Life Cycle Greenhouse Gas Emissions from Alternative Jet Fuels

PARTNER Project 28 report
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Partnership for Air Transportation Noise and Emissions Reduction Project 28

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The Partnership for AiR Transportation Noise and Emissions Reduction — PARTNER — is a cooperative aviation research organization, and an FAA/NASA/Transport Canada-sponsored Center of Excellence. PARTNER fosters breakthrough technological, operational, policy, and workforce advances for the betterment of mobility, economy, national security, and the environment. The organization's operational headquarters is at the Massachusetts Institute of Technology.

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Executive Summary

The long-term viability and success of a transportation fuel depends on both economic and environmental sustainability. These include, but are not limited to, the environmental impacts on global climate and air quality, the efficient usage of water and land resources, technical feasibility and the economic cost of fuel production. This report focuses on aspects of environmental sustainability, with an emphasis on life cycle greenhouse gas emissions as they relate to impacts on global climate. Through a life cycle accounting of the GHG emissions starting with the well, field, or mine where the fuel feedstock is extracted, and extending to the wake behind the aircraft, one can ascertain the change in GHG emissions that result from the use of an alternative fuel.

The focus of this manuscript is a comparison of the well-to-wake, life cycle GHG emissions from select feedstock-to-jet fuel pathways for the United States. Select fuel pathways have been considered using a consistent methodology to facilitate equitable comparisons. The emphasis throughout is on maximizing the transparency of assumptions and establishing a fundamental understanding for the reader of the pivotal factors defining fuel production for each feedstock and how these relate to life cycle analysis. As there is considerable variability in the life cycle GHG emissions from existing fuel production and many of the fuel pathways considered in this report have not been commercialized, a range of life cycle GHG emissions has been provided for each feedstock-to-fuel pathway. In addition to the examination of life cycle GHG emissions, the manuscript provides a first order examination of the land and water usage that could accompany the development of a biofuel industry for aviation.

The fuel options considered herein are "drop-in" alternatives in that they have the potential to serve as a direct replacement for conventional jet fuel, requiring little or no modification to existing infrastructure or aircraft. The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) framework (versions 1.8b and 1.8a) and its supporting data were the primary tool used in the well-to-wake life cycle GHG analysis. The GREET model was developed for ground transportation; hence, this analysis involved modifying the underlying data and framework of the model to reflect jet fuel production. In cases where a fuel production pathway was not preexisting within the GREET framework, the relevant process inputs and production characteristics were developed from the open literature. All results are presented in terms of GHG emissions (CO_2, CH_4 and N_2O) per unit of energy (lower heating value) consumed by the aircraft.

The fuel pathways considered herein span petroleum based jet fuel from conventional and unconventional sources (oil and sands and oil shale), Fisher-Tropsch jet fuel from natural gas, coal and biomass and hydroprocessed jet fuels from soy oil, palm oil, rapeseed oil, jatropha oil, algae oil, and salicornia oil. In most fuel pathways, the choice of allocation methodology and potential for GHG emissions from land use change were found to have the largest impact on the results. The scope of this work was limited to only quantifying the impacts of direct land use change; emissions from indirect land use change were not considered. Many fuel pathways were found to result in life cycle GHG emissions that are either lower or higher than conventional jet fuel depending on the specifics of fuel production. For this reason, it is essential not to simply assume that biofuels are environmentally beneficial without knowing the specifics of how the fuel is produced.

A few of the key results are outlined below:

- *Life cycle GHG emissions are but one of many considerations when evaluating the feasibility and sustainability of an alternative fuel option.*

- *The data do not include all of the feedstock-to-fuel pathways that could be use to create jet fuel. Some interesting options not covered include camelina oil to jet fuel, fuels created from pyrolysis oils and advanced fermentation of sugars to*

hydrocarbons. These will be addressed as part of the ongoing work and will appear in future revisions to this report.

o *Of the fuel options considered herein, conventional petroleum has the lowest emissions of any jet fuel pathway that that relies exclusively on fossil fuel resources.*

o *Few biofuels have zero life cycle GHG emissions.*

o *There is considerable variability in the life cycle GHG emissions; emissions from land use change contribute the most to this for the biofuel pathways considered.*

o *Water availability could be a limiting factor for biofuel production in certain regions of the US.*

o *Biofuel expansion within the US would require a significant increase in crop and feedstock production.*

o *The possibility that non-indigenous species will be introduced into an unprepared ecosystem should not be ignored.*

o *The most significant challenge in developing viable alternative fuels that could reduce aviation's GHG emissions lies in developing and commercializing large-scale production of next generation biomass feedstocks that could be grown in a sustainable manner.*

As part of the ongoing research that is presented in this report, a more complete assessment of land use change emissions that includes indirect effects is being developed. Proper evaluation of the indirect effects of alternative fuels within aviation requires modeling of the demand for renewable energy resources within the transportation sector, including aviation, as well as the demand for renewable energy resources from the energy sector as a whole. Most indirect effects are expected to occur on an international scale; hence, domestic analyses, such as those in this work, should be done in the context of the global market.

Aviation is not the only potential user of renewable biomass resources, and it will have to compete for these limited resources. Furthermore, large land area requirements indicate that it is unlikely that a single region could create sufficient biomass to meet worldwide demand for biofuels. Hence, it is probable that large-scale implementation of biofuels would arise as a superposition of regionally appropriate feedstocks. Current actions with regard to biofuel expansion are important in realizing the potential of this industry.

Biomass feedstocks do need to have the potential to displace large volumes of petroleum fuel to have considerable benefit. Any feedstock that is produced in a sustainable manner today could not only provide a benefit to those involved in the supply chain of fuel production and use, but it would also lead to valuable experience to the biofuels industry providing essential lessons in production and processing techniques. This experience should prove invaluable to the development of future sustainable feedstocks that could be used to create transportation fuels, such as jet fuel, with a minimum of arable land and fresh water.

Acknowledgements

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Units, Notation, Abbreviations and Acronyms

Units

Bpd	Barrels per day	LHV	Lower Heating Value
Btu	British Thermal Unit	Mg	Megagram (metric tonne)
Bu	Bushel	MJ	Megajoule
Ha	Hectare (10000 m^2)	mmBtu	Million British Thermal Units
HHV	Higher Heating Value	Ton	Imperial Ton (2000 pounds)
kWh	Kilowatt-hour	Tonne	Metric Ton (megagram)

Chemical Species Notation

C_n	Hydrocarbon chain of length 'n'	K_2O	Potassium Oxide
CH_4	Methane	N_2O	Nitrous Oxide
CO	Carbon Monoxide	NH_3	Ammonia
CO_2	Carbon Dioxide	NO_x	Nitrogen Oxide
CO_2e	Carbon Dioxide equivalent	P_2O_5	Phosphorous Pentoxide
H_2	Hydrogen Gas	SO_x	Sulfur Oxides

Acronyms and Abbreviations

AFLCAWG	Air Force Life Cycle Analysis Working Group	IPCC	Intergovernmental Panel on Climate Change
API	American Petroleum Institute	ISO	International Organization for Standardization
ASP	Aquatic Species Program	LCA	Life Cycle Analysis
BTL	Biomass to Liquids (via F-T)	LPG	Liquefied Petroleum Gas
CBTL	Coal and Biomass to Liquids (via F-T)	LUC	Land Use Change
CCS	Carbon Capture and Sequestration	MEA	Monoethanolamine
CRP	Conservation Reserve Program	NETL	National Energy Technology Laboratory
CTL	Coal to Liquids (via F-T)	NREL	National Renewable Energy Laboratory
DME	Dimethyl Ether	PARTNER	Partnership for Air Transportation Noise and Emissions Reduction
DOE	Department of Energy (US)	PFEI	Payload Fuel Energy Intensity
EIA	Energy Information Agency	PKO	Palm Kernel Oil
EISA	Energy Independence and Security Act	POME	Palm Oil Mill Effluent
FAPRI	Food and Agricultural Policy Research Institute	SAGD	Steam Assisted gravity Drainage
F-T	Fischer Tropsch	SOR	Steam to Oil Ratio
FFB	Fresh Fruit Bunches (from oil palm)	SOC	Soil Organic Carbon
GHG	Greenhouse Gas	SPK	Synthetic Paraffinic Kerosene
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation	TSS	Total Suspended Solids (mass percent)
GTL	Gas to Liquids (via F-T)	TTW	Tank to Wake
GWP	Global Warming Potential (IPCC)	ULS	Ultra Low Sulfur
HRD	Hydroprocessed Renewable Diesel	USDA	United States Department of Agriculture
HRJ	Hydroprocessed Renewable Jet	WTT	Well to Tank
ICP	In-situ Conversion Process (Shell)	WTW	Well to Wake
IGCC	Integrated Gasification Combined Cycle		

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1 Introduction

Both economic and environmental sustainability are required for any transportation fuel to be viable in the long term. An expansion of our energy portfolio to include alternative fuels would also result in the desirable consequence of energy diversity. This report presents results from ongoing research within the Partnership for AiR Transportation Noise and Emissions Reduction at MIT on alternative fuels. As shown by the diagram of Figure 1, the PARTNER alternative fuels research portfolio is considering many aspects of alternative fuel sustainability. This report focuses on aspects of environmental sustainability, with an emphasis on life cycle greenhouse gas emissions as they relate to impacts on global climate.
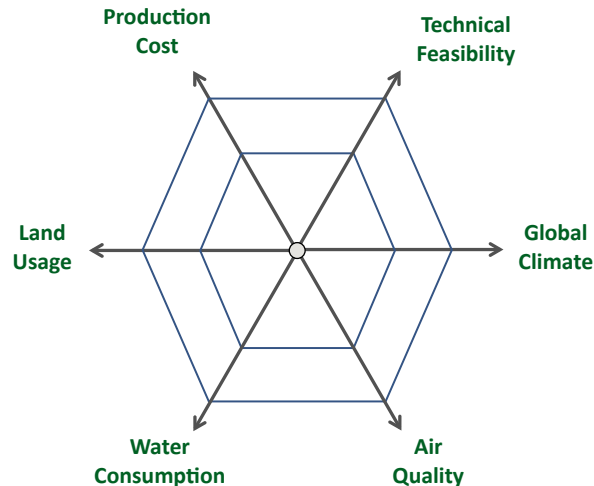
Figure 1: Aspects of alternative fuels that are being considered for alternative fuel feasibility and sustainability. This report has an emphasis on life cycle greenhouse gas emissions

Alternative jet fuels created from renewable resources offer the potential to reduce the greenhouse gas (GHG) emissions from aviation. This is not due to a change in fuel composition nor is it due to a change in engine efficiency; instead the reduction is due to a change in the GHG emissions that result from the extraction, production and combustion of the alternative fuel relative to conventional jet fuel. Through a life cycle accounting of the GHG emissions that starts with the well, mine, or field where the fuel feedstock is extracted, and extending to the wake behind the aircraft, one can ascertain the change in GHG emissions that result from the use of an alternative fuel. To emphasize the importance of life cycle analysis for an accurate comparison of GHG emissions, consider that the combustion of synthetic fuels, which are covered extensively in this work, results in about 4% less CO_2 emissions (per unit mass of fuel) as compared to conventional jet fuel. However, as will be shown in this report, the life cycle GHG emissions from various alternative fuels can vary by two orders of magnitude depending on the feedstock and the details of production.

There are many other issues that need to be considered when evaluating the potential of a specific alternative fuel. These include, but are not limited to, the efficient usage of water and land resources, the environmental impacts on air quality, and the economic cost of fuel production. This work touches on water and land usage, but not air quality or economic costs. The interested reader is directed to Hileman et al. (2009) for an extended discussion of these and other aspects of alternative jet fuel feasibility. Furthermore, this work does not consider whether our limited biomass resources would provide greater societal benefit if they were used elsewhere, for example to create fuels for ground transportation or to generate heat and/or electricity. A recent analysis by Hedegaard et al. (2008) indicates that scarce biomass resources could be used more effectively, from perspectives of energy efficiency and CO_2 mitigation, for heat and electricity

rather than ethanol for transportation. These topics are being considered as part of the ongoing PARTNER research portfolio.

The focus of this manuscript is a comparison of the well-to-wake, life cycle GHG emissions from select feedstock-to-jet fuel pathways for the United States; this is a partial list of potential alternative jet fuel options and future revisions to this report will expand upon this list. In some cases, details for a given fuel production pathway are expected to differ for other countries. The underlying data for each analysis have come from the literature, but each fuel pathways has been considered using a consistent methodology. This work is the first of its kind in that that a broad range of alternative fuels are analyzed for aviation using consistent methodologies that facilitate equitable comparisons. Each pathway is presented in such a way that transparency of assumptions is maximized and the reader is able to identify the pivotal factors defining fuel production for each feedstock. As there is considerable variability in the life cycle GHG emissions from existing fuel production, and many of the fuel pathways considered in this report have not been commercialized, a range of life cycle GHG emissions has been provided for each feedstock-to-fuel pathway. In addition to the examination of life cycle GHG emissions, the manuscript provides a first order examination of the land and water usage that could accompany the development of a biofuel industry for aviation.

The contents of the report are structured as follows. Chapter 2 presents the fuel pathways considered in this work while Chapter 3 provides background information on creating a life cycle GHG emissions inventory. Chapters 4 through 7 present life cycle GHG estimates for conventional jet fuel and various other alternative jet fuel pathways. Chapter 4 considers standard and Ultra Low Sulfur (ULS) jet fuel from conventional petroleum. Chapter 5 considers conventional jet from unconventional petroleum resources of oil sands and oil shale. Chapter 6 examines Fischer-Tropsch synthesis as a means to create synthetic paraffinic kerosene (SPK) fuel from natural gas, coal, and biomass. Chapter 7 presents the analysis of hydroprocessed renewable jet fuels (HRJ) from soy, palm, rapeseed, jatropha, algae, and salicornia. Chapter 8 presents a comparison of the life cycle GHG emissions established in Chapters 4 through 7 with a discussion of additional constraints that should be considered to when assessing environmental viability of alternative fuel options. These include potential damages caused by the introduction of invasive species and land and water availability. Conclusions from this work are presented in Chapter 9.

2 Alternative Jet Fuel Pathways

The fuel options considered herein are "drop-in" alternatives in that they have the potential to serve as a direct replacement for conventional jet fuel, requiring little or no modification to existing infrastructure or aircraft. Alcohols and biodiesel (fatty acid methyl esters) are not considered due to both the safety issues and the decrease in fuel efficiency that would accompany their use in aircraft operations (Hileman et al., 2009). The fuel compositions analyzed in this work are as follows:

- Conventional jet fuel (e.g., Jet A) from conventional petroleum (crude oil) and unconventional petroleum (oil sands and oil shale)
- Ultra Low Sulfur (ULS) jet fuel from conventional petroleum
- Synthetic Paraffinic Kerosene (SPK) fuels created via Fischer-Tropsch (F-T) synthesis or hydroprocessing of renewable oils.

This should not be viewed an all-encompassing list of potential fuel compositions that could be used as a replacement or a blend stock in aviation gas turbine engines. For example, advanced fermentation or pyrolysis can both be used to create a jet fuel blendstock,[1] but neither of these fuels is considered here.

ULS jet fuels are considered as they could reduce aviation's impact on air quality. The processing used to remove sulfur from conventional jet fuel (discussed in detail in Section 4.3) changes the fuel properties such that an additive package may be required to improve lubricity. Considerable research is ongoing in regards to the analysis of ULS jet fuels (e.g., Miller et al, 2009; Hileman et al., 2010; PARTNER Project 27, 2010).

SPK fuels have similar molecular composition to conventional jet fuel with the primary difference being a lack of aromatic compounds; conventional jet fuel typically contains 20% aromatic compounds. Because certain types of seals require aromatic compounds for proper swelling, SPK fuels have been certified in blends up to 50% with conventional jet fuel to maintain a minimum aromatic content.

The fuel production pathways analyzed in this work were jet fuel from conventional crude oil, jet fuel from Canadian oil sands, jet fuel from oil shale, Fischer-Tropsch jet fuel from natural gas, coal and biomass, and jet fuel created by hydroprocessing soy oil, palm oil, rapeseed oil, algal oil, jatropha oil and salicornia oil. As of September 2009, at 50% blend of SPK fuel derived via Fischer-Tropsch synthesis with conventional jet fuel was certified for use in commercial aviation under ASTM spec D7566-09 (ASTM D7566-09, 2009). The fuel specification and certification division of the Commercial Aviation Alternative Fuels Initiative (CAAFI) played a key role in the process of establishing D7566 and has outlined plans for a expanding the specification to include a 50% HRJ blend with conventional jet by 2010, 100% F-T derived SPK by 2011 and 100% HRJ by 2013 (Rumizen, 2010).

For each pathway, three potential scenarios (low emissions case, baseline case and high emissions case) were identified and life cycle GHG emissions were calculated for each of these scenarios. The fuel pathways are summarized in Table 1.

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[1] Note that pyrolysis is being examined to create synthetic aromatics that could be blended with SPK fuels to create a fully synthetic jet fuel while advanced fermentation is being examined to create a paraffinic fuel for aviation gas turbines.

Table 1: Fuel pathways investigated

Source	Feedstock	Recovery	Processing	Final product
Petroleum	Conventional crude	Crude extraction	Crude refining	Jet Fuel
	Conventional crude	Crude extraction	Crude refining	Ultra low Sulfur Jet Fuel
	Canadian oil sands	Bitumen mining/ extraction and upgrading	Syncrude refining	Jet Fuel
	Oil shale	In-situ conversion	Shale oil refining	Jet Fuel
Natural gas	Natural gas	Natural gas extraction and processing	Gasification, F-T reaction and upgrading	F-T Jet Fuel (GTL)
Coal	Coal	Coal mining	Gasification, F-T reaction and upgrading (with and without carbon capture)	F-T Jet Fuel (CTL)
Coal and Biomass	Coal and Biomass	Coal mining and biomass cultivation	Gasification, F-T reaction and upgrading (with carbon capture)	F-T Jet Fuel (CBTL)
Biomass	Biomass	Biomass cultivation	Gasification, F-T reaction and upgrading	F-T Jet Fuel (BTL)
	Soy oil	Cultivation and extraction of soy oils	Hydroprocessing	HRJ Fuel (Hydroprocessed Renewable Jet)
	Palm oil from Southeast Asia	Cultivation and extraction of palm oils	Hydroprocessing	HRJ Fuel
	Rapeseed Oil	Cultivation and extraction of soy oils	Hydroprocessing	HRJ Fuel
	Algae oil	Cultivation and extraction of algae oils	Hydroprocessing	HRJ Fuel
	Jatropha oil	Cultivation and extraction of jatropha oils	Hydroprocessing	HRJ Fuel
	Salicornia oil and solid biomass	Cultivation of salicornia and extraction of salicornia oils	Gasification, F-T reaction and upgrading (with carbon capture); Hydroprocessing	F-T Jet Fuel and HRJ Fuel

3 Procedural Overview of Life Cycle Analysis of GHG Emissions

3.1 *Life cycle Greenhouse Gas Emissions*

The life cycle analysis of alternative jet fuels encompasses emissions from the complete fuel cycle. This includes recovery and transportation of the feedstock from the well, field, or mine to the production facility, processing of these materials into fuels, transportation and distribution of the fuel to the aircraft tank, and finally, the combustion of the fuel in the aircraft. The steps of such a well-to-wake life cycle analysis are shown schematically in Figure 2. These "well-to-wake" (WtW) steps can be broadly grouped into fuel production and distribution, "well-to-tank" (WtT), and fuel combustion, "tank-to-wake" (TtW). The interested reader is directed to a recent guidance document created by a consortium that was assembled by the US Air Force (AFLCAWG, 2009) to learn more about the details of creating a life cycle GHG inventory for transportation fuels.
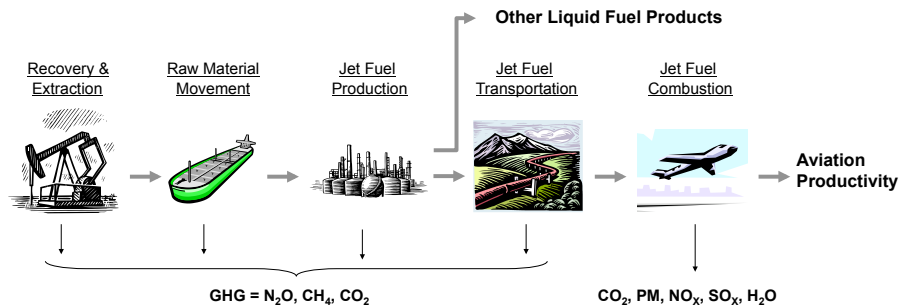
Figure 2: Steps considered, in the well-to-wake, life cycle GHG inventory of conventional jet fuel

For each step of the life cycle, GHG emissions are assessed and reported on the basis of per-unit energy consumed by the aircraft (per megajoule). The GHG covered in this analysis are carbon dioxide, methane and nitrous oxide using their 100-year global warming potentials (Solomon et al., 2007). This analysis did not cover non-CO_2 combustion emissions from aircraft - for example NO_X, SO_X, soot, and water - that directly or indirectly impact global climate change. These are currently being considered as part of ongoing research and will presented in future publications. This study also did not consider energy or GHG emissions associated with the initial creation of infrastructure such as extraction equipment, transportation vehicles, farming machinery, processing facilities, etc. The impact of such emissions on the total life cycle GHG emissions of the pathway is usually relatively small, and within the uncertainty range of the analysis. (Hill et al., 2006, Edwards et al., 2007).

Fossil feedstocks such as crude oil, coal or natural gas are created from geologically sequestered carbon sources, and the carbon is released as CO_2 when the fuel products are burned. Such combustion CO_2 has to be taken into account in the life cycle analysis (see Figure 2). Biomass feedstocks absorb CO_2 from the atmosphere when they grow and the CO_2 emitted during fuel combustion is equal to that absorbed during biomass cultivation. Hence, many biofuels have a "biomass credit" that offsets the combustion CO_2 in the life cycle analysis (see Figure 3). *This biomass credit is the primary difference between biomass and fossil fuels in terms of their GHG emissions.* However, a biofuel does not necessarily have GHG emissions that are below a fossil fuel.
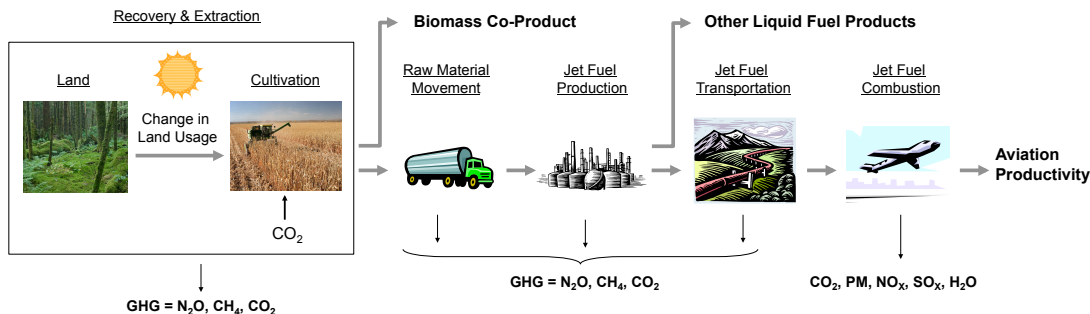
Figure 3: Steps considered in the well-to-wake, life cycle GHG inventory of bio-based alternative jet fuels

Biomass feedstocks also have the potential for CO_2 emissions or CO_2 sequestration from changes in land use (see first step of Figure 3). The CO_2 emissions or sequestration are due to changes in the biomass, soil and organic waste contained on and within the land. In some instances, these emissions can dominate the life cycle GHG emissions of the biofuel pathway. The land use change can be a direct land conversion, (e.g., tropical rainforest being cleared for cropland to grow feedstocks), or it can be an indirect conversion resulting from land being converted elsewhere in the world due to economic signals induced by increased demand for agricultural products. In either case, it is assumed that a fixed quantity of biomass (e.g., vegetable oil) needs to be supplied to global food markets and that additional production (for biofuel creation) is met by land that has been converted from some previous use. The magnitude of land use change emissions depends primarily on the type of land being converted to cropland and the type of crops being grown. For fossil feedstocks, where conversion of land (e.g. forest land, grass land) for extraction of fossil resources (e.g. extraction of bitumen) or placement of fuel processing facilities (e.g. oil refineries) takes place, land use change emissions are negligible compared to other components of the fuel pathway. This is because a large throughput of fuel volume or mass (as well as energy) is created per unit area of converted land.

Only emissions from direct land use change, where land is converted to facilitate biofuel production, were considered in this work. No attempt was made to quantify the magnitude of indirect land use change emissions resulting from fluctuations in supply and demand of other crops because of increased biofuel production. In order to properly capture these effects, a detailed economic model is required which falls beyond the capabilities of GREET and the scope of this work. Other academic and government groups have invested a significant amount of effort to quantify indirect land use change emissions and the reader is directed to their publications for estimates of these effects (EPA, 2010; Melillo et al., 2009).

For biofuels produced from algae, sufficient growth rates cannot be achieved without the direct feeding of CO_2 during growth. This is because the atmospheric concentration of CO_2 is too dilute to support an economically viable growth rate (Putt, 2007). The CO_2 used to feed the biomass must be abundant and come from a source external to the biofuel production system. In this study, fossil energy resource-based electricity generation was chosen to meet these needs. One can imagine a coupled system in which the CO_2 source and algae facility are linked to one another, where a fossil fuel is the primary input and both electricity and algal biofuel are primary outputs. This concept is shown schematically in Figure 4 with the system boundary for a conventional biofuel pathway expanded to include an outside source of CO_2. In addition, direct land use changes should be small as compared to crop-based biofuels and indirect land use changes should be minimal because the necessary infrastructure can be created in wasteland and desert areas. Algae also have the capability to grow in saline water, meaning that fresh water is not a pre-requisite.
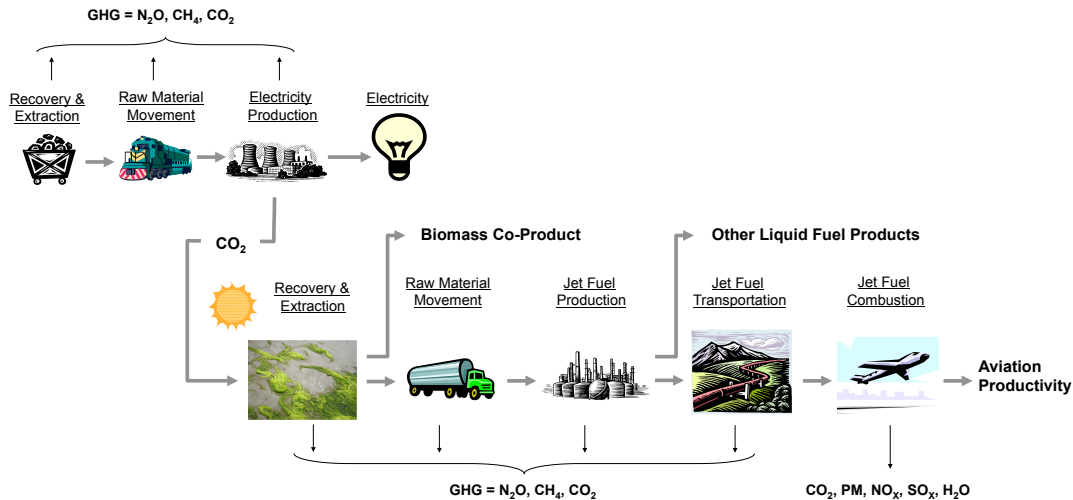
Figure 4: Steps considered in the well-to-wake, life cycle GHG inventory of bio-based fuels from algae

Fuel production generally results in the creation of co-products in addition to the primary fuel product. For example, the systems shown in Figures 2-4 generate biomass and liquid fuel co-products. These co-products have embodied value that can be quantified based on physical metrics, or their ability to displace some other product elsewhere in the greater marketplace. Four methods can be used to assign life cycle GHG emissions between the primary fuel product and any co-products that are created:

- Mass allocation
- Energy allocation
- Market-value allocation
- Displacement (or substitution, or system expansion)

The international organization for standardization (ISO) states in ISO 14044:2006(E) that processes shared with other product systems shall be identified and dealt with by preferentially using process disaggregation, system expansion, allocation by an underlying physical relationship and economic value, in this order. Inventories are based on material balances between input and output; therefore, allocation procedures should attempt to approximate such fundamental input/output relationships and characteristics (ISO, 2006).

The mass and energy allocation approaches distribute the life cycle GHG emissions based on either the mass or energy content, respectively, of the co-products and the fuel. In this work, the energy allocation method was used to allocate energy and emissions between co-products of the Fischer-Tropsch process as well as those in the hydroprocessing of renewable oils to make Hydroprocessed Renewable Jet (HRJ); this is because these co-products have utility as energy resources.

The market allocation approach apportions emissions based on the market prices of the co-products and primary fuel product. Unlike the mass or energy allocation approaches, the market value allocation can change with time. The sensitivity to market forces could be particularly useful when co-products are generated in quantities that stand to flood existing markets and drive the co-product price to zero. For example, if a fuel has a co-product that displaces some existing product, then the market value method will capture the diminished utility of creating additional co-product by allocating more of the emissions to the fuel being produced. This is because increasing alternative fuel production will not change the price of the alternative fuel as this is set by the price of conventional fuel net subsidies and taxes. Co-product creation does, however, have the potential to alter the price of similar commodities. In this work, market valuation was

used to allocate emissions between co-products leaving the system after oil extraction within the HRJ pathways.

The displacement method assumes that the production of the incidental co-product displaces the production of a substitute product. As a result, an emissions credit from the displaced product that is no longer produced is applied to the primary product. Although this methodology is desirable because it is time-invariant and it could in theory be applied to any co-product, it is hard to implement. This is because of difficulties in identifying a suitable product to be displaced, calculating the life cycle GHG emissions of that displaced product and determining the displacement ratio (Huo et al., 2008). In the case of biofuels, the issue of how to appropriately allocate land use change emissions further complicates the application of the displacement method. The life cycle analysis of algae in this work applied the displacement method to the fossil based electricity used as a CO_2 source.

The use of different allocation methodologies can lead to substantially different results, particularly in regards to biofuel pathways where significant quantities of co-products are being produced. The appropriate method may depend to a large extent on the question that the life cycle analyst attempting to answer. Regardless of which method is applied, it is important that those conducting life cycle analyses for any purpose clearly state the allocation approach adopted.

3.2 Analysis Procedure

Analyses of life cycle GHG emissions for several jet fuel production pathways were carried out based on available information in the scholarly and technical literature. The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) framework (versions 1.8b and 1.8a) and its supporting data, both developed and maintained by Argonne National Laboratory, was the primary tool used in the well-to-wake life cycle GHG analysis.[2] A simulation year of 2015 was used and default GREET assumptions were used in the analysis of the pathways, except where more recent data were obtained. For example, the average efficiencies of coal-fired power plants (utility boiler) and coal integrated gasification combined cycle (IGCC) plants were assumed to be 36% and 41.5%, respectively, on a lower heating value basis[3] (Deutch and Moniz, 2007).

A key limitation of the GREET framework is that it is designed for ground transportation fuels and vehicle systems and does not include jet fuel production pathways. Also, not all of the feedstocks analyzed in this work are available in GREET (e.g. oil shale). Hence, this work utilized data from the literature on jet fuel and jet fuel alternatives where available (e.g. fuel properties, refining efficiency) and incorporated them into the GREET framework to derive life cycle GHG emissions. Where supporting data are presented in this report, mixed units are used for consistency with GREET version 1.8a/1.8b.

The GREET framework was primarily used as a database and calculation platform, where the quality of results depends on the quality of input assumptions such as energy efficiencies, fuel properties and emission allocation method for co-products. Hence, a de novo approach was taken in identifying and reviewing key inputs and assumptions for each pathway. Specifically, default GREET input assumptions were examined for the fuel pathways available in GREET. Key parameters with a significant impact on the life cycle GHG emissions of the pathway were identified. Default GREET values for these key parameters were updated wherever necessary

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[2] The specific version of GREET used for each pathway within a section is stated at the beginning of the section; however, the impact on the results of this work of using GREET version 1.8a or 1.8b is negligible compared to the inherent uncertainties of life cycle analysis.

[3] From Deutch and Moniz, 2007, the US coal fleet average generating efficiency is about 33% (HHV) and the generating efficiency for coal IGCC plants is 38.4% (HHV). Since the difference between HHV and LHV range from 2 to 4%, a 3% difference is assumed in this report. Hence, the efficiency of an average coal-fired power plant is assumed to be (33+3) 36% and the efficiency of a coal IGCC plants is assumed to be (38.4+3) 41.5%.

using reviews of recent information available in the literature. Where a specific pathway was not available in GREET, the pathway was built within the GREET framework with all relevant input parameters gathered from the open literature.

The analysis methodology used in this work differs from that adopted by the US Environmental Protection Agency (EPA) in their Renewable Fuel Standard (RFS2) recently released in February 2010. Specifically, life cycle assessments can be categorized as either attributional or consequential. As defined by the EPA:

"An attributional approach to GHG emissions accounting provides information about the GHG emitted directly by a product and its life cycle. The product system includes processes that are directly linked to the product by material, energy flows or services through the supply-chain. A consequential approach to GHG emissions accounting in products provides information about the GHG emitted, directly or indirectly, as a consequence of changes in demand for the product. This approach typically describes changes in GHG emissions levels from affected processes, which are identified by linking causes with effects." (EPA, 2010)

Attributional and consequential life cycle analyses will tend to yield different results for an identical product; hence, comparing results from the two methodologies is inappropriate. This work implements an attributional methodology while the EPA has used a consequential analysis to more broadly consider the impact of future policy scenarios.

3.3   Goals and Practices

A life cycle analysis should be consistent with the goals of the study. As discussed in the "Framework and Guidance for Estimating Greenhouse Gas Footprints of Aviation Fuels", different levels of analysis fidelity originate primarily from robustness of assumptions, data quality and level of model completeness (AFLCAWG, 2009). These differences are summarized in Figure 5. As indicated, this work focuses on conducting high quality screening-level analyses using assumptions that attempt to capture industry averages.
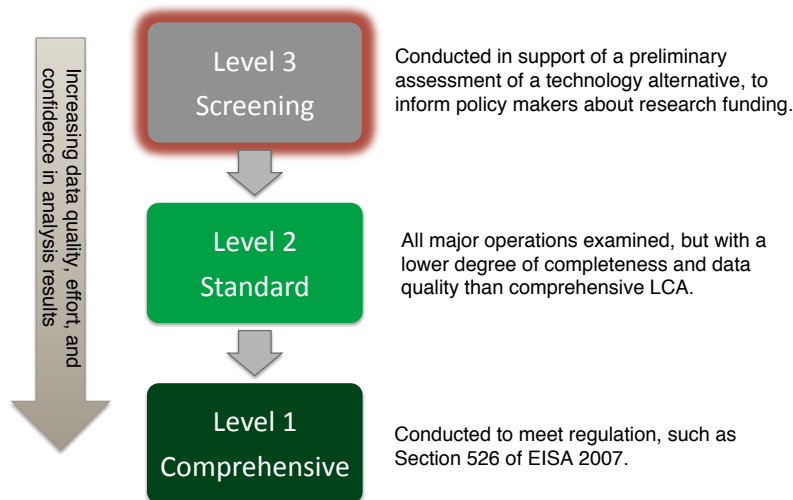
Figure 5: Levels of life cycle GHG studies used to ensure that the adopted practices are consistent with the goal of the work

Although a screening level analysis approach was chosen, uncertainty ranges were established using optimistic, nominal and pessimistic scenarios such that the result is presented as a range instead of a single point. In most cases, the GHG emissions inventories established for each pathway are not representative of an existing production configuration. As such, site-specific examinations of individual fuel pathways are still essential to quantifying specific GHG footprints

such as would be required to meet Section 526 of EISA 2007. The results of this work do not replace such an analysis.

3.4 Evaluation of Variability

To explore the impact of variability in key parameters values, three different scenarios – low GHG emissions, baseline or nominal GHG emissions, and high GHG emissions – were envisaged for each pathway. Key parameters were identified through examination of the GHG emissions that resulted from each of the individual steps of the life cycle (see Figures 1 through 3). The engineering judgment of the authors was used to identify parameters that had both variability as well as a considerable influence on the life cycle GHG emissions. Input parameters such as process efficiency and biomass feedstock yield have both of these qualities in that they exert considerable influence on the life cycle GHG emissions of the fuel pathway and their value a decade into the future could have considerable variability; hence, these parameters were varied as part of the three scenarios. Well defined parameters with a large impact on the life cycle emissions (such as the mass of CO_2 emitted per unit of fuel consumed by the jet engine) and parameters with large variability but only a small impact on the life cycle emissions (such as the distance the feedstock needs to travel from the source to the refinery) were generally not examined.

By using key parameters to define the low, baseline, and high emissions scenarios, a range of GHG emissions, rather than a single value, was derived for each fuel pathway. Appropriate values for the key parameters were determined through literature review and consultation with relevant experts. In general, industry average values, rather than marginal values, were sought. If a marginal value for a key parameter was found that fell outside of typical values and if the marginal value indicates a potential industry trend, then the value was examined as a separate case study. Variation of the key parameter values across the three scenarios could arise from differences in time frame (e.g. historical data versus future projections), different feedstocks (e.g. bituminous coal versus sub-bituminous coal), different technologies or changes in process designs. While the upper and lower bounds of values found in the literature were generally used in the low and high emissions cases, baseline values were usually those which were deemed most likely, most frequently occurring, or were the average or mid-point of the range of values reported in the literature.

Some of the pathways under consideration result in nitrous oxide emissions that represent more than 50% of the total life cycle GHG emissions. This work applied IPCC Tier 1 methodology to calculate N_2O emissions from each fuel pathway and is therefore subject to the full range of uncertainty associated with IPCC correlations. In many cases, the emission factors developed by the IPCC have uncertainty ranges in excess of 100% (De Klein et al., 2006). While the focus of this work was not to assess the uncertainties within the IPCC methodology, the reader should be aware of their existence and that their impact is of larger consequence for pathways where N_2O is a significant contributor to the total life cycle GHG emissions.

3.5 Functional Unit: g CO_2e/MJ and g CO_2e/kg-km

Consistent with (AFLCAWG, 2009), the life cycle GHG emissions are presented using a metric that captures the mass of GHG per unit of energy (lower heating value) consumed by the aircraft. This is given in units of g CO_2e/MJ where

$$CO_2e = \left(CO_2 + GWP_{CH_4} \cdot CH_4 + GWP_{N_2O} \cdot N_2O\right)_{\text{well-to-tank}} + \left(CO_2\right)_{\text{tank-to-wake}} \qquad \text{Equation 1}$$

As discussed in Section 3.1, non-CO_2 combustion emissions were not considered in this report. A metric of life cycle GHG intensity, with units of gCO_2e/kg-km, could be generated from this metric via multiplication by an energy efficiency metric such as Payload Fuel Energy Intensity (PFEI) with units of energy per payload distance flown (Hileman et al., 2008). This metric will be considered in Section 8.2.

4 Conventional Petroleum Pathways

Because this work focuses on fuels that could be used in gas turbine powered aircraft, the baseline for analysis is jet fuel from conventional petroleum. For the purposes of this report, jet fuel could represent JP-8, Jet A, or Jet A-1, which are the fuels in use by the US Air Force, commercial aviation in the US, and commercial aviation in Europe as well as much of the rest of the world, respectively. Section 4.1 considers the extraction of conventional crude oil for jet fuel while Section 4.2 considers petroleum refining to produce jet fuel. Section 4.3 examines an ultralow sulfur jet fuel (ULSJ) with a maximum fuel sulfur content of 15 parts per million. ULSJ differs from the other alternative fuels considered in this report in that it is derived from conventional petroleum and it would be used specifically to reduce aviation's impact on air quality as opposed to diversifying energy sources or reducing greenhouse gas emissions.

The steps involved in the production of jet fuel from conventional petroleum sources include crude oil extraction, transportation of crude oil to US refineries, refining of crude oil to jet fuel, and the transportation of jet fuel to the aircraft tank. The GHG emissions resulting from crude oil extraction, crude oil transportation and jet fuel transportation were obtained using data from two recently published National Energy Technology Laboratory (NETL) studies on the life cycle GHG emissions of petroleum-based transportation fuels (Skone and Gerdes, 2008; Skone and Gerdes, 2009). The emissions that result from crude oil refining were calculated using both a top-down and a bottoms-up perspective for conventional and an ultra low sulfur jet (ULSJ) fuel within GREET version 1.8a (Sections 4.2 and 4.3, respectively). This work differs from that of Skone and Gerdes (2008, 2009) in that the jet fuel pathway considers only jet fuel refined within the US and excludes jet fuel made from unconventional petroleum sources, such as oil sands, from the final result (jet fuel from unconventional sources is treated individually in section 5). Jet fuel refined within the US comprised 88.7% of all domestic jet fuel consumption in 2005 (Skone and Gerdes, 2008).

4.1 Crude Oil Recovery and Transportation

The source of crude oil is important in order to properly represent the range of resulting GHG emissions. The GHG emissions from crude oil recovery and crude oil transportation are designated origin specific GHG emissions. The variation in these emissions by crude oil source is primarily due to specific hydrocarbon flaring and venting practices during extraction, the emissions resulting from local electricity production, equipment efficiency and the transportation distance of crude oil to a US port.

Imported crude oils are on average heavier (lower API gravity[4]) and contain higher levels of sulfur than domestic products (Skone and Gerdes, 2009). The changes in crude oil properties as well as processing technique drive a variation in processing emissions of converting crude oil into finished fuel products. The GHG emissions associated with the processing of crude oil to jet fuel and ULS jet fuel were developed in this study using the GREET framework. These results are presented in conjunction with the origin specific GHG emissions and finished fuel transportation emissions derived from Skone and Gerdes (2008 and 2009) to establish GHG inventories for conventional jet fuel and ULS jet fuel produced at US refineries.

Of the crude oil mix fed into US refineries in 2005, only 34% was domestically produced. The other 66% was imported from other counties located around the world. When including domestic production, over 90% of the crude oil mix came from only 11 countries (Skone and Gerdes, 2008). Throughout this work, the remaining fraction of imported crude is designated 'other' and corresponds to the weighted average of all imported crude (excluding Canadian oil sands).

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[4] API gravity is a measure of the density of a petroleum liquid relative to water. An API gravity greater than 10 indicates lighter than water while an API gravity less than 10 indicates heavier than water. *API gravity = 141.5/SG - 131.5, where SG = specific gravity*

Table 2: Country specific average waterborne round trip travel distance of crude oil to US domestic ports

Crude Oil Source	Country-Specific Average Round-Trip Travel Distance (nautical miles)	Fraction of Imported Crude Transported by Ocean Tanker
Saudi Arabia	24036	100%
Kuwait	25052	100%
Iraq	24740	100%
Venezuela	3578	100%
United States	0	0%
Ecuador	11306	100%
Algeria	8904	100%
Canada	1350	28%
Mexico	2122	100%
Other	20000	100%
Angola	13472	100%
Nigeria	11344	100%
Notes:		
1) Skone and Gerdes (2008)		
2) Imported crude from Canadian oils sands ignored in this section because it is explicitly dealt with in Section 5.1		
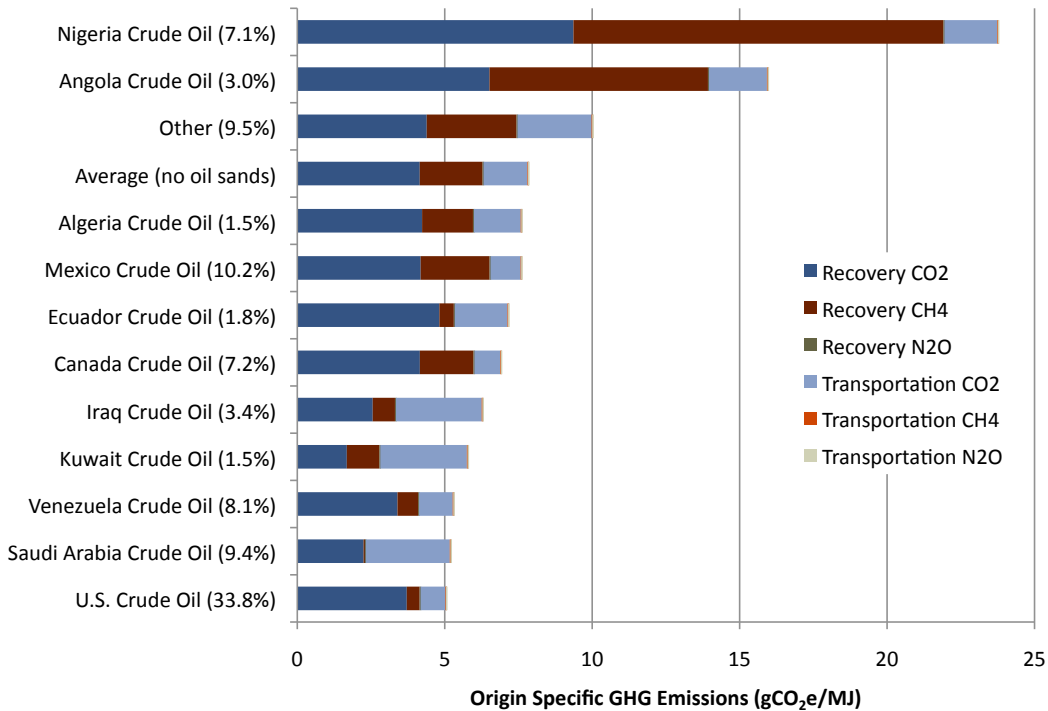
Figure 6: Origin specific GHG emissions by species of crude oil entering US refineries in 2005 (fraction of total imports in parentheses). Based on country profiles published in Skone and Gerdes (2008, 2009)

In their 2008 life cycle GHG analysis of petroleum based fuels, Skone and Gerdes developed crude oil extraction profiles, including methane flaring and venting data, for each of these 11 countries. They also developed a transportation profile for each country by accounting for the transport of imported crude oil from its point of extraction to foreign ports, ocean tanker transport of waterborne imported crude oil to domestic ports and crude oil transport within the US. Domestic crude oil is only subject to transport within the US and 72% of Canadian crude imports

are transported via pipeline and do not incur waterborne transport emissions; all other imports are subject to all three forms of transportation. Listed in Table 2 are the round trip distances used to calculate waterborne GHG emissions and the fraction of imports transported by ocean tanker for each crude oil source. The interested reader is directed to Skone and Gerdes (2008) for the emissions factors for each leg of the transportation process.

The significance of the data reported in Table 2 is that importing crude oil from Saudi Arabia and Kuwait results in substantially more GHG emissions from transportation than crude from other sources. The recovery and total transportation GHG emissions (by species) from 2005 are given for crude oil from each source in Figure 6. Beside the label for each source, the volumetric fraction of total crude fed into US refineries imported from that country is given in parentheses. The volumetric fractions do not add up to 100% because Canadian oil sands are not listed.

These data highlight that domestically produced crude oil results in lower GHG emissions than any other source because of reduced transportation emissions. Even though CO_2 emissions resulting from domestic oil extraction are higher than Iraq, Kuwait, Venezuela and Saudi Arabia, the combination of transportation emissions and methane venting causes these regions to have higher origin specific GHG emissions than the US. In the cases of Iraq, Kuwait and Saudi Arabia, transportation emissions (primarily ocean transport emissions) represent 47%, 51% and 55% of the total origin specific GHG emissions.

The primary driver for countries with excessive origin-specific GHG emissions is methane venting. Methane emissions from Nigerian and Angolan crude extraction exceed all other origin specific GHG emissions. Mexican and Canadian crude also have non-negligible methane emissions from venting.

The variation in profiles shown in Figure 6 was used to establish the low emissions, baseline and high emissions scenarios for jet fuel and ULS jet fuel from conventional crude. The low emissions scenario was composed of purely domestic crude oil, the baseline scenario adopted the weighted average of all crude oil fed into US refineries, excluding Canadian oil sands, and the high emissions scenario used only Nigerian crude. The transportation of jet fuel from US refineries to the aircraft tank are independent of the source of crude oil; hence, a single result was used for all three emissions scenarios.

4.2   Conventional Jet Fuel from Crude Oil

The conventional jet fuel production pathway forms the baseline against which the life cycle GHG emissions of alternative jet fuels are compared. The properties of conventional crude oil were based on the projected average crude oil received by US refineries in 2015, obtained using historical data provided by the EIA. There is a definite trend for crudes to become heavier and more sour (more sulfur) in the future; therefore, a business as usual scenario would likely see jet fuel production becoming more energy intensive as more hydroprocessing is required to maintain current product quality. This means that the energy intensity of refining may increase beyond the values used in this study. Some discussion of the impacts of crude oil quality on GHG emission from refining is given section 4.2.3 (EIA, 2008a).[5]

The key parameter in analyzing the GHG emissions associated with the production of jet fuel from crude oil is fuel-refining efficiency. Two methods were employed in the derivation of jet fuel refining efficiency. The first method was a top-down approach, which derived the jet fuel refining energy efficiency from the overall US refining energy efficiency. This formed the baseline case. The second method was a bottom-up approach, which estimated jet fuel-refining efficiency by summing the energy requirements for the individual refining processes. Specifically, two extreme

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[5] From data on sulfur content and API gravity of average crude oil input to US refineries from 1995 to 2006 given by EIA (EIA, 2008a), it was estimated that there was approximately a 2% annual increase in sulfur content and 0.25% annual decrease in API gravity. From these trends, the average crude oil quality received by US refineries in 2015 was estimated.

cases were examined: straight-run fuel production and hydroprocessing of crude. The refining efficiencies obtained using the bottom-up approach were used for the low and high emissions cases, respectively, providing a bound on the range of possible values.

4.2.1 Top-Down Approach (Baseline Case)

The overall US refinery efficiency as estimated by Wang (2008[6]) was 90.1% (LHV), based on statistics of process fuel use in US refineries, and 2006 refinery fuel inputs and outputs provided by the EIA. Overall refinery efficiency is defined as (Wang, 2008):

$$\eta_0 = \frac{E_{products}}{E_{inputs}}$$

Equation 2

where η_o = petroleum refinery energy efficiency,
$E_{products}$ = energy of all petroleum products,
E_{inputs} = energy in crude input, other feedstock inputs, and process fuels.

The jet fuel-specific refining efficiency was calculated from the overall refinery efficiency using the kerosene relative energy intensity[7] and Equation 3, developed by Wang et al. (2004). The relative energy intensity of the production of kerosene (jet fuel), X_S, based on an energy-content process allocation method, is 62.4%, and the overall refining efficiency, η_0, is 90.1%.

$$\eta_s = \frac{\eta_0}{\eta_0 + X_s(1 - \eta_0)}$$

Equation 3

From Equation 3, the refining energy efficiency of jet fuel is 93.5% (LHV). This refining efficiency was used in the baseline case for the life cycle analysis of GHG emissions from the production of jet fuel from conventional petroleum.

Based on 2006 US refinery data published by the EIA, Wang (2008) calculated the shares of process fuel used in US petroleum refineries (see Table 3). These shares were used as inputs to the GREET model for the baseline case of jet fuel production.

Table 3: Type of process fuel and fuel share in the refining of jet fuel and ULS jet fuel

Type of process fuel	Process energy share (%)
Electricity	3.5
Natural Gas	41.3
Refinery Gas	39.6
Coke	14.3
Residual Oil	1.3
Total	100

4.2.2 Bottom-up Approach (Low and High Emissions Cases)

The jet fuel produced by a refinery may be straight-run, produced from hydroprocessed stocks, or a blend of both. The refining efficiency of each production method was estimated by summing the energy inputs of individual processes. These two cases formed the low and high emissions

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[6] This work was used to update the refining efficiency of gasoline, diesel, LPG, residual oil and naphtha in GREET (version 1.8b). (GREET, 2008)

[7] Relative energy intensity was defined as "the ratio of total energy use share to the mass share of a given fuel." It provides a measure of how energy intensive the production of a particular fuel is relative to the mass share of that fuel produced. A relative energy intensity of more than 100% for a particular fuel means that the production of that fuel consumes a greater share of overall process energy than the mass share of that fuel produced. The energy intensity of the overall refinery is 100%.

scenarios, respectively, for the life cycle analysis of GHG emissions in the production of jet fuel from conventional petroleum.

The energy for the processes involved in refining jet fuel was taken from a 2007 Department of Energy sponsored report (Pellegrino et al., 2007). The report provided both a range of refining process energy use, as well as average energy use. The average energy use data for the relevant refining processes were used to calculate the jet fuel refining efficiency.

Straight-Run Jet Fuel
The main processes involved in the production of straight-run jet fuel are crude desalting and atmospheric distillation, followed by chemical treatments (such as the Merox process) to remove contaminants like mercaptans and organic acids, etc. The estimated process energy in crude desalting and atmospheric distillation is shown in Table 4. As no data were found in the literature regarding the energy needed for chemical treatment, it was assumed that the energy needed for this process was negligible by comparison.

Table 4: Energy requirement in the production of straight-run jet fuel

Refining process	Energy required (J/MJ product)
Crude desalting and atmospheric distillation	20,055
Chemical treatment	Assumed to be negligible
Total	20,055
Overall refining efficiency (LHV)	98%

Under these assumptions, the refining efficiency of straight-run jet fuel is about 98% (LHV). This refining efficiency likely represents the maximum efficiency for the production of jet fuel from conventional crude and was used in the low emissions scenario. The corresponding process fuel and fuel shares for the production of straight-run jet fuel are shown in Table 5.

Table 5: Process fuel and fuel shares for the production of straight-run jet fuel

Type of process fuel	J/MJ of jet fuel	Process fuel share (%)
Electricity	423	2.1
Natural Gas	5,772	28.8
Refinery Gas	9,669	48.2
Coke	3,548	17.7
Residual Oil	643	3.2
Total	20,055	100

Hydroprocessed Jet Fuel
The refining processes involved in producing jet fuel from hydroprocessing include crude desalting, atmospheric and vacuum distillation, hydrotreating and/or hydrocracking. Since the production of this hydroprocessed jet fuel was considered as the high emissions scenario, it was assumed that all the above processes were required (including both hydrotreating and hydrocracking). The energy needed for the refining processes to produce hydroprocessed jet fuel is shown in Table 6.

Table 6: Energy requirement in the production of jet fuel from hydroprocessing

Refining process	Energy required (J/MJ product)
Crude desalting and atmospheric distillation	20,055
Vacuum distillation	16,379
Hydrotreating (to S content of ~500ppm)	24,368
Hydrocracking	75,092
Total	135,894
Overall refining efficiency (LHV)	88%

The refining efficiency of hydroprocessed jet fuel is approximately 88% (LHV). As mentioned earlier, this refining efficiency was assumed in the high emissions scenario. The corresponding process fuel shares for the production of hydroprocessed jet fuel are shown in Table 7.

Table 7: Process fuel and fuel shares for the production of jet fuel from hydroprocessing

Type of process fuel	J/MJ of jet fuel	Process fuel share (%)
Electricity	9,137	6.7
Natural Gas	82,683	60.9
Refinery Gas	30,713	22.6
Coke	11,294	8.3
Residual Oil	2,067	1.5
Total	135,894	100

4.2.3   Impact of Crude Oil Quality Compared to Processing Technique

The scope of section 4.2 thus far has been limited to establishing a range of processing emissions for conventional jet fuel while assuming US average crude oil properties. As shown by Skone and Gerdes (2009), imported crude oils are on average heavier and contain higher levels of sulfur than domestic products. Skone and Gerdes used the API gravity and sulfur content of crude oils fed into US refineries to establish origin-specific processing GHG emissions for diesel fuel in 2005. Based on these data, the origin-specific processing GHG emissions per barrel of crude oil were calculated and subsequently related to origin-specific processing GHG emissions for jet fuel.[8] The upper and lower bounds on the variation from crude oil quality correspond to Mexican crude (API gravity of 23.8 and sulfur content of 3.0%) and Algerian crude (API gravity of 44.8 and sulfur content of 0.1%), respectively (Skone and Gerdes, 2009).

The variation in processing GHG emissions introduced from crude oil quality and processing technique are shown graphically in Figure 7. The variation in processing emissions resulting from processing techniques is 70% larger than the variation from crude oil quality. This analysis emphasizes that there are two factors that impact the processing emissions of making jet fuel. Although they have been considered separately, the impacts of crude oil quality and processing technique on processing emissions are not necessarily independent. Namely, the impact of crude oil quality on straight run fuel production may not be the same as its impact on hydroprocessed jet fuel.
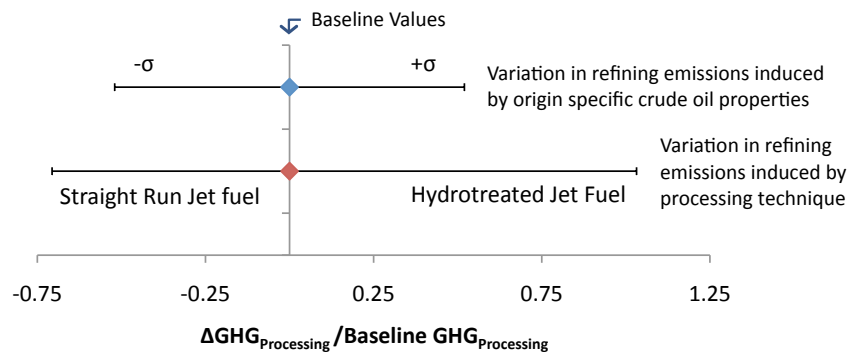
Figure 7: Variation in jet fuel processing emissions induced by origin specific crude oil properties and processing technique

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[8] This calculation was based on the assumption that the difference in processing emissions per barrel of diesel between each source and the average is equal to the difference in processing emissions per barrel of jet between each source and the average. Average processing emissions for diesel are 9.0 gCO_2e/MJ while those for jet are 5.65 gCO_2e/MJ.

4.2.4   Conventional Jet Fuel Results

The life cycle GHG emissions from the production of jet fuel from conventional crude are shown in Table 8. These results incorporate the recovery (crude extraction) and transportation results discussed in Section 4.1 to complete the life cycle GHG inventory. A comparison of the domestic results from this study with the average results presented by Skone and Gerdes (2008) is shown in the far right column of the table. Despite using a different approach to derive the GHG emissions in the processing of feedstock in the baseline case (top-down) from that used in the NETL study (bottom-up approach), similar results were obtained. These results assume average crude oil properties in all three scenarios; hence, they do not include any variation in processing emissions from crude oil quality. In addition, the combustion CO_2 equivalent emissions used by Skone and Gerdes are slightly higher than those calculated in this study. This is due to their estimates of CH_4 and N_2O emissions from jet fuel combustion. These emissions were excluded in this study due to the high level of uncertainty associated with their estimation. Overall, the life cycle GHG emissions of jet fuel from conventional crude obtained by NETL (88.0 gCO_2e/MJ) are about 0.7% higher than the baseline results (87.5 gCO_2e/MJ) developed herein.

Table 8: Summary of results for jet fuel from conventional crude and a comparison of results to the NETL petroleum baseline study

	MIT Conventional Jet Fuel			NETL
	Low	**Baseline**	**High**	**Baseline**
Key Assumptions				
Crude oil origin	US	Average	Nigeria	n/a
Processing Technique	Straight Run	Average	Hydro-processed	n/a
Refining efficiency (LHV)	98.0%	93.5%	88.0%	n/a
Life Cycle CO$_2$ Emissions by Stage				
Recovery of feedstock (gCO$_2$/MJ)	3.7	4.2	9.4	4.3
Transportation of feedstock (gCO$_2$/MJ)	0.8	1.5	1.8	1.3
Processing of feedstock to fuel (gCO$_2$/MJ)	1.6	5.5	11.0	5.5
Transportation of jet fuel (gCO$_2$/MJ)	0.8	0.8	0.8	0.9
Combustion CO$_2$ (gCO$_2$/MJ)	73.2	73.2	73.2	73.7
WTT GHG Emissions by Species				
WTT CO$_2$ emissions (gCO$_2$/MJ)	7.0	11.9	22.9	12.0
WTT CH$_4$ emissions (gCO$_2$e/MJ)	0.5	2.3	13.0	2.3
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.1	0.1	0.1	0.1
Total WTW GHG Emissions (gCO$_2$e/MJ)	80.7	87.5	109.3	88.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.92**	**1.00**	**1.25**	1.01

4.3   Ultra-Low Sulfur Jet Fuel from Conventional Crude

The production of ULS jet fuel from conventional crude oil was approached with the same techniques and assumptions that were applied to conventional jet fuel. Specifically straight-run fuel production and hydroprocessing of crude were considered for the low and high emissions scenarios, respectively. The additional hydroprocessing that would be required to desulfurize the petroleum stream to create the ULS jet fuel was modeled based on that required for ULS diesel. The required hydroprocessing likely exceeds simple hydrotreatment, and it may instead involve deep hydrogenation and/or hydrocracking.

Although the technology to produce a ULS jet fuel is known, it is important to note that the current hydroprocessing capacity in the United States is only sufficient to meet ULS diesel needs. It would need to be expanded to have sufficient capacity for both ULS jet and ULS diesel fuels. Such an expansion would require the construction of additional infrastructure, which was not included in the life cycle analysis.

4.3.1 Top-Down Approach (Baseline case)

General Motors et al. (2001) estimated a 2% energy penalty for reducing sulfur content in gasoline and diesel fuel from 350ppm to 5ppm. The baseline assumption in this work assumed the same 2% energy penalty applies for the production of ULS jet fuel compared to conventional jet fuel. Hence, the refining energy efficiency of ULS jet fuel was assumed to be 2% less than that of jet fuel, i.e. 91.5% (LHV). The process fuel shares outlined for conventional jet fuel (Table 3) were also used as inputs to the GREET model for the production of baseline ULS jet fuel.

4.3.2 Bottom-up approach (low and high emissions cases)

The bottom-up approach for ULS jet fuel builds upon that from the baseline case wherein the energy for the processes involved in refining jet fuel was taken from a 2007 Department of Energy sponsored report (Pellegrino et al., 2007). The report provided both a range of refining process energy use and average energy use. The average energy use data for the relevant refining processes were used to calculate the jet fuel refining efficiency.

Straight-Run ULS Jet Fuel

The production of straight-run ULS jet fuel requires crude desalting and atmospheric distillation followed by hydrotreatment to remove sulfur and other impurities. The estimated energy for these processes is shown in Table 9.

Table 9: Energy requirement in the production of straight-run ULS jet fuel

Refining process	Energy required (J/MJ product)
Crude desalting and atmospheric distillation	20,055
Hydrotreatment (to S content of ~5ppm)	48,184
Total	68,239
Overall refining efficiency (LHV)	93.5%

Hence, the refining efficiency of straight-run ULS jet fuel is about 93.5% (LHV). This refining efficiency was used in the ULS jet fuel low emissions scenario. The corresponding process fuel shares for the production of straight-run ULS jet fuel are shown in Table 10.

Table 10: Process fuel and fuel shares for the production of straight-run ULS jet fuel

Type of process fuel	J/MJ of jet fuel	Process fuel share (%)
Electricity	3,806	5.6
Natural Gas	42,570	62.4
Refinery Gas	15,235	22.3
Coke	5,603	8.2
Residual Oil	1,025	1.5
Total	68,239	100

Hydroprocessed ULS Jet Fuel

The calculation of the process energy required in the production of hydroprocessed ULS jet fuel is similar to that of jet fuel, except that additional hydrotreating is required to reduce the sulfur content of the fuel from about 500ppm to 5ppm. The energy needed for the refining processes is shown in Table 11.

Table 11: Energy requirement in the production of ULS jet fuel from hydroprocessing

Refining process	Energy required (J/MJ product)
Crude desalting and atmospheric distillation	20,055
Vacuum distillation	16,379
Hydrotreating (to S content of ~5ppm)	47,578
Hydrocracking	75,092
Total	159,104
Overall refining efficiency (LHV)	86%

The refining efficiency of hydroprocessed ULS jet fuel is about 86% (LHV). This refining efficiency was assumed in the high emissions scenario for the production of ULS jet fuel from conventional crude oil. The corresponding process fuel shares for the production of hydroprocessed ULS jet fuel are shown in Table 12.

Table 12: Process fuel and fuel shares for the production of ULS jet fuel from hydroprocessing

Type of process fuel	J/MJ of jet fuel	Process fuel share (%)
Electricity	10767	6.8
Natural Gas	100408	63.1
Refinery Gas	33394	21.0
Coke	12284	7.7
Residual Oil	2251	1.4
Total	159104	100

4.3.3 ULS Jet Fuel Results

The life cycle GHG resulting from the production of ULS jet fuel from conventional crude are shown in Table 13. These results incorporate the recovery and transportation results discussed in section 4.1 to complete the life cycle GHG inventory.

Table 13: Summary of results for ULS jet fuel from conventional crude pathway

	Low	Baseline	High
Key Assumptions			
Crude oil origin	US	Average	Nigeria
Processing Technique	Straight Run	Average	Hydro-processed
Refining efficiency (LHV)	93.5%	91.5%	86%
Life Cycle CO_2 Emissions by Stage			
Recovery of feedstock (gCO_2/MJ)	3.7	4.2	9.4
Transportation of feedstock (gCO_2/MJ)	0.8	1.5	1.8
Processing of feedstock to fuel (gCO_2/MJ)	5.5	7.3	13.1
Transportation of jet fuel (gCO_2/MJ)	0.8	0.8	0.8
Combustion CO_2 (gCO_2/MJ)	72.9	72.9	72.9
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	10.9	13.7	25.0
WTT CH_4 emissions (gCO_2e/MJ)	0.7	2.4	13.2
WTT N_2O emissions (gCO_2e/MJ)	0.1	0.1	0.1
Total WTW GHG Emissions (gCO_2e/MJ)	84.6	89.1	111.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.97**	**1.02**	**1.27**

In the baseline case, where a refining energy efficiency penalty of 2% was assumed in the production of ULS jet fuel relative to conventional jet fuel, the production of ULS jet fuel results in life cycle GHG emissions about 2% greater than those of conventional jet fuel. In the low

emissions case, the life cycle GHG emissions from the production of ULS jet fuel are decreased by 3% relative to the baseline conventional jet fuel. This effect arises from reduced origin-specific emissions relative to the baseline case and an additional energy intensive hydrotreating step compared to the production of straight-run conventional jet fuel. Conversely, the increase in life cycle GHG emissions between high emissions scenario ULS jet fuel and baseline conventional jet fuel is 27%. The energy penalty in this case results from further hydrotreating the jet fuel from about 500ppm to 5ppm (~2% difference in refining efficiency). The main reason driving the increase are the methane venting and transportation emissions associated with the Nigerian crude oil. In all cases, CO_2 emissions from combustion of ULS jet fuel are 0.3 gCO_2e/MJ less that conventional jet fuel.

5 Unconventional Petroleum Pathways

This manuscript considers two unconventional petroleum resources, oil sands from Alberta, Canada and oil shale from the Green River Formation in Colorado, Utah and Wyoming. While current production of oil sands is approximately 1.3 million bbl/day, oil shale development is presently limited to laboratory and pilot-plant stages (Hileman et al., 2009). Oil sands are deposits of tar-like petroleum, known as bitumen, within sand or porous rock while oil shale is a solid sedimentary rock that contains kerogen, a mixture of organic compounds, including bitumen, which can be refined to oil. Both of these resources require extensive work to convert the raw resource to a synthetic crude and this results in increased life cycle GHG emissions relative to conventional petroleum. All of the work regarding jet fuel from unconventional crude oil was performed with GREET version 1.8a.

5.1 Conventional Jet Fuel from Oil Sands

Alberta's oil sands resources constitute one of the world's largest proven oil reserves, second only to Saudi Arabia (Lacombe and Parsons, 2007). Bitumen production from oil sands takes place primarily by two methods, depending on the depth of the oil sand deposits. Relatively shallow deposits (75m or less) are recovered through surface mining, which involves the removal of soil and rock overlying the mineral deposit, mining of the oil sands ore and extraction of bitumen from oil sands using hot water. Deeper deposits employ in-situ production methods, which typically inject steam underground to reduce the viscosity of the bitumen and allow it to be pumped to the surface (AEUB, 2007). The bitumen produced can be upgraded to lighter synthetic crude oil (syncrude) through hydrogen-addition processes (e.g. hydrocracking) and/or carbon-subtraction processes (e.g. delayed coking).

The life cycle GHG emissions for the extraction of bitumen by both surface mining and in-situ technologies were considered. In particular, steam assisted gravity drainage (SAGD) was the main technology assumed for the in-situ production of oil sands. This technique has proven to be "technically feasible and economically attractive" for the extraction of bitumen oil sands deposits below 200 meters (Isaacs, 2007a). It was further assumed that all bitumen was upgraded to a light synthetic crude oil (syncrude).[9] Oil sands projects have traditionally used natural gas as the main source of process energy for bitumen extraction and upgrading as well as for producing hydrogen required for the upgrading process. This stems from the era when natural gas was cheap and abundant; however, this practice may not be sustainable as bitumen production expands. To reduce the dependence on natural gas, alternative process energy and hydrogen sources are being explored. These include coal combustion, coal gasification, nuclear energy, and the use of internally generated fuels such as bitumen residues (ACR, 2004). Additional energy demands are also being met using diesel, gasoline and grid electricity.

5.1.1 Bitumen Production through Surface Mining

The GHG emissions from bitumen production through surface mining are largely dependent on the quantity and source of the process energy. This work explored the use of natural gas and coal combustion to generate the steam and electrical power required for bitumen extraction. The total process energy data for surface mining were taken from the average values of individual industry projects (Deer Creek Energy, 2006; Synenco Energy, 2006; Shell Canada, 2007a). The process energy sources and values for the production of bitumen through surface mining for the low emissions, baseline and high emissions scenarios are summarized in Table 14.

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[9] It was assumed that about 1,600 standard cubic feet of hydrogen was used per barrel of syncrude produced through bitumen upgrading. The quality of syncrude produced (~35-49 °API) is generally lighter than the average crude oil received by US refineries (~30 °API). However, syncrude tends to produce lower quality distillate oil due to its low hydrogen content and requires more complex refining operations to produce transportation fuels. This report assumed that the refining efficiency of jet fuel from the lighter, but lower hydrogen content syncrude was the same as that from using heavier crude oil with higher hydrogen content.

Table 14: Process energy assumptions for the production of bitumen through surface mining for the low emissions, baseline and high emissions cases

Parameter	Low	Baseline	High
Process energy for steam and power generation (J/MJ bitumen)	52058	26029 (NG) 26029 (coal)	52058
Source of process energy	NG	NG and coal	Coal
Grid electricity (J/MJ bitumen)	3213	3213	3213
Diesel (J/MJ bitumen)	10723	10723	10723
Gasoline (J/MJ bitumen)	319	319	319
Total process energy required (J/MJ bitumen)	66313	66313	66313

5.1.2   Bitumen Production through In Situ Recovery

The efficiency of bitumen recovery from oil sands using in situ recovery techniques is characterized by the steam-to-oil ratio (SOR). SOR is the volume of steam required to extract a given volume of bitumen. The SOR of SAGD technology ranges from 2.0 to greater than 3.0, (Nieuwenburg, 2006; Lacombe and Parsons, 2007) with a 2006 average of about 2.5 (Canadian National Energy Board, 2006). The lower and upper bounds were adopted in the low and high emissions scenarios while the average was used in the baseline scenario. These assumptions translate to natural gas usages of 840 scf, 1050 scf and 1260 scf per barrel of bitumen in the low baseline and high emissions scenarios, respectively.[10] Other energy sources, such as coal, can also be used to supply an equivalent amount of energy. Electricity used in the in-situ production of bitumen was consistent with the average value provided by the National Energy Board (2006).

In the low and baseline cases, it was assumed that the process energy for steam generation was provided by natural gas, and in the high case, by coal. The process energy values for the production of bitumen through in-situ production for the three scenarios are summarized in Table 15.

Table 15: Process energy assumptions for the production of bitumen through in situ recovery for the low emissions, baseline and high emissions cases

Parameter	Low	Baseline	High
SOR (steam-to-oil ratio)	2.0	2.5	3.5
Process energy needed to make steam (J/MJ bitumen)	138544	173180	242451
Source of steam energy	NG	NG	Coal
Grid electricity (J/MJ bitumen)	5700	5700	5700

5.1.3   Bitumen Upgrading to Syncrude

The parameters analyzed in the upgrading of bitumen to syncrude included the yield of syncrude, amount of hydrogen used for the upgrading process, the source of hydrogen and the process energy. The yield of syncrude from bitumen upgrading can vary depending on the upgrading processes. The average volumetric yield is about 1 barrel of syncrude per 1.16 barrel of bitumen input, or 1 MJ of syncrude for every 1.27 MJ of bitumen input (EIA, 2006).[11] This yield formed the baseline upgrading analysis of this work. In the low scenario, it was assumed that 1.1 MJ of bitumen input is required per MJ of syncrude, based on the Shell Scotford upgrading project (Shell Canada, 2007b). The high scenario assumed that 1 MJ of syncrude was produced from 1.36 MJ of bitumen input based on the Synenco Energy Northern Lights Upgrading project.

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[10] Assuming that about 420 standard cubic feet of natural gas is needed to make a barrel of steam (Lacombe and Parsons, 2007).

[11] Assuming that the LHV of bitumen is 39.5 MJ/L (Shell Canada, 2007b), and the LHV of syncrude is 36.1 MJ/L (from Table 97).

In the low and baseline cases, it was assumed that 1626 scf of hydrogen was needed per barrel of syncrude produced (Isaacs, 2007b, ACR, 2004)[12] and that natural gas was the source of the hydrogen. In the high case, 1800 scf of hydrogen was assumed to be used per barrel of syncrude produced, as predicted by the Alberta Chamber of Resources as the amount needed to produce high quality light syncrude in the future (ACR, 2004). It was further assumed that this hydrogen was produced from the gasification of asphaltene, a by-product of the bitumen upgrading process, with no carbon capture.[13] A sensitivity analysis was conducted on the high emissions case to examine the impact of carbon capture and sequestration on the CO_2 resulting from the asphaltene gasification.[14]

Natural gas and electricity are the main process energy requirements for the upgrading processes. The range of values used for the low to high cases is based on data from the Alberta Chamber of Resources (baseline case) and individual industry upgrading projects (Low Case: Sturgeon upgrader; High case: Northern Lights upgrader[15]) (ACR, 2004; Petro-Canada, 2006; Synenco Energy, 2006). The parameters used in analyzed in the upgrading of bitumen to syncrude are summarized in Table 16.

Table 16: Assumed parameters in the upgrading of bitumen to syncrude

Parameter	Low	Baseline	High
Yield of SCO (MJ bitumen per MJ syncrude produced)	1.1	1.27	1.36
Amount of H_2 needed (J/MJ syncrude)	86468	86468	98780
Source of H_2	NG	NG	asphaltene gasification
Process energy (NG) (J/MJ syncrude)	4747	14400	25207
Electricity (J/MJ syncrude)	8323	4200	34846
Jet fuel refining efficiency from syncrude (%)	93.5	93.5	93.5

5.1.4   Oil Sands to Jet Fuel Results

The life cycle GHG emissions from the production of jet fuel from Canadian oil sands using surface-mining and in-situ production are shown in Table 17.

Carbon-capture and sequestration (CCS) of emissions from the gasification of asphaltene for hydrogen production was examined as a sensitivity study to the high emissions scenario of Table 15. Only emissions from the upgrading of bitumen to syncrude are affected by the use of carbon capture on the hydrogen production process. The results for the high emissions scenario of surface and in situ production of bitumen are shown with and without CCS in Table 18.

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[12] From ACR (2004) ~0.4 volume units of natural gas are needed to produce a volume unit of hydrogen.

[13] It is assumed that there are no energy use or emissions associated with the production of asphaltene, which is a by-product of bitumen upgrading. The efficiency of asphaltene gasification is also assumed to be the same as that of coal gasification (default GREET value of 62%).

[14] A 90% efficiency in the capture of carbon emitted from the gasification of asphaltene and carbon capture energy requirement of 250 kwh/ton C are assumed.

[15] In addition to syncrude, the Northern Lights upgrader project also produces butane and excess hydrogen. The emission credit for the butane co-product was ignored because it is negligible compared to the other well-to-tank emissions (~1%). It was also assumed that the emissions associated with asphaltene gasification to produce any excess hydrogen are offset by the emissions credit given when this hydrogen displaces conventional hydrogen on the market. The net effect is to ignore the excess hydrogen produced in the process. Such assumptions are deemed appropriate for the high emissions scenario.

Table 17: Life cycle GHG emissions for jet fuel from Canadian oil sands using surface-mining and in-situ recovery

	Low		Baseline		High	
	Surface	In situ	Surface	In situ	Surface	In situ
Life Cycle CO$_2$ Emissions by Stage						
Recovery of feedstock (gCO$_2$/MJ)	14.6	19.8	16.5	23.9	30.1	54.4
Transportation of feedstock (gCO$_2$/MJ)	1.3	1.3	1.3	1.4	1.5	1.7
Processing of feedstock to fuel (gCO$_2$/MJ)	5.5	5.5	5.5	5.5	5.5	5.5
Transportation of jet fuel (gCO$_2$/MJ)	0.5	0.5	0.5	0.6	0.6	0.7
Combustion CO$_2$ (gCO$_2$/MJ)	73.2	73.2	73.2	73.2	73.2	73.2
WTT GHG Emissions by Species						
WTT CO$_2$ emissions (gCO$_2$/MJ)	21.9	27.1	23.8	31.4	37.6	62.3
WTT CH$_4$ emissions (gCO$_2$e/MJ)	2.7	3.1	2.7	3.5	2.5	3.3
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.1	0.1	0.1	0.1	0.1	0.2
Total WTW GHG Emissions (gCO$_2$e/MJ)	97.9	103.6	99.8	108.2	113.5	139.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.12**	**1.18**	**1.14**	**1.24**	**1.30**	**1.59**

Table 18: Life cycle GHG emissions for the high emissions scenario of jet fuel from Canadian oil sands with and without carbon capture and sequestration of emissions from asphaltene gasification

	Surface Mining		In situ recovery	
High Emissions Scenario:	w/o CCS	w/ CCS	w/o CCS	w/ CCS
Life Cycle CO$_2$ Emissions by Stage				
Recovery of feedstock (gCO$_2$/MJ)	30.1	19.4	54.4	43.8
Transportation of feedstock (gCO$_2$/MJ)	1.5	1.3	1.7	1.6
Processing of feedstock to fuel (gCO$_2$/MJ)	5.5	5.5	5.5	5.5
Transportation of jet fuel (gCO$_2$/MJ)	0.6	0.5	0.7	0.7
Combustion CO$_2$ (gCO$_2$/MJ)	73.2	73.2	73.2	73.2
WTT GHG Emissions by Species				
WTT CO$_2$ emissions (gCO$_2$/MJ)	37.6	26.7	62.3	51.5
WTT CH$_4$ emissions (gCO$_2$e/MJ)	2.5	2.5	3.3	3.3
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.1	0.1	0.2	0.2
Total WTW GHG Emissions (gCO$_2$e/MJ)	113.5	102.6	139.0	128.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.30**	**1.17**	**1.59**	**1.47**

In developing a single value for each of the scenarios, the proportion of bitumen extracted through surface-mined oil sands versus that of bitumen extracted through in-situ production for processing into jet fuel was varied. Specifically, the low case assumes that all bitumen was extracted from surface-mined oil sands and the high case assumes all bitumen was extracted entirely through in-situ processes. The Alberta Energy and Utilities Board (AEUB) forecast that in 2015, about 56.5% of crude bitumen production will come from surface mining and the remaining 43.5% from in-situ production (AEUB, 2007). These proportions were adopted in the baseline case. The inputs for the scenario analysis are summarized in Table 19.

Table 19: Assumptions regarding the use of surface and in situ bitumen recovery in the low emissions, baseline, and high emissions scenarios of jet fuel from Canadian oil sands

	Low	Baseline	High
Proportion of bitumen used in jet fuel production from surface-mining processes	100%	56.5%	0%
Proportion of bitumen used in jet fuel production from in-situ processes	0%	43.5%	100%

The corresponding life cycle GHG emissions from producing jet fuel from Canadian oil sands are shown in Table 20. Recall that the low emissions value represents the low case for surface mining while the high emissions value represents the high case for in-situ recovery.

Table 20: Summary of result for jet fuel from Canadian oil sands

	Low	Baseline	High
Life Cycle CO$_2$ Emissions by Stage			
Recovery of feedstock (gCO$_2$/MJ)	14.6	19.7	54.4
Transportation of feedstock (gCO$_2$/MJ)	1.3	1.3	1.7
Processing of feedstock to fuel (gCO$_2$/MJ)	5.5	5.5	5.5
Transportation of jet fuel (gCO$_2$/MJ)	0.5	0.5	0.7
Combustion CO$_2$ (gCO$_2$/MJ)	73.2	73.2	73.2
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	21.9	27.1	62.3
WTT CH$_4$ emissions (gCO$_2$e/MJ)	2.7	3.0	3.3
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.1	0.1	0.2
Total WTW GHG Emissions (gCO$_2$e/MJ)	97.9	103.4	139.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.12**	**1.18**	**1.59**

The life cycle GHG emissions of the production and use of jet fuel from Canadian oil sands range from 1.12 to 1.59 times higher than those from the production and use of conventional jet fuel.

This analysis has only considered the use of natural gas as a process fuel and source of hydrogen. Although natural gas is currently the main source of energy for oil sands production, its use may not be sustainable in the long term. To reduce dependence on natural gas, coal and asphaltene (a bitumen residue) were considered as an alternative energy sources. Though abundant, the use of coal and bitumen upgrading residues can result in greater GHG emissions compared to the use of natural gas. Specifically, using coal energy to power bitumen production and asphaltene gasification to provide hydrogen for bitumen upgrading, the life cycle GHG emissions from surface-mining and in-situ production are 1.3 times and 1.6 times greater than those of conventional jet fuel, respectively. Even when carbon capture is used to reduce emissions from the gasification of asphaltene, the life cycle GHG emissions from surface-mining and in-situ production are still about 1.2 times and 1.5 times greater than those of conventional jet fuel, respectively.

5.2   Conventional Jet Fuel from Oil Shale

Oil shale is a sedimentary rock formation of mostly carbonate and silicate minerals containing a solid organic material called kerogen. This analysis of jet fuel production from oil shale is based on the Shell in situ conversion process (ICP), in which oil shale is heated while still in the earth using electrically heated wells for a prolonged period (two to three years). The heat releases a combination shale oil and shale gas from the kerogen in the oil shale.[16] Shale oil is a light crude composed largely of middle distillates while shale gas is mixture of natural gas, propane and butane. The yield relationship between the two products is estimated as two-thirds shale oil and one-third shale gas on an energy basis. The area surrounding the extraction site is frozen to form an impermeable barrier, preventing groundwater from disturbing the heating process and preventing products from escaping (Bartis et al., 2005; Mut, 2005).

Ex situ conversion is the alternative to the Shell ICP. Ex situ heating of the kerogen occurs above ground and requires the shale to be mined before it is processed (Brandt, 2008). The conversion

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[16] Note that *oil shale* is the geologic rock formation that contains kerogen, whereas *shale oil* is the light crude that is released from the oil shale.

of oil shale to shale oil and shale gas on the surface takes place at higher temperatures than the in situ technique. The higher temperatures could result in the decomposition of the carbonate minerals contained within the oil shale, potentially releasing considerably more CO_2 than the Shell ICP process.[17]

5.2.1 Analysis Methodology

The pathway for the extraction of shale oil from oil shale using the Shell ICP is not available in GREET and was analyzed based on incorporating information available from the literature into the GREET framework. Specifically, the process energy for the production of shale oil from oil shale, as well as the yield of oil and gas products using the Shell ICP, were adopted from Brandt's analysis of the Green River Formation in Colorado, Utah, and Wyoming (Brandt, 2008). This analysis was specifically focused on the Green River Formation; however, many other oil shale resources are dispersed around the world.[18]

Only major process energy demands, namely the in situ heating energy and the energy to maintain the frozen wall (about one order of magnitude less than the in situ heating energy) were considered.[19] The effects of different amounts and sources of in-situ heating energy on the overall GHG emissions of this pathway were explored with the low emissions, baseline and high emissions scenarios. The low and baseline cases assumed that 25% of the in situ heating energy needed was provided through the recycling of waste heat (as was assumed in the low carbon case in Brandt's analysis), while no recycling of waste heat was assumed in the high case. As a large amount of electrical energy is needed to provide in-situ heating for the Shell ICP, dedicated coal-based power generation facilities were assumed to be near the extraction site due to the vast abundance of coal resources in the Green River Formation. The locality of electricity was assumed to reduce transmission losses to 5% from the GREET default value of 8%. For the low case, the use of coal IGCC electricity with carbon capture (90% efficiency) was assumed;[20] while the baseline and high cases employ traditional pulverized coal-fired electricity without carbon capture.

Brandt (2008) provided low and high estimates of the electrical energy required for in situ heating, the amount of shale oil output and the amount of shale gas co-produced. There is no natural gas consumption within the process; therefore, energy and emission credits were given to the natural gas co-produced using the displacement method.[21] The low emissions scenario used the low value for in situ heating energy with high values for shale oil and shale gas production; the baseline scenario used mean values; and the high emissions scenario used high value for in situ heating energy with low values for shale oil and shale gas production. In addition, shale oil is much lighter and contains almost no heavy ends compared to traditional crude oil; hence, the

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[17] Typical oil shale from the Green River Formation is composed of 23% dolomite (calcium-magnesium bicarbonate) and 16% calcite (calcium carbonate). Surface conversion processes require up to 750°C while the decomposition of dolomite and calcite occur around 575°C and 650°C, respectively. The Shell ICP process takes place at temperatures between 340°C and 400°C. Because of this increased temperature, production of shale oil using surface retorting could result in between 1.2 to 3 times more GHG emissions than if the production had been performed using the Shell ICP process. (Hileman et al., 2009)

[18] There are known oil shale formations in Australia, Brazil, Canada, China, Estonia, Israel, Jordan, Morocco, Russia, Sweden, Syria, Thailand, Turkey and the United States (Geology.com, 2009)

[19] Other minor process energy requirements like drilling and pumping energy (< 1% of retorting energy), as well as energy needed for infrastructural construction, are ignored in this study.

[20] From Deutch and Moniz (2007), the estimated efficiency of an IGCC plant with 90% carbon capture is 31.2% (HHV) or about 34% (31.2+3) on a LHV basis. The efficiency of 34% (LHV) was adopted in this study.

[21] The displacement (system expansion) method was used to account for energy and emission credits to natural gas, i.e., the natural gas produced was assumed to displace the recovery and processing of conventional petroleum based natural gas in a separate, independent facility.

refining efficiency of processing shale oil to jet fuel was assumed to be higher (~2-3%) than that of refining traditional crude oil.[22]

The key processes and assumptions involved in the production of jet fuel from oil shale using the Shell ICP are summarized in Table 21.

Table 21: Input assumptions for the production of jet fuel from oil shale for low emissions, baseline and high emissions cases

	Low	Baseline	High
Process conditions	Use 25% recycled heat for retorting; use coal IGCC electricity with CCS; capture efficiency of 90%	Use 25% recycled heat for retorting; use 100% coal-fired electricity; no carbon capture	No recycled heat used; 100% coal-fired electricity; no carbon capture
Electrical energy input (J/MJ shale oil produced)	134600	148100	211900
Natural gas co-produced (J/MJ shale oil produced) (LHV)	223700	189600	152500
Refining efficiency (LHV)	96%	96%	96%

5.2.2 Oil Shale to Jet Fuel Results

The GHG emissions from the production and use of jet fuel from oil shale using the Shell ICP are shown in Table 22.

Table 22: Summary of results for jet fuel from in situ oil shale pathway

	Low	Baseline	High
Life Cycle CO$_2$ Emissions by Stage			
Recovery of feedstock (gCO$_2$/MJ)	3.6	41.2	59.7
Transportation of feedstock (gCO$_2$/MJ)	0.5	0.6	0.7
Processing of feedstock to fuel (gCO$_2$/MJ)	3.3	3.3	3.3
Transportation of jet fuel (gCO$_2$/MJ)	0.5	0.6	0.7
Combustion CO$_2$ (gCO$_2$/MJ)	73.2	73.2	73.2
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	7.8	45.8	64.4
WTT CH$_4$ emissions (gCO$_2$e/MJ)	2.4	2.5	3.2
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.6	0.2	0.2
Total WTW GHG Emissions (gCO$_2$e/MJ)	84.1	121.6	141.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.96**	**1.39**	**1.61**

With carbon capture (low emissions case), the life cycle GHG emissions of jet fuel from oil shale are reduced to slightly less than baseline conventional jet fuel. Without the capture of carbon dioxide from coal-based electricity plants providing the in situ heating energy, the production of jet fuel from oil shale produces life cycle GHG emissions 1.4 to 1.6 times greater than baseline conventional jet fuel.

In addition to increased GHG emissions, oil shale development also presents other adverse impacts to the environment. Though less intrusive to the surface topography than ex situ processes and not requiring the disposal of spent shale, in-situ conversion will still cause displacement of all other land uses in the area and disruptions to the local ecological community

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[22] Jim Bartis, interview with Hsin Min Wong, July 16, 2007.

(Bartis et al., 2005). In situ methods also have the potential to cause ground water contamination. Though the freeze wall protects groundwater during production, contamination may occur post-production. As the Green River formation lies within the Colorado River drainage basin, water contamination could impact millions of downstream users (Gruenwald, 2006).

6 Fischer-Tropsch Jet Fuel

The Fischer Tropsch (F-T) process first involves the steam reforming or gasification of any carbon containing feedstock (e.g. natural gas, coal or biomass) to synthesis gas (syngas), which is a mixture of hydrogen and carbon monoxide. The syngas is subsequently converted to paraffinic hydrocarbons in the presence of an iron- or cobalt-based catalyst (Fischer-Tropsch synthesis). A third upgrading step cracks the longer hydrocarbon chains to maximize the production of synthetic paraffinic liquid fuels like diesel and jet fuel. Syngas must be cleaned before Fischer-Tropsch synthesis step to remove contaminants, particularly sulfur, to avoid poisoning the catalyst. Hence, the resultant Fischer-Tropsch liquid fuels are virtually free of contaminants and the jet fuel fraction of the product slate falls into the category of synthetic paraffinic fuels.

All jet fuels produced using F-T synthesis have similar characteristics, independent of feedstock type. Any small variations in fuel properties are primarily associated with the operating conditions (e.g., catalyst, temperature, and pressure) within the synthesis reactors and how the direct products of the synthesis are treated and processed. All jet fuels produced using the F-T process share common characteristics with regard to compatibility with existing infrastructure and aircraft, combustion emissions, and their relative merit for use in aviation. Feedstock choice, however, does have a strong influence on fuel production capacity, production cost, life cycle greenhouse gas emissions, and technology readiness (Hileman et al., 2009).

Fischer-Tropsch fuels created from natural gas, coal, biomass and combinations of coal and biomass were analyzed in this work. Gas-to-Liquid (GTL) production is currently limited to Malaysia where Shell has been producing approximately 15,000 bbl/day since 1993. Sasol as well as Shell in collaboration with Qatar Petroleum are both constructing GTL facilities in Qatar with design capacities of 34,000 bbl/day and 140,000 bbl/day, respectively. Existing coal-to-liquids (CTL) capacity is limited to Sasol in South Africa where a production capacity of 160,000 bbl/day of oil equivalent has been consistently maintained. There is no commercial scale production of F-T fuels using biomass as the feedstock (BTL). This technology is still in the development phase; however, a German firm, CHOREN, began start-up operations of a 300 bbl/day facility in 2008 and Solena Group, with Rentech, announced plans for a 1,800 bbl/day BTL facility located in Gilroy, California. Experience with simultaneously gasifying a combination of coal and biomass in a single gasifier is presently limited to successful tests at an IGCC plant in the Netherlands (Hileman et al., 2009). The analysis of F-T jet fuel from coal and natural gas was conducted using GREET version 1.8a while F-T jet fuel from biomass and the combination of coal and biomass was considered using GREET version 1.8b.

6.1 Carbon Capture and Sequestration with Fischer-Tropsch Facilities

By comparison to conventional power plants, F-T plants are well suited for the implementation of carbon capture and sequestration (CCS). This opportunity occurs because the F-T process results in a relatively pure stream of CO_2 upstream of the F-T synthesis step, whereas coal power plants produce flue gas, which must be scrubbed to obtain a pure CO_2 stream.

This work considered cases with CCS implemented on a gas-to-liquids plant, a coal-to-liquids plant and a coal and biomass-to-liquids plant. Carbon dioxide is already captured within F-T plants as part of the process; therefore, the only difference for standard CCS implementation is the addition of CO_2 compression, transport and storage capital and operating costs (Tarka, 2009). It was assumed that the energy needed for CO_2 compression was provided by electricity internally generated within the F-T process. This results in a reduction of the overall process efficiency because some of the energy from the feedstock is used to generate this additional energy. The electrical energy needed to compress the captured carbon dioxide to a pipeline

ready pressure of about 15 MPa was assumed to be 250 kWh/ton carbon[23] (GREET, 2008; Kreutz et al., 2008). The transportation of compressed carbon dioxide and the energy required for sequestration in a storage site were not considered.

The amount of carbon available for capture is equal to the difference between the carbon present in the feedstock and the carbon present in the final products. The term carbon capture efficiency is used to define the percentage of available carbon that is actually captured. Capture efficiencies of 80% (Kreutz et al., 2008), 85% (SSEB, 2006) and 90% (Bartis et al., 2008; Deutch and Moniz, 2007; Tarka, 2009) were assumed for the high emissions, baseline and low emissions cases, respectively. Recent analyses at National Energy and Technology Laboratory (NETL) have indicated that capital expenditures associated with systems designed to capture CO_2 at the 80% level would not vary significantly from those capturing at the 90% level (Dilmore and Skone, 2009). A capture efficiency of 85% was chosen for the baseline case in this work to reflect the lack of commercial experience surrounding widespread implementation of CCS. Carbon emitted from the combustion of process fuels is not captured.

6.2 Fischer-Tropsch Fuel from Natural Gas

The life cycle GHG emissions for the production of F-T fuels using natural gas sourced outside North America were considered. The Gas to Liquids (GTL) pathway is available in GREET and well-documented based on current industry data (Wang, 1999; Wang and Huang, 1999; Wang, 2002). Specifically, this analysis assumed non-North American natural gas was supplied to an internationally located stand-alone F-T liquid fuels plant designed to maximize liquid fuels production (e.g. through recycling of tail gas from F-T reactors). Sufficient electrical energy was produced to fuel all internal processes, with negligible excess electricity produced for export. The process included the upgrading (hydroprocessing) of long-chain liquids to a final product slate of diesel (C_{18}), jet fuel (C_{12}) and naphtha (C_4-C_6). The energy allocation method was adopted for assigning energy and emissions to various liquid products. Default GREET assumptions were adopted regarding the transportation profiles of F-T fuel from international plants to the US (5000 miles by ocean tanker plus domestic transport via pipeline, barge and rail).

Most studies in the literature focus on F-T reactor designs and conditions that produce diesel and naphtha. To produce jet fuel instead of diesel, additional hydrocracking and greater syngas recycle are needed, resulting in a small increase in hydrogen and power requirements for the plant. Furthermore, a moderate decrease in the CO_2 associated to jet fuel compared to diesel would ensue due to changes in the allocation fractions. As these additional energy requirements do not lead to substantial increases in CO_2 emissions from the facility (Gray et al., 2007), they were ignored in this analysis, (i.e. the production of F-T jet fuel is assumed to have the same emissions as the production of F-T diesel). This assumption applies to all of the F-T jet fuel pathways analyzed in this study, namely the production of F-T jet fuel from natural gas, coal and biomass. Although F-T jet fuel can be made without added burdens, it is not possible to have a product slate of 100% F-T jet fuel[24] (a value of 25% is taken as the preferred value in these studies, and a sensitivity study is shown for a co-fed coal and biomass F-T facility).

The F-T process efficiency is a key parameter affecting the life cycle GHG emissions of the production of F-T jet fuel from natural gas. Equation 4 defines process efficiency for a general F-T facility. GTL process efficiencies from the literature vary from 60 to 65% (Edwards et al., 2007;

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[23] Kreutz et al. assume 90.5 kWh per tonne of CO_2, which converts to 300 kWh per ton of carbon. GREET default value in 2010 is 300 kWh per ton of carbon but an improved efficiency of 250 kWh per ton of carbon is assumed for 2015.

[24] An F-T plant configured to produce 70% diesel and 30% naphtha should theoretically be able to undergo modifications such that it could yield 60% jet fuel and 40% naphtha (Gray et al., 2007). Sasol is developing the ability to produce a joint Battlefield-Use Fuel of the Future (BUFF) using F-T synthesis. This fuel could be used in place of JP-8 in military aircraft and they report a yield of ~30% that conforms to the freezing point standards of JP-8. The rest of the product slate is composed of "heavy" diesel and naphtha (Lamprecht, 2007).

Green Car Congress, 2006a; Wang, 2002). The low emissions, baseline and high emission scenarios assumed LHV process efficiencies of 65%, 63% and 60%, respectively.

$$\text{Process Efficiency} = \frac{1\ \text{MJ}_{\text{fuel}}}{1\ \text{MJ}_{\text{feedstock}} + \text{Process Energy}} \qquad \text{Equation 4}$$

The important assumptions and corresponding life cycle GHG emissions in the production and use of F-T jet fuel from natural gas are shown in Table 23.

Table 23: Summary of inputs and results for F-T jet fuel from natural gas pathway

	Low	Baseline	High
Key Assumptions			
Process efficiency (LHV)	65%	63%	60%
Life Cycle CO$_2$ Emissions by Stage			
Recovery of feedstock (gCO$_2$/MJ)	4.6	4.6	4.6
Transportation of feedstock (gCO$_2$/MJ)	0.0	0.0	0.0
Processing of feedstock to fuel (gCO$_2$/MJ)	19.4	20.2	21.4
Transportation of jet fuel (gCO$_2$/MJ)	1.2	1.2	1.2
Combustion CO$_2$ (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	25.2	26.0	27.2
WTT CH$_4$ emissions (gCO$_2$e/MJ)	4.4	4.6	4.8
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.0	0.0	0.0
Total WTW GHG Emissions (gCO$_2$e/MJ)	100.1	101.0	102.4
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.14**	**1.15**	**1.17**

The life cycle GHG emissions of the production and use of F-T jet fuel made from natural gas range from 1.14 times to 1.17 times higher than those from conventional petroleum-based jet fuel.

6.2.1 Case Study: Impact of Carbon Capture on GHG Emissions from GTL Facilities

A distinguishing factor of GTL facilities is the reduced CO$_2$ emitted as compared to CTL and BTL. From the perspective of the operator this could potentially lead to higher capture costs per ton of carbon; hence the discussion of CCS with a GTL plant is often overlooked. Nonetheless, this work explored the effect of capturing CO$_2$ produced from GTL plants in the baseline scenario configuration. A capture efficiency of 85% was assumed for consistency with the baseline assumptions in the other F-T pathways.

A comparison of results from the baseline scenario with and without CCS is given in Table 24. CCS could reduce the processing GHG emissions of F-T jet fuel from natural gas by 15 gCO$_2$/MJ, such that the WTT emissions from the GTL pathway become comparable to those from conventional jet fuel. As is the case with all fossil based fuels, combustion emissions dominate the life cycle GHG inventory, making it difficult to improve upon conventional jet fuel.

Table 24: Life cycle GHG emissions from natural gas F-T jet fuel with and without carbon capture

	Baseline	Baseline with CCS
Key Assumptions		
Carbon capture efficiency	--	85%
Process Efficiency (LHV)	63%	62.7% (63% without CCS)
Life Cycle CO_2 Emissions by Stage		
Recovery of feedstock (gCO_2/MJ)	4.6	4.6
Transportation of feedstock (gCO_2/MJ)	0.0	0.0
Processing of feedstock to fuel (gCO_2/MJ)	20.2	5.4
Transportation of jet fuel (gCO_2/MJ)	1.2	1.2
Combustion CO_2 (gCO_2/MJ)	70.4	70.4
WTT GHG Emissions by Species		
WTT CO_2 emissions (gCO_2/MJ)	26.0	11.2
WTT CH_4 emissions (gCO_2e/MJ)	4.6	4.6
WTT N_2O emissions (gCO_2e/MJ)	0.0	0.0
Total WTW GHG Emissions (gCO_2e/MJ)	101.0	86.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.15**	**0.99**

6.3 Fischer-Tropsch Fuel from Coal

This examination of life cycle GHG performance of F-T jet fuel production from coal was conducted for scenarios both with and without carbon capture and sequestration. Similarly to GTL plants, a stand-alone F-T liquid fuels plant designed to maximize liquid fuels production with no excess electricity was examined. The upgrading (hydroprocessing) of long-chain liquid products to desired fuel products was also included. Energy allocation was used to assign energy and emissions among liquid fuel products.

6.3.1 CTL without Carbon Capture

Process efficiency and coal type were judged to be key parameters having a significant impact on the overall GHG emissions of the Coal to Liquids (CTL) pathway. Both bituminous and sub-bituminous coal based processes were examined.[25] In 2007, almost 65% of bituminous coal production in the US occurred via underground mining while the remainder was surface-mined. On the other hand, sub-bituminous coals are almost exclusively surface-mined. (EIA, 2008b) Underground mining results in substantially greater methane emissions compared to surface mining processes. Methane emissions arising from the production of underground-mined and surface-mined coals were estimated using aggregate coal mining methane emissions data (EIA, 2007; Bartis et al., 2008).[26]

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[25] Lignite coal can possibly be used as a feedstock to CTL plants but lignite production in the US is much lower (~7% of total coal production in 2007, EIA 2008c) compared to bituminous and sub-bituminous coal production (46.7% and 46.3% of total coal production in 2007, respectively, EIA 2008c). Reliable data on methane emissions associated with the mining of lignite coal are not available. For these reasons, lignite coal is not analyzed in this work.

[26] Bartis et al., 2008 estimated methane emissions of 338 pounds of carbon dioxide equivalent per ton of underground-mined coal and methane emissions of 42.4 pounds of carbon dioxide equivalent per ton of surface-mined coal based on EIA data (EIA, 2008b). The methane emissions per MJ of coal production were calculated from the lower heating values of bituminous coal and sub-bituminous coal.

The low emissions case assumed a lower heating value process efficiency of 53% with surface-mined bituminous coal.[27] In the baseline case, a process efficiency of 50% was used (Bartis et al., 2008; Deutch and Moniz, 2007; Van Bibber et al., 2007; Marano and Ciferno, 2002) with the 2007 average US coal mix of underground-mined and surface-mined bituminous coal and surface-mined sub-bituminous coal[28]; the use of anthracite or lignite coal was not considered in this analysis. The high emissions case assumed a process efficiency of 47% and underground-mined bituminous coal (based on case 1 of Southern States Energy Board CTL study, SSEB, 2006). For comparison, the first Sasol CTL plant built in the 1950s had process efficiencies under 40% (UK DTI, 1999; Gray and Tomlinson, 2001).

The inputs assumed for the three scenarios are summarized in Table 25.

Table 25: Input assumptions for the production of F-T jet fuel from coal (without carbon capture) for low emissions, baseline and high emissions cases

	Low	**Baseline**	**High**
Process efficiency (LHV)	53%	50%	47%
Coal input	Surface-mined bituminous coal	Average bituminous and sub-bituminous coal mix[29]	Underground-mined bituminous coal
Coal mining methane emissions (g CO_2e/MJ coal)	0.80	2.8	6.4

The life cycle GHG emissions from the production and use of F-T jet fuel from a CTL plant without carbon capture and sequestration are shown in Table 26.

Table 26: Summary of results for F-T jet fuel from coal pathway (without carbon capture)

	Low	**Baseline**	**High**
Life Cycle CO_2 Emissions by Stage			
Recovery of feedstock (gCO_2/MJ)	0.8	0.8	0.8
Transportation of feedstock (gCO_2/MJ)	0.1	0.1	0.0
Processing of feedstock to fuel (gCO_2/MJ)	100.5	117.2	122.4
Transportation of jet fuel (gCO_2/MJ)	0.6	0.6	0.6
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	102.1	118.8	123.9
WTT CH_4 emissions (gCO_2e/MJ)	1.5	5.7	13.7
WTT N_2O emissions (gCO_2e/MJ)	0.0	0.0	0.0
Total WTW GHG Emissions (gCO_2e/MJ)	174.0	194.9	208.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.99**	**2.23**	**2.38**

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[27] Thomas Tarka, interview with David Ortiz, October 3, 2008.

[28] Compared to 2007, the coal production mix in 2017 is projected to comprise a larger proportion of surface-mined sub-bituminous coal from Western coal production, particularly the Powder River Basin. (EIA, 2008d)

[29] From EIA (2008b) about 49% of total bituminous and sub-bituminous coal production in 2007 is made up of surface-mined sub-bituminous coal, 18% is made up of surface-mined bituminous coal, and the remaining 33% is made up of underground-mined bituminous coal. Coal methane emissions, lower heating value and carbon content of the average coal mix used in the baseline case are based on weighted average values.

6.3.2   CTL with Carbon Capture

As discussed in Section 6.1, carbon capture efficiencies of 80%, 85% and 90% were assumed for the high emissions, baseline and low emissions cases, respectively. These input assumptions are summarized in Table 27.

Table 27: Input assumptions for the production of F-T jet fuel from coal (with carbon capture) for low emissions, baseline and high emissions cases

	Low	**Baseline**	**High**
Carbon compression energy	250 kWh/ton C	250 kWh/ton C	250 kWh/ton C
Process efficiency (LHV)	51.3% (53% without CCS)	48.4% (50% without CCS)	45.6% (47% without CCS)
Coal input	Surface-mined bituminous coal	Average bituminous and sub-bituminous coal mix	Underground-mined bituminous coal
Carbon capture efficiency	90%	85%	80%

The life cycle GHG emissions of the production and use of F-T jet fuel from a CTL plant with carbon capture and sequestration are shown in Table 28.

Table 28: Summary of results for F-T jet fuel from coal pathway (with carbon capture)

	Low	**Baseline**	**High**
Life Cycle CO$_2$ Emissions by Stage			
Recovery of feedstock (gCO$_2$/MJ)	0.8	0.8	0.8
Transportation of feedstock (gCO$_2$/MJ)	0.1	0.1	0.1
Processing of feedstock to fuel (gCO$_2$/MJ)	11.4	19.4	26.6
Transportation of jet fuel (gCO$_2$/MJ)	0.6	0.6	0.6
Combustion CO$_2$ (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	12.9	20.9	28.1
WTT CH$_4$ emissions (gCO$_2$e/MJ)	1.6	5.9	14.1
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.0	0.0	0.0
Total WTW GHG Emissions (gCO$_2$e/MJ)	84.9	97.2	112.6
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.97**	**1.11**	**1.29**

The production of F-T jet fuel from a CTL plant without carbon capture results in life cycle GHG emissions that are between 2.0 (low emissions case) to 2.4 (high emissions case) times higher than conventional jet fuel. When carbon sequestration is implemented, the life cycle GHG emissions relative to those of conventional jet fuel range from almost 1.0 to 1.3. The use of carbon capture and sequestration can allow for an approximate 50% reduction in GHG emissions for a coal-to-liquids facility.

6.4   Fischer-Tropsch Fuel from Biomass

A self-sufficient BTL plant was assumed for the GHG analysis presented here. This assumes that biomass feedstock is used to meet internal process energy needs, with little or no excess electricity produced for export. Similar to the other F-T plants analyzed above, the BTL plant was assumed to produce commercial quality liquid fuels like diesel and jet fuel, and the energy allocation method was adopted for assigning energy and emissions between liquid fuel products.

The biomass feedstocks examined include waste biomass (e.g. forest residue, agricultural residue) and non-food energy crops (e.g. herbaceous biomass) that were assumed to be grown on land that would not incur adverse direct or indirect land use change emissions (e.g. idle or abandoned cropland). Specifically, three types of biomass feedstocks were considered: switchgrass, corn stover and forest residue. Switchgrass was considered as the primary biomass feedstock in all cases of the scenario analysis (low emissions, baseline and high emissions) because of its better yield and improved scalability relative to waste products. Furthermore, if waste products were used on a large scale then they would become a market commodity, as they are no longer waste. Such a change in classification leads to questions as to where system boundaries should be drawn with respect to the crop that initially produced the waste. Such issues were not considered in the scope of this report.

6.4.1 Switchgrass

Switchgrass is a perennial warm season grass native to North America, found in remnant prairies, native grass pastures, and naturalized along roadsides. Other forms of herbaceous biomass include mixed prairie grasses, wheat, hay and leaves, among others. As a replacement for annual crops, warm season grasses have also been shown to provide important habitat for wildlife, including game birds and other species threatened by the loss of tall grass prairie habitat (McLaughlin et al., 2002). The assumptions regarding the yield, energy and emissions associated with switchgrass cultivation were based on a survey of existing cultivation data from the literature. Also considered is the potential for long term changes in carbon contained within the soil on which the switchgrass is grown. The greatest potential for long-term soil carbon sequestration is in those situations where agricultural practices have led to a progressive and historical decline in soil carbon stocks (McLaughlin el al., 2002). The properties (e.g. lower heating value, carbon content) of switchgrass used in this work are reported in Table 97

6.4.1.1   Switchgrass Yield And Cultivation Trends

The approach used in this work is similar to that taken by the National Academies in their 2009 report on Liquid Fuels from Coal and Biomass. Figure 8 shows a distribution of annual switchgrass yields taken from Gunderson et al. (2008) that were used to establish predictive maps of potential yields across the continental United States. The data set comprises approximately 1400 observations with a mean of approximately 4.9 tons/acre/yr. For each data point, the specific cultivar, crop management information, ecotype, precipitation and temperature in the long-term climate record were documented. Using their model, Gunderson et al. (2008) predicted yields in excess of 8.9 tons/acre/yr for lowland ecotypes[30] in the Appalachian region and 5.4-6.2 tons/acre/yr in the Nebraska/South Dakota region. Similarly, yields for upland ecotypes in the Appalachian region were predicted to be greater than 6.2 tons/acre/yr and 3.1-4.5 tons/acre/yr in the Nebraska/South Dakota region. Gunderson et al. (2008) openly discuss that their model predicts the theoretical maximum yield for a given set of input conditions; hence, experimental yields for these regions will most likely be lower in practice.

Other studies have focused on establishing estimates for a national average yield. Heaton et al. (2004) found an average switchgrass yield of 4.6 tons/acre/yr (+/- 0.3 tons/acre/yr) and McLaughlin et al. (2002) projected a national average annual yield of 4.2 tons/acre/yr. Vadas et al. (2008) adopted a nominal yield of 4.0 tons/acre/yr and an optimistic yield of 5.8 tons/acre/yr based on data from large field plots in southern Wisconsin while Adler et al. (2007) simulated switchgrass production in Pennsylvania as 4.3 tons/acre/yr using DAYCENT.[31] Finally, the GREET herbaceous biomass production pathway assumes a yield of 6.0 tons/acre/yr.

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[30] Lowland and upland ecotypes are defined by position relative to the level where water flows or where flooding occurs

[31] DAYCENT is the daily time-step version of the CENTURY biogeochemical model. From weather (daily maximum and minimum air temperature, precipitation), soil-texture class, and land-use inputs, DAYCENT simulates fluxes of carbon (C) and nitrogen (N) between the atmosphere, vegetation, and soil while predicting crop production, soil organic-matter changes, and trace-gas fluxes.
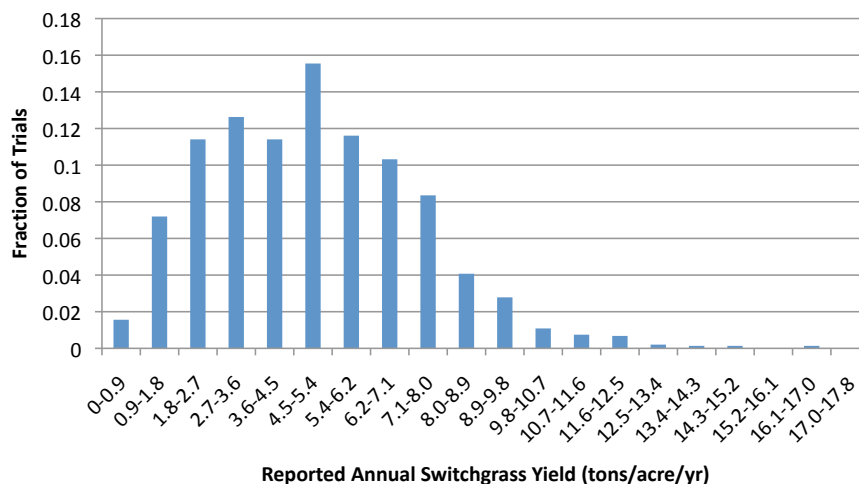
Figure 8: Distribution of reported switchgrass yields across the United States (data from Gunderson et al., 2008)

While numerous studies have estimated the potential yield of switchgrass, many of these results are based on small plots (less than 5m^2) and the results are not necessarily indicative of what can be expected of farm-scale production (National Academies, 2009). Schmer et al. (2008) managed switchgrass as a biomass energy crop in field trials of 7.4 to 22.2 acres on marginal cropland from 10 farms across a wide precipitation and temperature gradient in the mid-continental US. The actual farm-scale production resulted in harvested yields about 35% to 50% lower than those of small-scale plots. It is possible that the lower yields from large-scale production can be attributed to farmers' inexperience with the cropping system or differences in cropland quality; however, farmers worked closely with the researchers in collecting this data and the land had been in active crop production before being converted to switchgrass production (National Academies, 2009). Actual yield data from Schmer et al. (2008) ranged from 2.3 to 5.0 tons/acre/yr with a mean of 3.2 tons/acre/yr.

In their analysis, Gunderson et al. (2008) concluded that switchgrass yield is most influenced by ecotype (upland or lowland) and the relationship of precipitation and temperature. Lower yields were attributed to factors that were not quantified across the data set, such as soil pH, inherent soil fertility, total solar radiation (vs. long periods of cloud cover) and others that are artifacts of each individual growing site and cannot be aggregated for a generalized result. The specific rate of nitrogen application was found not to have a significant influence on yield. Very high levels of fertilization certainly did not guarantee increased biomass production, and in many cases, cases the zero fertilizer plantings did as well as any fertilized stands. Based on these conclusions, the yields adopted for the low emissions, baseline and high emissions scenarios (as shown in Table 29) were assumed to be independent input parameters from other cultivation inputs (on a per ton basis).

Table 29: Switchgrass yields assumed in the low, baseline and high emissions scenarios

	Low[1]	Baseline[2]	High[3]
Yield (tons/acre/yr)	5.8	4.6	3.2
Notes:			
1) Optimistic yield from Vadas et al. (2008) based on large field plots in southern Wisconsin			
2) Projected national average from McLaughlin et al. (2002)			
3) Average farm-scale yield from Schmer et al. (2008) based on mid-continental US			

6.4.1.2   Cultivation and Transportation of Switchgrass

The key inputs for switchgrass production are the process fuels and electricity used in farming, fertilizer inputs and herbicide usage. These parameters have been identified in several studies but are the subject of much uncertainty. Although it was determined that these inputs do not have a substantial impact on yield, they are essential for estimating the GHG emissions associated with switchgrass production. A summary of the available data is given in Table 30.

These data show considerable variation in the application rates of non-nitrogen fertilizers and herbicides. The phosphorous and potassium application rates quoted by Vadas et al. (2008) are an order of magnitude larger than those given by Adler, which are in turn an order of magnitude larger than the GREET default values.

Using the same arguments, which were previously made in considering yield as an independent parameter, the process fuel usage, nitrogen fertilizer application and other fertilizer and herbicide application were decoupled from their respective data sets. Hence, they were also considered as independent parameters for the purposes of the scenario analysis. The input parameters used for the low emissions, baseline and high emissions scenario are outlined in Table 31. The GREET default parameters were not used at all in this work because they are a decade older than the other results and they are not consistent with the 2015 timeframe of this study.

Default GREET transportation and distribution assumptions were adopted for switchgrass. Specifically that bailed switchgrass is transported 40 miles by truck from the field to an F-T processing facility in loads of 24 tons.

Table 30: Reported cultivation inputs for switchgrass

	Adler et al. (2007)[1]	Vadas et al. (2008)[2]	Schmer et al. (2008)[3]	GREET (2008)[4]
Process Fuels (Btu/ton)				
Diesel	82874	113046	107533	201589
Gasoline	0	22609	0	0
Electricity	0	7536	0	15641
Crop Management (g/ton)				
Nitrogen	5218	11348	7701	10635
P_2O_5	1236	10387	0	142
K_2O	2488	24607	0	226
Limestone	9491	0	0	0
Herbicides	6.4	0	185	28

Notes:
1) Actual fertilizer application rates were only given for nitrogen. All others were given in terms of CO_2e with application rates calculated using production emissions from GREET 1.8b.
2) Actual phosphorous and potassium usage was given in terms of elemental weight and converted to P_2O_5 and K_2O using molar mass fractions. Lube oil consumed is expressed in terms of diesel equivalent on an energy basis.
3) Data is the average of 10 field scale plots in Nebraska, North Dakota and South Dakota.
4) From Oak Ridge National Laboratory in 1998.

Table 31: Cultivation inputs for switchgrass in the low, baseline and high emissions scenarios

	Low	Baseline	High
Process Fuels (Btu/ton)			
Diesel	82874	107533	113046
Gasoline	0	0	22609
Electricity	0	0	7536
Nitrogen Fertilizer (g/ton)	5218	7701	11348
Other Fertilizers (g/ton)			
P_2O_5	0	1236	10387
K_2O	0	2488	24607
Limestone	0	9491	0
Herbicides	185	6.4	0

6.4.1.3 Nitrous Oxide Emissions

Nitrous oxide emissions can either be estimated using specialty software or through simple IPCC emissions factors. Estimates from Adler et al. (2007) using DAYCENT included both direct emissions of N_2O through nitrification and denitrification processes in the soil and indirect emissions of N_2O through soil nitrogen losses in forms other than N_2O (e.g. NO_x, NH_3, NO_3), which were subsequently converted to N_2O elsewhere. Conversely, the GREET method employs the IPCC 2006 conversion factor for direct and indirect N_2O emissions from switchgrass production, as shown in using Equation 5.

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{ton} \right) = \left(\frac{g_{nitrogen\ fertilizer}}{ton} \right) \cdot 0.01325 \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right) \qquad \text{Equation 5}$$

The N_2O emissions from Adler et al. (2007) were found to be 43% higher than those predicted by Equation 5 from the same application rate. The principle reason for this discrepancy is that annual nitrogen in crop residues (above-ground and below-ground) was not included. Estimates of the nitrogen deposited on the soil in the form of crop residues was obtained using the IPCC Tier 1 methodology for perennial grasses (De Klein et al., 2006) and implemented through Equation 6.

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{ton} \right) = \left(\left(\frac{g_{nitrogen\ fertilizer}}{ton} + 6025 \right) \cdot 0.01_{direct} + \left(\frac{g_{nitrogen\ fertilizer}}{ton} \right) \cdot 0.001_{indirect} \right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right) \quad \text{Equation 6}$$

where 6025 is the nitrogen in crop residues, 0.01_{direct} is the emissions factor for N_2O from nitrogen and $0.001_{indirect}$ is the emissions factor for volatilized NH_3 and NO_x from synthetic nitrogen converted to N_2O. The N_2O emissions as calculated using the new IPCC methodology are within 2% of those estimated by Adler et al. (2007) for the same nitrogen application rate. As such, Equation 6 was used for all N_2O calculations within the switchgrass production pathway.

6.4.1.4 Long Term Soil Carbon Sequestration from Switchgrass

Soil carbon sequestration is a potential strategy for offsetting CO_2 emissions to the atmosphere. The capacity of perennial energy crops to offset CO_2 emissions through soil carbon sequestration depends on the rate of soil carbon additions, the long-term capacity of soil for carbon storage, and the stability of sequestered soil carbon over time. The greatest potential for long-term soil carbon sequestration is in those situations where agricultural practices have led to a progressive and historical decline in soil carbon stocks (McLaughlin el al., 2002). Two land use change scenarios were considered for switchgrass production. The first is switchgrass grown on Conservation Reserve Program (CRP) land and second is switchgrass grown on carbon-depleted soils. The carbon contained within soil can become depleted over time as a result of farming traditional row crops with conventional tilling practices. The CRP compensates farmers for removing lands from crop production for environmental and economic reasons (National Academies, 2009). After stopping crop production, the land naturally re-establishes its carbon

content due to the growth of native perennial grasses; hence, switchgrass grown on CRP land was assumed to cause no net change of carbon in the soil. The change in soil carbon resulting from switchgrass grown on carbon-depleted soils was based on results from McLaughlin et al. (2002) and Adler et al. (2007).

McLaughlin et al. (2002) developed an empirically derived soil carbon dynamics model that estimates soil carbon accumulation rates for contrasting soil types and climates. After accounting for regional soil carbon gains by prior cropping history and climatic region, they found the average soil carbon sequestration level of traditional cropland converted to switchgrass production to be 0.21 Mg of C/acre/yr over a 30-year period. These sequestration rates are based on annual rates of aboveground production over a 6-year period, ranging from 5.3 to 7.9 tons/acre/yr. The variation between individual sites in the fist decade ranged from a minimum of -0.2 Mg of C/acre/yr for southeastern pastures to a maximum of 0.57 Mg of C/acre/yr for south-central croplands (McLaughlin et al., 2002). Therefore, increased soil carbon from switchgrass cultivation is not guaranteed and in some cases a carbon release could occur.

Adler et al. (2007) also estimated the net change in soil carbon associated with the cultivation of switchgrass. As was the case with McLaughlin et al. (2002), the land use change estimates made by Adler et al. (2007) were sensitive to initial soil carbon levels, which are influenced by previous vegetation cover and land management. To acquire realistic modern soil carbon levels, 1800 years of native vegetation followed by tree clearing, plowing, and 200 years of cropping were simulated. The consequence of this effort is that the initial conditions included legacy effects of 215 years of conventional tillage cropping. The result of 15-year switchgrass cycle in DAYCENT was an average carbon sequestration of approximately 0.17 Mg of C/acre/yr (Adler et al., 2007).

Changes in soil carbon arising from changes in land use can constitute a major component of the life cycle GHG emissions; however, their quantification involves high levels of uncertainty. Therefore, a single value was chosen to illustrate the potential impact of land use change emissions on this pathway. Specifically, the estimate of 0.17 Mg of C/acre/year presented by Adler et al. (2007) was adopted as it represents a conservative value with respect the work of McLaughlin et al. (2008).

6.4.2 Corn Stover and Forest Residue

Input assumptions on the energy and emissions associated with the recovery and transportation of corn stover and forest residues were based on default GREET assumptions (GREET, 2008). Default GREET assumptions were also adopted for the energy and carbon content of corn stover and forest residues[32] (see Table 97. While forest residues require no fertilizer and their removal is assumed to have negligible impact on the surrounding environment, corn stover is usually left on the field to replenish soil nutrients and mitigate erosion; however, there is little consensus regarding the quantification of the impacts on productivity, soil structure, and nutrient cycling of removing stover from the field. In addition to maintaining soil carbon levels, stover is a source of nitrogen; hence, synthetic fertilizer would be needed to supply the incremental amount of nutrients needed for corn cultivation when stover is removed (GREET, 2007).

The yield attainable for corn stover is directly proportional to the fraction of stover that must be left on the field to mitigate these aforementioned negative impacts. The quantity of available stover can then be calculated as the difference between the total stover yield and quantity left on the field. The National Academies reported an average yield of corn stover across the United States in 2007 of 151 bu/acre/yr[33], or 4.24 tons/acre/yr. This yield is based on the assumption of a 1:1 ratio of dry weight of corn grain to stover (National Academies, 2009). Results from Wilhelm et al. (2007) have shown that the stover needed to maintain soil carbon, and thus productivity, is a

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[32] A significant variation of forest residue properties can be found in the literature, due to different assumptions of the type of biomass (e.g. hard wood vs. soft wood), which constitute the forest residues.

[33] A bushel of corn or corn stover is defined as 56 lbs at 15.5% moisture content by mass.

greater constraint to an environmentally sustainable harvest than that needed to control water and wind erosion. They predicted that 3.4 tons/acre/yr are required to mitigate erosion and soil carbon loss with moldboard plowing while only 2.3 tons/acre/yr are required if no-till practices were employed. Sheehan et al. (2004) considered the USDA's regional tolerable soil-loss limits as the constraint for leaving stover on the field. They found that 2.2 tons/acre/yr are required with conventional tillage while only 1.1 tons/acre/yr are required if no-till practices are employed.

The corn stover yield adopted in this work was based on the total yield from the National Academies (2009) and the constraints for soil carbon and erosion under no-till practices from Wilhelm et al. (2007). These resulted in a yield of 1.65 dry tons/acre/yr (equivalent to 1.94 tons/acre/yr at 15% moisture). The key input assumptions for the recovery and transportation of corn stover and forest residues are shown in Tables 32 and 33, respectively.

Although the corn stover yields adopted in this work were chosen to prevent a reduction of the natural carbon stock of the soil, an examination of the magnitude of potential land use change emissions from improperly harvested corn stover is still relevant. The analysis from Sheehan et al. (2004) imposed no constraint on maintaining soil carbon levels and the evolution of carbon stored in the soil was a central part of their results. Under maximum stover removal conditions subject to maintaining USDA's tolerable soil constraints, an average emissions rate of 13 kg of C/acre/yr was calculated over the first 30 years. The emissions profile over that time period is strongly positive for the first 10 years but becomes negative after year 20 as the soil begins to approach a new equilibrium state.

Table 32: Input parameters for the recovery and transportation of corn stover

Input parameter[1]	Value
Yield	1.65 dry ton/acre
Collection Energy	254,190 Btu/dry ton (100% diesel fuel use)

Fertilizer Use
    Nitrogen
    Phosphorus
    Potassium
	
	4.50 kg/dry ton
	1.63 kg/dry ton
	8.35 kg/dry ton

N_2O emissions[2]	13.3 g/dry ton
Transportation distance	30 miles
Truck payload	24 ton
Notes: 1) All input parameters were based on GREET 1.8b (GREET, 2008) simulation year of 2015. 2) Includes N_2O emissions from nitrification and denitrification in the soil, as well as N_2O credit from corn stover removal based on GREET defaults.[34]	

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[34] Incremental fertilizer use (including nitrogen fertilizer) is accounted for to make up for the loss in soil nutrients from the removal of corn stover from the field. On the other hand, if left on the field, a fraction of the nitrogen in corn stover will be converted to N_2O and emitted from the soil. This N_2O emission is avoided when corn stover is removed from the field. In this case, the N_2O credit from the removal of corn stover slightly outweighs the N_2O emissions resulting from the incremental application of nitrogen fertilizers, resulting in net negative N_2O emissions.

Table 33: Input parameters for the collection and transportation of forest residue.

Input parameter[1]	Value
Diesel fuel consumption	3.4 gal/dry ton
Collection energy	459,200 Btu/dry ton
Transportation distance	75 miles
Truck payload	17 ton
Note:	
1) These input parameters were based on GREET 1.8b (GREET, 2008)	

6.4.3 Analysis of Biomass Feedstocks

A sensitivity analysis was carried out to investigate the effect of varying feedstock type (switchgrass, corn stover and forest residue) and the influence of changes in soil carbon due to switchgrass growth or corn stover removal on the overall life cycle GHG emissions of the BTL pathway. A process efficiency of 45% was assumed in all the cases. The results are summarized in Table 34.

The sensitivity study shows that the life cycle GHG emissions from the use of switchgrass, without considering soil carbon change credit, are the highest of the three feedstocks. This is largely due to the large N_2O emissions arising from the use of nitrogen fertilizers. However, when a soil carbon change credit is included, the life cycle GHG emissions are -0.02 times those of conventional jet fuel. Hence, the pathway becomes carbon negative because the CO_2 sequestration arising from land use change can dominate the GHG emissions of the BTL pathway. Overall, the use of all three types of feedstock in a biomass to liquids facility results in significant reductions (~80-100% reduction) of life cycle GHG emissions compared to conventional jet fuel.

Table 34: Variation of life cycle GHG emissions of BTL pathway with type of feedstock

Feedstock	Life cycle CO_2 emissions (gCO_2/MJ)	Life cycle N_2O emissions (gCO_2e/MJ)	Life cycle GHG emissions (gCO_2e/MJ)	Compared to baseline conventional jet fuel
Corn stover (without soil carbon change emissions)	7.8	1.0	9.0	0.10
Corn stover (with soil carbon change emissions)	12.4	1.0	13.6	0.16
Forest residue	11.7	0.1	12.2	0.14
Switchgrass (without soil carbon change credit)	7.3	10.3	17.7	0.20
Switchgrass (with soil carbon change credit)	-12.5	10.3	-2.0	-0.02
Note:				
1) Process efficiency of 45% was assumed in all cases.				

For the low emissions case, a process efficiency of 52% was assumed, based on the analysis by Kreutz et al. (2008) of a 4400 bpd BTL plant. The baseline case assumes a process efficiency of 45% from the Choren process based on the "self-sufficient basis scenario" where all required process energy is provided by biomass feedstock (Baitz et al., 2004). Choren currently operates the only commercial-scale BTL plant in the world, producing 300 bpd of F-T liquids. This is further confirmed by estimates of roughly a 5% efficiency drop for BTL plants compared to CTL plants due to additional processing energy for biomass grinding and drying.[35] The high emissions case directly applies a 5% drop in process efficiency compared to 47% assumed in the CTL case, resulting in a process efficiency of 42%.

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[35] Thomas Tarka, interview with David Ortiz, October 3, 2008.

The input assumptions and life cycle GHG emissions for the production and use of F-T jet fuel from switchgrass with and without a soil carbon sequestration credit are shown in Tables 35 and 36 respectively. The 'biomass credit' represents the CO_2 that is absorbed from the atmosphere during biomass growth. Note that the CO_2 emitted during the combustion of process fuels and the F-T fuel is approximately equal to the CO_2 absorbed from the atmosphere during growth of the biomass feedstock.

Table 35: Summary of results for F-T jet fuel from switchgrass with no soil carbon sequestration

Land Use Change Scenario B0	Low	Baseline	High
Key Assumptions			
Feedstock	Switchgrass	Switchgrass	Switchgrass
Process Efficiency (LHV)	0.52	0.45	0.42
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO_2/MJ)	-192.7	-222.7	-238.6
Recovery of feedstock (gCO_2/MJ)	3.8	6.4	11.4
Transportation of feedstock (gCO_2/MJ)	0.6	0.6	0.6
Processing of feedstock to fuel (gCO_2/MJ)	122.1	152.1	168.0
Transportation of jet fuel (gCO_2/MJ)	0.5	0.5	0.5
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	-65.8	-63.1	-58.2
WTT CH_4 emissions (gCO_2e/MJ)	0.1	0.2	0.5
WTT N_2O emissions (gCO_2e/MJ)	7.2	10.3	13.3
Total WTW GHG Emissions (gCO_2e/MJ)	11.9	17.7	26.0
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.14**	**0.20**	**0.30**

Table 36: Summary of results for F-T jet fuel from switchgrass with soil carbon sequestration

Land Use Change Scenario B1	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO_2/MJ)	-13.6	-19.8	-30.5
WTW CO_2 emissions (gCO_2/MJ)	-9.0	-12.5	-18.2
Total WTW GHG Emissions (gCO_2e/MJ)	-1.7	-2.0	-4.4
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**-0.02**	**-0.02**	**-0.05**
Notes: 1) All other input assumptions (cultivation of switchgrass, F-T processing and carbon capture efficiency) are based on those in the B0 emissions case of the corresponding scenario.			

The life cycle GHG emissions of the BTL pathway using switchgrass as a feedstock are 0.14 times to 0.3 times those of conventional jet fuel with no soil carbon sequestration. When the soil carbon sequestration credit is included, the emissions from the pathway range from -0.02 to -0.05 times those of conventional jet fuel. The larger value for land use change emissions in the high scenario occurs due to the lower assumed yield per acre.

Nitrous oxide emissions represent more than 50% of the total life cycle GHG emissions from the switchgrass to F-T jet pathway. As a result, life cycle GHG emissions for this pathway are strongly influenced by any uncertainty associated with the IPCC correlations that were used in the nitrous oxide estimates. The reader should be aware of these inherent uncertainties when comparing different pathways for GHG reduction potential in Sections 8.1 and 8.2.

6.5   Fischer-Tropsch Fuel from Coal and Biomass

While both CTL and BTL hold promise as alternative jet fuels, they also have flaws. Even with 85% carbon capture, a CTL plant has life cycle GHG emissions over 110% of conventional jet fuel (i.e., the life cycle GHG emissions are 10% higher than conventional jet fuel); without carbon capture, CTL has 220% of the emissions of conventional jet fuel. If the goal is to reduce GHG emissions, then coal alone appears to be a poor choice. Biomass-to-liquids plants without carbon capture have life cycle GHG emissions that are less than 10% of conventional jet fuel; however, considerable logistical challenges exist in obtaining sufficient biomass to operate at large scales because of the relatively low energy density of biomass. Because current F-T plant designs are capital intensive, it is not economically feasible to build many small plants that are dispersed among the regions where biomass is being grown. Biomass must therefore be accumulated from considerable areas and transported to large central plants; hence, the infrastructure to move the biomass becomes a limiting factor. Emissions from the transportation of the biomass to the processing facility are included in the life cycle analysis, but represent a negligible fraction of the total emissions. Since both biomass and coal are processed into an F-T fuel using similar technology, they could be processed at a single F-T plant. The biomass offsets the high emissions from coal and coal offsets the low energy density and production limitations of biomass. A coal and biomass to liquid (CBTL) plant with carbon capture and storage (CCS) was considered in this analysis. A configuration without CCS was examined as a case study; however, the focus herein is on a configuration with CCS as the primary goal is to reduce GHG emissions.

The coal and biomass can be gasified either in parallel with the syngas streams being mixed afterwards, or in the same unit (co-gasification). Since the parallel configuration is a superposition of the results of sections 6.3 and 6.4, this section examines co-gasification. Parallel processing also requires additional infrastructure, as separate gasifiers are needed for each feedstock. Co-gasification was assumed to occur in an entrained flow gasifier. Such technology is already commercially available for large scale processing of coal and biomass (van der Drift et al., 2004). Before entering the gasifier, biomass must be milled down to particles of diameter 1mm or less. Currently, the most energy efficient method of milling the biomass is via torrefaction pre-treatment, which is a mild thermal treatment where the biomass is heated to ~250°C yielding a solid uniform product with lower moisture content and higher energy content. Studies have shown that torrified wood can be milled to the required size using only 10-20 $kW_{electricity}/MW_{biomass}$ and that capacity expansion factors between 2 and 6.5 can be achieved (van der Drift et al., 2004; Bergman et al., 2005). Moreover, torrified particles can be pneumatically transportable, which is considered impossible for a bed of untreated biomass particles. Efficiencies for torrefaction range from 85% to 97%, with 90% assumed in the baseline case (Bergman et al., 2005).

Because of the pre-processing of biomass, the overall F-T plant efficiency depends on the weight percent of biomass that is being co-gasified. This study explored a range from 10% to 40% biomass feed with 40%, 25% and 10% chosen for the low, baseline and high emissions cases, respectively. Due to the CO to H_2 ratio resulting from gasification, feeds with biomass weight fractions below 50% lead to syngas favorable for methanol and hydrocarbon fuels whereas biomass weight fractions above 50% lead to syngas favorable for dimethyl ether (DME) synthesis. Additionally, higher biomass feeding rates lead to the formation of less char and tars (Kumabe et al., 2007). CTL plant efficiencies were modified to account for the extra power consumption of pre-processing the biomass; hence, the reduction of the CTL plant efficiency, η_{CTL}, depends on both the fraction of biomass feed and the torrefaction efficiency. The plant is assumed to be self sufficient in terms of electricity production with no extra power exported to the grid. Higher efficiencies could be possible if additional electricity were generated for grid export. The implementation for CCS comes at the cost of 250kWh per ton of carbon captured, as was the case with CTL. This energy requirement is also accounted for in the CBTL process efficiency plant. In total, the CBTL plant efficiency was examined as a function of CTL efficiency, biomass weight percentage, torrefaction efficiency, biomass grinding energy and CCS efficiency, as expressed in Equation 7.

$$\eta_{CBTL} = f\left(\eta_{CTL}, \text{ biomass wt\%}, \eta_{\text{torrefaction}}, \text{ biomass grinding energy}, \eta_{\text{CCS}}\right) \quad \text{Equation 7}$$

Assumptions about feedstock type are similar to those made when coal and biomass were considered separately. Specifically, surface-mined sub-bituminous coal and switchgrass were assumed in the low emissions case; average US coal from 2007 (mix of underground-mined and surface-mined bituminous coal and surface-mined sub-bituminous coal[36]; anthracite or lignite coal was not considered) and switchgrass were assumed in the baseline case and underground-mined bituminous coal (from case 1 of Southern States Energy Board CTL study, SSEB, 2006) and switchgrass were assumed in the high case. Sub-bituminous coal is used in the low emissions case because it reduces GHG emissions by increasing the fraction of input energy provided by biomass for a given biomass/coal mass ratio. Input assumptions for the combined coal and biomass pathway are summarized in Table 37.

Table 37: Input assumptions for the production of F-T jet fuel from coal and biomass (with carbon capture) for low emissions, baseline and high emissions cases

	Low	**Baseline**	**High**
CTL Process Efficiency (LHV)	53%	50%	47%
Biomass Weight Fraction	40%	25%	10%
Carbon Capture Efficiency	90%	85%	80%
Carbon Compression Energy	250 kWh/ton C	250 kWh/ton C	250 kWh/ton C
Torrefaction Efficiency	97%	90%	85%
Coal Input	Surface-mined sub-bituminous coal	Average bituminous and sub-bituminous coal mix	Underground-mined bituminous coal
Coal Mining Methane Emissions (gCO$_2$e/MJ$_{Coal}$)	0.8	2.8	6.4
Biomass Type	Switchgrass	Switchgrass	Switchgrass

6.5.1 Allocation Methodology

As previously stated, an F-T facility can produce a wide product slate. For example, the study carried out by NETL on F-T diesel from CBTL assumed an output of 70% diesel and 30% naphtha (Tarka, 2009). In this analysis, F-T jet fuel is the product of interest but does not consist of more than roughly a third of the total plant output to prevent excessive naphtha production. Since other F-T fuels are made as a result of producing F-T jet fuel, the emissions from processing and all other upstream activities must be allocated among the fuels being produced. Even when configured to make jet fuel, diesel fuel will likely be the primary output of the F-T facility due to economic drivers. For this reason, it is sensible to allocate emissions among the liquid products (i.e., jet fuel, diesel fuel, and naphtha) on the basis of their respective energy content.

Figure 9 demonstrates problems that could result from using the displacement method to assign emissions 'credits' to a fuel that is not the primary product. When the yield of jet fuel is 25% by volume, there are 3 liters of other F-T fuels produced for every 1 liter of jet fuel. When the yield of jet fuel is reduced to 5% by volume, there are 19 liters of other F-T fuels produced for every 1 liter of jet fuel. In the limit where the yield of jet fuel approaches zero, the quantity of other F-T fuels, relative to jet fuel, asymptotically approaches infinity. If the other F-T fuels represent a reduction in emissions as compared to their petroleum equivalents then the displacement method attributes these emissions reductions to the jet fuel; therefore, the displacement methodology results in a jet

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[36] Compared to 2007, the coal production mix in 2017 is projected to comprise a larger proportion of surface-mined sub-bituminous coal from Western coal production, particularly the Powder River Basin. (EIA, 2008d)

fuel which appears to have life cycle emissions that approach negative infinity as the yield of jet fuel is reduced towards zero. The results from Figure 9 show that an energy allocation scheme prevents a product that is responsible for a third, or less, of total output from the facility receiving the emissions 'credit' from the entire product slate.
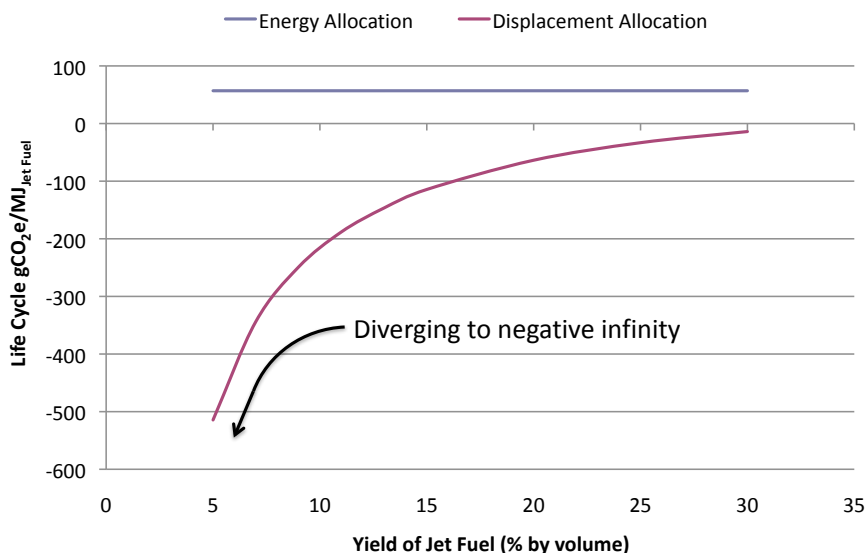
Figure 9: The effects of product slate composition on life cycle greenhouse gas emissions from F-T jet fuel

The diverging effect resulting from use of the displacement method is unavoidable because jet fuel is not the primary product from the F-T plant. This highlights the important conclusion that life cycle GHG emissions from a given fuel are as much a function of subjective choices of the analyst/operator regarding allocation methodologies, as they are of specific production characteristics and process inputs. In order to maintain consistent results, which cannot be skewed by choices such as the distribution of products leaving the F-T facility, energy allocation was chosen. This method comes with the caveat that all fuels produced in addition to jet fuel, such as diesel and naphtha, can also carry environmental benefits that are only captured when the entire system is considered as a whole. The specific product slate of this analysis was 25% F-T jet fuel, 55% F-T diesel and 20% F-T naphtha.

6.5.2   Results: F-T Jet Fuel from Coal and Biomass

Local sensitivity analysis was conducted on the feedstock type, the quantity of GHG emissions from land use change, CTL process efficiency, biomass weight percentage, torrefaction efficiency, CCS efficiency and CCS compression energy. Each parameter was varied with all others held at their baseline values with the impact on life cycle GHG emissions quantified as a percent change from the baseline value. Figure 10 presents this information in a manner that allows the magnitude of each change to be seen in comparison to the others. The biomass feed rate has the dominant influence. Since the biomass feed rate is a parameter that is chosen by the operator, the emissions from CBTL facilities will be dictated by practical, as opposed to technological, limitations. Overall, the choice of feedstock and the potential for soil carbon sequestration were found to have a larger impact on life cycle GHG emissions than the process efficiencies.

When examining the impacts of changing the plant efficiency (this can be achieved either directly or by changing the energy consumption for biomass pre-processing or carbon dioxide compression) it was found that lower efficiencies lead to lower net emissions. This counter-intuitive result occurs because the plants were chosen to be self-sufficient and CCS is used to capture emissions from gasifying additional biomass to supply syngas for process fuel. The

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capture and sequestration of carbon contained in the biomass leads to a net carbon removal from the atmosphere and hence having less biomass converted to fuel results in more of the carbon in the biomass going to sequestration.
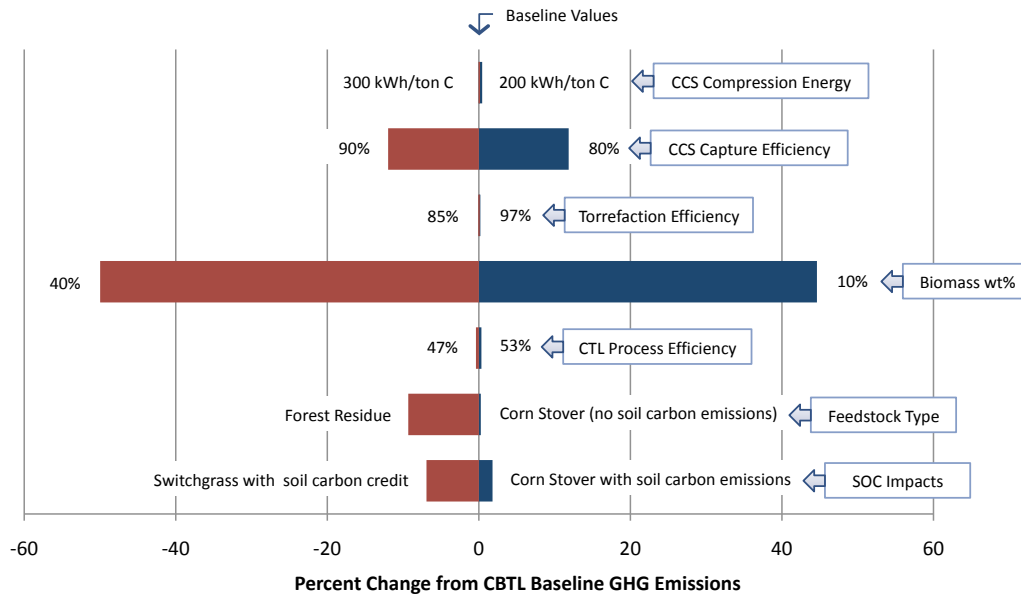
Figure 10: Sensitivity analysis of operational specifications and configurations of F-T jet fuel from coal and biomass

The results for the low emissions, baseline and high emissions scenarios for CBTL using switchgrass as the biomass feedstock are shown in Tables 38 and 39. Table 38 gives the results when no soil carbon sequestration credit is given to the switchgrass while Table 39 gives the results when the soil carbon sequestration credit is included. The 'biomass credit' represents the CO_2 that is absorbed from the atmosphere during biomass growth. Increasing biomass credit reflects larger amounts of biomass being used as feedstock.

Table 38: Summary of results for F-T jet fuel from coal and biomass without soil carbon sequestration credit

Land Use Change Scenario B0	Low	Baseline	High
Key Assumptions			
Biomass Weight Fraction	40%	25%	10%
Biomass Input	Switchgrass	Switchgrass	Switchgrass
Carbon Capture Efficiency	90%	85%	80%
CBTL Process Efficiency (LHV)	48.9% (53% without CCS or biomass processing)	46.0% (50% without CCS or biomass processing)	44.1% (47% without CCS or biomass processing)
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO_2/MJ)	-78.6	-44.3	-15.3
Recovery of feedstock (gCO_2/MJ)	1.2	1.2	1.1
Transportation of feedstock (gCO_2/MJ)	0.3	0.2	0.1
Processing of feedstock to fuel (gCO_2/MJ)	14.7	21.9	28.6
Transportation of jet fuel (gCO_2/MJ)	0.5	0.5	0.5
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	-62.0	-20.5	14.9
WTT CH_4 emissions (gCO_2e/MJ)	1.1	4.9	13.6
WTT N_2O emissions (gCO_2e/MJ)	2.9	2.0	0.9
Total WTW GHG Emissions (gCO_2e/MJ)	12.4	56.9	99.8
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.14**	**0.65**	**1.14**

Table 39: Summary of results for F-T jet fuel from coal and switchgrass with soil carbon sequestration credit

Land Use Change Scenario B1	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO_2/MJ)	-5.5	-3.9	-2.0
WTW CO_2 emissions (gCO_2/MJ)	2.9	46.0	83.4
Total WTW GHG Emissions (gCO_2e/MJ)	6.9	53.0	97.8
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.08**	**0.61**	**1.12**
Notes:			
1) All other input assumptions (cultivation of switchgrass, F-T processing and carbon capture efficiency) are based on those in the B0 emissions case of the corresponding scenario.			

The life cycle GHG emissions of the CBTL pathway range from 0.14 to 1.14 times those of conventional jet fuel when no soil carbon sequestration credit is given. The emissions for this pathway range from 0.08 to 1.12 times those of conventional jet when the soil carbon sequestration credit is included. The large range of this pathway is primarily driven by the variation in biomass weight fraction of the feedstock.

In their assessment of F-T diesel production, Tarka (2009) used a displacement (system expansion) scheme instead of energy-based allocation to account for the benefit of making a reduced carbon, biomass-based F-T naphtha in addition to the F-T diesel. As such, their results differ from those given here. When a common allocation approach and productions assumptions are implemented, both analyses yield similar results.
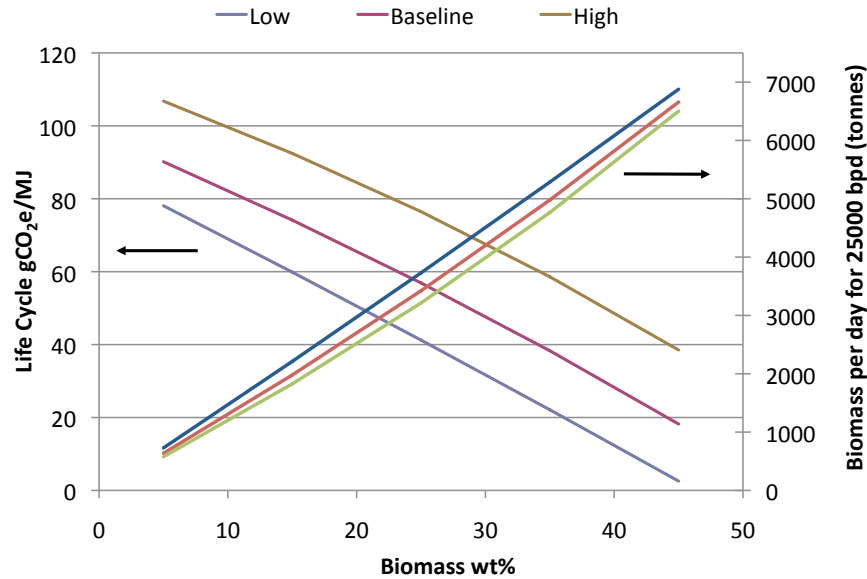
Figure 11: Dependence of cumulative life cycle emissions and biomass requirements for varied biomass utilization within CBTL

Figure 11 presents the implication of varying biomass weight over a range of 5% to 45%. Life cycle GHG emissions can be reduced to a fraction of conventional jet fuel with considerable biomass usage. For example, provided sufficient CCS is available, a CBTL jet fuel created from 45% biomass could have life cycle GHG emissions that are only 20% of conventional jet fuel; however, roughly 245 railroad cars of biomass would be needed every day to create sufficient CBTL jet fuel to power the aircraft at Boston Logan airport.[37] This large amount of biomass highlights the importance of considering GHG reductions for high biomass weight percentages in conjunction with biomass feeding requirements; it also points to lower biomass percentages being more realistic. Future work will consider the economics of CBTL fuels.

6.5.3 Case Study: Impact of Carbon Capture on GHG Emissions from CBTL Facilities

The analysis presented above assumed the use of CCS to reduce GHG emissions. Two additional cases were examined to emphasize the importance of using CCS to reduce GHG emissions from CBTL facilities. The first is the baseline CBTL case without CCS and without soil carbon sequestration. The second is the baseline CBTL case without CCS where the biomass feed rate has been adjusted such that the life cycle GHG emissions are on parity with conventional jet fuel.

A comparison of results from the baseline scenario with and without CCS is given in Table 40. Not having CCS leads to an increase in GHG emissions of 106.1 gCO_2/MJ, such that the WTW emissions from the CBTL pathway become 186% of conventional jet fuel. Without CCS, one needs to use 70% biomass, by weight, for F-T fuel from coal and switchgrass to reach GHG parity with conventional jet fuel. This understates the importance of CCS to getting reduced GHG emissions from CBTL fuels.

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[37] 6,500 tonnes of biomass would be needed per day to provide 25,000 barrels per day of jet fuel at a biomass feed rate of 45% (this is roughly the consumption of Boston Logan Airport). A typical railroad car can carry 26.5 tonnes of biomass (Mahmudi, 2006).

Table 40: Life cycle GHG emissions from F-T jet fuel from coal and switchgrass with and without CCS

	Baseline	Baseline (w/o CCS)	Biomass for GHG Parity (w/o CCS)
Key Assumptions			
Biomass Weight Fraction	25%	25%	70%
Biomass Input	Switchgrass	Switchgrass	Switchgrass
Carbon capture efficiency	85%	0%	0%
Life Cycle CO$_2$ Emissions by Stage			
Biomass credit (gCO$_2$/MJ)	-44.3	-42.0	-141.1
Recovery of feedstock (gCO$_2$/MJ)	1.2	1.2	1.9
Transportation of feedstock (gCO$_2$/MJ)	0.2	0.2	0.4
Processing of feedstock to fuel (gCO$_2$/MJ)	21.9	126.1	146.5
Transportation of jet fuel (gCO$_2$/MJ)	0.5	0.5	0.5
Combustion CO$_2$ (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	-20.5	86.0	8.3
WTT CH$_4$ emissions (gCO$_2$e/MJ)	4.9	4.7	2.3
WTT N$_2$O emissions (gCO$_2$e/MJ)	2.0	1.9	6.5
Total WTW GHG Emissions (gCO$_2$e/MJ)	56.9	163.0	87.5
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	0.65	1.86	1.00

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7 Hydroprocessed Renewable Jet Fuel from Renewable Oils

Renewable oils can be processed into a fuel that has properties similar to those of F-T fuels. The processing involves hydrotreatment to deoxygenate the oil with subsequent hydrocracking to create hydrocarbons that fill the distillation range of jet fuel (Hileman et al., 2009). This work examined the life cycle GHG emissions from the production and use of Hydroprocessed Renewable Jet fuel (HRJ) from soybean oil, palm oil, rapeseed oil, algae oil, jatropha oil and salicornia oil. As of the writing of this report, the production of hydroprocessed jet fuel from any oil feedstock is still limited to quantities suitable for flight-testing. Numerous flight tests have been successfully conducted of fuel composed of 50% HRJ from mixes of jatropha, algae and camelina blended with conventional jet fuel.

In all cases, HRJ will have to compete with hydroprocessed renewable diesel (HRD) and biodiesel for feedstock availability. Biodiesel is currently the only biofuel produced at commercial scales from renewable oils; however, facilities to hydroprocess renewable oils are being built worldwide with a production capacity of nearly 60,000 bbl/day (Hileman et al., 2009). While biodiesel is not appropriate for use in gas turbines at any blending ratio, biodiesel production can be used to understand the resource potential of renewable oils for biofuel because it is made from the same feedstock. Soy and palm oil were examined using GREET version 1.8a whereas all other HRJ pathways used GREET version 1.8b.

Soybean oil is of interest as it is used extensively in the US and Europe for biodiesel production. The US and Europe used a total of 48,700 bbl/day (2.6 million metric tons per year) of soy oil in 2009 for biodiesel, which represents only 20% of total soy oil use in these regions. Similarly, rapeseed oil is the main feedstock used for biodiesel production in Europe. Approximately 35% of all rapeseed oil used in Europe went to biodiesel, amounting to 58,000 bbl/day (3.1 million metric tons per year). Palm oil, on the other hand, has grown to become the most produced oil in the world. Its production has increased rapidly in the past 20 years and the production of palm oil surpassed soy oil for the first time in 2006. Almost 90% of global palm oil production occurs in Indonesia and Malaysia (FAPRI, 2009). Biodiesel production is expanding based on the palm oil resources in this area. Malaysia has issued approval of 91 companies to build domestic palm oil based biodiesel plants; however, total planned capacity of the entire region is still less than 30,000 bbl/day (1.5 million metric tons per year) (Virki, 2007; Forbes.com, 2007; Mission Biofuels Limited, 2007).

Soybeans, palm and rapeseed are edible food crops requiring fertile cropland for cultivation. The use of these crops in fuel production could result in direct or indirect land use change emissions. As noted previously in the text, large uncertainties exist in estimating the GHG emissions from both direct and indirect land use change. This work only quantifies direct land use change because the economic models required to analyze indirect effects are beyond the scope of this effort. Similar to the treatment of switchgrass in Sections 6.3 and 6.4, the following sections consider a range of direct land use change scenarios using multiple land conversion scenarios to establish upper and lower bounds for these values. These scenarios were created based upon existing data from the literature. Since indirect land use change emissions are the integrated impact of direct land use change resulting from increased crop prices, the range of direct land use change emissions presented herein should bracket any potential indirect land use change emissions occurring because of these feedstocks.

Jatropha curcus is a small tree or large shrub, up to 5-7 meters tall, which can grow without irrigation in a broad spectrum of rainfall regimes. All subsequent references to jatropha are with regard to *jatropha curcas*. Under normal conditions, the jatropha plant flowers only once a year during the rainy season; however, in permanently humid regions or under irrigated conditions it can be made to flower almost all year round (Achten et al., 2008). The resulting fruit is composed of an outer capsule containing two or three seeds. Each seed has a shell and a kernel, which

contains oil. Jatropha plants have higher oil yields than many other oil yielding crops; however, the husk and the seed shells result in more co-product per unit mass of jatropha oil than both algae and palm fresh fruit bunches.

The potential for jatropha was recently demonstrated by successful gas turbine test flights using a hydroprocessed mix of jatropha and other oils. Jatropha cultivation is ideal for regions of the world with the highest rates of poverty and plenty of hot, dry land because it is well adapted to the tropics and subtropics. India has been recently pushing to expand biodiesel production from jatropha oil; the government has announced plans to subsidize an intensive program to plant jatropha for biofuels on 27 million acres of "wastelands". Despite these recent efforts, global production in 2008 was limited to 242 cultivation projects, amounting to only 2.2 million acres. Furthermore, most jatropha grown for biofuels is cultivated locally on plots of less than 12 acres (Luoma, 2009). The major limitations to expanding from local cultivation to large scale production is that farmers are in poverty and only allow jatropha to grow on their land because of its ability to grow in the arid conditions. In areas where jatropha is not naturally occurring, farmers do not wish to take the initiative to restructure their farms to prevent other revenue generating species from dominating the natural growth of jatropha. New plantations takes 3-5 years for jatropha production mature such that it can be grown commercially and farmers don't have the capital to invest in projects where the potential for revenue is several years away. Due to the general poverty of the regions where jatropha is considered promising, there is little to no loan availability or presence of long term contracts to guarantee a return on their investment (Mani, 2010). While it appears that jatropha may be an appropriate solution to provide fuel to the villages in which it is grown, there is currently minimal evidence to indicate that jatropha will expand beyond the village level to become an energy resource on the global scale.

Algae were first examined as a biofuel feedstock by the Department of Energy during the Aquatic Species Program (ASP) from 1978 to 1996. The ASP focused most of its attention on identifying a specific factor that would stimulate the algae to have a high lipid weight fraction (Sheehan et al., 1998b). Much discussion still surrounds the possibility of genetically modifying certain strains of algae to produce more oils; however, the present analysis focuses only on previously documented strains that currently exist. Furthermore, it is important to differentiate between micro-algae and macro-algae. Microalgae, as the name suggests, are tiny organisms which grow in water with concentrations ~0.2-0.4g/L and have the appearance of tinting the water green; these are the algae considered in this work. Macro algae are the classical long strands that grow on the bottom of ponds and lakes, (a.k.a. seaweed). While some research has been conducted using macro-algae as a fuel source, it is not considered further in this work; hereafter, algae will refer to microalgae.

No company is currently producing commercial quantities of algal oil for use as a transportation fuel; instead they are producing quantities that are appropriate for various stages of research and development. Furthermore, the industry is not sufficiently defined to have well defined best practices for cultivating and extracting the oil from the algal cells. This report examines relatively conventional methods of oil extraction wherein the algae are dried prior to chemical treatment; however, several research efforts are ongoing to evaluate whether the oil could be extracted via electrical pulses or if the algae can be genetically modified to naturally excrete oil. Although no commercial scale production of algae currently exists, significant investments are being made into developing it as a feedstock for alternative fuels. For example, the Department of Defense recently ordered 20,000 gallons of algae fuel from Solazyme at a cost of $8.5 million USD ($425 per gallon) for advanced testing as F-76 Naval Distillate (Green Car Congress, 2009). This is a relatively small sum compared to the 600 million US$ that Exxon Mobil has invested in research and development of algal transportation fuels (Mouawad, 2009).

Halophytes are considered an oil seed crop for fuel production because they prosper in sea or brackish water on marginal lands. Other benefits of halophyte agriculture include freeing up arable land for freshwater resources, cleansing the environment, decontaminating soils, desalinating brackish waters and soil carbon sequestration. Considering that 43% of the earth's

landmass is arid or semi arid and 97% of the earth's water is seawater (Hendricks and Bushnell, 2008), halophytes are an attractive option for large-scale production. From the halophyte family, *Salicornia bigelovii* emerged as an oilseed crop from a screening of wild halophytes and was selected for seawater trials due to its seed yield and oil fraction (Anwar et al., 2002). All further references to halophyte or salicornia refer specifically to *Salicornia bigelovii*.

The development of salicornia for fuel production is still in the experimental stage; no fuel testing has been conducted on fuel produced from the salicornia plant. Despite our inexperience, a major UNFCCC project in the Sonoran desert has been cultivating salicornia on 30 hectares of coastal land since 1996 with the goal of developing cost-effective cultivation processes on a commercial basis, and optimizing agronomic methods for irrigation and harvesting (UNFCCC, 1998). The GCC (Cooperation Council for the Arab States of the Gulf) countries have also established research programs for halophytes in the form of an experimental 200-hectare integrated seawater agricultural system near Abu Dhabi. The Masdar Institute of Science and Technology (MIST) is collaborating with UOP, Boeing and Etihad Airways to realize this project (Green Car Congress, 2010).

7.1 Energy Requirements for HRJ from Renewable Oils

The hydrotreatment process for the production of HRJ from renewable oils was based on the UOP hydrodeoxygenation process, which primarily produces "green diesel" (Marker et al., 2005; Huo et al., 2008). Other techniques are available to produce hydrocarbon fuels from renewable oils beyond the approach developed by UOP; however, the UOP process is currently the most established for jet fuel production. Similar to F-T fuels, additional hydroprocessing is needed for the production of jet fuel instead of diesel, resulting in increased hydrogen and power requirements. The assumption that additional processing requirements for F-T jet fuel relative to diesel are negligible is justified in the literature (Gray et al., 2007); however, in the case of HRJ, using diesel as a surrogate for jet is only appropriate for crude estimates.

The UOP process used for the creation of Hydroprocessed Renewable Diesel (HRD) is described by the following chemical reaction:

$$C_nH_{2n+1}COOH + 3H_2 \rightarrow C_{n+1}H_{2n+4} + 2H_2O$$ Equation 8
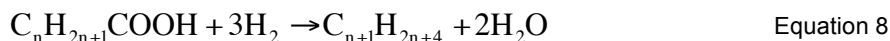
where *n* is the carbon chain length of the fatty acids within the triglyceride molecules used as a feedstock for the process. Triglycerides are formed from a single molecule of glycerol, combined with three fatty acids. This simplified analysis assumed that all renewable oils, regardless of type (soy, pal, rapeseed, jatropha, algae or salicornia) are identical and contain only fatty acids with a carbon chain length of 18. The error introduced by this assumption is likely small compared to the uncertainty in quantifying cultivation inputs for each feedstock; however, not all oils are chemically equal and this analysis does not reflect the physical properties of any single oil type.

Table 41 presents the actual distributions of fatty acid carbon chain lengths for most of the oils considered in this work.[38] An example of a shortcoming of the assumption used in this analysis is the treatment of oils containing unsaturated carbon chains. The presence of double bonds within a carbon chain would lead to additional hydrogen consumption during the deoxygenation process in order to saturate the molecule, however, such effects are ignored herein.

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[38] The notation for denoting carbon chain length and number of double bonds is (chain length):(number of double bonds). For example, a chain length of 18 with 2 double bonds is expressed as 18:2.

Table 41: Component fatty acid profiles for renewable oils considered in this work

Fatty Acid Components (weight %)	Caprylic (8:0)	Capric (10:0)	Lauric (12:0)	Myristic (14:0)	Palmitic (16:0)	Palmitoleic (16:1)	Stearic (18:0)	Oleic (18:1)	Linoleic (18:2)	Linolenic (18:3)	Arachidic (20:0)	Gadoleic (20:1)	Behenic (22:0)	Erucic (22:1)
Soybean	—	—	—	—	11	—	4	22	53	8	—	—	—	—
Palm	—	—	—	—	44	—	4	39	11	—	—	—	—	—
Palm Kernel	3	7	47	14	9	—	1	19	1	—	—	—	—	—
Rapeseed (B. campestris)	—	—	—	—	4	—	2	33	18	9	—	12	—	22
Rapeseed (B. napus)	—	—	—	—	3	—	1	17	14	9	—	11	—	45
Jatropha *curcas*[3]	—	—	—	—	13	—	8	45	34	—	—	—	—	1
Salicornia *bigevolii*[4]	—	—	—	—	7	—	3	18	73	—	—	—	—	—

Notes:
1) Unless otherwise indicated, this information comes from: DeMan et al. (1999)
2) Additional data for fatty acid components of coconut oil, canola oil, sunflower oil, cottonseed oil, peanut oil, olive oil, mustard seed, lard, yellow grease is also available from DeMan et al. (1999)
3) From Shweta et al. (2004)
4) Salicornia oil is similar to safflower oil in fatty acid composition (Glenn et al., 1998). This profile is averaged values for safflower oil from: Coşge et al. (2007)

Equation 8 can be re-written in a mass balance form for easier comparison to experimental data.

1.12 lb Oil + **0.024 lb** Hydrogen → **1.00 lb** HRD + **0.14 lb** Water
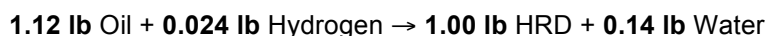
The feedstock, key products and process energy needed per pound of HRD are summarized in Table 42.

Table 42: Experimental and theoretical requirements for the creation of renewable diesel

Feedstock (lb)	Experimental			Theoretical
	Low	Baseline	High	
Oil	100	100	100	100
H$_2$ (51586 Btu/lb)	1.5	2.72	3.8	2.14
Key Products (lb)				
HRD (18908 Btu/lb)	83.0	84.19	86	89.28
Propane Mix Gas (18568 Btu/lb)	2.0	4.75	5	0.00
Process Energy (Btu)				
Electricity	5785	6942	8099	--
Natural Gas	8950	8950	8950	--

Notes:
1) Steam is assumed to be produced from natural gas at 80% efficiency
2) Energy contents are taken from GREET (2008)
3) Experimental data taken from Appendix 2 of Huo et al. (2008) with modifications per recommendations from UOP (Kalnes, 2009)

Further refinement of HRD is required for the creation of HRJ. The strategy to estimate the process requirements of HRJ was to use the experimental data for the creation of HRD and subsequently estimate the additional requirements to convert the HRD into HRJ. For this analysis, HRD and HRJ are assumed to be symmetrical distributions of straight chains hydrocarbons centered on C$_{12}$ and C$_{18}$ respectively. As shown schematically in Figure 11, the cracking from diesel fuel to jet fuel was assumed to occur via the addition of gaseous hydrogen.
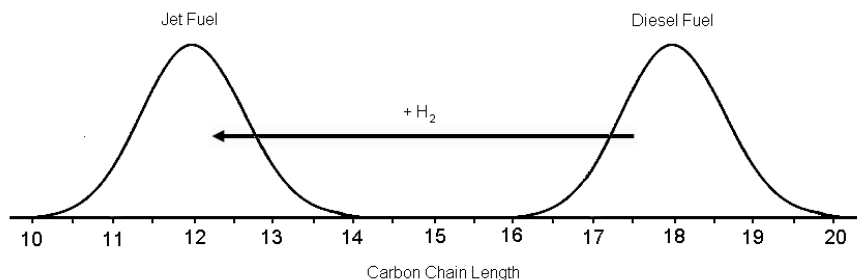
Figure 12: Schematic showing the change in hydrocarbon composition between HRD and HRJ fuels that results from additional hydroprocessing.

The mechanism by which hydrocarbon chains crack is through smaller molecules, (e.g., pentane (C_5H_{12}) and hexane (C_6H_{14})) breaking off the end. The dominant effect that takes place is the reduction of C_{18} to C_{13} by cracking pentane and C_{12} by cracking hexane (Kalnes, 2009). Other reactions are also occurring where molecules from the distribution about C_{18} crack to those from the distribution about C_{12}. To account fully for this effect would entail including the statistical nature by which chemical reactions are more likely to take place. In keeping with the level of detail required of this analysis, the aforementioned effects were assumed to cancel out if the two distributions have the same shape about their mean.

Making use of these arguments simplifies the analysis to two chemical reactions converting diesel fuel to jet fuel:

$$\frac{\begin{aligned} C_{18}H_{38} + H_2 &\rightarrow C_{12}H_{26} + C_6H_{14} \\ C_{18}H_{38} + H_2 &\rightarrow C_{13}H_{28} + C_5H_{12} \end{aligned}}{2C_{18}H_{38} + 2H_2 \rightarrow C_{12}H_{26} + C_{13}H_{28} + C_5H_{12} + C_6H_{14}}$$
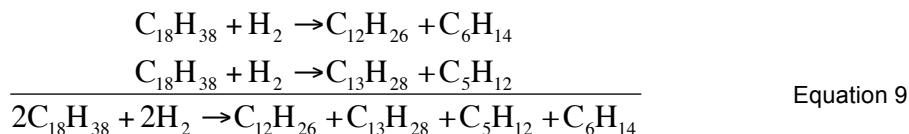
Equation 9

Written in mass balance form and normalized for one lb of HRD, the overall equation governing the formation of HRJ from renewable oil can be expressed as:

1.00 lb HRD + **0.0079 lb** Hydrogen → **0.697 lb** HRJ + **0.311 lb** Naphtha
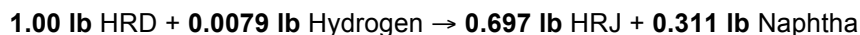
Although variations in carbon chain length are not captured in this analysis, fatty acids with carbon chain length distributions around 12 are better suited for use as feedstock for jet fuel because higher blending percentages can be used without the need for hydrocracking. Furthermore, carbon chain lengths closer to 12 would result in a higher yield of jet fuel per unit mass of oil input.

Naphtha in this case is a combination of 46% C_5H_{12} and 54% C_6H_{14} by mass. Using these ratios of HRD to HRJ, the process energies from Table 42 were modified to reflect the energy requirements to create HRJ as shown in Table 43. Based on discussions with experts at UOP (Kalnes, 2009), it is assumed that total process energies (natural gas and electricity) will increase by 10% to 30% per pound of renewable feedstock when including the hydrocracking required for the formation of HRJ. The total hydrogen consumption is the sum of that needed to first make HRD and then to crack it to HRJ. In all cases where renewable oils are processed into finished fuel products, energy and emissions were allocated based on energy content.

Table 43: Energy requirements for the creation of HRJ

Feedstock (lb)	Emissions Scenarios		
	Low	Baseline	High
Oil	100	100	100
H_2 (51586 Btu/lb)	2.15	3.38	4.48
Key Products (lb)			
HRJ (18950 Btu/lb)	57.8	58.7	59.9
Naphtha (19215 Btu/lb)	25.8	26.2	26.8
Propane Mix Gas (18568 Btu/lb)	2.0	4.8	5.0
Process Energy (Btu)			
Electricity	6364	8330	10529
Natural Gas	9845	10740	11635

The hydroprocessing step of converting renewable oil into HRD has a mass yield of 84% and results in 8.7 gCO_2/MJ. After making the aforementioned changes, hydroprocessing renewable oil into HRJ has a mass yield below 60% and results in 10.3 gCO_2/MJ. The emissions associated with hydrogen production in this work are representative of steam reforming natural gas and are consistent with default GREET assumptions. While these results provide a first approximation of mass and energy inputs, they do not reflect the impact of oil composition on process inputs or differences in real world production scenarios.

The decrease in yield is accompanied by an increase in naphtha production. Although naphtha is used as a blending stock in gasoline, it has a lower economic value than HRD, which is a high performance diesel fuel. Because petroleum refineries have on-site hydrogen production to meet internal demands that could be expanded to supply the hydrogen for hydroprocessing renewable oils, hydroprocessing facilities would likely have lower emissions if they were integrated into existing petroleum refineries. With such a configuration, excess naphtha resulting from HRJ production can be integrated into the refinery naphtha stream and catalytically reformed to high value hydrocarbons or steam reformed to supplement the internal hydrogen needs of the process. However, if the value of the HRJ and naphtha stream is less than the HRD stream, a fuel producer would likely focus on HRD production unless they could charge a premium for HRJ fuel. Under such a situation, the aviation community could consider using HRD as a dilute blend stock in gas turbine engines (~10%). This option would require further research to ensure viability with the current fleet of aircraft. Further research is being devoted to understand the tradeoffs among HRD, HRJ, and FAME production.

7.2 HRJ from Soybean Oil

The production of HRD from soybean oil is available in GREET (as documented in Huo et al., 2008). Default assumptions on farming energy, fertilizer use, yield, etc, were updated where necessary using data from the literature (mainly the US Department of Agriculture (USDA) and Hill et al., 2006). The key parameters chosen for establishing the low emissions, baseline, and high emissions scenarios were yield per hectare, N_2O emissions, liming emissions and the processing emissions whereby soy oil is converted into HRJ (from Section 7.1). While one case assumed zero land use change emissions, two other independent land use change scenarios were examined; within which each of the aforementioned key parameters were varied.

7.2.1 Cultivation of Soybeans

The farming energy data used was based on 2002 survey results provided by USDA's Economic Research Service.[39] These data are shown in Table 44.

Table 44: Farming energy required for average US soybean production

Fuel type	Farming Fuels (gal/ha)	Farming energy (Btu/ha)[2]	Fuel share (%)
Diesel	11.9[1]	1523500	70.1
Gasoline	3.2	372900	17.1
LPG	1.0	83960	3.9
Electricity	19.3 (kWh/ha)	65780	3.0
NG	3.7 (m^3/ha)	127500	5.9
Total	n/a	2174000	100
Notes: 1) Diesel use included custom work of 1.7 gal/acre (Hill et al., 2006; Sheehan et al., 1998a). 2) LHV values of fuels (except electricity) were obtained from GREET (GREET, 2008).			

Soybean production and yield[40] data were based on historical and projected US data. The baseline case assumes a projected soybean yield in 2015 of 110 bu/ha[41] (FAPRI, 2009). The low and high emissions scenarios were based on analysis of historical soybean yield data from 1987 through 2007 (USDA, 2008a). The variation between the lowest yield and the mean was -28% while that between the highest yield and the mean was +17%.[42] Based on these historical data, it was assumed that the yield in some future drought year could be 28% lower than in the baseline case, corresponding to the high emissions case of 79.5 bu/ha. Similarly, a yield in some future ideal growing year could be 17% higher than in the baseline case, corresponding to the low emissions case of 129 bu/ha.

Using the farming energy per hectare and the soybean yield per hectare, the farming energy per bushel of soybean was determined as shown in Table 45. The high emissions case includes the energy used to grow and process the soybean seed (i.e. seeding energy) as estimated by Hill et al. (2006) to be an additional 4.5% of total energy inputs (including farming and fertilizer energy).

The data on fertilizer (nitrogen, phosphorus and potassium), herbicide and insecticide use in soybean cultivation from recent years was obtained from the USDA (USDA, 2008b). Glyphosate was the most commonly applied herbicide in soybean cultivation. The energy data for the manufacture of glyphosate are not available in GREET, so the energy requirements provided by Hill et al. (2006) (475 MJ/kg) were used. Default GREET assumptions for herbicide energy use were applied for the remaining proportion of non-glyphosate herbicide use. Insecticide and fungicide are also applied in soybean cultivation, but in negligible amounts (e.g. 0.4 g/bu of insecticide and 0.1 g/bu of fungicide were applied in 2006) compared to fertilizer and herbicide application. Hence, these were not considered here. The average agrichemical use from 2002, 2004 and 2006 was adopted in all cases (see Table 46).

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[39] Via e-mail from Kathleen Kassel to Hsin Min Wong on February 28, 2008.

[40] Yield is defined as total production divided by total planted area.

[41] Based on average of projections in the years 2014/15 and 2015/16. One bushel of soybeans is defined as 60lbs.

[42] The average yield between 1987 and 2007 was 90.2 bu/ha; the lowest and highest yields within this time period were 65.0 bu/ha and 105.0 bu/ha, respectively.

Table 45: Farming energy for US soybean production for low emissions, baseline and high emissions cases

		Low	Baseline	High[1]
Farming fuel energy (Btu/bu)	Soybean yield (bu/ha)	129	110	79.5
	Diesel	11,789	13,793	20,009
	Gasoline	2,886	3,376	4,898
	LPG	650	760	1,103
	Electricity	509	596	864
	NG	987	1154	1,675
	Total	16,820	19,680	28,549
Notes: 1) Includes an additional 4.5% energy to grow and process seeds.				

Table 46: US soybeans agrichemical use

Agrichemical use (g/bu)	Year			
	2002	**2004**	**2006**	**Average**
Nitrogen	52	64	31	49
Phosphorus	154	196	114	155
Potash	311	310	214	278
Herbicide (Percentage of glyphosate in herbicide use)	15 (73.6%)	13 (83.1%)	15 (88.8%)	14 (81.8%)

Emissions from N_2O were estimated using a combination of GREET and IPCC Tier 1 methodology (De Klein et al., 2006) in the low and baseline scenarios and from Hill et al. (2006) in the high emissions scenario. Specifically, GREET calculated the total amount of nitrogen (in the form of soybean biomass) left on the field as 200.7 g/bushel of soybeans harvested. The IPCC Tier 1 methodology estimates the combined direct and indirect conversion rate for nitrogen from synthetic fertilizers as 1.325% and nitrogen from crop residues as 1.225%. These rates include the atmospheric deposition of nitrogen volatilized from managed soils as well as nitrogen from leaching and runoff. The formula for calculating N_2O emissions from soybean cultivation is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{bu}\right) = \left(\left(\frac{g_{\text{nitrogen fertilizer}}}{bu}\right) \cdot 0.01325 + \left(200.7 \frac{g_{\text{N-crop residue}}}{bu}\right) \cdot 0.01225\right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N}\right) \qquad \text{Equation 10}$$

Assuming a nitrogen fertilizer application rate of 49 g/bu, total N_2O emissions are 4.9 g/bu. Hill et al. (2006) assumed N_2O emissions to be an all-encompassing number of 1760 g/ha based on conventional tillage and rotation of corn, soybean and wheat. Under a soybean yield of 110 bu/ha, emissions from N_2O would be 16 g/bu, roughly 3 times higher than those estimated by GREET.

The scenario analysis adopts zero nitrogen fertilizer use in the low emissions case,[43] the estimates of GREET in the baseline case and those from Hill et al. in the high emissions case. Hence, N_2O emissions were 3.9 g/bu in the low case, 4.9 g/bu in the baseline case and 22 g/bu in the high emissions case (the high emissions case assumes a soybean yield of 79.5 bu/ha).

Lime (composed primarily of calcium carbonate) can be added to soil to reduce acidity and increase nutrient uptake. However, the use of lime must be accounted for in the overall life cycle analysis of soybean farming emissions. GREET does not account for lime use in soybean

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[43] Growing condition for soybeans around the US vary substantially. While the average nitrogen usage across the US is 49g/bu, it is uncommon for soybeans to respond to pre-plant fertilizer N applications in the central US soybean belt (Franzen, 1999). In many cases, soybeans fix nitrogen into the soil and are planted prior to corn to increase soil nitrogen in soy/corn/wheat rotation farms.

cultivation. Hill et al. (2006) estimate 230 kg/ha of CO_2 from liming. The energy and emissions associated with the manufacture of lime were assumed negligible.[44]

The low emission case assumes zero liming and the high emissions case uses the full CO_2 emissions from Hill et al. (230 kg/ha); the baseline case uses the average of these extremes (115 kg/ha). The soybean yields from Table 45 result in 1045 gCO_2/bu (using a yield of 110 bu/ha) in the baseline case and 2893 gCO_2/bu (using a yield of 79.5 bu/ha) in the high emissions case. A summary of the parameters used in the cultivation of soybeans is given in Table 47.

Table 47: Input assumptions for the cultivation of soybeans for the low emissions, baseline and high emissions cases

Input parameter	Low	Baseline	High
Soybean yield (bu/ha)	129	110	79.5
Farming energy (Btu/bu)	16,820	19,680	28,549
Fertilizer inputs (g/bu)			
Nitrogen	0	49	n/a
Phosphorus	155	155	155
Potash	278	278	278
Herbicide (g/bu)	14	14	14
Percentage of glyphosate in herbicide use (%)	81.8	81.8	81.8
N_2O emissions (g N_2O/bu)	3.9	4.9	22
Liming emissions (g/bu)	0	1,045	2,893

7.2.2 Extraction of Oil from Soybeans

The energy needed for the crushing and extraction of oil from soybeans was based on GREET default assumptions (~5,800 Btu/lb soy oil produced). GREET data is based on the study by Sheehan et al. (1998a), but adjusted to reflect potential technological improvements (about 10% less total process energy than the value estimated by Sheehan et al., 1998a). The inputs, outputs and process energy needed per lb of processed soy oil are summarized in Table 48.

Table 48: Input, output and process energy for extraction of oil from soybeans

Input (lb)	
Soybean	5.70
Output (lb)	
Soy oil	1
Soy bean meal	4.48
Process energy (Btu)	
Electricity	551
Natural Gas	5134
N-hexane	182
Total	5867[1]
Note:	
1) GREET default value of 5867 Btu for simulation year 2010 was adjusted to 5800 Btu for the simulation year 2015 and used here.	

7.2.3 Co-Product Usage and Allocation Methodology

In the extraction of soy oil from soybeans, soybean meal is produced in large quantities (by mass relative to soy oil) as a co-product. Four methods were explored to allocate the energy and

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[44] The errors introduced by this assumption are likely small because the energy and emissions that arise from the manufacture of lime are small compared to those of other fertilizers and herbicides, (e.g., nitrogen fertilizer manufacture energy and CO_2 emissions are 42 mmBtu/ton and 2.5 Mg/ton, respectively, compared to 7 mmBtu/ton and 0.5 Mg/ton for lime).

emissions of soybean farming and soy oil extraction between soy oil and soybean meal. These were allocation by mass, energy, market value, and the displacement (system expansion) method. The resultant allocation of GHG emissions between soy oil and soybean meal based on the four methods are given in Table 49.

Soybean meal is primarily used as an animal feed and could potentially displace barley, corn, and soybean. The amount of product that soybean meal would displace was calculated on a protein equivalence basis (i.e. matching of protein content). Soybean meal was assumed to have protein content of ~48% (Ahmed et al., 1994). Avoidance of land use change emissions associated with the cultivation of the displaced crops was taken into account in the analysis for illustrative purposes.

7.2.3.1 Displacement of Barley

The protein content of barley was assumed to be 12% (USDA, 2008c; OMAFRA, 2003; Kennelly et al., 1995). Hence, 1 lb of soy meal displaces 4 lbs of barley on a protein equivalence basis. As 0.38 lbs of soy meal is produced per MJ of fuel, about 1.5 lbs of barley is displaced per MJ of fuel. From the life cycle assessment of GHG emissions of barley cultivation conducted by Lechón et al. (2005), the CO_2 credit is 71.5 g/lb of barley displaced (71.3 gCO_2/MJ fuel) and N_2O credit is 0.36 g/lb of barley displaced (107.0 gCO_2e/MJ fuel). Lechón et al. did not include land use change emissions associated with the farming of barley. This work assumed the land use change credit associated with the displaced barley was the avoidance of the conversion of Brazilian Cerrado grassland to cropland (~2,833,333 gCO_2/ha when emissions were amortized over 30 years, Fargione et al, 2008). Assuming a barley yield of 5940 lb/ha (Lechón et al., 2005), the resultant land use change credit was 477 gCO_2/lb of barley displaced (476g CO_2/MJ).

7.2.3.2 Displacement of Corn

Corn is a widely used animal feed that could be displaced by soy meal. The protein content of corn is about 9% (OMAFRA, 2003; Parish, 2007; Kennelly et al., 1995). Hence, 1 lb of soybean meal displaces 5.3 lbs of corn on a protein equivalence basis. As 0.38 lbs of soybean meal is produced per MJ of fuel, about 2.0 lbs of corn is displaced per MJ of fuel. The life cycle GHG emissions for corn farming were analyzed in GREET using default GREET values (version 1.8b, GREET, 2008). Specifically, the CO_2, CH_4 and N_2O credits per lb of corn displaced were 100.6 g/lb (133.3 gCO_2/MJ fuel), 0.15 g/lb (5.0 gCO_2e/MJ fuel) and 0.22 g/lb (86.9 gCO_2e/MJ fuel), respectively. In the case of corn displacement, land use change credit was assumed to result from the avoidance of worldwide conversion of non-cropland to cropland for corn cultivation as simulated by Searchinger et al. (2008) (11.7 Mg CO_2/ha when emissions were spread over 30 years). Assuming a corn yield of 150 bu/acre (average yield from 2003-2007, USDA, 2008a), the land use change credit was estimated as 1393 g/lb of corn displaced (1846.0 gCO_2/MJ fuel).

7.2.3.3 Displacement of Soybeans

Soybean is the default product assumed to be displaced by soybean meal in GREET. The protein content of soybeans was assumed to be 40% (GREET, 2008). Hence, 1 lb of soybean meal displaces 1.2 lbs of soybean on a protein equivalence basis. As 0.38 lbs of soybean meal is produced per MJ of fuel, 0.46 lbs of soybean is displaced per MJ of fuel produced. Based on the assumptions of the baseline scenario, the CO_2, CH_4 and N_2O credits per lb of soybeans displaced were 71.0 g/lb (21.3 gCO_2/MJ fuel), 0.08 g/lb (0.57 gCO_2e/MJ fuel) and 0.09 g/lb (7.9 gCO_2e/MJ fuel), respectively. In considering land use change emissions, worldwide conversion of non-cropland to cropland for corn cultivation as simulated by Searchinger et al. (2008) was used (11.7 Mg CO_2/ha when emissions were spread over 30 years). This scenario was assumed to broadly apply to the case of soybean farming. Assuming a soybean yield of 110 bu/ha (as in the baseline case described above), the land use change credit was estimated to be 1773 g/lb of soybean displaced (530 gCO_2/MJ fuel).

7.2.3.4   Results

Table 49 shows six different approaches to allocation between soy oil and soy meal. The first three rows show allocation by mass, market value and energy while the rest shows the displacement calculation described above.

Table 49: Allocation of GHG emissions between soy oil and soy meal using various methodologies

Allocation Approach		Soy Oil (%)	Soy Meal (%)
Mass		18.2	81.8
Market Value[1]		44.7	55.3
Energy[2]		33.9	66.1
Displacement of barley	LUC included for soy meal and barley	-5.4	105.4
	LUC included for soy meal but not barley	71.3	28.7
	no LUC	-207.6	307.6
Displacement of corn	LUC included for soy meal and corn	-232.9	332.9
	LUC included for soy meal but not corn	63.8	36.2
	no LUC	-287.4	387.4
Displacement of soybean	LUC included for both soy meal and displaced soybeans	9.8	90.2
	LUC included for soy meal but not displaced soybeans	95.3	4.7
	no LUC	49.5	50.5
Notes: 1) Market value of soy oil and soy meal are $1.05/kg and $0.29/kg (FAPRI, 2009) 2) Energy content of soy oil and soy meal are 37.2 MJ/kg and 16.2 MJ/kg (GREET, 2008) 3) Life cycle GHG emissions from soybean production, soy oil extraction and land use change are 31.0 gCO_2e/MJ, 26.9 gCO_2e/MJ, 562.1 gCO_2e/MJ of fuel produced 4) Conversion ratio used is 0.587 lb_{HRJ}/lb_{oil} and energy allocation to HRJ after hydroprocessing is 65.3% 5) LUC refers to land use change			

7.2.3.5   Discussion

The use of different co-product allocation approaches can result in substantially different results, particularly when significant amounts of co-product are produced and different allocation approaches are appropriate for different scenarios and pathways. The displacement method, as applied here, is not appropriate because of the large variation in life cycle GHG emissions that would result from the choice of feedstock (i.e., barley, corn, and soybean) displaced by soy meal and how land use change are addressed. A more appropriate use of the displacement method would be achieved by modeling worldwide agriculture; however, that is beyond the scope of the analysis presented here.

In this case, the mass or energy allocation method may not be most appropriate as soy meal is not valued based on its mass, or sold as a commercial energy product. Further, allocation by mass would benefit a fuel producer, as it will result in most of the emissions being assigned to the meal instead of the oil. As discussed with regard to the product slate from a Fischer Tropsch facility, a general shortcoming in the displacement approach arises when a large quantity of co-product is generated relative to the primary product. This causes an overestimation of credits for the co-product and may even lead to net negative emissions. Because the high protein content of soy meal leads to commercial value as an animal feed, market value allocation was adopted here

and in all cases where oil extraction from an oilseed results in meal leaving the system and an independent product. Market value allocation is further explored in the next section.

7.2.3.6   Market Value Allocation with Soy Oil to HRJ

In using the market value allocation approach, temporal fluctuations in the market prices of co-products could cause temporal changes in the allocation values. The sensitivity to market forces is apt in this case as it allocates shares of the energy and emissions from production based on the current utility (of which price is the most reasonable measure) for each product. If a market becomes saturated with a co-product then the utility and market value will both approach zero. Market fluctuations make getting a consistent result challenging. Figure 13 shows the temporal variation in the allocation fraction between soy oil and soy meal from 2005 through 2015. Price projections are based on the database maintained by FAPRI (FAPRI, 2009) and physical quantities are based on Table 48. The dashed lines correspond to the constant values used within GREET (GREET, 2008).
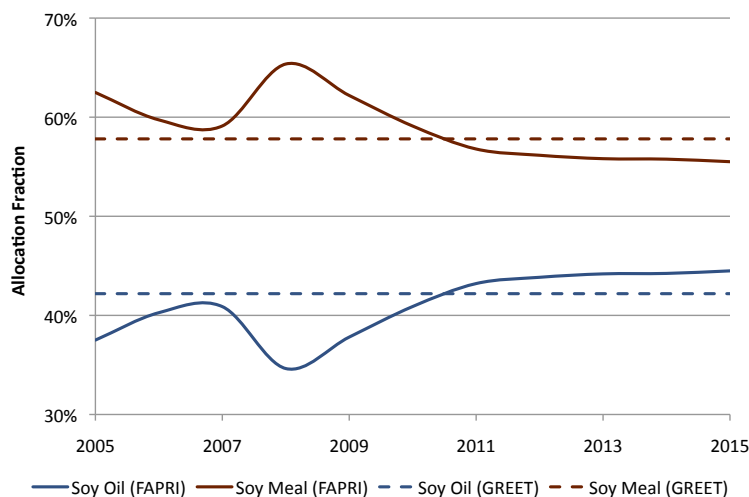
Figure 13: Temporal variation in the soy oil and soy meal allocation fractions when using market value allocation. Constant dashed lines correspond to default GREET prices

Although the GREET default values turn out to be a reasonable mean of actual prices over time, this study used the 2015 FAPRI projects prices of $1.05/kg and $0.29/kg for soy oil and soybean meal, respectively (FAPRI, 2009).

7.2.4   Transportation of Soy Oil to HRJ Production Facilities

The estimation of GHG emissions arising from transportation (of feedstock and fuel) is based on default GREET assumptions.

7.2.5   Land Use Change Emissions from Soy Oil Production

The use of food crops as a source of renewable oils for fuel production may lead to emissions from land use change. The magnitude of land use change emissions can depend on the type of converted land, the type of crops grown and the farming practices employed. In addition to a scenario where no land use change emissions were incurred, two scenarios of land use change were chosen to explore the range of magnitudes of GHG emissions due to land use change. The goal of this section is to provide the reader an understanding of how land use change emissions compare to the emissions from the other five life cycle stages. It is not intended to explicitly quantify the specific land use change emissions that would result from expanded soy oil production.

The first scenario considers the conversion of Cerrado grassland in Brazil to soybean fields and the second scenario considers converting tropical rainforests in Brazil to soybean fields (Fargione et al., 2008). Using these Brazilian estimates with the present analysis of soybean production implies there is little difference between soybean cultivation in Brazil and soybean cultivation in the US. It further implies a transportation profile for the transportation of soy oil from Brazil to the US; however, the magnitude of the GHG emissions resulting from this additional transportation were assumed to be negligible relative to the GHG emissions from either LUC scenario. In all scenarios, the impact of varying soybean yield was explored on land use change emissions amortized over 30 years. The impacts per bushel of soybeans are shown in Table 50.

Table 50: Land use change emissions arising from soybean production

Scenario	Land use change emissions (Mg CO$_2$/ha)	Soybean yield (bu/ha)	Land use change emissions (g CO$_2$/bu)[3]
Grassland conversion[1]	85[2]	79.5 110 129	35,600 25,800 22,000
Tropical rainforest conversion[1]	737[2]	79.5 110 129	309,000 223,300 190,400
Notes: 1) Soybean yields in Brazil were assumed to be similar to soybean yields in the US (FAPRI, 2009) 2) From Fargione et al., 2008 3) Amortized over 30 years (no discounting)			

7.2.6 Results

The life cycle GHG emissions from the production and use of HRJ from soy oil for each land use change option are summarized Tables 51-53. Without land use change emissions, the life cycle GHG emissions of the production of HRJ from soy oil range from 0.31 to 0.68 times those of conventional jet fuel. However, as illustrated in the various land use change scenarios, land use change emissions can dramatically increase the overall life cycle GHG emissions of the fuel to several times that of conventional jet fuel. Specifically, the land use change scenarios investigated in this work result in life cycle GHG emissions ranging from 0.9 to nearly 9 times those of conventional jet fuel.

Table 51: Life cycle emissions for the soy oil to HRJ assuming zero land use change emissions

Land Use Change Scenario S0	Low	Baseline	High
Key Assumptions			
Soybean yield (bu/ha)	129	110	79.5
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO_2/MJ)	-73.7	-70.5	-68.9
Recovery of feedstock (gCO_2/MJ)	17.5	20.1	25.9
Transportation of feedstock (gCO_2/MJ)	1.2	1.2	1.1
Processing of feedstock to fuel (gCO_2/MJ)	7.1	10.3	13.2
Transportation of jet fuel (gCO_2/MJ)	0.6	0.6	0.6
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	-47.3	-38.4	-28.1
WTT CH_4 emissions (gCO_2e/MJ)	1.2	1.3	1.5
WTT N_2O emissions (gCO_2e/MJ)	3.0	3.6	15.4
Total WTW GHG Emissions (gCO_2e/MJ)	27.3	37.0	59.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.31**	**0.42**	**0.68**

Table 52: Life cycle emissions for the soy oil to HRJ assuming conversion of Cerrado grassland to soybean field

Land Use Change Scenario S1	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO_2/MJ)	54.4	60.8	82.5
WTW CO_2 emissions (gCO_2/MJ)	77.4	92.8	124.9
Total WTW GHG Emissions (gCO_2e/MJ)	81.7	97.8	141.7
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.93**	**1.12**	**1.62**

Notes:
1) All other input assumptions (cultivation of soybeans, extraction of soy oil, processing of soy oil to HRJ) are based on those in the S0 emissions case of the corresponding scenario.

Table 53: Life cycle emissions for the soy oil to HRJ assuming conversion of tropical rainforest

Land Use Change Scenario S2	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO_2/MJ)	471.5	527.2	715.5
WTW CO_2 emissions (gCO_2/MJ)	494.5	559.2	757.8
Total WTW GHG Emissions (gCO_2e/MJ)	498.8	564.2	774.7
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**5.70**	**6.45**	**8.85**

Notes:
1) All other input assumptions (cultivation of soybeans, extraction of soy oil, processing of soy oil to HRJ) are based on those in the S0 emissions case of the corresponding scenario.

7.3   HRJ from Palm Oil

A palm oil to HRJ pathway was created within the GREET framework using information on farming energy, fertilizer use, yield, oil extraction energy, etc., obtained from the literature, mainly from the Malaysian Palm Oil Board and Wicke et al. (2007).[45] The extraction of oil from palm

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[45] Palm oil is not a preexisting pathway within GREET. As such, a new pathway was built within the GREET framework using the soy oil to renewable diesel pathway as a guide.

fresh fruit bunches (FFB) yields two types of oil: palm oil and palm kernel oil. Both oil types were assumed to be main products in the FFB extraction process and used in the production of HRJ.[46] The only co-product from the process is palm kernel expeller, which can be used as an animal feed. The key parameters chosen for establishing the low emissions, baseline, and high emissions cases were palm fresh fruit bunch yield per acre, farming energy, methane emissions from palm oil mill effluent treatment and processing of palm oil into HRJ[47] (from Section 7.1). In addition to a scenario where land use change emissions were assumed zero, three land use change scenarios were examined where the aforementioned key parameters were varied.

7.3.1   Cultivation of Palm Fresh Fruit Bunches (FFB)

Using a process similar to that of soybean-to-HRJ, the yields of palm FFB were derived from historical data spanning 1987 through 2007 in Malaysia (MPOB, 2008a). The data was extrapolated using a linear fit to estimate the yield in 2015.[48] The resultant yield of 21.3 ton/ha (all references in this section to tons refer to short tons) was adopted in the baseline case. The percentage variations in the data between the lowest yield and mean, and the highest yield and mean were found to be -14% and +9%, respectively. The high emissions case assumed a yield 14% lower than the baseline case, corresponding to 18.2 ton/ha. The low emissions case assumed a yield 9% higher than the baseline case, corresponding to 23.1 ton/ha.

In Southeast Asian oil plantations, most of the cultivation and harvest of the palm FFB are done manually with the help of animals. The main fossil energy used is diesel fuel for powering farm machinery and equipment. This work assumed that 3.0 mmBtu/ha of diesel was consumed in the low emissions and baseline cases, while 4.9 mmBtu/ha was used in the high emissions case (Wicke et al., 2007). The farming energy per ton of FFB harvested was estimated by combining the farming energy per hectare with the yield of palm FFB (see Table 54).

Only nitrogen fertilizer was considered in this study as the GHG emissions contributions from the production of phosphate and potash fertilizers, as well as herbicides, were negligible compared to those of nitrogen fertilizers (Wicke et al., 2007). The application of organic fertilizers (e.g. empty fruit bunches) was not considered because their emissions are carbon-neutral. The nitrogen fertilizer application rate adopted in all three cases (low, baseline and high) was 139 kg/ha (Wicke et al., 2007). The corresponding nitrogen fertilizer application in terms of grams of nitrogen per ton of FFB was calculated based on the yields in each scenario.

Emissions from N_2O were estimated using IPCC Tier 1 methodology (De Klein et al., 2006). Due to the lack of data, N_2O emissions from nitrogen in above and below ground crop residues were not accounted for in this analysis. The IPCC Tier 1 methodology estimates the combined direct and indirect conversion rate for nitrogen from synthetic fertilizers as 1.325%. This rate includes the atmospheric deposition of nitrogen volatilized from managed soils as well as nitrogen from leaching and runoff. The formula for calculating N_2O emissions from the cultivation of palm fresh fruit bunches is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{ton_{FFB}} \right) = \left(\left(\frac{g_{nitrogen\,fertilizer}}{ton_{FFB}} \right) \cdot 0.01325 \right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right) \qquad \text{Equation 11}$$

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[46] This is in contrast to other studies where diesel is the desired product from renewable oil processing and palm kernel oil, whose carbon number distribution centered on C_{12} is outside the diesel range, is considered a by-product instead of a main product in the FFB extraction process.

[47] Although palm kernel oil has a shorter carbon length (mostly 12 carbons in length) than palm oil (mostly 16 and 18 in length), for the purposes of this study, they are both assumed to be 18 carbon atoms in length. This was discussed briefly in Section 7.1.

[48] From the linear fit, the yield was estimated to increase at a rate of about 0.0435 tonnes per year from 18.08 tonnes in 1987.

A summary of the parameters assumed in the cultivation of palm FFB discussed above is given in Table 54.

Table 54: Assumptions in the cultivation of palm fresh fruit bunches for the low emissions, baseline and high emissions cases

Input parameter	Low	Baseline	High
FFB yield (ton/ha)	23.1	21.2	18.2
Farming energy (Btu/ton FFB)	131,300	143,100	270,800
Nitrogen fertilizer inputs (g/ton FFB)	6,020	6,560	7,640
N_2O emissions (g N_2O/ton FFB)	125.3	136.6	159.1

7.3.2 Extraction of Oil from Palm FFB

Palm oil and palm kernel oil are produced from palm FFB at a processing mill. During the extraction process, palm kernel shells (PKS), fiber, empty fruit bunches (EFB) and kernels are also produced. Palm Kernel Effluent and fiber are burnt to generate electricity to fuel mill processes, while EFB are used as organic fertilizers. Palm kernels can be further processed to produce palm kernel oil and palm kernel expeller, which can be used as an animal feed. A waste stream called palm oil mill effluent (POME) is also produced from the extraction of oil from FFB. POME treatment (through anaerobic digestion) results in the emission of CH_4.

The palm oil extraction rate, kernel recovery rate, palm kernel oil extraction rate, palm kernel expeller extraction rate and the energy inputs needed for the crushing and extraction of oil from palm FFB and palm kernels were derived from data from the Malaysian Palm Oil Board (MPOB, 2008b) and Wicke et al. (2007). The input parameters, energy consumption and product yield are shown in Table 55. The total process energy of 199.5 Btu/lb oil was used in all the three scenarios (low emissions, baseline and high emissions cases).

The treatment of palm oil mill effluent results in methane production. For every ton of FFB processed, between 1.5 to 2.2 kg of CH_4 is emitted from POME treated in an open digesting tank (Reijnders and Huijbregts, 2008). This work assumed methane emissions of 1.5 kg CH_4/ton FFB in the low emissions case, 1.85kg CH_4/ton FFB (average value) in the baseline case, and 2.2 kg CH_4/ton FFB in the high emissions case. The resultant CH_4 emissions per pound of oil are given in Table 56.

7.3.3 Co-Product Usage and Allocation Methodology

The market allocation method was used to apportion emissions among palm oil, palm kernel oil and palm kernel expeller[49] to be consistent with the approach taken in the soy oil pathway. The market prices assumed were $0.98/kg for palm oil, $1.22/kg for palm kernel oil and $0.14/kg for PKE, based on average export prices between January and October 2008 (MPOB, 2008b).

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[49] In the extraction of palm oil from FFB and palm kernel oil from kernels, co-products such as palm kernel shell, fiber, empty fruit bunches and palm kernel expeller are formed. As palm kernel shell and fiber are used for electricity production in the mill and empty fruit bunches as fertilizers in palm FFB cultivation to reduce energy requirement and fertilizer input, respectively, they are recycled streams within the process rather than co-products. In addition, the emissions generated by the use of these products are considered carbon-neutral as they were previously absorbed from the atmosphere during growth. Hence, palm kernel expeller is the only co-product considered in the allocation of process energy and emissions.

Table 55: Inputs, outputs and process energy in the extraction of palm oil and palm kernel oil

Extraction of palm oil from FFB	
Palm oil extraction rate (ton palm oil/ton FFB)[1]	0.20
Amount of kernels produced (ton kernels/ton palm oil)[1]	0.25
Energy input Diesel (litre/ton FFB)[2,3]	1.36
Extraction of palm kernel oil from kernels	
Palm kernel oil extraction rate (ton PKO/ton kernel)[1]	0.45
Palm kernel expeller extraction rate (ton PKE/ton kernel)[1]	0.51

Energy input[2]
    Electricity (kWh/ton kernel)
    Diesel (litres/ton kernel)
	
	77.3
	17.3

Overall energy use and product yields	
Input FFB (lb)	 4.5

Output
    Palm oil and palm kernel oil (lb)
    Palm kernel expeller (lb)
	
	1
	0.11

Process energy
    Electricity (Btu/lb oil)
    Diesel (Btu/lb oil)
    Total (Btu/lb oil)
	
	29.7
	169.8
	199.5

Notes:
1) MPOB, 2008b.
2) Wicke et al., 2007.
3) The amount of diesel energy needed for electricity generation is reduced by the use of palm kernel shell and fiber (by-products of palm oil extraction) for electricity production.

Table 56: Methane emissions from POME treatment

	Low	Baseline	High
CH_4 emissions from POME treatment (kg CH_4/ton FFB)	1.5	1.85	2.2
CH_4 emissions from POME treatment (g CH_4/lb oil)	3.32	4.15	4.98

7.3.4   Transportation of Palm Oil and Palm Kernel Oil to HRJ Production Facilities

It is assumed that palm oil and palm kernel oil are produced in facilities near palm plantations in Southeast Asia and then transported to the United States for hydroprocessing to HRJ. This transportation pathway is not available in GREET. The key assumptions used in this work are summarized in Table 57.

Table 57: Assumptions in the transportation of palm oil and palm kernel oil from Southeast Asia to the US

Transportation of palm oil and PKO from mill to harbor in Southeast Asia	
Mode (%)	
Truck	100
Distance (miles)	63[1]
Transportation of palm oil and PKO from Southeast Asia to US ports[2]	
Mode (%)	
Ocean tanker	100
Distance (miles)	
Gulf Coast (50%)	13500
East Coast (25%)	12300
West Coast (25%)	7700
Average	11750
Transportation of palm oil and PKO from US ports to HRJ production facilities[3]	
Mode (%)	
Truck	50
Rail	50
Distance (miles)	
By truck	100
By rail	500
Notes:	
1) Wicke et al., 2007.	
2) Shipping distances from http://www.asaim.org.tw/Ocean%20Shipping%20Distances.pdf	
3) Authors' own estimates.	

7.3.5 Land Use Change Emissions from Palm Oil Production

Like soy oil, palm oil requires the use of arable land for growth and its use in fuel production may lead to land use change emissions. In addition to a scenario where no land use change emissions were incurred, three scenarios of land use change were considered using available data in the literature to illustrate the possible range of land use change emissions. The scenarios are regionally appropriate for Southeast Asia and are thus consistent with all the present analysis of palm oil production. The goal of this section is to provide an understanding for the reader of how land use change emissions compare the emissions from the other five life cycle stages. It is not intended to explicitly quantify the specific land use change emissions that would result from expanded palm oil production.

The first case assumed direct land use change emissions from the conversion of previously logged forest to palm plantations (Wicke et al., 2007). The second and third cases assumed land use change emissions resulting from the conversion of tropical rainforest and peat land rainforest in Southeast Asia, respectively, to palm plantations (Fargione et al., 2008). In all scenarios, the total land use change emissions were uniformly allocated over 30 years with no discounting, roughly the life-span of a palm plantation. In addition, the impact of varying palm FFB yield was explored in each land use change scenario. The resultant land use change emissions per ton of FFB are shown in Table 58.

Table 58: Land use change emissions in the cultivation of palm fresh fruit bunches

Scenario	Land use change Emissions (Mg CO$_2$/ha)	Assumed palm FFB yield (ton/ha)	Land use change emissions (g CO$_2$/ton FFB)[1]
Conversion of previously logged over forest	--	--	78,400[2]
Conversion of tropical rainforest	702[3]	18.2 21.2 23.1	1,285,700 1,103,800 1,013,000
Conversion of peat land rainforest	3,452[3]	18.2 21.2 23.1	6,322,300 5,427,700 4,981,200
Notes: 1) Assumed to be amortized over 30 years 2) Derived from data in Wicke et al., 2007, assuming crude palm oil energy content of 36 MJ/kg and that 1 lb of FFB produces 0.2 lbs of crude palm oil. 3) Fargione et al., 2008			

7.3.6 Results

The key assumptions and corresponding life cycle GHG emissions in the production and use of HRJ from palm oil with and without land use change emissions are outlined in Tables 59-62. Excluding land use change emissions, the life cycle GHG emissions of the production of HRJ from palm oil and palm kernel oil range from 0.2 to 0.4 times those of conventional jet fuel. However, when including land use change emissions, they can dominate GHG emissions of the entire pathway. Specifically, the land use change scenarios investigated in this work result in life cycle GHG emissions ranging from 0.4 to over 9 times those of conventional jet fuel.

Table 59: Life cycle emissions for the palm oil to HRJ assuming zero land use change emissions

Land Use Change Scenario P0	Low	Baseline	High
Key Assumptions			
Palm FFB yield (ton/ha)	23.1	21.2	18.2
Life Cycle CO$_2$ Emissions by Stage			
Biomass Credit (gCO$_2$/MJ)	-73.7	-70.5	-68.9
Recovery of feedstock (gCO$_2$/MJ)	4.8	4.9	6.6
Transportation of feedstock (gCO$_2$/MJ)	3.3	3.1	3.1
Processing of feedstock to fuel (gCO$_2$/MJ)	7.1	10.3	13.2
Transportation of jet fuel (gCO$_2$/MJ)	0.6	0.6	0.6
Combustion CO$_2$ (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	-58.0	-51.6	-45.5
WTT CH$_4$ emissions (gCO$_2$e/MJ)	5.2	6.3	7.4
WTT N$_2$O emissions (gCO$_2$e/MJ)	4.9	5.1	5.8
Total WTW GHG Emissions (gCO$_2$e/MJ)	22.5	30.1	38.1
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.26**	**0.34**	**0.44**

Table 60: Life cycle emissions for the palm oil to HRJ assuming conversion of logged over forest

Land Use Change Scenario P1	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO$_2$/MJ)	10.1	9.6	9.5
WTW CO$_2$ emissions (gCO$_2$/MJ)	22.5	28.5	34.4
Total WTW GHG Emissions (gCO$_2$e/MJ)	32.6	39.8	47.6
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.37**	**0.45**	**0.54**
Notes: 1) All other input assumptions (cultivation of palm FFB, extraction of palm oils, processing of palm oils to HRJ) are based on those in P0 emissions case of corresponding scenario.			

Table 61: Life cycle emissions for the palm oil to HRJ assuming conversion of tropical rainforest

Land Use Change Scenario P2	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO$_2$/MJ)	130.7	135.8	155.2
WTW CO$_2$ emissions (gCO$_2$/MJ)	143.1	154.6	180.1
Total WTW GHG Emissions (gCO$_2$e/MJ)	153.2	166.0	193.3
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**1.75**	**1.90**	**2.21**
Notes: **1)** All other input assumptions (cultivation of palm FFB, extraction of palm oils, processing of palm oils to HRJ) are based on those in P0 emissions case of corresponding scenario.			

Table 62: Life cycle emissions for the palm oil to HRJ assuming conversion of peatland rainforest

Land Use Change Scenario P3	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO$_2$/MJ)	642.8	667.9	763.1
WTW CO$_2$ emissions (gCO$_2$/MJ)	655.2	686.7	788.0
Total WTW GHG Emissions (gCO$_2$e/MJ)	665.3	698.0	801.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**7.60**	**7.98**	**9.16**
Notes: 1) All other input assumptions (cultivation of palm FFB, extraction of palm oils, processing of palm oils to HRJ) are based on those in P0 emissions case of corresponding scenario.			

7.4   HRJ from Rapeseed Oil

The production of HRJ from rapeseed oil was analyzed with the GREET framework using cultivation and processing data from the literature.[50] Rapeseed has been grown for the production of animal feed and vegetable oils for both human consumption and biofuel production. The leading producers of rapeseed are currently China, Canada, India and the European Union (FAO, 2010). The use of rapeseed oil as a feedstock for biofuels is of particular interest in Europe, where Rapeseed Methyl Ester is one of the two main biofuels under consideration (CONCAWE, 2002). This analysis assumes that rapeseed oil is produced in Europe and subsequently imported to the United States to be hydroprocessed into HRJ. The key parameters used to form the low, baseline and high emissions scenario were the rapeseed yield, oil content, farming energy, fertilizer application, transportation distance and oilseed drying energy requirements. In addition to a scenario where land use change emissions were assumed zero, a scenario where rapeseed

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[50] Rapeseed oil is not a preexisting pathway within GREET. As such, a new pathway was built within the GREET framework using the soy oil to renewable diesel pathway as a guide.

is grown on set-aside land[51] was examined; the aforementioned key parameters were varied within each land use change scenario.

7.4.1 Cultivation of Rapeseed

Rapeseed cultivation was assumed to take place predominantly in the United Kingdom (UK) and France based on data from Mortimer and Elsayed (2006), Edwards et al. (2007), Richards (2000) and Prieur et al. (2008). The analysis was supplemented by additional data from Sweden and Denmark from Bernesson et al. (2004) and Schmidt (2007). This was deemed appropriate due to the relative similarity in climate among southern Sweden, Denmark, France and the UK.

Rapeseed yield was estimated using data for the UK and France from 1999 through 2009. The baseline scenario adopted a projected rapeseed yield in 2015 of 3.35 Mg/ha (Eurostat, 2010) using linear regression on the historical data. The low and high emissions scenarios were developed using the same method employed to estimate soybean and palm yields. Specifically, based on historical rapeseed yield data from 1999 through 2009 (Eurostat, 2010), the variation between the lowest yield and the line of best fit was -16.8% (UK in 2001) while that between the highest yield and the line of best fit was +15.7% (France in 2009). Based on these historical data, it was assumed that yield fluctuations in some future year could be 16.8% lower than in the baseline case, corresponding to the high emissions case of 2.79 Mg/ha. Similarly, a yield in some future ideal growing year could be 15.7% higher than in the baseline case, corresponding to the low emissions case of 3.89 Mg/acre.

Yearly data from both France and the UK is shown in Figure 14. The weighted average corresponds to the ratio of total harvested weight to total planted area from both countries. Although there is substantial fluctuation in yield from year to year, the underlying trend is increasing over time at a rate of 23.7 kg/ha/year.
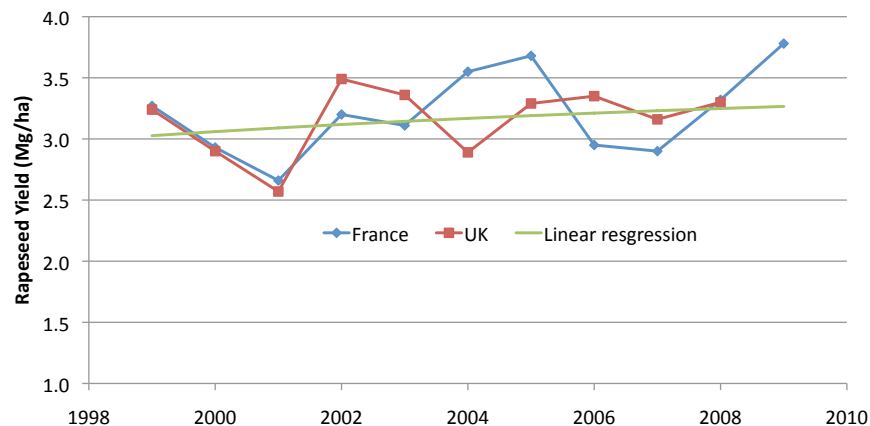
Figure 14: Yearly rapeseed yield for France and the UK between 1999 and 2009

The oil fraction of rapeseed ranges from 40% to 45% by mass based on the sources used to model cultivation. The oil yield per kilogram of rapeseed is higher than any other renewable oil feedstock considered in this work. In this analysis, oil fractions of 45% (Prieur et al., 2008; Bernesson et al., 2004), 44% (Schmidt, 2007) and 41% (Richards, 2000; Mortimer and Elsayed, 2006) were assumed in the low, baseline and high emissions scenarios.

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[51] From 1988 through 2009, the EU government compensated farmers to remove 10%-15% of their land from production to deliver some environmental benefits following considerable damage to agricultural ecosystems and wildlife as a result of the intensification of agriculture. The program has since changed such that the set aside system is on a voluntary basis with no compensation. Land that was set aside by farmers could be available for increased rapeseed cultivation (Gray, 2009).

Inputs to rapeseed cultivation are fuels for farming operations, nitrogen fertilizers, phosphate, potash and herbicides. The energy consumption of seed crop growth and processing of seeds is less than 1% of all energy used in cultivation and harvesting and was considered negligible for the purposes of this analysis (Richards, 2000). The usage per hectare of each of these resources in the low, baseline and high emissions scenario is given in Table 63. Nitrogen fertilizer application was assumed to occur in the form of 50% ammonia and nitrogen solutions and 50% ammonium nitrate and ammonium sulfate. Defualt GREET assumptions for soybeans were adopted regarding the types of herbicides applied during rapeseed cultivation. The largest variation was found in the use of diesel fuel on the farm. The low emissions scenario was based on data from Mortimer and Elsayed (2006) whose estimate for North East England was by far the most optimistic. The baseline scenario was based on French data from Prieur et al. (2008), although Bernesson (2004) gave a similar estimate in his analysis of rapeseed production in Sweden. The high emissions scenario employs data from Richards (2000) for arable lands in England, which is confirmed by Schmidt (2007) with his farming energy estimates of rapeseed production in Denmark.

The corresponding production inputs per Mg of rapeseed for each scenario were calculated by combining the production inputs and yields per hectare. The production of rapeseed oil results in straw biomass production. The average ratio of oilseed production to straw is approximately 0.96:1.[52] This analysis assumed that the straw was ploughed back into the fields after harvest. This leads to minimized depletion of soil nutrients and fertilizer savings, which were accounted for in the cultivation inputs (Prieur et al., 2008). Where there is a nearby heat or power generation facility that is outfitted to accommodate biomass feedstocks, the straw represents a potential energy source (Richards, 2000); however, straw from rapeseed is rarely harvested because of burning problems with the newer varieties and lower yields than grasses or wheat. Lower yield makes rapeseed straw more expensive to harvest so it is simply tilled back into the soil by most farmers (Bernesson et al., 2004).

Table 63: Farming energy, fertilizer and herbicide usage for the production of rapeseed in the low emissions, baseline and high emissions scenarios

	Low	Baseline	High
Rapeseed Yield (Mg/ha)	2.79	3.35	3.89
Rapeseed Oil Fraction (mass)	45%	44%	41%
Fuel Usage			
Diesel (MJ/ha)[1]	1857	2310	3934
Fertilizer Usage			
Nitrogen(kg-N/ha)[2]	140	164	180
Phosphate (kg-P_2O_5/ha)[3]	34	47	56
Potash (kg-K_2O/ha)[4]	35	43	82
Herbicides[5]	1.8	2.3	2.8
Notes:			

Notes:
1) Mortimer and Elsayed (2006) – Low Case; Prieur et al. (2008) – Baseline Case; Richards (2000) –High Case
2) Bernesson et al. (2004) and Schmidt (2007) – Low Case; Prieur et al. (2008) – Baseline Case; Edwards et al. (2007) – High Case
3) Bernesson et al. (2004) – Low Case; Prieur et al. (2008) – Baseline Case; Mortimer and Elsayed (2006) and Schmidt (2007) – High Case
4) Prieur et al. (2008) – Low Case; Richards (2000) and Bernesson et al. (2004) – Baseline Case; Mortimer and Elsayed (2006) and Schmidt (2007) – High Case
5) Richards (2000) – Low Case; Prieur et al. (2008) – Baseline Case; Mortimer and Elsayed (2006) – High Case

[52] The straw and oilseed production from Richards (2000) were 4 Mg/ha and 4.08 Mg/ha, respectively (ratio of 0.98:1). The straw and oilseed production from Schmidt (2007) were 2.93 Mg/ha and 3.13 Mg/ha, respectively (ratio of 0.94:1).

Emissions from N_2O were estimated using IPCC Tier 1 methodology (De Klein et al., 2006). Nitrogen in above and below ground crop residues was estimated by applying the aforementioned crop residue production ratio to estimate the amount of straw tilled back into the soil. Rapeseed straw has been characterized as 0.75% nitrogen by mass (Karaosmanoglu et al., 1999) leading to 7125 g of nitrogen reapplied to the field in the form of straw biomass per megagram of oilseed production. The IPCC Tier 1 methodology estimates the combined direct and indirect conversion rate for nitrogen from synthetic fertilizers as 1.325% and nitrogen from crop residues as 1.225%. These rates include the atmospheric deposition of nitrogen volatilized from managed soils as well as nitrogen from leaching and runoff. The formula for calculating N_2O emissions from rapeseed cultivation is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{Mg} \right) = \left(\left(\frac{g_{nitrogen\ fertilizer}}{Mg} \right) \cdot 0.01325 + \left(7125 \frac{g_{N\text{-}crop\ residue}}{Mg} \right) \cdot 0.01225 \right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right) \qquad \text{Equation 12}$$

The least defined aspects of rapeseed production are the drying and storage practices. In practice, there is little consensus on the oilseed moisture content at harvest, which has implications for energy consumption during drying. A recent survey of current harvesting, drying and storage practices of oilseed rape in the UK found that most farmers harvest above 12% moisture content, which is the threshold for Ochratoxin B production (Armitage et al., 2005).[53] After harvest the rapeseed must be dried to a moisture content of 9% for storage (Prieur et al., 2008; Richards, 2000; Schmidt, 2007). Most rapeseed is stored for about 3 months before being sold. This tendency is driven mainly by market strategy and cash flow. Longer storage periods may require lower moisture contents to minimize mite infestation and deterioration through rancidity. Mites are the greatest problem faced by rapeseed famers and were observed on more than 25% of sites. From the perspective of seed crushers, moisture content (high or low) and admixture of stores were the most common reason for rejection or price reductions (Armitage et al., 2005). This analysis assumed moisture contents of 13%, 14% and 15% at harvest and 9% after drying in the low emissions, baseline and high emissions scenario, respectively. Losses due to mites, rancidity and admixture were neglected due to lack of quantitative data but these could become important for ill-maintained rapeseed stored over long periods of time.

Table 64: Rapeseed drying and storage assumptions in the low emissions, baseline and high emissions scenario

	Low	Baseline	High
Moisture Content			
Harvest[1]	13%	14%	15%
Storage[2]	9%	9%	9%
Drying Energy Consumption[3]			
Diesel (MJ/L$_{evaporated}$)	n/a	4.7652	6
Grid Electricity (MJ/L$_{evaporated}$)	n/a	0	3.6
Diesel (MJ/Mg)	165.0	277.0	423.5
Grid Electricity (MJ/Mg)	0.0	0.0	254.1
Notes:			
1) Mortimer and Elsayed (2006) – Low Case; Average of low and high cases – Baseline case; Bernesson et al. (2004) – High Case			
2) Prieur et al. (2008), Richards (2000), Mortimer and Elsayed (2006), Schmidt (2007)			
3) Mortimer and Elsayed (2006) – Low Case; Bernesson et al. (2004) – Baseline case; Schmidt (2007) – High Case			

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[53] Ochratoxin B is the most abundant food contaminating mycotoxin in the world. Human exposure occurs primarily through consumption of improperly stored food products (Armitage et al., 2005)

Approximately one third of farmers surveyed in the UK use ambient air drying to reduce the moisture content of their seed from the value at harvest to that required for storage. Drying using ambient air can take from 2 to 4 weeks, although 2 weeks is the most common. The other two thirds of farmers use hot air dryers to reduce drying times at the expense of increased energy consumption (Armitage et al., 2005). This work assumed the use of hot air drying according to energy consumption estimates from Mortimer and Elsayed (2006), Bernesson et al. (2004) and Schmidt (2007) in the low, baseline and high emissions scenarios, respectively. The assumptions regarding the drying and storage of rapeseed used in this work are summarized in Table 64.

7.4.2 Extraction of Oil from Rapeseed

A modified version of the process for oil extraction from soybeans established by Sheehan et al. (1998a), including only the processes relevant to rapeseed in an N-hexane extraction facility, was used to model the process inputs to extracting oil from rapeseed. Ozata et al. (2009) also used this approach in their analysis of biodiesel from rapeseed. The changes to the data from Sheehan et al. (1998a) were limited to removing the energy demands for drying as this has been explicitly quantified for rapeseed in Table 64. The process energies were converted from energy per unit mass of oilseed to energy per unit mass of oil using the oil fractions from the low emissions, baseline and high emissions scenarios. The outputs and energy consumption assumed in the extraction of oil from rapeseed are shown in Table 65.

Table 65: Process inputs for extracting oil from rapeseeds

	Low[1]	Baseline[1]	High[1]
Receiving and Storage			
Electricity	18.60	19.02	20.41
Rapeseed Preparation			
Electricity	68.76	70.32	75.46
Steam	153.35	156.83	168.31
Oil Extraction			
Electricity	11.46	11.73	12.58
N-hexane	94.15	96.29	103.34
Meal Processing			
Electricity	63.56	65.01	69.77
Steam	492.54	503.74	540.59
Oil Recovery			
Electricity	1.21	1.24	1.33
Steam	77.04	78.79	84.55
Solvent Recovery			
Electricity	1.66	1.69	1.82
Steam	0.00	0.00	0.00
Oil Degumming			
Electricity	5.38	5.50	5.91
Steam	60.53	61.91	66.44
Waste Treatment			
Electricity	1.82	1.86	1.99
Steam	32.56	33.30	35.74
Totals			
Electricity	172.45	176.37	189.27
Natural Gas[2]	1020.03	1043.21	1119.54
N-hexane[3]	94.15	96.29	103.34
Notes:			
1) All values are in Btu per pound of oil			
2) Steam is generated from natural gas with an efficiency of 80%.			
3) GREET uses Liquefied Petroleum Gas (LPG) as a surrogate for N-hexane when calculating emissions			

7.4.3   Co-Product Usage and Allocation Methodology

In the extraction of oil from rapeseed, rapeseed meal is produced in large quantities as a co-product (1.22-1.44 kg of meal per kg of oil). This is similar to the extraction of oil from soybeans; hence, the same challenges with respect to allocation arise. Rapeseed meal is primarily used as an animal feed and could potentially displace barley, corn, and soybean meal. One kg of rapeseed meal is equivalent to 0.87 kg of soybean meal on a protein basis (Prieur et al., 2008).

Based on the results from section 7.2.3, the displacement method, as applied here, is not appropriate because of the large variation in life cycle GHG emissions that will result from the choice of feedstocks (i.e., barley, corn, and soybean meal) displaced by rapeseed meal. The mass or energy allocation method may not be most appropriate as rapeseed meal is not valued based on its mass, or sold as a commercial energy product. The commercial value of the protein in rapeseed meal for animal feed resulted in market value allocation being adopted in this work. This is internally consistent with the other pathways considered in this work as well as with Prieur et al. (2008), Mortimer and Elsayed (2006) and Ozata et al. (2009).

7.4.4   Transportation of Rapeseed Oil to HRJ Production Facilities

Rapeseed cultivation, harvesting and oil extraction were assumed to occur primarily in France and the UK. The one-way distance from the farm to the oil extraction facility was assumed to be 115 km (Prieur et al., 2008).[54] It was then assumed that the United States imports this oil to a domestic hydroprocessing facility where it is converted to jet fuel. This transportation profile is not available in GREET and was created using the data from Table 66. Default GREET assumptions were used for the details of each transportation mode.

Table 66: Transportation profile of Rapeseed Oil from Europe to the United States

Transportation of Rapeseed Oil to European Shipping Ports[1]	
Mode (%)	
Truck	100
Distance (kilometers)	150
Transportation of Rapeseed Oil from European Ports to US ports[2]	
Mode (%)	
Ocean tanker	100
Distance (kilometers)	
Western UK to Eastern US (50%)	5520
Western France to Eastern US (25%)	5780
Southern France to Western US (25%)	7170
Average	6000
Transportation of Rapeseed Oil from US ports to HRJ production facilities[1]	
Mode (%)	
Truck	50
Rail	50
Distance (miles)	
By truck	160
By rail	800
Notes:	
1) Author's own estimates	
2) Shipping distances from http://www.searates.com/reference/portdistance/	

7.4.5   Land Use Change Emissions from Rapeseed Oil Production

Two scenarios of land usage were considered for the cultivation of rapeseed in France and the UK. Rapeseed is an established crop in these areas where over 2 million hectares were

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[54] Richards (2000) and Mortimer and Elsayed (2006) estimated one-way transport distances of 90 km and 130 km, respectively.

harvested in 2008 (Eurostat, 2010). The first scenario represents established rapeseed production with an assumption that land use change emissions were zero. A second scenario resulting in positive GHG emissions was envisioned where rapeseed production is expanded for biofuel production on set aside land thus resulting in land use change emissions. Set aside land is land that was removed from agricultural production as a result of government mandates to re-establish some environmental benefits to agricultural ecosystems and wildlife. This is similar to the Conservation Reserve Program (CRP) in the United States. While removed from production, the natural carbon and nitrogen stocks of the land are replenished. Participation by farmers in the set aside system has recently changed to voluntary and these lands could be available for rapeseed cultivation (Gray, 2009). A return to crop production would cause the accumulated carbon and nitrogen in the soils to be depleted over time. The goal of this section is to provide an understanding for the reader of how land use change emissions compare to emissions from the other five life cycle stages. It is not intended to explicitly quantify the specific land use change emissions that would result from expanded palm oil production.

The estimate of GHG emissions resulting from the conversion of set aside land to rapeseed cultivation was based on Schmidt (2007). The total land use change emissions were amortized over 30 years with no discounting. The corresponding land use change emissions per Mg of rapeseed were calculated by combining the GHG estimate with the yield per hectare; this is presented in Table 67.

Table 67: Land use change emissions from rapeseed cultivation on set aside lands in Europe

Scenario	Land use change Emissions (Mg CO_2e/ha)	Assumed Rapeseed yield (Mg/ha)	Land use change emissions (g CO_2e/Mg)[1]
Conversion of set-aside land[2]	94.6	2.79 3.35 3.89	1,129,700 940,800 810,200
Notes: 1) Assumed to be amortized over 30 years with no discounting 2) Estimate from Schmidt (2007)			

7.4.6 Results

The life cycle GHG emissions from the production and use of HRJ fuel from rapeseed oil are given in Tables 68 and 69. When no land use change emissions are present, the life cycle GHG emissions range from 0.45 to 0.87 times those of conventional jet fuel; however, nitrous oxide emissions represent between 39% and 44% of the total. Edwards et al. (2007) found nitrous oxide emissions from rapeseed production to be of similar magnitude using an independent, well-validated soil chemistry model (DNDC, version 82N). When rapeseed is grown on set aside land, the life cycle GHG emissions range from 0.87 to 1.47 times those of conventional jet fuel. The variation in the biomass credit is due to minor changes in the allocation scheme through the pathway. The transportation of oil across the Atlantic is responsible for only 0.6 g CO_2e/MJ; these emissions would not have been incurred had the fuel been processed and used within Europe.

Nitrous oxide emissions represent more than approximately 40% of the total life cycle GHG emissions from the rapeseed to HRJ pathway. As such, the consequences of the uncertainty associated with IPCC correlations are more important for this pathway. Although the magnitude of N_2O emissions in this work compare favorably with the detailed model used by Edwards et al. (2007), the reader should be aware of these inherent uncertainties when comparing different pathways for GHG reduction potential in sections 8.1 and 8.2.

Table 68: Summary of results from renewable jet fuel production and use from rapeseed

Land Use Change Scenario R0	Low	Baseline	High
Key Assumptions			
Total Biomass Yield (Mg/ha/yr)	2.79	3.35	3.89
Seed Oil Fraction	45%	44%	41%
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO_2/MJ)	-73.7	-70.5	-68.9
Recovery of feedstock (gCO_2/MJ)	13.6	17.2	26.4
Transportation of feedstock (gCO_2/MJ)	3.2	3.1	3.1
Processing of feedstock to fuel (gCO_2/MJ)	7.1	10.3	13.2
Transportation of jet fuel (gCO_2/MJ)	0.6	0.6	0.6
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	-49.2	-39.2	-25.7
WTT CH_4 emissions (gCO_2e/MJ)	1.0	1.3	1.7
WTT N_2O emissions (gCO_2e/MJ)	17.6	22.4	29.5
Total WTW GHG Emissions (gCO_2e/MJ)	39.8	54.9	75.9
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	0.45	0.63	0.87

Table 69: Life cycle GHG emissions for production and use of renewable jet fuel from rapeseed assuming cultivation on set-aside land

Land Use Change Scenario R1	Low[1]	Baseline[1]	High[1]
LUC CO_2 emissions (gCO_2/MJ)	35.8	40.0	48.9
LUC N_2O emissions (gCO_2e/MJ)	2.7	3.0	3.6
Land use change emissions (gCO_2e/MJ)	38.4	43.0	52.6
Total WTW GHG Emissions (gCO_2e/MJ)	78.2	97.9	128.5
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	0.89	1.12	1.47
Notes:			
1) All other input assumptions (rapeseed cultivation, extraction of oil, processing of oil to HRJ) are based on those in the R0 emissions case of the corresponding scenario.			

7.5 HRJ from Jatropha Oil

The creation of HRJ from jatropha oil is not a pathway available in GREET; hence, supporting information was obtained from the literature and a pathway was constructed within the GREET framework. The jatropha fruit is composed of an outer capsule containing two or three seeds. Each seed has a shell and a kernel, which contains oil. This structure differs from other oil seed crops discussed thus far because additional co-products beyond meal result from the oil extraction process.

There are also concerns with jatropha cultivation for biofuels that do not apply to soy, palm, or rapeseed. An overarching concern of jatropha cultivation for fuel production is that the biomass co-products are toxic to both humans and animals. Further, there are questions regarding the introduction of a non-native invasive species to the North American ecosystem. The toxicology of jatropha oil is discussed later in this section while more details on the impacts of invasive species within the North American ecosystem are provided in Section 8.4.

7.5.1 Yield and Plant Characterization

Jatropha plants are well adapted to semi-arid conditions, although more humid environments are shown to result in higher crop yields. The plant can tolerate high temperatures but it does not tolerate frost, which causes immediate damage. Upon removal of the outer capsule, the primary

products are seeds and the co-products are husks; at this stage, the seeds can be processed directly or the shells can be removed through decortication.[55] If the seeds are left intact, oil is extracted leaving de-oiled cake as the co-product. If the shells are removed before oil extraction, oil and meal are created from the kernel and the shells are considered a separate co-product. The parts of the jatropha fruit are characterized in Table 70.

Upon examination of 28 growth sites worldwide, a correlation of 0.22 was found between the quantity of precipitation and seed yield (Achten et al., 2008). This means that although more precipitation is moderately connected to higher seed yields, there are many cases where excellent yields have been realized in dry conditions and poor yields realized in wet conditions. The majority of data used in this work came from cultivation details and physical characterization of jatropha by Reinhardt et al. (2008) and Achten et al (2008).

Based on these data, it was concluded that an average yield of 2500kg/ha/yr of dry seeds is a representative estimate while 5000kg/ha/yr could be realized under optimal management practices; 1000kg/ha/yr appears to be a reasonable lower bound (Achten et al., 2008; Reinhardt et al., 2008). These values provide the bounds on yield for the low and high emissions cases. Note that yields are quoted in terms of seed weight. To fully characterize the fruit, mass ratios of oil to seed, husk to seed and kernel to seed were developed for the low emissions, baseline and high emissions cases. These are shown Table 71 in conjunction with yield assumptions.

Table 70: Parts of the jatropha fruit including processed states

Product	Energy Content (MJ/kg)	Sub-fractions	Description
Capsule	--	Husk + seeds	Entire fruit
Husk	15.5	--	Outer core of fruit, surrounding seeds; green and 'fleshy' in fresh state, later brown and dry
Seed	--	Shell + kernel	Compact unit inside fruit, consists of shell and kernel, usually 2-3 per capsule
Shell	19.0	--	Brown or black shell surrounding the kernel
Kernel	--	--	White compact nucleus of seed, actual oil-containing part of the fruit
Oily cake	19.5	Seeds (processed)	Leftovers from the mechanical oil extraction from seeds (incl. shells), contains residual oils
De-oiled cake	17.5	Seeds (processed)	Leftovers from the mechanical and solvent-aided oil extraction from seeds
Meal	18.0	Kernels (processed)	Leftovers from the mechanical and solvent-aided oil extraction from seeds
Notes: 1) Reinhardt et al (2007 and 2008)			

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[55] Decortication is a procedure involving the removal of a surface layer, membrane or fibrous cover. In the case of jatropha, this refers to the removal of the shells from the kernels.

Table 71: Yields and mass fractions characterizing the jatropha fruit

	Low	Baseline	High
Jatropha Yield (kg$_{seed}$/ha/yr)	5000	2500	1000
Oil Ratio (kg$_{oil}$/kg$_{seed}$)	0.37	0.35	0.34
Husk Ratio (kg$_{husk}$/kg$_{seed}$)	0.48	0.60	0.60
Kernel Ratio (kg$_{kernel}$/kg$_{seed}$)	0.67	0.63	0.63

7.5.2 Cultivation of Jatropha Fruit

The input assumptions surrounding jatropha cultivation are based on a 30-hectare test plot in India documented by Reinhardt et al. (2008). The inputs required for growth are seedlings, irrigation water (first three years only), diesel fuel (for tractor and irrigation pump) and mineral fertilizers in the form of nitrogen, phosphorous pentoxide (P_2O_5) and potassium oxide (K_2O). Since irrigation water is needed for only the first three years of growth, this analysis does not include the diesel fuel required for pumping irrigation water in calculating life cycle GHG emissions of HRJ from jatropha. The cultivation assumptions adopted in this work are listed in Table 72.

Table 72: Cultivation inputs for the growth of jatropha

	Low	Baseline	High
Cultivation Diesel (Btu/kg$_{seed}$)	1163	1320	1419
Pesticides (g/kg$_{seed}$)	0	0	0
Nitrogen (g/kg$_{seed}$)	31.8	34.0	35.3
P_2O_5 (g/kg$_{seed}$)	12.6	13.0	13.4
K_2O (g/kg$_{seed}$)	31.3	37.4	37.4

Nitrous oxide emissions from jatropha cultivation were estimated using the IPCC Tier 1 methodology (De Klein et al., 2006). Due to the lack of data, N_2O emissions from nitrogen in above and below ground crop residues were not accounted for in this analysis. The IPCC Tier 1 methodology estimates the combined direct and indirect conversion rate of nitrogen from synthetic fertilizers to N_2O emissions as 1.1%. These rates include the atmospheric deposition of nitrogen volatilized from managed soils; however, nitrogen from leaching and runoff was assumed to be negligible, in terms of N_2O conversion, as jatropha is ideally grown in well-drained or gravelly soils[56] (Achten et al., 2008). The formula for calculating N_2O emissions from jatropha cultivation is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{kg_{seed}} \right) = \left(\left(\frac{g_{nitrogen\ fertilizer}}{kg_{seed}} \right) \cdot 0.011 \right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right) \qquad \text{Equation 13}$$

7.5.3 Toxicity of Jatropha Fruit and Oil

Due to the toxicity of jatropha fruit and oils, attention is warranted to the impacts on human health and work environment. The fruits contain irritants affecting pickers and manual dehuskers. In addition, accidental consumption of the seeds or oils can lead to severe digestion problems. For these reasons, intercropping edible crops with jatropha is only recommended during the period before any fruit is borne (Achten et al., 2008). Gandhi et al. (1995) evaluated the crude oil in vivo and in vitro for toxicity, skin irritation and haemolytic activity. The toxic fraction containing phorbol esters was subsequently isolated from the oil. Locals use the oil as a cathartic purgative and for treating skin ailments. The seeds are also used for the treatment of dropsy, gout, paralysis and rheumatism. Upon administering tests on mice, rats and rabbits, jatropha oil was found to be

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[56] Nitrous oxide emissions from leaching and run off only apply to soils where the soil water-holding capacity is exceeded (De Klein et al., 2006)

acutely toxic after oral administration, leading to diarrhea, bloodshot eyes and inflammation of the gastro-intestinal tract. Topical application not only had an irritant effect but also caused diarrhea and mortality in the animals. Gandhi et al. (1995) summarize the impacts of ingestion and contact with jatropha oil as a severe health hazard to livestock and humans. The phorbol esters in the oil are irritants of skin and mucous membranes produce haemolysis of red blood cells and are reported to be tumor promoters (Gandhi et al., 1995). These results lead to the conclusion that complete removal of toxins is essential before any industrial applications are considered for jatropha oil. The removal of toxins was not modeled as a source of GHG emissions within this analysis.

7.5.4   Transportation of Jatropha Fruit to Production Facilities

Even though there are questions regarding the wisdom of introducing potentially invasive species such as jatropha to non-native environments, the production of jatropha in this analysis is assumed to take place in the southwestern United States. If the oil were produced overseas, then there would be an increase in the GHG emissions from transportation, comparable to that discussed in regards to rapeseed oil (see Section 7.4.3). After harvesting, assumptions of the transportation mechanisms and distances are consistent with GREET defaults for the soybeans to HRJ pathway. Specifically, the transportation of capsules is by truck to a local storage area and subsequently to a local oil extraction facility. The capsule processing, oil extraction and hydroprocessing of oil to HRJ are assumed to occur at the same location with on site capabilities for power generation using biomass co-products. As was the case in other HRJ pathways, transportation elements of the life cycle GHG emissions are sufficiently small to be within the margin of error; therefore, assumptions in this area carry little consequence.

7.5.5   Extraction of Oil from Jatropha Fruit

Extracting oil from the jatropha fruit has a large influence on the life cycle emissions. Before beginning oil extraction, a dehusker must remove the husks and obtain the seeds. In small-scale production facilities, the seeds are then crushed in a screw press to extract the oil. This method can only obtain up to 80% of the oil so larger production facilities mill the seeds into small particles and N-hexane chemical solvent is used to obtain up to 99% of the oils (Achten et al., 2008). The deterministic factor of this life cycle is how the co-products are used (husks, shells, meal). It is not necessary to remove the shells from the kernels before the solvent treatment, but it is more energy efficient to do so because the additional burden of processing the added material from the shells through the chemical solvent is more than the burden of removing the shells beforehand. If the shells are removed prior to milling, they are obtained independently of the meal and the oil. If the shells were not removed, the seeds could be split into a de-oiled cake and the oil itself. Regardless of whether the shells are removed, the resultant product is not suitable for animal and human consumption because of its toxicity (see Section 7.5.3).

The co-products from oil extraction can be used in a multitude of capacities. Figure 15 shows the products resulting from this process and potential uses for each.
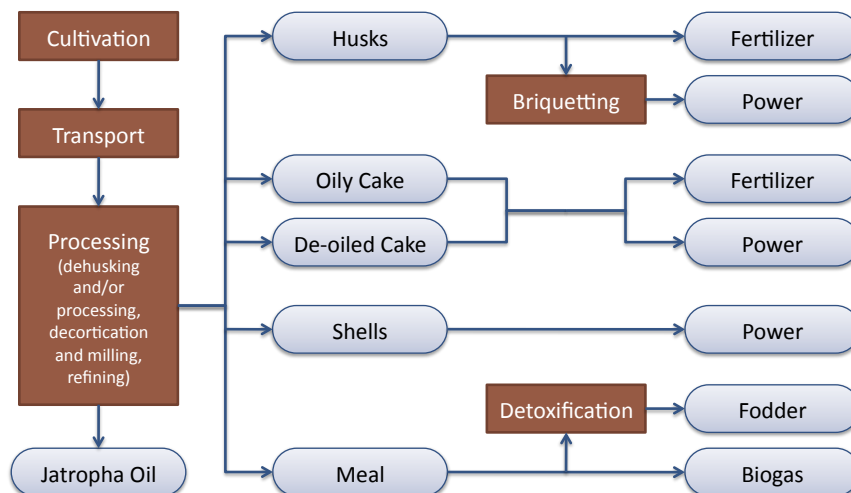
Figure 15: Utilization of by-products from jatropha cultivation and oil extraction (adapted from Reinhardt, 2007)

When using the co-products from jatropha, caution must be maintained due to its toxicity. For example, the use of shells and meal as fertilizer in edible crop production raises bio-safety questions and warnings have been issued to the serious lack of information surrounding the health effects of burning the oil in closed quarters (Achten et al., 2008).

To estimate the emissions from oil extraction, this work combined the process inputs from Reinhardt et al. (2008) with modified processes for oil extraction from soybeans established by Sheehan et al. (1998a) to model oil extraction from jatropha fruit in an N-hexane extraction facility. The assumed process energies are summarized Table 73.

7.5.6   Co-Product Usage and Allocation Methodology

To understand the influence of co-product type and usage on the life cycle emissions of the fuel, four scenarios, outlined in Table 74, were considered. These scenarios examined how the life cycle GHG emissions of HRJ from jatropha oil change depending on the use of co-products for animal feed, fertilizer or electricity production. The electricity production was further broken down to examine displacement of US average grid electricity and energy allocation between oil and electricity. The life cycle GHG emissions resulting from each scenario are summarized in Figure 16.

As explained below, Scenario 1 was chosen for this analysis. Using the products for power generation seems a more logical choice for the biomass co-product because the displacement of fertilizer on large scales with a product that is toxic to humans was deemed undesirable due to the potential for ground water contamination (this eliminates Scenario 2). Energy allocation was chosen over displacement due to the large quantities of co-product generated per unit of oil production (this eliminates Scenarios 3 and 4). As shown in Figure 16, the displacement of fossil based electricity results in the unrealistic result of negative life cycle GHG emissions for the fuel. It is important to understand that the allocation method used in scenario 4 attributes energy and emissions based on the electricity produced from the biomass co-products and the energy contained in the oil. This is equivalent to expanding the system boundary of the oil extraction stage to include the power generation unit. Assumptions regarding the generation of renewable electricity from biomass are outlined in Table 75.

Table 73: Process inputs for extracting oil from jatropha fruit

	Low[1,3]	Baseline[1]	High[1]
Receiving and Storage			
Electricity	33.5	38.3	39.4
Jatropha Preparation			
Electricity (briquetting)	93.2	122.8	126.6
Electricity (decorticator)	23.2	24.6	25.3
Electricity (dehusking)	51.6	59.0	60.7
Electricity (cleaning)	3.0	2.9	3.0
Steam (cleaning)	125.9	123.2	127.0
Oil Extraction			
Electricity	9.4	9.2	9.5
N-hexane[4]	77.3	75.7	77.9
Meal Processing			
Electricity	38.1	37.3	38.4
Steam	368.7	361.0	371.9
Oil Recovery			
Electricity	1.0	1.0	1.0
Steam	63.2	61.9	63.8
Solvent Recovery			
Electricity	1.4	1.3	1.4
Steam	0.0	0.0	0.0
Oil Degumming			
Electricity	4.4	4.3	4.5
Steam	49.7	48.6	50.1
Waste Treatment			
Electricity	1.5	1.5	1.5
Steam	26.7	26.2	27.0
Totals			
Electricity	260.3	302.1	311.2
Natural Gas[2]	792.8	776.3	799.6
N-hexane[4]	77.3	75.7	77.9

Notes:
1) All values are in Btu per pound of oil
2) Steam is generated from natural gas with an efficiency of 80%.
3) The low emissions case appears to have higher process energy than the baseline; however, this arises due a larger fraction of the seed weight going through the extraction processes (see Table 71) and is counteracted in the fuel production stage.
4) GREET uses Liquefied Petroleum Gas (LPG) as a surrogate for N-hexane when calculating emissions

Table 74: Co-product creation and allocation scenarios from the oil extraction process for jatropha capsules. The corresponding life cycle emissions for each scenario are shown in Figure 16

Scenario	Co-Product	Use	Emissions Allocation Method
1	Husks, shells and meal	Burned for electricity	Energy allocation via energy of electricity produced and energy contained in the oil
2	Husks, shells and meal	Fertilizer	Displacement method (system expansion) where fertilizers are displaced in subsequent cultivation practices
3	Husks and shells	Burned for electricity	Displacement method (system expansion) where electricity from US grid is displaced
3	Meal	Detoxified and sold for animal feed	Market value allocation between oil and meal.
4	Husks, shells and meal	Burned for electricity	Displacement method (system expansion) where electricity from US grid is displaced
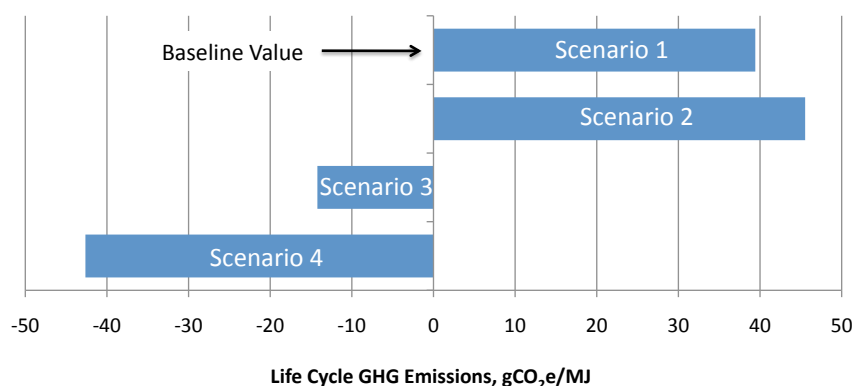
Figure 16: Sensitivity of life cycle emissions of HRJ from jatropha to co-product utilization and allocation scheme. Scenarios are described in Table 74

Table 75: Assumptions used in establishing average US biomass conversion efficiency to electricity

Generation Method	Production Share	Generation Efficiency
Biomass Utility Boiler	99%	32.1%
Biomass IGCC	1%	43.0%
Note: 1) Transmission losses of 8% were also included in the overall efficiency of generation		

7.5.7   Results

The life cycle emissions from the production and use of HRJ fuel from jatropha oil are given in Table 76. The life cycle GHG emissions resulting from the production and use of jatropha HRJ range from 0.36 to 0.52 times those from conventional jet fuel. The variation in the biomass credit is due to minor changes in the allocation scheme through the pathway. Because marginal land

was assumed and no estimates of root carbon sequestration from jatropha were available, the GHG emissions from land use change were assumed to be zero and no other land use change scenarios were created.

Table 76: Life cycle emissions from the jatropha oil to HRJ pathway

	Low	Baseline	High
Key Assumptions			
Jatropha Seed Yield (kg/ha/yr)	1000	2500	5000
Jatropha Seed Oil Fraction	0.34	0.35	0.37
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO$_2$/MJ)	-73.7	-70.5	-68.9
Recovery of feedstock (gCO$_2$/MJ)	16.1	16.7	17.6
Transportation of feedstock (gCO$_2$/MJ)	1.5	1.5	1.5
Processing of feedstock to fuel (gCO$_2$/MJ)	7.1	10.3	13.2
Transportation of jet fuel (gCO$_2$/MJ)	0.6	0.6	0.6
Combustion CO_2 (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO$_2$/MJ)	-48.5	-41.3	-36.1
WTT CH_4 emissions (gCO$_2$e/MJ)	1.0	1.2	1.4
WTT N_2O emissions (gCO$_2$e/MJ)	8.9	9.1	9.4
Total WTW GHG Emissions (gCO$_2$e/MJ)	31.8	39.4	45.1
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	0.36	0.45	0.52

Nitrous oxide emissions represent more than 20% of the total life cycle GHG emissions from the jatropha to HRJ pathway. As such, the consequences of the uncertainty associated with IPCC correlations are more pronounced for this pathway. The reader should be aware of these inherent uncertainties when comparing different pathways for GHG reduction potential in sections 8.1 and 8.2.

7.6   HRJ from Algae Oil

The creation of HRJ from algae oil is not a pathway available in GREET; hence, supporting information was obtained from the literature.[57] Algae can be grown in an open pond, a controlled bioreactor, or a combination of both. A typical open pond approach consists of a raceway (oval) shaped pond using a paddlewheel to circulate the water and mix the algae for even light exposure and growth. In bioreactors, the algae are grown in sheets or tubes, allowing for much higher growth rates per unit area than open ponds. Bioreactors shield the algae from weather variations and facilitate growth in vertical geometries thereby reducing land requirements; however, these designs are cost intensive relative to open ponds. Only open pond technologies were examined in this analysis because of their reduced capital costs, the relative abundance of experimental documentation, and their increased technological readiness (relative to bioreactors). Given time, the capital costs of bioreactors could decrease as technological advances are made; these concepts will be examined further as a part of this continuing research effort. Furthermore, only conventional means of oil extraction using chemical solvents were considered in this report. As was discussed in the introduction of Chapter 7, there is currently much research devoted to other methods of oil extraction including genetic modification of the algae such that they naturally excrete oil and using electric shocks to open the algal cells. Such methods were not considered here.

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[57] Algae oil is not a preexisting pathway within GREET. As such, a new pathway was built within the GREET framework using the soy oil to renewable diesel pathway as a guide.

7.6.1 Algal Yield and Lipid Content

The two defining characteristics for algae as a biofuel are its growth rate (generally given in grams/m^2/day) and lipid content (generally given as a weight percent of total biomass). Both quantities vary greatly within the literature as they depend on variables including algae type and weather conditions, among many others. Generally higher algal growth rates reported in the literature represent bioreactor technology and not open ponds. Some recent presentations have employed yields equivalent to 140g/m^2/day at 25% lipids content; an estimate over 550% of the yields adopted in this study (Daggett, 2007). The current analysis is based on the engineering judgment of the authors gained from their literature review of open ponds. During peak periods of growth, 50g/m^2/day could be achieved, but a yearly average of 20 g/m^2/day appears to be more reasonable (Ben-Amotz, 2008; Sheehan et al., 1998b). A survey of algal strains also returned a range of lipid contents up to 40% (Becker, 2007). Assuming that technology will improve by the simulation year of 2015, 50 g/m^2/day and 40% lipids by weight was adopted for the low emissions case, 25 g/m^2/day and 25% lipids by weight was adopted for the baseline case and 20 g/m^2/day at 15% lipids by weight was adopted for the high emissions case.

7.6.2 System Expansion (Displacement) for Electricity Emissions

While much of the methodology for the analysis of algal HRJ is similar to that discussed for other HRJ pathways, the life cycle is complicated by the need to feed CO_2 to the algae to sustain acceptable growth rates. As schematically shown in Figure 17, system expansion (displacement method) was used instead of energy allocation when apportioning emissions between the fuel and any electricity generated in providing the CO_2 required for growth. Justification for this choice is rooted in the argument that electricity used for CO_2 production operates independently of the algae cultivation and hence, should be treated differently than co-products physically created from the cultivation itself. The system boundary was expanded to include both the electricity and emissions from a power plant producing equivalent electricity to that within the original system boundary. The expanded system had both HRJ and biomass co-products leaving the system boundary, but zero net electricity exiting the expanded system. Thus, the expanded system could be treated in a similar fashion as the other HRJ pathways.

The life cycle GHG emissions from the type of electricity used to supply CO_2 for algal growth and the type of electricity being displaced both impact the life cycle emissions of the algal fuel. This is due to the variation that could exist in the emissions per kilowatt-hour of different power generation technologies (e.g. a coal utility boiler has higher emissions than an integrated gasification combined cycle which, in turn, has higher emissions than a nuclear power plant). The options in Table 77 outline the effects on the life cycle emissions that these choices could cause. When CO_2 created from 'dirty' electricity is used to feed algal growth, but 'clean' electricity is included in the expanded system then the CO_2 emissions credit is less than the CO_2 used to grow the algae; this results in a fuel that appears to be 'dirty' in terms of higher CO_2 emissions. When CO_2 created from 'clean' electricity is used to feed the algae but 'dirty' electricity is included in the expanded system then the CO_2 emissions credit is greater than the CO_2 used to grow the algae; this results in a fuel that appears to be 'clean' in terms of lower CO_2 emissions. In this analysis, the electricity used to provide the CO_2 was assumed to be the same as that in the expanded system. For this configuration, the CO_2 emissions credit was approximately equal to the biofuel combustion emissions, which is the assumption used for biofuels that have not been 'fed' CO_2 for enhanced growth.
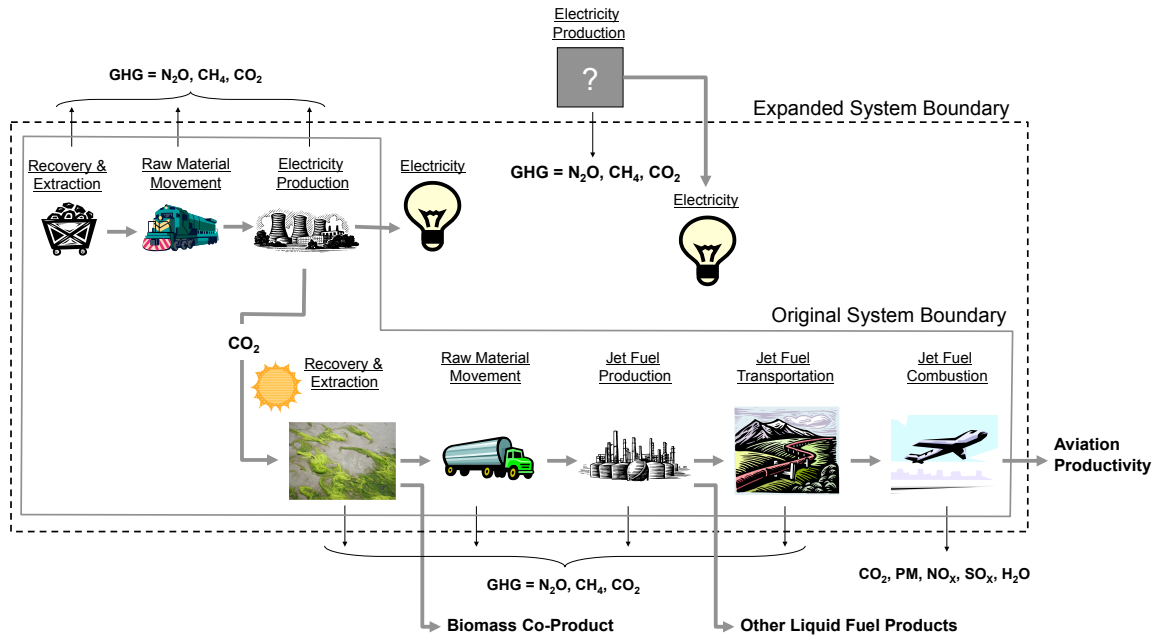
Figure 17: System boundary expansion of the algal HRJ fuel pathway

Table 77: Impact of CO_2 source and electricity choice on the biomass credit given to algal HRJ

CO_2 Source for Algae Growth	Electricity displaced in expanded system boundary	CO_2 "Biomass Credit" allocated to the algal fuel
Conventional Coal Electricity	Conventional Coal	Biomass Credit ≈ Combustion CO_2
	US Average Grid	Biomass Credit < Combustion CO_2
	Nuclear (zero CO_2)	Biomass Credit ≈ 0
US Average Grid Electricity	Conventional Coal	Biomass Credit > Combustion CO_2
	US Average Grid	Biomass Credit ≈ Combustion CO_2
	Nuclear (zero CO_2)	Biomass Credit ≈ 0

Carbon dioxide usage in algae cultivation can vary depending on the lipid content and meal carbon content of the algae. A simpler approach was adopted in this analysis where data from the literature were used to conclude that algae as a whole is approximately 60% carbon (Weissman and Goebel, 1987; Kadam, 2001). The supporting literature indicates that CO_2 requirements range from 2.16-2.2 kg per kg of algae grown. Emissions factors for power plant technologies from GREET are listed in Table 78 (these are not the full life cycle GHG emissions from power generation, rather they are just the CO_2 available to be captured and used from the power plant itself). Depending on the power plant, between 0.23 and 0.5 kg of algae could ideally be grown from the CO_2 resulting from each kWh of electricity. This includes electricity generated from biomass.

Table 78: US power generation data as it relates to algae cultivation

Generation Technology	Fraction of US Mix (%)	Power Plant Efficiency (%)	Power Plant Emissions (g_{CO2}/kWh)	Plant Capacity for Algae Cultivation (kg_{algae}/kWh)
Oil-Fired	2.6	34.8	834	0.38
NG-Fired	22.5	43.5	495	0.23
Coal-Fired	48.7	36.1	1027	0.47
Biomass Fired	1.3	32.2	1084	0.50
Nuclear	17.6	--	0	0
Other (hydro, wind, etc.)	7.3	--	0	0
Notes: 1) All data from GREET (2008) 2) Based on 2.18 kg CO_2 being required per kg of algal growth				

7.6.3 Algal Carbonation Options and Technologies

Algae cultivation using an open pond configuration can be supplied CO_2 as a part of flue gas either from an adjacent power plant or from a purified CO_2 stream. Flue gas is approximately 14% CO_2 (Kadam, 2001) but can be purified to more than 99% CO_2 using the recyclable solvent monoethanolamine (MEA). The process known as MEA extraction is specifically targeted to remove large amounts of CO_2 from flue gas. The greatest advantage of MEA extraction is its relatively high carrying capacity in terms of the amount of CO_2 absorbed per unit volume of solvent. Less solvent circulation is required for a given system performance specification, leading to lower capital and operating costs than other solvent extraction methods (Herzog et al., 1991).

Little data exists documenting the benefits, if any, of using pure CO_2 compared to flue gas as a carbon source for algae cultivation. The assumption here is that both methods result in the same algae yield. The disadvantage of flue gas manifests from the compression and transportation of over seven times more gas volume for the same quantity of CO_2. For this reason, flue gas usage is only considered an option for algae cultivation facilities co-located with their CO_2 source. The disadvantage of pure CO_2 is that flue gas must be scrubbed to remove all non-CO_2 elements and subsequently compressed for transportation; flue gas requires only a simple blower. Scrubbing flue gas using the MEA process requires substantial quantities of steam and energy intensive compression. The compression can represent between 13% and 30% of the total process energy consumption (Herzog et al., 1991). Table 79 compares the energy usage of direct flue gas injection with two studies estimating energy usage for the MEA process. In both MEA studies, 98% of the energy results from producing steam to strip CO_2 from the amine solution. The remainder of the MEA process energy, and all energy for direct injection, comes from electricity.

Table 79: Comparison of energy inputs of direct flue gas injection and MEA extraction

	Total Energy (MJ/t_{CO2})	Total Energy (MJ/kg_{algae})	Emissions (g_{CO2}/kg_{algae})
Direct Injection			
Kadam (2001)	80	0.18	35
MEA			
Kadam (2001)	6650	14.63	911
Herzog et al. (1991)	6097-10248	13.41-22.55	835-1404
Note: 1) Steam production is assumed to occur using an 80% efficient industrial boiler			

The emissions to create a pure CO_2 stream are over 25 times higher than those from using flue gas from a co-located source. For this reason, the low and baseline scenarios adopted the assumption that algae cultivation occurs adjacent to a power plant, thus using flue gas as the

CO_2 source. The high emissions scenario assumed that CO_2 must be scrubbed from the flue gas using MEA extraction prior to compression for transportation and storage, according to the estimate by Kadam (2001). This assumption is appropriate for facilities that are not located in the immediate vicinity of a CO_2 source. Furthermore, transportation energy is not included in the estimates of Table 79 and would be a function of distance between the CO_2 source and algae cultivation facility.

7.6.4 Cultivation of Algae in Open Ponds

The process energy and nutrient requirements for algae cultivation were established from two sources: (1) a simulated open pond algae farm that was used to capture flue gas from an adjacent fossil power plant and (2) design specifications for an open pond system used for fuel production. The simulated open pond algae farm parameters were obtained for a 1000-hectare algal production system based on primary bench scale data and process modeling (Kadam, 2001). Weissman and Goebel generated the open pond design specifications for a 192-hectare system in 1987 (Weissman and Goebel, 1987). While Kadam presents all energy inputs of cultivation and harvesting as a single value, Weissman and Goebel show the inputs for the individual steps of mixing, pumping, primary harvesting and secondary harvesting. The data from both studies are in good agreement and were amalgamated to form the inputs to this analysis.

Since both studies quote similar electrical inputs, all electricity estimates were taken from Weissman and Goebel due to their higher degree of detail. In their design specifications, electric motors were assumed to operate with a drive efficiency of 70%. Drive efficiency encompassed the motor, an in-line speed reducer and a two-stage chain and sprocket reduction. This study considers a simulation year of 2015; hence, modern motors and gearing mechanisms could be incorporated. A modern three-phase (variable speed) electric motor meeting similar cost and performance requirements is rated as up to 92.4% efficient (Marathon Electric, 2009). The mechanical efficiency of a chain and sprocket has been found to reach 98.6% in ideal conditions and 81% in poor conditions (Spicer et al., 2001). Assuming both a motor and connection efficiency of 90%, the drive efficiency climbs to 81%, corresponding to an improvement of 15.7% above Weissman and Goebel's estimates. This efficiency improvement was applied to all process inputs related to electric motors or electric pumps.

The cultivation inputs used for the low, baseline and high emissions scenarios are shown in Table 80 (normalized per kilogram of algae) while harvesting and drying are dealt with in subsequent sections.

Table 80: Cultivation inputs per kilogram of algae for algae growth in an open pond

	Low	Baseline	High
Nutrients			
CO_2 (kg)	2.16	2.2	2.2
Nitrogen (kg)	0.044	0.053	0.053
Superphosphate (kg)	0.019	0.029	0.039
Potassium Sulfate (kg)	0.030	0.030	0.030
Process Energy			
Mixing (Btu)	281.8	281.8	281.8
Water Supply (Btu)	229.8	229.8	229.8
Nutrient Supply (Btu)	16.2	16.2	16.2

Nitrogen from fertilizer use was assumed to come only from ammonia. As a well-established fertilizer in the farming industry, ammonia has a production pathway within the GREET framework. The production pathways of superphosphate and potassium sulfate are not established within GREET. The process energies required for the production of superphosphate were adopted from Anderi Silva and Alexandre Kuley (2003). Specific inputs for potassium sulfate were not available in the literature. A final emissions inventory from potassium sulfate production

was taken from Kadam (2001). The emissions resulting from the production of nitrogen fertilizer, superphosphate and potassium sulfate are summarized in Table 81.

Table 81: Life cycle GHG emissions from the production of nutrients used in algae cultivation

	Nitrogen	**Superphosphate**	**Potassium Sulfate**[1]
CO_2	2537	235	-370[1]
CH_4	62.5	7.5	1.5
N_2O	5.6	1.3	1.5
Total	2605	244	-367
Notes:			
1) Negative process energy flows occur due to displacement allocation being used in their life cycle assessment			
2) Units are g CO_2e/kg of nutrient			

Nitrous oxide emissions from algae cultivation were estimated using IPCC Tier 1 methodology for flooded rice fields (De Klein et al., 2006). It was assumed that the conversion rate of nitrogen contained in a flooded rice field is similar to the conversion rate of nitrogen from an open pond. Open ponds were the only source of N_2O emissions from algae considered in this work. The IPCC Tier 1 methodology for flooded rice fields estimates the direct conversion rate of nitrogen from synthetic fertilizers as 0.3%. No mechanisms for indirect emissions were considered due to a lack of information. The formula used in this study for calculating N_2O emissions from algae cultivation is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{kg} \right) = \left(\left(\frac{g_{nitrogen\ fertilizer}}{kg} \right) \cdot 0.003 \right) \cdot \left(\frac{44 g_{N_2O}}{28 g_N} \right)$$

Equation 14

7.6.5 Nutrient Recycling via Anaerobic Digestion

Fertilizer production is a non-negligible GHG source in the algae to HRJ pathway. As can be seen from the data in Table 82, which were derived from Tables 80 and 81, the nitrogen fertilizer emissions are comparable to those from the baseline soybean to HRJ pathway. High quantities of fertilizer in any algal effluent leaving the system can also lead to eutrophication of natural waters. The additional fertilizer can over-stimulate the production of organic compounds leading to negative environmental effects such as hypoxia and reductions in water quality, fish, and other animal populations. If not properly contained, the environmental impacts of this process include decreased biodiversity, changes in species composition and dominance, and toxicity. These considerations warrant an investigation into nutrient recycling systems to reduce fertilizer demand. The analysis that follows assumes that all nitrogen and phosphorous is contained in the meal after oil extraction.

Table 82: GHG emissions resulting from fertilizer production within the algae to HRJ pathway for the three emissions cases

	Low	**Baseline**	**High**
Nitrogen (gCO_2e/kg$_{algae}$)	114.6	130.3	138.1
Superphosphate (gCO_2e/kg$_{algae}$)	4.6	7.1	9.5
Potassium Sulfate (gCO_2e/kg$_{algae}$)	-11.0	-11.0	-11.0

One means of nutrient recycling is anaerobic digestion, wherein microorganisms break down carbon-based matter in the absence of oxygen. It is widely used as a renewable energy source because the process produces a methane-rich biogas. Other products of anaerobic digestion are a liquid effluent and a solid digestate. Of interest here is the nutrient-rich liquid effluent that can be used to supplement fertilizers. A conventional reactor is maintained at an operating temperature of approximately 35°C. Retention times for conversion range from 20 to 30 days where about 60% of organic carbon is converted to biogas. The biogas composition is typically 60% methane and 40% CO_2 with traces of hydrogen sulfide and water vapor (Chynoweth et al, 2001). By using the algal meal co-products to feed a digester, a substantial nutrient fraction could

be recovered. Such systems would also allow for the recycle of carbon contained in the meal, thus reducing the demand on an external CO_2 source.

Several factors affect the rate of digestion and biogas production. The most important is temperature. Anaerobic bacteria communities can endure temperatures ranging from below freezing to above 57°C, but they thrive best at temperatures between 37°C and 54°C. Bacteria activity, and thus biogas production, falls off gradually from 35°C to 0°C (DOE, 2008). In some cases, the conventional design discussed above is being replaced by more innovative designs influenced primarily by the suspended solids content of the feed. Designs for feed with intermediate solids contents (such as sewage sludge or aquatic plants) involve recycling the solids following settling within the digester. Such designs have increased loading rates 20-fold and improved process stability. Furthermore, the biodegradability of certain feedstocks has been seen to exceed 90% (Chynoweth et al., 2001).

The process flows for an algae system that includes an anaerobic digester are shown in Figure 18. It is assumed that the algae are being grown in areas with sufficient heat and sunshine to sustain the required temperature of 35°C and the energy required to maintain a suitable temperature is negligible from the life cycle analysis standpoint. Furthermore, any additional heating requirements would be low-grade and could be met using flue gas before injection into the pond or exhaust gas from the drying facility.
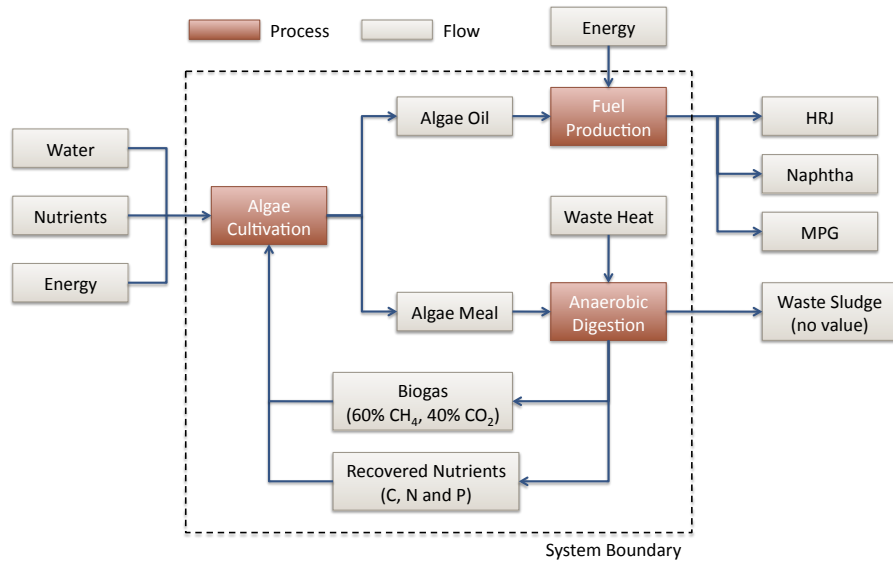
Figure 18: Process flows for algae oil HRJ using anaerobic digestion to recover nutrients from algae meal

Weissman and Goebel explored the potential for nutrient recovery from algae meal using anaerobic digestion. They projected that 50% of carbon not converted to biogas, 75% of all nitrogen and 50% of all phosphorous could be recovered in the liquid effluent. The remainders would be contained in the digestate, which is considered a waste of zero value.

The carbon content of algae meal can vary depending on the total carbon content and lipid faction of the algae. It is also directly proportional to the methane production potential from anaerobic digestion. Algae was assumed to have a constant carbon content of 60% while lipids were assumed to have a constant carbon content of 76.1% (based on their molecular formula). Under these assumptions, the carbon content of meal, and thus the methane production potential, is a function of lipid fraction.[58]

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[58] This assumption is most likely only valid for lipid contents between 0 and 50%; however, that is irrelevant because algae strains considered in this work do not exceed 50% lipids.

When digested, the resulting biogas can be used directly to meet internal heat demands or to fuel any other heat engine. Methane within the biogas can be upgraded to the same standards as fossil natural gas. If allowed, the biogas may be utilized within the local gas distribution networks. Gas must be cleaned to reach pipeline quality, and be of the correct composition for the local distribution network to accept it. These restrictions on raw biogas usage outside the system lead to the conclusion that on-site usage is the most practical use for biogas produced from anaerobic digestion.

7.6.6   Dewatering and Drying

After cultivation, algae can represent as little as 1 part in 2000 in water (Kadam, 2001; Weissman and Goebel, 1987). Conventional chemical oil extraction technologies require a feedstock that is approximately 90% dry. The task of extracting and drying the solid algae is the most energy intensive step of the cultivation process. As shown schematically in Figure 19, the algae must first be harvested, then dewatered and finally dried to reach the desired concentration. The applicability of algae as an environmentally beneficial biofuel when using conventional oil extraction facilities can be highly dependent on the extent of dewatering and the method of drying.
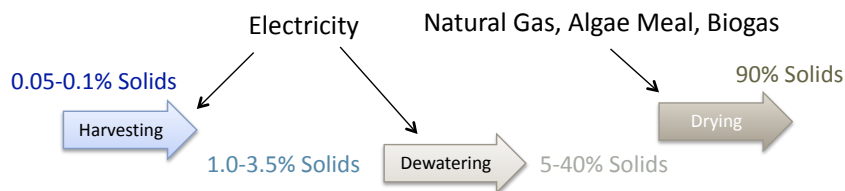
Figure 19: Flow chart showing the steps involved in dewatering and drying of algae

Generally, harvesting and initial dewatering is performed by mechanical or gravitational force while drying relies on direct heating to evaporate the remaining water. Due to the high latent heat of water, dewatering and drying have considerable impact on the life cycle GHG emissions of the pathway. Additional dewatering and less drying generally lead to reduced emissions. Solar, natural gas, algae meal and biogas from the anaerobic digestion of algae meal were considered as options for providing the drying energy. Common types of dewatering procedures are flocculation[59] or settling, filtering with a vacuum or press and centrifugation. The performance of each method is characterized by the total suspended solids (TSS) of the initial feed (given as a percentage by mass), the concentration factor,[60] energy usage per m^3 of throughput and energy usage per kg of processed solids. The initial TSS relates the energy usage per kg of processed solids to the energy usage per m^3 of throughout.

Weissman and Goebel consider three different two-stage harvesting and dewatering configurations. Each begins with a solids concentration of 0.075% and is comprised of primary harvesting using a micro strainer, vacuum belt filter or settling pond followed by secondary dewatering using a centrifuge to achieve 10% solids. The centrifuge dominates energy consumption from this step; hence, a higher concentration factor in the primary harvesting step reduces energy consumption in the centrifuge. Specifications for a belt-filter, micro-strainer and settling pond are given in Table 83 (Shelef et al., 1984). Notice that the concentration factor of the belt filter and settling pond are higher than that of the micro-strainer. In all cases, flocculent must be used to aid in the harvesting process; however, the environmental impacts of the flocculent were assumed to be negligible in terms of the life cycle GHG emissions.

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[59] Flocculation is a process where particles come out of suspension in the form of floc or flakes. The action differs from precipitation in that, prior to flocculation, the particles are merely suspended in a liquid and not actually dissolved in a solution.

[60] Concentration factor is defined as the ratio of initial feed volume to concentrated volume after separation.

Table 83: Energy consumption and performance specifications for primary algae harvesting mechanisms

Micro-strainer		Vacuum Belt Filter		Settling pond	
Electricity (Btu/kg)	Concentration Factor	Electricity (Btu/kg)	Concentration Factor	Electricity (Btu/kg)	Concentration Factor
23.8	10	47.6	70	46.9	50

It is desirable to extend the centrifuge energy consumption estimates to explore the impacts of varying the %TSS after dewatering. The result of combining the energy consumed per m^3 of throughput with the initial TSS is the energy consumed per kg of processed solids. The relationship between the energy consumed per kg of processed solids and concentration factor is plotted in Figure 18 and was subsequently used to define a continuous function for the centrifuge energy consumption. Data points originate from the three configurations described by Weissman and Goebel and independent average data for a solids ejecting disc centrifuge from Shelef et al. (1984). The advantage of this type of centrifuge for algae harvesting is its ability to reliably dewater to 12-25% solids, although, solids finer than algae may be retained in the overflow stream. Based on a review of the technology survey conducted by Molina Grima et al. (2003) and the conclusions of Shelef et al. (1984), even the best centrifuges appear to have a concentration limit of 25% TSS. More common results place the final TSS level around 15%. While following the energy consumption mapped in Figure 20, this analysis assumed that dewatering can occur to a maximum of 25% TSS.
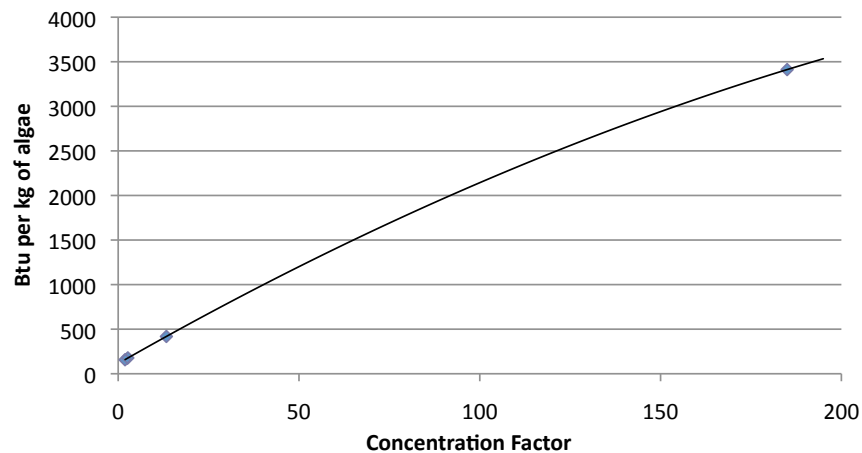
Figure 20: Energy consumption of a centrifuge secondary harvesting mechanism. Line is a fit through four data points

The values in Figure 20 are based on historical studies and may be pessimistic for a simulation year of 2015. As such, the efficiency improvement of 15.7% discussed with regards to cultivation was applied to all electrical inputs of harvesting and dewatering.

Each method must be considered within the context of reliability and scalability. The advantages of micro-strainers are their operation, low energy consumption, and high filtration ratios. Their problems include incomplete solids removal and the buildup of bacterial and algae slime on the micro fabric (Shelef et al., 1984). Regardless of their merits, micro-strainers result in high-energy consumption at the centrifugation stage due to their low concentration factor, making them an undesirable choice. Belt filters and settling ponds are both promising from the perspective of energy consumption and they have been assessed as a reliable method for harvesting (given the use of a flocculent). Furthermore, the concentration factors estimated by Weissman and Goebel are reasonable when compared to data from Shelef at al. (1984). Because some experiments have indicated that belt filters may be less successful at harvesting small algae such as *Chlorella* (Weissman and Goebel, 1987), settling ponds were chosen for the low, baseline and highemissions scenarios.
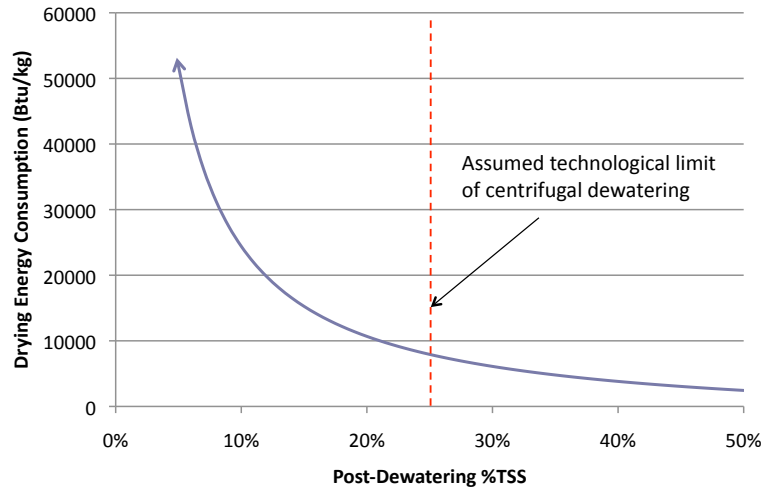
Figure 21: An examination of the post-dewatering moisture content impact on energy consumption of drying algae to 90% total suspended solids

The impacts of dewatering on drying energy requirements are demonstrated in Figure 21. As the extent of dewatering increases, there is a sharp reduction in the energy required for drying to 90% TSS. Drying energy must be in the form of direct heat in order to evaporate water from the algae slurry. Solar drying is considered to be an optimistic choice but is not out of the realm of possibility, with some reports estimating a drying area only 12% of that required for growing (Shelef et al., 1984). The low emissions scenario therefore adopted solar drying while the baseline and high emissions scenarios require the combustion of a process fuel for heat. The process fuel may be natural gas, algae meal or biogas from the anaerobic digestion of algae meal.

As discussed in the previous section, an opportunity for system integration is available by using energy contained in the algae meal to supply internal energy demands. This energy can be obtained by burning the meal directly in a boiler or by combusting the biogas produced from anaerobic digestion. Depending on the quantity of meal produced per kg of algae, some or all of the drying energy can be supplied by these methods. This threshold point, where all drying energy is supplied by the meal, is fundamentally related to the lipid content and the post-dewatering %TSS content. Designing a sustainable drying technique that does not rely on the combustion of non-renewable fuels is essential in creating a sustainable fuel.

The threshold lipid content is also dependent on whether the meal is burned directly or whether anaerobic digestion is used to obtain biogas. These relationships are explored in Figure 22 where two different biomass conversion factors from anaerobic digestion are compared to directly burning the meal in a boiler. While both methods result in zero net GHG emissions from the drying process, nutrient recovery is not possible with meal combustion. The opportunity to recover nutrients leads to the conclusion that an algae cultivation facility concerned with minimizing GHG emissions would be equipped with an anaerobic digester. Therefore, the low and baseline emissions scenario assumed the use of an anaerobic digestion system with biomass conversion factors of 80% and 70% respectively. The high emissions scenario assumed the meal is sold as animal feed.
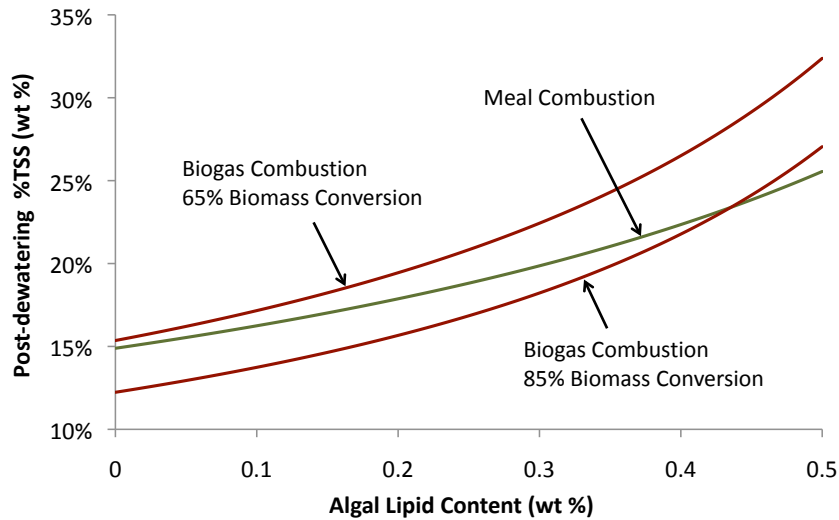
Figure 22: Algal lipid content and post-dewatering moisture content that result in a sustainable drying process. Points above each line require additional energy to be added (e.g. from natural gas)

For an algae strain comprised of 25% oil by mass, sufficient meal is produced to supply the energy to dry other algae from a post-dewatering %TSS content of 19% via direct burning and from 17% to 21% via anaerobic digestion (depending on anaerobic digester efficiency). If the post-dewatering %TSS content were above these threshold values then the algae meal would need to be supplemented by natural gas to supply sufficient energy for drying to 90% TSS.

The decision to use algae meal or natural gas as a drying fuel must be made in the context of the economic, technological and energy usage limitations of dewatering. The trade-offs between these factors and other system parameters can only be understood by considering the entire algal production system.

7.6.7 Transportation of Algae to Production Facilities

The production of algae in this analysis is assumed to take place in the southwestern United States. After harvesting, assumptions about the transportation mechanisms and distances have been modified from GREET defaults for the soybeans to HRJ pathway. When meal is used as an energy source, oil extraction must be collocated with the cultivation facility to enable the recycling and integration of energy flows. In these configurations, only the oil is transported from the cultivation site to a hydroprocessing facility. Specifically, the transportation is by tanker truck in 25-ton shipments; raw algae transportation would occur in truckloads of only 15 tons. In cases where meal is sold as a supplement to animal feed, standard GREET transportation assumptions were adopted. As was the case in every other pathway considered, the contribution from transportation elements in the life cycle analysis are sufficiently small to be within the margin of error; therefore, the importance of these assumptions is minimal. As such, the geographic location of the facility is of secondary importance to the overall results. However, as was discussed in Section 7.6.3, the relative location of the algae facility to its CO_2 source is very important to the life cycle GHG emissions.

7.6.8 Extraction of Oil from Algae

The extraction of oil from algae is currently an area of considerable research. This analysis used a modified version of the process for oil extraction from soybeans established by Sheehan et al. (1998a) that includes only the processes relevant to algae in an N-hexane extraction facility. The process energies from Sheehan et al. were converted from energy per unit mass of biomass to energy per unit mass of oil using the oil fractions from the low emissions, baseline and high emissions scenarios. The outputs and energy consumption assumed in the extraction of oil from algae are shown in Table 84.

Table 84: Outputs and process energy for N-hexane oil extraction from algae

	Low[1]	Baseline[1]	High[1]
Algae Lipid Content	40%	25%	15%
Meal Produced (lb_{meal}/lb_{oil})	1.5	3	5.67
Receiving and Storage			
Electricity	20.9	33.5	55.8
Algae Preparation			
Electricity	4.0	6.5	10.8
Steam	172.5	276.0	460.0
Oil Extraction			
Electricity	12.9	20.6	34.4
N-hexane[3]	105.9	169.5	282.5
Meal Processing			
Electricity	52.2	83.5	139.1
Steam	505.4	808.6	1347.7
Oil Recovery			
Electricity	1.4	2.2	3.6
Steam	86.7	138.7	231.1
Solvent Recovery			
Electricity	1.9	3.0	5.0
Steam	0.0	0.0	0.0
Oil Degumming			
Electricity	6.1	9.7	16.1
Steam	68.1	109.0	181.6
Waste Treatment			
Electricity	2.0	3.3	5.4
Steam	36.6	58.6	97.7
Total			
Electricity	101.4	162.2	270.3
Natural Gas[2]	1086.6	1738.6	2897.6
N-hexane[3]	105.9	169.5	282.5
Notes:			
1) All values are in Btu per pound of oil			
2) Steam is assumed to be generated from natural gas with an efficiency of 80%.			
3) GREET uses Liquefied Petroleum Gas (LPG) as a surrogate for N-hexane when calculating emissions			

As discussed previously, algae meal resulting from oil extraction can be burned in a utility boiler, used as a feed to an anaerobic digester or sold as a substitute product to soy meal consumed as animal feed. When burned in a boiler, the direct heat can be used to fuel the drying process. If there is excess meal after drying is completed, the rest is sold as animal feed; if meal is unable to provide all the drying energy then natural gas could be used to supply the difference. When digested, the resulting biogas can be used directly to fuel the drying process or any other heat engine. Methane within the biogas can be upgraded to the same standards as fossil natural gas. If allowed, the biogas may be utilized within the local gas distribution networks. Gas must be cleaned to reach pipeline quality, and be of the correct composition for the local distribution network to accept it. These restrictions on raw biogas usage outside the system lead to the conclusion that on-site electricity generation is the most practical use for biogas not consumed for drying. Any electricity generated over and above the internal demand of cultivation is considered exportable to the grid and emissions from cultivation and oil extraction were allocated between oil and exported electricity based on energy. If insufficient biogas is produced to provide all the drying energy then natural gas could be used to supply the difference.

When sold as an independent product, market value allocation was used in partitioning emissions between algae oil and meal. As discussed with regards to soy oil to HRJ, using market value allocation captures temporal changes in the allocation fractions between products. Since no established market price exists for algae oil, or algae meal, they were assumed related to the prices of soy oil and soy meal. Prices assumed for soy oil and soybean meal are the 2015 projections made by FAPRI (FAPRI, 2009). Algae oil was assumed to have the same market price as soy oil, $1.05/kg, while algae meal was assumed to have a price related to that of soy meal based on protein equivalency. Protein content serves as a common denominator because both meals would be used as animal feed. Soybean meal was assumed to have a protein content of 48% (Ahmed et al., 1994) and algae meal to have an average protein content of 52% (Becker, 2007).[61] Based on these protein contents, a price of $0.31/kg was calculated for algae meal.

7.6.9   Sensitivity Analysis and System Design

For algal facilities yielding a fixed lipid content, the life cycle GHG emissions of fuel production are most sensitive to the extent of dewatering. Using the relationships and assumptions discussed in the previous section, life cycle GHG emissions were explored for a range of post-dewatering %TSS contents. Using assumptions consistent with the baseline case, the results are plotted in Figure 23 for three different system configurations. It can be seen that maximum dewatering is always optimum for minimizing life cycle GHG emissions, regardless of the source of drying energy and use of meal. If the post-dewatering %TSS content is less than 15%, algal HRJ results in higher GHG emissions on a life cycle basis than jet fuel from conventional petroleum. Recall that dewatering to 25% TSS content was assumed as the maximum dewatering achievable with current technology. By comparing the two natural gas drying cases, one can see the benefit of nutrient recycling as cases using fertilizer to meet all nutrient demands resulted in approximately twice the GHG emissions, for higher post-dewatering %TSS contents, as those where nutrients were recycled. The lowest life cycle GHG emissions were achieved by using biogas from the anaerobic digester and recycled nutrients.

The low emissions, baseline and high emissions scenarios assumed post-dewatering %TSS contents of 25%, 20% and 15% respectively. While the high emissions scenario assumed natural gas as the fuel used for drying, the baseline scenario was constructed with an anaerobic digester producing biogas configured to supply the drying energy. The low scenario assumed solar drying with an anaerobic digester producing biogas, which would then be used for on-site electricity generation to meet internal demands with excess power being exported to the gird.

Local sensitivity analysis was conducted on the anaerobic digestion biomass conversion efficiency, lipid content and CO_2 injection method. Each parameter was varied with all others held at their baseline values with the impacts quantified as a percent change from the baseline value. Figure 24 presents this information in a manner that allows the magnitude of each change to be seen in comparison to the others. Section 7.6.3 discussed the use of MEA extraction to create a pure CO_2 stream from flue gas. The importance of collocating the cultivation facility to its CO_2 source is further emphasized by these results. The unexpected result that life cycle GHG emissions are reduced by using algae with lower lipid contents can be understood by considering that additional meal, and hence more biogas and by extension more energy, is available per kg of oil if the algae have lower lipid contents. Finally, the conversion efficiency is directly proportional to the quantity of energy available from the meal, making it the second most sensitive operational specification of the system (second to post-dewatering %TSS content).

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[61] The meal is assumed to be the only oil-containing portion of algae. This hypothesis is verified by considering soybeans and soy meal. Soybeans have a protein content of 40% (GREET, 2008) and an oil fraction of 18.25% (Sheehan et al., 1998a) leading to a soy meal protein content of 48.9%. The average protein and lipid contents of the strains documented by Becker (2007) were 45.1% and 13.2%, respectively.
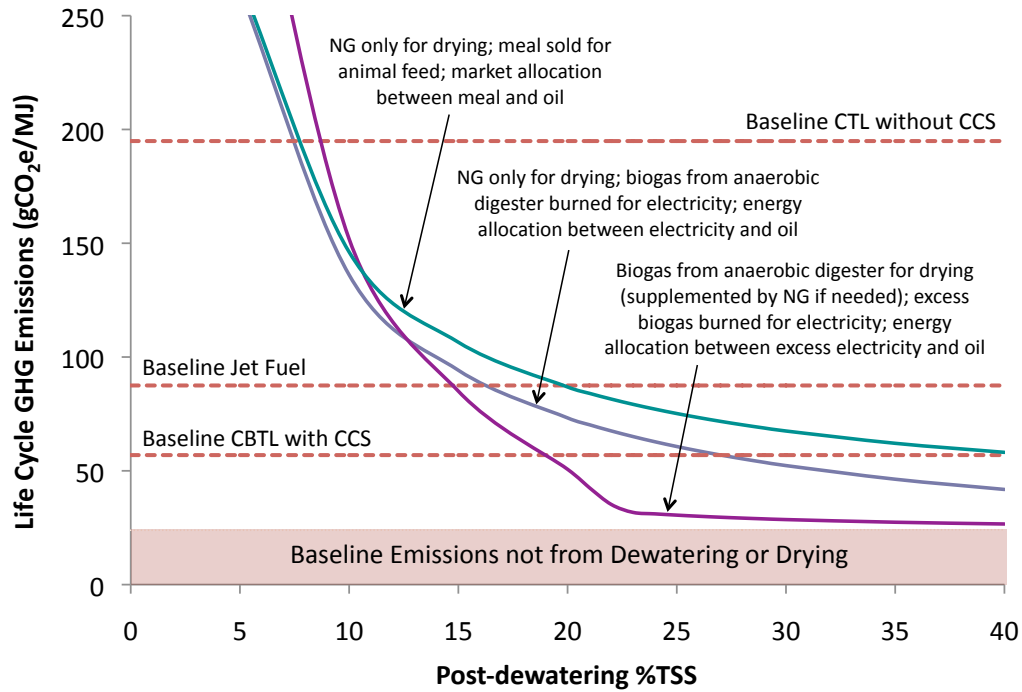
Figure 23: Life cycle GHG of HRJ production from algae as a function of the extent of dewatering. Three different system configurations were explored which are described more fully in the figure
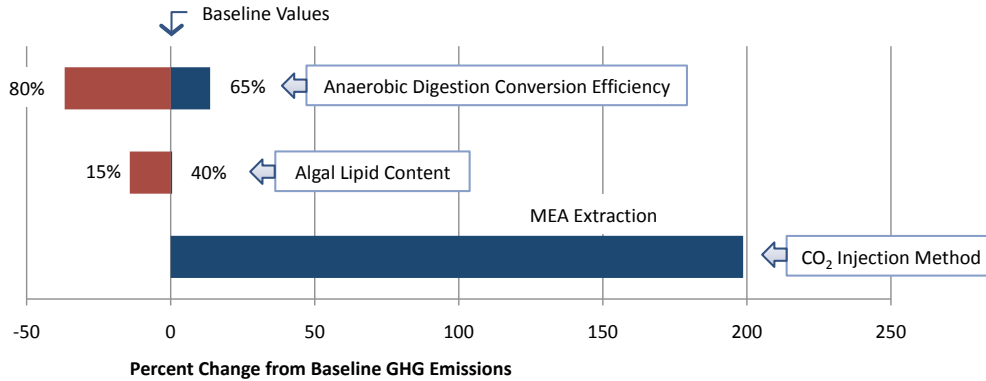
Figure 24: Sensitivity analysis of operational specifications and configurations of HRJ production from algae

7.6.10   Results

The key assumptions and corresponding life cycle GHG emissions in the production and use of HRJ from algae oil are outlined in Table 85. It should be noted that although water usage was not quantified in this section, it is a part of ongoing research efforts. The most notable differences between the cases lie in the recovery step and the WTT CH_4. The recovery step includes emissions from CO_2 injection, dewatering and drying. The dominant factors of each life cycle stage were described in previous sections and are the driving sources of variation between scenarios. The increased WTT CH_4 in the high emissions scenario is a result of natural gas usage for MEA extraction and drying.

The life cycle GHG emissions resulting from the production and use of HRJ from renewable algae oil range from 0.16 to 2.2 times those from conventional jet fuel. Variation in the biomass credit is due to minor changes in the allocation schemes used throughout the pathway.

Table 85: Life cycle emissions from the algae oil to HRJ pathway

	Low	Baseline	High
Key Assumptions			
Algae Yield (g/m^2/day)	40	25	20
Algal Lipid Content	40%	25%	15%
Anaerobic Conversion Efficiency	80%	70%	n/a
Life Cycle CO_2 Emissions by Stage			
Biomass Credit (gCO_2/MJ)	-73.7	-70.5	-68.9
Recovery of feedstock (gCO_2/MJ)	5.4	29.6	143.1
Transportation of feedstock (gCO_2/MJ)	0.3	0.3	1.2
Processing of feedstock to fuel (gCO_2/MJ)	7.1	10.3	13.2
Transportation of jet fuel (gCO_2/MJ)	0.6	0.6	0.6
Combustion CO_2 (gCO_2/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO_2 emissions (gCO_2/MJ)	-60.3	-29.7	89.2
WTT CH_4 emissions (gCO_2e/MJ)	0.7	1.8	27.7
WTT N_2O emissions (gCO_2e/MJ)	3.3	8.1	5.8
Total WTW GHG Emissions (gCO_2e/MJ)	14.1	50.7	193.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.16**	**0.58**	**2.21**

The uncertainty surrounding N_2O emissions from algae cultivation is larger that other pathways due to the assumption that algae ponds have the same emission factor as flooded rice fields. While there is little existing information on N_2O formation from algae ponds, the results from Table 85 indicate that N_2O represents less than 16% of the total life cycle GHG emissions. The reader should be aware of the additional uncertainty with respect to N_2O emissions from algae cultivation when comparing different pathways for GHG reduction potential in sections 8.1 and 8.2.

7.7   HRJ from Salicornia Oil

The creation of HRJ from salicornia oil is not a pathway available in GREET; hence, supporting information was obtained from the literature and a pathway was created within the GREET framework. Salicornia is both a wild and cultivated annual shrub that germinates, grows and reproduces in areas of high salinity such as coastal shorelines, marshes or inland lakes. The plant itself is leafless with green jointed and succulent stems that form terminal fruiting spikes in which seeds are borne. In subtropical regions, salicornia can grow up to 50cm in height with most of the seed spikes on the upper one third of the plant (Anwar et al., 2002). Since oil containing seeds represent only a small fraction of the total plant, the deterministic factor in the life cycle analysis of jet fuel from Salicornia is the usage of non-oil containing biomass. The cultivation of halophytes on arid or semi-arid land, where there is little or no carbon stock naturally present in the soil, can lead to substantial net long-term carbon storage. Salicornia falls into the category of coastal halophytes because of its ability to grow in saltwater.

This work considers the production of HRJ from Salicornia oil while varying the crop productivity, oil yield, cultivation inputs and nutrient usage to establish the low emissions, baseline and high emissions scenarios. Due to the nature of Salicornia growth, the combination of either electricity generation or Fischer-Tropsch synthesis with an HRJ facility were explored. Long-term carbon sequestration potential is quantified through scenarios that vary the beneficial GHG credit from land use change.

7.7.1 Biomass, Seed and Oil Yield

In a similar manner to jatropha, additional products are created while growing seeds for oil production. When considering the yield of salicornia, three yield parameters define productivity: total biomass yield per hectare, seed yield per kilogram of total biomass and seed oil fraction. Total biomass is defined as seeds and straw biomass (similar to other forms of herbaceous biomass considered for BTL facilities). Data used to establish growth and oil yields was taken from field trials conducted at Puerto Penasco, Sonora, Mexico in an extreme coastal desert environment at the northern Gulf of California (Glenn et al., 1991). The ranges for the emissions scenarios were established using data from multiple plots taken from 1982 through 1988. The baseline total biomass yield and seed yield were defined by taking a weighted average (weighted by the number of plots) of the data from Glenn et al. (1991). The low emissions and high emissions data were defined as plus and minus one standard deviation from the weighted mean, respectively (standard deviation also weighted by the number of plots). The oil fractions adopted were taken directly from the range quoted by Glenn et al. (1991). The assumptions regarding total biomass yield, seed yield and oil fractions for Salicornia are given in Table 86.

The total biomass yield is almost an order of magnitude higher than the seed yield (per hectare). In the baseline case, this translates to 7.22 kg of straw biomass for every kilogram of seeds, 25.6 kg of straw biomass per kilogram of oil and 43.5 kg of straw biomass per kilogram of HRJ. The most important point highlighted by these numbers is that the production of salicornia will not be driven by the demand for its seeds, but rather the demand for its straw biomass. Were salicornia grown to use its straw biomass for either electricity production or as feedstock to a Fischer-Tropsch facility, the production of hydroprocessed renewable fuels from its oil seeds would be a high value co-product from the process.

Table 86: Salicornia yield and oil fraction assumptions

	Low	Baseline	High
Total Biomass Yield (kg/ha/yr)	17614	16247	14880
Seed production ($g_{seed}/kg_{total\ biomass}$)	142	122	101
Seed Yield (kg/ha/yr)	2506	1977	1504
Seed Oil Fraction	33%	28.2%	26%

In section 7.7.5, a system is considered where straw biomass is used for renewable power generation and seeds are used for HRJ production. The heating value of salicornia biomass varies with the species, the chief variable being the ash content, but in general salicornia biomass falls in the range of Lignite A or B coals (Glenn et al., 1992).

In section 7.7.6, a system is considered where straw biomass is used as a feedstock for a BTL facility and seeds are used for HRJ production. The BTL plant is similar to that discussed in section 6.4; namely, that biomass feedstock is used to meet internal process energy needs, with little or no excess electricity produced for export, and commercial quality middle distillate fuels such as diesel and jet fuel are produced.

7.7.2 Cultivation of Salicornia

The cultivation of salicornia is generally motivated by the opportunity to directly sequester carbon from the atmosphere. This occurs by reversing the trend towards desertification through cultivation of salicornia on saline lands and re-vegetation of degraded rangelands (Glenn et al., 1993). For this reason, process energy requirements for the cultivation of salicornia are sometimes presented in the form of fossil carbon emitted per carbon absorbed during growth. Glenn et al. (1992) calculated the carbon costs of salicornia assuming that all farm operations, from pumping water to planting, harvesting, baling and hauling the crop were performed using diesel fuel. They found that salicornia required 225kg to 300kg of fossil carbon for each 1000kg of atmospheric carbon fixed. The low emissions and high emissions case were defined as the lower and upper extremes, respectively, while the mean is assumed in the baseline case. All calculations assumed diesel fuel composed of 85.6% carbon (see Table 87) and salicornia composed of 24.7% carbon (Glenn et al., 1992).

The use of fertilizers in the cultivation of salicornia is a subject of debate. The test plots created by Glenn et al. in the Sonoran Desert were irrigated solely with hyper saline seawater that had first been used in a shrimp aquaculture facility (O'Leary et al., 1985), which added nitrogen and other nutrients to the water. In later trials, fields irrigated with un-enriched seawater received fertilizer additions equivalent to 200kg of nitrogen per hectare as urea, di-ammonium phosphate or ammonium nitrate (Glenn et al., 1991). The degree to which fertilizer must be added is a function of the coupling between aquaculture farms and salicornia farms. Large-scale coastal shrimp farms have caused algal blooms and disease problems in rivers and basins that receive their nutrient-rich effluent. The same problems can be expected to occur from the large volume of highly saline drainage water containing unused fertilizer discharging from salicornia farms (Glenn et al. 1998). When coupled together, salicornia farms could help mitigate the problem if aquaculture effluent is recycled onto a salicornia farm instead of being discharged directly to the sea; hence, any assumptions made regarding fertilizer use carry a degree of uncertainty. The assumptions adopted in this work to establish a reasonable range deal with varying the fraction of irrigation water that comes from aquaculture facilities. The low emissions case assumed 100% of the irrigation water was pre-enriched from an aquaculture facility; the baseline case assumed 50% was pre-enriched, leading to 100kg of nitrogen usage per hectare, and the high emissions case assumed the full 200kg of nitrogen usage per hectare.

Irrigation is the single most expensive production cost in growing salicornia. In general, seawater irrigation requires copious and frequent, sometimes even daily, irrigation to prevent salt from building up in the root zone. Partially offsetting this effect is that increased salinity of irrigation water can lead to higher water usage efficiencies; therefore, higher salinity leads to less water being required to produce a kilogram of dry biomass (Glenn et al., 1992). During cultivation experiments in the Sonoran Desert, it was determined that salicornia can thrive when water salinity exceeds 100ppm (three times the normal ocean level); however, the volume of seawater required for irrigation is 35% more than the value that would be required if freshwater were being used. The additional volume is needed to control salt levels in the soil (Glenn et al., 1998). Although more volume is required for salicornia cultivation, seawater farms often require less water lifting than conventional farms, which may lift water from wells far deeper than 100 meters. This difference results in seawater farms using less energy for water pumping than freshwater farms. The life cycle GHG emissions calculated in this work include the emissions from pumping irrigation water but secondary effects of salt-water irrigation were not quantified. Glenn et al. (1998) showed that normal farm and irrigation equipment could be modified such that it is protected from salt damage from seawater.

The input assumptions discussed above are summarized in Table 87.

Table 87: Input assumptions regarding the cultivation of salicornia

	Low	Baseline	High
Process Fuels			
Diesel (kg/kg$_{C\ Fixed}$)	225	262.5	300
Diesel (Btu/kg$_{seed}$)[1]	18557	25316	34831
Fertilizer Use			
Nitrogen (kg/ha/yr)	0	100	200
Nitrogen (g/kg$_{seed}$)[1]	0	50.6	133.0
Notes:			
1) Calculated using the yield assumptions from Table 86			

Emissions from N_2O were estimated using IPCC Tier 1 methodology (De Klein et al., 2006). Due to the lack of data, N_2O emissions from nitrogen in above and below ground crop residues were not accounted for in this analysis. The IPCC Tier 1 methodology estimates the combined direct and indirect conversion rate for nitrogen from synthetic fertilizers as 1.325%. These rates include the atmospheric deposition of nitrogen volatilized from managed soils as well as nitrogen from leaching and runoff. The formula for calculating N_2O emissions from salicornia cultivation is given by:

$$N_2O \text{ Emissions} \left(\frac{g_{N_2O}}{kg_{seed}} \right) = \left(\left(\frac{g_{nitrogen\ fertilizer}}{kg_{seed}} \right) \cdot 0.01325 \right) \cdot \left(\frac{44g_{N_2O}}{28g_N} \right) \qquad \text{Equation 15}$$

7.7.3 Transportation of Salicornia Biomass and Seeds to Production Facilities

The production of salicornia biomass and seeds was assumed to take place on the southwestern coast of the United States. After harvesting, GREET default values from the soybeans to HRJ pathway were assumed in regards to transportation mechanisms and distances travelled for the seeds. The assumptions of transportation mechanisms and distances for straw biomass are consistent with GREET defaults for herbaceous biomass in the BTL pathway. Specifically, the transportation of seeds is by truck to a local storage area and subsequently to a local oil extraction facility. The transportation of straw biomass is also by truck directly to an F-T or renewable power generation facility assumed to be the same distance as the HRJ facility. The difference between seed and straw biomass transportation is that seeds are loose and moved in loads of 15 short tons while straw biomass is baled and moved in loads of 24 short tons. Even after accounting for transportation of both seeds and straw biomass, the transportation elements of the life cycle GHG emissions are sufficiently small to be within the margin of error; therefore, assumptions in this area carry little consequence.

7.7.4 Oil Extraction from Salicornia Seeds

As there is no commercially available process for extracting oil from salicornia seeds, the extraction process was modeled with the soybean oil extraction process described by Sheehan et al. (1998a). Glenn et al. (1998) classify salicornia oil as highly poly-unsaturated and similar to safflower oil in fatty-acid composition (as presented in Table 41). They also conclude that it can be extracted from the seed and refined using conventional oilseed equipment. This same method was applied to both algae and jatropha oil extraction. Salicornia seeds more closely resemble soybeans than either algae or jatropha; hence, less modification to the soybean process was required. Specifically, pre-drying of seeds was eliminated and seed shell removal energy was included. Field drying of the seeds, prior to harvest was assumed and the energy for removing shells from jatropha seeds was applied to salicornia seeds. The process inputs assumed in this analysis are summarized Table 88.

7.7.5 Co-Product Usage and Allocation Methodology

Based on the yields adopted in the baseline case, 7.22 kg of straw biomass is produced for each kilogram of seed. Although salicornia could be grown for its seeds with the straw biomass tilled back into the field to facilitate higher rates of long-term carbon sequestration in the soil, it is doubtful that this would be done in practice. In an economy where 85% of all energy is derived from fossil carbon sources, the waste biomass could be used as a renewable energy source thus displacing fossil-based electricity or heat generation. Assuming the energy content of straw biomass is 16.3 MJ/kg, within the range of lignite A and B coals, the energy contained in straw biomass created per pound of oil is 10.5 times that contained in the oil itself.

7.7.5.1 Electricity Production from Biomass Co-products

The use of straw biomass co-product for electricity production was modeled with the assumption that to the electricity production occurred at the same location as the oil extraction and HRJ production. Three allocation methodologies were envisaged for this configuration. A summary of each is given in Table 89. The first methodology, which is the most straightforward, would displace average US grid electricity with the result being the lowest life cycle emissions for HRJ. The second methodology, system level energy allocation of recovery and feedstock transportation emissions, is the most complicated and is shown schematically in Figure 25. The unallocated cultivation and transportation emissions are first divided between HRJ production and electricity production based on the energy of the final products (39.9% and 60.1% respectively); the emissions associated to HRJ production are then allocated between oil and meal based on market value (61.5% and 39.5% respectively); both are then summed back together and allocated based on energy between HRJ and all other energy products from the system (15.4%

and 84.6% respectively). The other energy products from the system are comprised of renewable naphtha, mixed propane gas and electricity. The third allocation methodology involves initially allocating only the cultivation emissions between the seeds and straw biomass based on market valuation (67.2% and 32.8% respectively).[62] The life cycle GHG emissions resulting from each of the aforementioned systems are compared in Figure 26.

Table 88: Process inputs for extracting oil from salicornia seeds. All values are in Btu/lb of oil

	Low[1]	Baseline[1]	High[1]
Receiving and Storage			
Electricity	20.9	33.5	55.8
Salicornia Preparation			
Electricity	4.0	6.5	10.8
Steam	172.5	276.0	460.0
Oil Extraction			
Electricity	12.9	20.6	34.4
N-hexane	105.9	169.5	282.5
Meal Processing			
Electricity	52.2	83.5	139.1
Steam[3]	505.4	808.6	1347.7
Oil Recovery			
Electricity	1.4	2.2	3.6
Steam	86.7	138.7	231.1
Solvent Recovery			
Electricity	1.9	3.0	5.0
Steam	0.0	0.0	0.0
Oil Degumming			
Electricity	6.1	9.7	16.1
Steam	68.1	109.0	181.6
Waste Treatment			
Electricity	2.0	3.3	5.4
Steam	36.6	58.6	97.7
Totals			
Electricity	101.4	162.2	270.3
Natural Gas[2]	1086.6	1738.6	2897.6
N-hexane[3]	105.9	169.5	282.5
Notes:			
1) All values are in Btu per pound of oil 2) Steam is assumed to be generated from natural gas with an efficiency of 80%. 3) GREET uses Liquefied Petroleum Gas (LPG) as a surrogate for N-hexane when calculating emissions			

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[62] Market value of seeds is the sum of market value of oil and meal individually. Value of salicornia oil was taken as equivalent to soy oil while the value of salicornia meal was found by scaling the value of soy meal by the ratio of protein contents. Soy meal has an average protein content of 48% (Ahmed et al., 1994) while salicornia meal has a protein content of 42% (Glenn et al., 1992). The market value of straw biomass was assumed to be proportional to the cost of coal based on energy value. The average cost of coal for electric utilities in 2008 was $2.07/mmBtu (EIA, 2009a).

Note: "HR" denotes Hydroprocessed Renewable
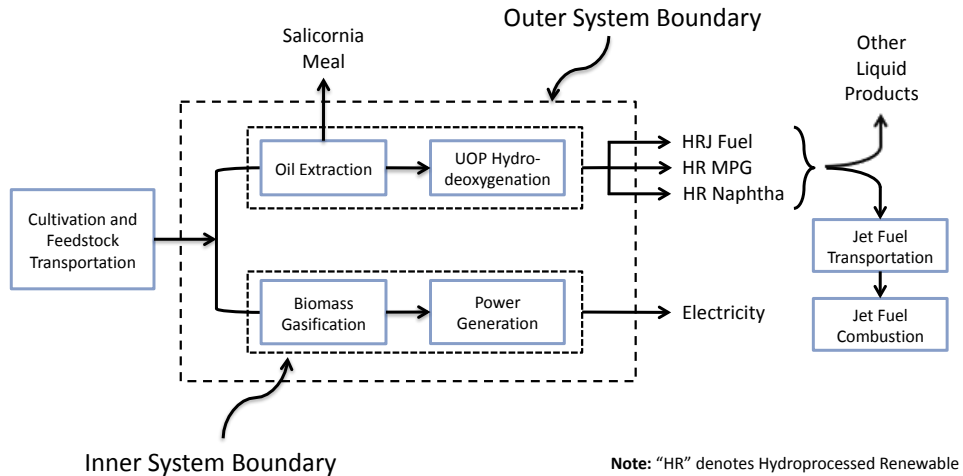
Figure 25: System boundary definitions for system level energy allocation between HRJ production from salicornia oilseeds and electricity generation from salicornia straw biomass

Table 89: Allocation methodologies examined for the production of HRJ from salicornia oil when using the straw biomass co-product for electricity generation

Scenario	Co-Product	Use	Emissions Allocation Method
1	Straw biomass	Burned for electricity production	Displacement method (system expansion) where electricity from US grid is displaced. The corresponding emissions credits are given to the fuel production process.
2	Straw biomass	Burned for electricity production	Electricity production considered a parallel process to HRJ production. Cultivation and feedstock transportation emissions are allocated using a system level energy allocation scheme between all energy products from the system.
3	Straw biomass	Burned for electricity production	Cultivation and feedstock transportation emissions are allocated prior to any electricity production based on market valuation of the unprocessed seeds and straw biomass.
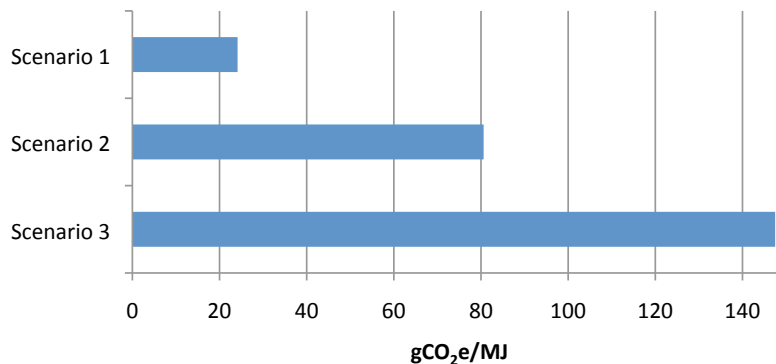
Figure 26: Comparison of life cycle GHG emissions from the allocation methodologies of Table 74 based on the production of HRJ from salicornia oil when using the straw biomass co-product for electricity generation

As was observed for other fuel pathways, different allocation schemes result in varied emissions within the same pathway. The most important point to realize from this configuration is that there

are 5 MJ of electricity generated for each MJ of HRJ. Hence, giving a liquid fuel credit for this electricity will reduce its emissions considerably. Additionally, when only the oilseeds are used for fuel production, the yield of HRJ per hectare is small, being less than even soybeans. This underscores the importance of the straw biomass as the driving renewable energy resource from salicornia cultivation.

7.7.5.2   Fischer-Tropsch Fuel Production from Biomass Co-products

If the straw biomass co-product from salicornia oilseeds is used for electricity production then jet fuel represents only 15% of the total energy products created. This is a smaller fraction than any other oilseed crops considered in this work and is not conducive to being a part of a large-scale biofuel production network. In section 6.4, jet fuel from solid terrestrial biomass via the Fischer-Tropsch process was discussed with switchgrass as the feedstock. Any carbon containing matter could theoretically be used to create syngas, which contains primarily H_2, CO and CO_2. This section discusses the implementation of a fuel production configuration where salicornia oilseeds are processed through an HRJ facility and all of the straw biomass is used as feedstock to a BTL facility.

By incorporating a BTL facility producing 25% jet fuel, the total yield of jet fuel grows to be three times larger than that from HRJ alone for the same quantity of feedstock consumed. Furthermore, F-T diesel and naphtha are also created which contribute to GHG mitigation efforts in other energy consuming sectors. For these reasons, this work focused on a coupled HRJ and F-T facility. All assumptions discussed thus far were applied to the HRJ section, while input assumptions pertinent specifically to the BTL section are summarized in Table 90.

Table 90: Input assumptions relevant to a BTL facility using salicornia straw biomass as feedstock

|	Low	Baseline	High
BTL Plant Efficiency	52%	45%	42%
F-T Jet Fuel Yield (fraction of all F-T fuel products)	30%	25%	20%
Straw Biomass Energy Content (MJ/kg)	17.4	16.3	15.1
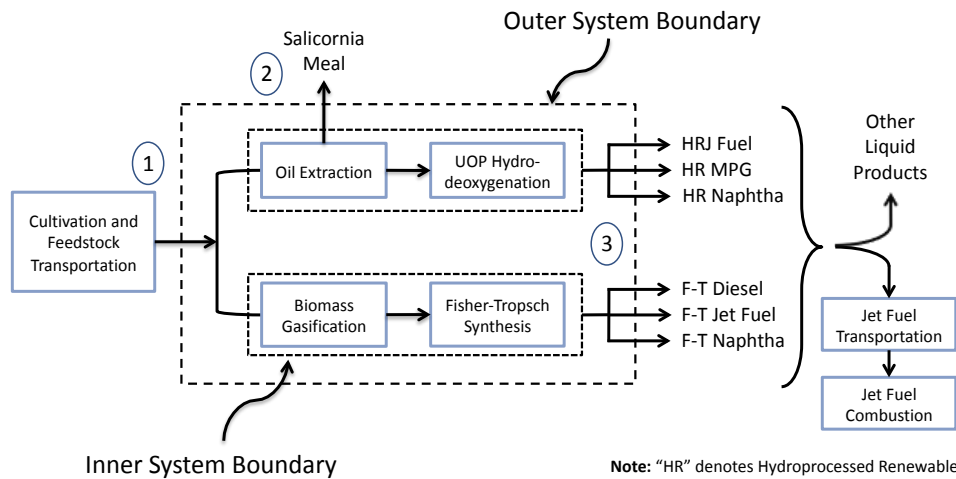
Figure 27: System boundary definitions for system level energy allocation of coupled HRJ and F-T fuel production from salicornia oilseeds and straw biomass

As was the case when electricity was produced, the allocation methodology is an important choice in determining the final life cycle GHG emissions of the jet fuel. Figure 27 outlines how the coupling of an HRJ facility with an F-T facility was arranged for this analysis. Straight black lines correspond to emissions flows following one or more products. This methodology allocates energy and emissions common to both processes on a pseudo-system wide level by energy content and attributes energy and emissions coming from a specific process to the products from

the process. The unallocated cultivation and transportation emissions are first divided between HRJ production and F-T fuel production based on the final energy products (allocation ratio 1); those associated to HRJ production are then allocated between oil and meal based on market value (allocation ratio 2); the HRJ and F-T emissions are then summed back together and allocated based on energy between jet fuel and all other energy products from the system (allocation ratio 3). The other energy products from the system are comprised of F-T diesel, F-T naphtha, hydroprocessed renewable naphtha and mixed propane gas. The allocation ratios and combined facility product slate resulting in the low emissions, baseline and high emissions scenarios are presented in Table 91.

Table 91: Allocation ratios and product slates describing a coupled HRJ and F-T facility processing salicornia oilseeds and straw biomass

	Low	Baseline	High
Product Slate (MJ/MJ$_{Jet}$)			
F-T Diesel	1.2	1.5	1.8
F-T Naphtha	0.30	0.49	0.75
Renewable Naphtha	0.16	0.16	0.16
Mixed Propane Gas	0.012	0.027	0.028
Salicornia Meal	0.52	0.64	0.71
Allocation Ratios			
Allocation Ratio 1	32.1% HRJ 67.9% F-T	30.9% HRJ 69.1% F-T	27.8% HRJ 72.2% F-T
Allocation Ratio 2	66.7% Oil 33.3% Meal	61.5% Oil 38.5% Meal	58.8% Oil 41.2% Meal
Allocation Ratio 3	37.1% Jet Fuel 62.9% Other Fuels	31.9% Jet Fuel 68.1% Other Fuels	26.6% Jet Fuel 73.4% Other Fuels

7.7.6 Land Use Change Emissions from Salicornia Cultivation

The degree to which carbon can be sequestered through salicornia cultivation depends on the initial state of the soil in which they are grown and the penetration of the roots in this soil. Dry land soils are typically low in organic carbon and could conceivably hold greater carbon under restored conditions (Glenn et al., 1993).

Estimates of soil organic carbon, root carbon and charcoal carbon storage of semi-permanent cultures conducted by Sommer et al. (2000) have been previously used to estimate potential carbon sequestration from halophyte cultivation (Hendricks, 2008; Hendricks and Bushnell, 2008). These estimates are for deep soils under small farmer land use systems in the Eastern Amazon region that contain conventional vegetation and crops, not salicornia. They were intended to quantify current carbon stock; hence, using the carbon storage values from Sommer et al. as a sequestration potential implicitly assumes similar soil carbon storage capacity and zero soil carbon prior to any halophyte cultivation. Soil carbon stored for semi-permanent secondary vegetation range in the Eastern Amazon region range from 146 to 167 tC ha^{-1}, where approximately 90% is soil organic carbon, 5% is root carbon and the remaining 5% is charcoal in the soil. Furthermore, although this range applies to the top 6m of soil, an average of 67% of the carbon is stored in the top 1.2 meters (Sommer et al., 2000).

A second independent analysis of carbon sequestration from salicornia was carried out for a joint project between the United States and Mexico. The project is called "Project Salicornia: Halophyte Cultivation in Sonora" and is operated under the United Nation Framework convention for Climate Change. Project developers have a preliminary estimate that the cultivation area would reach a steady state soil carbon content of 49 tC ha^{-1} after 100 years (UNFCCC, 1998). This value happens to correspond to half the carbon stock in the top 1.2 meters of soil as estimated by Sommer et al. (2000), namely 48.7 to 55.7 tC ha^{-1}. The UNFCCC estimate was adopted for this study as a land use change scenario.

Under the optimistic assumption that all of the soil carbon sequestration occurs in the first 30 years, the impact of long-term soil carbon sequestration on life cycle GHG emissions are given in Table 92. The sequestered carbon was amortized over 30 years with no discounting. The values are on the basis of carbon sequestered per MJ of jet fuel created from salicornia oil and straw biomass

Table 92: Long term soil, root and charcoal carbon sequestration from the cultivation of salicornia for fuel production

Long Term Carbon Sequestration (tC ha^{-1})	Long Term Carbon Sequestration (gCO$_2$/MJ)		
	Low	**Baseline**	**High**
49	49.7	41.9	33.9
Note: 1) Since values represent carbon sequestration, larger values correspond to lower emissions.			

Soil carbon sequestration estimates lead to emissions credits ranging from 0.39 to 0.57 times the life cycle GHG emissions from the production and use of conventional jet fuel. Recall that these estimates assume that the carbon content of the soil used for cultivation is depleted prior to cultivation occurring. Were salicornia grown on land with non-depleted carbon stocks then it is unlikely that any long-term soil carbon sequestration would result.

7.7.7 Results

The life cycle GHG emissions resulting from the production and use of HRJ and F-T jet fuel from salicornia seeds and straw biomass are summarized in Table 93. Results incorporating long-term carbon sequestration are given in Table 94.

These results reflect the sum of emissions from HRJ production and F-T jet production for one total megajoule of jet fuel. Based on the input assumption adopted in this work and the allocation methodologies discussed above, the life cycle emissions of the salicornia to HRJ and F-T jet pathway range from 0.35 to 0.76 times those from the production and use of conventional jet fuel. If long-term carbon sequestration occurs because of the salicornia growth, the life cycle GHG emissions range between -0.2 to 0.37 times those from conventional jet fuel.

Table 93: Summary of results from renewable and F-T jet fuel production and use from salicornia

Land Use Change Scenario H0	Low	Baseline	High
Key Assumptions			
Total Biomass Yield (kg/ha/yr)	17614	16247	14880
Seed production (g$_{seed}$/kg$_{total biomass}$)	142	122	101
Seed Yield (kg/ha/yr)	2506	1977	1504
Seed Oil Fraction	33%	28.2%	26%
Life Cycle CO$_2$ Emissions by Stage			
Biomass Credit (gCO$_2$/MJ)	-90.9	-105.3	-116.9
Recovery of feedstock (gCO$_2$/MJ)	26.7	36.8	47.1
Transportation of feedstock (gCO$_2$/MJ)	1.0	1.1	1.2
Processing of feedstock to fuel (gCO$_2$/MJ)	21.6	38.3	51.5
Transportation of jet fuel (gCO$_2$/MJ)	0.5	0.5	0.5
Combustion CO$_2$ (gCO$_2$/MJ)	70.4	70.4	70.4
WTT GHG Emissions by Species			
WTT CO$_2$ emissions (gCO$_2$/MJ)	-41.0	-28.6	-16.5
WTT CH$_4$ emissions (gCO$_2$e/MJ)	1.0	1.3	1.7
WTT N$_2$O emissions (gCO$_2$e/MJ)	0.2	4.6	10.5
Total WTW GHG Emissions (gCO$_2$e/MJ)	30.5	47.7	66.1
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**0.35**	**0.55**	**0.76**

Table 94: Life cycle GHG emissions for production and use of renewable and F-T jet fuel salicornia assuming long-term carbon sequestration

Land Use Change Scenario H1	Low[1]	Baseline[1]	High[1]
Land use change emissions (gCO$_2$/MJ)	-49.7	-41.9	-33.9
WTW CO$_2$ emissions (gCO$_2$/MJ)	-20.3	-0.1	20.0
Total WTW GHG Emissions (gCO$_2$e/MJ)	-19.2	5.8	32.2
Life Cycle GHG Emissions Relative to Baseline Conventional Jet Fuel	**-0.22**	**0.07**	**0.37**
Notes:			
1) All other input assumptions (salicornia cultivation, extraction of oil, processing of oil to HRJ and straw biomass to F-T jet) are based on those in the H0 emissions case of the corresponding scenario.			

Without the opportunity to obtain nutrients from the recycled streams of aquaculture farms, substantial increases in nitrous oxide emissions are likely to occur. An increase in N$_2$O emissions translates to more pronounced consequences from the uncertainty associated with the IPCC correlations. The reader should be aware of the potential for these inherent uncertainties when comparing different pathways for GHG reduction potential in sections 8.1 and 8.2.

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8 Summary of Results

8.1 Life Cycle GHG Emissions

Table 95 summarizes the results of the life cycle GHG emissions for the baseline scenario of the fuel pathways considered in this work. Figure 28 presents the results of the analysis in graphical form. It is important to note that this plot does not show cumulative totals, but rather it displays the contribution of emissions from each step in the fuel life cycle. The impact of the land use change scenarios, which are summarized in Table 96, is included in the form of two pathways for switchgrass fed F-T jet, three pathways for soy oil to HRJ, four pathways for palm oil HRJ, two pathways for rapeseed to HRJ and two pathways for salicornia oil HRJ. These results highlight the need to avoid land usage changes that result in positive GHG emissions. This method of presentation displays the 'biomass credits' that are given to biofuels from the CO_2 absorbed during biomass growth; these credits are largely the reason why these fuels offer the potential for reduced GHG emissions. With the exception of BTL and CBTL, the biofuel pathways all have similar 'biomass credits' and the magnitude of these credits is approximately[63] the magnitude of the combustion emissions. The 'biomass credit' for CBTL is smaller because the fuel is created from a combination of coal and biomass. The 'biomass credit' for BTL is larger because biomass is used to power the entire fuel production process.

The baseline life cycle GHG emissions values in Figure 28 were combined with the low and high emissions scenario values to create Figure 29. The uncertainty bars represent the range of emissions as given by the low and high emissions cases. Many pathways have baseline life cycle GHG emissions that are lower than conventional jet fuel but have the potential to have GHG emissions that are higher than conventional jet fuel. For this reason, it is essential not to simply assume that biofuels are environmentally beneficial without knowing the specifics of how the fuel is produced.

A few of the key results are outlined below:

- *Life cycle GHG emissions are but one of many considerations when evaluating the feasibility and sustainability of an alternative fuel option.*

- *The data do not include all of the feedstock-to-fuel pathways that could be use to create jet fuel. Some interesting options not covered include camelina oil to jet fuel, fuels created from pyrolysis oils and advanced fermentation of sugars to hydrocarbons. These will be addressed as part of the ongoing work and will appear in future revisions to this report.*

- *Of the fuel options considered herein, conventional petroleum has the lowest emissions of any jet fuel pathway that that relies exclusively on fossil fuel resources.*

- *Few biofuels have zero life cycle GHG emissions.*

- *There is considerable variability in the life cycle GHG emissions; emissions from land use change contribute the most to this for the biofuel pathways considered.*

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[63] In some of the high and low emissions scenarios, the 'biomass credit' is not quite equal to the combustion emissions due to variations in the allocation ratios throughout the pathway. Since the biomass credit' is given in the first life cycle stage (feedstock recovery), it is subject to all allocation ratios.

Table 95: Baseline life cycle GHG emissions for all fuel pathways studied. Land use change scenarios are described in Table 96

	Biomass Credit	Recovery	Feedstock Transport	Processing	Fuel Transport	Combustion	WTT N_2O	WTT CH_4	Land Use Change	Total
Crude to conventional jet fuel	0.0	4.2	1.5	5.5	0.8	73.2	0.1	2.3	0.0	87.5
Crude to ULS jet fuel	0.0	4.2	1.5	7.3	0.8	72.9	0.1	2.4	0.0	89.1
Oil sands to jet fuel	0.0	19.7	1.3	5.5	0.5	73.2	0.1	3.0	0.0	103.4
Oil shale to jet fuel	0.0	41.2	0.6	3.3	0.6	73.2	0.2	2.5	0.0	121.5
Natural gas to F-T fuel	0.0	4.6	0.0	20.2	1.2	70.4	0.0	4.6	0.0	101.0
Coal to F-T fuel (no carbon capture)	0.0	0.8	0.1	117.2	0.6	70.4	0.0	5.7	0.0	194.8
Coal to F-T fuel (with carbon capture)	0.0	0.8	0.1	19.4	0.6	70.4	0.0	5.9	0.0	97.2
Switchgrass to F-T fuel (LUC-B0)	-222.7	6.4	0.6	152.1	0.5	70.4	10.3	0.2	0.0	17.7
Switchgrass to F-T fuel (LUC-B1)	-222.7	6.4	0.6	152.1	0.5	70.4	10.3	0.2	-19.8	-2.0
Coal and Switchgrass to F-T fuel with CCS (LUC-B0)	-44.3	1.2	0.2	21.9	0.5	70.4	2.0	4.9	0.0	56.9
Coal and Switchgrass to F-T fuel w/o CCS (LUC-B1)	-44.3	1.2	0.2	21.9	0.5	70.4	2.0	4.9	-3.9	53.0
Soy oil to HRJ (LUC-S0)	-70.5	20.1	1.2	10.3	0.6	70.4	3.6	1.3	0.0	37.0
Soy oil to HRJ (LUC-S1)	-70.5	20.1	1.2	10.3	0.6	70.4	3.6	1.3	60.8	97.8
Soy oil to HRJ (LUC-S2)	-70.5	20.1	1.2	10.3	0.6	70.4	3.6	1.3	527.2	564.2
Palm oils to HRJ (LUC-P0)	-70.5	4.9	3.1	10.3	0.6	70.4	5.1	6.3	0.0	30.1
Palm oils to HRJ (LUC-P1)	-70.5	4.9	3.1	10.3	0.6	70.4	5.1	6.3	9.6	39.8
Palm oils to HRJ (LUC-P2)	-70.5	4.9	3.1	10.3	0.6	70.4	5.1	6.3	135.8	166.0
Palm oils to HRJ (LUC-P3)	-70.5	4.9	3.1	10.3	0.6	70.4	5.1	6.3	667.9	698.0
Rapeseed oil to HRJ (LUC-R0)	-70.5	17.2	3.1	10.3	0.6	70.4	22.4	1.3	0.0	54.9
Rapeseed oil to HRJ (LUC-R1)	-70.5	17.2	3.1	10.3	0.6	70.4	22.4	1.3	43.0	97.9
Jatropha oil to HRJ	-70.5	16.7	1.5	10.3	0.6	70.4	9.1	1.2	0.0	39.4
Algae oil to HRJ	-70.5	29.6	0.3	10.3	0.6	70.4	8.1	1.8	0.0	50.7
Salicornia to HRJ and F-T Fuel (LUC-H0)	-105.3	36.8	1.1	38.3	0.5	70.4	4.6	1.3	0.0	47.7
Salicornia to HRJ and F-T Fuel (LUC-H1)	-105.3	36.8	1.1	38.3	0.5	70.4	4.6	1.3	-41.9	5.8

Table 96: Land use change scenarios considered in this work

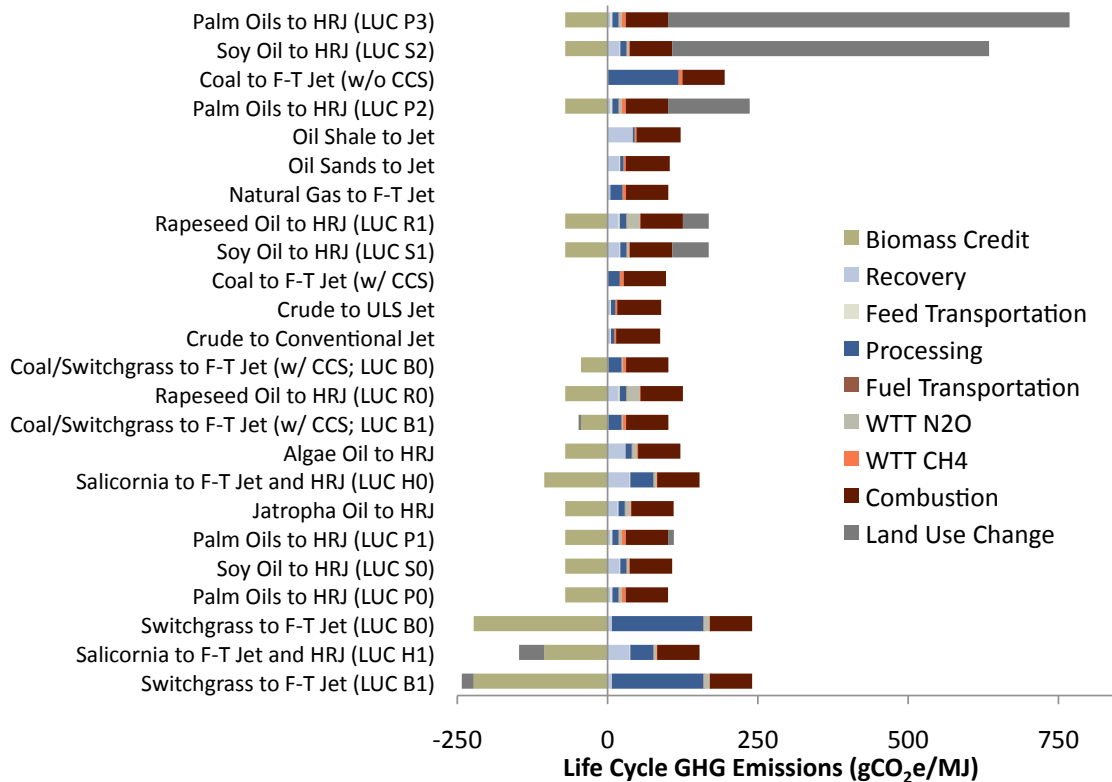| Land use change	Scenario 0	Scenario 1	Scenario 2	Scenario 3
Switchgrass	None	Carbon depleted soils converted to switchgrass cultivation	n/a	n/a
Soy oil	None	Grassland conversion to soybean field	Tropical rainforest conversion to soybean field	n/a
Palm oil	None	Logged over forest conversion to palm plantation field	Tropical rainforest conversion to palm plantation field	Peat land rainforest conversion to palm plantation field
Rapeseed oil	None	Set-aside land converted to rapeseed cultivation	n/a	n/a
Salicornia	None	Desert land converted to salicornia cultivation field	n/a	n/a

Figure 28: Life cycle GHG emissions for the alternative jet fuel pathways under consideration. Note: CCS denotes carbon capture and sequestration and the land use change (LUC) scenarios are defined in Table 96
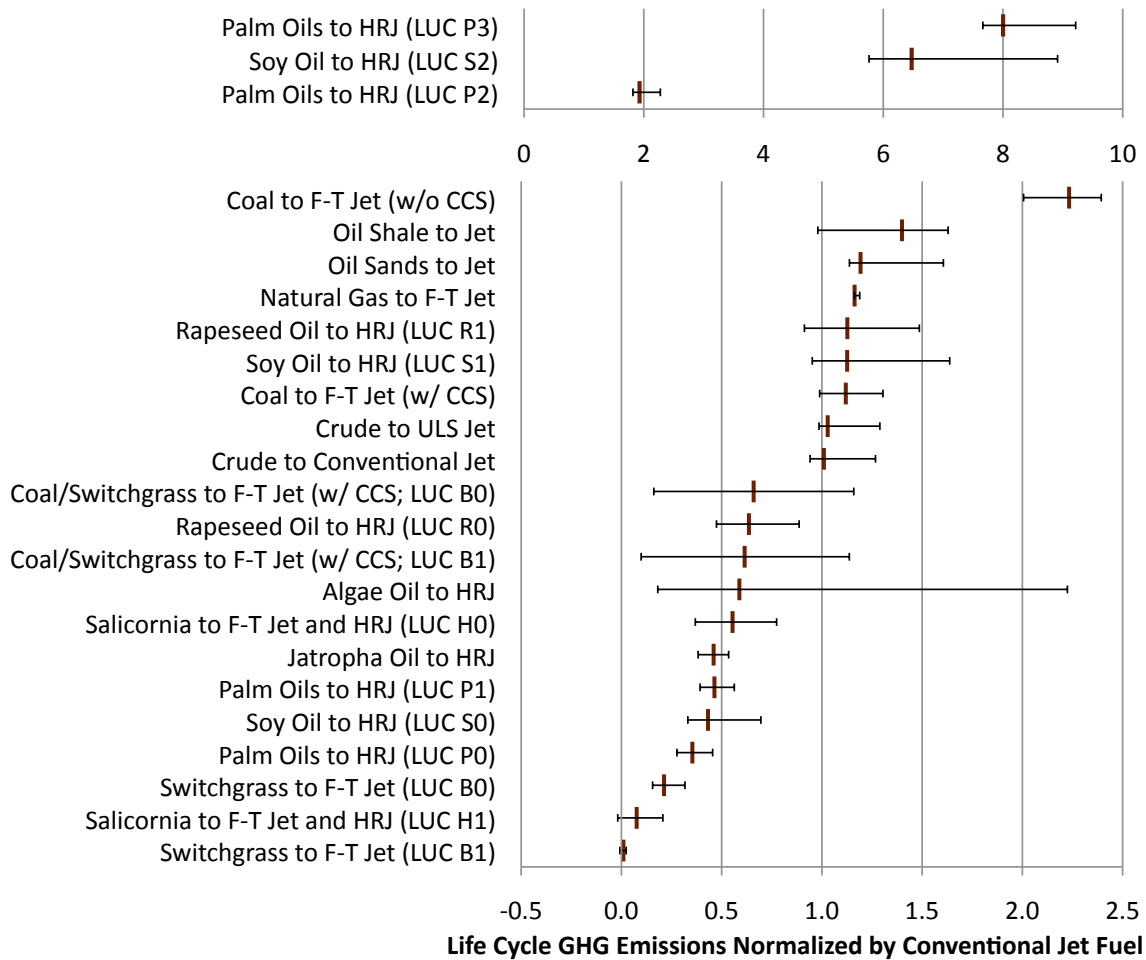
Figure 29: Life cycle GHG emissions for the alternative jet fuel pathways under consideration. Uncertainty bars represent the low emissions, baseline, and high emissions scenarios. Please note the different scales for the top and bottom portions of the figure. Note: CCS denotes Carbon Capture and Storage and Land Use Change (LUC) scenarios are defined in Table 96

8.2 Life Cycle GHG Intensity

As was discussed in Section 3.6, the life cycle GHG emissions from Table 95 as well as Figures 28 and 29 can be combined with an aircraft energy efficiency to yield a metric of life cycle GHG intensity. With a Payload Fuel Energy Intensity (PFEI) of 0.015 MJ / kg-km, the US commercial fleet had a life cycle GHG intensity of 1.3 g CO_{2e}/kg-km in 2007.[64] Because SPK fuels have increased specific energy (energy per unit mass) relative to conventional jet fuel, the fleet wide PFEI will decrease by 0.3% (Hileman et al., 2010). Despite the decrease in energy usage, there would be a 3% increase in fuel volume relative to conventional jet fuel. The change in energy with the use of SPK fuels is sufficiently small relative to the change in life cycle GHG emissions that have been discussed in this document that a life cycle GHG intensity value could be obtained for any of the fuels considered in this document by multiplying the life cycle GHG emissions by the PFEI. Other fuel efficiency metrics could similarly be converted to life cycle GHG metrics using the same method.

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[64] This value was calculated based on calculations performed by Hileman et al. (2008) using the mean energy content of JP-8 and 2007 data from the Bureau of Transportation Statistics (1.17 x 10^{11} revenue ton miles flown and 1.99 x 10^{10} gallons of jet fuel consumed, http://www.transtats.bts.gov/).

8.3 Biofuel Yield

The focus of this work was to establish life cycle GHG emissions inventories for a variety of alternative jet fuels and to determine the scales at which these fuels would need to be implemented in order to achieve emissions reduction targets. The former is covered in the results from Section 8.1 while the latter requires additional analysis of biofuel yield.

Figure 30 summarizes the fuel production potential for all of the biofuel pathways presented in Figures 28 and 29. The range in yields of fuel per kilogram of oilseed feedstock arises from the both the crop yield per acre of land and the variation in oil fraction. Rapeseed seed yields the most oil per kilogram (44%) followed by jatropha seeds (35%), algae (25%), palm kernels (22%) and finally soybeans (18%). Recall that there can be considerable variability in biomass oil yields, (e.g., potential algal yields up to 390% of those used in this work have been quoted by stakeholders in the industry). The fuel production from salicornia is the result of the combined jet fuel production from HRJ and BTL facilities. When configured in this manner, each megajoule of jet fuel is composed of 65% F-T jet and 35% HRJ.
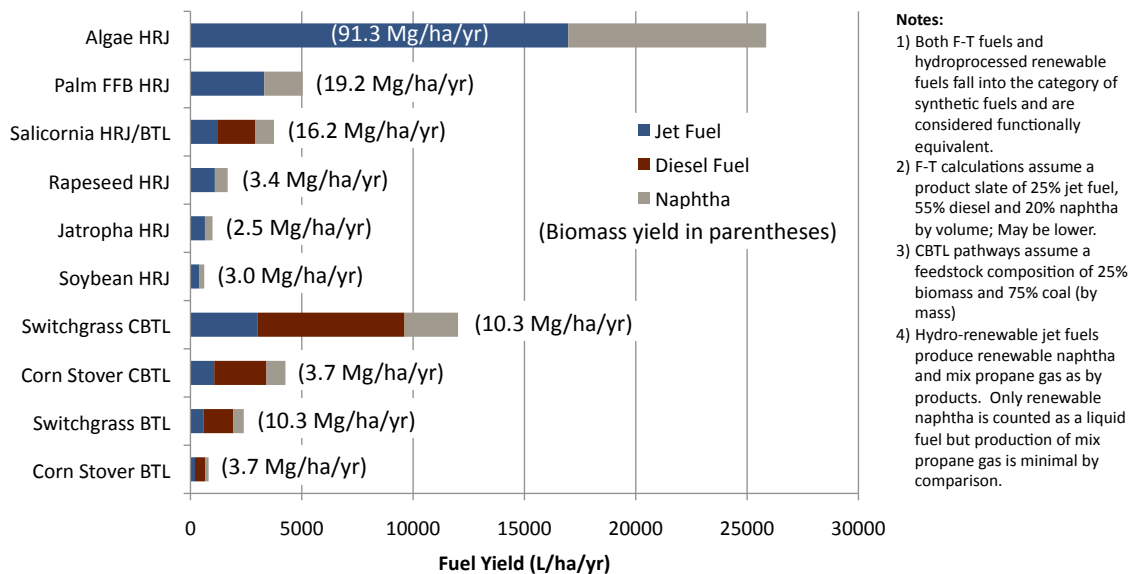

Figure 30: Fuel production potential for various alternative jet fuels that could be derived from biomass. This is not an all-encompassing list of alternative jet fuel options; it merely represents those examined by the authors as part of their ongoing research

A subtle but important point surrounding the F-T jet fuel results is that only 25% of the fuel output from the F-T fuel facility was assumed to be jet fuel. Using switchgrass CBTL as an example, there would be 12,000 (3000 × 4) liters of liquid hydrocarbons produced for every hectare of switchgrass, only 3000 liters of which would be jet fuel (the rest could be fuels such as diesel, gasoline, and naphtha). All 12000 liters of fuel produced carry an environmental benefit; therefore, all liquid fuel products must be considered to capture the total CO_2 mitigation. The switchgrass (or other biomass) used as feedstock for pure BTL plants have very low energy densities. This causes a large quantity to be required in order to make a relatively small quantity of jet fuel (which has a high energy density). Although the same feedstock could be used for both pure BTL and CBTL, when supplemented by coal, the same fuel output can be obtained from a smaller quantity of biomass. This increase is only due to supplementing the biomass feedstock with coal and not to an increase in biomass usage efficiency. Corn stover or forest residue can also be used as a feedstock while maintaining life cycle GHG emissions within 9% of the value found using switchgrass.

The life cycle GHG emissions from Figure 29 and production potentials from Figure 30 can be combined to select fuel pathways that hold the most potential for reducing aviation's GHG emissions. This combination is needed to reduce aviation's GHG emissions because fuel pathways having both low life cycle GHG emissions as well as large fuel production potential are required. A graphical representation of the land requirements to supply the entire 2009 US jet fuel market with 100% SPK and a 50/50 blend of SPK and conventional jet fuel is shown in Figure 31.[65] Three representative fuel yields were chosen to span the relevant range from Figure 30.
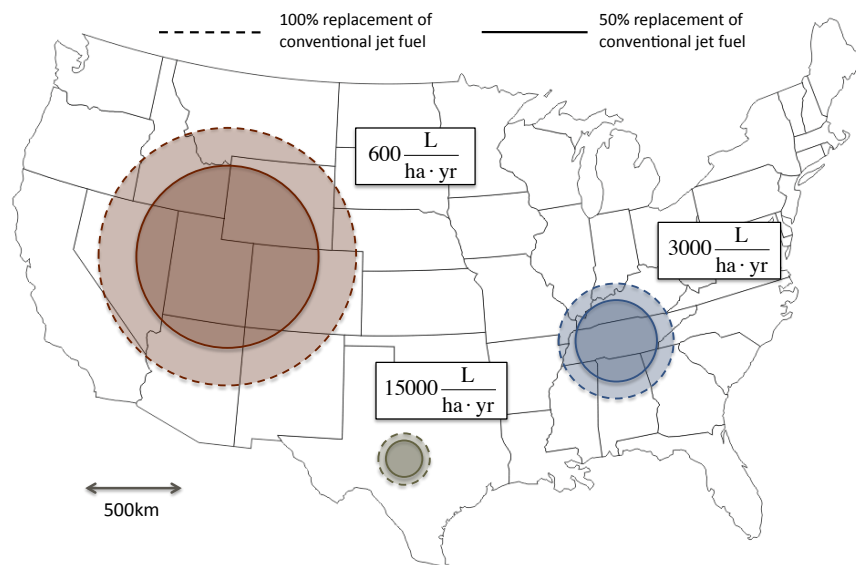

Figure 31: Land area requirements to replace conventional jet fuel use within the US with 100% SPK and 50/50 blend of SPK with conventional jet fuel. Average US conventional jet fuel consumption in 2009 is 1.4 million bbl/day

Fossil-to-jet fuel pathways have large production potential, but they have comparable or higher emissions than conventional jet fuel; therefore, their use will not reduce GHG emissions. BTL fuels have low GHG emissions, but they also have limited fuel production potential due to the large capital costs for F-T production facilities. With the use of excess rapeseed, palm or soy (available after food needs are met) for HRJ production, rapeseed to HRJ, soy to HRJ and palm to HRJ have low life cycle GHG emissions; however, there is little excess available and new cropland is required for additional production. Current global production of soy, palm and rapeseed oil translate to only 34%, 43% and 18% of US jet fuel demand, respectively (FAPRI, 2009). As such, expanded production of soy oil and palm oil for large-scale HRJ production could result in significant GHG emissions from land use change. Because of its toxic characteristics and low yield, jatropha is likely limited to small regional applications making it inappropriate to replace considerable quantities of conventional jet fuel. Hence, BTL fuels as well as HRJ fuels from soy, palm, and jatropha have limited potential for reducing GHG emissions. The production potential of CBTL is largely dependent on the biomass weight fraction of the feedstock. Switchgrass was found to yield three times more volume of jet per hectare than corn stover when fed with a weight fraction of 25%. Salicornia holds promise if it is used to make both HRJ and BTL (or CBTL) fuels as it could reduce life cycle GHG emissions by 25% compared to conventional jet fuel and has a production potential equivalent to one third of palm. The most obvious opportunity for large-scale production is using algae to make HRJ, which explains the recent wave of investments in the algae industry.

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[65] Average US consumption of jet fuel thus far in 2009 has been roughly 1.4 million barrels per day (EIA, 2009b)

8.4 Biofuel Impact on Domestic Water Resources

Water plays an essential role in developing and utilizing energy resources. It is used in energy-resource extraction, refining and processing, and transportation. The dependence of biofuels on water extends even further to include water used for feedstock growth. This work has thus far only focused on life cycle GHG emissions and yield as the bounding factors to large-scale biofuel production. Water requirements and regional availability serve as a third bound to the trade space of biofuel development.

It is important to distinguish between water withdrawal and water consumption. Water withdrawal involves withdrawing water from its source and subsequently returning it after use. The quality of the returned water may not be the same as when it was originally removed. Electricity generation is the best example where water use is dominated by withdrawn water used for cooling (some cooling water is lost to evaporation and this fraction is considered consumed). Water consumption involves withdrawing water from its source and not returning it after use. Crop irrigation represents the largest fraction of water consumption within the US. Figure 32 shows water withdrawals and water consumption by sector for the US in 2000 and 1995 respectively. Notice that the volume of consumed water is only 30% of that withdrawn.
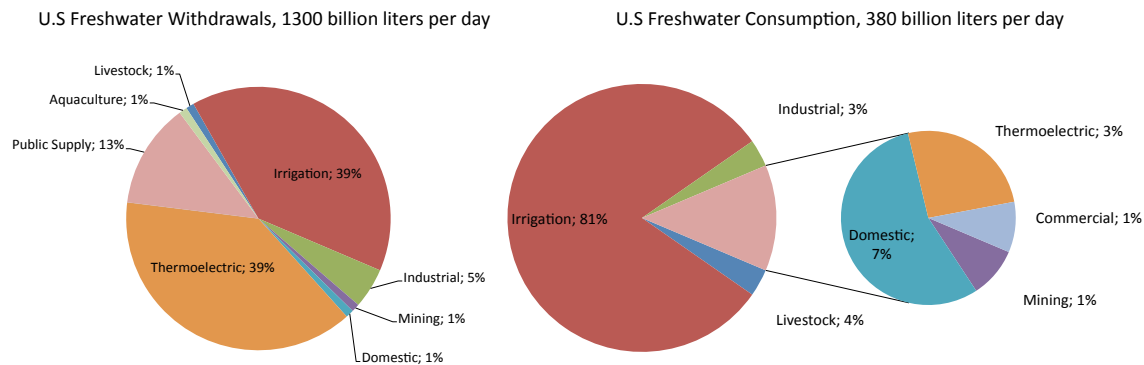

Figure 32: Water consumption and water withdrawals in the US by sector (data adapted from DOE, 2006)

The water required to refine one liter of petroleum fuels was found to be between 1 and 2.5 liters (King and Webber, 2008). Jet fuel consumption in the US is currently at 1.4 million barrels per day, or 222 million liters per day (EIA, 2009b). This means that the aviation industry alone is responsible for consuming between 222 and 556 million liters of water each day. Most alternative fuels will require at least the quantity of water that refineries currently consume and have the potential to consume several orders of magnitude more (DOE, 2006). This fact leads us to define the life cycle water consumption in fuel production using a similar metric to that used to quantify life cycle GHG emissions. The units of liters consumed per MJ of fuel energy delivered to the tank are used to encompass the water consumed during the recovery and transportation of the feedstock from the well, field, or mine to the production facility, processing of these materials into fuels, transportation and distribution of the fuel to the aircraft tank, and combustion of the fuel in the aircraft. Without considering any specific examples, water consumption from feedstock recovery and processing will dominate the life cycle water consumption while water consumption from feedstock and fuel transportation will be indirect through consuming fuels that required water for their production. Water consumption from the combustion stage is zero.

The primary focus of this work was on life cycle GHG emissions; hence no new data has been generated concerning water usage within fuel production pathways. While this is an active part of our ongoing research, this work makes use of a study concerning energy demand on water resources from the US Department of Energy (2006) and a study on the water intensity of transportation from King and Webber (2008). The relevant data from both papers is presented in Figure 33. Notice that corn and soybeans requiring irrigation consume 3 orders of magnitude more water than refining of conventional petroleum based fuels. The irrigation needed for crop

production varies greatly depending on the region. For example, water use for irrigated soy production varies from 600,000 liters per hectare for Pennsylvania to about 4.3 million liters per hectare for Colorado, with a national average of 2.4 million liters per hectare (DOE, 2006). The overall average is deceiving because many areas use no irrigation while others have much higher demands.

Few new reservoirs have been built since 1980, and fresh surface-water withdrawals have peaked at about just over 1 trillion liters per day. Many regions depend on groundwater to meet increasing water demands, but declining groundwater tables could severely limit future water availability. Some regions have seen groundwater levels drop as much as 300 to 900 feet over the past 50 years from pumping water from aquifers at a rate faster than the natural rate of recharge (DOE, 2006). The expansion of biofuel production could exacerbate non-uniform regional water stresses between states; hence, the impact of biofuels on water consumption will be regionally dependent and cannot be summarized by a single value.

If feedstocks can be found that require no additional water above the status quo, their processing to jet fuel will still place additional strain on the current system. Converting natural gas and coal to F-T diesel are 5 and 7 times more water intensive than refining of conventional petroleum, respectively. Similarly, water consumption from steam reforming of natural gas to make hydrogen for hydroprocessing renewable oils to HRJ will be at least comparable to refining conventional petroleum.
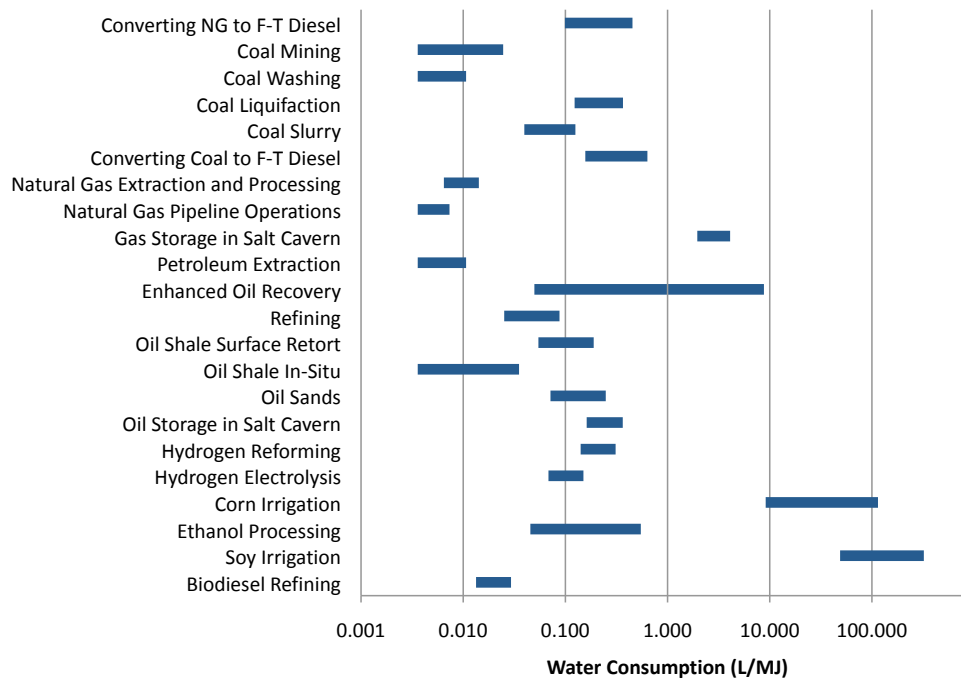
Figure 33: Water consumption for the production of various fuels (data adapted from DOE, 2006 and King and Webber, 2008)

Depending on the water quality needs for particular applications, freshwater supplies can be augmented with degraded or brackish water. This applies to the algae and salicornia pathways discussed in sections 7.6 and 7.7. Algae grown in open ponds have the additional complication of evaporation from the pond surface. Algae are most effectively grown in shallow, high surface area ponds. Assuming an evaporative rate of 1.5 cm/day (Weissman and Goebel, 1987), 100 liters of water per hectare need to be replenished each minute. This constraint will limit the location of algae farms to being adjacent to a plentiful source of degraded or brackish water.

As previously discussed, biofuel production would aggravate regional strains on freshwater

supply and local infrastructure. Further insight as to where these strains will be felt can be gained by considering the data from DOE (2006) in conjunction with the regions within the US where biofuel feedstock cultivation expansion is likely. In their 2006 report on the interdependency of energy and water, the US Department of Energy (DOE) showed that freshwater withdrawal in most regions of the US exceeds the available precipitation (precipitation minus evapotranspiration[66]). While the ratio of freshwater withdrawals to available precipitation is less than 30 for most of the northeast, midwest and southeast, values between 100 and 500 are not uncommon for western states. The magnitude of this ratio indicates the degree to which current water demands are being met with groundwater pumping or transport of surface water from other regions (DOE, 2006).

Locations most conducive to next generation biofuel feedstock cultivation are the Southwest and Southeast for algae, Southwest for salicornia, and Midwest and Southeast for switchgrass. With the exception of Midwest and part of the southeast, these are regions where groundwater withdrawal is well in excess of the available precipitation each year. The southwest should receive particular attention when discussing the expansion of domestically produced biofuels because of its large water constraints.

These data demonstrate that water availability could be a limiting factor for biofuel production in certain regions of the US. Water usage is part of this ongoing work and will be addressed further in subsequent reports.

8.5 Invasive Species with Respect to Biofuels

Crops ideal for large-scale biofuel production are those with high yield that do not require fresh water irrigation and can maintain high yields when grown on marginal lands. These are also general traits that describe invasive species. More specifically, the National Invasive Species Information Center (NISIC) defines the following characteristics as some of the traits of species likely to be invaders (NISIC, 2006).

- Rapid growth and short life cycle: go from seed to producing seed very rapidly – sometimes within a few weeks.
- Able to grow in a wide range of habitats
- High number of seeds produced
- Long seed dormancy and staggered germination
- Efficient method of seed dispersal
- Benefit from allelopathy, which is the release of chemicals into the surrounding soil that prohibit the growth of other plants.

As defined by most government organizations, invasive species are non-indigenous and adversely affect the habitats they invade economically, environmentally or ecologically. There are approximately 50,000 non-indigenous species in the United States. Some of these are noninvasive and beneficial to society, such as corn, wheat and rice. Others have caused major economic losses in agriculture and forestry and have irreversibly damaged the ecosystems to which they were introduced (Pimentel et al., 2000).

Historically most plant introductions, with the exception of agricultural weeds, have been intentional while microbe introductions have been accidental. Regardless of initial rationale, an additional 700,000 hectares of US wildlife habitat are invaded each year by non-indigenous weeds. As an example, 4 million hectares of grassland in northern California have been lost to yellow star thistle (Campbell, 1994). These are lands that may otherwise have been used for energy crop cultivation such as switchgrass. Invaded lands can also become predisposed to fires. Cheatgrass growth in the Great Basin in Idaho has increased the frequency of fires by more than an order of magnitude (Whisenant, 1990).

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[66] Describes the sum of evaporation and plant transpiration from the Earth's land surface to the atmosphere.

The accidental introduction of weeds and microbes often occurs via crop seeds and other parts of host plants. The agriculture industry suffers an overall reduction of 12% in crop yields due to weeds with each 1% decrease in crop yield being accompanied by a 4.5% increase in crop cost to the farmer. Ecologists agree that non-indigenous weeds are a greater risk than native ones. Plant pathogens carried by foreign microbes result in crop losses of approximately 65% those of weeds. Similarly, pathogens of forest plants cause the loss of 9% of forest products each year (Pimentel et al., 1997).

The majority of the aforementioned impacts occurred because of intentional introduction of foreign plants to the North American ecosystem. Many of the feedstocks considered in this work have never before been grown at large scale and/or require deliberate introduction to an ecosystem. The financial burden of losses and control due to invasive plants and microbes has been estimated as several tens of billions of dollars each year (Pimentel et al., 2000). While the authors are not attempting to imply that the expansion of biofuels could result in such significant economic losses, it is important to understand that controlling invasive plants is not a negligible consideration in the decision making process. The awareness that invasive species can be a direct financial burden highlights the potentially negative economic consequences of introducing new species to the environment. Maintaining environmentally sustainable feedstock production that meets food and energy demands is essential to the economic success of meeting large-scale biofuel demands.

The expansion of biofuel consumption within the US will require a significant increase in crop and feedstock production and this presents the possibility that non-indigenous species will be introduced into an unprepared ecosystem. Ironically, the impact of these invasives could be to inhibit crop production, which could hurt the industry that was responsible for their introduction.

9 Conclusions

As part of continuing research on alternative jet fuels, a screening level life cycle assessment of a wide range of potential drop-in alternative jet fuels was conducted. Three scenarios were developed for each pathway corresponding to optimistic, nominal and pessimistic assumptions regarding specific process inputs and production characteristics. In most fuel pathways, the choice of allocation methodology and potential for GHG emissions from land use change were found to have the largest impact on the results.

Consistent allocation methodologies were implemented across multiple pathways to facilitate equitable comparisons of different alterative fuels. In all analyses, the use of displacement or system expansion was minimized to reduce the variability of the results to subjective assumptions. The displacement method was only implemented where assumptions could be made that minimize the impact of allocation on the result (e.g. algae electricity). Energy allocation was used for any process that resulted in a product slate of liquid hydrocarbons (e.g., jet, diesel and naphtha from F-T synthesis). Market value allocation was used for processes resulting in a combination of oil and meal (e.g. separation of soybeans into soy oil and soy meal). In all such cases, the oil price was assumed equal to that of soy oil and the price of meal assumed relative to soy meal based on relative protein content. The use of mass or volume allocation was avoided.

The treatment of land use change emissions (both positive and negative) in this work was developed to provide an understanding for the reader of how land use change emissions compare with the emissions from the other five life cycle stages. It was not intended to explicitly quantify the specific land use change emissions that would result from expanded production of any given feedstock. The scope of this work was limited to only quantifying the impacts of direct land use change; emissions from indirect land use change were not considered. Multiple scenarios were used to explore the range of magnitudes of GHG emissions due to land use change; these included a scenario where no land use change emissions were incurred. This approach allowed the impacts of different land use change scenarios to be isolated from the other emissions from fuel production.

As part of ongoing research, a more complete assessment of land use change emissions that includes indirect effects is being developed. Proper evaluation of the indirect effects of alternative fuels within aviation requires modeling of the demand for renewable energy resources within the transportation sector, including aviation, as well as the demand for renewable energy resources from the energy sector as a whole. Most indirect effects are expected to occur on an international scale; hence, domestic analyses, such as those in this work, need to be done in the context of the global market.

Given their reduced life cycle GHG emissions relative to conventional jet fuel, some alternative fuels could play a central role in mitigating aviation's contribution to climate change, including helping aviation to achieve carbon-neutral growth when combined with improved technology and operations that are more efficient. If appropriate renewable feedstocks were used, both Fischer-Tropsch (F-T) fuels and Hydroprocessed Renewable Jet (HRJ) fuel could provide aviation with modest (~10%) to large (~50%) reductions in emissions that contribute to global climate change. If projections of soil carbon sequestration prove valid, a salicornia-based biofuel could have a 100% reduction in life cycle GHG emissions.

Aviation is not the only potential user of renewable biomass resources, and it will have to compete for these limited resources. Furthermore, large land area requirements indicate that it is unlikely that a single region could create sufficient biomass to supply the entire planet with biofuels. Hence, it is probable that large-scale implementation of biofuels would arise as a superposition of regionally appropriate feedstocks. It is critical to continue to examine feedstocks that could be used to create transportation fuels, such as jet fuel, without the need for arable land, with a minimum of fresh water, and with large yields per hectare.

Although this analysis has identified several shortcomings that could prevent biofuels from being a complete environmental solution for the aviation industry, they could still be an important part of the aviation industry's strategy for reducing life cycle GHG emissions. Current actions with regard to biofuel expansions are important in realizing the potential of this industry. Not all feedstocks need to have the potential to displace large volumes of petroleum fuel. Any feedstock produced today can lead to valuable experience through benefitting the local economy and providing essential lessons in production and processing techniques. This experience would be invaluable should a higher yield crop, such as algae, become commercially viable.

The most significant challenge is not in developing viable alternative fuels that could reduce aviation's GHG emissions -- the technology exists; rather, the challenge lies in developing and commercializing the large scale production of next generation of biomass feedstocks that could be grown in a sustainable manner.

Appendix A – General Feedstock and Fuel Properties

The properties of the main feedstocks and fuels used in this analysis are given in Table 97. In most cases, these properties represent only a single value within the typical range of each characteristic. While these data are appropriate for the level of detail of this work, sample specific data should be used (where available) in conducting analyses of individual production configurations.

Table 97: Feedstock and Fuel Properties

Feedstock or Fuel	LHV (MJ/kg)	Density (kg/L)	Carbon content (wt %)	Sulfur content (wt ppm)	Source(s)
Crude oil or syncrude	41.3[1]	0.878[2]	84.6[1]	16,900[2]	EIA, 2008a
Conventional Jet A	43.2	0.802	86.2	600	Hileman et al., 2010
ULS Jet A	43.3	0.791	85.9	15	Hileman et al., 2010
F-T Jet Fuel / HRJ	44.1	0.76	84.7	0	Hileman et al., 2010
F-T Diesel / HRD	44.0	0.78	84.9	0	Norton et al., 1998; GREET, 2008
F-T Naphtha / HR Naphtha	44.4	0.70	84.2	0	GREET, 2008
Coal (US average)[4]	22.7	-	59.0	11,100	GREET 2007; EIA, 2006
Bituminous coal[4]	26.4	-	64.8	29,400	SSEB, 2006
Sub-bituminous coal[4]	18.4	-	49.2	3,500	SSEB, 2006
Petroleum coke[5]	33.2	-	92.3	68,000	EIA, 2006; GREET, 2008
Biomass (forest residue)[6]	15.4	-	51.7	0	GREET, 2008
Biomass (corn stover)[6]	16.3	-	44.5	0	GREET, 2008
Biomass (switchgrass)[6]	17.6	-	47.0	900	GREET, 2008
Natural gas	47.1	0.00078	72.4	6	GREET, 2007
Hydrogen	120	0.00009	0	0	GREET, 2007

Notes:
1) Energy content of crude oil assumed to be 5.8 million Btu per barrel (HHV); carbon content calculated from formula: percent carbon = 76.99 + (10.19×Specific Gravity) + (-0.76×Sulfur Content). (EIA, 1999)
2) Density and sulfur content derived using historical data (1995-2006) provided in EIA, 2008a
3) As source of process energy (e.g. electricity generation). LHV and sulfur content from GREET, 2007; carbon content derived from coal HHV and US average coal carbon emission factor of 26.0 million metric tons per quadrillion Btu for the electric power sector in 2004. (EIA, 2006)
4) For Coal-To-Liquids (CTL) process.
5) Used as a source of process energy in the refining of jet fuel. LHV and carbon content from EIA, 2006, sulfur content from GREET, 2008.
6) For Biomass-To-Liquids (BTL) process.

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