06	0.47	0.53
Economic Scenario	1	10% OME	0.73	0.87	0.13
	2	20% OME	0.57	0.78	0.21
	3	30% OME	0.41	0.70	0.30
	4	40% OME	0.24	0.62	0.38
	5	50% OME	0.08	0.54	0.46
	6	60% OME	-0.08	0.46	0.54
	7	70% OME	-0.24	0.38	0.62
	8	80% OME	-0.41	0.30	0.70
	9	90% OME	-0.57	0.21	0.79
	10	100% OME	-0.73	0.13	0.87

3.3.6.3 Sensitivity of Preference and Indifference Thresholds

The preference and indifference threshold values are varied within a range of ±10% and their impacts on the ranking of the alternatives are determined for both the base case and the OME-diesel blend case. Threshold values are changed in four ways: a 10% increase in both preference (P) and indifference (Q) thresholds, a 10% increase in the preference threshold and a 10% decrease in the indifference threshold, a 10% increase in the indifference threshold and a 10% decrease in the preference threshold, and, finally, a 10% decrease in both preference and indifference thresholds. In both the base case and the OME-diesel blend case, the ranking is not altered in any of the four above-mentioned scenarios (Table 3.14). Hence, it can be said that both the base case and OME-diesel blend case rankings are not sensitive to the assigned threshold values.

Table 3. 14: Sensitivity analysis rankings for the base case and the OME-diesel blend case

	Ranking of Base Case*		**Ranking of OME-Diesel Blend Case***									
Scenarios	WT	FR	10%	20%	30%	40%	50%	60%	70%	80%	90%	100%
+10% P*, Q*	2	1	10	9	8	7	6	5	4	3	2	1
+10%P -10% Q	2	1	10	9	8	7	6	5	4	3	2	1
-10% P	2	1	10	9	8	7	6	5	4	3	2	1

	Ranking of Base Case*		Ranking of OME-Diesel Blend Case*									
+10% Q -10% P*, Q*	2	1	10	9	8	7	6	5	4	3	2	1
+10% GHG Emission	2	1	10	9	8	7	6	5	4	3	2	1
-10% GHG Emission	2	1	10	9	8	7	6	5	4	3	2	1
+10% Soot Emission	2	1	10	9	8	7	6	5	4	3	2	1
-10% Soot Emission	2	1	10	9	8	7	6	5	4	3	2	1
+10% Water Depletion	2	1	10	9	8	7	6	5	4	3	2	1
+10% Water Depletion	2	1	10	9	8	7	6	5	4	3	2	1
+10% Employment Potential	2	1	10	9	8	7	6	5	4	3	2	1
-10% Employment	2	1	10	9	8	7	6	5	4	3	2	1

	Ranking of Base Case*		Ranking of OME-Diesel Blend Case*									
Potential +10%	2	1	10	9	8	7	6	5	4	3	2	1
Employee wage & benefit -10%	2	1	10	9	8	7	6	5	4	3	2	1
Employee wage & benefit +10%	2	1	10	9	8	7	6	5	4	3	2	1
Capital Cost -10%	2	1	10	9	8	7	6	5	4	3	2	1
Capital Cost +10%	2	1	10	9	8	7	6	5	4	3	2	1
Operational Cost -10%	2	1	10	9	8	7	6	5	4	3	2	1
Operational Cost +10%	2	1	10	9	8	7	6	5	4	3	2	1

	Ranking of Base Case*		**Ranking of OME-Diesel Blend Case***									
Overall Cost												
-10% Overall Cost	2	1	10	9	8	7	6	5	4	3	2	1

*WT= Whole Tree, FR= Forest Residue, OME-Diesel Blend Case = % of OME in Diesel, P= Preference Threshold, Q= Indifference Threshold

3.3.6.4 Impact Values of Indicators

As uncertainty may also exist in assessment impact values, we conducted a sensitivity analysis by changing the impact values by ±10% for each indicator. The impacts on overall ranking are determined for both cases (see Table 3.14). One indicator is changed at a time. We found that rankings are not sensitive to changes in any indicator, that is, the rankings remain the same after the changes.

3.3.6.5 OME for Biomass Transportation

We developed a scenario assuming OME was the transportation fuel for biomass collection from the harvest area. It is expected that when the truck delivers biomass to the OME plant, it will be refilled with OME, thus the truck does not use conventional fossil fuel for biomass transportation. Table 3.15 shows the impact values on transportation for this scenario. For the whole tree pathway, the changes are less than 2% for all the indicators. For the forest residue pathway, however, a 90%-95% reduction in GHG and soot emissions is possible if OME is used for biomass transportation instead of diesel. Water consumption for biomass transportation in the whole tree pathway is negligible (0.0000006 L H_2O / MJ of OME from road construction), whereas in the forest residue pathway water required for biomass transportation is 0, as no external water is required for OME production from biomass. Indicators such as employment potential, labor cost, capital cost, and operational cost are not affected in this scenario. However, transportation costs increase by 5% and 1.62% in the forest residue and the whole tree pathways, respectively. The higher increase in the forest residue pathway is mainly due to the longer biomass collection distance compared to the other pathway. Change in overall cost is almost negligible for both pathways. Overall costs increase by 0.3% in the forest residue pathway and 0.07% in the whole tree pathway. As a result, the ranking remains the same as it is in the base case: the forest residue is preferred over the whole tree pathway.

Table 3. 15: Indicator values for OME as a transportation fuel scenario

		Whole Tree		**Forest Residue**	
		Base Case Transp.	**OME as Transp. Fuel**	**Base Case Transp.**	**OME as Transp. Fuel**
Energy Consumption	GJ/MJ	0.017	0.013	0.022	0.010

		Whole Tree		Forest Residue	
		Base Case Transp.	**OME as Transp. Fuel**	**Base Case Transp.**	**OME as Transp . Fuel**
GHG Emissions	kg CO_2eq/MJ	2.96	2.49	1.69	0.109
Soot Emissions	gm/MJ	0.0002	0.0001	0.0002	0.00001
Water Depletion	L H_2O/MJ	0.0003	0.0000006	0.001	0
Employment Potential	hours/m^3	0.033	0.033	0.135	0.135
Labor Cost	$/MJ	0.002	0.002	0.007	0.007
Capital Cost	S/MJ	0.013	0.013	0.010	0.010
Operational Cost	S/MJ	0.061	0.061	0.056	0.056
Overall Cost	S/MJ	0.074	0.074	0.067	0.067

3.4 Conclusions

The use of alternate fuels or fuel additives instead of fossil sources can improve environmental, economic, and social performances globally. This study developed a life cycle sustainability assessment (LCSA) framework for oxymethylene ether (OME) production from two types of forest biomass, whole tree and forest residue, to be used as a diesel additive by integrating the environmental, economic, and social impact assessments. Through the multi-criteria decision analysis method, the forest residue pathway was found to be strongly preferred over the whole tree pathway for OME synthesis in all sustainability impacts considered. The whole tree pathway is less preferred, as its GHG emissions were significantly higher (20.69 gCO_2eq/MJ) than in the forest residue pathway (13.37 gCO_2eq/MJ) due energy-intensive road construction operations. All cost indicators are higher for the whole tree pathway, thus making it a less preferred pathway to produce OME from a cost perspective. From a social perspective, all the indicators also favor the forest residue pathway. A second case was developed to select the most sustainable blend of OME with diesel. Based on environmental, economic, and social assessments, a blend with a higher OME percentage is preferred. In this study, all the impacts were assigned equal weights for both cases. However, the sensitivity analysis of different model parameters (e.g., preference functions, threshold values, and weights) found that the variation in values is almost negligible. The developed LCSA framework can be used to assess different types of energy pathways to evaluate their environmental, economic and social viability before commercialization. The framework can also be used to rank different energy pathways, thus assisting policy-makers to develop energy sector policies that are environmentally, economically, and socially sustainable.

References

Afgan, N. and M. da Graça Carvalho (2000). "Sustainable Assessment Method for Energy Systems: indicators, criteria and decision making procedure", Springer Science & Business Media.

Afgan, N. H., M. G. Carvalho and N. V. Hovanov (2000). "Energy system assessment with sustainability indicators." Energy Policy **28**(9): 603-612.

Afgan, N. H. and M. Darwish (2011). "Multi-criteria sustainability assessment of water desalination and energy systems—Kuwait case." Desalination and Water Treatment **25**(1-3): 241-250.

Agbor, E., X. Zhang and A. Kumar (2014). "A review of biomass co-firing in North America." Renewable and Sustainable Energy Reviews **40**: 930-943.

Aspen-Icarus (2014). Aspen Process Economic Analyzer. Version 8.8. Cambridge, Aspen Technology Inc.

Assefa, G. and B. Frostell (2007). "Social sustainability and social acceptance in technology assessment: A case study of energy technologies." Technology in Society **29**(1): 63-78.

Beer, T. and T. Grant (2007). "Life-cycle analysis of emissions from fuel ethanol and blends in Australian heavy and light vehicles." Journal of Cleaner Production **15**(8): 833-837.

Beer, T., T. Grant, G. Morgan, J. Lapszewicz, P. Anyon, J. Edwards, P. Nelson, H. Watson and D. Williams (2001). Comparison of Transport Fuels: Final Report to the Australian Greenhouse

Office on the Stage 2 Study of Life-Cycle Emissions—Analysis of Alternative Fuels for Heavy Vehicles, Report No. EV45A/2/F3C, Commonwealth Scientific and Industrial Research Organisation, Aspendale, Victoria, Australia.

Beer, T., T. Grant, D. Williams and H. Watson (2002). "Fuel-cycle greenhouse gas emissions from alternative fuels in Australian heavy vehicles." Atmospheric Environment **36**(4): 753-763.

Beer, T., D. Olaru, M. Van der Schoot, T. Grant, B. Keating, S. Hatfield Dodds, C. Smith, M. Azzi, P. Potterton and D. Mitchell (2003). Appropriateness of a 350 Million Litre Biofuels Target: Report to the Australian Government Department of Industry, Tourism and Resources, CSIRO, ABARE, BTRE, Canberra.

Behzadian, M., R. B. Kazemzadeh, A. Albadvi and M. Aghdasi (2010). "PROMETHEE: A comprehensive literature review on methodologies and applications." European Journal of Operational Research **200**(1): 198-215.

Benoît Norris, C., M. Traverso, S. Valdivia, G. Vickery-Niederman, J. Franze, L. Azuero, A. Ciroth, B. Mazijn and D. Aulision (2013). "The Methodological Sheets for Sub-Categories in Social Life Cycle Assessment (S-LCA)." United Nations Environment Programme (UNEP) and Society for Environmental Toxicology and Chemiastry (SETAC).

Bond, T. C., S. J. Doherty, D. Fahey, P. Forster, T. Berntsen, B. DeAngelo, M. Flanner, S. Ghan, B. Kärcher and D. Koch (2013). "Bounding the role of black carbon in the climate system: A scientific assessment." Journal of Geophysical Research: Atmospheres **118**(11): 5380-5552.

Brans, J.-P. and B. Mareschal (2005). "PROMETHEE methods." Multiple criteria decision analysis: state of the art surveys: 163-186.

Brans, J.-P. and P. Vincke (1985). "Note—A Preference Ranking Organisation Method: (The PROMETHEE Method for Multiple Criteria Decision-Making)." Management science **31**(6): 647-656.

Brans, J.-P., P. Vincke and B. Mareschal (1986). "How to select and how to rank projects: The PROMETHEE method." European Journal of Operational Research **24**(2): 228-238.

Canada-Visa. (2014). "Canada Salary Wizard." Retrieved January 25 2015, from http://www.canadavisa.com/canada-salary-wizard.html.

Carrera, D. G. and A. Mack (2010). "Sustainability assessment of energy technologies via social indicators: Results of a survey among European energy experts." Energy Policy **38**(2): 1030-1039.

Cherubini, F. (2010). "The biorefinery concept: using biomass instead of oil for producing energy and chemicals." Energy Conversion and Management **51**(7): 1412-1421.

Chum, H. L. and R. P. Overend (2001). "Biomass and renewable fuels." Fuel Processing Technology **71**(1): 187-195.

Ciroth, A., M. Finkbeier, J. Hildenbrand, W. Klöpffer, B. Mazijn and S. Prakash (2011) "Towards a life cycle sustainability assessment: Making informed choices on products." http://wedocs.unep.org/bitstream/handle/20.500.11822/8001/UNEP_LifecycleInit_Dec_FINAL.pdf?sequence=3&isAllowed=y Accessed.

Deutsch, D., D. Oestreich, L. Lautenschütz, P. Haltenort, U. Arnold and J. Sauer (2017). "High Purity Oligomeric Oxymethylene Ethers as Diesel Fuels." Chemie Ingenieur Technik **89**(4): 486-489.

Dincer, I. (2007). "Environmental and sustainability aspects of hydrogen and fuel cell systems." International journal of energy research **31**(1): 29-55.

Dominguez-Faus, R., S. E. Powers, J. G. Burken and P. J. Alvarez (2009). "The water footprint of biofuels: A drink or drive issue?" Environmental science & technology **43**(9): 3005-3010.

Elghali, L., R. Clift, P. Sinclair, C. Panoutsou and A. Bauen (2007). "Developing a sustainability framework for the assessment of bioenergy systems." Energy Policy **35**(12): 6075-6083.

Fülöp, J. (2005). Introduction to decision making methods. BDEI-3 Workshop, Washington, Citeseer.

Genc, T. (2014). "Sensitivity analysis on PROMETHEE and TOPSIS weights." International Journal of Management and Decision Making **13**(4): 403-421.

Government-of-Canada (2016). Forestry: Region of Western Canada and the Territories: 2014-2016http://www.edsc.gc.ca/img/edsc-esdc/jobbank/SectoralProfiles/WT/2A71-RPT-SectProf_WT_Forestry_EN--GEN-20150827-VF-MR.pdf.

Guinée, J. (2016). "Life Cycle Sustainability Assessment: what is it and what are its challenges? Taking Stock of Industrial Ecology". Springer**:** 45-68.

Hoekstra, A. Y., A. K. Chapagain, M. M. Aldaya and M. M. Mekonnen (2011). The Water Footprint Assessment Manual: Setting the Global Standard London, Earthscan.

IEA (2016) "CO_2 emissions from fuel combustion." https://www.iea.org/publications/freepublications/publication/CO2EmissionsfromFuelCombustion_Highlights_2016.pdf Accessed.

Kabir, M. R. and A. Kumar (2011). "Development of net energy ratio and emission factor for biohydrogen production pathways." Bioresource Technology **102**(19): 8972-8985.

Kabir, M. R. and A. Kumar (2012). "Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways." Bioresource Technology **124**: 394-405.

King, C. W. and M. E. Webber (2008). "Water intensity of transportation." Environmental science & technology **42**(21): 7866-7872.

Kloepffer, W. (2008). "Life cycle sustainability assessment of products." The International Journal of Life Cycle Assessment **13**(2): 89.

Klöpffer, W. and I. Renner (2007). "Lebenszyklusbasierte Nachhaltigkeitsbewertung von Produkten." Technikfolgenabschätzung–Theorie und Praxis (TATuP) **16**(3): 32-38.

Kumar, A., J. B. Cameron and P. C. Flynn (2003). "Biomass power cost and optimum plant size in western Canada." Biomass and bioenergy **24**(6): 445-464.

Kumar, A., S. Sokhansanj and P. C. Flynn (2006). "Development of a multicriteria assessment model for ranking biomass feedstock collection and transportation systems." Applied Biochemistry and Biotechnology **129**(1-3): 71-87.

Kumar, M., A. O. Oyedun and A. Kumar (2017). "Hydrothermal liquefaction of biomass for the production of diluents for bitumen transport." Biofuels, Bioproducts and Biorefining **11**: 811-829.

Li, X., J. Grace, C. Lim, A. Watkinson, H. Chen and J. Kim (2004). "Biomass gasification in a circulating fluidized bed." Biomass and bioenergy **26**(2): 171-193.

Löfvenberg, U. (2010). "The BEST experiences with low blends in diesel and petrol fuels." BEST (BioEthanol for Sustainable Transport) Deliverable(D3).

Luk, J., H. Fernandes and A. Kumar (2010). "A conceptual framework for siting biorefineries in the Canadian Prairies." Biofuels, Bioproducts and Biorefining **4**(4): 408-422.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**(Supplement C): 1249-1262.

Mann, M. K. and P. L. Spath (1997). Life cycle assessment of a biomass gasification combined-cycle power system, National Renewable Energy Lab., Golden, CO (US).

Mareschal, B. (2013). "Visual PROMETHEE 1.4 Manual." Retrieved **16**: 2013.

McKendry, P. (2002). "Energy production from biomass (part 1): overview of biomass." Bioresource Technology **83**(1): 37-46.

Mohamadabadi, H. S., G. Tichkowsky and A. Kumar (2009). "Development of a multi-criteria assessment model for ranking of renewable and non-renewable transportation fuel vehicles." Energy **34**(1): 112-125.

Moore, F. T. (1959). "Economies of scale: Some statistical evidence." The Quarterly Journal of Economics: 232-245.

Myhre, G., D. Shindell, F. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J. Lamarque, D. Lee and B. Mendoza (2013). "Anthropogenic and natural radiative forcing." Climate change **423**.

Niven, R. K. (2005). "Ethanol in gasoline: environmental impacts and sustainability review article." Renewable and Sustainable Energy Reviews **9**(6): 535-555.

Oestreich, D., L. Lautenschütz, U. Arnold and J. Sauer (2017). "Reaction kinetics and equilibrium parameters for the production of oxymethylene dimethyl ethers (OME) from methanol and formaldehyde." Chemical Engineering Science **163**: 92-104.

OMNR (1994). Reasons for Decision and Decision: Class Environmental Assessment by the Ministry of Natural Resources for Timber Management on Crown Lands in Ontario, Environmental Assessment Board: Ontario Ministry of Natural Resources.

Pellegrini, L., M. Marchionna, R. Patrini and S. Florio (2013). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car, SAE Technical Paper.

Pellegrini, L., R. Patrini and M. Marchionna (2014). Effect of POMDME Blend on PAH Emissions and Particulate Size Distribution from an In-Use Light-Duty Diesel Engine, SAE Technical Paper.

Peters, M. S., K. D. Timmerhaus and R. E. West (2003). Plant Design and Economics for Chemical Engineers. New York, MCGraw-Hill.

Rahman, M. M., C. Canter and A. Kumar (2015). "Well-to-wheel life cycle assessment of transportation fuels derived from different North American conventional crudes." Applied Energy **156**: 159-173.

Safari, H., M. S. Fagheyi, S. S. Ahangari and M. R. Fathi (2012). "Applying Promethee method based on entropy weight for supplier selection." Business management and strategy **3**(1): 97-106.

Sarkar, S. and A. Kumar (2009). "Techno-economic assessment of biohydrogen production from forest biomass in Western Canada." Transactions of the ASABE **52**(2): 519-530.

Sarkar, S. and A. Kumar (2010). "Biohydrogen production from forest and agricultural residues for upgrading of bitumen from oil sands." Energy **35**(2): 582-591.

Sarkar, S. and A. Kumar (2010). "Large-scale biohydrogen production from bio-oil." Bioresource Technology **101**(19): 7350-7361.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2015). "Net energy ratio for the production of steam pretreated biomass-based pellets." Biomass and bioenergy **80**(0): 286-297.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2016). "Comparative net energy ratio analysis of pellet produced from steam pretreated biomass from agricultural residues and energy crops." Biomass and bioenergy **90**: 50-59.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2016). "Techno-economic assessment of pellets produced from steam pretreated biomass feedstock." Biomass and bioenergy **87**: 131-143.

Singh, S. and A. Kumar (2011). "Development of water requirement factors for biomass conversion pathway." Bioresource Technology **102**(2): 1316-1328.

Singh, S., A. Kumar and S. Jain (2014). "Impact of biofuel production on water demand in Alberta." Canadian Biosystems Engineering **56**.

Smith, D. M., B. C. Larson, M. J. Kelty and P. M. S. Ashton (1997). The practice of silviculture: applied forest ecology, John Wiley and Sons, Inc.

Spath, P., A. Aden, T. Eggeman, M. Ringer, B. Wallace and J. Jechura (2005). "Biomass to Hydrogen Production Detailed Design and Economics Utilizing the Battelle Columbus Heated Gasifier." Golden, Colorado, United States: National Renewable Energy Laboratory (NREL).

Spath, P. L. and M. K. Mann (2000). Life cycle assessment of hydrogen production via natural gas steam reforming, National Renewable Energy Laboratory Golden, CO.

Stripple, H. (2001). Life Cycle Assesment of Road—A Pilot Study for Inventory Analysis—2nd Revised Edition, B1210E.

Sultana, A. and A. Kumar (2011). "Development of energy and emission parameters for densified form of lignocellulosic biomass." Energy **36**(5): 2716-2732.

Sultana, A. and A. Kumar (2012). "Ranking of biomass pellets by integration of economic, environmental and technical factors." Biomass and Bioenergy **39**: 344-355.

Sultana, A., A. Kumar and D. Harfield (2010). "Development of agri-pellet production cost and optimum size." Bioresource Technology **101**(14): 5609-5621.

Swanson, R. M., A. Platon, J. A. Satrio, R. C. Brown and D. D. Hsu (2010). Techno-Economic Analysis of Biofuels Production Based on Gasification. Golden, Colorado, National Renewable Energy Laboratory**:** 165.

Thakur, A., C. E. Canter and A. Kumar (2014). "Life-cycle energy and emission analysis of power generation from forest biomass." Applied Energy **128**: 246-253.

Valente, C., R. Spinelli and B. G. Hillring (2011). "LCA of environmental and socio-economic impacts related to wood energy production in alpine conditions: Valle di Fiemme (Italy)." Journal of Cleaner Production **19**(17): 1931-1938.

Van Vliet, O. P., A. P. Faaij and W. C. Turkenburg (2009). "Fischer–Tropsch diesel production in a well-to-wheel perspective: a carbon, energy flow and cost analysis." Energy Conversion and Management **50**(4): 855-876.

Visual-PROMETHEE (2015). Multicriteria Decision Aid - Methods, Modeling and Software. Belgium, Solvay Brussels School of Economics & Managementhttp://www.promethee-gaia.net/index.html.

Winkler, N. (1998). "A manual for the planning, design and construction of forest roads in steep terrain." Food and Agriculture Organization of the United Nations, Forest Harvesting Case Study **10**.

Wong, A., H. Zhang and A. Kumar (2016). "Life cycle water footprint of hydrogenation-derived renewable diesel production from lignocellulosic biomass." Water research **102**: 330-345.

Wong, A. J. L. (2015). Life Cycle Assessment of Lignocellulosic Biomass Conversion Pathways to Hydrogenation Derived Renewable Diesel MSc Thesis, University of Alberta, Edmonton, Canada.

Yang, J., M. Xu, X. Zhang, Q. Hu, M. Sommerfeld and Y. Chen (2011). "Life-cycle analysis on biodiesel production from microalgae: water footprint and nutrients balance." Bioresource Technology **102**(1): 159-165.

Zhang, H. and K. R. Haapala (2014). "Integrating sustainable manufacturing assessment into decision making for a production work cell." Journal of Cleaner Production.

Zhang, X., A. Kumar, U. Arnold and J. Sauer (2014). "Biomass-derived Oxymethylene Ethers as Diesel Additives: A Thermodynamic Analysis." Energy Procedia **61**: 1921-1924.

Zhang, X., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2016). "An optimized process design for oxymethylene ether production from woody-biomass-derived syngas." Biomass and bioenergy **90**: 7-14.

Chapter 4: The life cycle greenhouse gas emission benefits from alternative uses of biofuel coproducts[3]

4.1 Introduction

Around 71% of global anthropogenic greenhouse gas (GHG) emissions are caused by energy production and its subsequent use, which also includes transportation emissions (U.S. Environmental Protection Agency 2014). The growing concern over climate change and fossil fuel dependency has encouraged people to look for renewable energy sources. The share of renewables in the total global energy demand has increased in the past decades. As of 2016, renewable energy comprised around 18% of world energy mix (World Energy Council 2016). Bioenergy accounts for 14% of the total renewable sources energy production and it has been considered among the most effective energy transformation and climate change mitigation options in many counties (World Energy Council 2016). Its future production is projected to constitute up to 35% of global energy by 2050 (Rose et al. 2014). Bioenergy has been widely used in number of applications namely, transportation fuel, space heating from domestic to industrial scale, domestic cooking, water heating, electricity generation, and combined heat and power generation and the growth of these application areas are gradually increasing in recent years (World Energy Council 2016, REN21 2017).

[3] A version of Chapter 4 has been published as Mahbub, N., E. Gemechu, H. Zhang and A. Kumar (2019). "The life cycle greenhouse gas emission benefits from alternative uses of biofuel coproducts." *Sustainable Energy Technologies and Assessments* **34**: 173-186.

Among the liquid transportation biofuels, ethanol and biodiesel are the two widely produced in the global market (REN21 2017). They are mainly sourced from agricultural products such as corn, palm oil, soybean, sugarcane and wheat. Representing around 4% of global transportation fuel, ethanol and biodiesel industries are emerging as the largest biofuel industries among all the renewables worldwide (Sawin et al. 2013). Global ethanol and biodiesel productions were reported to be 98.6 billion litres and 30.8 billion litres, respectively for the year 2016 (Sawin et al. 2013), with 9% increase of biodiesel production from 2015 (Sawin et al. 2013). The United States (U.S.) and Brazil are the world leading biofuel producers, together contributing to 80% of total biofuels worldwide (World Energy Council 2016). The global biofuel production is expected to increase while the cost to reduce in the comping decades due to renewable energy policy development in many countries and global trends in diesel and gasoline demands (OECD/FAO 2018). Among the policy measures that strongly support biofuel expansion is the U.S. Renewable Fuel Standard (RFS). RFS sets a target of yearly volume of renewable fuels production that will be blended with conventional transportation fuels, up to 36 billion gallons of renewable fuel yearly by 2022 (U.S. Department of Energy 2011, U.S. Department of Energy 2018). According to RFS, U.S. has to increase its biofuel production from 15 billion litres in 2006 to 136 billion litres in 2020 (World Energy Council 2016). The European Union (EU) has also established policies for the production and use of energy from renewable sources. Directive 2009/28/EC requires 20% of the energy consumed in the EU to come from renewable sources (Commission 2009), while Directive 2003/30/EC requires Member States to set target of 5.75% share of biofuels in its road transportation fuel mix by 2010 (Commission 2003). Canadian federal government has also mandated a 5% ethanol blend in fossil gasoline since late 2010 and 2% biodiesel in conventional diesel since July 1, 2011 (Maps and Matters 2012). In support of this mandate, Canada is gradually producing increasing amount of biofuels internally with a target of reducing Canada's yearly GHG emissions up to four million tonnes (Maps and Matters 2012). The Government of Canada is also developing a clean fuel standard that aims to achieve 30 million tonnes of annual GHG emissions reduction by 2030 through the increase use of low carbon fuels (ECCC 2018). The biofuel industry can play an important role (Littlejohns et al. 2018).

Bioenergy could offer environmental benefits compared with fossil fuel-based electricity, heating or transportation (Staples et al. 2017) due to their renewable nature and potential to reduce GHG emissions. However, some studies suggest that the environmental impact from bioenergy could be worse than fossil fuel depending on how the direct and indirect land use change are accounted, the coproduct allocation option, system boundary definition and other considerations (Searchinger et al. 2008, Plevin et al. 2010, Kendall and Yuan 2013, Zaimes and Khanna 2014, Carneiro et al. 2017). The environmental and economic benefits of biofuel are not limited to the use of the main fuel, but also from the use of coproducts. In most cases, biofuel processes are multi-functional, i.e. they produced coproducts that have a wide range of potential applications. Evaluating the effects of coproduct application on the overall environmental performance of biofuel production is challenging and needs to be handled carefully. This has been a topic of discussion in several LCA studies (Kendall and Chang 2009, Kendall and Yuan 2013, Zaimes and Khanna 2014, Canter et al. 2016, Yizhen 2018). Due to the perspective nature of biofuel technologies and lack in proper consideration of coproduct use, inconsistency in system boundary selection, assumptions and allocation approaches; there is high uncertainty and variability of LCA results of biofuel. Depending on the feedstock type, the coproducts yield and potential applications could vary greatly. For example in Canada, ethanol is primarily produced from dry milling of wheat (Saha 2010), from which dry distiller grains (DDGs) are simultaneously produced as coproducts (Saha 2010, Moore 2011). Around 3.5 kg of DDGs can be obtained per litre of ethanol (Moore 2011). Canadian wheat-based ethanol industries provide 1.46 billion litres of ethanol and 1.16 million tonnes of DDGs annually (Saha 2010). Some ethanol industries also coproduce acetic acid, electricity, glycerol, lactic acid and pulp (Energy 2013, Falano et al. 2014). On the other hand, biodiesel from oil seeds such as canola or soybean seeds produces seed meals as a coproduct, around 60-80% of the feedstock (Moore 2011). Biodiesel industries also coproduce propylene glycol depending on the feedstock (Energy 2013, Falano et al. 2014).

Common biofuel coproducts such as DDGs, soybean meal, and canola meal have around 26%, 47%, and 35% crude protein, respectively (Moore 2011) . Crude protein is a measure of the protein content in a food and it is fundamental in animal feed. DDGs contain important nutrients such as protein and fiber that are left unused after the starch in wheat grains is extracted to produce ethanol (Energy 2013). It is widely available in market in different forms such as dry distiller grains with soluble, modified distiller grains with soluble, wet distiller grains with soluble, and condensed distillers soluble to be used as animal feed (Lardy and Anderson 2014). Around 42% of the total DDGs supplied to the local and global feedstock market by the US ethanol industries are used as dairy cattle feed, followed by beef cattle feed (42%) and swine feed (11%). The rest is used as poultry feed (5%) (Lardy and Anderson 2014). DDGs have a metabolizable energy of 7013 BTU/ lb dry matter, which is comparable to that of corn, a commonly used animal feed (7178 BTU/ lb dry matter) (Lory et al. 2008). In addition, DDGs have a crude protein concentration of 32.2%, four times higher than that of corn (8.3%) (Lory et al. 2008). This makes DDGs highly preferable sources of animal feed.

DDGs could also be used as a stand-alone heat source or as a compliment to coal in furnaces for space heating (Carmel , Saha 2010, Eriksson et al. 2012). There are several studies that evaluate the combustion and fuel characterization of DDGs (Eriksson et al. 2012), highlighting their high sulphur and nitrogen content, high potassium concentration and low calcium and magnesium concentrations which are comparable with most other agricultural fuels. Recent study by Mansur et al. (2018) (Tago and Masuda 2018) investigate the high potential of DDGs to be converted into biocrude oil, 38.2% oil recovery. Because of high moisture content and high oxygen levels, biomass has a lower heating value than conventional coal (Saha 2010). Cofiring densified biomass with coal can reduce combustion emissions significantly compared to conventional coal firing (Demirbas 2004, Ruhul Kabir and Kumar 2012). Biofuel coproducts such as DDGs can also be densified in cubes, pellets, or briquettes form to increase the bulk density and ultimately reducing the transportation and storage costs (Saha 2010). They can be cofired with coal or burned on their own to provide heating, similar to conventional wood pellets (Saha 2010). DDGs pellets can achieve high bulk density, hardness, durability, and lower moisture and ash content, which make them ideal to be used as an energy substitute (Saha 2010).

Organic fertilizer is another emerging potential market for biofuel coproducts. Because of high concentrations of plant nutrients, i.e., boron (B), calcium (Ca), iron (Fe), magnesium (Mg), manganese (Mn), nitrogen (N), phosphorus (P), potassium (K), sulfur (S), and zinc (Zn), both DDGs and oilseed meals are of great interest to those in the organic fertilizer market (Moore

2011). The high nitrogen content and the relatively less expensiveness, make DDGs preferable sources of nitrogen fertilizer than urea (Shroyer et al. 2011).

The canola meal potentially displaces soybean meal as an animal feed in the market (Moore 2011), and the GHG emissions savings from replacing the soybean meal as animal feed were determined. Moreover, due to the rapid expansion of biodiesel production and its potential as an alternate fuel, biodiesel industries are considering different uses of main coproducts such as crude glycerine or glycerol (Crandall 2004, Donkin et al. 2009). Three alternative applications are considered: (a) synthetic glycerine, (b) animal feed and (c) fertilizer.

Glycerine is initially obtained in a crude form during biodiesel production, which is assumed to be upgraded and sold as synthetic glycerine in the market. When used in cattle feed, glycerine displaces the starch content in the cattle feed rations. Schroder and Sudekum showed that 10% dry glycerine displaces around 50% of the starch content in the feed ration of steers (Schröder and Südekum 1999) . Glycerine replaces corn starch when used in high and low forage diets for sheep and steers with an energy value ranging from 0.90 to 1.05 Mcal/lb (Linn and Raeth-Knight 2016). Glycerine at 3.1% of dry matter in feed rations increases milk production rates and protein percentages (Bodarski et al. 2005) . Around 3-15% glycerine in feed ration dry matter has no lethal impact on cattle digestion, feed intake, milk composition, or milk production (Khalili et al. 1997, Schröder and Südekum 1999, Linke et al. 2004, Donkin et al. 2009). In addition, the energy value provided by the starch level in crude glycerine (2000-2300 kcal/kg) is equal to the energy provided by corn starch 2000 kcal/kg used in animal feed (Preston 2005, Inc. 2013, Donkin and Doane 2008). Hence, crude glycerine obtained during biodiesel processing can replace animal feed in the market. Increased biodiesel production also created a new market for glycerine as an alternative green energy source replacing petroleum-based fossil fuels such as natural gas (Andrew and Forgie 2008, Energy 2013).

A number of studies have been conducted on different applications of biofuel coproducts and their associated energy and emissions impacts (Lory et al. 2008, Bremer et al. 2010, Saha 2010, Moore 2011). Saha evaluated and compared the on-site environmental impacts from DDG pellets and commercial wood pellets when used as energy source in a furnace (Saha 2010). However, the life cycle environmental impacts of producing the DDGs pellets commercially to be used for space heating were not covered. Bremer et al. found that coproduct credits represent around 19-38% of the total life cycle GHG emissions in the corn to ethanol pathway when DDGs are used as animal feed mostly in cattle, poultry, and swine diets (Bremer et al. 2010). Henderson showed that life cycle GHG emissions decreased significantly (4-45%) from ethanol-blended gasoline when coproduct credits are obtained from DDGs and CO_2 produced during ethanol production from corn (Henderson 2000). Falano et al. evaluated the potential GHG emissions savings when acetic acid, electricity and lactic acid from ethanol production are considered. The study highlighted the possibility of around 72-87% GHG emissions reduction compared with conventional fuels (Falano et al. 2014). While most of the existing LCA studies are based on ethanol production from corn (Levelton Engineering Ltd. & $(S\&T)^2$ Consultants Inc. 1999, Farrell et al. 2006, Kendall and Chang 2009, Bremer et al. 2010), studies on coproducts from wheat processing are very limited. Furthermore, the studies on biofuel coproduct credits did not thoroughly discuss the commercial feasibility and the long-term policy implications of the coproduct uses. Moreover, most of the studies considered a single use of coproduct in the biofuel life cycle (Bremer et al. 2010, Lardy and Anderson 2014), alternative use of the coproducts and the potential environmental trade-off were not evaluated. This paper, therefore, aims to answer research questions associated with the use of coproduct from biofuel pathway as a potential substitute for animal feed, fuel or fertilizer. The paper attempts to address the following issues: Are the coproducts from wheat to ethanol and canola to biodiesel energy conversion pathway environmentally viable sources of alternative? What are the environmental benefits of substituting those coproducts? How to assign credit to the coproducts and how it affects the overall results? What are the long-term policy implications of coproduct uses from bioenergy pathways? The insight from the study will provide valuable information to decision makers as it highlights the potential consequences of coproducts from a rapidly growing bioenergy production in an economy.

4.2 Method

LCA is the method followed in this study. According to ISO, LCA has four stages: goal and scope definition, life cycle inventory analysis; life cycle impact assessment; and interpretation (ISO 2006, Burchart-Korol 2013). The research context according the ISO framework and principle is explained in this section.

Goal and Scope of the Study

The main purpose of this study is to identify alternative environmental impacts related to potential applications of coproducts from two biofuel pathways: DDGs from wheat to ethanol and canola meal and glycerine from canola to biodiesel conversions. The effect of coproduct allocation on the overall life cycle GHG emissions will be investigated. The main findings from this study will provide the energy industry, mostly biofuel producers, insights on the environmental benefits of effective coproduct use. The information will also help policy makers to develop long-term policies on the commercialization of environmentally friendly valorization of biofuel coproducts.

The GHG emissions saving from the use of coproducts is evaluated per the functional unit of 1MJ energy from a biofuel. This will allow an easy comparability with other similar studies (Kabir and Kumar 2011). All energy and material input requirements are scaled to match the functional unit.

Figure 4.1 shows the flow chart of the method followed to obtain the coproduct credits. First, an appropriate allocation approach is decided among all possible approaches. Allocation approaches are explained in later section. Different coproduct applications, the corresponding energy and emission factors associated with the displaced products, and the displacement ratios between the coproducts and the displaced products are explained in sections later for both the pathways. The coproduct credits were calculated by multiplying the amount of displaced products with the emission factors associated with the displaced products. The amount of the displaced products was determined by using the displacement ratios. Finally, a number of scenarios were obtained considering different uses of coproducts. The scenarios include both single and combined uses of different coproduct applications to determine the impact of coproduct credits on the overall life cycle GHG emissions of the biofuels.

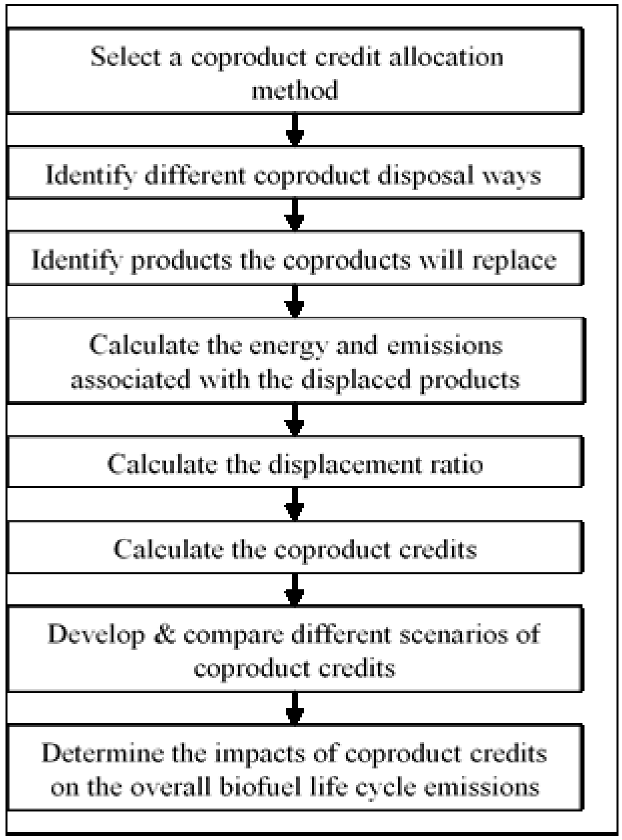
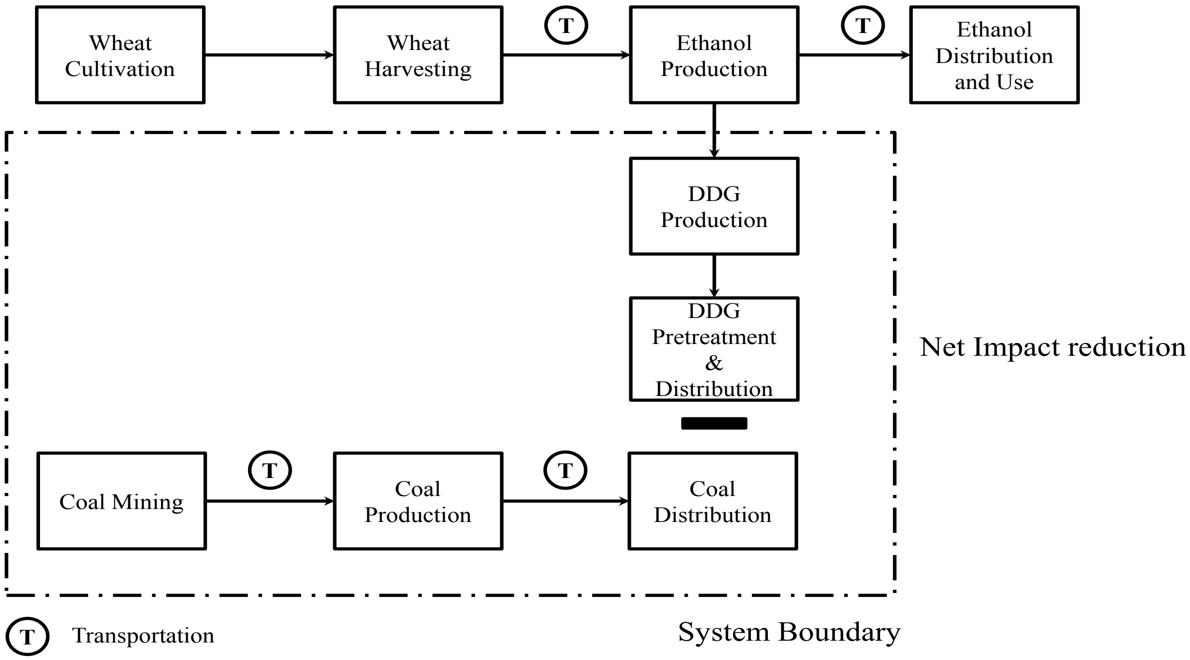
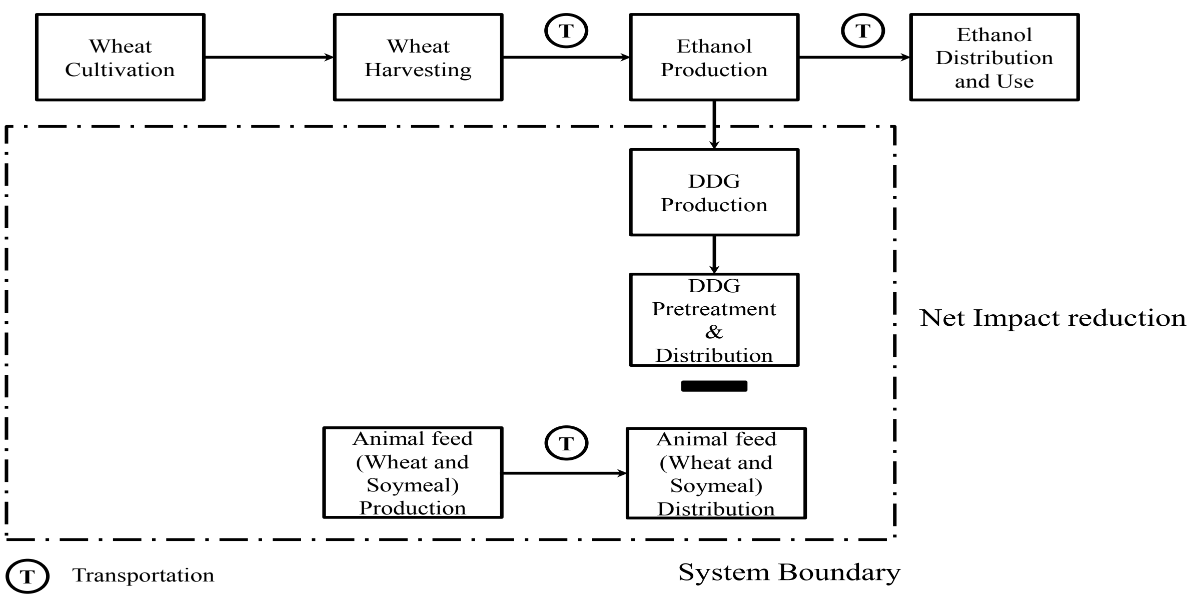


Figure 4. 1: Method followed to obtain biofuel coproduct credits

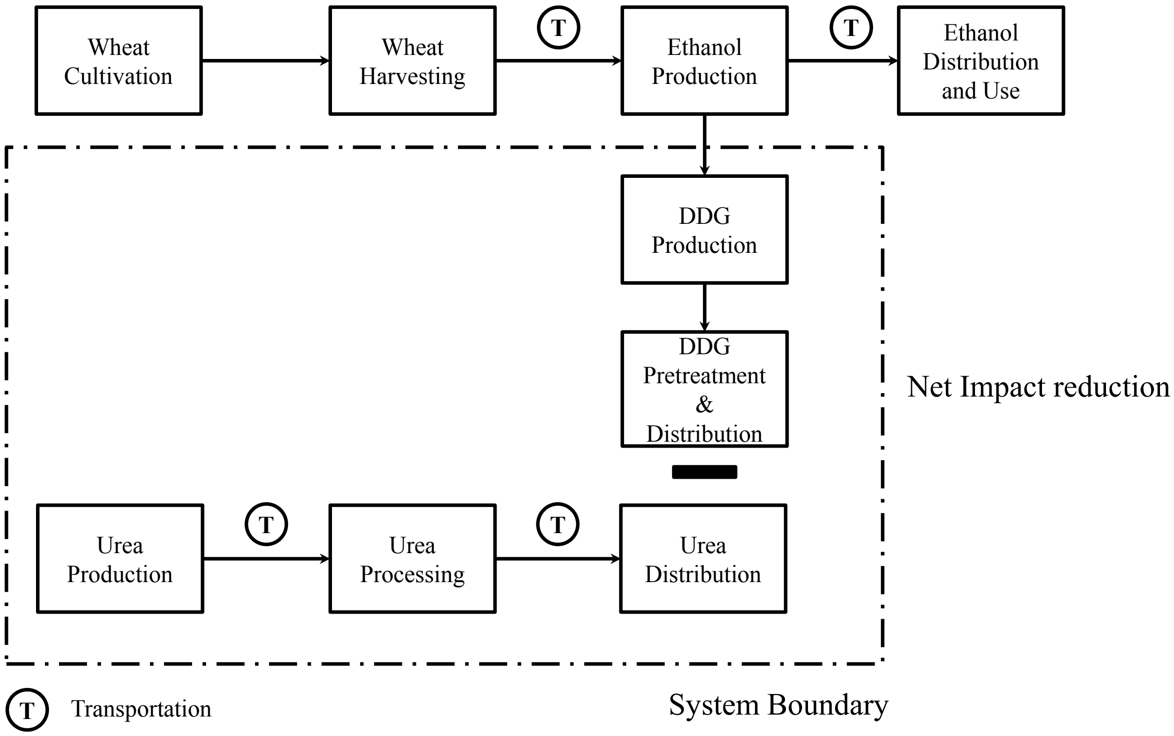
Three application of bioethanol coproduct DDG has been explored in the study. Figures 2.2 a, b, c illustrates the cradle-to-grave wheat to ethanol pathway system boundaries(Kodera 2007, $(S\&T)^2$ Consultants Inc. 2011), when the coproduct is used to substitute: (a) animal feed, (b) coal, and (c) fertilizer, respectively.


Wheat Cultivation
Wheat Harvesting
T
Ethanol Production
T
Ethanol Distribution and Use
DDG Production
DDG Pretreatment & Distribution
Net Impact reduction
Coal Mining
T
Coal Production
T
Coal Distribution
T Transportation
System Boundary

(a)


Wheat Cultivation
Wheat Harvesting
T
Ethanol Production
T
Ethanol Distribution and Use
DDG Production
DDG Pretreatment & Distribution
Net Impact reduction
Animal feed (Wheat and Soymeal) Production
T
Animal feed (Wheat and Soymeal) Distribution
T Transportation
System Boundary

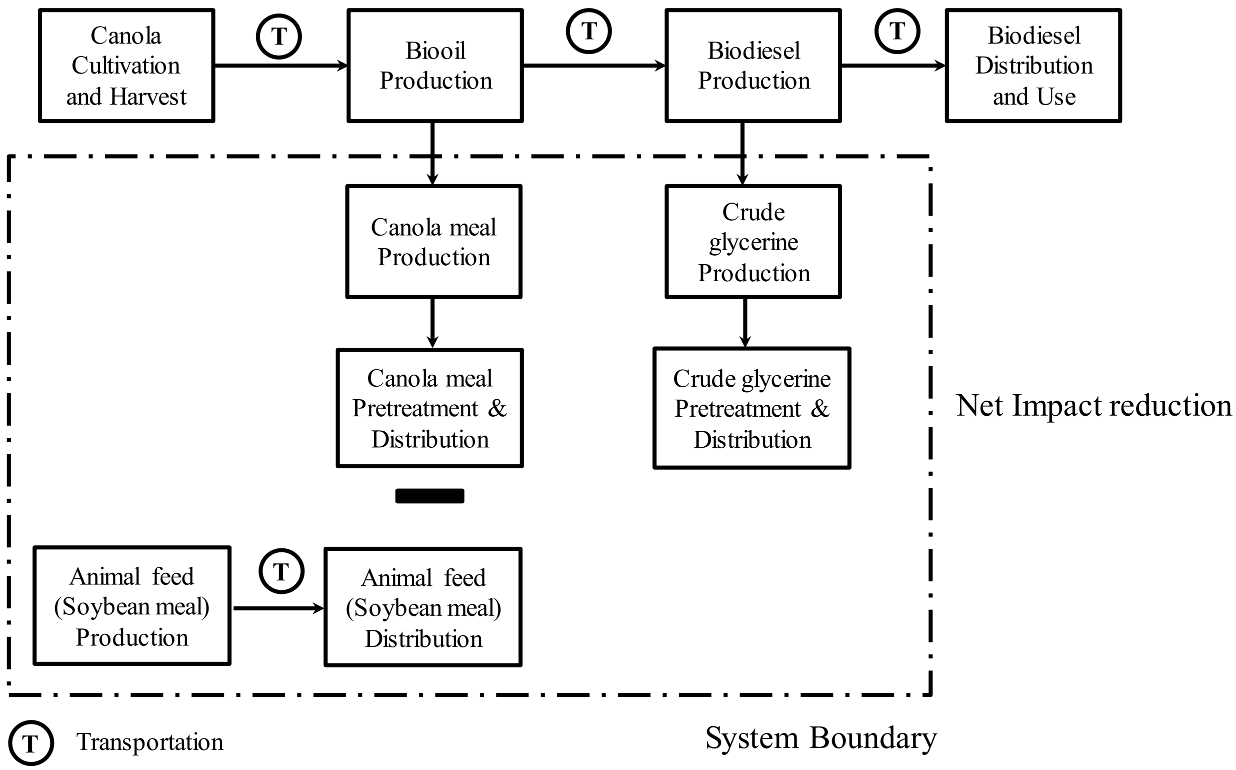
(b)



(c)

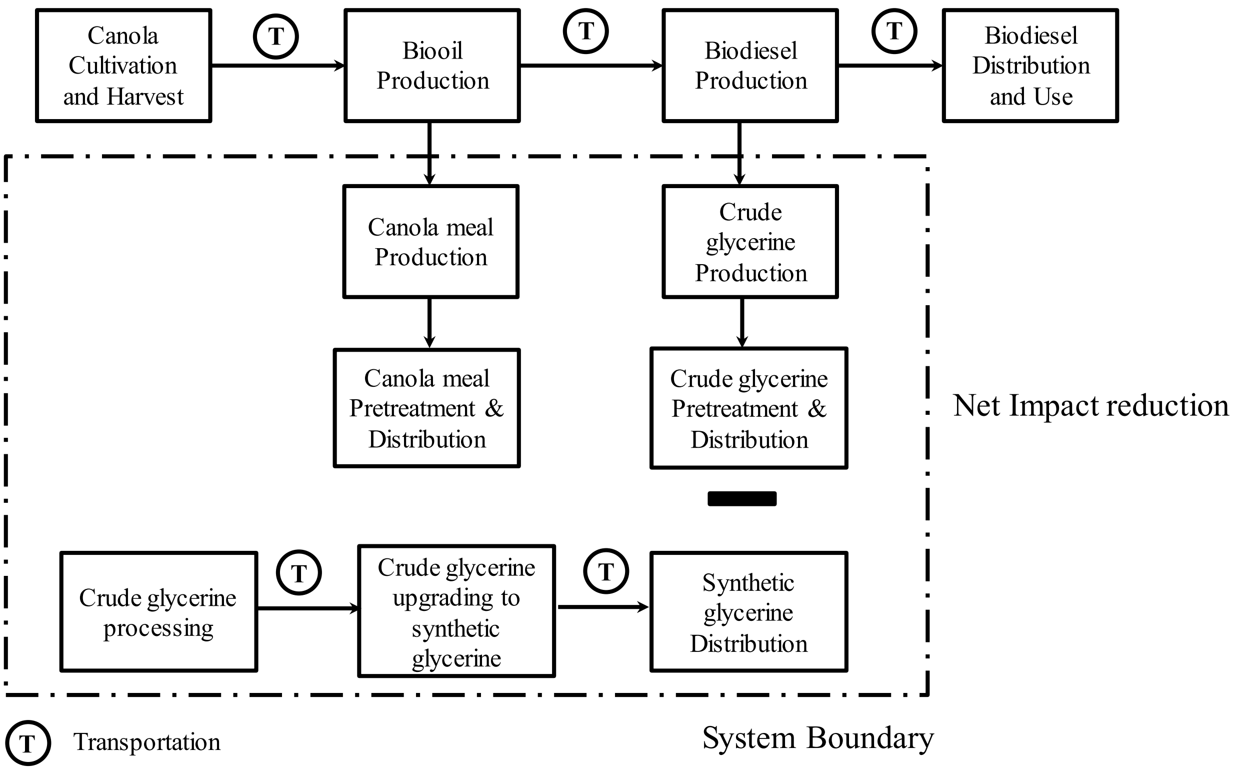
Figure 4. 2: Cradle-to-grave wheat to ethanol pathway system boundaries when the coproduct substitutes (a) coal, (b) animal feed, and (c) fertilizer

As illustrated in Figures 2.3 a, b, c, d, canola meal is a coproduct from canola seed crushing in canola oil production process, while glycerine is from canola oil to biodiesel conversion process ((S&T)^2 Consultants Inc. 2004, (S&T)2 Consultants Inc. 2011). One application of canola meal as animal feed and three applications of glycerin such as upgraded synthetic glycerin, animal feed, and fertilizer have been considered in this study.



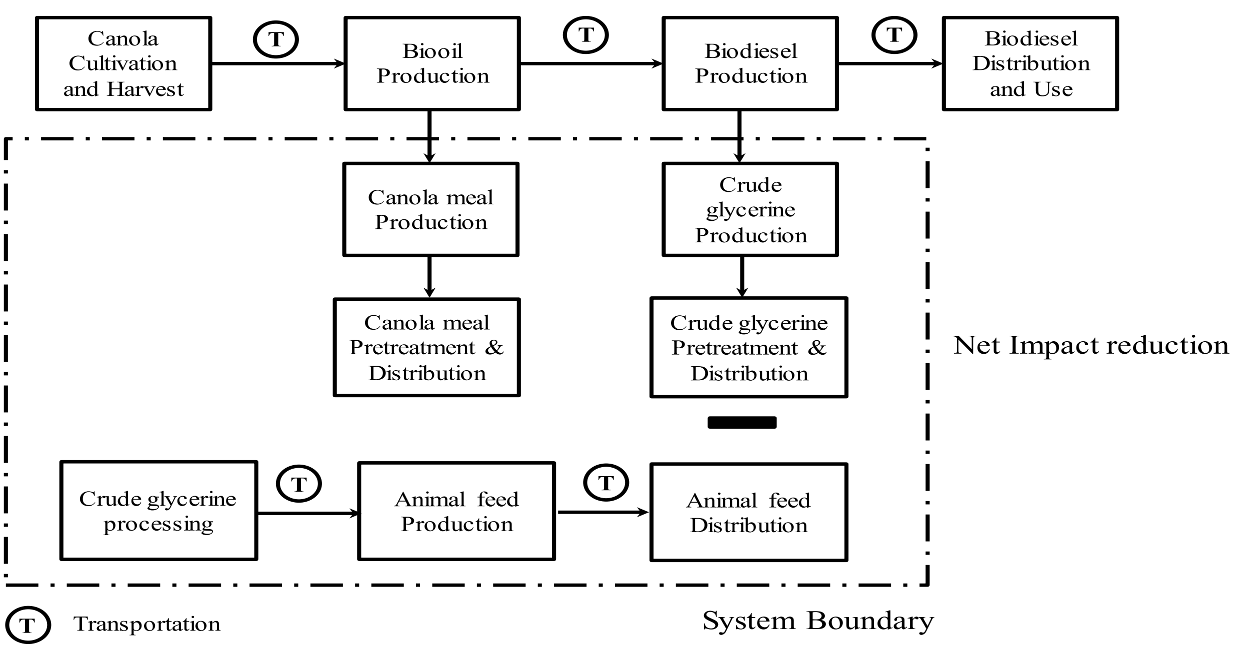
Canola Cultivation and Harvest
T
Biooil Production
T
Biodiesel Production
T
Biodiesel Distribution and Use
Canola meal Production
Crude glycerine Production
Canola meal Pretreatment & Distribution
Crude glycerine Pretreatment & Distribution
Net Impact reduction
Animal feed (Soybean meal) Production
T
Animal feed (Soybean meal) Distribution
T
Transportation
System Boundary

(a)



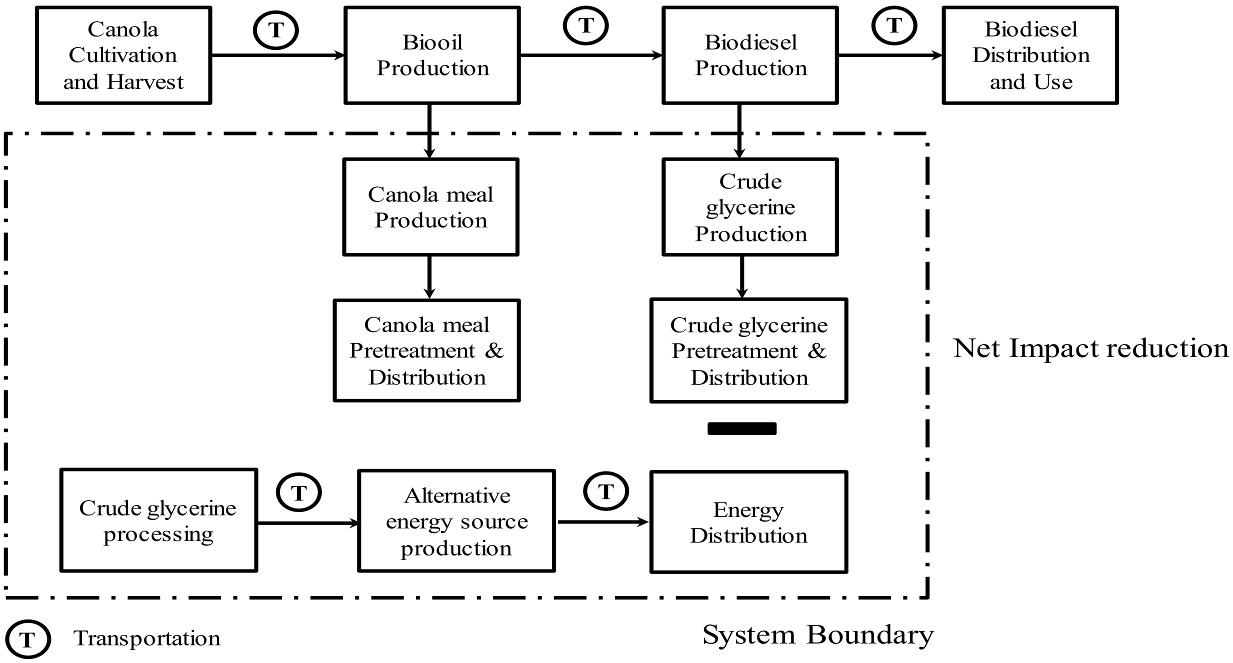
Canola Cultivation and Harvest
T
Biooil Production
T
Biodiesel Production
T
Biodiesel Distribution and Use
Canola meal Production
Crude glycerine Production
Canola meal Pretreatment & Distribution
Crude glycerine Pretreatment & Distribution
Net Impact reduction
Crude glycerine processing
T
Crude glycerine upgrading to synthetic glycerine
T
Synthetic glycerine Distribution
T Transportation
System Boundary

(b)



Canola Cultivation and Harvest
T
Biooil Production
T
Biodiesel Production
T
Biodiesel Distribution and Use
Canola meal Production
Crude glycerine Production
Canola meal Pretreatment & Distribution
Crude glycerine Pretreatment & Distribution
Net Impact reduction
Crude glycerine processing
T
Animal feed Production
T
Animal feed Distribution
T Transportation
System Boundary

(c)



(d)

Figure 4. 3: Cradle-to-grave canola to biodiesel pathway system boundaries when the coproducts substitute (a) soybean meal, (b) synthetic glycerine, (c) animal feed, and (d) fossil energy

4.3 Life Cycle Inventory Analysis

The data requirements and the allocation procedures followed to calculate the coproduct credits for the biofuel pathways are discussed in this section (Zaimes and Khanna 2014).

4.3.1 Allocation Approach to Coproduct Credit

There are different ways of assigning environmental burden between the main product and its coproducts (Andrew and Forgie 2008). According the ISO recommendation, allocation should be avoided whenever possible by partitioning the input and output requirements among the main product and its coproduct(ISO 2006). When this is not the case, the allocation hierarchy is to perform system expansion. It is a preferred allocation method in LCA for coproduct credit calculation (Levelton Engineering Ltd. & $(S\&T)^2$ Consultants Inc. 1999, Weidema 1999, Kim and Dale 2002, Kodera 2007, Heirigs et al. 2010, (S&T)2 Consultants Inc. 2011). System expansion approach followed to evaluate the effect of coproduct credit on the overall life cycle GHG emissions of the biofuels.

System expansion approach extends the boundary to include alternative products from the market that can be substituted or replaced by the coproduct (Andrew and Forgie 2008). The main advantage of system expansion approach is that it takes into account the indirect effects of the coproduct use, thereby making the LCA results more reliable (Ekvall and Finnveden 2001, Kodera 2007). That said, there are arguments in favor of the system expansion approach in cases where fuel is a byproduct and non-fuel products are the main product, and when different fuel production pathways are compared (Kim and Dale 2002, Kodera 2007, Heirigs et al. 2010, Wang et al. 2011). According to Wang et al., the selection of a coproduct credit allocation method should be based on the particular biofuel pathway ((S&T)2 Consultants Inc. 2011, Wang et al. 2011). However, mass allocation, energy allocation, and market value allocations have been mentioned in the literature for coproduct credit calculation as well (Malça and Freire 2004, Bernesson et al. 2006, Hill et al. 2006, Malça and Freire 2006, Kodera 2007). But limitations such as curtailed system boundaries, which exclude coproduct use, lower value of displaced products, and inconsistency in price, result in improper allocation of emissions between the main product and its coproduct (Kodera 2007, (S&T)2 Consultants Inc. 2011).

In system expansion approach, the coproduct credit is obtained based on the assumption that the displaced product and the coproduct have the same energy and GHG emissions credit. The obtained coproduct credit is then subtracted from the total life cycle GHG emissions of the corresponding biofuel pathways (Heirigs et al. 2010, (S&T)2 Consultants Inc. 2011). To apply this approach, the following are necessary: quantity of products to be displaced, displacement ratio between the coproduct and displaced product, and finally, energy and emission impacts from the displaced products (Kodera 2007, (S&T)2 Consultants Inc. 2011).

4.3.2 Data Inventory: Wheat to Ethanol Pathway

Table 4.1 shows the inputs and assumptions to calculate coproduct credits when using DDGs from wheat to ethanol pathway in the market as animal feed, fuel, and fertilizer.

Table 4. 1: Data and assumptions for different uses of ethanol coproduct DDGs

Assumptions/ Properties	Units	Value	References
Wheat grain needed	kg/ L	2.66	((S&T)^2 Consultants Inc. 2004)
Displacement ratio (DDG to animal feed)		0.8	((S&T)^2 Consultants Inc. 2004, Thacker 2006, Feed Opportunities from Biofuels Industries Network 2013)

Assumptions/ Properties	Units	Value	References
Fraction of wheat in animal feed to be displaced by DDGs		0.45	((S&T)[2] Consultants Inc. 2004)
Fraction of soybean meal in animal feed to be displaced by DDGs		0.55	((S&T)[2] Consultants Inc. 2004)
DDGs yield from ethanol plant	kg per kg wheat grain	0.38	((S&T)[2] Consultants Inc. 2004)
Avoided methane emissions	g per kg DDG	3.74	((S&T)[2] Consultants Inc. 2004)
Natural gas required	kWh per L of ethanol	0.32	((S&T)[2] Consultants Inc. 2004)
Coal upstream emission factor	g/mm Btu	6,178	(Argonne National Laboratory 2016)
Coal combustion emission factor	g/ mm Btu	100,002	(Argonne National Laboratory 2016)
C footprint of 1 tonne DDG	tonne CO_2eq	0.330	Calculated

Assumptions/ Properties	Units	Value	References
C footprint of 1 tonne coal	tonne CO_2eq	2.42	Calculated
Ethanol higher heating value	MJ/L	23.57	((S&T)^2 Consultants Inc. 2004)
Combustion energy of coal	MJ/kg	24.024	((S&T)^2 Consultants Inc. 2004)
Combustion energy f DDG pellet	MJ/kg	18.84	(Saha 2010)
Displacement ratio (coal to DDGs)		1.28	Calculated
GHG emission from urea production	g CO_2eq/kg urea	1,329.43	(Munawar et al. 2003)
Displacement ratio (urea to DDGs)		11	(Shroyer et al. 2011)

The GHGenius model was used to calculate the coproduct credit when DDGs from wheat to ethanol pathway displace animal feed ((S&T)^2 Consultants Inc. 2004). The displacement ratio between DDGs and animal feed is calculated as 0.8 ((S&T)^2 Consultants Inc. 2004). The animal feed is composed of 45% wheat and 55% soybean meal ((S&T)^2 Consultants Inc. 2004). The displacement factor is calculated based on the digestible energy in kcal/kg of animal feed in pig diets (Thacker 2006, FOBI 2013). DDGs have digestible energy of 3924 kcal/kg, compared with wheat and soybean meal that provide around 3350 and 3280 kcal/kg, respectively (Thacker 2006, FOBI 2013). The energy and GHG emissions associated with the production and transportation of wheat and soybean for animal feed are included in the coproduct credit calculation. Methane emission savings of 3.74 grams per kg of DDGs consumed as animal feed is assumed ($(S\&T)^2$ Consultants Inc. 2004, Energy 2013). It is considered that the ruminants being fed DDGs in their cattle feed ratio produces 3.74 grams less methane compared to animals that are fed regular cattle feed.

When used as an energy fuel for space heating, a displacement ratio of 1.28 between DDGs and coal is calculated based on the combustion energy of DDGs pellets and industrial coal (Table 1) ($(S\&T)^2$ Consultants Inc. 2004, Saha 2010). 1.28 kg of DDGs is needed to provide the same amount of energy from 1 kg of Coal. The upstream and combustion emissions from coal are considered to calculate the GHG emission savings from substituting coal with DDGs. The life cycle emissions from the consumption of 1 kg of coal is determined using the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET 1) model (Laboratory 2016). The displacement ratio of 11.0 is considered for DDGs when used as a fertilizer, urea. This ratio is based on experimental data that shows a 11.0 kg of DDGs provides the same amount of nitrogen a 1 kg of urea could provide (0.5 kg of nitrogen per kg of urea) (Shroyer et al. 2011). The direct emissions mainly due to the combustion of fossil sources like natural gas during urea production and indirect emissions such as unused CO_2 released to the atmosphere during urea production are considered while calculating the coproduct credit (Munawar et al. 2003).

4.3.3 Data Inventory: Canola to Biodiesel Pathway

Table 4.2 shows the data and assumptions used for calculating coproduct credits for biodiesel coproducts, such as canola meal and glycerine.

Table 4. 2: Data and Assumptions for Different Uses of the Biodiesel Coproducts Canola Meal and Glycerine.

Assumptions/ Properties	Units	Value	References
Canola required per kg canola oil	kg/ L	0.88	((S&T)^2 Consultants Inc. 2004)
GHG emissions from soy bean milling	g CO_2eq/kg soy bean	385	((S&T)^2 Consultants Inc. 2004)
GHG emissions from canola milling	g CO_2eq/kg canola	247	((S&T)^2 Consultants Inc. 2004)
Canola meal fraction		0.57	((S&T)^2 Consultants Inc. 2004)
Canola oil fraction		0.43	((S&T)^2 Consultants Inc. 2004)
Biodiesel higher heating value	MJ/L	35.4	((S&T)^2 Consultants Inc. 2004)
Glycerine yield	kg/L biodiesel	0.09	(Donkin et al. 2009)
Displaced emission value for producing crude glycerine	g CO_2eq/kg	6590	((S&T)^2 Consultants Inc. 2004)

Assumptions/ Properties	Units	Value	References
Displaced emission value for glycerine used as fossil fuel	g CO_2eq/kg	300	((S&T)^2 Consultants Inc. 2004)
Displaced emission value for glycerine used as animal feed	g CO_2eq/kg	400	((S&T)^2 Consultants Inc. 2004)

For the canola to biodiesel pathway, the GHGenius model was used to calculate the coproduct credit and the system expansion approach was used to assess the credits for both coproducts in the canola to biodiesel pathway ((S&T)^2 Consultants Inc. 2004). The GHG emissions for processing canola meal and soybean meal are estimated to be 247 g CO_2eq per kg canola and 385 g CO_2eq per kg soybean, respectively ((S&T)^2 Consultants Inc. 2004). These two emission values were used to calculate the coproduct credit for canola meal when it replaces soybean meal as animal feed.

Based on existing literature, it is assumed that 0.09 kg crude glycerine is obtained from 1 liter of biodiesel from canola (Donkin et al. 2009). The crude glycerine is assumed to be upgraded in order to obtain refined food-grade glycerine that would replace synthetic glycerine in the market (Energy 2013). The energy and emission factors for alternative production of synthetic glycerine are determined using GHGenius model version 4.03 ((S&T)^2 Consultants Inc. 2004). Around 6590 gram CO_2eq per kg glycerine is saved when crude glycerine during the biodiesel production is upgraded and used to displace synthetic glycerine in the market ((S&T)^2 Consultants Inc. 2004). The emission factor includes all fossil energy required to process synthetic glycerine from crude glycerine.

Glycerine is a common ingredient in dairy cattle feed rations (Andrew and Forgie 2008, Donkin 2008, Donkin et al. 2009, Energy 2013). Energy value of 2000-2300 kcal/ kg has been found to be comparable to starch from corn in feed ration (Andrew and Forgie 2008, Donkin 2008, Energy 2013, Linn and Raeth-Knight 2016). This study assumes that glycerine replaces corn in cattle feed rations with a displaced emission value of 400 g CO_2eq/kg of glycerine ((S&T)^2 Consultants Inc. 2004, Energy 2013). The emission factor has been calculated considering all the fossil energy sources involved in producing cattle feed from crude glycerine.

The displaced emission value for glycerine as an energy substitute for natural gas is assumed to be 300 g CO_2eq/kg of glycerine ((S&T)^2 Consultants Inc. 2004, Energy 2013), which means around 300 g CO_2eq emissions are saved when glycerine is used as a fossil energy source, natural gas. All the emission factors include the fossil energy sources required to process crude glycerine into synthetic glycerine, animal feed and /or alternative energy sources.

4.4 Life Cycle Impact Assessment

A number of scenarios were developed by considering individual and combined impacts from different uses of the coproducts. The best scenario was identified based on the maximum coproduct credit or maximum GHG emission savings. Predominantly Alberta-specific numbers and assumptions were used to develop the model. The three primary GHG emissions are considered: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) to estimate the environmental impact in this study. The global warming potential values are based the Intergovernmental Panel on Climate Change (IPCC) 100 years' time-horizon (Flato et al. 2013, Mahbub et al. 2017). A GHGenius model was used to calculate the coproduct credits and the life cycle GHG emission savings (Kumar et al. 2009, $(S\&T)^2$ Consultants Inc. 2013). The model uses system expansion approach for coproduct credit (Kodera 2007, Heirigs et al. 2010, (S&T)2 Consultants Inc. 2011).

4.5 Sensitivity and Uncertainty Analysis

Since the model inputs and assumptions were taken from a wide range of literature, sensitivity and uncertainty analysis were conducted to prove the robustness of the model. Sensitivity analysis was conducted to identify the sensitive inputs and uncertainty analysis was done to identify the possible output range caused due to uncertain inputs. An excel based add-in software R-studio was used to conduct both the sensitivity and to run the Monte Carlo simulation for uncertainty analysis (Gentleman et al. 2009). For the sensitivity analysis, the input values were changed within a range of ±25% to determine the sensitive and insensitive inputs to output results (Di Lullo et al. 2017). This study investigated several applications of biofuel coproducts, most of which are not commonly produced or used. Hence, data availability was a big challenge when calculating the coproduct credits. Due to the lack of sufficient data, normal and lognormal distributions were not used in this study. Rather, a more conservative distribution, triangular distribution, was selected for the model inputs, since it supports extreme value points resulting in a moderate output distribution (Di Lullo et al. 2017). The triangular distribution was generated by using three point estimates for the input variables, the maximum, minimum, and most likely values (Di Lullo et al. 2017). The maximum and minimum values were determined by assuming ±10% of the input values, which are good enough to obtain a conservative output distribution based on results of previous studies related to biofuels (Di Lullo et al. 2017, Mahbub et al. 2017, Mahbub et al. 2018). 50,000 samples were run for the Monte Carlo simulation, which results in an error less than 0.1 gCO_2eq/MJ with 99% probability (G. Angevine and Oviedo 2012, Di Lullo et al. 2017). The results were obtained in the 5^{th} and 95^{th} percentiles, and the most uncertain inputs were observed using tornado plots (G. Angevine and Oviedo 2012, Di Lullo et al. 2017).

4.6 Results

This section presents and discusses the main findings of the study. First the results from the study are summarized. The significance of biofuel coproduct credits from different coproduct applications on the environment is discussed.

The coproduct credits or GHG emission savings were determined by varying the percentage of the coproduct use from 20% to 100% by increments of 20%, to understand the significant impacts of coproduct use (see Table 3). The results are presented per the functional unit of 1 MJ of energy produced from biofuel. Because of the higher moisture level and lower heating value of DDGs than conventional coal, it produces less GHG emissions per unit energy compared to industrial coal combustion. As a result, the highest coproduct credits were obtained from using DDGs pellets for heating as an alternative to coal firing. The GHG emissions saving ranges from 13.43 (20% use) to 67.14 (100% use) g CO_2eq/MJ of ethanol. The second highest emissions savings were obtained from using DDGs as animal feed followed by as fertilizer. Due to high protein content and digestible energy of DDGs, higher coproduct credit is obtained when used as animal feed compared to land application. The GHG emissions savings from using DDGs as animal feed ranged from 8.31 to 17.29 g CO_2eq/MJ of ethanol, whereas when used as fertilizer the emission savings were 1.72 to 8.61 g CO_2eq/MJ of ethanol, respectively. Table 3 shows the percentage of use, amount of products displaced by coproducts, and associated GHG emissions saved for the three DDGs applications.

Table 4. 3: GHG emissions savings per MJ of ethanol from different uses of DDGs

GHG emissions savings when DDG displaces animal feed (wheat and soybean meal)						
% of quantity of DDG displacing animal feed	%	100	80	60	40	20
Amount of animal feed displaced by DDG	ton	112.23	89.78	67.34	44.89	22.45
GHG emissions savings from the use of DDG as animal feed	g CO_2eq/MJ	17.29	15.79	12.80	9.80	8.31
GHG emissions savings when DDG displaces fertilizer (urea)						
% of quantity of DDG displacing fertilizer	%	100	80	60	40	20
Amount of fertilizer displaced by DDG	tonne	8,610	6,888	5,166	3,444	1,722
GHG emissions savings from the use of DDG as fertilizer	gCO_2eq/MJ	5.18	4.15	3.11	2.20	1.04
GHG emissions savings when DDG displaces fossil fuel (coal)						
% of quantity of DDG displacing fossil fuel	%	100	80	60	40	20

Amount of fossil fuel displaced by DDG	tonne	74,272	59,417	44,563	29,709	14,854
GHG emissions savings from the use of DDG as fuel	gCO_2eq/MJ	67.14	53.71	40.28	26.86	13.43

A number of scenarios have been developed in this study considering different potential uses of biofuel coproducts. Fig. 4.4 shows the GHG emission savings from combining different DDGs uses. Since, DDGs are mostly used as animal feed; it is considered common while developing the combined scenarios. Coproduct credit increases with increasing percentages of coproduct use as fuel in combinations. Comparatively fewer GHG emissions were saved when DDGs displaces animal feed or/and urea compared to coal.

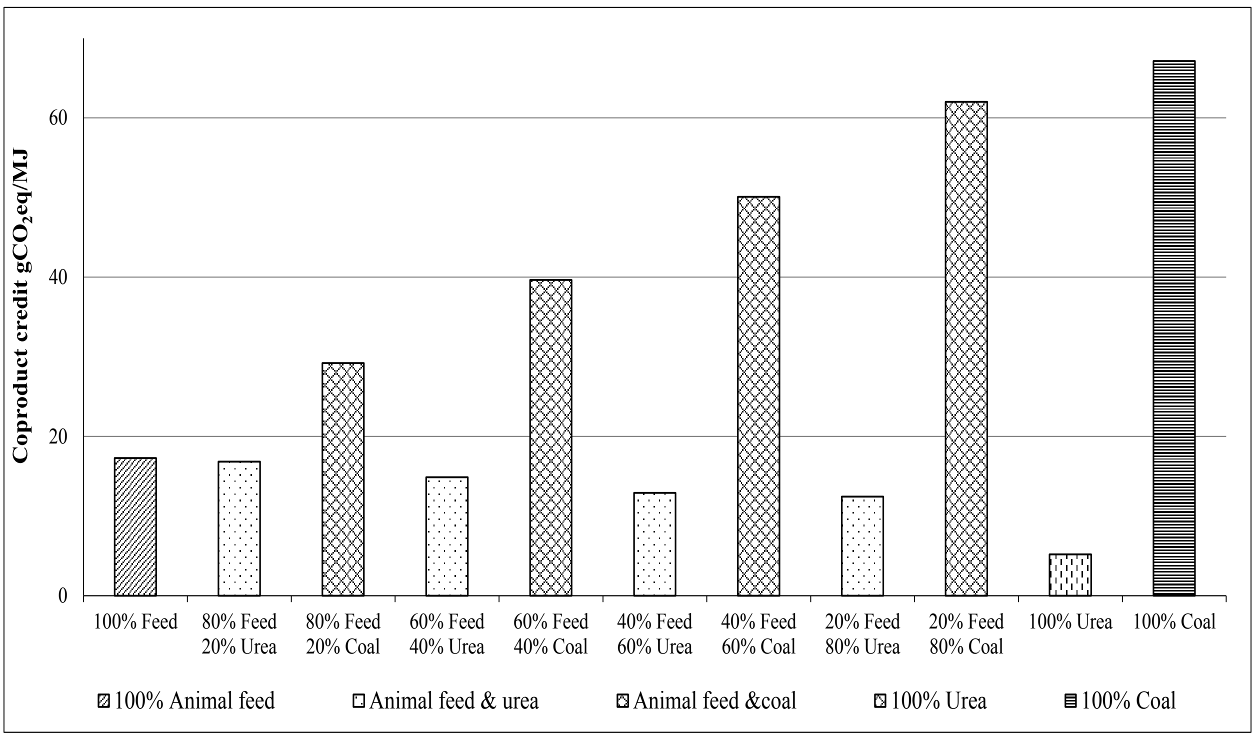


Figure 4. 4: Coproduct credit or GHG emissions savings from different combinations (by mass percentage) of DDG use in the wheat to ethanol pathway

Table 4.4 shows the coproduct credits from the use of canola meal and glycerine from biodiesel. The coproduct credit ranges from 2.23 to 11.14 g CO2eq/MJ of biodiesel based on the use percentage of canola meal. For glycerine, the highest credit was earned when the crude glycerine produced in biodiesel production replaces synthetic glycerine. The GHG emission savings ranged from 17.13 to 3.95 g CO_2eq/MJ when crude glycerine was processed into synthetic glycerine based on their percentage use. The displaced emission value of synthetic glycerine (6,590 g CO_2eq/kg glycerine) is the largest, due to higher energy consumption and higher emissions associated with synthetic glycerine production from crude glycerine compared to the other glycerine uses such as animal feed (0.40 g CO_2eq/MJ of biodiesel) or fuel (0.30 g CO_2eq/MJ of biodiesel) (Energy 2013). The GHG emission savings ranged from 1.04 to 0.21 g CO2eq/MJ and 0.78 to 0.16 gCO2eq/MJ of biodiesel when glycerine replaces corn in the cattle feed ration and petroleum-based natural gas as an energy source, respectively.

Table 4. 4: GHG emissions savings per MJ of biodiesel from different uses of the biodiesel coproducts canola meal and glycerine.

GHG emissions savings when canola meal displaces animal feed						
% of quantity of canola meal displacing animal feed	%	100	80	60	40	20
GHG emissions savings from the use of canola meal as animal feed per kg of canola meal	gCO_2eq/ kg canola	335	268	201	134	67

GHG emissions savings from the use of canola meal as animal feed per MJ of biodiesel	gCO_2eq/MJ	11.14	8.92	6.69	4.46	2.23
GHG emissions savings when crude glycerine displaces synthetic glycerine						
% of quantity of crude glycerine displacing synthetic glycerine	%	100	80	60	40	20
Amount of synthetic glycerine displaced by crude glycerine	kg/ L biodiesel	0.092	0.074	0.055	0.037	0.018
GHG emissions savings from the use of crude glycerine	gCO_2eq/MJ	17.13	13.70	10.28	6.85	3.43
GHG emissions savings when crude glycerine displaces fuel (natural gas)						
% of quantity of crude glycerine displacing fossil fuel	%	100	80	60	40	20
Amount of fossil fuel displaced by crude glycerine	kg/L biodiesel	0.092	0.074	0.055	0.037	0.018
GHG emissions savings from the use of crude glycerine	gCO_2eq/MJ	0.78	0.62	0.47	0.31	0.16
GHG emissions savings when crude glycerine displaces animal feed						
% of quantity of crude glycerine displacing animal feed	%	100	80	60	40	20
Amount of animal feed displaced by crude glycerine	kg/L biodiesel	0.092	0.074	0.055	0.037	0.018

GHG emissions savings from the use of crude glycerine	gCO_2eq/MJ	1.04	0.83	0.62	0.42	0.21

Figure 4.5 compares different combinations of glycerine use. Due to the highest displaced emission factor of synthetic glycerine, it was found that coproduct credit gradually decreases with deceases in the percentage of glycerine replacing synthetic glycerine. The lowest coproduct credit was earned from the use of glycerine as an energy source.

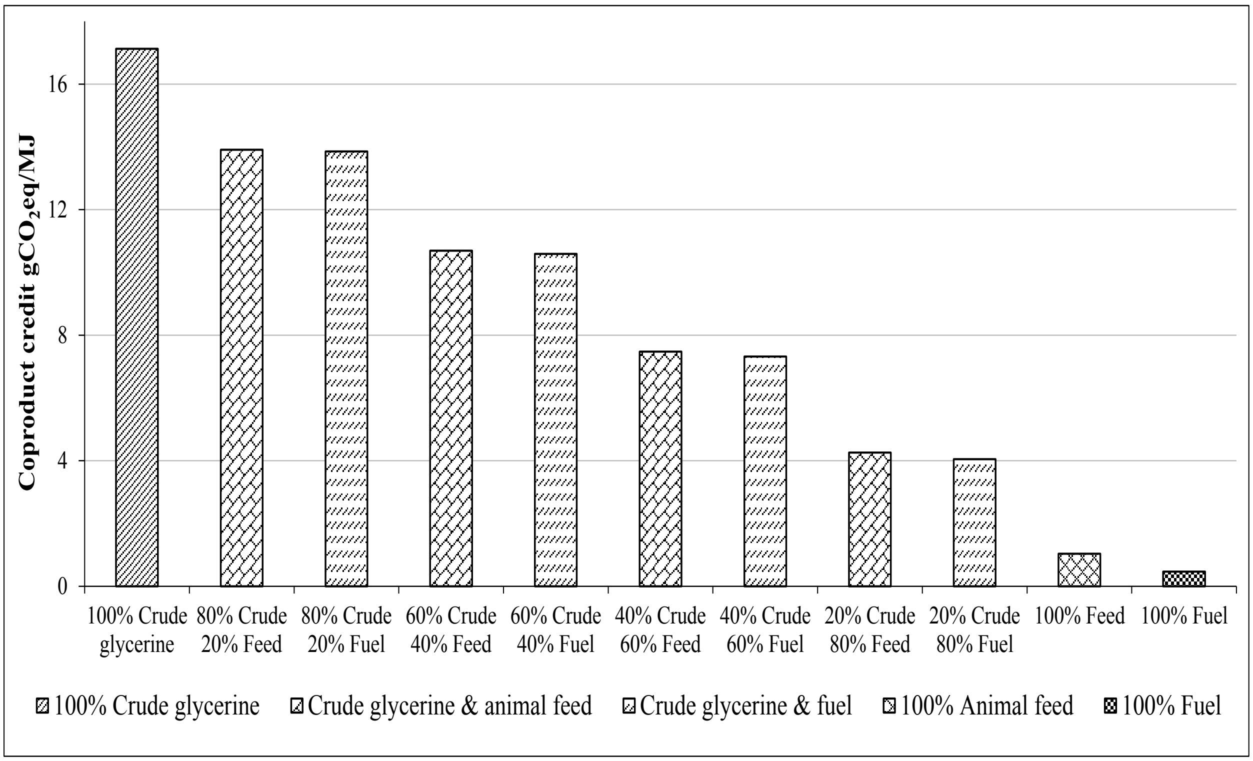


Figure 4. 5: Coproduct credit or GHG emissions savings from different combinations (by mass percentage) of crude glycerine use in the canola to biodiesel pathway

Impact of Coproduct Credit on Overall Life Cycle Emissions of Biofuels

Life cycle GHG emissions decrease significantly when biofuel pathways are credited for coproduct production and use. Falano et al. assessed the cradle-to-gate GHG emissions for ethanol production from wheat in a bio refinery. 1 MJ ethanol production is responsible for 82.70 gCO_2eq (Falano et al. 2014). The system boundary includes the production of coproducts like acetic acid, lactic acid, and electricity along with the main product, and emission savings from the use of coproducts are evaluated using system expansion approach (Falano et al. 2014). However, the study does not consider the transportation and use of the main product and the coproducts within the system boundary (Falano et al. 2014). The wheat to ethanol life cycle GHG emissions reported by Falano et al. has been used as a reference in this study, to assess the percentage reduction in the total life cycle GHG emissions for different applications of DDGs (shown in Table 4.5). The life cycle GHG emissions can be decreased by 16-81%, 10-21%, and 1-6% when DDGs replace fuel, animal feed, and fertilizer, respectively, based on their use ranging from 20-100% (by mass).

Table 4. 5: Impact of coproduct use on overall life cycle GHG emissions in the wheat to ethanol pathway.

Total life cycle GHG emissions in wheat to ethanol pathway (Falano et al. 2014)	% of quantity of DDG displacing other products	Net GHG emissions when DDG replaces fuel (coal)	% of total life cycle emission reduced	Net GHG emissions when DDG replaces fertilizer (urea)	% of total life cycle GHG emission reduced	Net GHG emissions when DDG replaces animal feed-wheat and soymeal	% of total life cycle GHG emission reduced
gCO_2eq/MJ	%	gCO_2eq/MJ	%	gCO_2eq/MJ	%	gCO_2eq/MJ	%
82.70	100	15.56	81	77.52	6	65.41	21
82.70	80	28.99	65	78.55	5	66.91	19
82.70	60	42.42	49	79.59	4	69.90	15
82.70	40	55.84	32	80.63	3	72.90	12
82.70	20	69.27	16	81.66	1	74.40	10

The life cycle GHG emissions reductions from combined applications of DDGs were also determined (Table 4.6). Due to higher emission intensity of coal, the total GHG emissions can be reduced by 35-75% when DDGs replace fuel and animal feed in combination. On the other hand, the GHG emission reductions ranged from 15-20% when DDGs replace animal feed and fertilizer in combination depending on the use percentage.

Table 4. 6: Impact of different types of coproduct (DDG) use in combination on overall life cycle GHG emissions in the wheat to ethanol pathway.

Total life cycle GHG emissions in the wheat to ethanol pathway (Falano et al. 2014)	% of combined use when DDG displaces other products	Net GHG emissions when DDG is used as animal feed and fertilizer in a combination	% of total life cycle GHG emissions reduced	Net GHG emissions when DDG used as animal feed and fuel in a combination	% of total life cycle GHG emissions reduced
gCO_2eq/MJ	%	gCO_2eq/MJ	%	gCO_2eq/MJ	%
82.70	80%-20%	65.87	20	53.48	35
82.70	60%-40%	67.83	18	43.05	48
82.70	40%-60%	69.80	16	32.61	61
82.70	20%-80%	70.25	15	20.68	75
82.70	0%-100%	77.52	6	15.56	81
82.70	100%-0%	65.41	21	65.41	21

Table 4.7 shows the impact of canola meal and glycerine uses on the overall life cycle GHG emissions of the canola to biodiesel pathway. A study by S&T consultant (Energy 2013) that evaluates the cradle-to-grave GHG emissions of biodiesel production from canola in Alberta, has

been used as a reference. The study shows that the production of 1 MJ biodiesel from canola in Alberta emits around 42.28 g CO_2eq (Energy 2013), when the pathway is not credited for any coproducts production and use.

Table 4. 7: Impact of coproduct use on overall life cycle GHG emissions in the canola to biodiesel pathway.

Total life cycle GHG emissions in canola to biodiesel pathway (Energy 2013)	gCO_2eq /MJ	42.28	42.28	42.28	42.28	42.28
Percentage of quantity of canola meal/ glycerine displacing other products	%	100%	80%	60%	40%	20%
Net GHG emissions when canola meal replaces animal feed	gCO_2eq/ MJ	31.14	33.37	35.59	37.82	40.05
Percentage of total life cycle emissions reduced	%	26	21	16	11	5
Net GHG emissions when glycerine replaces synthetic glycerine	gCO_2eq/ MJ	25.15	28.58	32.00	35.43	38.86
Percentage of total life cycle GHG emissions reduced	%	41	32	24	16	8
Net GHG emissions when glycerine replaces fuel	gCO_2eq/ MJ	41.50	41.66	41.81	41.97	42.12

Percentage of total life cycle GHG emissions reduced	%	2	1	1	1	0
Net GHG emissions when glycerine replaces animal feed	gCO_2eq/MJ	41.24	41.45	41.66	41.86	42.07
Percentage of total life cycle GHG emissions reduced	%	2	2	1	1	0

GHG emissions decreased by 5-26% from the use of canola meal as animal feed and by 8-41% when crude glycerine replaced synthetic glycerine in the market. However, due to very low emission intensity, the GHG emissions reduction was quite small (around 1-2%) when crude glycerine replaced either fuel or fertilizer in the market. Similarly, due to higher emission intensity of synthetic glycerine, the combined GHG emissions reductions ranged from 10-33% for either combination.

Table 4. 8: Impact of different types of coproduct (glycerine) uses in combination on overall life cycle GHG emissions in the canola to biodiesel pathway.

Total life cycle GHG emissions in the canola to biodiesel pathway (Energy 2013)	% of combined use when glycerine displaces other products	Net GHG emissions when glycerine displaces synthetic glycerine and fuel in combination	% of total life cycle GHG emissions reduced	Net GHG emissions when glycerine displaces synthetic glycerine and animal feed in combination	% of total life cycle GHG emissions reduced
gCO_2eq/MJ	%	gCO_2eq/MJ	%	gCO_2eq/MJ	%
42.28	80%-20%	28.42	33	28.37	33
42.28	60%-40%	31.70	25	31.59	25
42.28	40%-60%	35.00	17	34.81	18
42.28	20%-80%	38.23	10	38.02	10
42.28	0%-100%	41.50	2	41.24	2
42.28	100%-0%	25.15	41	25.15	41

Sensitivity and uncertainty analysis

From both pathways, the applications with the largest GHG emissions savings are presented in this paper.

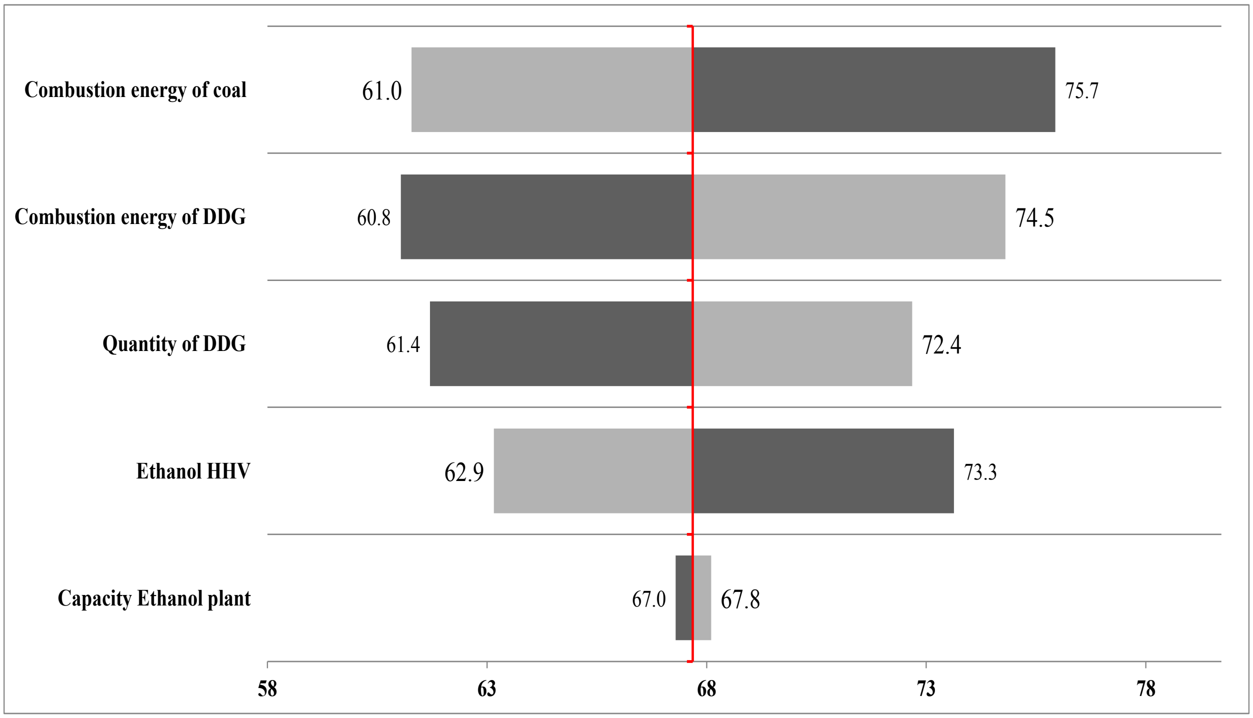


Figure 4. 6: Uncertainty analysis of coproduct credits in gCO_2eq/MJ when DDG replaces fuel (fossil coal)

In wheat to ethanol pathway when DDGs replaced fossil coal the combustion energy of coal and DDGs were found to be the most sensitive and uncertain inputs (Figure 4.6).

Similarly, in canola to biodiesel pathway, when crude glycerine replaces synthetic glycerine, the quantity of glycerine produced per litre of biodiesel found to be the most sensitive and uncertain input (Figure 4.7) that needs further investigated.

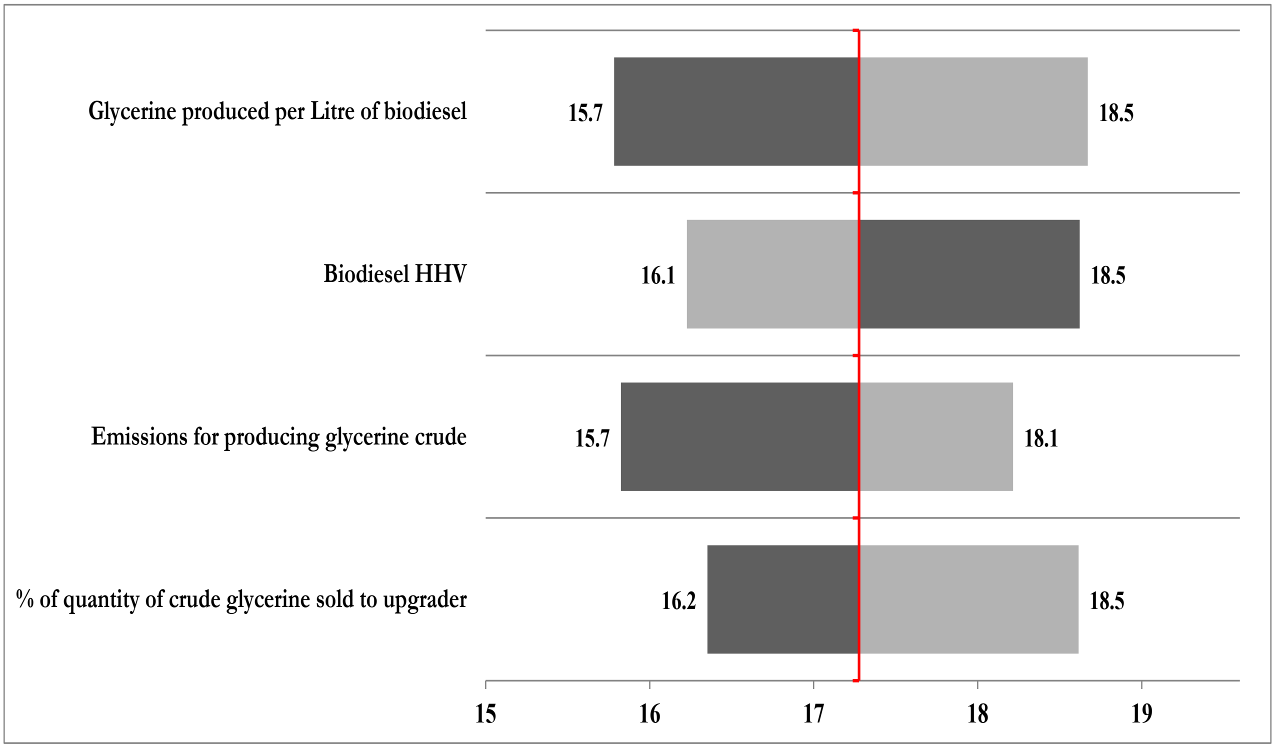


Figure 4. 7: Uncertainty analysis of coproduct credits in gCO_2eq/MJ when crude glycerine replaces synthetic glycerine

4.7 Discussion

The study results highlight that the overall environmental performances of biofuel production are highly affected by the use of coproducts. The importance of considering coproduct uses in evaluating the life cycle environmental impacts of biofuel has been highlighted by different authors (Luo et al. 2009, Kaufman et al. 2010, Wang et al. 2011). As a climate change mitigation option, biofuel use has been practiced as a potential solution. USA, one of the largest global producers, has mandated biofuel production and its blending with conventional transportation fuels under the RFS federal program. With the increasing production of biofuels, the industries are also producing unavoidably huge amount of coproducts every year, which are often overlooked in environmental assessment of the main products. Biofuel coproducts have the potential to be used as animal feed, fertilizer and also replace fossil based energy products in the market, thereby reducing GHG emissions. Globally, biofuel industry is producing around 52 million tonnes of byproducts that can be used for animal feed, mainly from ethanol production (Shurson 2017). Coproducts from corn to ethanol conversion have been widely applied in feeding dairy and beef cattle, but with substantial increase in ethanol market and associated coproducts, the applications has been expanded in swine, poultry and aquaculture feeds (Shurson GC et al. 2012, Shurson 2017). Fertilizer and fuel energy substitute potential of biofuel coproducts are another applications areas. The net environmental benefit from bioenergy production is then highly depends on the applications of its coproducts and the corresponding substitution ratio. When the coproduct substitutes a product with high GHG emission intensity, the overall biofuel life cycle emissions will decrease significantly.

Different types of coproducts such as DDG, CO_2, acetic acid, electricity and lactic acid simultaneously produced during ethanol production have been found to reduce total life cycle GHG ranging from 4 to 87% (Henderson 2000, Bremer et al. 2010, Falano et al. 2014). This study also showed different use of coproducts produced in the bioethanol and biodiesel lifecycle. The most common use of ethanol coproduct DDG is as animal feed, which reduced the life cycle GHG emissions from 8.31 to 17.29 g CO_2eq/GJ of ethanol based on percentage of DDG used (20-100%). However, the highest GHG emissions reductions were obtained from using DDGs pellets for heating as an alternative to coal firing. The GHG emissions saving ranges from 13.43 (20% use) to 67.14 (100% use) g CO_2eq/MJ of ethanol due to lower heating value and higher moisture level than conventional coal. GHG emissions savings were comparatively less from DDG's use as fertilizer ranging from 1.72 to 8.61 g CO_2eq/MJ of ethanol. Several scenarios were also developed to determine the coproduct credit from different use of coproducts in combination. It was found that the emissions savings were higher with higher percentage of DDG use as animal feed and fuel in combination. No other study has determined the combined impact of different coproduct use before. The emission factors for displaced products include all the life cycle stages mostly. Further research is required, to handle all the life cycle stages carefully.

For canola to diesel pathway, two different coproducts were obtained in two different life cycle stages, namely canola meal and glycerin. The highest coproduct credits were obtained from crude glycerin displacing synthetic glycerin in the market. The GHG emissions savings ranged from 3.95 to 17.13 g CO_2eq/MJ. The lowest GHG emissions were saved when glycerin replaces natural gas as a fuel source ranging from 0.16 to 0.78 gCO_2eq/MJ of biodiesel.

However, the actual impact could be subjected to different factors, such as market availability, economic benefit, environmental, and other concerns while considering the realistic marginal technologies and competing products. For example, in the case of DDGs' use as animal feed, even though they have important features in terms of nutrient value, there are some concerns that could potential affect their wide applications. The high phosphorus concentration, improper digestion due to high fiber concentration, high sulfur content, presence of carcinogenic toxins (Belyea et al. 1989, Morse et al. 1992, Arosemena et al. 1995, Blanco-Canqui et al. 2002, Belyea et al. 2004, Rausch and Belyea 2006), and variable protein content (25-35%) (Belyea et al. 1989) are issues that could have negative impacts on animal health. Further research is required to overcome these issues. Hence, the biofuel industries are showing interest in other markets for biofuel coproducts.

The ethanol industry sells distiller grains in different forms, including wet distiller grains, distillates, dry distiller grains, etc. The shelf life of the wet distiller grain (65-70% moisture content) is very short, i.e., less than a week (Saha 2010), which is not enough to transport coproducts to local markets. This can result in a huge economic loss for the ethanol industry. Hence, most of the ethanol industries dry the distiller grains to 10% moisture content (Saha 2010), so that the shelf time is increased to two weeks and the DDGs can be transported to markets across the country or worldwide.

The two biggest challenges in producing DDGs are drying and transporting the DDGs to the market. Studies have found that drying the distiller grains consumes around 50% of the total energy used in the bioethanol life cycle (Inc. (S & T)2 Consultants 2013). Because of its low bulk density, transporting granular DDGs is very expensive. Densified DDGs, such as in the form of pellets, cubes, or briquettes, can increase the bulk density, thereby reducing the transportation cost. DDG pellets are burned with coal or individually for heating. However, the production and use of DDGs commercially as a fuel source is still challenging as several factors like plant size, transportation to the target market, market availability, market acceptance, etc., need to be considered

Evaluating the economic viability of biofuel coproduct use as animal feed, fertilizer and fuel energy substitute is beyond the scope of this study. A detailed process based techno-economic assessment is required to provide more insight on the impacts of coproducts on the overall cost competitiveness of biofuel development. However, the economic values of coproducts are also other key elements that need to be investigated in detail. Due to very limited production and market supply, DDGs are very expensive when they are used to substitute animal feeds like corn grain, rice bran, and wheat bran. This is mainly due to their higher protein content (USDA 2016). On the other hand, biofuel coproducts seemed to have less economic value as fertilizer than as feed which discourages their application as fertilizers. Lory et al. showed that DDGs' market value as animal feed is 99 dollars per tonne more than as a fertilizer. DDGs have 39 dollars per tonne value when used as fertilizer and around 172 dollars per tonne when used as animal feed (Lory et al. 2008). Government initiatives and incentives are indispensable to foster the market penetration of bioenergy coproduct to overcome these economic barriers and ensure a more GHG emissions reduction option. The short-term implications such as huge investment requirements and societal barriers might seem challenging, however, the long-term implications can help the government to meet their target aligned with the national and global policies related to climate change mitigations such as, climate leadership plan, renewable fuels regulations, Paris agreement etc.

The other important aspect that could to be discovered is the long-term environmental and economic consequences of biofuel expansion in a given economy. This study highlights the role of coproduct use in reducing the environmental impacts of biofuel. The outlook of renewable market in the coming decades suggests a remarkable growth in biofuel production and the associated coproducts. The agriculture, energy and transportation sector are among the most affected sectors from the wide development and deployment of biofuel. However, this study is limited to evaluating the share of GHG emissions attributed from the use of coproducts, but the impacts on the environmental performance of other sectors are not covered. Hence, the future perspective of this study is to assess the economy-wide and global environmental and socio-economic consequences of coproduct from bioenergy by extending the scope of the study.

4.8 Conclusion

This study identified potential applications of biofuel coproducts and analyzed the energy and emission impacts of coproduct use on the overall life cycle GHG emissions of the two most largely produced and commonly used biofuels, ethanol from wheat grain and biodiesel from canola. DDGs are produced as coproducts of ethanol from wheat grain, and canola meal and crude glycerine are produced along with biodiesel from canola. Three applications were considered for DDGs in the wheat to ethanol pathway: animal feed, fuel, and fertilizer. DDGs replace canola meal and soymeal as animal feed, coal as a fuel, and urea as a fertilizer. Results show that GHG emissions decrease by 181% from individual applications of DDGs and by 35-75% from different combined applications of DDGs. Similarly, in the canola biodiesel pathway, GHG emissions decrease by 5-26% when canola meal replaces animal feed and 8-41% when crude glycerine replaces synthetic glycerine. GHG emissions decrease by 10-33% when crude glycerine replaces synthetic glycerine and fuel or animal feed in combination. Hence, it can be said that GHG emissions decreases significantly when coproduct utilization or their disposal is included in the system boundary of the biofuel life cycle. The biofuel pathways can be credited for other applications of the identified coproducts or for other coproducts as well. The developed model can be used to determine the effects of coproduct use on the overall lifecycle emissions for other biofuels as well. The model can be updated to include all the emission factors associated with the life cycle stages of the alternative coproduct applications. This study will help to inform stakeholders in bioenergy sector regarding the environmental consequences of potential coproduct use. The model also provides insight for policy makers on the long-term perspective if biofuel coproducts use as climate change mitigation option. The study generally provides a Canada specific model for coproduct credit based on a comprehensive process-based LCA approach. Though it is applied to coproducts from wheat to ethanol and canola to biodiesel production, the model could be applied to other energy pathways with minor adjustments. Finally, it is worth mentioning that there are some limitations that need to be considered while interpreting the results. These are, for example, the carbon footprint of the replaced products can be updated to factorize all the stages involved in their life cycle (if needed). The model predominantly uses Alberta specific assumptions and data, but it can be used for any other region and country. More applications or disposal ways of the coproducts can be identified and embedded on the model for wider application.

References

$(S\&T)^2$ Consultants Inc. (2004). GHGenius A model for lifecycle assessment of transportation fuels. A model for lifecycle assessment of transportation fuels Accessed 06/06/2017.

(S&T)2 Consultants Inc. (2011) "Co-Products Report." http://www.ghgenius.ca/reports.php?tag=Co-products Accessed 17/11/2016.

$(S\&T)^2$ Consultants Inc. (2013) "GHGenius Manual 4.03 Volume 1 " http://www.ghgenius.ca/reports.php Accessed17/11/2016.

Andrew, R. and V. Forgie (2008). "Life Cycle Assessment of Canola Biodiesel in New Zealand" Landcare Research and Costing of Canola Biodiesel in New Zealand. Centre for Ecological Economics.

Argonne National Laboratory (2016). GREET® Model. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation Model https://greet.es.anl.gov/.

Arosemena, A., E. DePeters and J. Fadel (1995). "Extent of variability in nutrient composition within selected by-product feedstuffs." Animal feed science and technology **54**(1): 103-120.

Belyea, R., K. Rausch and M. Tumbleson (2004). "Composition of corn and distillers dried grains with solubles from dry grind ethanol processing." Bioresource Technology **94**(3): 293-298.

Belyea, R., B. Steevens, R. Restrepo and A. Clubb (1989). "Variation in composition of by-product feeds." Journal of Dairy Science **72**(9): 2339-2345.

Bernesson, S., D. Nilsson and P.-A. Hansson (2006). "A limited LCA comparing large-and small-scale production of ethanol for heavy engines under Swedish conditions." Biomass and Bioenergy **30**(1): 46-57.

Blanco-Canqui, H., C. J. Gantzer, S. H. Anderson, E. Alberts and F. Ghidey (2002). "Saturated hydraulic conductivity and its impact on simulated runoff for claypan soils." Soil Science Society of America Journal **66**(5): 1596-1602.

Bodarski, R., T. Wertelecki, F. Bommer and S. Gosiewski (2005). "The changes of metabolic status and lactation performance in dairy cows under feeding TMR with glycerin [glycerol] supplement at periparturient period." Electronic Journal of Polish Agricultural Universities. Series Animal Husbandry **8**(4).

Bremer, V. R., A. J. Liska, T. J. Klopfenstein, G. E. Erickson, H. S. Yang, D. T. Walters and K. G. Cassman (2010). "Emissions savings in the corn-ethanol life cycle from feeding coproducts to livestock." Journal of environmental quality **39**(2): 472-482.

Burchart-Korol, D. (2013). "Life cycle assessment of steel production in Poland: a case study." Journal of Cleaner Production **54**: 235-243.

Canter, C. E., J. B. Dunn, J. Han, Z. Wang and M. Wang (2016). "Policy Implications of Allocation Methods in the Life Cycle Analysis of Integrated Corn and Corn Stover Ethanol Production." BioEnergy Research **9**(1): 77-87.

Carmel, I. "Ag Fuel and Feed." Waste Diversion & Recycling Retrieved 17 November, 2016, from http://www.waste.net/wastehaulers/aa1138251.html

Carneiro, M. L. N. M., F. Pradelle, S. L. Braga, M. S. P. Gomes, A. R. F. A. Martins, F. Turkovics and R. N. C. Pradelle (2017). "Potential of biofuels from algae: Comparison with fossil fuels, ethanol and biodiesel in Europe and Brazil through life cycle assessment (LCA)." Renewable and Sustainable Energy Reviews **73**: 632-653.

Commission, E. (2003). Directive 2003/30/EC on the promotion of the use of energy from renewable sources amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union**:** 42.

Commission, E. (2009). Directive 2009/28/EC on the promotion of the use of energy from renewable sources amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union**:** 16.

Crandall, L. (2004). "Glycerol abundance cause for concern." Inform **15**: 146-147.

Demirbas, A. (2004). "Combustion characteristics of different biomass fuels." Progress in energy and combustion science **30**(2): 219-230.

Di Lullo, G., H. Zhang and A. Kumar (2017). "Uncertainty in well-to-tank with combustion greenhouse gas emissions of transportation fuels derived from North American crudes." Energy **128**: 475-486.

Donkin, S. and P. Doane (2008). Glycerol from Biodiesel Production: The New Corn for Dairy Cattle http://ag.arizona.edu/ANS/swnmc/Proceedings/2008/14Donkin_08.pdf.

Donkin, S., S. Koser, H. White, P. Doane and M. Cecava (2009). "Feeding value of glycerol as a replacement for corn grain in rations fed to lactating dairy cows." Journal of Dairy Science **92**(10): 5111-5119.

Donkin, S. S. (2008). "Glycerol from biodiesel production: the new corn for dairy cattle." Revista Brasileira de Zootecnia **37**(SPE): 280-286.

ECCC (2018). Clean Fuel Standard, Regulatory Design Paper, Environment and Climate Change.

Ekvall, T. and G. Finnveden (2001). "Allocation in ISO 14041—a critical review." Journal of Cleaner Production **9**(3): 197-208.

Energy, A. (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta.

Eriksson, G., A. Grimm, N. Skoglund, D. Boström and M. Öhman (2012). "Combustion and fuel characterisation of wheat distillers dried grain with solubles (DDGS) and possible combustion applications." Fuel **102**: 208-220.

Falano, T., H. K. Jeswani and A. Azapagic (2014). "Assessing the environmental sustainability of ethanol from integrated biorefineries." Biotechnology journal **9**(6): 753-765.

Farrell, A. E., R. J. Plevin, B. T. Turner, A. D. Jones, M. O'Hare and D. M. Kammen (2006). "Ethanol Can Contribute to Energy and Environmental Goals." Science **311**(5760): 506-508.

Feed Opportunities from Biofuels Industries Network. (2013). "Wheat DDGS Feed Guide: Wheat Dried Distiller Grains with Solubles." Retrieved September 2017, from https://cigi.ca/wp-content/uploads/2013/02/DDGS-Feed-Guide_Revised_Jan.-2013.pdf.

Flato, G., J. Marotzke, B. Abiodun, P. Braconnot, S. C. Chou, W. J. Collins, P. Cox, F. Driouech, S. Emori and V. Eyring (2013). "Evaluation of climate models. In: climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change." Climate Change 2013 **5**: 741-866.

FOBI (2013). Wheat DDGS Feed Guide: Wheat Dried Distiller Grains with Solubles, Feed Opportunities from Biofuels Industries (FOBI) https://cigi.ca/wp-content/uploads/2013/02/DDGS-Feed-Guide_Revised_Jan.-2013.pdf Accessed 25/02/2018.

G. Angevine and V. Oviedo (2012). Ensuring canadian access to the oil markets in the asia-pacific region, Fraser institute**:** 1-62https://www.fraserinstitute.org/studies/ensuring-canadian-access-oil-markets-asia-pacific-region.

Gentleman, R., R. Ihaka, D. Bates, J. Chambers, J. Dalgaard and K. Hornik (2009). "The R project for statistical computing." URL: http://www. r-project. org/254.

Heirigs, P., P. Hodson, S. Kaffka, D. O'Connor, M. Stowers and O. Vessia (2010). "Issues Related to Accounting for Co-Product Credits in the California Low Carbon Fuel Standard." Subgroup on Issues Related to Co-Product Credits, State of California Air Resources Board, Sacramento, CA.

Henderson, S. (2000). "Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario." Agriculture and Agri-Food Canada.

Hill, J., E. Nelson, D. Tilman, S. Polasky and D. Tiffany (2006). "Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels." Proceedings of the National Academy of sciences **103**(30): 11206-11210.

Inc., S. T. C. (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions.

ISO (2006). Environmental Management–Life Cycle Assessment–Principles and Framework (ISO 14040). Geneva, The International Organization for Standardization (ISO).

Kabir, M. R. and A. Kumar (2011). "Development of net energy ratio and emission factor for biohydrogen production pathways." Bioresource Technology **102**(19): 8972-8985.

Kaufman, A. S., P. J. Meier, J. C. Sinistore and D. J. Reinemann (2010). "Applying life-cycle assessment to low carbon fuel standards—How allocation choices influence carbon intensity for renewable transportation fuels." Energy Policy **38**(9): 5229-5241.

Kendall, A. and B. Chang (2009). "Estimating life cycle greenhouse gas emissions from corn–ethanol: a critical review of current U.S. practices." Journal of Cleaner Production **17**(13): 1175-1182.

Kendall, A. and J. Yuan (2013). "Comparing life cycle assessments of different biofuel options." Current Opinion in Chemical Biology **17**(3): 439-443.

Khalili, H., T. Varvikko, V. Toivonen, K. Hissa and M. Suvitie (1997). "The effects of added glycerol or unprotected free fatty acids or a combination of the two on silage intake, milk production, rumen fermentation and diet digestibility in cows given grass silage based diets." Agricultural and Food Science in Finland **6**(5-6).

Kim, S. and B. E. Dale (2002). "Allocation procedure in ethanol production system from corn grain I. system expansion." The International Journal of Life Cycle Assessment **7**(4): 237-243.

Kodera, K. (2007). "Analysis of allocation methods of bioethanol LCA." Internship report, Faculty of Earth and Life Science, MSc in Environmental Resource and Management, Amsterdam.

Kumar, P., R. Britter and N. Gupta (2009). "Hydrogen fuel: Opportunities and barriers." Journal of fuel cell science and technology **6**(2): 021009-021009-021007.

Laboratory, A. N. (2016). GREET® Model. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation Model.

Lardy, G. P. and V. L. Anderson (2014). Feeding coproducts of the ethanol industry to beef cattle, NDSU Extension Service.

Levelton Engineering Ltd. & (S&T)^2 Consultants Inc. (1999). Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario https://ghgenius.ca/reports/AAFCCornEthanol.pdf Accessed 05/10/2015.

Linke, P., J. DeFrain, A. Hippen and P. Jardon (2004). "Ruminal and plasma responses in dairy cows to drenching or feeding glycerol." J. Dairy Sci **87**(Suppl 1): 343.

Linn, J. and M. Raeth-Knight. (2016). "A Biodiesel byproduct may substitute for corn" Retrieved 17 November, 2016, from http://www.extension.umn.edu/agriculture/dairy/feed-and-nutrition/a-biodiesel-byproduct-may-substitute-for-corn/.

Littlejohns, J., L. Rehmann, R. Murdy, A. Oo and S. Neill (2018). "Current state and future prospects for liquid biofuels in Canada." Biofuel Research Journal **5**(1): 759-779.

Lory, J. A., R. E. Massey, C. D. Fulhage, M. C. Shannon, R. L. Belyea and J. M. Zulovich (2008). "Comparing the Feed, Fertilizer, and Fuel Value of Distiller's Grains." Crop Management **7**(1): 0-0.

Luo, L., E. van der Voet, G. Huppes and H. A. Udo de Haes (2009). "Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol." The International Journal of Life Cycle Assessment **14**(6): 529-539.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**: 1249-1262.

Malça, J. and F. Freire (2004). Life cycle energy analysis for bioethanol: allocation methods and implications for energy efficiency and renewability. 17th International Conference on Efficiency, Costs, Optimization, Simulation and Environmental Impact of Energy and Process Systems.

Malça, J. and F. Freire (2006). "Renewability and life-cycle energy efficiency of bioethanol and bio-ethyl tertiary butyl ether (bioETBE): assessing the implications of allocation." Energy **31**(15): 3362-3380.

Maps, B. E. and W. E. Matters (2012). "How Biofuels Work."

Moore, A. (2011). Fertilizer potential of biofuel byproducts, INTECH Open Access Publisher.

Morse, D., H. Head, C. Wilcox, H. Van Horn, C. Hissem and B. Harris (1992). "Effects of concentration of dietary phosphorus on amount and route of excretion." Journal of Dairy Science **75**(11): 3039-3049.

Munawar, E., M. Ubaura, N. Goto and K. Fujie (2003). "Estimation CO2, Non-CO2 GHGs and Other Gas pollutant Emissions of Indonesia's Urea Fertilizer Factories." World **45**: 100.

OECD/FAO (2018). OECD-FAO Agricultural Outlook 2018-2027, OECD and Food and Agriculture Organization of the United Nationshttps://doi.org/10.1787/agr_outlook-2018-en.

Plevin, R. J., M. O'Hare, A. D. Jones, M. S. Torn and H. K. Gibbs (2010). "Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated." Environmental Science & Technology **44**(21): 8015-8021.

Preston, R. (2005). 2005 Feed Composition Tables. Beef. http://beefmagazine.com/mag/beef_feed_composition_tables/

Rausch, K. D. and R. L. Belyea (2006). "The future of coproducts from corn processing." Applied Biochemistry and Biotechnology **128**(1): 47-86.

REN21 (2017). Renewables 2017 Global Status Report. Paris, REN21 Secretariat.

Richmond, B. (2000). "Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario R-2000-1." Assessment.

Rose, S. K., E. Kriegler, R. Bibas, K. Calvin, A. Popp, D. P. van Vuuren and J. Weyant (2014). "Bioenergy in energy transformation and climate management." Climatic Change **123**(3-4): 477-493.

Ruhul Kabir, M. and A. Kumar (2012). "Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways." Bioresource technology **124**: 394-405.

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan.

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan.

Sawin, J. L., F. Sverrisson, K. Seyboth, R. Adib, H. E. Murdock, C. Lins, I. Edwards, M. Hullin, L. H. Nguyen and S. S. Prillianto (2013). "Renewables 2017 Global Status Report."

Schröder, A. and K.-H. Südekum (1999). Glycerol as a by-product of biodiesel production in diets for ruminants. INTERNATIONAL RAPESEED CONGRESS, Regional Institute Gosford.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes and T.-H. Yu (2008). "Use of U.S. Croplands for Biofuels Increases Greenhouse Gases through Emissions from Land-Use Change." Science **319**(5867): 1238-1240.

Shroyer, K. J., S. A. Staggenborg and J. Propheter (2011). "Utilization of dry distillers grains and charcoal as nitrogen fertilizer in corn." Agronomy journal **103**(5): 1321-1328.

Shurson, G. C. (2017). "The Role of Biofuels Coproducts in Feeding the World Sustainably." Annual Review of Animal Biosciences **5**(1): 229-254.

Shurson GC, Zijlstra RT, Kerr BJ and S. HH. (2012). Feeding biofuels co-products to pigs. Biofuel Coproducts as Livestock Feed: Opportunities and Challenges,. Makkar HPS. Rome:, Food Agric. Organ**:** 175-207.

Staples, M. D., R. Malina and S. R. H. Barrett (2017). "The limits of bioenergy for mitigating global life-cycle greenhouse gas emissions from fossil fuels." Nature Energy **2**: 16202.

Tago, T. and T. Masuda (2018). "Utilization of DDGS using ethanol solution for biocrude oil production by hydrothermal liquefaction AU - Mansur, Dieni." Biofuels **9**(3): 325-330.

Thacker, P. (2006). "Nutrient digestibility, performance and carcass traits of growing-finishing pigs fed diets containing dried wheat distillers grains with solubles." Canadian journal of animal science **86**(4): 527-529.

Thacker, P. A. (2006). "Nutrient digestibility, performance and carcass traits of growing-finishing pigs fed diets containing dried wheat distillers grains with solubles." Canadian journal of animal science **86**(4): 527-529.

U.S. Department of Energy. (2011). "Biomass Program" Retrieved March 25, 2018, from https://www.nrel.gov/docs/fy11osti/51241.pdf.

U.S. Department of Energy. (2018). "Alternative Fuels and Advanced Vehicles." Retrieved April 1, 2018, from https://www.afdc.energy.gov/fuels/.

U.S. Environmental Protection Agency (2014). Climate Change Indicators in the United States: Global Greenhouse Gas Emissionshttps://www3.epa.gov/climatechange/pdfs/print_global-ghg-emissions-2014.pdf. Accessed 17 March 2018.

USDA (2016). Feed Grains Database, United States Department of Agriculture (USDA)http://www.ers.usda.gov/data-products/feed-grains-database.aspx.

Wang, M., H. Huo and S. Arora (2011). "Methods of dealing with co-products of biofuels in life-cycle analysis and consequent results within the U.S. context." Energy Policy **39**(10): 5726-5736.

Wang, M., H. Huo and S. Arora (2011). "Methods of dealing with co-products of biofuels in life-cycle analysis and consequent results within the US context." Energy Policy **39**(10): 5726-5736.

Weidema, B. P. (1999). System expansions to handle co-products of renewable materials. Presentation summaries of the 7th LCA case studies symposium SETAC-Europe, Citeseer.

World Energy Council (2016). World Energy Resources Bioenergy. 2016 https://www.worldenergy.org/wp-content/uploads/2017/03/WEResources_Bioenergy_2016.pdf Accessed 10/04/2018.

Yizhen, Z. (2018). Life Cycle Environmental Impacts of Biofuels: The Role of Co-products. Doctor of Philosophy, University of California.

Zaimes, G. G. and V. Khanna (2014). "The role of allocation and coproducts in environmental evaluation of microalgal biofuels: How important?" Sustainable Energy Technologies and Assessments **7**: 247-256.

Chapter 5: A Literature Review on Consequential Life Cycle Assessment of Biofuel Energy systems[4]

5.1 Introduction

There is a growing global concern on climate change and how to combat its adverse impacts (Fuss et al. 2014, Millar et al. 2017). The anthropogenic greenhouse gas (GHG) emissions increased annually by 2.2% on average between 2005 and 2015 (Le Quéré et al. 2018, Le Quéré et al. 2019), mainly due to rapid increase in global population and the demand for fossil-based energy. In order to limit global warming well below 2^{o} C, there should be massive transition towards low carbon economy (Van Vuuren et al. 2016). The transportation sector emitted 23% of the global GHG emissions in 2014. The majority of emissions came from the production and use of the fossil fuels in vehicles (IEA 2017). Canada, the world's 9th largest emitter, contributed around 1.6% of the global GHG emissions in 2017, around 716 $MtCO_2$-eq, an 8 $MtCO_2$-eq increase from 2016 (Environment and Climate Change Canada 2018, 2019a, 2019b). The transportation sector and the oil and gas sector were the major GHG emitters and together contributed more than 50% of the emissions (Environment and Climate Change Canada 2019a). Alberta, the fourth most populous province in Canada, produced 273 $MtCO_2$-eq in 2017 (Environment and Climate Change Canada 2019a), around 38% of the country's total, which is alarming. Alberta's transportation sector is one of the major contributors of the province's total GHG emissions, next to oil and sands sector (Environment and Climate Change Canada 2019a).

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[4] A version of Chapter 5 will be submitted to a peer-reviewed journal for publication, titled as "Nafisa Mahbub, Eskinder Gemechu, and Amit Kumar. A Literature Review on Consequential Life Cycle Assessment Methodology and Applications in Biofuel Energy systems."

A number of GHG mitigation options have been introduced to combat climate change. Different environmental strategies, regulations, and policies have been proposed on local, regional, and global scale to support the effective application of the GHG mitigation options. Examples include the Paris Agreement with a target to keep the increase in the global average temperature below 2°C above pre-industrial levels (United Nations Climate Change 2018), Canada's Clean Fuel Standard that encourage the use of low carbon fuels (Environment and Climate Change Canada 2018), Alberta's Specified Gas Reporting Program (Government of Alberta 2019b), and Alberta's Renewable Fuels Standards, which mandates blending of 5% ethanol in conventional gasoline used in Alberta (Government of Alberta 2019a). A thorough quantitative assessment of the environmental, social, economic, and technological viability, and market acceptance of the proposed strategies or policy decisions are mandatory before application, which can be evaluated using life cycle assessment (LCA).

LCA is a thorough environmental impact assessment tool, used to evaluate the environmental, economic and social performance of a product system taking into account its full life cycle stages, from resource extraction to final disposal. Depending on the types of question to be addressed, LCA can be conducted using two modelling approaches, attributional or consequential (Thomassen et al. 2008). Attributional LCA (A-LCA) assigns inputs and outputs from a particular unit process of a product system to a defined functional unit using a normative allocation and market average data. Consequential LCA (C-LCA) is a change oriented approach which aims at evaluation of the potential consequences of a product system due to change in the demand of the functional unit. Marginal data and system expansion approach are used in the C-

LCA to quantify the long-term environmental consequences of changes in the product system (Thomassen et al. 2008, Reinhard and Zah 2009, Vázquez-Rowe et al. 2014). The concept of consequential LCA was first proposed in the year of 1993 by Weidema (Weidema 1993), who mentioned about the requirement of assessing the marginal technologies while comparing different alternative technologies on a life cycle basis, which is the underlying principle of C-LCA. The use of the term C-LCA, its' definition, methodological background, historical evolution, and difference from A-LCA came into discussion since late 20th and beginning of 21st century (Frischknecht 1998, Ekvall 1999, Ekvall 2002, Ekvall and Weidema 2004). Debates and disagreements have been evident in literature published during the last decade regarding A-LCA and C-LCA modelling approaches (Thomassen et al. 2008, Dalgaard and Muñoz 2014).

This chapter focuses on reviewing how C-LCAs, methodologies and applications, have been used to address different policy questions related to the wide deployment of biomass based energy system. Biomass, a form of renewable energy, is most commonly used as an alternative source of energy (fuels, electricity, space heating etc.) resulting in lesser GHG emissions and environmental pollution compared to conventional fossil sources. There are different biomass sources such as, solid wood and wood waste used for space heating and electricity, forest biomass, forest residues, agricultural crops and residues, sawdust, and wood waste used to produce transportation biofuels, food waste in garbage burned to produce electricity or biogas, animal manure and municipal solid waste converted to biofuels. Biofuel is one of the most sustainable ways for reducing GHG emissions from the transportation sector as it makes the combustion cleaner when mixed with conventional fuels. Although these renewable energy sources are environmentally sound but they have not been always economically sustainable, which makes the commercial application of these technologies challenging in some jurisdictions. Policy decision needs to be both environmentally friendly and economically viable which can be determined or verified by applying C-LCA.

Several A-LCA studies have been conducted to calculate the life cycle environmental burdens attributed to a specific amount of functional unit of the biofuel produced from the system investigated (Adler et al. 2007, Cherubini et al. 2009, Cherubini and Strømman 2011, Collet et al. 2011, Mahbub et al. 2017). However, limited studies have been published addressing the impacts of producing any additional biofuel to meet any particular policy or regulations applied on a regional, national, or global basis. Any additional biofuel demand can produce significant environmental, economic and social consequences locally, regionally, or globally, which can be captured using C-LCA. Any additional production of biofuel will impact the supply and production of other fuels (including both renewables and non-renewables) in the market and other interlinked economic sectors such as food, agriculture, and so on in the global market, which cannot be captured by the A-LCA method.

C-LCA is a change oriented approach that captures the consequences from a change in the investigated system; hence it is preferred over A-LCA approach to determine the long-term environmental consequences of any decision or changes (Marvuglia et al. 2013, Hamelin et al. 2017). However, a recent review study showed the challenges in conducting C-LCA of bioenergy system. A recent study conducted a review on the C-LCA modelling approach in bioenergy system since 2007 with the aim of identifying the main topics covered, the methodological approach and scenarios covered (Roos and Ahlgren 2018). However, the study did not discuss the results/findings of the different C-LCA studies in detail, which could be helpful to understand the application of C-LCA in the field of bioenergy systems (Roos and Ahlgren 2018). Furthermore, the study did not elaborate how different C-LCA approaches helped resolving different policy questions previously, what are the crucial aspects to consider while conducting a C-LCA of biofuel energy systems, how to overcome the methodological limitations of C-LCA applied in the bioenergy systems, and how to enhance the implementation of C-LCA in different bioenergy applications.

The application of C-LCA as a decision-making tool has been in discussion since last decade and gradually it is getting a lot of attention as it can evaluate the possible consequences of any large-scale environmental policy or decision implementation in future (Earles and Halog 2011). However, C-LCA is still an emerging approach compared to the conventional A-LCA. Also, there is no concrete framework for its effective and validated application. Limited empirical or application oriented C-LCA studies on biofuels have been published. As a result, modelers still encounter uncertainties regarding the system delimitation, identification of the marginal technology also known as the affected technologies, identification of competing products/ substitutes, modelling approaches and so on. The objective of this paper is to review the current state of the work on C-LCA of biofuels, explore the research areas covered, identify the research gaps and recommend areas which needs further investigation. The review paper discusses the following issues crucial to the application of C-LCA in the field of biofuel energy systems including:

- different C-LCA approaches used in studies resolving different policy questions related to biofuel policy or program deployment,

- the crucial methodological aspects to consider while conducting a C-LCA such as marginal technology, allocation issues, competing technology, market information, C-LCA modelling, application areas covered, uncertainty,
- the major methodological limitations while applying an C-LCA, and
- recommendations on how to enhance the implementation of C-LCA in different biofuel applications.

The key findings in this chapter can be helpful to the researchers, modelers, or policy makers interested in biofuel C-LCA as it highlights and discusses the recent biofuel C-LCA studies, uncertainties involved in the methodological framework, research gaps, areas to still explore and application of the study findings.

5.2 Method

5.2.1 Systematic Literature Review

A systematic literature review has been conducted. The review process involves: establishing the main research question, developing a set of criteria to include/ exclude articles to be reviewed, asses the quality of research on the selected papers to be included, critically reviewing the papers, narrating the key results with arguments, identifying the knowledge gaps and recommendations for future research (Ramdhani et al. 2014). The steps followed in conducting the systematic literature review have been acknowledged by number of authors (Magarey 2001, Ramdhani et al. 2014, Machi and McEvoy 2016, Roos and Ahlgren 2018) in literature. The environmental benefits from using biofuels have been discussed frequently in literature; however, the long-term consequences of biofuel production technologies and use, when applied in a large-scale addressing either any climate change mitigation policy target are not discussed precisely. There are limited number of studies on C-LCA of biofuels, however, methodological discrepancy and lack of standard framework hinders its appropriate application in the biofuel area. Hence, we selected the topic of C-LCA of biofuel energy systems to conduct a systematic literature review. We sorted and finalized papers that have discussed either the C-LCA methodological aspects or applications in the biofuel areas. Peer reviewed journal articles (both original papers and review articles that discuss the methodological framework, empirical or application of C-LCA) mostly published during the last 12 years were considered for the review. Relevant technical reports, standards, and book chapters published within the same time frame were also taken into account. The keywords used to search articles for the review include life cycle consequential assessment, C-LCA, bioenergy, biofuels etc. A table/ synthesis matrix has been used to organize the major findings from different reviewed articles.

The research question was to analyze different articles and research papers that discussed different methodological developments of C-LCA and the long-term consequences of biofuel production and applications due to a large-scale policy implementation or any climate change mitigation-oriented decision. The main purposes of this chapter are to: provide a comprehensive review on the current state of the art of C-LCA conducted on biofuel energy systems; highlight research gaps; analyze and synthesize findings to identify the areas of improvement; and provide recommendations for future research to enhance the quality or accuracy of the C-LCA studies on bioenergy and biofuels. To meet this objective, we analyzed the current state of the art of C-LCA on biofuels, identified the major gaps, limitations from the existing studies and discussed them under different categories such as system boundary and allocation, (in the result sections) along with the areas of improvement for effective and random application of C-LCA in the field of biofuel energy systems. The steps that have been followed to conduct the literature review in this paper are shown in figure 1.

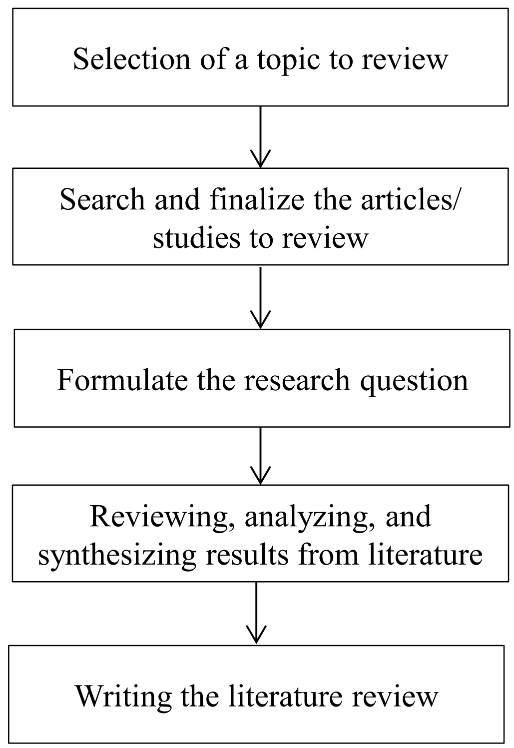

Figure 5. 1: Methodology for the literature review

5.3 C-LCA Results and Discussion

This section discusses the issues which are found to be crucial to the applications of C-LCA in biofuel and bioenergy systems. Both methodological and empirical aspects have been discussed along with the knowledge gaps and recommendations to address the gaps.

5.3.1 Allocation Approach and System Boundary/ System Delimitation

Allocation is among the most widely discussed and explored topics by the LCA practitioners both for the A-LCA and C-LCA (Weidema 2003, Guinée et al. 2004, Thomassen et al. 2008). LCA results from the same pathway/ product system can be significantly different due to its multifunctional nature and applying different allocation approaches to it (Inc. (S & T)2 Consultants 2013).

Literature showed that, A-LCA studies mostly use physical allocation between the main products and coproducts, whereas the C-LCA studies mandatorily use the system expansion approach to include the coproduct use in the system hereby avoiding allocation (Ekvall and Andrae 2006, Thomassen et al. 2008). These certainly agree with the ISO (International Organization for Standardization) guidelines for LCA (International Organization for Standardization 2006-07, International Organization for Standardization 2006-07). According to the ISO guideline, primarily the allocation should be avoided whenever possible using either subdivision or system expansion. Secondly, if it can't be avoided allocation factors based on physical causal relation should be used to allocate energy and emissions between main product and coproducts. Finally, if physical causal factors can't be determined other factors should be used for energy and emissions allocation (International Organization for Standardization 2006-07, International Organization for Standardization 2006-07). As C-LCA seeks to determine the future consequences of any actions (like changes in demand/production) on the subsequent processes, system expansion approach is used mandatorily. Hence, C-LCA needs to include all the affected processes within the system by expanding the boundary.

However, researchers have found that, the ISO guidelines ragarding the allocation of multifunctional processes cannot be applied to different situations in a similar way (Ekvall and Tillman 1997, Baumann 1998). Different allocation approaches apply to different systems to be investigated depending on the goal and scope of the study (Thomassen et al. 2008). A-LCA approach uses physical factors based on energy content, monetary value, or mass to allocate the environmental burdens among main product and its coproducts (Thomassen et al. 2008). System expansion has also been used in A-LCAs (Inc. (S & T)2 Consultants 2013, Mahbub et al. 2019). For example, Mahbub et al. have assessed the environmental benefits of using coproduct credits in two different biofuel life cycle, namely bioethanol and biodiesel (Mahbub et al. 2019). Canola meal and DDGs (dry distiller grains) were coproduced with bioethanol, whereas canola meal and crude glycerine were produced as coproducts in the biodiesel conversion process. The authors identified different alternative uses of the biofuel coproducts using literature information and experts' judgement and calculated the avoided emissions by determining the displacement factors based on energy content between the coproduct and the replaced product (Mahbub et al. 2019). The replaced products were assumed to have the same functionality as the biofuel coproducts (Mahbub et al. 2019). However, the authors did not include any actual market analysis, any change in biofuel demand, and the marginal processes that can be impacted by the demand change within the system boundary. Lack of these practical economic causal factors made the results volatile and less reflective of the actual coproduct credits and their market value. C-LCA approach can be used further to determine the actual market penetration of these biofuel coproducts considering the processes impacted by any demand changes for biofuel.

Very few studies have shown demonstration and comparison of both A-LCA and C-LCA modelling approaches on the same case (Ekvall and Andrae 2006, Thomassen et al. 2008). The authors recommended to demonstrate both type of modelling for similar cases, as the outcomes were significantly different. Thomassen et al. found that the environmental impacts varied significantly when two different LCA approaches were used to assess the life cycle emissions from average milk production in The Netherlands (Thomassen et al. 2008). The energy use and climate change impacts determined by the C-LCA approach were found 63% and 44% less than the values obtained by the A-LCA, respectively (Thomassen et al. 2008). The reduced numbers were mainly due to the avioided emisisons from the alternative use of the coproducts such as beef production in the C-LCA and the difference in the animal feed types required by the animals for increased milk production. In the A-LCA approach, the concentrates produced in the farm were used as animal feed whereas the C-LCA approach used system expansion to include soybean meal and spring barley as animal feed C-LCA. The energy and protein requirements were used to calculate the amount of the animal feed (soybean meal and spring barley) in the C-LCA approach, whereas the average real life data were used for the in-house concentrates. Although the soybean meal and spring barley produced less environmental burdens, the animal feed experts did not recommend such feed compositions as they lack in proper nutrition when used for long period resulting in less milk production (Thomassen et al. 2008). However, C-LCA approach was found less practical and acceptable for the long run by the feed experts or nutrionists compared to the A-LCA. We need to come up with more alternate solutions or ways to fill such gaps between the theotical estimates and practical acceptance; for which further research is needed. Similarly, we need diversified examples on simultaneous modelling of A-

LCA and C-LCA approaches for specific biofuel cases in order to get more transparent and reliable life cycle results.

In C-LCA approach, the allocation of environmental burdens across a multifunctional system is decided by the response of the system to the demand change (Ekvall and Weidema 2004). Ekvall and Weidema have briefly mentioned about different multifunctional scenarios along with relevant recommendations on how to delimit the system boundary to allocate energy and emissions across multiple output products (Ekvall and Weidema 2004). For example, when the main product and coproduct in a multifunctional system are produced independently, any changes in demand will result in a major change in the production of the main product only rather than the coproduct. The authors suggested subdivisions of the multifunctional system for allocation or physical or causal allocation similar to the one used in A-LCA approach. Secondly, when the production depends on demand for the main product, any changes in the demand will significantly impact the production of both products. In this case, the authors suggested the system boundary to include all the processes related to the use/disposal of the coproduct, the competing products, and the main product (Ekvall and Weidema 2004). Finally, when the production depends on the demand for the coproduct, any changes in demand will have very small impact on both the production systems. The researchers recommended, the system boundary should include the use and disposal of the main product along with all the processes related to the production, use, and disposal of the competing products (Ekvall and Weidema 2004). Although Ekvall and Weidema covered a range of possible multifunctional scenarios and suggested how to delimit the system boundary in the C-LCA, demonstration of these scenarios throughout the life cycle of any product or system were not included. The authors also declared the suggestions to be merely theoretical (Ekvall and Weidema 2004), which lack in reflecting practical situations. The authors further claimed the suggested scenarios to be only effective while assessing the consequence due to the demand change for the main product and large uncertainties are involved while deciding to what extent the processes are affected due to the demand change. The dependence or independence between the production and demand varies with time and market conditions (Ekvall and Weidema 2004), which makes the suggested scenarios ambiguous and volatile to apply for system delimitation.

While conducting the literature review, none of the papers discusses the system delimitation for different types of biofuel multifunctional systems. There is very limited discussion in these studies on why, what, and how to include/exclude processes related to the production, use, and disposal of the main product, coproducts, and the competing products as a consequence of a demand change or any other change in the biofuel system throughout the life cycle. Therefore, it can be concluded that, further research is required to get realistic examples on system delimitations in the biofuel multifunctional systems across the life cycle.

5.3.2 Marginal Technology, Coproducts and Competing Products

Marginal technology refers to a technology that is affected by a small change in the demand for the product under investigation (Weidema et al. 1999). Weidema et al. suggested a five-step method for identifying the marginal technology in the life cycle and demonstrated the method by identifying long-term marginal technology for the paper and pulp industry and European electricity generation under the ceteris paribus condition (Weidema et al. 1999). However, the examples did not reflect any specific local or regional situation, which compromised the accuracy of the results. Additionally, the study did not demonstrate any case that goes beyond the ceteris paribus condition, which occurs when the demand change requires a large substitution or capacity adjustment affecting the overall market trend, the production costs of the technologies involved, and other constraints (Weidema et al. 1999). In this case, the change is not referred as a marginal change and a totally different product/technology substitutes the product under study as a consequence of the demand change, which was not covered by Weidema et al. However, the authors emphasized the importance of identifying the marginal technology as it can reflect the actual consequence of a demand change in the system under investigation (Weidema et al. 1999).

Weidema et al. claimed that marginal data are less uncertain and more stable than average data in comparative life cycle studies (Weidema et al. 1999). While using average data to determine the product substitution in the life cycle assessment, wide range of data for different product/technologies from different sources are collected which adds the uncertainty (Weidema et al. 1999). Furthermore, the average data lacks in considering time variation making the result less accurate. Additionally, in case of multifunctional system, an arbitrary allocation factor needs to be determined to divide the burdens among the main products and coproducts while using the average data adding more uncertainty, which can be avoided by using marginal data (Weidema et al. 1999). However, it is not possible to find marginal technology for all type of the products/technologies so easily, which makes the LCA modelers rely on the average data to identify the marginal technology, hence adding uncertainty. Since, the marginal technologies are only affected by the boundary constraints, long term production cost, and development in the marginal technology, using marginal data gives more stable results than the average data since it keeps changing even with a smaller capacity adjustments (Weidema et al. 1999).

A number of studies (Ekvall and Weidema 2004, Earles and Halog 2011, Hamelin et al. 2017) have mentioned about the five-step method for identifying the marginal technology (Weidema et al. 1999). Earles and Halog even mentioned that the success of the C-LCA application largely depends on how effectively the affected technology (marginal technology) is decided and included in the system boundary, since C-LCA determines any future potential consequence due to the demand change of the product under study (Earles and Halog 2011). However, a detailed demonstration of this method has not been found in application-oriented life cycle studies except in Weidema et al 1999 and 2003 (Weidema et al. 1999, Weidema 2003).

Studies have applied the five-step method and derived conclusions on how to finalize the marginal technology and include in the study, which is briefly summarized here. Ekvall and Weidema and Hamelin et al. concluded that constrained technologies can't be chosen as marginal technology both in short-term and long-term. For short-term, the technologies that can change/ use the capacity partially in response to the demand change keeping the existing capacity constant are identified as the marginal technologies (Ekvall and Weidema 2004, Hamelin et al. 2017). Whereas in long-term, the technology capacity adapts to the changed demand keeping the capacity use constant (Ekvall and Weidema 2004). The identified long-term marginal technologies are either soon to be phased out due to higher short-term expenses or the technologies with additional capacity installed due to the increasing demand trend as well as lower long-term operating cost (Earles and Halog 2011, Hamelin et al. 2017). However, these studies have not demonstrated the observations with life cycle study examples specific to local or regional situations so that it can used as a reference for similar scenarios.

Most of the C-LCA studies on biofuels or bioenergy system lack discussing the assumptions or its rationale while identifying and selecting marginal technologies, which impedes the reliability of the results. Mathiesen et al. also classified marginal technologies into three different groups simple, dynamic, and complex marginal system depending on their adaptability to demand changes (Mathiesen et al. 2009). The researchers used this classification to determine the marginal technology for the Danish electricity production based on the historical publications on Danish energy systems. The authors have identified the business as usual, planned/ proposed, and the actual marginal technologies (either simple, complex, or dynamic) from the studies related to policy, government energy plans or other long-term energy perspectives such as capacity installment or demand change (including both conventional and bioenergy/ biofuel systems) from 1976 to 2006 (Lund and Mathiesen 2009, Mathiesen et al. 2009). However, the historical studies used for assessment were mostly the economic feasibility studies that used projections and historical statistics on regional energy systems to identify the marginal technologies (Mathiesen et al. 2009). The authors found that the actual marginal technology identified was not identical to the theoretical one and the marginal technologies were not consistent, they varied with time (Mathiesen et al. 2009). The observation from the study indicates the requirement to invest in more than one marginal technology based on the nature of the energy system involved (Mathiesen et al. 2009), however, the authors did not explain any specific solution to this complex situation rather than some generic recommendations.

It has been further argued that, in reality the marginal effects are comprised of both short-term and long-term effects which are very complex to model (Ekvall and Weidema 2004). Hence as a simplification, the modelled systems either include long term or short-term effects, preferably the long term since the C-LCA studies seek to determine the long-term environmental consequences of a demand change on the market and the cumulative short-term effects result in the long-term effects (Ekvall and Weidema 2004, Mathiesen et al. 2009). This is certainly not reflective of real world scenario, hence affecting the accuracy of the life cycle results.

Ekvall and Finnveden have recommended to include competing products in the multifunctional systems where either the coproduct demand determines the production of the main product or where constrained or competitive production resources (renewables/non-renewables) are used in C-LCA (Ekvall and Finnveden 2001). Another step-by-step method has been recommended by Ekvall and Weidema (Ekvall and Weidema 2004) to identify the competing/ substituted products, which include: establishing the main function or the obligatory properties of the coproduct, identifying the market affected due to the production and use of the coproduct, and finally, identifying the competing products substituted/displaced by the coproduct in the market selected in the previous step. Although the method emphasized to identify and include all potential competing product substituted by the coproduct, the biofuel LCA studies mostly either exclude coproduct use stage or include a single competing product. A few biofuel LCA studies have identified a range of applications for biofuel coproducts. Bioethanol coproduct DDG can substitute animal feed or fertilizer or can be used as a space heating source replacing natural gas (Andrew and Forgie 2008, Saha 2010, Mahbub et al. 2019). Biodiesel coproduct crude glycerine can be upgraded and can substitute synthetic glycerine, fuel, or animal feed in the market (Linn and Raeth-Knight 2016, Mahbub et al. 2019). These biofuel coproducts have been found to substitute more than one competing products in the relevant market which needs to be considered in the C-LCA modelling as well for enhancing the model accuracy and applicability.

Further research is necessary since application-oriented studies can explain the practical allocation facts and the uncertainties associated with the identifying the marginal technology and competing product better than the theoretical recommendations. This is crucial to develop a strong foundation of C-LCA framework so that it can be applied in practical situations.

5.3.3 C-LCA Scenarios and Modelling Approach

Marvugila et al. mentioned that C-LCA can model different types of scenarios related to biofuel production and large-scale policy implementation without compromising the actual magnitude of the implementation under the study (Marvuglia et al. 2013). As the A-LCA assumes the ceteris paribus conditions (fully elastic market) in case of any demand change and it determines the environmental burden for a specific amount of product ignoring the surrounding relevant product systems and markets (Marvuglia et al. 2013). Different studies suggest that C-LCA approach overcomes these limitations by modelling different biofuel scenarios, namely normative, predictive, and explorative scenarios supporting policy or decision making (Börjeson et al. 2006, Finnveden et al. 2009, Marvuglia et al. 2013). . The normative scenarios assess the environmental consequences from a biofuel technology with a specific objective whereas; a predictive scenario predicts the consequence of a large-scale application of a new technology in the market (Finnveden et al. 2009, Marvuglia et al. 2013). Explorative scenario can assess the environmental impacts from large scale application of a particular biofuel technology from different perspectives such as industrialist or policy makers and can model it either for countrywide or worldwide applications (Finnveden et al. 2009, Marvuglia et al. 2013). Although these classification of biofuel scenarios, gave us the idea about the type of cases the C-LCA studies can assess, the authors did not suggest any specific modelling approach particular to this classification or the importance of the classification while selecting a C-LCA model.

Furthermore, the authors suggested three different modelling approaches which have been mentioned in literature to model different C-LCA scenarios (Schmidt 2008, Earles et al. 2013, Marvuglia et al. 2013, Vázquez-Rowe et al. 2014), such as simplified model, Partial Equilibrium (PEM) model, and Computable General Equilibrium (CGE) model. A number of criteria are mentioned by Marvuglia et al. while selecting a C-LCA modelling approach, namely, market delimitation, production scale, time scope, constraints related to market, multifunctionality, policy, and technology, multifunctional process, direct indirect land use changes and finally, assessing the demand change or any other decision change. However, no C-LCA studies have followed these criteria while selecting a modelling approach to solve a particular scenario. Moreover, the researchers also did not include any examples or demonstration to clarify the role of these criteria to select a modelling approach (Marvuglia et al. 2013). Further research is needed so that more scenarios are assessed to have a better idea on how to select a particular C-LCA modelling approach based on the above mentioned criteria.

5.3.3.1 Simplified approach

Many researchers have mentioned about the simplified approach as a C-LCA modelling tool, which is based on assumptions like fully elastic market where supply meets demand equally

(Schmidt 2008, Weidema et al. 2009, Marvuglia et al. 2013). The simplified approach, mostly used to model the consequences from a biofuel demand change on the agricultural sector (Marvuglia et al. 2013). The advantage of using the simplified approach is that it can assess the consequences in both constrained and unconstrained market based on either long-term or short-term context assuming a fully elastic market where any additional demand can be fulfilled by the available supply. However, the simplified approach assumes the ceteris paribus condition, which does not consider the relevant product systems and markets affected by the demand change. As a result, the simplified approach is not preferable while modelling the consequence of any complex decision context.

5.3.3.2 Partial Equilibrium Model

On the other hand, partial equilibrium model (PEM) considers the affected economy. However, it assesses the environmental consequences of a decision based on a particular economic sector (Witzke et al. 2008, Marvuglia et al. 2013). The limitation of using a PEM is that, it can't model the interaction among multiple economic sectors due to a demand change in a particular economic sector. However, PEM can assess the consequences of a decision at different levels ranging from local, industry, regional, to national level (Earles and Halog 2011). For example, Vázquez-Rowe et al. have used a PEM model at the industry level to maximize the farm/ industry profits, activities, or functions or minimizes loss (Vázquez-Rowe et al. 2013, Vázquez-Rowe et al. 2014) whereas, other researchers have used the PEM model at regional or national level to determine the consequential impacts from a large-scale application of a biofuel related policy such as national biofuel policy (Francois and Reinert 1997, Adams et al. 2005).

PEM is actually an economic model that uses the relationship between price and demand-supply elasticity (Marvuglia et al. 2013), and incorporates market information into the LCA (Earles and Halog 2011). However, some researchers have argued that PEMs should be simultaneously used with LCA, but they should be applied individually (Bouman et al. 2000), whereas some authors have emphasized integration of these two models (Ekvall 2002, Ekvall and Andrae 2006). When PEM is integrated with LCA, it can better assess the impacts including both direct and indirect consequences resulting from a decision/ change in the system. PEMs can be both simpler and complex depending on the scope of the study. The simpler models assess the consequence of a change on particular activities. For example, the impact of biofuel demand increase on the consumption of regular conventional fossil fuels such as gasoline, diesel etc. On the contrary, PEMs can also be larger and complex as well considering hundreds of different types of goods and commodities across multiple markets, regions or countries, also known as Multi-Market, Multi-Region Partial Equilibrium Model (MMMR-PE) (Roningen 1997, Adams et al. 2005). Some examples of MMMR-PE models include Food and Agricultural Sector Optimization Model (FASOM, Food and Agricultural Policy Research Institute (FAPRI), AGLINK/COSIMO, IMPACT models which have been found to determine the environmental impacts associated with indirect land use changes (ILUC) from agricultural sectors due to increased biofuel demand as well as large scale biofuel policy implications, such as the national biofuel policy by US Environmental Protection Agency (EPA) (Searchinger et al. 2008, Fabiosa et al. 2010, Msangi et al. 2010, Sissine 2010). All these models are developed with a focus on the agricultural sector mostly; further research is required to have some PEMs focusing on other sectors such as transportation, industrial etc. since biofuels are mostly consumed in these sectors.

5.3.3.3 Computable General Equilibrium Model

Researchers have applied another C-LCA modelling CGE (Computable General Equilibrium) which is a nonlinear optimization model that includes all the sectors impacted within an economy to determine the consequence of a change in demand/supply/policy or any other change in the system (Ekvall 2002, Earles and Halog 2011, Yang and Heijungs 2018). Similar to PE models, the CGE (Computable General Equilibrium) models mostly assess the consequences of a large-scale change related to energy, environment, market, tax, trade, or policy application (Fortuna and Rege 2010, Earles and Halog 2011, Marvuglia et al. 2013). Although CGE is broader than PE, the model still lacks accuracy as it uses highly aggregated data at macroeconomic level. Sector level detail information are neither modelled nor be traced from the results of the CGE which impedes the realistic application of C-LCA. The most commonly used CGE model in the C-LCA literature is the GTAP (Global Trade Analysis Project) (Kloverpris et al. 2008, Kløverpris 2009, Dandres et al. 2011). Kloverpris is et al. have used the GTAP model to determine the ILUC impacts in terms of GHG emissions and other pollutant emissions from the agricultural sector due to demand/supply shock (change) (Kloverpris et al. 2008), and land conversion by affected ecosystem biomes (Kløverpris 2009). Researchers have also used the GTAP model to assess large-scale biofuel energy policy implication such as European bioenergy policy, national biofuel policy by US EPA etc. (Sissine 2010, Dandres et al. 2011). The GTAP models used in these studies have been found to determine the changes in the productions from up to 57 economic sectors across 87 to 113 regions globally (Kloverpris et al. 2008, Dandres et al. 2011). Different statistical, government, and public databases along with publicly available literature are used to collect data on production from different economic regions to run GTAP (Dandres et al. 2011), however the high level data aggregation from different sources or inventories adds high uncertainty to the GTAP model results. In case of biofuel on of the inevitable consequence is the impact on the land use change due to increased biofuel demand or production. The C-LCA modelers mostly use three common approaches to determine the consequence of a biofuel demand change on land use both in the PEM and GGE modelling, such as, expansion, intensification, and displacement (Kløverpris 2008, Reinhard and Zah 2009, Earles et al. 2013). However, Reinhard and Zah also mentioned that it is not practical to apply these approaches due to the extent of the consequence caused by the demand change and the size of the system (Reinhard and Zah 2009). Additionally, these approaches can be used to determine the consequences on land use from biofuel demand change produced from agricultural feedstocks only, rather than other feedstocks such as forest biomass, energy crops etc.

5.3.3.4 Linear and Non-linear CLCA Model

Researchers Yang and Heijungs have classified the C-LCA modelling approaches differently, based on the mathematical computational level, such as linear and nonlinear model. Examples of linear models include process based LCA, Input Output based LCA and PEM, whereas CGE models are the nonlinear optimization models (Yang and Heijungs 2018). Linear models assume a fully elastic market supply and a linearly proportional relation between the input and demand requirement resulting in a fixed input-output coefficient which contradicts the concept of economies of scale (West 1995, Yang 2016) and fails to model the demand consequence for complex product system.

Nonlinear models, on the other hand, consider the economies of scale and can model the demand interaction across multiple sectors (Yang and Heijungs 2018). However, both types of models have been argued to be based on assumptions with higher level of uncertainty and limitations to apply in practical situations (Yang and Heijungs 2018). Hence, the researchers have recommended not to rely on a particular C-LCA model result rather than the results from different types of C-LCA models to enhance the credibility of the decision making (Yang and Heijungs 2018). As results from different C-LCA modelling approaches will help to identify the pattern (such as scattered means poor model agreement and vice versa) of the consequence which makes the result more accurate with better predictive capability. This recommendation is particularly helpful for the policy/regulation makers or sustainability researchers who rely on the results from the modelling of large-scale policy application or any other change in the biofuel life cycle. Apart from the above-mentioned models which are commonly used in C-LCA modelling, some other infrequently used models are system dynamics integrated assessment model (IAM), and linear optimization model (Dowlatabadi 1995, Duchin and Levine 2011, Stasinopoulos et al. 2012). However, the application of these models are very rare, hence not encouraged to implement for any biofuel C-LCA scenario at this point.

Different economic modelling approaches combined with LCA models have been developed and used in different C-LCA studies on biofuel systems based on the scope and intent of the study (Witzke et al. 2008, Dandres et al. 2011, 2012). However, there is no structured and consistent C-LCA framework yet that can be used as a standard guideline to C-LCA application. Thus, more examples of application oriented C-LCA studies combined with economic equilibrium models or any other mathematical models related to biofuels are required to demonstrate practical and robust application of C-LCA in the field of bioenergy systems.

5.3.4 Environmental Impacts and Applications of C-LCA in Biofuel Energy Systems

The C-LCA models can determine the environmental consequences from different applications related to the production and use of different biofuel energy systems. Some of the common areas of application include assessing different biofuel production and processing technologies, comparing different biofuel pathways based on different feedstock, and assessing the tradeoff between different applications of the same biofuel/ biomass feedstock (Melamu and Von Blottnitz 2011, Prapaspongsa et al. 2017). Most of these studies were conducted with a view to either policy analysis or researching different biofuel production and processing technologies.

Majority of the biofuel C-LCA models have mostly focused on the environmental consequences like GHG emissions (global warming/ climate change indicator), direct indirect land use changes to assess the consequence of a demand change in the system under investigation (Pehnt et al. 2008, Reinhard and Zah 2009, Garraín et al. 2016, Prapaspongsa and Gheewala 2016, Tonini et al. 2016). Researchers used these indicators to assess the future consequence of any demand change triggered by any demand change, as they create the highest impact. However, other impact categories including eutrophication, acidification, fossil depletion, human health damage, ozone depletion, ecotoxicity, resource depletion etc. are also determined in different biofuel C-LCA studies (Vázquez-Rowe, Marvuglia et al. 2014, Styles, Dominguez et al. 2016) depending on the decision context assessed by the C-LCA model.

Number of biofuel C-LCA studies have been driven by analysis of different regional, global, and national level policy implementation such as US renewable fuel standard, US Energy Policy Act, EU energy policy, EU biofuel policy, Danish renewable energy target, Luxembourg energy policy and so on (Melamu and Von Blottnitz 2011, Dandres et al. 2012, Karlsson et al. 2015). The C-LCA studies on biofuels also analyzed different production technologies or crop systems and make recommendations based on the measured consequences to aid large scale biofuel policy implementation and biofuel program deployment. Marvugila et al. have assessed the environmental impacts associated with the land use change, crop production pattern change, and land quality transformation impacts due to producing additional 80,000 tonnes of maize for bioenergy production in Luxembourg with a view to achieving the 20/20/20 target of the European bioenergy policy which mandates producing at least 20% of total energy share using renewable sources by 2020 (Marvuglia et al. 2013, Vázquez-Rowe et al. 2014). Vazquez-Rowe et al. considered the entire bio methane production system up to national grid supply along with the import/export flow of commodities due to expansion and displacement of crops to meet the shock of 80,000 tonnes of maize (Vázquez-Rowe et al. 2014). These types of holistic scenario assessment results are helpful to policy makers and stakeholders as it can give a bigger picture of the entire production system prior to implementation.

Broadening the scope of analysis, another researcher, Macintosh et al. developed eight different scenarios to assess different forest management strategies (including fossil replacement by forest bioenergy) of the Southern Forestry Region (SFR) of New South Wales (NSW), Australia with respect to three policy assumptions, such as baseline, global, and national. The study showed how different policy context can produce different results from the forest management strategies (Macintosh et al. 2015). The advantage of this kind of analysis is that it helps the decision maker to amend the scope of the strategies based on the pros and cons obtained from the diversified scenarios.

When the biofuel demand increases, local productions may not always be the solution. This triggers the dependence on the regional and global import export to meet the extra demand. Reinhard and Zah assessed the environmental consequence of importing soybean methyl ester (SME) from Brazil and palm methyl ester (PME) from Malaysia to Switzerland and replacing the current diesel consumption of Switzerland by 1% (Reinhard and Zah 2009). The study showed very interesting results of increased environmental burdens from using the imported biofuels rather than local production (Reinhard and Zah 2009). This is mainly due to considering the impacts associated with the coproducts, marginal (affected) vegetable oil in the world market, and land use changes within the system boundary. It was observed that, the marginal oil in the 1world market and the marginal land area impacted created more environmental burdens which offset the emissions reductions due to imports, resulting in higher GHG emissions compared to local production (Reinhard and Zah 2009). Thus, it can be said that C-LCA gives more accurate and realistic predictability in the long term compared to regular A-LCA studies, since it considers the marginal technologies and processes affected by the demand change in the system.

Different application areas assessed by C-LCA include production and processing technologies (Prapaspongsa and Gheewala 2016, Styles et al. 2016), policy (Dandres et al. 2011, Vázquez-Rowe et al. 2014), feedstock comparison (Reinhard and Zah 2009), market development (Van Stappen et al. 2016), recycling (Moore et al. 2017) and so on. However, majority of the biofuel C-LCA studies have assessed the environmental consequences in the context of different policy applications (Dandres et al. 2012, Hamelin et al. 2014, Van Stappen et al. 2016). The number of empirical studies on C-LCA of biofuels is still limited due to data unavailability, uncertain assumptions, lack of reliable method, and so on (McManus and Taylor 2015, Roos and Ahlgren

2018). Several biofuel C-LCA studies based on the CGE approach using the GTAP model, are mostly focused on the determining the ILUC impacts from the agricultural sector due to the increased biofuel production (Vázquez-Rowe et al. 2014, Garraín et al. 2016, Prapaspongsa and Gheewala 2016, Tonini et al. 2016). The consequence of biofuel demand increase across other economic sectors such as transportation, industrial emissions etc. have not been discussed in the literature, although biofuels are mostly consumed in these sectors. Additionally, biofuels are produced from variety of feedstocks other than the agricultural feedstocks, such as forest-based feedstock, biogas, or municipal solid waste. Thus, more examples of application oriented C-LCA studies combined with either economic equilibrium models, market analysis, energy assessment tool, or any other computational models, in the field of biofuel energy are required to demonstrate practical and robust application of C-LCA in the field of bioenergy systems.

5.3.5 Uncertainties and Challenges in the Application of C-LCA on Biofuels

C-LCA methodology and its applications to practical situations have been criticized of being highly uncertain attributed to complex interaction between economy and technological improvement (Hamelin et al. 2017, Plevin 2017). Researchers mentioned that, the A-LCA results are precise having less uncertainty, since A-LCA considers normative allocations attributed to a specific amount of product ignoring the indirect impacts (Reinhard and Zah 2009, Brandão et al. 2014, Hamelin et al. 2017). Though A-LCA is more accurate with less uncertainty, C-LCA has been highly recommended by researchers in the context of decision making as C-LCA considers an economy wide coverage by expanding the system boundary to include all the marginal technologies affected/ substituted due to the additional production (Hamelin et al. 2017). However, the C-LCA method has been criticized by researchers as an immature method with no consistent framework and very limited empirical studies that can be referred as a guidance (Hamelin et al. 2017). On the contrary, Weidema et al. claimed that the C-LCA models are less uncertain and more stable than the A-LCA model for comparative life cycle studies, as C-LCA uses marginal data whereas A-LCA uses average data (Weidema et al. 1999). While using average data to determine the marginal technology, wide range of data from different sources are collected which adds the uncertainty (Weidema et al. 1999). Furthermore, the average data lacks in considering time variation making the result less accurate. Since the marginal technologies are only affected by the boundary constraints, long-term production cost, and development in the marginal technology, using marginal data in C-LCA gives more stable results than using average data in A-LCA since it keeps changing even with a smaller capacity adjustments (Weidema et al. 1999). However, it is not possible to find marginal technology for all type of the products/ technologies so easily, which makes the LCA modelers rely on the average data to identify the marginal technology, hence adding uncertainty. Biofuel C-LCA studies face challenges with data inventory. Due to lack of detail information on different biomass feedstock or crop systems or newly developing biomass processing technologies, aggregated data are used to develop the C-LCA model which induces higher level of uncertainty (Marvuglia et al. 2013).

Adding further uncertainty, C-LCA predicts the long term consequence of any decision or change in the future assuming the future technologies are well defined by the current technologies assessed in the system (Ekvall and Weidema 2004). Most of the biofuel C-LCA studies do not account for the technological improvement in future since the assumption is highly uncertain and volatile which can impede the results (Mathiesen et al. 2009). Dynamic modelling and extrapolation techniques are recommended to develop scenarios considering future technological improvement as well as the system boundary delimitation should happen up to that point beyond which the higher uncertainties and lower consequences are reasonable enough to ignore the rest of the processes (Ekvall and Weidema 2004). Development of multiple C-LCA scenarios by varying the model assumptions such as considering different marginal technology alternatives affected coupled with uncertainty and sensitivity analysis have been recommended to minimize the level of uncertainty in C-LCA studies (Brandão et al. 2014, Hamelin et al. 2017).

5.4 Key Observations

The systematic literature review was conducted on the current state of C-LCA of biofuel energy systems with the aim of identifying research gaps and areas for future research. The issues crucial to the application of C-LCA to biofuel and bioenergy systems, in particular the methods and empirical aspects, were discussed. The major findings are listed below:

- One of the fundamental elements in a C-LCA approach is how to identify and select the marginal product or technology. Several studies mention the five steps approach proposed by Weidema et al. for identifying the marginal technology (Weidema et al. 1999). However, application-oriented demonstration of this method is limited. Different studies have made different conclusions, for instance, that constrained technologies are not suitable as marginal technology both in the short- and long-term. Some studies found that the technologies that can change or use the capacity partially in response to changes in demand are chosen as the marginal technologies. The diverse findings regarding marginal technologies make C-LCA modelling results ambiguous. The assumptions or rationales used to identify and select the marginal technologies need detailed discussion, since the marginal technology chosen significantly affects the reliability of the results.

- Marginal technologies (business as usual, planned/proposed, and actual) have been discussed in studies related to policy, government energy plans, and other long-term energy considerations such as capacity installment or demand change (including both conventional and bioenergy/biofuel systems). However, the studies are mostly economic feasibility studies that use projections and historical statistics on regional energy systems to identify the marginal technologies. Hence, our review concluded that there is no proper demonstration or guidance of C-LCA application on energy systems in general.
- Application-oriented studies can explain practical allocation facts, system delimitation, how to include/exclude processes related to coproduct use, marginal technology, competing products, and associated uncertainties better than theoretical recommendations do.
- C-LCA can model different types of scenarios such as normative, predictive, and explorative with multidirectional objectives assessing the consequences of a wide range of scenarios triggered by demand change. For example, normative scenarios determine the environmental consequences of a biofuel technology with a specific objective; a predictive scenario forecasts the consequence of the large-scale application of a new technology in the market; and an explorative scenario assesses the environmental impacts of the large-scale application of a particular biofuel technology in relation to other perspectives such as industrialist or policy makers' viewpoints, locally or globally.
- Three different models are commonly mentioned in the literature to model C-LCA: the simplified model, partial equilibrium model, and computable general equilibrium model.

The selection of a C-LCA modelling approach depends on a number of criteria, for instance, market delimitation, production scale, time scope, constraints related to market, multifunctionality, policy, technology, and direct/indirect land use changes. It also includes assessing demand change or any other decision change. However, most biofuel C-LCA studies fail to properly explain the criteria used to select a particular modelling approach to solve the problem.

- In terms of mathematical computation, C-LCA modelling approaches are classified as linear (process-based LCA, input-output LCA or partial equilibrium models) and nonlinear (computable general equilibrium models). The scope of the model varies based on assumptions such as market supply, input-output coefficient, and economies of scale. However, both models assume high levels of uncertainty and limitations in practical situations. Hence, to enhance the credibility of the decision making, it is important to evaluate the results from different models rather than rely on results from a particular C-LCA model.
- The application areas assessed by C-LCA include production and processing technologies, policy, feedstock comparison, market development, and recycling. However, most biofuel C-LCA studies assess the environmental consequences in the context of policy applications.
- Biofuel C-LCA studies face challenges with data inventory. Given the lack of detailed information on different biomass feedstocks, crop systems, and newly developing biomass processing technologies, aggregated data are used to develop most of the C-LCA models, which increases uncertainty.

- Uncertainty in C-LCA studies could be minimized by developing multiple C-LCA scenarios by varying model assumptions such as considering different marginal technologies or products affected, along with uncertainty and sensitivity analyses.

In summary, inconsistent methods, ambiguous modelling assumptions, lack of standard guidelines, empirical results, and application oriented C-LCA studies in the biofuel field were found as a result of the literature review. More application oriented C-LCA studies related to the biofuel production and use can be useful for the modelers to get a better idea regarding the methodological framework, model assumptions, system boundary, and data inventory crucial to a successful application of a C-LCA model. Moreover, a few comparative analyses related to biofuels production or uses have been conducted using the C-LCA approach. Comparative analysis adds comprehensiveness to the decision context which helps better the policy makers, strategy analysts or researchers for robust decision making. Hence, further research is recommended to explore comparative aspects of different biofuel production and uses.

Table 5. 1: Findings from the reviewed C-LCA papers on biofuel energy systems

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
(Reinhard and Zah 2009)	Case study	Environmental consequence of two imported biofuels, soybean methyl ester (SME) and palm methyl ester (PME), replacing the current fuel mix (fossil diesel) in Switzerland by one percent	System expansion approach to model the multifunctional system.	Replacing fossil fuels with biofuels.	GHG emissions, land use, acidification, eutrophication, ozone depletion, ecotoxicity, human toxicity, and abiotic depletion.
(Silalertruksa et al.	Case	Comparative analysis of six ways of	C-LCA modelling	Environmental trade-offs from	GHG emissions and direct land

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
2009)	study	producing cassava to meet the Thai bio-ethanol policy target; used to determine the most environmentally sustainable pathway	approach including both the region of increased demand and other regions impacted due to increased biofuel demand.	different alternatives to obtain the additional biofuel feedstock and subsequent changes on the agricultural system.	use change.
(Dandres et al. 2011)	Case study	The environmental consequences in the global economy due to the large-scale application of two European energy policies for electricity production.	Global economic general equilibrium model and LCA modelling	A large-scale biofuel policy application considering all the impacted economic sectors globally and including the trading partner countries compared to conventional fossil fuels used for electricity production.	GHG emissions, human health damage, natural resource depletion, ecosystems damage.
(Dandres et al. 2012)	Method + Case study	Development of a new LCA approach, macro life cycle assessment (M-LCA), to assess the environmental and economic consequences of large-scale biofuel policy application on a medium- or long-term basis. The M-LCA considers technological improvement and economic evolution.	Global economic general equilibrium model, LCA model, and SimaPro.	More promising and more helpful with decision making than conventional C-LCA as it can model the future economic and technological evolutions known as prospective LCA (P-LCA).	GHG emissions, human health damage, natural resource depletion, ecosystems damage

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
(Garraín et al. 2016)	Case study	Assessment of the indirect environmental impacts to meet the additional biofuel demand in Spain.	Linear equations for cropland area calculation and ILUC factor (“carbon foregone”) were developed to calculate carbon loss from land use change.	Factors for successful C-LCA modelling caused by additional biofuel demand involving different types of energy crops as feedstock, coproducts, and land use changes.	GHG emissions from indirect land use change and global agricultural land area affected
(Kallio et al. 2013)	Case study	Environmental consequences of achieving the RFS (Renewable fuel standard) policy target by using trees to produce heat, electricity, and biodiesel to replace fossil fuels in Finland by 2035. The forest carbon sink is found to decrease with the excessive use of the trees and above ground biomass.	A partial equilibrium model for Finland’s forest sector (SF-GTM), MELA 2009 (large-scale Finnish forestry model)	Impacts of large-scale biofuel policy targets on GHG emissions reduction and climate change mitigation.	Change in GHG emissions in the atmosphere
(Karlsson et al. 2015)	Case study	Environmental consequences from different alternative uses of faba bean (i.e., as animal feed, to produce ethanol, or as roughage feed in Sweden).	Cradle-to-grave C-LCA modelling through the system expansion approach.	Sustainable practices related to bio refineries that promote alternate uses of crop feedstock as well as crop fractionation to produce protein concentrate, liquid, and solid biofuels.	GHG emissions, arable land use change, and fossil energy use

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
(Khoshnevisan et al. 2016)	Case study	Comparison of the environmental consequences of bioethanol and biogas production from pinewood using two different pretreatment technologies, N-methylmorpholine-N-oxide (NMMO) and steam explosion.	SimaPro 8.1	Environmental and economic viability of bioethanol and biogas production technologies. Comparative analysis based on economic measures such as profitability index along with environmental indicators.	Human health damage, ecosystem quality, resources depletion and climate change
(Lim and Lee 2011)	Case study	Energy and emission impacts of coproducing biodiesel and bioethanol to maximize the outputs from a palm oil processing plant.	Seed-to-wheel consequential modelling through the system expansion approach.	Sustainable norms and practices related to palm oil processing in bio refineries continuing more than a century considering the palm oil tree growth cycle.	GHG emissions and energy efficiency measures such as net energy ratio, net carbon emission ratio, carbon emission savings, and carbon payback period.
(Melamu and Von Blottnitz 2011)	Case study	Environmental consequences of using sugar mill bagasse to produce 2nd generation ethanol instead of producing process heat.	SimaPro 7.1.2	Alternative use of sugar mill bagasse, sustainable biofuel research, and biofuel policy making.	GHG emissions, fossil energy use, acidification, and eutrophication.
(Menten et al. 2015)	Method + Case study	Development of a bottom-up long term optimization model (MIRET) to assess the environmental impacts of different energy pathways. The	Energy optimization model (MIRET) developed using the TIMES	Useful information generated by the integrated energy model help strategic environmental	Global warming potential.

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
		model is demonstrated through a case study of producing synthetic biodiesel in France up to 2030.	economic model.	planning, industry practices, and biofuel policy making	
(Renouf et al. 2013)	Case study	Environmental impacts from producing diversified bio-based products such as ethanol, electricity, and polylactide (PLA) plastics from Australian sugarcane along with the main product (sugar) and identification of the most environmentally sustainable pathway.	SimaPro (V7.1)	Eco-efficient and sustainable sugar cane growing practices in Australia and environmental trade-offs of producing diversified biofuels and bio-based products in the sugarcane system.	GHG emissions, acidification, water use, land use, non-renewable energy use, eutrophication, respiratory inorganics (RI) and respiratory organics (RO).
(Parajuli et al. 2017)	Case study	The environmental consequences of integrating two bio refineries to produce bio-based ethanol and lactic acid and comparing the results from the A-LCA and C-LCA approaches.	SimaPro 8.0.4	System efficiency improvement and environmental benefits from the integration of bio refineries, use of recirculated material for bioethanol production, and plant energy recovery.	Global warming potential, eutrophication, non-renewable energy use and agricultural land occupation (ALO).
(Prapaspongsa and Gheewala 2016)	Case study	Environmental risks involved in applying a large-scale bioethanol policy by 2021 in Thailand from cassava and molasses compared to	A global systematic indirect land use change (ILUC) model coupled with C-LCA in	Benefits from proper coproduct use and relevant suppliers considered in the system	GHG emissions and indirect land use change (ILUC).

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
		conventional gasoline.	SimaPro.	boundary to reduce biofuel life cycle GHG emissions.	
(Prapaspongsa et al. 2017)	Case study	Modelling the environmental impacts of biodiesel production from palm oil using both A-LCA and C-LCA approaches and comparing the results with conventional diesel.	ReCipe 2008 method and SimaPro version 8.0.3	Support and inform renewable energy and sustainable strategy/policy makers on the pros and cons of different modeling choices, proper coproduct use, direct-indirect land use change impacts, and technological evolutions in biodiesel production processes from palm oil.	GHG emissions, human toxicity, acidification, photochemical oxidation, and eutrophication.
(Rege et al. 2015)	Methodology + Case study	Two types of C-LCA partial equilibrium models (PEM) to capture realistic environmental impacts such as indirect land use changes along with the direct consequences of a demand change for maize used in biogas production in Luxembourg.	Mathematical formulations of non-linear programming-(NLP) and positive mathematical programming (PMP) approaches.	Comparison of two alternative PEM approaches, non-linear programming-(NLP) and positive mathematical programming (PMP) in C-LCA in terms of capturing market demand elasticity, and technical, social, and economic	N/A

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
				constraints.	
(Reinhard and Zah 2011)	Case study	Environmental tradeoffs from producing additional rapemethylester (RME) biofuel that displaces local food production (i.e., barley) or edible oil production. In either case, the displaced product is imported from other countries with subsequent environmental consequences.	CML method and Swiss method of ecological scarcity.	Environmental tradeoffs associated with an additional biofuel demand on the local and global supply chain.	CML indicators such as GHG emissions, human toxicity, acidification, eutrophication, land-use changes, and UBP points (by the Swiss method of ecological scarcity).
(Moore et al. 2017)	Case study	Environmental consequences of replacing chemical fertilizer used in sugarcane production in Brazil with environmentally friendly nutrients such as vinasse and filter cake produced as residues from bioethanol production.	Cumulative energy demand (CED) version 1.08 and ReCipe midpoint (H) version 1.11	Inform and motivate the relevant stakeholders on sustainable sugarcane production using bioethanol processing residues such as vinasse and filter cake instead of chemical fertilizer.	GHG emissions, depletion of fossil fuels, terrestrial acidification, human toxicity, eutrophication and terrestrial ecotoxicity
(Tonini et al. 2016)	Case study	Development of 554 GHG emission factors from 24 types of biomass ranging from crops to residues for three types of bioenergy: bioethanol, bioelectricity, and biomethane.	IPCC 2007 method using Ecoinvent v3.1 consequential database.	Identify sustainable bioenergy production practice from different biomasses using different thermal and biological production	GHG emissions per functional unit (kWh or MJ)

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
				pathways including land use change impacts in the life cycle.	
(Tonini et al. 2016)	Method + Case study	Environmental consequences of producing bioethanol and biogas from eight types of agro-industrial residues through four conversion pathways. 32 scenarios were developed.	A deterministic ILUC model and a biochemical energy conversion model developed using Matlab (©, The MathWorks Inc., version R2012) with the C-LCA model.	The developed models/framework can be used by future consequential bioenergy research assessing alternate uses of residues (animal feed/food) and additional demand for arable land.	Global warming, aquatic eutrophication-nitrogen, acidification, and phosphorous resource-saving from indirect land use changes.
(Tonini et al. 2017)	Case study	Identification of environmentally sustainable energy conversion pathways of domestic biomass (agricultural and forest residues, municipal waste, and food residues) to meet Denmark's annual energy demand by 2030 and national energy policy targets.	Matrix-based LCA model coupled with a linear programming based optimization model.	Inform/guide policy/decision makers, technology developers, industries, market analysts, and energy system analysts regarding the optimum use of different domestic biomass resources for energy production and thereby meeting different national environment policy target.	GHG emissions and cumulative fossil energy demand

Reference	**Case study/ Method**	**Study objective/focus**	**Modelling approach/tool**	**Application of study findings**	**Environmental consequence measures**
(Styles et al. 2015)	Case study	Comparative analysis of environmental consequences of different bioenergy options – biofuel, biogas, and biomass – used on a large arable farm,	LCAD tool coupled with a ILUC (indirect land use change) module.	Sustainable practices in anaerobic digestion farming, and biofuel policy decision related to bioenergy subsidies, resource utilization, ecosystem protection, and climate change mitigation	GHG emissions. eutrophication, acidification, and resource depletion
(Styles et al. 2016)	Case study	Environmental consequences of producing biogas through the anaerobic digestion of five waste feedstocks in the UK between 2014 and 2017.	LCAD EcoScreen tool	Educate the policy makers regarding different anaerobic digestion processes, its efficiency, waste feedstock requirement, plant scale, plant capacity, and impacts on GHG abatement and land use.	Global warming potential, eutrophication, acidification, and fossil resource depletion
(Mathiesen et al. 2009)	Method + Case study	Discussion of the procedure used to identify marginal/affected technologies in a C-LCA study and associated uncertainties in the bioenergy field.	EnergyPLAN model.	Guidance on analyzing and selecting different potential marginal technologies for effective C-LCA results. Energy system analysis consisting different types	N/A

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
				of renewable energy, varied proportion of energy mix, energy savings, and costings is recommended to analyze different future energy scenarios.	
(Escobar et al. 2014)	Case study	Environmental consequences of meeting the increased biodiesel demand in Spain in response to the European Directive 2009/28/CE through two alternative pathways, producing biodiesel from cooking oil and importing it from Argentina.	GaBi 6 Software	Biofuel policy making, decision based on tradeoffs between in-house biofuel production and imports	Global warming potential, abiotic depletion, acidification, eutrophication, freshwater aquatic ecotoxicity, human toxicity, marine aquatic ecotoxicity, ozone layer depletion, photochemical ozone creation, and terrestrial ecotoxicity potential.
(Marvuglia et al. 2013)	Method +Case study	A brief guideline on the practical implementations of different LCA modelling approaches followed by a case study of the C-LCA of bioenergy production.	PEM model combined with C-LCA model developed in SimaPro	Educate/inform C-LCA modelers or researchers on the most commonly used C-LCA modelling approaches (simplified, PEM and CGM), their assumptions, scopes, data inventories,	GHG emissions

Reference	Case study/ Method	Study objective/focus	Modelling approach/tool	Application of study findings	Environmental consequence measures
				method, and applications with a focus on the most critical aspects of C-LCA application (delimitation of relevant market boundaries, economies of scale, multifunctional processes, direct and indirect land use changes, etc.).	

Reference

Adams, D., R. Alig, B. A. McCarl, B. C. Murray, L. Bair, B. Depro, G. Latta, H.-C. Lee, U. A. Schneider and J. Callaway (2005). FASOMGHG conceptual structure, and specification: documentation, College Station, TX: Texas A&M University http://agecon2.tamu.edu/people/faculty/mccarl-bruce/papers/1212FASOMGHG_doc.pdf Accessed 02/01/2019.

Adler, P. R., S. J. Del Grosso and W. J. Parton (2007). "Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems." Ecological Applications **17**(3): 675-691.

Andrew, R. and V. Forgie (2008). "Life Cycle Assessment of Canola Biodiesel in New Zealand." Landcare Research and Costing of Canola Biodiesel in New Zealand. Centre for Ecological Economics.

Baumann, H. (1998). Life cycle assessment and decision making: theories and practices, Chalmers University of Technologyhttps://is4ie.org/resources/dissertations/112 Accessed 03/02/2019.

Börjeson, L., M. Höjer, K.-H. Dreborg, T. Ekvall and G. Finnveden (2006). "Scenario types and techniques: towards a user's guide." Futures **38**(7): 723-739.

Bouman, M., R. Heijungs, E. Van der Voet, J. C. van den Bergh and G. Huppes (2000). "Material flows and economic models: an analytical comparison of SFA, LCA and partial equilibrium models." Ecological economics **32**(2): 195-216.

Brandão, M., R. Clift, A. Cowie and S. Greenhalgh (2014). "The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on “Using Attributional Life Cycle

Assessment to Estimate Climate-Change Mitigation…”." Journal of Industrial Ecology **18**(3): 461-463.

Cherubini, F., N. D. Bird, A. Cowie, G. Jungmeier, B. Schlamadinger and S. Woess-Gallasch (2009). "Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations." Resources, Conservation and recycling **53**(8): 434-447.

Cherubini, F. and A. H. Strømman (2011). "Life cycle assessment of bioenergy systems: State of the art and future challenges." Bioresource Technology **102**(2): 437-451.

Collet, P., A. Hélias Arnaud, L. Lardon, M. Ras, R. A. Goy and J. P. Steyer (2011). "Life-cycle assessment of microalgae culture coupled to biogas production." Bioresource Technology **102**(1): 207-214.

Dalgaard, R. and I. Muñoz (2014). Consequential and attributional modeling in life cycle assessment of food production systems. Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2014), San Francisco, California, USA, 8-10 October, 2014, American Center for Life Cycle Assessment.

Dandres, T., C. Gaudreault, P. Tirado-Seco and R. Samson (2011). "Assessing non-marginal variations with consequential LCA: application to European energy sector." Renewable and Sustainable Energy Reviews **15**(6): 3121-3132.

Dandres, T., C. Gaudreault, P. Tirado-Seco and R. Samson (2012). "Macroanalysis of the economic and environmental impacts of a 2005–2025 European Union bioenergy policy using the GTAP model and life cycle assessment." Renewable and Sustainable Energy Reviews **16**(2): 1180-1192.

Dowlatabadi, H. (1995). "Integrated assessment models of climate change: An incomplete overview." Energy Policy **23**(4-5): 289-296.

Duchin, F. and S. H. Levine (2011). "Sectors may use multiple technologies simultaneously: The rectangular choice-of-technology model with binding factor constraints." Economic Systems Research **23**(3): 281-302.

Earles, J. M. and A. Halog (2011). "Consequential life cycle assessment: a review." The International Journal of Life Cycle Assessment **16**(5): 445-453.

Earles, J. M., A. Halog, P. Ince and K. Skog (2013). "Integrated economic equilibrium and life cycle assessment modeling for policy-based consequential LCA." Journal of Industrial Ecology **17**(3): 375-384.

Ekvall, T. (1999). "System expansion and allocation in life cycle assessment with implications for wastepaper management." Doktorsavhandlingar vid Chalmers Tekniska Hogskola(1509): 1-54.

Ekvall, T. (2002). "Cleaner production tools: LCA and beyond." Journal of Cleaner Production **10**(5): 403-406.

Ekvall, T. and A. Andrae (2006). "Attributional and consequential environmental assessment of the shift to lead-free solders (10 pp)." The International Journal of Life Cycle Assessment **11**(5): 344-353.

Ekvall, T. and G. Finnveden (2001). "Allocation in ISO 14041—a critical review." Journal of Cleaner Production **9**(3): 197-208.

Ekvall, T. and A.-M. Tillman (1997). "Open-loop recycling: criteria for allocation procedures." The International Journal of Life Cycle Assessment **2**(3): 155.

Ekvall, T. and B. P. Weidema (2004). "System boundaries and input data in consequential life cycle inventory analysis." The International Journal of Life Cycle Assessment **9**(3): 161-171.

Environment and Climate Change Canada (2018). Clean Fuel Standard, Regulatory Design Paper, Environment and Climate Change Canada,https://www.canada.ca/content/dam/eccc/documents/pdf/climate-change/clean-fuel-standard-regulatory-design-paper-2018-en-1.pdf Accessed 10/01/2019.

Environment and Climate Change Canada (2019a). National Inventory Report 1990-2017: Greenhouse Gas Sources and Sinks in Canada Environment and Climate Change Canada (ECCC),https://www.canada.ca/en/environment-climatechange/services/climate-change/greenhouse-gas-emissions.html Accessed 05/05/2019.

Environment and Climate Change Canada (2019b). Canadian Environmental Sustainability Indicators : Greenhouse gas emissions, Environment and Climate Change Canada (ECCC),https://www.canada.ca/en/environment-climate-change/services/environmental-indicators/greenhouse-gas-emissions.html Accessed 01/05/2019.

Escobar, N., J. Ribal, G. Clemente and N. Sanjuán (2014). "Consequential LCA of two alternative systems for biodiesel consumption in Spain, considering uncertainty." Journal of Cleaner Production **79**: 61-73.

Fabiosa, J. F., J. C. Beghin, F. Dong, A. Elobeid, S. Tokgoz and T.-H. Yu (2010). "Land allocation effects of the global ethanol surge: Predictions from the international FAPRI model." Land Economics **86**(4): 687-706.

Finnveden, G., M. Z. Hauschild, T. Ekvall, J. Guinée, R. Heijungs, S. Hellweg, A. Koehler, D. Pennington and S. Suh (2009). "Recent developments in life cycle assessment." Journal of environmental management **91**(1): 1-21.

Fortuna, M. J. A. and S. R. Rege (2010). Computable General Equilibrium Models: Theory and Applications, Universidade dos Açores, Centro de Estudos de Economía Aplicad do Atlântico ….

Francois, J. F. and K. A. Reinert (1997). Applied methods for trade policy analysis: a handbook, Cambridge University Press.

Frischknecht, R. (1998). "Life cycle inventory analysis for decision-making." The International Journal of Life Cycle Assessment **3**(2): 67-67.

Fuss, S., J. G. Canadell, G. P. Peters, M. Tavoni, R. M. Andrew, P. Ciais, R. B. Jackson, C. D. Jones, F. Kraxner and N. Nakicenovic (2014). "Betting on negative emissions." Nature Climate Change **4**(10): 850.

Garraín, D., C. de la Rúa and Y. Lechón (2016). "Consequential effects of increased biofuel demand in Spain: Global crop area and CO2 emissions from indirect land use change." Biomass and Bioenergy **85**: 187-197.

Government of Alberta. (2019a). "Renewable Fuels Standard." Retrieved 01/05/2019, from https://www.alberta.ca/renewable-fuels-standard.aspx?utm_source=redirector

Government of Alberta (2019b). Carbon Competitiveness Incentive Regulation. Alberta, Canadahttps://www.alberta.ca/carbon-competitiveness-incentive-regulation.aspx Accessed 20/06/2019.

Guinée, J., M. Gorrée, R. Heijungs, G. Huppes, R. Kleijn, A. De Koning, L. Van Oers, A. Wegener Sleewijk, S. Suh and U. de Haes HA (2004). "Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards (Eco-Efficiency in Industry and Science)."

Hamelin, L., I. Naroznova and H. Wenzel (2014). "Environmental consequences of different carbon alternatives for increased manure-based biogas." Applied Energy **114**: 774-782.

Hamelin, L., P. A. Z. Puławy and E. Ecoinnovazione (2017). "Consequential Life Cycle Assessment: What, How, and Why?" Encyclopedia of Sustainable Technologies: 277.

IEA (2017). CO2 Emissions from Fuel Combustion 2017 https://webstore.iea.org/co2-emissions-from-fuel-combustion Accessed 05/05/2019.

Inc. (S & T)2 Consultants (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions, Alberta Innovates - Energy and Environment Solutions, https://books.google.ca/books/about/Life_Cycle_Assessment_of_Selected_Biofue.html?id=tT1pngEACAAJ&redir_esc=y.

International Organization for Standardization (2006-07). ISO 14040:2006 Environmental Management–Life Cycle Assessment–Principles and Framework International Organization for Standardization (ISO),https://www.iso.org/standard/37456.html Accessed 05/01/2018.

International Organization for Standardization (2006-07). ISO 14044:2006 Environmental management -- Life cycle assessment -- Requirements and guidelines, International Organization for Standardization (ISO),https://www.iso.org/standard/38498.html Accessed 05-09-2018.

Kallio, A., O. Salminen and R. Sievänen (2013). "Sequester or substitute—consequences of increased production of wood based energy on the carbon balance in Finland." Journal of forest economics **19**(4): 402-415.

Karlsson, H., S. Ahlgren, I. Strid and P.-A. Hansson (2015). "Faba beans for biorefinery feedstock or feed? Greenhouse gas and energy balances of different applications." Agricultural Systems **141**: 138-148.

Khoshnevisan, B., M. Shafiei, M. A. Rajaeifar and M. Tabatabaei (2016). "Biogas and bioethanol production from pinewood pre-treated with steam explosion and N-methylmorpholine-N-oxide (NMMO): A comparative life cycle assessment approach." Energy **114**: 935-950.

Kloverpris, J., H. Wenzel and P. H. Nielsen (2008). "Life cycle inventory modelling of land use induced by crop consumption-Part 1: Conceptual analysis and methodological proposal." International Journal of Life Cycle Assessment **13**(1): 13-21.

Kløverpris, J. H. (2008). "Consequential life cycle inventory modelling of land use induced by crop consumption." Industrial PhD Thesis. April.

Kløverpris, J. H. (2009). "Identification of biomes affected by marginal expansion of agricultural land use induced by increased crop consumption." Journal of Cleaner Production **17**(4): 463-470.

Le Quéré, C., R. M. Andrew, P. Friedlingstein, S. Sitch, J. Pongratz, A. C. Manning, J. I. Korsbakken, G. P. Peters, J. G. Canadell and R. B. Jackson (2018). "Global carbon budget 2017." Earth System Science Data Discussions **10**: 405-448.

Le Quéré, C., J. I. Korsbakken, C. Wilson, J. Tosun, R. Andrew, R. J. Andres, J. G. Canadell, A. Jordan, G. P. Peters and D. P. van Vuuren (2019). "Drivers of declining CO 2 emissions in 18 developed economies." Nature Climate Change **9**(3): 213-217.

Lim, S. and K. T. Lee (2011). "Parallel production of biodiesel and bioethanol in palm-oil-based biorefineries: life cycle assessment on the energy and greenhouse gases emissions." Biofuels, Bioproducts and Biorefining **5**(2): 132-150.

Linn, J. and M. Raeth-Knight. (2016). "A Biodiesel byproduct may substitute for corn" Retrieved 17 November, 2016, from http://www.extension.umn.edu/agriculture/dairy/feed-and-nutrition/a-biodiesel-byproduct-may-substitute-for-corn/.

Lund, H. and B. V. Mathiesen (2009). "Energy system analysis of 100% renewable energy systems—The case of Denmark in years 2030 and 2050." Energy **34**(5): 524-531.

Machi, L. A. and B. T. McEvoy (2016). The literature review: Six steps to success, Corwin Press.

Macintosh, A., H. Keith and D. Lindenmayer (2015). "Rethinking forest carbon assessments to account for policy institutions." Nature Climate Change **5**(10): 946.

Magarey, J. M. (2001). "Elements of a systematic review." International Journal of Nursing Practice **7**(6): 376-382.

Mahbub, N., E. Gemechu, H. Zhang and A. Kumar (2019). "The life cycle greenhouse gas emission benefits from alternative uses of biofuel coproducts." Sustainable Energy Technologies and Assessments **34**: 173-186.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**: 1249-1262.

Marvuglia, A., E. Benetto, S. Rege and C. Jury (2013). "Modelling approaches for consequential life-cycle assessment (C-LCA) of bioenergy: critical review and proposed framework for biogas production." Renewable and Sustainable Energy Reviews **25**: 768-781.

Mathiesen, B. V., M. Münster and T. Fruergaard (2009). "Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments." Journal of Cleaner Production **17**(15): 1331-1338.

McManus, M. C. and C. M. Taylor (2015). "The changing nature of life cycle assessment." Biomass and Bioenergy **82**: 13-26.

Melamu, R. and H. Von Blottnitz (2011). "2nd Generation biofuels a sure bet? A life cycle assessment of how things could go wrong." Journal of Cleaner Production **19**(2-3): 138-144.

Menten, F., S. Tchung-Ming, D. Lorne and F. Bouvart (2015). "Lessons from the use of a long-term energy model for consequential life cycle assessment: the BTL case." Renewable and Sustainable Energy Reviews **43**: 942-960.

Millar, R. J., J. S. Fuglestvedt, P. Friedlingstein, J. Rogelj, M. J. Grubb, H. D. Matthews, R. B. Skeie, P. M. Forster, D. J. Frame and M. R. Allen (2017). "Emission budgets and pathways consistent with limiting warming to 1.5 C." Nature Geoscience **10**(10): 741.

Moore, C. C. S., A. R. Nogueira and L. Kulay (2017). "Environmental and energy assessment of the substitution of chemical fertilizers for industrial wastes of ethanol production in sugarcane cultivation in Brazil." The International Journal of Life Cycle Assessment **22**(4): 628-643.

Msangi, S., M. Ewing and M. Rosegrant (2010). Biofuels and agricultural growth: challenges for developing agricultural economies and opportunities for investment. Handbook of Bioenergy Economics and Policy, Springer**:** 73-90.

Parajuli, R., M. T. Knudsen, M. Birkved, S. N. Djomo, A. Corona and T. Dalgaard (2017). "Environmental impacts of producing bioethanol and biobased lactic acid from standalone and integrated biorefineries using a consequential and an attributional life cycle assessment approach." Science of the Total Environment **598**: 497-512.

Pehnt, M., M. Oeser and D. J. Swider (2008). "Consequential environmental system analysis of expected offshore wind electricity production in Germany." Energy **33**(5): 747-759.

Plevin, R. J. (2017). "Assessing the climate effects of biofuels using integrated assessment models, part I: Methodological considerations." Journal of Industrial Ecology **21**(6): 1478-1487.

Prapaspongsa, T. and S. H. Gheewala (2016). "Risks of indirect land use impacts and greenhouse gas consequences: an assessment of Thailand's bioethanol policy." Journal of Cleaner Production **134**: 563-573.

Prapaspongsa, T., C. Musikavong and S. H. Gheewala (2017). "Life cycle assessment of palm biodiesel production in Thailand: Impacts from modelling choices, co-product utilisation, improvement technologies, and land use change." Journal of Cleaner Production **153**: 435-447.

Ramdhani, A., M. A. Ramdhani and A. S. Amin (2014). "Writing a Literature Review Research Paper: A step-by-step approach." International Journal of Basic and Applied Science **3**(1): 47-56.

Rege, S., M. Arenz, A. Marvuglia, I. Vázquez-Rowe, E. Benetto, E. Igos and D. Koster (2015). "Quantification of Agricultural Land Use Changes in Consequential Life Cycle Assessment

Using Mathematical Programming Models Following a Partial Equilibrium Approach." Journal of Environmental Informatics **26**(2).

Reinhard, J. and R. Zah (2009). "Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment." Journal of Cleaner Production **17**: S46-S56.

Reinhard, J. and R. Zah (2011). "Consequential life cycle assessment of the environmental impacts of an increased rapemethylester (RME) production in Switzerland." Biomass and Bioenergy **35**(6): 2361-2373.

Renouf, M. A., R. J. Pagan and M. K. Wegener (2013). "Bio-production from Australian sugarcane: an environmental investigation of product diversification in an agro-industry." Journal of Cleaner Production **39**: 87-96.

Roningen, V. O. (1997). "Multi-market, multi-region partial equilibrium modeling." Applied methods for trade policy analysis: a handbook: 231-257.

Roos, A. and S. Ahlgren (2018). "Consequential life cycle assessment of bioenergy systems–A literature review." Journal of Cleaner Production **189**: 358-373.

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan

Schmidt, J. H. (2008). "System delimitation in agricultural consequential LCA." The International Journal of Life Cycle Assessment **13**(4): 350.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes and T.-H. Yu (2008). "Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change." Science **319**(5867): 1238-1240.

Silalertruksa, T., S. H. Gheewala and M. Sagisaka (2009). "Impacts of Thai bio-ethanol policy target on land use and greenhouse gas emissions." Applied Energy **86**: S170-S177.

Sissine, F. (2010). Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis, Technical Report EPA-420-R-10-006. Assessment and Standards Division, Office ….

Stasinopoulos, P., P. Compston, B. Newell and H. M. Jones (2012). "A system dynamics approach in LCA to account for temporal effects—a consequential energy LCI of car body-in-whites." The International Journal of Life Cycle Assessment **17**(2): 199-207.

Styles, D., E. M. Dominguez and D. Chadwick (2016). "Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment." Science of the Total Environment **560**: 241-253.

Styles, D., J. Gibbons, A. P. Williams, J. Dauber, H. Stichnothe, B. Urban, D. R. Chadwick and D. L. Jones (2015). "Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation." Gcb Bioenergy **7**(6): 1305-1320.

Thomassen, M. A., R. Dalgaard, R. Heijungs and I. de Boer (2008). "Attributional and consequential LCA of milk production." The International Journal of Life Cycle Assessment **13**(4): 339-349.

Tonini, D., L. Hamelin, M. Alvarado-Morales and T. F. Astrup (2016). "GHG emission factors for bioelectricity, biomethane, and bioethanol quantified for 24 biomass substrates with consequential life-cycle assessment." Bioresource Technology **208**: 123-133.

Tonini, D., L. Hamelin and T. F. Astrup (2016). "Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes." Gcb Bioenergy **8**(4): 690-706.

Tonini, D., C. Vadenbo and T. F. Astrup (2017). "Priority of domestic biomass resources for energy: Importance of national environmental targets in a climate perspective." Energy **124**: 295-309.

United Nations Climate Change. (2018). "The Paris Agreement." Retrieved 05/05/2018, from https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement.

Van Stappen, F., M. Mathot, V. Decruyenaere, A. Loriers, A. Delcour, V. Planchon, J.-P. Goffart and D. Stilmant (2016). "Consequential environmental life cycle assessment of a farm-scale biogas plant." Journal of environmental management **175**: 20-32.

Van Vuuren, D. P., H. van Soest, K. Riahi, L. Clarke, V. Krey, E. Kriegler, J. Rogelj, M. Schaeffer and M. Tavoni (2016). "Carbon budgets and energy transition pathways." Environmental Research Letters **11**(7): 075002.

Vázquez-Rowe, I., A. Marvuglia, S. Rege and E. Benetto (2014). "Applying consequential LCA to support energy policy: land use change effects of bioenergy production." Science of the Total Environment **472**: 78-89.

Vázquez-Rowe, I., S. Rege, A. Marvuglia, J. Thénie, A. Haurie and E. Benetto (2013). "Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg." The International Journal of Life Cycle Assessment **18**(8): 1593-1604.

Weidema, B. P. (1993). "Market aspects in product life cycle inventory methodology." Journal of Cleaner Production **1**(3-4): 161-166.

Weidema, B. P. (2003). Market information in life cycle assessment, Miljøstyrelsen.

Weidema, B. P., T. Ekvall and R. Heijungs (2009). "Guidelines for application of deepened and broadened LCA." Deliverable D18 of work package **5**: 17.

Weidema, B. P., N. Frees and A.-M. Nielsen (1999). "Marginal production technologies for life cycle inventories." The International Journal of Life Cycle Assessment **4**(1): 48-56.

West, G. R. (1995). "Comparison of input–output, input–output+ econometric and computable general equilibrium impact models at the regional level." Economic Systems Research **7**(2): 209-227.

Witzke, P., M. Banse, H. Gömann, T. Heckelei, T. Breuer, S. Mann, M. Kempen, M. Adenäuer and A. Zintl (2008). "Modeling of Energy-Crops in Agricultural Sector Models–A Review of Existing Methodologies." JRC Scientific and Technical Reports.

Yang, Y. (2016). "Two sides of the same coin: consequential life cycle assessment based on the attributional framework." Journal of Cleaner Production **127**: 274-281.

Yang, Y. and R. Heijungs (2018). "On the use of different models for consequential life cycle assessment." The International Journal of Life Cycle Assessment **23**(4): 751-758.

Chapter 6: Conclusions and recommendations

6.1 Conclusions

This chapter summarizes the key outcomes of the thesis.

An LCA model was developed to determine the environmental performance of oxymethylene ether (OME) production, a newly emerging diesel additive biofuel. A comparative LCA was performed to determine the environmental performance of OME from whole tree and forest residue biomass in Western Canada. The scope of the LCA model was broadened to include economic and social impacts by developing an LCSA framework. The framework was used to evaluate the sustainability performances of OME from two pathways and to identify the most sustainable pathway. Biofuel systems are multifunctional; that is, they produce coproducts along with the biofuels. This was followed by assessment of different alternative applications of coproducts from two biofuel pathways, DDGs from wheat to ethanol and canola meal and glycerine from canola to biodiesel. A framework was developed to evaluate the environmental benefits of different potential applications of coproducts on the overall biofuel life cycle. LCA studies commonly do not consider proper displacement values to calculate the coproduct credits, which leads to the overestimation of biofuel life cycle GHG emissions. LCA can determine the environmental or sustainability impacts from different technologies, but long-term consequences of any large-scale technology or policy implementation are determined through a consequential LCA (C-LCA). Finally, an extensive systematic literature review on the C-LCA of biofuels was conducted to determine the current state of C-LCA of biofuels along with the research gaps, scope, challenges, limitations, and areas for further investigation. The main findings from each chapter are discussed briefly in the following sections.

6.1.1 Life Cycle Energy and Emission Impacts of OME Production

This study developed a cradle-to-grave LCA model to estimate the environmental performances of OME production from two different types of forest biomass. The forest residue pathway was found to produce a lower environmental burden than the whole tree. The upstream GHG emissions from the forest residue pathway (18g CO_2eq/MJ) were significantly lower than those of the whole tree pathway (27 g CO_2eq/MJ) because the forest residue pathway does not have emission-intensive road construction operation. Although the transportation distance for biomass collection in the forest residue pathway (21.75 km) was almost 5 times higher than in the whole tree pathway (4.56 km), biomass transportation energy consumption in the whole tree pathway was twice as high as in the forest residue pathway, again due to the emission-intensive road construction operation. Biomass production energy consumption was also higher in the whole tree pathway because harvesting operations like skidding, felling, and chipping, and skidding are more energy-intensive than forest residue harvesting operations such as forwarding and chipping. For both pathways, more than 75% of the life cycle GHG emissions were from fuel combustion. However, since OME was produced from biomass, combustion emissions were considered to be carbon neutral as it is assumed that the same amount of $CO_2$is absorbed by the trees during its growth. Hence, the upstream emissions formed the life cycle emissions. The life cycle GHG emissions were 86% lower from forest residue feedstock and 79% lower from whole tree feedstock than from diesel. When OME is used as a diesel additive (10% OME blended with 90% diesel), life cycle GHG emissions were 5% and 5.35% lower than diesel's for the whole tree and forest residue pathways, respectively.

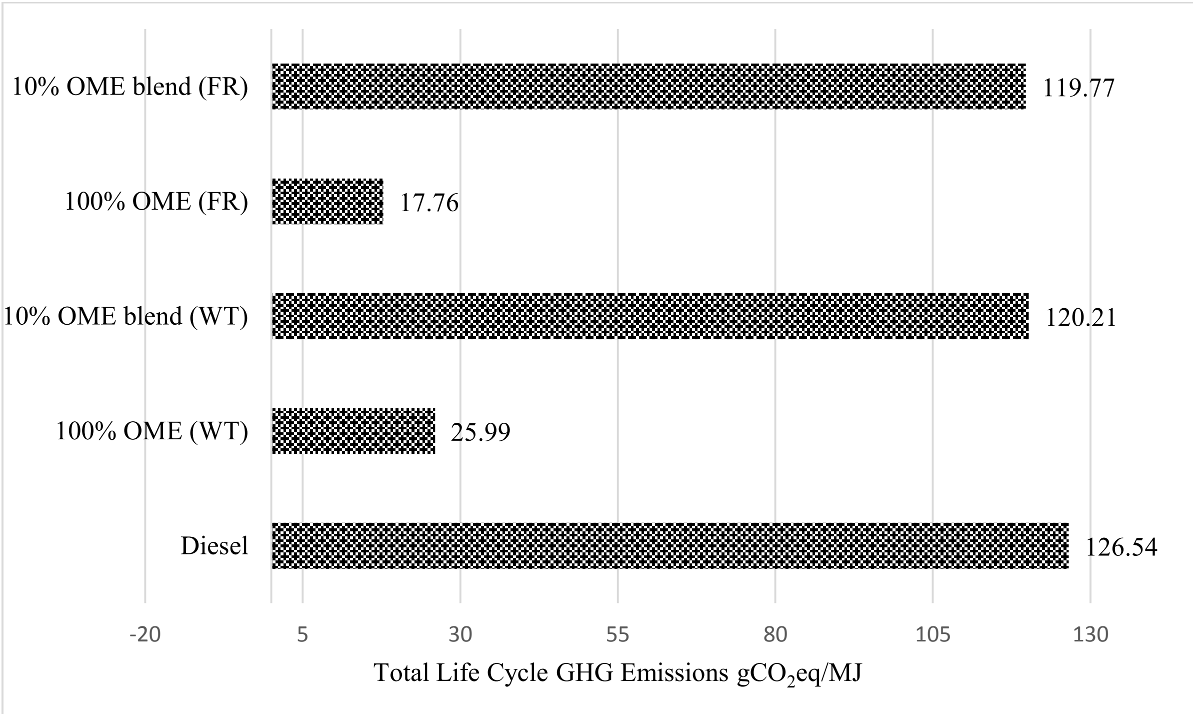

Fig. 6.1: OME and OME blends from whole tree and forest residue pathways' GHG emissions compared with conventional diesel.

Soot emissions were 30% and 89% less than diesel's for the 10% OME blend with diesel and 100% OME, respectively. Since the data inventory and assumptions were taken from different sources, several scenarios were developed for both pathways with varying parameters and assumptions for the upstream operations, i.e., plant scale, capacity factor, biomass moisture content, and biomass yield, as well as the exclusion of road construction and the inclusion of silviculture operations in the whole tree pathway. The scenarios were developed independently of each other and compared with the base case scenario. The results were the same in every scenario as the life cycle GHG emissions were insensitive to parameter variations, which proved the robustness of the developed model. However, when road construction was not included in the whole tree pathway, energy consumption and GHG emissions dropped significantly, by around 33% compared to the base case scenario.

6.1.2 Life cycle Sustainability Assessment of OME Production

A life cycle sustainability assessment (LCSA) framework was developed to evaluate the environmental and socio-economic performances of OME production from two types of forest biomass, whole tree and forest residue. Different scenarios were developed in order to identify the most sustainable OME and diesel blend ratio. A multicriteria outranking method known as PROMETHEE was used to compare the alternative OME production pathways. A combination of linear and usual preference functions was used in the model. The indicators were assigned equal weights in the base case. A sensitivity analysis was conducted to investigate the outcomes from different decision-making scenarios based on changes in weight. The Visual PROMETHEE software was used to generate the alternate ranking results. The environmental impact assessment results indicate that the forest residue pathway has lower net GHG emissions (13 g CO_2eq/MJ) than the whole tree pathway (21 g CO_2eq/MJ). When we look at the contribution analysis by life cycle stages, in both pathways, fuel combustion in vehicles contributes more than 80% of the total life cycle GHG emissions. However, since OME was derived from biomass-based feedstock, fuel combustion in the chemical conversion plant and vehicle was assumed to be zero due to its carbon neutrality over life cycle. The chemical conversion process emitted significantly low GHG emissions, around 4% (≈4 g CO_2eq/MJ). The emissions are mainly from the ash disposal operation and the use of natural gas. The residues were considered to be readily available in the forest and harvested using the existing road network. These are produced as part of logging operations; hence, the biomass harvest emissions were assumed to be zero and the biomass transportation emissions were relatively lower (1.69 gCO_2eq/MJ) in the forest residue pathway than the whole tree (2.96 gCO_2eq/MJ).

The soot emissions were higher in the forest residue pathway (0.004 gm/ MJ) than the whole tree pathway (0.003 gm/MJ) because of higher diesel consumption in the trucks used to transport the biomass. The trucks travelled around 36.17 km to collect the biomass residues from the forest and only 8.65 km to harvest whole trees. The water footprint in both pathways was from biomass growth (almost 99.99%). The contribution from other operations was insignificant. Since OME production is a new energy technology and not yet developed at a commercial scale, the initial investment cost (capital cost) is high. The forest residue pathway has a relatively lower production cost (1.71 $/liter) than the whole tree (1.92 $/liter), which has high capital and operational costs. Biomass yield has a significant impact on transportation cost. Forest residues, with a yield of around 0.247 dry tonne/hectare, has a higher transportation cost ($14.83 / dry tonne forest residue) than whole tree ($11.10 / dry tonne whole tree), which has a higher yield, 84 dry tonne/hectare. Raw material costs, utilities, and labor costs were the major cost components in the life cycle.

The social indicators were evaluated to determine the social acceptance of the new technology. The forest residue pathway has higher employment potential (around 0.27 hours/ m^3 of woody biomass or 0.0004 hours/MJ of OME) than the whole tree (0.15 hours/ m^3 biomass or 0.0003 hours /MJ). The forest residue pathway also offers higher employee wages and benefits ($67.26/dry tonne) than the whole tree ($62.58/dry tonne). Hence, several jobs with different skill categories can be created throughout the different life cycle stages of OME production when this technology is implemented commercially.

Fig. 6.2 shows the base case ranking results in a GAIA (geometrical analysis for interactive assistance) plane. The preferred pathway is shown by the decision axis (the triangle in Fig. 6.2) using the position of the pathways and the orientation of the indicator axes towards the pathways.

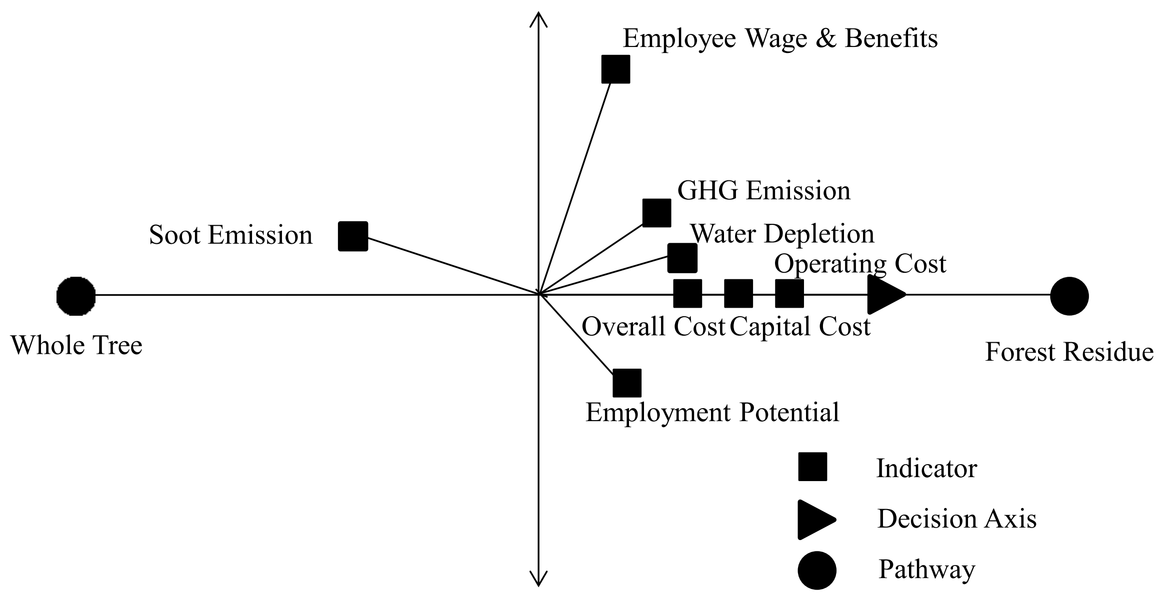

Fig. 6.2: PROMETHEE ranking results for the two pathways

All the indicators excluding the soot emissions were directed towards the decision axis, which indicated that the forest residue is the preferred pathway. The ranking results were presented by three outranking flows: positive, negative, and net (Table 6.1). The forest residue pathway outranked the whole tree pathway in both the PROMETHEE I partial ranking and PROMETHEE II complete ranking. For OME-diesel blend scenarios, higher OME ratios were less preferred than lower ones in all three sustainability dimensions.

Table 6.1 Ranking results for the whole tree and forest residue pathways

Pathways	$\emptyset^{+}$(positive flow)	$\emptyset^{-}$(negative flow)	$\emptyset$(net flow)
Whole Tree	0.13	0.88	-0.75
Forest Residue	0.88	0.13	0.75

Sensitivity analysis was performed to assess the impact of different variables on the final results and to reflect decision makers' preferences. A stability interval was used to conduct a weight sensitivity analysis, across which the ranking was not altered or remained stable. For the base case, the ranking was found to be sensitive only to the soot emissions. The GHG and soot emissions had smaller sensitivity intervals in the OME diesel blends case, which means the ranking was sensitive to the GHG and soot emissions only. Other sensitivity scenarios were developed by changing the weights of different sustainability impacts, threshold values, and indicator values. The rankings remained insensitive to all the changes, which validated the robustness of the LCSA framework. An additional scenario was analyzed in which OME was used as a fuel to transport biomass feedstock from the forest to the chemical conversion plant. In the forest residue pathway, a 90-95% reduction in GHG and soot emissions was possible when OME was used for biomass transportation instead of diesel, but changes in the whole tree pathway were negligible. Based on all the sustainability criteria, the forest residue pathway was found to be the preferred pathway.

6.1.3 Life Cycle Environmental Benefits from Biofuel Coproducts

This study discussed the environmental significance of applying credits to bio-ethanol and biodiesel coproducts based on their potential applications as energy substitutes, animal feed, and fertilizer. The objective was to identify potential alternative applications of coproducts from two different biofuel pathways, dry distiller grains (DDGs) from the wheat to ethanol and canola meal and crude glycerine from the canola to biodiesel pathways. A detailed framework was developed to identify the alternate applications and investigate the impact of coproduct allocation on overall biofuel life cycle GHG emissions. This kind of analysis is important to assess the environmental consequences of long-term policies or the large-scale application of technologies related to biofuels. Biofuel coproduct credits or GHG emission savings were determined by varying the percentage of the coproduct use by mass from 20% to 100%. The largest coproduct credits were found from using DDGs pellets for heating (as an alternative to coal firing). Because of their higher moisture content and lower heating value than fossil coal, DDGs used for heating could emit lower GHGs per unit energy. The coproduct credits ranged from 13.43-67.14g CO_2eq/MJ of ethanol based on the percentage of use.

The second largest coproduct credits were obtained from using DDGs as animal feed, followed by as an organic fertilizer. DDGs are rich in protein content and the digestible energy of DDGs is comparable to that of corn used as animal feed, hence a higher coproduct credit is obtained when DDGs are used as animal feed compared to land applications. The coproduct credits from using DDGs as animal feed ranged from 8.31-17.29 g CO_2eq/MJ of ethanol, and when used as fertilizer, the credits were 1.72-8.61 g CO_2eq/MJ of ethanol.

The coproduct credits from combining different DDGs uses were also determined. Since the highest coproduct credits were obtained from their use as fuels, the credits increased with increasing percentages of coproduct use as fuel in combinations (see Fig. 6.3). Comparatively lower GHG emissions were saved when DDGs displace animal feed or/and urea compared to using coal.

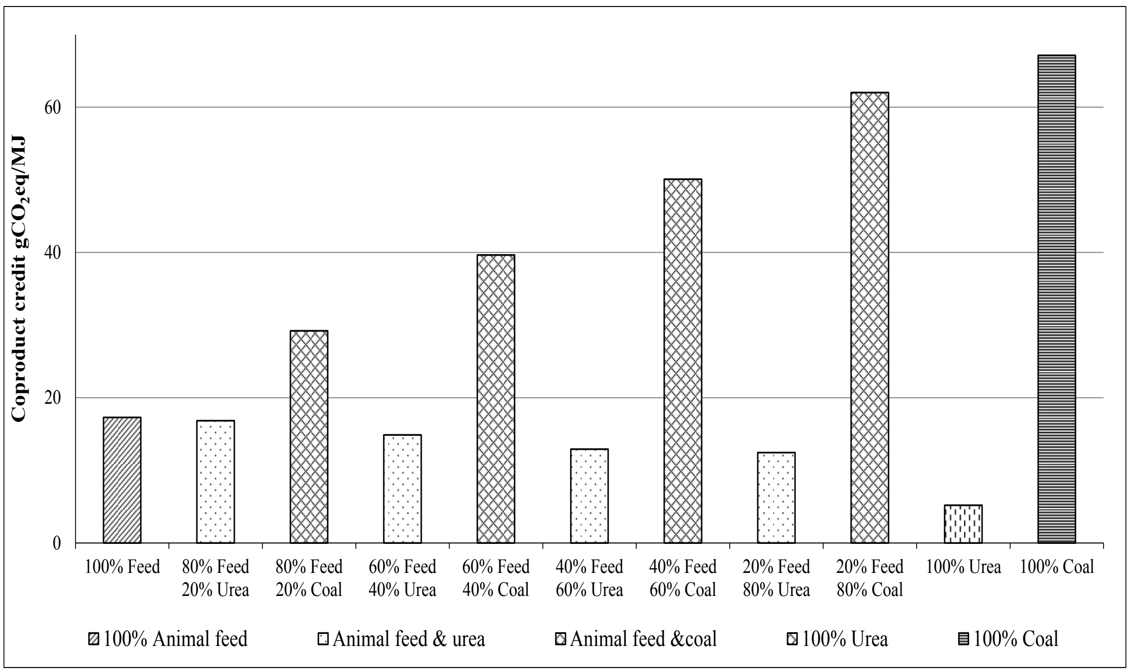

Fig.6.3: Coproduct credit or GHG emissions savings from different combinations (by mass percentage) of DDG use in the wheat to ethanol pathway

In the canola to biodiesel pathway, the coproduct credit ranged from 2.23 to 11.14 g CO_2eq/MJ of biodiesel depending on the percentage of canola meal use. The highest credit was earned when the crude glycerine obtained from the biodiesel processing replaced synthetic glycerine. The coproduct credits ranged from 17.13-3.95 g CO_2eq/MJ when crude glycerine was processed into synthetic glycerine depending on its percentage use. Because of the higher energy consumption and higher emissions associated with synthetic glycerine production from crude glycerine, the displaced emissions for synthetic glycerine (6,590 g CO_2eq/kg glycerine) were the largest, compared to other glycerine uses such as animal feed (0.40 g CO_2eq/MJ of biodiesel) or fuel (0.30 g CO_2eq/MJ of biodiesel). The coproduct credits ranged from 1.04-0.21 g CO_2eq/MJ and 0.78-0.16 gCO_2eq/MJ of biodiesel when glycerine replaced corn in the cattle feed ration and petroleum-based natural gas as an energy source, respectively.

As shown in Fig. 6.4, the biodiesel coproduct credits gradually decreased with deceases in the percentage of glycerine replacing synthetic glycerine in the combination because of the higher displacement emission values compared to the other applications such as fuel or animal feed. The smallest coproduct credit was earned when crude glycerine replaced fuel in the market.

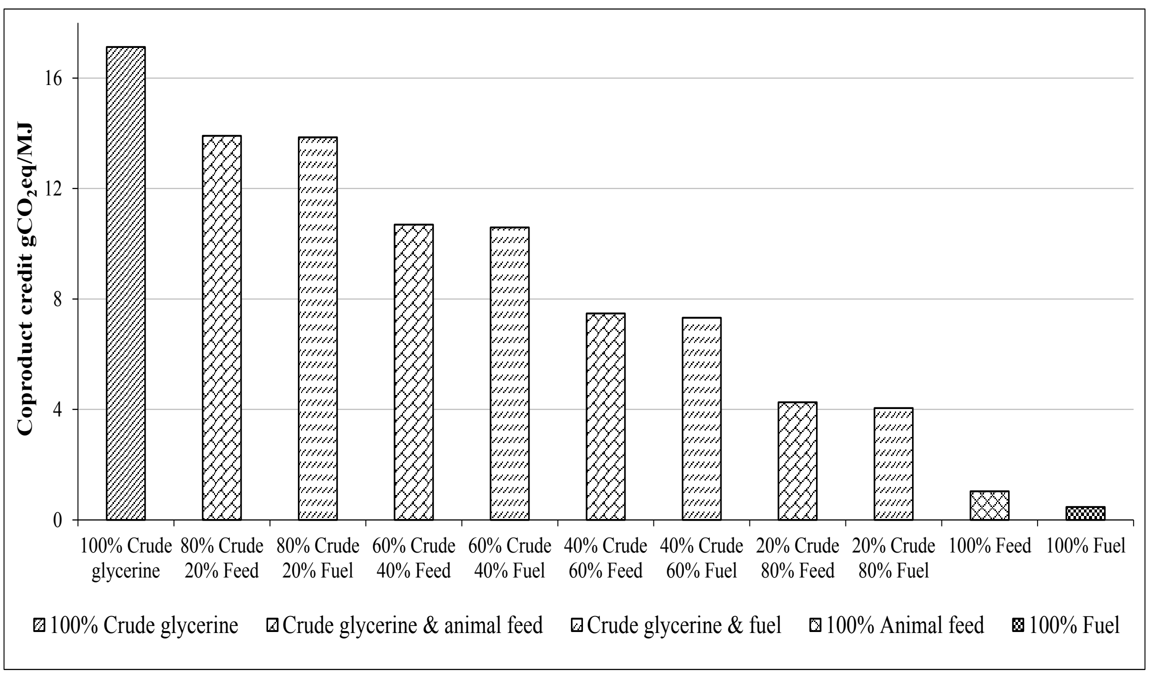

Fig. 6.7 Coproduct credit or GHG emissions savings from different combinations (by mass percentage) of crude glycerine use in the canola to biodiesel pathway

The coproduct credits affect the overall biofuel life cycle depending on the application. When DDGs replace fuel, animal feed, and fertilizer, the ethanol life cycle GHG emissions can decrease by 16-81%, 10-21%, and 1-6%, respectively. Similarly, crude glycerine can decrease the life cycle GHG emissions by 5-26% when canola meal is used as animal feed and by 8-41% when crude glycerine replaces synthetic glycerine in the market. The combined applications of DDGs result in further GHG emissions reduction. Life cycle GHG emissions can be reduced by around 35-75% when DDGs replace fuel and animal feed in combination, and by 15-20% when DDGs replace animal feed and fertilizer in combination. GHG emissions can by reduced by around 10-33% from the combined use of synthetic glycerine and fuel or synthetic glycerine and fertilizer. Results from the sensitivity and uncertainty analyses suggest the combustion energy of coal and DDGs to be the most sensitive and uncertain inputs in the application of DDG as coal for space heating in the wheat to ethanol pathway. Glycerine yield (g/liter of biodiesel) appears to be the most sensitive and uncertain input when crude glycerine replaces synthetic glycerine.

6.1.4 Applications of Consequential LCA on Biofuel Technologies

The systematic literature review was conducted on the current state of C-LCA of biofuel energy systems with the aim of identifying research gaps and areas for future research. The issues crucial to the application of C-LCA to biofuel and bioenergy systems, in particular the methods and empirical aspects, were discussed. One of the fundamental elements in a C-LCA approach is how to identify and select the marginal product or technology. Several studies mention the 5-step approach proposed by Weidema et al. for identifying the marginal technology (Weidema et al. 1999). However, application-oriented demonstration of this method is limited. Different studies have made different conclusions, for instance, that constrained technologies are not suitable as marginal technology both in the short and long term. Some studies found that the technologies that can change or use the capacity partially in response to changes in demand are chosen as the marginal technologies. The diverse findings regarding marginal technologies make C-LCA modelling results ambiguous. The assumptions or rationales used to identify and select the marginal technologies need detailed discussion, since the marginal technology chosen significantly affects the reliability of the results.

Marginal technologies (business as usual, planned/proposed, and actual) have been discussed in studies related to policy, government energy plans, and other long-term energy considerations such as capacity installment or demand change (including both conventional and bioenergy/biofuel systems). However, the studies are mostly economic feasibility studies that use projections and historical statistics on regional energy systems to identify the marginal technologies. Hence, our review concluded that there is no proper demonstration or guidance of C-LCA application on energy systems in general. Application-oriented studies can explain practical allocation facts, system delimitation, how to include/exclude processes related to coproduct use, marginal technology, competing products, and associated uncertainties better than theoretical recommendations do.

C-LCA can model different types of scenarios such as normative, predictive, and explorative with multidirectional objectives assessing the consequences of a wide range of scenarios triggered by demand change. For example, normative scenarios determine the environmental consequences of a biofuel technology with a specific objective; a predictive scenario forecasts the consequence of the large-scale application of a new technology in the market; and an explorative scenario assesses the environmental impacts of the large-scale application of a particular biofuel technology in relation to other perspectives such as industrialist or policy makers' viewpoints, locally or globally.

Three different models are commonly mentioned in the literature to model C-LCA: the simplified model, partial equilibrium model, and computable general equilibrium model. The selection of a C-LCA modelling approach depends on a number of criteria, for instance, market delimitation, production scale, time scope, constraints related to market, multifunctionality, policy, technology, and direct/indirect land use changes. It also includes assessing demand change or any other decision change. However, most biofuel C-LCA studies fail to properly explain the criteria used to select a particular modelling approach to solve the problem.

In terms of mathematical computation, C-LCA modelling approaches are classified as linear (process-based LCA, input-output LCA, partial equilibrium models) and nonlinear (computable general equilibrium models). The scope of the model varies based on assumptions such as market supply, input-output coefficient, and economies of scale. However, both models assume high levels of uncertainty and limitations in practical situations. Hence, to enhance the credibility of the decision making, it is important to evaluate the results from different models rather than rely on results from a particular C-LCA model.

The application areas assessed by C-LCA include production and processing technologies, policy, feedstock comparison, market development, and recycling. However, most biofuel CLCA studies assess the environmental consequences in the context of policy applications. Biofuel C-

LCA studies face challenges with data inventory. Given the lack of detailed information on different biomass feedstocks, crop systems, and newly developing biomass processing technologies, aggregated data are used to develop most of the C-LCA models, which increases uncertainty. Uncertainty in C-LCA studies could be minimized by developing multiple C-LCA scenarios by varying model assumptions such as considering different marginal technologies or products affected, along with uncertainty and sensitivity analyses.

6.2 Recommendations for Future Research

The following recommendations could be considered to advance the research:

- The LCA and LCSA models on OME production can be extended to include other biomass feedstocks such as wood waste and agricultural residues, as well as different modes of transportation. Forest biomass feedstock was considered in this study because forest biomass is a potential source of bioenergy and Alberta has an abundance of forests, which increases the potential further. It is also important to consider other forms of transporting biomass, such as in bales and pellets, and compare these to the transportation considered in this study to find the most environmentally friendly way of biomass transport.
- OME can be produced from natural gas following two conversion processes, steam methane reforming and auto thermal reforming. Thus, the developed LCSA framework can be used to assess the sustainability performance of OME produced from conventional fossil sources to compare it with biomass-derived sources. The coproduct credit model can be extended to include other biofuels to determine the impact of coproduct use on the overall life cycle. The ethanol and biodiesel pathways can be credited for other potential

uses of their coproducts such as acetic acid, electricity, lactic acid, pulp, propylene glycol, etc.

- Develop a comprehensive C-LCA framework for assessing the impacts of biofuel production and import in a particular jurisdiction.
- Conduct C-LCA assessment of biofuel production and import in Canadian environment which could help in policy formulation and decision making.

Reference

(S&T)[2] Consultants Inc. (2004). GHGenius A model for lifecycle assessment of transportation fuels. A model for lifecycle assessment of transportation fuelsAccessed 06/06/2017.

(S&T)[2] Consultants Inc. (2011) "Co-Products Report." http://www.ghgenius.ca/reports.php?tag=Co-products Accessed17 November 2016.

(S&T)[2] Consultants Inc. (2011) "GHGenius Co-Product Update." http://www.ghgenius.ca/reports.php?tag=Co-products Accessed17/11/2016.

(S&T)[2] Consultants Inc. (2013) "GHGenius Manual 4.03 Volume 1 " http://www.ghgenius.ca/reports.php Accessed17/11/2016.

Abas, N., A. Kalair and N. Khan (2015). "Review of fossil fuels and future energy technologies." Futures **69**: 31-49.

Adams, D., R. Alig, B. A. McCarl, B. C. Murray, L. Bair, B. Depro, G. Latta, H.-C. Lee, U. A. Schneider and J. Callaway (2005). FASOMGHG conceptual structure, and specification: documentation, College Station, TX: Texas A&M University http://agecon2.tamu.edu/people/faculty/mccarl-bruce/papers/1212FASOMGHG_doc.pdf Accessed 02/01/2019.

Adler, P. R., S. J. Del Grosso and W. J. Parton (2007). "Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems." Ecological Applications **17**(3): 675-691.

Afgan, N. and M. da Graça Carvalho (2000). Sustainable Assessment Method for Energy Systems: indicators, criteria and decision making procedure, Springer Science & Business Media.

Afgan, N. H., M. G. Carvalho and N. V. Hovanov (2000). "Energy system assessment with sustainability indicators." Energy Policy **28**(9): 603-612.

Afgan, N. H. and M. da Graça Carvalho (2000). Energy system assessment with sustainability indicators. Sustainable Assessment Method for Energy Systems, Springer**:** 83-125.

Afgan, N. H. and M. Darwish (2011). "Multi-criteria sustainability assessment of water desalination and energy systems—Kuwait case." Desalination and Water Treatment **25**(1-3): 241-250.

Agbor, E., X. Zhang and A. Kumar (2014). "A review of biomass co-firing in North America." Renewable and Sustainable Energy Reviews **40**: 930-943.

Andrew, R. and V. Forgie (2008). "Life Cycle Assessment of Canola Biodiesel in New Zealand." Landcare Research and Costing of Canola Biodiesel in New Zealand. Centre for Ecological Economics.

Andrews, E. S. (2009). Guidelines for social life cycle assessment of products: social and socio-economic LCA guidelines complementing environmental LCA and Life Cycle Costing, contributing to the full assessment of goods and services within the context of sustainable development, UNEP/Earthprint.

Argonne National Laboratory (2016). GREET® Model. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation Modelhttps://greet.es.anl.gov/.

Arosemena, A., E. DePeters and J. Fadel (1995). "Extent of variability in nutrient composition within selected by-product feedstuffs." Animal feed science and technology **54**(1): 103-120.

Aspen-Icarus (2014). Aspen Process Economic Analyzer. Version 8.8. Cambridge, Aspen Technology Inc.

Assefa, G. and B. Frostell (2007). "Social sustainability and social acceptance in technology assessment: A case study of energy technologies." Technology in Society **29**(1): 63-78.

Baumann, H. (1998). Life cycle assessment and decision making: theories and practices, Chalmers University of Technologyhttps://is4ie.org/resources/dissertations/112 Accessed 03/02/2019.

Beer, T. and T. Grant (2007). "Life-cycle analysis of emissions from fuel ethanol and blends in Australian heavy and light vehicles." Journal of Cleaner Production **15**(8): 833-837.

Beer, T., T. Grant, G. Morgan, J. Lapszewicz, P. Anyon, J. Edwards, P. Nelson, H. Watson and D. Williams (2001). Comparison of Transport Fuels: Final Report to the Australian Greenhouse Office on the Stage 2 Study of Life-Cycle Emissions—Analysis of Alternative Fuels for Heavy Vehicles, Report No. EV45A/2/F3C, Commonwealth Scientific and Industrial Research Organisation, Aspendale, Victoria, Australia.

Beer, T., T. Grant, D. Williams and H. Watson (2002). "Fuel-cycle greenhouse gas emissions from alternative fuels in Australian heavy vehicles." Atmospheric Environment **36**(4): 753-763.

Beer, T., D. Olaru, M. Van der Schoot, T. Grant, B. Keating, S. Hatfield Dodds, C. Smith, M. Azzi, P. Potterton and D. Mitchell (2003). Appropriateness of a 350 Million Litre Biofuels Target: Report to the Australian Government Department of Industry, Tourism and Resources, CSIRO, ABARE, BTRE, Canberra.

Behzadian, M., R. B. Kazemzadeh, A. Albadvi and M. Aghdasi (2010). "PROMETHEE: A comprehensive literature review on methodologies and applications." European journal of Operational research **200**(1): 198-215.

Belyea, R., K. Rausch and M. Tumbleson (2004). "Composition of corn and distillers dried grains with solubles from dry grind ethanol processing." Bioresource Technology **94**(3): 293-298.

Belyea, R., B. Steevens, R. Restrepo and A. Clubb (1989). "Variation in composition of by-product feeds." Journal of Dairy Science **72**(9): 2339-2345.

Benoît Norris, C., M. Traverso, S. Valdivia, G. Vickery-Niederman, J. Franze, L. Azuero, A. Ciroth, B. Mazijn and D. Aulision (2013). "The Methodological Sheets for Sub-Categories in Social Life Cycle Assessment (S-LCA)." United Nations Environment Programme (UNEP) and Society for Environmental Toxicology and Chemiastry (SETAC).

Bernesson, S., D. Nilsson and P.-A. Hansson (2006). "A limited LCA comparing large-and small-scale production of ethanol for heavy engines under Swedish conditions." Biomass and Bioenergy **30**(1): 46-57.

Bhawana Pathak, S. C. a. M. H. F. (2013). "Biomass - Resource for Sustainable Development." International Journal of Advancements in Research & Technology **2**(6): 271-289.

Blanco-Canqui, H., C. J. Gantzer, S. H. Anderson, E. Alberts and F. Ghidey (2002). "Saturated hydraulic conductivity and its impact on simulated runoff for claypan soils." Soil Science Society of America Journal **66**(5): 1596-1602.

Bodarski, R., T. Wertelecki, F. Bommer and S. Gosiewski (2005). "The changes of metabolic status and lactation performance in dairy cows under feeding TMR with glycerin [glycerol] supplement at periparturient period." Electronic Journal of Polish Agricultural Universities. Series Animal Husbandry **8**(4).

Bond, T. C., S. J. Doherty, D. Fahey, P. Forster, T. Berntsen, B. DeAngelo, M. Flanner, S. Ghan, B. Kärcher and D. Koch (2013). "Bounding the role of black carbon in the climate system: A scientific assessment." Journal of Geophysical Research: Atmospheres **118**(11): 5380-5552.

Börjeson, L., M. Höjer, K.-H. Dreborg, T. Ekvall and G. Finnveden (2006). "Scenario types and techniques: towards a user's guide." Futures **38**(7): 723-739.

Bouman, M., R. Heijungs, E. Van der Voet, J. C. van den Bergh and G. Huppes (2000). "Material flows and economic models: an analytical comparison of SFA, LCA and partial equilibrium models." Ecological economics **32**(2): 195-216.

Brandão, M., R. Clift, A. Cowie and S. Greenhalgh (2014). "The Use of Life Cycle Assessment in the Support of Robust (Climate) Policy Making: Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation…"." Journal of Industrial Ecology **18**(3): 461-463.

Brans, J.-P. and B. Mareschal (2005). "PROMETHEE methods." Multiple criteria decision analysis: state of the art surveys: 163-186.

Brans, J.-P. and P. Vincke (1985). "Note—A Preference Ranking Organisation Method: (The PROMETHEE Method for Multiple Criteria Decision-Making)." Management science **31**(6): 647-656.

Brans, J.-P., P. Vincke and B. Mareschal (1986). "How to select and how to rank projects: The PROMETHEE method." European Journal of Operational Research **24**(2): 228-238.

Bremer, V. R., A. J. Liska, T. J. Klopfenstein, G. E. Erickson, H. S. Yang, D. T. Walters and K. G. Cassman (2010). "Emissions savings in the corn-ethanol life cycle from feeding coproducts to livestock." Journal of Environmental Quality **39**(2): 472-482.

Budsberg, E., M. Rastogi, M. E. Puettmann, J. Caputo, S. Balogh, T. A. Volk, R. Gustafson and L. Johnson (2012). "Life-cycle assessment for the production of bioethanol from willow biomass crops via biochemical conversion." Forest Products Journal **62**(4): 305.

Burchart-Korol, D. (2013). "Life cycle assessment of steel production in Poland: a case study." Journal of Cleaner Production **54**: 235-243.

Canada-Visa. (2014). "Canada Salary Wizard." Retrieved January 25 2015, from http://www.canadavisa.com/canada-salary-wizard.html.

Canter, C. E., J. B. Dunn, J. Han, Z. Wang and M. Wang (2016). "Policy Implications of Allocation Methods in the Life Cycle Analysis of Integrated Corn and Corn Stover Ethanol Production." BioEnergy Research **9**(1): 77-87.

Carmel, I. "Ag Fuel and Feed." Waste Diversion & Recycling Retrieved 17 November, 2016, from http://www.waste.net/wastehaulers/aa1138251.html

Carneiro, M. L. N. M., F. Pradelle, S. L. Braga, M. S. P. Gomes, A. R. F. A. Martins, F. Turkovics and R. N. C. Pradelle (2017). "Potential of biofuels from algae: Comparison with fossil fuels, ethanol and biodiesel in Europe and Brazil through life cycle assessment (LCA)." Renewable and Sustainable Energy Reviews **73**: 632-653.

Carrera, D. G. and A. Mack (2010). "Sustainability assessment of energy technologies via social indicators: Results of a survey among European energy experts." Energy Policy **38**(2): 1030-1039.

Charlotte Helston. (2012, May 2012). "Energy Profiles: Biofuels " Retrieved November 17 2016, 2016, from http://www.energybc.ca/biofuels.html

Cherubini, F. (2010). "The biorefinery concept: using biomass instead of oil for producing energy and chemicals." Energy Conversion and Management **51**(7): 1412-1421.

Cherubini, F., N. D. Bird, A. Cowie, G. Jungmeier, B. Schlamadinger and S. Woess-Gallasch (2009). "Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations." Resources, Conservation and recycling **53**(8): 434-447.

Cherubini, F. and A. H. Strømman (2011). "Life cycle assessment of bioenergy systems: State of the art and future challenges." Bioresource Technology **102**(2): 437-451.

Chum, H. L. and R. P. Overend (2001). "Biomass and renewable fuels." Fuel Processing Technology **71**(1): 187-195.

Ciroth, A., M. Finkbeier, J. Hildenbrand, W. Klöpffer, B. Mazijn and S. Prakash (2011) "Towards a life cycle sustainability assessment: Making informed choices on products." http://wedocs.unep.org/bitstream/handle/20.500.11822/8001/UNEP_LifecycleInit_Dec_FINAL.pdf?sequence=3&isAllowed=y Accessed.

Ciroth, A., M. Finkbeier, J. Hildenbrand, W. Klöpffer, B. Mazijn, S. Prakash, G. Sonnemann, M. Traverso, C. M. L. Ugaya, S. Valdivia and G. Vickery-Niederman (2011). Towards a live cycle sustainability assessment: making informed choices on products, United Nations Environment Programma (UNEP)https://www.lifecycleinitiative.org/wp-content/uploads/2012/12/2011%20-%20Towards%20LCSA.pdf Accessed 02/05/2016.

Collet, P., A. Hélias Arnaud, L. Lardon, M. Ras, R. A. Goy and J. P. Steyer (2011). "Life-cycle assessment of microalgae culture coupled to biogas production." Bioresource Technology **102**(1): 207-214.

Commission, E. (2003). DIRECTIVE 2003/30/EC on the promotion of the use of energy from renewable sources amending and subsequently
repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union**:** 42.

Commission, E. (2009). DIRECTIVE 2009/28/EC on the promotion of the use of energy from renewable sources amending and subsequently
repealing Directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union**:** 16.

Crandall, L. (2004). "Glycerol abundance cause for concern." Inform **15**: 146-147.

Dalgaard, R. and I. Muñoz (2014). Consequential and attributional modeling in life cycle assessment of food production systems. Proceedings of the 9th International Conference on Life

Cycle Assessment in the Agri-Food Sector (LCA Food 2014), San Francisco, California, USA, 8-10 October, 2014, American Center for Life Cycle Assessment.

Dandres, T., C. Gaudreault, P. Tirado-Seco and R. Samson (2011). "Assessing non-marginal variations with consequential LCA: application to European energy sector." Renewable and Sustainable Energy Reviews **15**(6): 3121-3132.

Dandres, T., C. Gaudreault, P. Tirado-Seco and R. Samson (2012). "Macroanalysis of the economic and environmental impacts of a 2005–2025 European Union bioenergy policy using the GTAP model and life cycle assessment." Renewable and Sustainable Energy Reviews **16**(2): 1180-1192.

Demirbas, A. (2004). "Combustion characteristics of different biomass fuels." Progress in Energy and Combustion Science **30**(2): 219-230.

Deutsch, D., D. Oestreich, L. Lautenschütz, P. Haltenort, U. Arnold and J. Sauer (2017). "High Purity Oligomeric Oxymethylene Ethers as Diesel Fuels." Chemie Ingenieur Technik **89**(4): 486-489.

Di Lullo, G., H. Zhang and A. Kumar (2017). "Uncertainty in well-to-tank with combustion greenhouse gas emissions of transportation fuels derived from North American crudes." Energy **128**: 475-486.

Dincer, I. (2007). "Environmental and sustainability aspects of hydrogen and fuel cell systems." International journal of energy research **31**(1): 29-55.

Dominguez-Faus, R., S. E. Powers, J. G. Burken and P. J. Alvarez (2009). "The water footprint of biofuels: A drink or drive issue?" Environmental science & technology **43**(9): 3005-3010.

Donkin, S. and P. Doane (2008). Glycerol from Biodiesel Production: The New Corn for Dairy Cattle http://ag.arizona.edu/ANS/swnmc/Proceedings/2008/14Donkin_08.pdf.

Donkin, S., S. Koser, H. White, P. Doane and M. Cecava (2009). "Feeding value of glycerol as a replacement for corn grain in rations fed to lactating dairy cows." Journal of Dairy Science **92**(10): 5111-5119.

Donkin, S. S. (2008). "Glycerol from biodiesel production: the new corn for dairy cattle." Revista Brasileira de Zootecnia **37**(SPE): 280-286.

Dowlatabadi, H. (1995). "Integrated assessment models of climate change: An incomplete overview." Energy Policy **23**(4-5): 289-296.

Duchin, F. and S. H. Levine (2011). "Sectors may use multiple technologies simultaneously: The rectangular choice-of-technology model with binding factor constraints." Economic Systems Research **23**(3): 281-302.

Earles, J. M. and A. Halog (2011). "Consequential life cycle assessment: a review." The International Journal of Life Cycle Assessment **16**(5): 445-453.

Earles, J. M., A. Halog, P. Ince and K. Skog (2013). "Integrated economic equilibrium and life cycle assessment modeling for policy-based consequential LCA." Journal of Industrial Ecology **17**(3): 375-384.

ECCC (2018). Clean Fuel Standard, Regulatory Design Paper, Environment and Climate Change.

Ekvall, T. (1999). "System expansion and allocation in life cycle assessment with implications for wastepaper management." Doktorsavhandlingar vid Chalmers Tekniska Hogskola(1509): 1-54.

Ekvall, T. (2002). "Cleaner production tools: LCA and beyond." Journal of Cleaner Production **10**(5): 403-406.

Ekvall, T. and A. Andrae (2006). "Attributional and consequential environmental assessment of the shift to lead-free solders (10 pp)." The International Journal of Life Cycle Assessment **11**(5): 344-353.

Ekvall, T. and G. Finnveden (2001). "Allocation in ISO 14041—a critical review." Journal of Cleaner Production **9**(3): 197-208.

Ekvall, T. and A.-M. Tillman (1997). "Open-loop recycling: criteria for allocation procedures." The International Journal of Life Cycle Assessment **2**(3): 155.

Ekvall, T. and B. P. Weidema (2004). "System boundaries and input data in consequential life cycle inventory analysis." The International Journal of Life Cycle Assessment **9**(3): 161-171.

Elghali, L., R. Clift, P. Sinclair, C. Panoutsou and A. Bauen (2007). "Developing a sustainability framework for the assessment of bioenergy systems." Energy Policy **35**(12): 6075-6083.

Energy, A. (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta.

Environment and Climate Change Canada (2018). Clean Fuel Standard, Regulatory Design Paper, Environment and Climate Change Canada,https://www.canada.ca/content/dam/eccc/documents/pdf/climate-change/clean-fuel-standard-regulatory-design-paper-2018-en-1.pdf Accessed 10/01/2019.

Environment and Climate Change Canada (2019a). National Inventory Report 1990-2017: Greenhouse Gas Sources and Sinks in Canada Environment and Climate Change Canada (ECCC),https://www.canada.ca/en/environment-climatechange/services/climate-change/greenhouse-gas-emissions.html Accessed 05/05/2019.

Environment and Climate Change Canada (2019b). Canadian Environmental Sustainability Indicators : Greenhouse gas emissions, Environment and Climate Change Canada

(ECCC),https://www.canada.ca/en/environment-climate-change/services/environmental-indicators/greenhouse-gas-emissions.html Accessed 01/05/2019.

Eriksson, G., A. Grimm, N. Skoglund, D. Boström and M. Öhman (2012). "Combustion and fuel characterisation of wheat distillers dried grain with solubles (DDGS) and possible combustion applications." Fuel **102**: 208-220.

Escobar, N., J. Ribal, G. Clemente and N. Sanjuán (2014). "Consequential LCA of two alternative systems for biodiesel consumption in Spain, considering uncertainty." Journal of Cleaner Production **79**: 61-73.

Fabiosa, J. F., J. C. Beghin, F. Dong, A. Elobeid, S. Tokgoz and T.-H. Yu (2010). "Land allocation effects of the global ethanol surge: Predictions from the international FAPRI model." Land Economics **86**(4): 687-706.

Falano, T., H. K. Jeswani and A. Azapagic (2014). "Assessing the environmental sustainability of ethanol from integrated biorefineries." Biotechnology Journal **9**(6): 753-765.

Farrell, A. E., R. J. Plevin, B. T. Turner, A. D. Jones, M. O'Hare and D. M. Kammen (2006). "Ethanol Can Contribute to Energy and Environmental Goals." Science **311**(5760): 506-508.

Feed Opportunities from Biofuels Industries Network. (2013). "Wheat DDGS Feed Guide: Wheat Dried Distiller Grains with Solubles." Retrieved September 2017, from https://cigi.ca/wp-content/uploads/2013/02/DDGS-Feed-Guide_Revised_Jan.-2013.pdf.

Finnveden, G., M. Z. Hauschild, T. Ekvall, J. Guinée, R. Heijungs, S. Hellweg, A. Koehler, D. Pennington and S. Suh (2009). "Recent developments in life cycle assessment." Journal of environmental management **91**(1): 1-21.

Flato, G., J. Marotzke, B. Abiodun, P. Braconnot, S. C. Chou, W. J. Collins, P. Cox, F. Driouech, S. Emori and V. Eyring (2013). "Evaluation of climate models. In: climate change 2013: the physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change." Climate Change 2013 **5**: 741-866.

FOBI (2013). Wheat DDGS Feed Guide: Wheat Dried Distiller Grains with Solubles, Feed Opportunities from Biofuels Industries (FOBI)https://cigi.ca/wp-content/uploads/2013/02/DDGS-Feed-Guide_Revised_Jan.-2013.pdf Accessed 25/02/2018.

Fortuna, M. J. A. and S. R. Rege (2010). Computable General Equilibrium Models: Theory and Applications, Universidade dos Açores, Centro de Estudos de Economía Aplicad do Atlântico ….

Francois, J. F. and K. A. Reinert (1997). Applied methods for trade policy analysis: a handbook, Cambridge University Press.

Frischknecht, R. (1998). "Life cycle inventory analysis for decision-making." The International Journal of Life Cycle Assessment **3**(2): 67-67.

Fülöp, J. (2005). Introduction to decision making methods. BDEI-3 Workshop, Washington, Citeseer.

Fuss, S., J. G. Canadell, G. P. Peters, M. Tavoni, R. M. Andrew, P. Ciais, R. B. Jackson, C. D. Jones, F. Kraxner and N. Nakicenovic (2014). "Betting on negative emissions." Nature Climate Change **4**(10): 850.

G. Angevine and V. Oviedo (2012). Ensuring canadian access to the oil markets in the asia-pacific region, Fraser institute**:** 1-62
https://www.fraserinstitute.org/studies/ensuring-canadian-access-oil-markets-asia-pacific-region.

Garraín, D., C. de la Rúa and Y. Lechón (2016). "Consequential effects of increased biofuel demand in Spain: Global crop area and CO2 emissions from indirect land use change." Biomass and Bioenergy **85**: 187-197.

Genc, T. (2014). "Sensitivity analysis on PROMETHEE and TOPSIS weights." International Journal of Management and Decision Making **13**(4): 403-421.

Gentleman, R., R. Ihaka, D. Bates, J. Chambers, J. Dalgaard and K. Hornik (2009). "The R project for statistical computing." URL: http://www. r-project. org/254.

Government-of-Canada (2016). Forestry: Region of Western Canada and the Territories: 2014-2016http://www.edsc.gc.ca/img/edsc-esdc/jobbank/SectoralProfiles/WT/2A71-RPT-SectProf_WT_Forestry_EN--GEN-20150827-VF-MR.pdf.

Government of Alberta. (2019a). "Renewable Fuels Standard." Retrieved 01/05/2019, from https://www.alberta.ca/renewable-fuels-standard.aspx?utm_source=redirector

Government of Alberta (2019b). Carbon Competitiveness Incentive Regulation. Alberta, Canadahttps://www.alberta.ca/carbon-competitiveness-incentive-regulation.aspx Accessed 20/06/2019.

Guinee, J. (2016). Life cycle sustainability assessment: What is it and what are its challenges? Taking Stock of Industrial Ecology, Springer**:** 45-68.

Guinée, J., M. Gorrée, R. Heijungs, G. Huppes, R. Kleijn, A. De Koning, L. Van Oers, A. Wegener Sleewijk, S. Suh and U. de Haes HA (2004). "Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards (Eco-Efficiency in Industry and Science)."

Guinee, J. B., R. Heijungs, G. Huppes, A. Zamagni, P. Masoni, R. Buonamici, T. Ekvall and T. Rydberg (2010). "Life cycle assessment: past, present, and future." Environmental Science and Technology **45**(1): 90-96.

Hamelin, L., I. Naroznova and H. Wenzel (2014). "Environmental consequences of different carbon alternatives for increased manure-based biogas." Applied Energy **114**: 774-782.

Hamelin, L., P. A. Z. Puławy and E. Ecoinnovazione (2017). "Consequential Life Cycle Assessment: What, How, and Why?" Encyclopedia of Sustainable Technologies: 277.

Heirigs, P., P. Hodson, S. Kaffka, D. O'Connor, M. Stowers and O. Vessia (2010). "Issues Related to Accounting for Co-Product Credits in the California Low Carbon Fuel Standard." Subgroup on Issues Related to Co-Product Credits, State of California Air Resources Board, Sacramento, CA.

Henderson, S. (2000). Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario. Agriculture and Agri-Food Canadahttps://ghgenius.ca/reports/AAFCCornEthanol.pdf Accessed 05/02/2018.

Henderson, S. (2000). "Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario." Agriculture and Agri-Food Canada.

Hill, J., E. Nelson, D. Tilman, S. Polasky and D. Tiffany (2006). "Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels." Proceedings of the National Academy of sciences **103**(30): 11206-11210.

Hoekstra, A. Y., A. K. Chapagain, M. M. Aldaya and M. M. Mekonnen (2011). The Water Footprint Assessment Manual: Setting the Global Standard London, Earthscan.

IEA (2016) "CO_2 emissions from fuel combustion." https://www.iea.org/publications/freepublications/publication/CO2EmissionsfromFuelCombustion_Highlights_2016.pdf Accessed.

IEA (2017). CO2 Emissions from Fuel Combustion 2017 https://webstore.iea.org/co2-emissions-from-fuel-combustion Accessed 05/05/2019.

Inc. (S & T)2 Consultants (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions, Alberta Innovates - Energy and

Environment Solutions, https://books.google.ca/books/about/Life_Cycle_Assessment_of_Selected_Biofue.html?id=tT1pngEACAAJ&redir_esc=y.

Inc., S. T. C. (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions.

Inc.(S & T)^2 Consultants (2013). Life Cycle Assessment of Selected Biofuel Pathways in Alberta, Alberta Innovates - Energy and Environment Solutions, Alberta Innovates - Energy and Environment Solutions, https://books.google.ca/books/about/Life_Cycle_Assessment_of_Selected_Biofue.html?id=tT1pngEACAAJ&redir_esc=y.

International Organization for Standardization (2006-07). ISO 14040:2006 Environmental Management–Life Cycle Assessment–Principles and Framework International Organization for Standardization (ISO),https://www.iso.org/standard/37456.html Accessed 05/01/2018.

International Organization for Standardization (2006-07). ISO 14044:2006 Environmental management -- Life cycle assessment -- Requirements and guidelines, International Organization for Standardization (ISO),https://www.iso.org/standard/38498.html Accessed 05-09-2018.

ISO (2006). Environmental Management–Life Cycle Assessment–Principles and Framework (ISO 14040). Geneva, The International Organization for Standardization (ISO).

ISO International Organization for Standardization (2006-07a). ISO 14040:2006 Environmental Management–Life Cycle Assessment–Principles and Framework https://www.iso.org/standard/37456.html Accessed 05/01/2018.

ISO International Organization for Standardization (2006-07b). ISO 14044:2006 Environmental management -- Life cycle assessment -- Requirements and guidelineshttps://www.iso.org/standard/38498.html Accessed 05-09-2018.

Kabir, M. R. and A. Kumar (2011). "Development of net energy ratio and emission factor for biohydrogen production pathways." Bioresource Technology **102**(19): 8972-8985.

Kabir, M. R. and A. Kumar (2012). "Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways." Bioresource Technology **124**: 394-405.

Kabir, M. R., B. Rooke, G. Dassanayake and B. A. Fleck (2012). "Comparative life cycle energy, emission, and economic analysis of 100 kW nameplate wind power generation." Renewable Energy **37**(1): 133-141.

Kajaste, R. (2014). "Chemicals from biomass–managing greenhouse gas emissions in biorefinery production chains–a review." Journal of Cleaner Production **75**: 1-10.

Kallio, A., O. Salminen and R. Sievänen (2013). "Sequester or substitute—consequences of increased production of wood based energy on the carbon balance in Finland." Journal of forest economics **19**(4): 402-415.

Karlsson, H., S. Ahlgren, I. Strid and P.-A. Hansson (2015). "Faba beans for biorefinery feedstock or feed? Greenhouse gas and energy balances of different applications." Agricultural Systems **141**: 138-148.

Kaufman, A. S., P. J. Meier, J. C. Sinistore and D. J. Reinemann (2010). "Applying life-cycle assessment to low carbon fuel standards—How allocation choices influence carbon intensity for renewable transportation fuels." Energy Policy **38**(9): 5229-5241.

Keesom, W., S. Unnasch, J. Moretta and J. Consultancy (2009). Life cycle assessment comparison of North American and imported crudes, Alberta Energy Research Institute.

Kendall, A. and B. Chang (2009). "Estimating life cycle greenhouse gas emissions from corn–ethanol: a critical review of current U.S. practices." Journal of Cleaner Production **17**(13): 1175-1182.

Kendall, A. and J. Yuan (2013). "Comparing life cycle assessments of different biofuel options." Current Opinion in Chemical Biology **17**(3): 439-443.

Khalili, H., T. Varvikko, V. Toivonen, K. Hissa and M. Suvitie (1997). "The effects of added glycerol or unprotected free fatty acids or a combination of the two on silage intake, milk production, rumen fermentation and diet digestibility in cows given grass silage based diets." Agricultural and Food Science in Finland **6**(5-6).

Khoshnevisan, B., M. Shafiei, M. A. Rajaeifar and M. Tabatabaei (2016). "Biogas and bioethanol production from pinewood pre-treated with steam explosion and N-methylmorpholine-N-oxide (NMMO): A comparative life cycle assessment approach." Energy **114**: 935-950.

Kim, S. and B. E. Dale (2002). "Allocation procedure in ethanol production system from corn grain I. system expansion." The International Journal of Life Cycle Assessment **7**(4): 237-243.

King, C. W. and M. E. Webber (2008). "Water intensity of transportation." Environmental science & technology **42**(21): 7866-7872.

Kloepffer, W. (2008). "Life cycle sustainability assessment of products." The International Journal of Life Cycle Assessment **13**(2): 89.

Klöpffer, W. and I. Renner (2007). "Lebenszyklusbasierte Nachhaltigkeitsbewertung von Produkten." Technikfolgenabschätzung–Theorie und Praxis (TATuP) **16**(3): 32-38.

Kloverpris, J., H. Wenzel and P. H. Nielsen (2008). "Life cycle inventory modelling of land use induced by crop consumption-Part 1: Conceptual analysis and methodological proposal." International Journal of Life Cycle Assessment **13**(1): 13-21.

Kløverpris, J. H. (2008). "Consequential life cycle inventory modelling of land use induced by crop consumption." Industrial PhD Thesis. April.

Kløverpris, J. H. (2009). "Identification of biomes affected by marginal expansion of agricultural land use induced by increased crop consumption." Journal of Cleaner Production **17**(4): 463-470.

Kodera, K. (2007). "Analysis of allocation methods of bioethanol LCA." Internship report, Faculty of Earth and Life Science, MSc in Environmental Resource and Management, Amsterdam.

Kumar, A., J. B. Cameron and P. C. Flynn (2003). "Biomass power cost and optimum plant size in western Canada." Biomass and bioenergy **24**(6): 445-464.

Kumar, A., S. Sokhansanj and P. C. Flynn (2006). "Development of a multicriteria assessment model for ranking biomass feedstock collection and transportation systems." Applied Biochemistry and Biotechnology **129**(1-3): 71-87.

Kumar, M., A. O. Oyedun and A. Kumar (2017). "Hydrothermal liquefaction of biomass for the production of diluents for bitumen transport." Biofuels, Bioproducts and Biorefining **11**: 811-829.

Kumar, P., R. Britter and N. Gupta (2009). "Hydrogen fuel: Opportunities and barriers." Journal of fuel cell science and technology **6**(2): 021009-021009-021007.

Laboratory, A. N. (2016). GREET® Model. The Greenhouse gases, Regulated Emissions, and Energy use in Transportation Model.

Lardy, G. P. and V. L. Anderson (2014). Feeding coproducts of the ethanol industry to beef cattle, NDSU Extension Service https://www.ag.ndsu.edu/publications/livestock/feeding-coproducts-of-the-ethanol-industry-to-beef-cattle/as1242.pdf Accessed 07/08/2015.

Le Quéré, C., R. M. Andrew, P. Friedlingstein, S. Sitch, J. Pongratz, A. C. Manning, J. I. Korsbakken, G. P. Peters, J. G. Canadell and R. B. Jackson (2018). "Global carbon budget 2017." Earth System Science Data Discussions **10**: 405-448.

Le Quéré, C., J. I. Korsbakken, C. Wilson, J. Tosun, R. Andrew, R. J. Andres, J. G. Canadell, A. Jordan, G. P. Peters and D. P. van Vuuren (2019). "Drivers of declining CO 2 emissions in 18 developed economies." Nature Climate Change **9**(3): 213-217.

Levelton Engineering Ltd. & $(S\&T)^2$ Consultants Inc. (1999). Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario https://ghgenius.ca/reports/AAFCCornEthanol.pdf Accessed 05/10/2015.

Li, X., J. Grace, C. Lim, A. Watkinson, H. Chen and J. Kim (2004). "Biomass gasification in a circulating fluidized bed." Biomass and bioenergy **26**(2): 171-193.

Lim, S. and K. T. Lee (2011). "Parallel production of biodiesel and bioethanol in palm-oil-based biorefineries: life cycle assessment on the energy and greenhouse gases emissions." Biofuels, Bioproducts and Biorefining **5**(2): 132-150.

Linke, P., J. DeFrain, A. Hippen and P. Jardon (2004). "Ruminal and plasma responses in dairy cows to drenching or feeding glycerol." J. Dairy Sci **87**(Suppl 1): 343.

Linn, J. and M. Raeth-Knight. (2016). "A Biodiesel byproduct may substitute for corn" Retrieved 17 November, 2016, from http://www.extension.umn.edu/agriculture/dairy/feed-and-nutrition/a-biodiesel-byproduct-may-substitute-for-corn/.

Littlejohns, J., L. Rehmann, R. Murdy, A. Oo and S. Neill (2018). "Current state and future prospects for liquid biofuels in Canada." Biofuel Research Journal **5**(1): 759-779.

Löfvenberg, U. (2010). "The BEST experiences with low blends in diesel and petrol fuels." BEST (BioEthanol for Sustainable Transport) Deliverable(D3).

Longhurst, P. J. B.-I., Juliet; Manovic, Vasilije (2016). "Biomass resources and biofuels potential for the production of transportation fuels in Nigeria." Renewable and Sustainable Energy Reviews **63**: 172-192

Lory, J. A., R. E. Massey, C. D. Fulhage, M. C. Shannon, R. L. Belyea and J. M. Zulovich (2008). "Comparing the Feed, Fertilizer, and Fuel Value of Distiller's Grains." Crop Management **7**(1): 0-0.

Luk, J., H. Fernandes and A. Kumar (2010). "A conceptual framework for siting biorefineries in the Canadian Prairies." Biofuels, Bioproducts and Biorefining **4**(4): 408-422.

Lund, H. and B. V. Mathiesen (2009). "Energy system analysis of 100% renewable energy systems—The case of Denmark in years 2030 and 2050." Energy **34**(5): 524-531.

Luo, L., E. van der Voet, G. Huppes and H. A. Udo de Haes (2009). "Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol." The International Journal of Life Cycle Assessment **14**(6): 529-539.

Machi, L. A. and B. T. McEvoy (2016). The literature review: Six steps to success, Corwin Press.

Macintosh, A., H. Keith and D. Lindenmayer (2015). "Rethinking forest carbon assessments to account for policy institutions." Nature Climate Change **5**(10): 946.

Magarey, J. M. (2001). "Elements of a systematic review." International Journal of Nursing Practice **7**(6): 376-382.

Mahbub, N., E. Gemechu, H. Zhang and A. Kumar (2019). "The life cycle greenhouse gas emission benefits from alternative uses of biofuel coproducts." Sustainable Energy Technologies and Assessments **34**: 173-186.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**(Supplement C): 1249-1262.

Mahbub, N., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2017). "A life cycle assessment of oxymethylene ether synthesis from biomass-derived syngas as a diesel additive." Journal of Cleaner Production **165**: 1249-1262.

Mahbub, N., A. O. Oyedun, H. Zhang, A. Kumar and W.-R. Poganietz (2018). "A life cycle sustainability assessment (LCSA) of oxymethylene ether as a diesel additive produced from forest biomass." The International Journal of Life Cycle Assessment: 1-19.

Malça, J. and F. Freire (2004). Life cycle energy analysis for bioethanol: allocation methods and implications for energy efficiency and renewability. 17th International Conference on Efficiency, Costs, Optimization, Simulation and Environmental Impact of Energy and Process Systems.

Malça, J. and F. Freire (2006). "Renewability and life-cycle energy efficiency of bioethanol and bio-ethyl tertiary butyl ether (bioETBE): assessing the implications of allocation." Energy **31**(15): 3362-3380.

Mann, M. K. and P. L. Spath (1997). Life cycle assessment of a biomass gasification combined-cycle power system, National Renewable Energy Lab., Golden, CO (US).

Mansur, D., T. Tago and T. Masuda (2018). "Utilization of DDGS using ethanol solution for biocrude oil production by hydrothermal liquefaction." Biofuels **9**(3): 325-330.

Maps, B. E. and W. E. Matters (2012). "How Biofuels Work."

Mareschal, B. (2013). "Visual PROMETHEE 1.4 Manual." Retrieved **16**: 2013.

Marvuglia, A., E. Benetto, S. Rege and C. Jury (2013). "Modelling approaches for consequential life-cycle assessment (C-LCA) of bioenergy: critical review and proposed framework for biogas production." Renewable and Sustainable Energy Reviews **25**: 768-781.

Mathiesen, B. V., M. Münster and T. Fruergaard (2009). "Uncertainties related to the identification of the marginal energy technology in consequential life cycle assessments." Journal of Cleaner Production **17**(15): 1331-1338.

McKendry, P. (2002). "Energy production from biomass (part 1): overview of biomass." Bioresource Technology **83**(1): 37-46.

McManus, M. C. and C. M. Taylor (2015). "The changing nature of life cycle assessment." Biomass and Bioenergy **82**: 13-26.

Melamu, R. and H. Von Blottnitz (2011). "2nd Generation biofuels a sure bet? A life cycle assessment of how things could go wrong." Journal of Cleaner Production **19**(2-3): 138-144.

Menten, F., S. Tchung-Ming, D. Lorne and F. Bouvart (2015). "Lessons from the use of a long-term energy model for consequential life cycle assessment: the BTL case." Renewable and Sustainable Energy Reviews **43**: 942-960.

Millar, R. J., J. S. Fuglestvedt, P. Friedlingstein, J. Rogelj, M. J. Grubb, H. D. Matthews, R. B. Skeie, P. M. Forster, D. J. Frame and M. R. Allen (2017). "Emission budgets and pathways consistent with limiting warming to 1.5 C." Nature Geoscience **10**(10): 741.

Mohamadabadi, H. S., G. Tichkowsky and A. Kumar (2009). "Development of a multi-criteria assessment model for ranking of renewable and non-renewable transportation fuel vehicles." Energy **34**(1): 112-125.

Moore, A. (2011). Fertilizer potential of biofuel byproducts, INTECH Open Access Publisher.

Moore, C. C. S., A. R. Nogueira and L. Kulay (2017). "Environmental and energy assessment of the substitution of chemical fertilizers for industrial wastes of ethanol production in sugarcane cultivation in Brazil." The International Journal of Life Cycle Assessment **22**(4): 628-643.

Moore, F. T. (1959). "Economies of scale: Some statistical evidence." The Quarterly Journal of Economics: 232-245.

Morse, D., H. Head, C. Wilcox, H. Van Horn, C. Hissem and B. Harris (1992). "Effects of concentration of dietary phosphorus on amount and route of excretion." Journal of Dairy Science **75**(11): 3039-3049.

Msangi, S., M. Ewing and M. Rosegrant (2010). Biofuels and agricultural growth: challenges for developing agricultural economies and opportunities for investment. Handbook of Bioenergy Economics and Policy, Springer**:** 73-90.

Munawar, E., M. Ubaura, N. Goto and K. Fujie (2003). "Estimation CO2, Non-CO2 GHGs and Other Gas pollutant Emissions of Indonesia's Urea Fertilizer Factories." World **45**: 100.

Myhre, G., D. Shindell, F. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J. Lamarque, D. Lee and B. Mendoza (2013). "Anthropogenic and natural radiative forcing." Climate change **423**.

Nguyen, T. L. T., J. E. Hermansen and R. G. Nielsen (2013). "Environmental assessment of gasification technology for biomass conversion to energy in comparison with other alternatives: the case of wheat straw." Journal of Cleaner Production **53**: 138-148.

Niven, R. K. (2005). "Ethanol in gasoline: environmental impacts and sustainability review article." Renewable and Sustainable Energy Reviews **9**(6): 535-555.

NRCan. (2018, 26/09/2018). "How much forest does Canada have? ." Retrieved 06/05/2019, from https://www.nrcan.gc.ca/forests/report/area/17601

NRCan. (2019, 27/05/2019). "Energy and Greenhouse Gas Emissions (GHGs)." Retrieved 20/05/2019, 2019, from https://www.nrcan.gc.ca/energy/facts/energy-ghgs/20063

OECD/FAO (2018). OECD-FAO Agricultural Outlook 2018-2027, OECD and Food and Agriculture Organization of the United Nationshttps://doi.org/10.1787/agr_outlook-2018-en.

Oestreich, D., L. Lautenschütz, U. Arnold and J. Sauer (2017). "Reaction kinetics and equilibrium parameters for the production of oxymethylene dimethyl ethers (OME) from methanol and formaldehyde." Chemical Engineering Science **163**: 92-104.

OMNR (1994). Reasons for Decision and Decision: Class Environmental Assessment by the Ministry of Natural Resources for Timber Management on Crown Lands in Ontario, Environmental Assessment Board: Ontario Ministry of Natural Resources.

Parajuli, R., M. T. Knudsen, M. Birkved, S. N. Djomo, A. Corona and T. Dalgaard (2017). "Environmental impacts of producing bioethanol and biobased lactic acid from standalone and integrated biorefineries using a consequential and an attributional life cycle assessment approach." Science of the Total Environment **598**: 497-512.

Pehnt, M., M. Oeser and D. J. Swider (2008). "Consequential environmental system analysis of expected offshore wind electricity production in Germany." Energy **33**(5): 747-759.

Pellegrini, L., M. Marchionna, R. Patrini, C. Beatrice, N. Del Giacomo and C. Guido (2012). Combustion Behaviour and Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in a Light-Duty Diesel Engine. SAE 2012 World Congress & Exhibition United States, SAE International https://saemobilus.sae.org/content/2012-01-1053.

Pellegrini, L., M. Marchionna, R. Patrini and S. Florio (2013). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car, SAE Technical Paper.

Pellegrini, L., M. Marchionna, R. Patrini and S. Florio (2013). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car. SAE 2013 World Congress & Exhibition, United States, SAE International https://saemobilus.sae.org/content/2013-01-1035 Accessed 05/05/2015.

Pellegrini, L., R. Patrini and M. Marchionna (2014). Effect of POMDME Blend on PAH Emissions and Particulate Size Distribution from an In-Use Light-Duty Diesel Engine, SAE Technical Paper.

Pellegrini, L., R. Patrini and M. Marchionna (2014). Emission Performance of Neat and Blended Polyoxymethylene Dimethyl Ethers in an Old Light-Duty Diesel Car. SAE 2013 World Congress & Exhibition, United States, SAE International.

Peters, M. S., K. D. Timmerhaus and R. E. West (2003). Plant Design and Economics for Chemical Engineers. New York, MCGraw-Hill.

Plevin, R. J. (2017). "Assessing the climate effects of biofuels using integrated assessment models, part I: Methodological considerations." Journal of Industrial Ecology **21**(6): 1478-1487.

Plevin, R. J., M. O'Hare, A. D. Jones, M. S. Torn and H. K. Gibbs (2010). "Greenhouse Gas Emissions from Biofuels' Indirect Land Use Change Are Uncertain but May Be Much Greater than Previously Estimated." Environmental Science & Technology **44**(21): 8015-8021.

Prapaspongsa, T. and S. H. Gheewala (2016). "Risks of indirect land use impacts and greenhouse gas consequences: an assessment of Thailand's bioethanol policy." Journal of Cleaner Production **134**: 563-573.

Prapaspongsa, T., C. Musikavong and S. H. Gheewala (2017). "Life cycle assessment of palm biodiesel production in Thailand: Impacts from modelling choices, co-product utilisation, improvement technologies, and land use change." Journal of Cleaner Production **153**: 435-447.

Preston, R. (2005). 2005 Feed Composition Tables. Beef. http://beefmagazine.com/mag/beef_feed_composition_tables/

Rahman, M. M., C. Canter and A. Kumar (2015). "Well-to-wheel life cycle assessment of transportation fuels derived from different North American conventional crudes." Applied Energy **156**: 159-173.

Ramdhani, A., M. A. Ramdhani and A. S. Amin (2014). "Writing a Literature Review Research Paper: A step-by-step approach." International Journal of Basic and Applied Science **3**(1): 47-56.

Rausch, K. D. and R. L. Belyea (2006). "The future of coproducts from corn processing." Applied Biochemistry and Biotechnology **128**(1): 47-86.

Rege, S., M. Arenz, A. Marvuglia, I. Vázquez-Rowe, E. Benetto, E. Igos and D. Koster (2015). "Quantification of Agricultural Land Use Changes in Consequential Life Cycle Assessment Using Mathematical Programming Models Following a Partial Equilibrium Approach." Journal of Environmental Informatics **26**(2).

Reinhard, J. and R. Zah (2009). "Global environmental consequences of increased biodiesel consumption in Switzerland: consequential life cycle assessment." Journal of Cleaner Production **17**: S46-S56.

Reinhard, J. and R. Zah (2011). "Consequential life cycle assessment of the environmental impacts of an increased rapemethylester (RME) production in Switzerland." Biomass and Bioenergy **35**(6): 2361-2373.

REN21 (2013). Renewables 2017 Global Status Report, Renewable Energy Policy Network for the 21st century (REN21).

REN21 (2017). Renewables 2017 Global Status Report. Paris, REN21 Secretariat.

Renouf, M. A., R. J. Pagan and M. K. Wegener (2013). "Bio-production from Australian sugarcane: an environmental investigation of product diversification in an agro-industry." Journal of Cleaner Production **39**: 87-96.

Richmond, B. (1999). "ASSESSMENT OF NET EMISSIONS OF GREENHOUSE GASES FROM ETHANOL-GASOLINE BLENDS IN SOUTHERN ONTARIO."

Richmond, B. (2000). "Assessment of Net Emissions of Greenhouse Gases from Ethanol-Gasoline Blends in Southern Ontario R-2000-1." Assessment.

Roningen, V. O. (1997). "Multi-market, multi-region partial equilibrium modeling." Applied methods for trade policy analysis: a handbook: 231-257.

Roos, A. and S. Ahlgren (2018). "Consequential life cycle assessment of bioenergy systems–A literature review." Journal of Cleaner Production **189**: 358-373.

Rose, S. K., E. Kriegler, R. Bibas, K. Calvin, A. Popp, D. P. van Vuuren and J. Weyant (2014). "Bioenergy in energy transformation and climate management." Climatic Change **123**(3-4): 477-493.

Ruhul Kabir, M. and A. Kumar (2012). "Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways." Bioresource technology **124**: 394-405.

Safari, H., M. S. Fagheyi, S. S. Ahangari and M. R. Fathi (2012). "Applying Promethee method based on entropy weight for supplier selection." Business management and strategy **3**(1): 97-106.

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan.

Saha, S. (2010). Pelleting and characterization of dry distillers' grain with solubles pellets as bio-fuel. Master of Science University of Saskatchewan

Sanchez, S. T., J. Woods, M. Akhurst, M. Brander, M. O'Hare, T. P. Dawson, R. Edwards, A. J. Liska and R. Malpas (2012). "Accounting for indirect land-use change in the life cycle assessment of biofuel supply chains." Journal of The Royal Society Interface **9**(71): 1105-1119.

Sarkar, S. and A. Kumar (2009). "Techno-economic assessment of biohydrogen production from forest biomass in Western Canada." Transactions of the ASABE **52**(2): 519-530.

Sarkar, S. and A. Kumar (2010). "Biohydrogen production from forest and agricultural residues for upgrading of bitumen from oil sands." Energy **35**(2): 582-591.

Sarkar, S. and A. Kumar (2010). "Large-scale biohydrogen production from bio-oil." Bioresource Technology **101**(19): 7350-7361.

Sawin, J. L., F. Sverrisson, K. Seyboth, R. Adib, H. E. Murdock, C. Lins, I. Edwards, M. Hullin, L. H. Nguyen and S. S. Prillianto (2013). "Renewables 2017 Global Status Report."

Sawin, J. L., F. Sverrisson, K. Seyboth, R. Adib, H. E. Murdock, C. Lins, I. Edwards, M. Hullin, L. H. Nguyen and S. S. Prillianto (2017). "Renewables 2017 Global Status Report."

Schmidt, J. H. (2008). "System delimitation in agricultural consequential LCA." The International Journal of Life Cycle Assessment **13**(4): 350.

Schröder, A. and K.-H. Südekum (1999). Glycerol as a by-product of biodiesel production in diets for ruminants. INTERNATIONAL RAPESEED CONGRESS, Regional Institute Gosford.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes and T.-H. Yu (2008). "Use of U.S. Croplands for Biofuels Increases Greenhouse Gases Through Emissions from Land-Use Change." Science **319**(5867): 1238-1240.

Searchinger, T., R. Heimlich, R. A. Houghton, F. Dong, A. Elobeid, J. Fabiosa, S. Tokgoz, D. Hayes and T.-H. Yu (2008). "Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change." Science **319**(5867): 1238-1240.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2015). "Net energy ratio for the production of steam pretreated biomass-based pellets." Biomass and bioenergy **80**(0): 286-297.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2016). "Comparative net energy ratio analysis of pellet produced from steam pretreated biomass from agricultural residues and energy crops." Biomass and bioenergy **90**: 50-59.

Shahrukh, H., A. O. Oyedun, A. Kumar, B. Ghiasi, L. Kumar and S. Sokhansanj (2016). "Techno-economic assessment of pellets produced from steam pretreated biomass feedstock." Biomass and Bioenergy **87**: 131-143.

Shroyer, K. J., S. A. Staggenborg and J. Propheter (2011). "Utilization of dry distillers grains and charcoal as nitrogen fertilizer in corn." Agronomy Journal **103**(5): 1321-1328.

Shurson, G. C. (2017). "The Role of Biofuels Coproducts in Feeding the World Sustainably." Annual Review of Animal Biosciences **5**(1): 229-254.

Shurson GC, Zijlstra RT, Kerr BJ and S. HH. (2012). Feeding biofuels co-products to pigs. Biofuel Coproducts as Livestock Feed: Opportunities and Challenges,. Makkar HPS. Rome:, Food Agric. Organ.**:** 175-207.

Silalertruksa, T., S. H. Gheewala and M. Sagisaka (2009). "Impacts of Thai bio-ethanol policy target on land use and greenhouse gas emissions." Applied Energy **86**: S170-S177.

Singh, S. and A. Kumar (2011). "Development of water requirement factors for biomass conversion pathway." Bioresource Technology **102**(2): 1316-1328.

Singh, S., A. Kumar and S. Jain (2014). "Impact of biofuel production on water demand in Alberta." Canadian Biosystems Engineering **56**.

Sissine, F. (2010). Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis, Technical Report EPA-420-R-10-006. Assessment and Standards Division, Office ….

Smith, D. M., B. C. Larson, M. J. Kelty and P. M. S. Ashton (1997). The practice of silviculture: applied forest ecology, John Wiley and Sons, Inc.

Sokhansanj, S., S. Mani, M. Stumborg, R. Samson and J. Fenton (2006). "Production and distribution of cereal straw on the Canadian prairies." Canadian Biosystems Engineering **48**: 3.

Spath, P., A. Aden, T. Eggeman, M. Ringer, B. Wallace and J. Jechura (2005). "Biomass to Hydrogen Production Detailed Design and Economics Utilizing the Battelle Columbus Heated Gasifier." Golden, Colorado, United States: National Renewable Energy Laboratory (NREL).

Spath, P. L. and M. K. Mann (2000). Life cycle assessment of hydrogen production via natural gas steam reforming, National Renewable Energy Laboratory Golden, CO.

Staples, M. D., R. Malina and S. R. H. Barrett (2017). "The limits of bioenergy for mitigating global life-cycle greenhouse gas emissions from fossil fuels." Nature Energy **2**: 16202.

Stasinopoulos, P., P. Compston, B. Newell and H. M. Jones (2012). "A system dynamics approach in LCA to account for temporal effects—a consequential energy LCI of car body-in-whites." The International Journal of Life Cycle Assessment **17**(2): 199-207.

Stripple, H. (2001). Life Cycle Assesment of Road—A Pilot Study for Inventory Analysis—2nd Revised Edition, B1210E.

Styles, D., E. M. Dominguez and D. Chadwick (2016). "Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment." Science of the Total Environment **560**: 241-253.

Styles, D., J. Gibbons, A. P. Williams, J. Dauber, H. Stichnothe, B. Urban, D. R. Chadwick and D. L. Jones (2015). "Consequential life cycle assessment of biogas, biofuel and biomass energy options within an arable crop rotation." Gcb Bioenergy **7**(6): 1305-1320.

Sultana, A. and A. Kumar (2011). "Development of energy and emission parameters for densified form of lignocellulosic biomass." Energy **36**(5): 2716-2732.

Sultana, A. and A. Kumar (2012). "Ranking of biomass pellets by integration of economic, environmental and technical factors." Biomass and Bioenergy **39**: 344-355.

Sultana, A., A. Kumar and D. Harfield (2010). "Development of agri-pellet production cost and optimum size." Bioresource Technology **101**(14): 5609-5621.

Swanson, R. M., A. Platon, J. A. Satrio, R. C. Brown and D. D. Hsu (2010). Techno-Economic Analysis of Biofuels Production Based on Gasification. Golden, Colorado, National Renewable Energy Laboratory**:** 165.

Tago, T. and T. Masuda (2018). "Utilization of DDGS using ethanol solution for biocrude oil production by hydrothermal liquefaction AU - Mansur, Dieni." Biofuels **9**(3): 325-330.

Thacker, P. (2006). "Nutrient digestibility, performance and carcass traits of growing-finishing pigs fed diets containing dried wheat distillers grains with solubles." Canadian journal of animal science **86**(4): 527-529.

Thacker, P. A. (2006). "Nutrient digestibility, performance and carcass traits of growing-finishing pigs fed diets containing dried wheat distillers grains with solubles." Canadian journal of animal science **86**(4): 527-529.

Thakur, A., C. E. Canter and A. Kumar (2014). "Life-cycle energy and emission analysis of power generation from forest biomass." Applied Energy **128**: 246-253.

Thomassen, M. A., R. Dalgaard, R. Heijungs and I. de Boer (2008). "Attributional and consequential LCA of milk production." The International Journal of Life Cycle Assessment **13**(4): 339-349.

Tonini, D., L. Hamelin, M. Alvarado-Morales and T. F. Astrup (2016). "GHG emission factors for bioelectricity, biomethane, and bioethanol quantified for 24 biomass substrates with consequential life-cycle assessment." Bioresource Technology **208**: 123-133.

Tonini, D., L. Hamelin and T. F. Astrup (2016). "Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes." Gcb Bioenergy **8**(4): 690-706.

Tonini, D., C. Vadenbo and T. F. Astrup (2017). "Priority of domestic biomass resources for energy: Importance of national environmental targets in a climate perspective." Energy **124**: 295-309.

U.S. Department of Energy. (2011). "Biomass Program " Retrieved March 25, 2018, from https://www.nrel.gov/docs/fy11osti/51241.pdf.

U.S. Department of Energy. (2018). "Alternative Fuels and Advanced Vehicles." Retrieved April 1, 2018, from https://www.afdc.energy.gov/fuels/.

U.S. Environmental Protection Agency (2014). Climate Change Indicators in the United States: Global Greenhouse Gas Emissionshttps://www3.epa.gov/climatechange/pdfs/print_global-ghg-emissions-2014.pdf. Accessed 17 March 2018.

UNEP (2018). The Emissions Gap Report Nairobihttp://wedocs.unep.org/bitstream/handle/20.500.11822/26895/EGR2018_FullReport_EN.pdf?sequence=1&isAllowed=y Accessed 05/05/2019.

United Nation Climate Change. (2019a). "About the Secretariat." Retrieved 01/05/2019, from https://unfccc.int/about-us/about-the-secretariat.

United Nations Climate Change. (2019b). "Kyoto Protocol - Targets for the first commitment period." Retrieved 05/05/2019, from https://unfccc.int/process/the-kyoto-protocol.

United Nations Climate Change. (2018). "The Paris Agreement." Retrieved 05/05/2018, from https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement.

USDA (2016). Feed Grains Database, United States Department of Agriculture (USDA)http://www.ers.usda.gov/data-products/feed-grains-database.aspx.

Valente, C., R. Spinelli and B. G. Hillring (2011). "LCA of environmental and socio-economic impacts related to wood energy production in alpine conditions: Valle di Fiemme (Italy)." Journal of Cleaner Production **19**(17): 1931-1938.

Van Stappen, F., M. Mathot, V. Decruyenaere, A. Loriers, A. Delcour, V. Planchon, J.-P. Goffart and D. Stilmant (2016). "Consequential environmental life cycle assessment of a farm-scale biogas plant." Journal of environmental management **175**: 20-32.

Van Vliet, O. P., A. P. Faaij and W. C. Turkenburg (2009). "Fischer–Tropsch diesel production in a well-to-wheel perspective: a carbon, energy flow and cost analysis." Energy Conversion and Management **50**(4): 855-876.

Van Vuuren, D. P., H. van Soest, K. Riahi, L. Clarke, V. Krey, E. Kriegler, J. Rogelj, M. Schaeffer and M. Tavoni (2016). "Carbon budgets and energy transition pathways." Environmental Research Letters **11**(7): 075002.

Vázquez-Rowe, I., A. Marvuglia, S. Rege and E. Benetto (2014). "Applying consequential LCA to support energy policy: land use change effects of bioenergy production." Science of the Total Environment **472**: 78-89.

Vázquez-Rowe, I., S. Rege, A. Marvuglia, J. Thénie, A. Haurie and E. Benetto (2013). "Application of three independent consequential LCA approaches to the agricultural sector in Luxembourg." The International Journal of Life Cycle Assessment **18**(8): 1593-1604.

Visual-PROMETHEE (2015). Multicriteria Decision Aid - Methods, Modeling and Software. Belgium, Solvay Brussels School of Economics & Managementhttp://www.promethee-gaia.net/index.html.

Wang, M., H. Huo and S. Arora (2011). "Methods of dealing with co-products of biofuels in life-cycle analysis and consequent results within the U.S. context." Energy Policy **39**(10): 5726-5736.

Wang, M., H. Huo and S. Arora (2011). "Methods of dealing with co-products of biofuels in life-cycle analysis and consequent results within the US context." Energy Policy **39**(10): 5726-5736.

Weidema, B. P. (1993). "Market aspects in product life cycle inventory methodology." Journal of Cleaner Production **1**(3-4): 161-166.

Weidema, B. P. (1999). System expansions to handle co-products of renewable materials. Presentation summaries of the 7th LCA case studies symposium SETAC-Europe, Citeseer.

Weidema, B. P. (2003). Market information in life cycle assessment, Miljøstyrelsen.

Weidema, B. P., T. Ekvall and R. Heijungs (2009). "Guidelines for application of deepened and broadened LCA." Deliverable D18 of work package **5**: 17.

Weidema, B. P., N. Frees and A.-M. Nielsen (1999). "Marginal production technologies for life cycle inventories." The International Journal of Life Cycle Assessment **4**(1): 48-56.

West, G. R. (1995). "Comparison of input–output, input–output+ econometric and computable general equilibrium impact models at the regional level." Economic Systems Research **7**(2): 209-227.

Winkler, N. (1998). "A manual for the planning, design and construction of forest roads in steep terrain." Food and Agriculture Organization of the United Nations, Forest Harvesting Case Study **10**.

Witzke, P., M. Banse, H. Gömann, T. Heckelei, T. Breuer, S. Mann, M. Kempen, M. Adenäuer and A. Zintl (2008). "Modeling of Energy-Crops in Agricultural Sector Models–A Review of Existing Methodologies." JRC Scientific and Technical Reports.

Wong, A., H. Zhang and A. Kumar (2016). "Life cycle water footprint of hydrogenation-derived renewable diesel production from lignocellulosic biomass." Water research **102**: 330-345.

Wong, A. J. L. (2015). Life Cycle Assessment of Lignocellulosic Biomass Conversion Pathways to Hydrogenation Derived Renewable Diesel, University of Alberta.

World Energy Council (2016). World Energy Resources Bioenergy | 2016https://www.worldenergy.org/wp-content/uploads/2017/03/WEResources_Bioenergy_2016.pdf Accessed 10/04/2018.

Xue Lia, E. M., Satya Panigrahib, Lope Tabilb, Shahab Sokhansanj, Mark Stumborg (2012). "A review of agricultural crop residu esupply in Canada for cellulosic ethanol production." Renewable and Sustainable Energy Reviews **16**: 2954–2965.

Yang, J., M. Xu, X. Zhang, Q. Hu, M. Sommerfeld and Y. Chen (2011). "Life-cycle analysis on biodiesel production from microalgae: water footprint and nutrients balance." Bioresource Technology **102**(1): 159-165.

Yang, Y. (2016). "Two sides of the same coin: consequential life cycle assessment based on the attributional framework." Journal of Cleaner Production **127**: 274-281.

Yang, Y. and R. Heijungs (2018). "On the use of different models for consequential life cycle assessment." The International Journal of Life Cycle Assessment **23**(4): 751-758.

Yizhen, Z. (2018). Life Cycle Environmental Impacts of Biofuels: The Role of Co-products. Doctor of Philosophy, University of California.

Zaimes, G. G. and V. Khanna (2014). "The role of allocation and coproducts in environmental evaluation of microalgal biofuels: How important?" Sustainable Energy Technologies and Assessments **7**: 247-256.

Zhang, H. and K. R. Haapala (2014). "Integrating sustainable manufacturing assessment into decision making for a production work cell." Journal of Cleaner Production.

Zhang, H. and K. R. Haapala (2014). "Integrating sustainable manufacturing assessment into decision making for a production work cell." Journal of Cleaner Production **105**: 52-63.

Zhang, X., A. Kumar, U. Arnold and J. Sauer (2014). Biomass-derived oxymethylene ethers (OMEs) as diesel additives: A thermodynamic analysis. The 6th International Conference on Applied Energy, Taipei, Taiwan.

Zhang, X., A. Kumar, U. Arnold and J. Sauer (2014). "Biomass-derived Oxymethylene Ethers as Diesel Additives: A Thermodynamic Analysis." Energy Procedia **61**: 1921-1924.

Zhang, X., A. O. Oyedun, A. Kumar, D. Oestreich, U. Arnold and J. Sauer (2016). "An optimized process design for oxymethylene ether production from woody-biomass-derived syngas." Biomass and Bioenergy **90**: 7-14.

Zhang, Y. (2017). Life Cycle Environmental Impacts of Biofuels:The Role of Co-products. Doctor of Philosophy University of California.

Appendix A

Road Construction Estimation

We estimated road construction distances for this study based on a discussion with Fulton Smyl (Business Analyst, Alberta Innovates-Technology Futures, 2016 on June 28, 2016) on Alberta's forest management plans, roads classification, and design specifications. Three major road types are considered for construction: primary, secondary, and tertiary roads. In this study, the OME plant is assumed to be located at the center of a circular biomass harvest area. The harvest area is assumed to be divided into eight equal sections hence eight primary roads need to be constructed. The construction estimate for primary roads is given below.

$$\text{The total primary road length} = r \cdot 8 \tag{1}$$

Here, r = radius of the circular biomass harvest area= biomass collection distance including effects of tortuosity factor and geometric factor

Each sector is then divided into two equal areas to determine the length of the secondary roads. The secondary road lengths are obtained by using the arc length formula given below:

$$\text{Arc length} = \theta \cdot \frac{\pi}{180} \cdot r \tag{2}$$

where r = radius of the circular biomass harvest area and $\theta = 360°/8 = 45°$

$$\text{Secondary road length, } S_1 = \theta \cdot \frac{\pi}{180} \cdot r \tag{3}$$

$$\text{Secondary road length, } S_2 = \theta \cdot \frac{\pi}{180} \cdot (r/2) \tag{4}$$

$$\text{Total length for secondary road construction} = (S_1+S_2) \cdot 8 \tag{5}$$

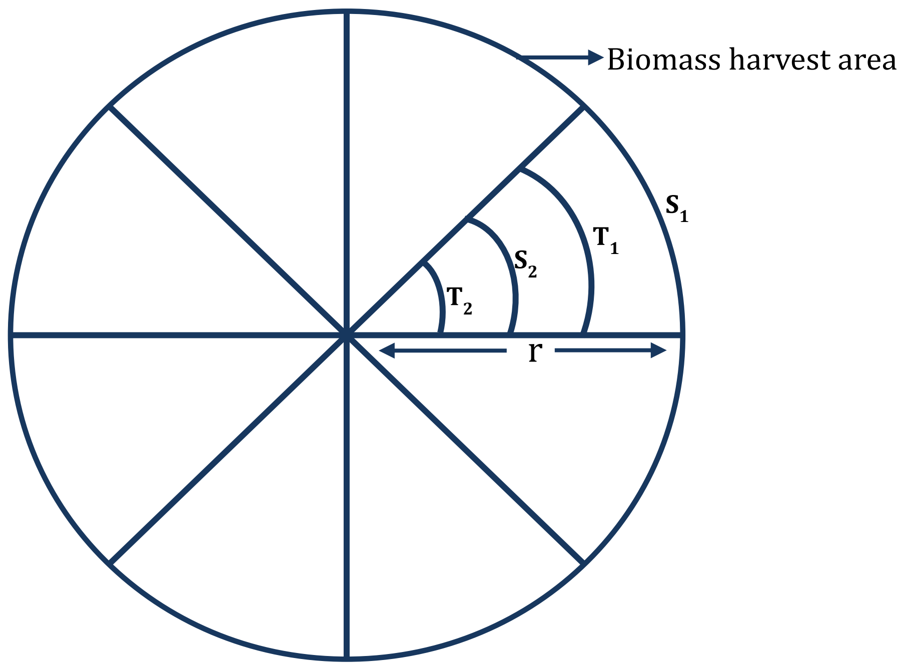

Figure A1: Primary road construction distance from the biomass harvest area

Each secondary road area is further divided equally into two areas to calculate the tertiary road lengths.

Tertiary road length, $T_1 = \theta \cdot \frac{\pi}{180} \cdot (r \cdot 3/4)$ (6)

Tertiary road length, $T_2 = \theta \cdot \frac{\pi}{180} \cdot (r/4)$ (7)

Total length for tertiary road construction = $(T_1+T_2) \cdot 8$ (8)

Here, r = radius of the circular biomass harvest area

S_1 and S_2 = secondary road lengths

T_1 and T_2 = tertiary road length

Appendix B

Table B1. Specifications of equipment used in the whole tree and forest residue pathways

Equipment Specification	Units	Whole Tree	Forest Residue	References
Type		Feller	Forwarder	(Kabir and Kumar 2011)
Equipment name	Model	John Deere 853J	Komatsu WA 250-6	
Lifetime productivity	dry tonne of biomass	95812.5	101200	
Lifetime fuel consumption	L diesel	514650	416000	
Lifetime	hours	10950	16000	
Dedicated equipment required	N/A	18	17	
Equipment weight	tonne	29.427	11.82	
Steel in each equipment	tonne	28.84	11.58	
Type		Chipper	Chipper	(Kabir and

Equipment Specification	Units	Whole Tree	Forest Residue	References
Equipment name	Model	Morbark 50/48	Nicholson WFP 3A	Kumar 2011)
Lifetime productivity	dry tonne of biomass	270000	252000	
Lifetime fuel consumption	L diesel	900000	990000	
Lifetime	hours	9000	9000	
Dedicated equipment required	N/A	6	7	
Equipment weight	tonne	28.74	59	
Steel in each equipment	tonne	28.16	57.82	
Type		Skidder	N/A	(Kabir and Kumar 2011)
Equipment name	Model	John Deere 748 H	N/A	
Lifetime productivity	dry tonne	90000	N/A	

Equipment Specification	Units	Whole Tree	Forest Residue	References
	of biomass			
Lifetime fuel consumption	L diesel	540000	N/A	
Lifetime	hours	12000	N/A	
Dedicated equipment required	N/A	19	N/A	
Equipment weight	tonne	14.64	N/A	
Steel in each equipment	tonne	14.35	N/A	

Table B2. Data inventory for raw materials, fuels, plant construction, and road construction

Impact Factors	Data Inventory	Units	Values	References
Steel energy & emissions factors	Energy required to produce 1 tonne of steel	GJ / tonne	34	(Kabir and Kumar 2011)
	CO_2eq emissions per tonne of steel	kg CO_2eq /tonne	2495.22	(Kabir and Kumar 2011)
Aluminum energy & emissions factors	Energy required to produce 1 tonne of steel	GJ / tonne	39.15	(Kabir and Kumar 2011)
	CO_2eq emissions per tonne of steel	kg CO_2eq /tonne	3467	(Kabir and Kumar 2011)
Concrete energy & emissions factors	Energy required to produce 1 tonne of steel	GJ / tonne	0.863	(Kabir and Kumar 2011)
	CO_2eq emissions per tonne of steel	kg CO_2eq /tonne	120	(Kabir and Kumar 2011)
Diesel	Lower heating value	MJ/L	35.98	(Pellegrini et al.

energy & emissions factors				2013)
	CO_2eq emissions per liter of diesel	kg CO_2eq /L	3.61	(Kabir and Kumar 2011)
	Energy required to produce 1 liter of diesel	GJ / L	0.046	(Kabir and Kumar 2011)
Plant construction material quantity	Steel	tonnes	2618	calculated
	Aluminum	tonnes	8092	calculated
	Concrete	tonnes	22	calculated
Primary road construction energy & emissions factors	Energy required to construct 1 km road	GJ /km	1731	(Stripple 2001, Kabir and Kumar 2012)
	CO_2eq emissions from construction of 1 km road	kg CO_2eq /km	403845	
Secondary road construction equipment	Crawler tractor operating hours	hours/km	70	(Winkler 1998)
	Dedicated tractor required	N/A	0.71	calculated

	Tractor fuel consumption	L/hour	23	(Winkler 1998)
Tertiary road construction equipment	Crawler tractor operating hours	hours/km	100	(Winkler 1998, Kabir and Kumar 2012)
	Dedicated tractor required	N/A	0.68	calculated
	Tractor fuel consumption	L/hour	23	(Winkler 1998)

Water use factor calculation

Wong et al. (2016) considered the average annual precipitation (rainfall) as the water source for the forest biomass growth in the Western province of Canada, Alberta. They assumed the overland flow from the Western forests small enough to be neglected and considered the amount of average annual precipitation to be roughly same as the amount of evapotranspiration. They calculated the water use factor for the forest biomass growth by using the equation given below;

$$\text{Water use factor} = \frac{\text{average annual rainfall (mm/year) x time to harvest forest biomass (years) x \% allocation}}{\text{biomass yield (dry kg/ hectare)}}$$

(B1)

Here, water use factor= water required for biomass growth (L H_2O/ kg dry wood)

% allocation= 100% allocation for whole tree and 20% allocation for forest residue (since forest residue yield is assumed to be 20% of whole tree harvest (Kumar et.al 2003; Mahbub et al. 2017)

time to harvest forest biomass = a100 year of rotation is required for whole tree harvest whereas forest residues are harvested every year (Mahbub et al. 2017)

biomass yield= amount of biomass growth per year with the average annual rainfall

PROMETHEE outranking method

The PROMETHEE outranking method has been used in this study to compare different alternatives or pathways. In this method, first all criteria (indicators) are assigned weights that have been decided by the decision-maker, then a preference index is calculated for all the pathways considering all the criteria and the pathways are ranked. In this study, the best pathway was selected based on the net outranking score. This method relies on the assumption that the higher the score the better the performance of the pathway (the more sustainable the pathway) (Fülöp 2005).

This study develops a base case scenario to compare two different OME pathways based on nine different sustainability criteria (indicators).

Step 1: Preference Function

The two pathways are compared in terms of each criterion (indicator) and the differences in estimates on a specific indicator are converted to a degree of preference from 0 to 1, 0 being not preferred at all to 1 (strictly preferred) by using a preference function.

The preference function of the whole tree pathway (WT) over the forest residue pathway (FR) on a particular criterion i is given by (Fülöp 2005) as

$0 \leq P_i(WT,FR) \leq 1$

where

$P_i(WT,FR)=0$; indicates incomparability between two pathways,

$P_i(WT,FR)\approx 0$; indicates weak preference of WT over FR,

$P_i(WT,FR)\approx 1$; indicates strong preference of WT over FR,

$P_i(WT,FR)=1$; indicates strict preference of WT over FR.

The preference function can thus be defined as the difference between the evaluations of the two pathways on a particular indicator (Fülöp 2005). Thus the preference function of the whole tree pathway (WT) over the forest residue pathway (FR) on a particular criteria i is given by (Fülöp 2005) as

$P_i(WT,FR)=p_i\,(WT_i\text{-}FR_i)$

where p_i is a non-decreasing function and $p_i\,(WT_i\text{-}FR_i)\ =0$ when $(WT_i\text{-}FR_i)\leq 0$ and $0 \leq p_i\,(WT_i\text{-}FR_i)\leq 1$ when $(WT_i\text{-}FR_i)>0$.

Usual and linear preference functions are used in this study. For a usual preference function, incomparability occurs only when the difference between the evaluations of the two pathways on a specific indicator is 0. When the deviation is different from 0 the pathway with the higher value is strictly preferred over the lower valued one (Brans and Vincke 1985).

Usual Preference Function: $p_i\left(WT_i\text{-}FR_i\right)=\begin{cases}0, p_i\,(WT_i\text{-}FR_i)\leq 0\\ 1, p_i\,(WT_i\text{-}FR_i)>0\end{cases}$

Linear preference functions require indifference (Q) and preference (P) thresholds to make the preference decision more realistic. In linear preference, indifference occurs until the deviation between evaluations exceed the indifference threshold, and above this the threshold preference increases progressively until the deviation equals the sum of the two thresholds (Brans and Vincke 1985).

Linear Preference Function: $p_i\left(WT_i\text{-}FR_i\right)=\begin{cases}0,\ p_i\left(WT_i\text{-}FR_i\right)\leq Q\\ (\left(p_i\left(WT_i\text{-}FR_i\right)\right)\text{-}Q)/P,\ Q\leq p_i\left(WT_i\text{-}FR_i\right)\leq P\\ 1, p_i\left(WT_i\text{-}FR_i\right)\geq Q+P\end{cases}$

Step 2: Multi-criteria Preference Index

A multi-criteria preference index compares a pair of alternatives over all criteria. Preference index $\pi(WT, FR)$ for WT over FR, taking into account nine criteria (indicators), is defined as

$$\pi(WT,FR)=\sum_{i=1}^{m} w_i\ P_i(WT,FR)$$

where $w_i>0$ and w_i is normalized weight assigned to criteria i and m=9.

The preference index is a value again between 0 and 1 to demonstrate the preference of WT over FR considering all the weighted criteria (indicators).

For example,

$\pi(WT,FR)=0$ indicates the sum of all the $P_i(WT,FR)$ values equal 0. WT is never preferred over FR for any criteria.

$\pi(WT,FR)\approx 0$ indicates a weak preference of WT over FR.

$\pi(WT,FR)\approx 1$ indicates a strong preference of WT over FR.

$\pi(WT,FR)=1$ indicates the sum of all the $P_i(WT,FR)$ values equal 1. WT is strictly preferred over FR for all criteria. The boundary conditions to calculate the preference indices are

$\pi(WT,WT)=0$

$0\leq\pi(WT,FR)\leq 1$

$0\leq\pi(WT,FR)+\pi(FR,WT)\leq 1$

Step 3: Partial and Complete Ranking of Alternatives

Two outranking flows are used to rank the alternative pathways, i.e., positive outranking flow and negative outranking flow. The positive and negative outranking flows for WT over FR are given by equations C1 and C2 (Fülöp 2005).

$$\emptyset^{+}(WT)=\frac{1}{n-1}\sum_{k=1}^{n}\pi(WT,FR) \quad \text{(C1)}$$

Here n stands for the number of pathways. In this study n=2. The positive outranking flow $\emptyset^{+}(WT)$ determines how much the WT pathway outranks the other pathway. The larger the value of $\emptyset^{+}(WT)$, the stronger the pathway. The negative outranking flow is given as

$$\emptyset^{-}(WT)=\frac{1}{n-1}\sum_{k=1}^{n}\pi(WT,FR) \quad \text{(C2)}$$

The negative outranking flow $\emptyset^{-}(\mathrm{WT})$ determines how much the WT pathway is outranked by the other pathway. The lower the value of $\emptyset^{+}(\mathrm{WT})$, the stronger the pathway.

Both the PROMETHEE I partial ranking and the PROMETHEE II complete ranking were used to rank the alternatives. According to the PROMETHEE I partial ranking, WT is preferred over the FR pathway if $\emptyset^{+}(\mathrm{WT}) \geq \emptyset^{+}(\mathrm{FR})$, $\emptyset^{-}(\mathrm{WT}) \leq \emptyset^{-}(\mathrm{FR})$ and one of them is a strict inequality. The WT and FR pathways are indifferent if $\emptyset^{+}(\mathrm{WT}) = \emptyset^{+}(\mathrm{FR})$ and $\emptyset^{-}(\mathrm{WT}) = \emptyset^{-}(\mathrm{FR})$. Otherwise, the WT and FR pathways are incomparable (Fülöp 2005).

According to the PROMETHEE II complete ranking, the net outranking flows for the whole tree pathway $\emptyset(\mathrm{WT})$ and the forest residue pathway $\emptyset(\mathrm{FR})$ given by equations C3 and C4 determine the preference of one pathway over the other (Fülöp 2005).

$$\emptyset(\mathrm{WT})=\emptyset^{+}(\mathrm{WT})-\emptyset^{-}(\mathrm{WT}) \tag{C3}$$

$$\emptyset(\mathrm{FR})=\emptyset^{+}(\mathrm{FR})-\emptyset^{-}(\mathrm{FR}) \tag{C4}$$

If $\emptyset(\mathrm{WT}) > \emptyset(\mathrm{FR})$, WT is preferred over FR. The pathways are indifferent if $\emptyset(\mathrm{WT}) = \emptyset(\mathrm{FR})$. The pathway with the largest net outranking flow value ($\emptyset$) is considered to be the best sustainable pathway over the others.

Table B3: Social impacts assessments for whole tree and forest residue pathway

Unit operations	Path ways	Volume of wood chips involved in the unit operation for 20 years	Total time involved in an unit operation in 20 years	Employment potential±	Biomass involved	Labor cost€
		m^3	hours	hours/m^3	dry tonne/hour	$/dry tonne
Biomass Growth	WT	20320944	876000	0.043	3.2	8.17
	FR	16990738.8	876000	0.05	3.2	8.17
Biomass Harvest	WT	20320944	786666.6667	0.038	3.56	7.33
	FR	16990738.8	542687.2043	0.03	5.16	5.06
Biomass Transportation	WT	20320944	672656.0203	0.03	4.16	6.27
	FR	16990738.8	2294751.977	0.14	1.22	21.4

Unit operations	Path ways	Volume of wood chips involved in the unit operation for 20 years	Total time involved in an unit operation in 20 years	Employment potential±	Biomass involved	Labor cost€
		m^3	hours	hours/m^3	dry tonne/hour	$/dry tonne
Chemical Conversion	WT	20320944	180979.6697	0.01	N/A	34.91
	FR	16990738.8	223258.9455	0.01	N/A	34.91
OME Transportation	WT	20320944	632206.16	0.03	4.43	5.89
	FR	16990738.8	632206.16	0.04	4.43	5.89

±The employment potential for a particular unit operation was assessed by dividing a ratio of operation time by the biomass volume (m^3) involved in the operation.

€For the harvesting and transportation operations, employee wages were calculated based on hours of operation and hourly labor rates. The wages and benefits for the harvesting and transportation operations are estimated to be $26.11/hour (Canada-Visa 2014), equivalent to the required skill level of the job.