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Peer-reviewed author version

Suresh, Pooja; MALINA, Robert; Staples, Mark D.; LIZIN, Sebastien; OLCAY, Hakan; Blazy, Damian; Pearlson, Matthew N. & Barrett, Steven R. H. (2018) Life cycle greenhouse gas emissions and costs of production of diesel and jet fuel from municipal solid waste. In: ENVIRONMENTAL SCIENCE & TECHNOLOGY, 52 (21), p. 12055-12065.

DOI: 10.1021/acs.est.7b04277
Handle: http://hdl.handle.net/1942/27603

Life cycle greenhouse gas emissions and costs of production of diesel and jet fuel from municipal solid waste

ACCEPTED FOR PUBLICATION IN ENVIRONMENTAL SCIENCE AND TECHNOLOGY

October 6 2018

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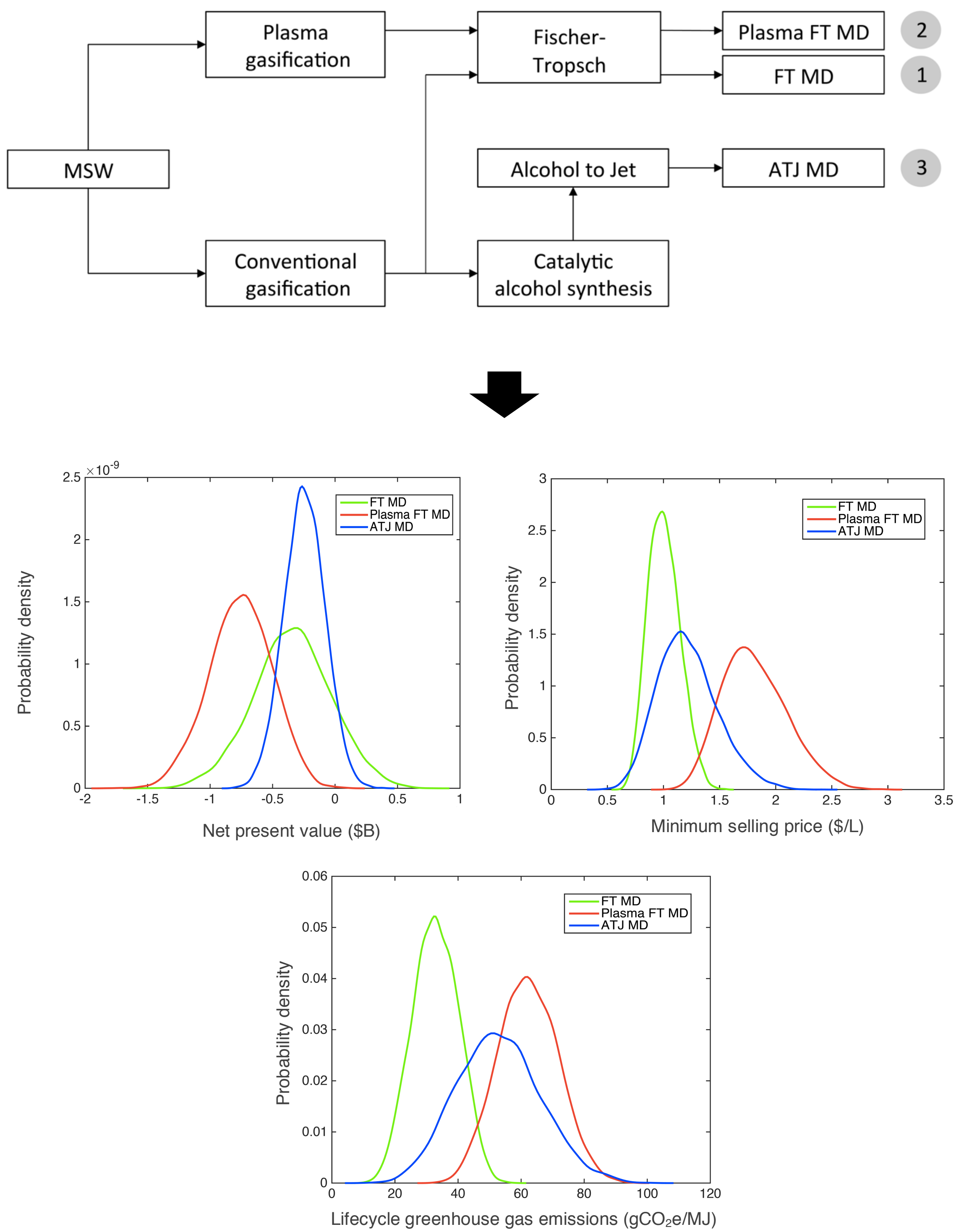
KEYWORDS

Biofuels; Aviation; Waste management; LCA; TEA; Uncertainty analysis

ABSTRACT

This paper quantifies and compares the life cycle GHG emissions and costs of production of diesel and jet fuel derived from municipal solid waste (MSW) in the United States via three thermochemical conversion pathways: conventional gasification and Fischer-Tropsch (FT middle distillate, MD), plasma gasification and Fischer-Tropsch (Plasma FT MD) and, conventional gasification, catalytic alcohol synthesis and alcohol-to-jet upgrading (ATJ MD). We use expanded system boundaries to capture the change in existing MSW use and disposal, and account for parameter uncertainty with Monte Carlo simulations. We estimate median life cycle GHG emissions of 32.9, 62.3 and 52.7 gCO_2e/MJ for FT, Plasma FT and ATJ MD fuels, respectively, compared to a baseline of 90 gCO_2e/MJ for conventional MD fuels. Median minimum selling prices are estimated at 0.99, 1.78 and 1.20 $ per litre with the probability of achieving a positive net present value of fuel production at market prices of 14%, 0.1% and 7% for FT, Plasma FT and ATJ MD fuels, respectively. If the societal perspective rather than an investor's perspective is evaluated the probability of positive net present value of fuel production increases to 93%, 67% and 92.5% for the FT, Plasma FT, and ATJ MD fuels, respectively

ABSTRACT ART



1 Introduction

Transportation accounted for approximately 27% of total greenhouse gas (GHG) emissions in the United States (US) in 2015,[1] with fossil fuels constituting over 95% of the sector's primary energy consumption.[2] Alternative fuels offer the potential to reduce GHG emissions from transportation compared to petroleum-derived fuels. Diesel and jet fuel make up approximately 35% of US transportation energy consumption, which is projected to increase to 44% in 2040,[3] and federal agencies in the US have put in place mandates and goals specific to alternative diesel and jet fuel (e.g. [4-5]).

Contrary to traditional crop-based feedstocks, waste-based feedstocks for alternative fuel production do not require additional land and do not directly compete with food production. Municipal solid waste (MSW), in particular, could offer a significant environmental advantage because the conversion of MSW to fuels would not only displace petroleum-derived fuels, but also avoid the GHG emissions associated with existing waste management strategies. In 2013, 34% of the total generated MSW in the US was recycled and composted, while the remaining 150 million tonnes (metric tons) of MSW, comprising food waste, residential rubbish and commercial waste, were discarded.[6] 80% of the discards were transferred to landfills,[6] where waste of biogenic origin releases anthropogenic methane, thereby making landfills the third-largest anthropogenic source of methane emissions in the US.[7]

Finally, in contrast to fuels from other waste streams such as waste fats and greases,[8] MSW derived fuels could replace relatively large shares of petroleum-derived MD fuel supply, as the energy content of the US national MSW discards in 2013 was equivalent to approximately 70% of same year US jet fuel consumption and 20% of the same year US transportation demand for all middle distillate fuels.[2,6,9]

This analysis evaluates three thermochemical pathways that convert MSW to middle distillate (MD), i.e. diesel and jet, fuel: conventional gasification and Fischer-Tropsch (FT MD), plasma gasification and Fischer-Tropsch (Plasma FT MD) and, conventional gasification, catalytic alcohol (ethanol) synthesis and alcohol-to-jet upgrading (ATJ MD). Figure S1 in the Supplemental Information (SI) illustrates the major technologies in each of the conversion pathways. These technologies are suited to the heterogeneity of MSW and have attracted commercial interest in recent years. They not only produce MD fuels but also other products that include naphtha/gasoline, higher alcohols (propanol and butanol), as well as electricity. The product slates vary based on the technology and the data source, as indicated in the SI.

Despite the potential advantages and commercial interest in MSW MD fuels, only two peer-reviewed studies have assessed components of environmental and/or economic feasibility for a limited number of pathways.[10-11] This is the first peer-reviewed analysis to quantify life cycle GHG emissions and costs of production for the MSW to Plasma FT MD and the MSW to ATJ MD pathways. Life cycle GHG emissions are reported in gCO_2e/MJ_{MD}, minimum selling price in $\$/L_{MD}$, and net present value in billion US dollars ($B) are the three impact assessment metrics utilized throughout this paper. Furthermore, in contrast to previous studies, this study expands the system boundary to capture the change in the overall MSW life cycle that results from using MSW to produce MD fuels rather than its existing use. Therefore, we account for changes in GHG emissions and costs associated with the replacement of existing waste management strategies, additional recycling, end-of-life combustion and co-products. Additionally, we quantify the uncertainty associated with the environmental and economic evaluation of these pathways through Monte Carlo Simulations and present results as probability distributions.

2 Methods

2.1 System boundary and functional unit

The analysis quantifies the change in GHG emissions and costs resulting from diverting MSW away from the prevailing waste management strategy to alternative MD fuel production.[12,13] In this respect, it differs from other biomass-to-fuel LCA studies. MSW has a pre-existing life cycle that is altered when it is used as fuel feedstock, whereas in the case of crop-based biomass, additional feedstock is cultivated for the purpose of fuel production. Therefore, this analysis excludes processes that occur irrespective of the waste management such as collection and existing sorting for recycling and composting. The system boundary does include the effects of eliminating waste management processes such as landfill, and accounts for the impacts of MSW conversion to MD fuels and their end-use.

The system boundary is shown in Figure 1. The system boundary is set where the MSW discards exit the sorting facility. Approximately 66% of the MSW generated in the US is discarded after the recycling and compost streams are separated out,[6] and only these discards are included in this analysis. The composition of the discarded MSW is described in the SI under 3.1. The system boundary also includes the impact of displacing the existing management strategy for discards, which is a combination of landfill (approximately 80% of the discards) and incineration (the remainder) in the US.[6] The model accounts for transport of the feedstock to the fuel production plant. At the plant, further classification is required to adjust the feedstock composition in order to prevent contamination of equipment.[11] Non-combustibles such as metals, glass and other inorganics are sorted out, resulting in a higher heating value feedstock, which is then sized to meet the requirements of the gasifier. The pre-processing system is based on the Refuse Derived Fuel

(RDF) facility model presented by Jones *et al*.[14] Assuming an average sorting efficiency of 90%,[10] approximately 15% of the MSW feed is separated out during the pre-processing.

The recyclable scrap metals and glass removed from the feedstock are sold for recycling, and the rejects are sent to landfill. The GHG emissions impacts, revenues and costs associated with material recovery and disposal of rejects are included within the system boundary. The pre-processed feedstock is then directed to the fuel production process. The material, energy and carbon balances for the conversion technologies are incorporated into the model to account for inputs such as utilities and catalysts, as well as the output fuels and co-products such as excess electricity, higher alcohols, sulfur and slag. The electricity generated by the plant is used to satisfy its own utility requirements, and any excess electricity is considered a co-product that can be sold to the grid. Slag is sold as construction aggregates.[15] Waste streams generated by the fuel production process, such as spent catalysts and ash, are disposed in landfills, which are included in the system boundary. Transportation and distribution, and combustion, of the finished MD product make up the last stages of the fuel life cycle.

The functional units of this analysis are one megajoule (based on lower heating value) of middle distillate fuel for the GHG emissions lifecycle analysis, and one liter of middle distillate fuel for the economic analysis. Functional units were chosen in order to allow for comparison of results between pathways analyzed in this manuscript, and with existing studies on other middle distillate fuels. The lifecycle GHG emission analysis, techno-economic analysis and uncertainty assessment were performed using MATLAB.

2.2 Conversion technology

We analyze three thermochemical pathways to convert MSW to MD fuels: FT MD, plasma Plasma FT MD and ATJ MD. Of the technologies across the three pathways, conventional gasification and FT in the FT MD fuel pathway are more mature, thereby typically achieving higher fuel yields than the other technologies. Gasification refers to partial oxidation of the pre-processed feedstock at elevated temperatures to produce syngas, which is then synthesized to fuels and wax using a FT catalyst, and the products are refined to yield naphtha, jet and diesel. In the case of plasma gasification, plasma torches are used to create the high temperatures necessary for decomposing and oxidizing the feedstock. Plasma gasification is a less mature technology but because it renders toxic substances non-hazardous and provides cleaner syngas with no tars, it has received considerable interest in waste management. Relative to the first two pathways, the ATJ MD pathway involves more conversion steps, with the syngas converted to ethanol followed by a series of chemical upgrading steps to produce jet fuel, diesel and naphtha. Due to low ethanol yields and losses in the additional conversion steps, the fuel yield is lower but catalytic synthesis of alcohols is less capital-intensive than FT.

Although classification to remove inorganics, sizing, and drying of feedstock is necessary for thermochemical pathways, these pre-processing steps can be less energy- and cost-intensive than the pretreatment required for biochemical pathways, wherein the biodegradable and non-biodegradable content of MSW must be separated.[16] The data to calculate the material and energy balances (MEB) for each pathway are obtained from literature for a facility size of approximately 3000 tonnes per day (tpd) of raw MSW feed (2000 tpd of dry, processed MSW), based on the size of large landfills in the US that receive more than 30% of the nation's MSW.[9] A brief explanation on each pathway and the references used to calculate their respective MEB can be found in the SI under 3.2. We use probability distributions to capture parameter uncertainty, as defined in Table S1.

2.3 Life cycle GHG emission analysis

GHG emissions are calculated as mass of GHG per unit of energy (lower heating value). We include carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions. CH_4 and N_2O emissions are converted into CO_2 equivalents (CO_2e) using the 100-year global warming potentials of the three gases.[17] Climate impacts of non-CO_2 MD fuel combustion emissions are not included in the analysis and this may underestimate climate benefits particularly in aviation, since paraffinic alternative jet fuels (such as those produced from FT synthesis) have been found to significantly reduce black carbon emissions.[18] We allocate GHG emissions amongst energy products on the basis of their relative energy contents[59] for the baseline analysis. System expansion is assessed for sensitivity analyses with regard to energy products. For non-energy products, i.e. elemental sulfur and construction aggregates, we use system expansion. Market-based allocation is an alternative method for allocating emissions between products that are used for different purposes (such as energy- and non-energy products) [58, 8] We assessed the effect of using market-based allocation to allocate emissions between energy and non-energy products instead of system expansion, and found the difference to be negligible. Details are presented in the SI in Table S11 and in Section 3.2.

The calculated mass and energy balances are integrated with life cycle inventories and databases to compute the GHG emissions. Fuel transportation and distribution emissions are obtained from Argonne National Laboratory's Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation model (GREET .NET 2015).[19] We use jet and diesel fuel combustion CO_2 emission factors from the Intergovernmental Panel on Climate Change (IPCC).[20] The MSW-related emission factors for feedstock transportation, landfill, incineration and recycling, as well as the life cycle GHG emissions for production of construction aggregates are obtained from the US EPA's Waste Reduction Model (WARM) model.[21] We use the default feedstock transport distance of 20 miles from the WARM model for the baseline analysis, and assess the sensitivity of the results to variations in the parameter. In order to capture the uncertainty associated with LCA parameters, the probability distributions shown in Table S1 are employed.

2.3.1 MSW feedstock characteristics

When using the method presented by the US Energy Information Administration (EIA), the lower heating value (LHV) of the US average MSW discards is found to be approximately 13 MJ/kg.[22] The LHV of the pre-processed and dried MSW is approximately 20 MJ/kg. This value and the LHV values calculated using other references are used to build the probability distribution for LHV.[23-26] Similarly, we calculate carbon, ash (inorganic), moisture and sulfur content.[25-29] One distinction between MSW and biomass as feedstocks for alternative MD fuel is that a portion of the carbon in MSW is not biogenic by origin, attributable to plastics and rubber.[22] Therefore, the non-biogenic proportion of carbon in the feedstock is calculated to determine the non-biogenic share of process and combustion emissions that have to be counted in the analysis. We do not vary the MSW composition for the stochastic analysis because we are considering the composition of the total discards in the US as an average representation. Due to lack of uncertainty estimates, we did not assign arbitrary bounds and instead, assess sensitivity of results to MSW composition in different cases.

2.3.2 Replaced waste management strategy

Converting MSW to MD fuels avoids the GHG emissions that would have otherwise resulted from landfilling and incinerating the MSW for energy recovery, but also eliminates existing GHG emission benefits that currently occur if landfill gas is recovered and displaces fossil energy use. Emission factors that account for all of the above effects are obtained from the WARM model for each material type.[21] The combustion emission factors for each material type take into account non-biogenic combustion CO_2 and N_2O emissions, transportation GHG emissions, avoided electric utility GHG emissions, and avoided emissions due to steel recycling, if applicable.[23] For landfill gas recovery we use the emission factors reported for US average landfill gas recovery rates, based on the average landfill type mix. Even after accounting for GHG benefits from landfill gas recovery and carbon sequestration, the net avoided landfill GHG emissions amount to 162 gCO_2e per tonne of raw MSW and the net avoided combustion GHG emissions amount to 5 gCO_2e per tonne of raw MSW.[21] The sum of these is used as the GHG credit from the replaced waste management strategy in this analysis. Lower and upper bounds for the GHG credit are applied for the stochastic analysis based on IPCC guidance.[30]

2.3.3 Classification and recycling

The energy requirements for classification and sizing of the MSW feedstock are derived from simulation models of refuse-derived fuel facilities by Pressley *et al*. and Caputo *et al*.[10, 31-32] The inorganics stream that is separated from the MSW feed comprises approximately 55% metals and 30% glass. The composition breakdown by material (ferrous, aluminum etc.) and product type (cans, packaging, durable goods etc.) is used in conjunction with the appropriate GHG emission benefit factors from recycling in the WARM model.[6, 21] For product types that have not been modeled in WARM we rely on similar products as proxies.[33] In order to capture the associated uncertainties we use a range of recycling rates. The rates vary from recycling only aluminum cans, steel cans and glass bottles at the lower bound (approximately 30% of total scrap by weight), to recycling approximately 80% of the total metals and glass by weight. The most likely estimate is assumed to correspond to recycling aluminum cans, aluminum in durable goods (as aluminum ingot), steel cans and glass bottles (approximately 40% of total scrap by weight).

2.3.4 Fuel production process

The MEB calculated for each conversion pathway are used to estimate the process-related GHG emissions, the GHG emissions associated with production of inputs, and the allocation of emissions amongst co-products. Process CO_2 emissions are calculated based on carbon balances. The carbon converted to fuels, alcohols, tars and dissolved hydrocarbons is accounted for and any remaining carbon from the input feedstock and catalysts is assumed to be converted to CO_2.[34] GHG emissions associated with production of inputs are determined from the GREET model.[19] The energy product slate used for allocation of emissions by share of energy for each pathway is given in Table S2 in the SI. Excess electricity generation is calculated from the literature, and correlated to fuel yield. The result is an inversely proportional relationship since lower fuel yield implies that more unconverted syngas can be combusted for electricity.[35]

If the simulation models used to calculate the energy balances for the conversion technologies are missing data on some of the material inputs and outputs, we estimate these with uncertainty ranges from other studies, referenced in Table S1 in the SI. Emissions associated with disposal of rejects, ash and spent catalysts in landfills are accounted for using the WARM model, and since these materials are inorganic they do not contribute to anthropogenic methane emissions.[36] The mass of rejects, ash, slag and sulfur are calculated from the feedstock composition based on sorting efficiency (90%), the calculated inorganic content and sulfur content in pre-processed MSW, as well as elemental sulfur recovery rates are from the literature.[37]

2.4 Techno-economic analysis

The TEA calculates production costs and NPV from the plant perspective, and changes to the MSW life cycle are accounted for if they lead to a change in the costs of inputs or to a change in revenues. The replaced waste management strategy affects the feedstock cost because replacing existing or new landfills may allow the plant to charge similar tipping fees for the MSW feedstock. At the same time this could lead to the commodification of MSW and, in the long run, result in a positive feedstock cost. MSW-to-fuel technologies have not entered large-scale commercial production in the US yet. However, existing empirical evidence shows the emergence of long-term, zero-cost MSW feedstock contracts.[38] We, therefore, follow Jones et al.[14] and assume zero feedstock cost for the baseline analysis but quantify the sensitivity of results to positive and negative feedstock costs. To quantify the costs of production of MD fuels produced from MSW, we calculate minimum selling prices (MSP) and net present value (NPV) of plant operation by adopting the discounted cash flow rate of return (DCFROR) model from Pearlson et al for a 20-year plean.[39] All prices are expressed in 2014 USD. Further financial assumptions are listed in Table S3. Facility capital cost estimates are obtained from the literature referenced for material and energy balances for each pathway.[11,14,57] In some cases, the estimates are supplemented with additional capital costs of the processes that are not modeled in the particular studies, such as MSW pre-processing, naphtha reforming to gasoline, and alcohol-to-jet conversion. The material and energy balances calculated for the LCA are carried over to the TEA in order to calculate operating costs and sales revenues. More detail on the cost and revenue calculations and underlying data are provided in Section 3.3 of the SI.

2.5 Uncertainty assessment

We implement stochastic analysis using Monte Carlo simulations, wherein parameters are randomly sampled from their probability distributions for 10,000 iterations. This translates the uncertainty in the input parameters to uncertainty in the results. Parameter uncertainty in this analysis stems primarily from data limitations. We assign uniform distributions when available data are considered equally likely. For example, in the cases of LHV, carbon, ash, moisture and sulfur content of the MSW feedstock, we calculate values using different methods from literature[22, 23-26, 25-29] (see 2.3.1). Since there is no data suggesting that one value is more likely than the other, we assume that all the values are equally likely and assign uniform distributions. When data is available to estimate minimum and maximum bounds, as well as a most likely value, we assign triangular or pert distributions. A second type of parameter uncertainty in this analysis is statistical uncertainty associated with availability of a large number of data samples, for example, availability of historical data for commodity prices. In this case, the uncertainty distributions are dictated by the samples, based on best fit using the Anderson-Darling test.[40] A detailed explanation of the methods we use to quantify the uncertainties associated with the conversion efficiency of the pathway, capital cost, and fuel and energy prices is provided in section 3.4 of the SI.

3 Results

3.1 Life cycle GHG emissions

The results for net life cycle GHG emissions for the three MSW to MD fuels pathways are summarized in Table 1. The median results of 32.86, 62.34 and 52.74 gCO_2e/MJ for FT, Plasma FT, and ATJ MD fuels, respectively indicate that they have the potential to reduce life cycle GHG emissions compared to the conventional MD baseline of 90 gCO_2e/MJ.[41] However, parameter uncertainty translates into ranges that 95% of the Monte Carlo simulation results lie within: 18.45 – 47.33, 43.55 – 81.47, and 26.44 – 79.32 gCO_2e/MJ, for FT, Plasma FT, and ATJ MD fuels, respectively. The probability density functions of the life cycle GHG emissions are provided in the upper part of Figure S2 in the SI. Cumulative probability curves of Figure S3 indicate that the probability of MSW-derived FT, Plasma FT and ATJ MD fuels satisfying the minimum 60% emissions reduction requirement (compared to conventional MD) under the US Renewable Fuel Standard (RFS2) is 65.7, 0.1 and 10.2%, respectively. Note that the 60% threshold under RFS2 is for cellulosic biomass. Based on the cellulosic content of the MSW, the produced fuels could be categorized as cellulosic or advanced. In the case of the latter, a less stringent threshold of 50% applies. Figure S4 compares the 95% confidence interval results with the life cycle GHG emissions of other biofuels.

The conventional gasification and FT pathway has the highest fuel yield of the three pathways; approximately 50-57% of the input MSW energy (LHV) is converted to fuels (with 54% as the mode of the fuel yield probability distribution). The other two pathways have lower fuel yields, implying that more of the non-biogenic carbon in the MSW feedstock is converted to CO_2 during the process. Higher emissions during fuel production, which are then allocated over lower fuel and energy co-product yields, results in higher net life cycle GHG emissions for the other two pathways.

Table S4 shows the results for each pathway broken out by life cycle step. The credits from the replaced waste management strategy and recycling are major contributors to the overall GHG emissions. These credits, as well as the emissions associated with feedstock transport, are the same for each of the three pathways on the basis of per tonne of input MSW, but vary when they are allocated over the fuel and energy co-product yield.

The fuel production and combustion steps are major sources of GHG emissions in all three pathways. The Plasma FT MD pathway has the highest fuel production emissions per tonne of input MSW. The mode of the fuel yield probability distribution for this pathway is approximately 38%. At this fuel yield, the pathway generates more excess electricity than the other pathways (almost 9% of the input MSW LHV) but at the upper bound of fuel yield of 46%, the plant has to import electricity to meet plasma power requirements. This results in increased net GHG emissions due to the high carbon intensity of the US average grid mix (160.1 gCO_2e/MJ).[19] The grid makeup is described in Table S5. Fossil fuel inputs such as petroleum coke and natural gas further increase the GHG emissions associated with fuel production in this pathway.

For the ATJ MD pathway, the fuel yields vary between 24 – 40% of the input MSW LHV with 31% as the mode, the lowest of the three pathways. Therefore, on a per MJ of MD fuel basis it has higher fuel production emissions, which are offset by higher credits per MJ from waste management strategy and additional recycling, resulting in a 15% lower median GHG emissions than the Plasma FT MD pathway. Fuel combustion emissions attributable to the non-biogenic portion of the MSW feedstock are similar for the three pathways and vary only because of the different proportions of diesel and jet fuel produced by each pathway.

In addition to the standard deviation measures listed in Table 1, we also quantify the contributions of the parameters that are sampled for the stochastic analysis to the overall variance of the results through a first-order sensitivity index analysis. Uncertainty associated with the non-biogenic proportion of carbon in the feedstock contributes approximately 47%, 41% and 41% of the total variance of life cycle GHG emissions for FT, Plasma FT, and ATJ MD fuels, respectively. This translates to larger standard deviations for the fuel production and combustion steps where the non-biogenic share of emissions is counted. Other major contributions to variance are 39%, 34% and 38% from the recycling credit for FT, Plasma FT, and ATJ MD fuels, respectively. Detailed results are presented in Figure S5.

We also conduct sensitivity analyses to quantify variability within the pathways. Figure 2 shows the five drivers that are assessed by varying each one in isolation. The parameter that produces the largest change in results for all three pathways is the MSW composition, characterized by the non-biogenic proportion of carbon in the feedstock and the LHV of the feedstock. The 0% non-biogenic case assumes absence of all plastics and rubber. This reduces the energy content of the feedstock to approximately 8 MJ/kg, and is accompanied by a reduction of almost 40% in the quantity of fuel produced per tonne of raw MSW, relative to the baseline. The absence of non-biogenic carbon emissions during fuel production and combustion reduces the median life cycle GHG emissions by 180-320% overall, depending on the pathway. This is because the contribution to the life cycle GHG emissions from the fuel combustion step is zero, and only emissions associated with input utilities and chemicals are counted from the fuel production step. The primary remaining contributors are the replaced waste management and further recycling credits.

The 65% non-biogenic case assumes the absence of food wastes, yard wastes and wood, and this reduces the replaced waste management credit since the landfill emissions due to these biogenic wastes are not avoided. Additionally, the non-biogenic CO_2 emissions from both fuel production and combustion are higher, resulting in a net increase of 60-100% in the median life cycle GHG emissions. These results reflect the sensitivity of life cycle GHG emissions of the MSW MD fuels to variability in the composition of MSW that may occur in different geographic regions.

In the baseline case, US national average landfill gas recovery rates have been assumed in order to calculate the replaced waste management credit in the baseline stochastic analysis. Figure 2 shows two other potential cases: one with no landfill gas recovery (replaced waste management credit of 603 $kgCO_2e$ per tonne of MSW) and the other with all replaced landfills assumed to have landfill gas recovery for energy with aggressive gas collection. In the latter case, replacing the waste management strategy results in GHG emissions of 23 $kgCO_2e$/tonne.[21]

For the conventional gasification and FT pathway, MEB are estimated for a larger facility scale from Larson *et al*.[11] and a different fuel yield case at the same 3000 tpd scale from Vliet *et al*.[42] At the larger 7000 tpd feed capacity scale, a lower fuel yield of 34% (23% excess electricity) leads to higher life cycle GHG emissions by approximately 11 gCO_2e/MJ compared to the baseline median. On the other hand, generating additional excess electricity while maintaining high fuel yield (52% fuels and 8% excess electricity) at the 3000 tpd scale, results in lower net GHG emissions (31.5 gCO_2e/MJ).

In the case of the plasma gasification and FT pathway, the 1000 tpd facility with a fuel yield similar to the baseline generates additional excess electricity, and therefore has lower life cycle GHG emissions (by 8%).[43-44] We assess the effect of increased fuel yield at the 3000 tpd scale that requires additional electricity to be imported from the grid,[44-45] and find that this would result in 56% higher life cycle GHG emissions. Due to data limitations, we assess only the effect of different conversion efficiencies for the ATJ MD fuel pathway. The higher fuel yield is based on future projections by Mu *et al*.[34] and the lower fuel yield is based on a conservative estimate by Jones *et* *al.*,[14] which also forms the lower bound in the uncertainty assessment. These cases and other conversion efficiency scenarios are detailed in Tables S6-S8 in the SI.

Using system expansion instead of energy allocation generates a carbon credit for the excess electricity exported to the grid, and the produced higher alcohols, making the Plasma FT and ATJ MD pathways more sensitive than the FT MD pathway to the emissions allocation method. Changing the feedstock transport distance based on literature estimates to 10 and 70 miles (16 and 113 km), respectively, reduces the associated emissions by 4-8%, respectively increases emissions by 21-37%.[46]

3.2 MSP and NPV

The MSP and NPV results for the three MSW to MD fuels pathways are summarized in Table 1. The median MSP results are 0.99, 1.78 and 1.20 $ per liter for FT, Plasma FT and ATJ MD fuels, respectively. Parameter uncertainty results in ranges of values that 95% of the Monte Carlo simulation results lie within: 0.72 – 1.28, 1.24 – 2.39 and 0.68 – 1.75 $ per liter for FT, Plasma FT, and ATJ MD fuels, respectively. These results, even at the lower bound, are above the approximate average US price of conventional middle distillate fuel in January 2016 of 0.27 $ per liter (refiner price).[47] However, there is volatility associated with fuel prices in the short and long term, and we account for this volatility in the NPV calculations. We fit a normal distribution to the year-to-year price variations of the past 20 years from 1996 to 2015. The short-term price volatility is predicted by sampling from a normal distribution fitted to the year-to-year price variations of the past 20 years from 1996 to 2015. The probability of achieving positive NPV for the project is calculated from the NPV results to be 14%, 0.1% and 7% for FT, Plasma FT, and ATJ MD fuels, respectively. The probability density functions are given in the middle and lower part of Figure S2.

Table S9 and S10 give the results for each pathway, disaggregated by type of cost and type of revenue. Capital costs and fixed operating expenses, which are a function of the capital costs, are the major cost contributors for all three pathways, making up 70-75% of total expenses. The net capital costs are highest for the Plasma FT MD pathway and the lowest for the ATJ MD pathway, but when normalized to the MD fuel yield, the FT MD pathway has the lowest median capital cost per liter of \$0.89/L.

The variable operating expenses attributable to water, catalysts, cleaning chemicals and disposal of wastes are only 2-3% of MSP for all three pathways. Comparison of the results indicates that revenues from the sale of gasoline, and of scrap metals and glass, vary among the three pathways due to technology-specific differences in conversion process product slates and plant feed capacities. The Plasma FT and ATJ MD pathways have higher co-product revenues from higher export of excess electricity and sale of higher alcohols, respectively (see Table S10).

We also quantify first order contributions to variance for the MSP and NPV results. The major contributions to variance of MSP are 73%, 70% and 54% from capital costs; 11%, 10% and 10% from fixed operating costs; and 3%, 12% and 24% from fuel yield for FT, Plasma FT, and ATJ MD fuels, respectively. The primary contributions to variance of NPV are 51%, 35% and 35% from year-to-year fuel price variations; 21%, 14% and 15% from the analysis start year (2017) fuel prices; 20%, 40% and 30% from capital costs; and 3%, 6% and 5% from fixed operating costs for FT, Plasma FT, and ATJ MD fuels, respectively. Detailed results are presented Figures S6 and S7. Figure S8 shows the probability of two of these technologies or all three of them to result in identical values when individual contributions of these parameters are considered on the overall variance. These joint probabilities range from 0 to 7.1%.

The majority of variance in the NPV results arises from uncertainty associated with fuel prices. Since the fuel yields are higher for the FT MD pathway, the total variance and standard deviation are also greater than that of the other two pathways. On the other hand, the MSP of the FT MD pathway has the lowest standard deviation (0.14 $/L) of the three pathways because calculation of the MSP divides the net costs over the fuel yield, thereby resulting in an inverse relationship. Following from the fuel yields and the capital costs (shown in Table S1 and explained in Section 3.1), the FT MD fuel has the lowest median MSP and the Plasma FT MD fuel has the highest median MSP of the three pathways. The ATJ MD pathway has the least negative median NPV because the relative reduction of net capital costs outweighs other costs compared to the other two pathways. However, to achieve a positive NPV, the ATJ MD pathway requires a higher selling price for the fuel than the FT MD pathway, because the lower fuel yield implies that each unit of fuel needs to be sold at a higher price.

Figure 3 presents the results of the sensitivity analysis for the MSP and NPV results in terms of discount rate, income tax rate, feedstock cost, plant scale and associated technology parameters, and carbon pricing as an example of a policy driver. The discount rate, which is dictated by the rate of required return for equity and loan interest rate for debt, has the greatest impact on the results. From an investor's perspective, Blazy *et al.* suggest that the discount rate could be up to approximately 22% for novel alternative fuel technologies with significant associated risks.[48] This reduces the probability of positive NPV to 0-0.4% and increases the MSP by 40-60% depending on the conversion pathway. To assess sensitivity in the opposite direction, we use the social opportunity cost of capital based on long-term treasury bond rates from the US Office of Management and Budget as the discount rate (3.2% nominal).[49] This decreases the MSP by 50-70%, and increases the probability of positive NPV above 80% for the FT and ATJ MD pathways.

The plant scale and conversion yield cases assessed for NPV and MSP are the same as for the LCA sensitivity analysis, except for the FT MD pathway, wherein the 3000 tpd case (other than the baseline) evaluated for the economic sensitivity analysis is based on data from Zhu *et al*.[50] At larger feed input capacities, economies of scale are achieved for the conversion technologies. At the same feed capacity and level of capital investment, improvements in fuel yield increase the probability of positive NPV to greater than 50%. In the case of the FT MD pathway, at the same feed capacity, lower fuel yield (39%) and 8% higher capital costs than the baseline results in a decrease of the probability of positive NPV to 0.4%.

In order to quantify the impact of feedstock cost, we use the 2013 US average landfill tipping fees,[6] first as a source of revenue that lowers the median MSP by 20-46% and raises the probability of positive NPV to 2.5-55% and second, as a positive cost associated with the feedstock as it gains value due to end-use as fuels, yielding the opposite effect. The latter increases the median MSP by 25-50% and lowers the probability of positive NPV to 0-2%. Income tax expenses have a similar effect, raising the median MSP by 24-32% when the tax rate is raised to match the 2015 US combined corporate income tax rate of 39%, as reported by the Organization for Economic Co-operation and Development (OECD).[51] In the discount rate and feedstock cost cases, the ATJ MD pathway demonstrates the lowest median MSPs ($0.34/L, $0.64/L) and highest probability of positive NPV (87%, 55%) compared to the other two pathways.

We also present the result of implementing a carbon price of $48.56/tonne (2014 dollars) based on the revised social cost of carbon guidance provided by the US Interagency Working Group on Social Cost of Carbon.[52] This ties together the results of the LCA and TEA analyses. The carbon price improves the median MSP of the FT MD pathway by 11% compared to 3-4% for the other two pathways because of its greater life cycle GHG savings potential of 63% compared to 30-40% (median estimates).

4 Discussion

The results from the LCA show that drawing a larger system boundary allows for analysis of the change in GHG emissions that occurs due to conversion of MSW to fuels, relative to the existing waste management strategy. Therefore, the total life cycle GHG emissions of the MSW MD fuels are dependent on the waste management being replaced, credits from additional recycling, and combustion emissions attributable to the non-biogenic content of the feedstock. We note that the results in this paper represent the current US average characteristics of MSW feedstock; the MSW composition and credit from replaced waste management strategies may vary significantly at different spatial and temporal scopes.[9, 53] Furthermore, this work only quantifies life cycle GHG emissions, whereas additional analyses could include criteria such as air quality and non-GHG climate impacts.[54] We also note that we compare the GHG benefits of MSW MD fuels only to conventional petroleum-derived fuels, and not to MSW-to-electricity or MSW-to-ethanol or other alternative MD fuel pathways. The GHG benefits of different waste management strategies depend on a number of factors, such as MSW composition, carbon intensity of the grid electricity, conversion efficiencies and feedstock pretreatment requirements.[55-56] The mature conventional gasification and FT technologies demonstrate higher conversion efficiencies in the literature, leading to the lowest median lifecycle GHG emissions of the three pathways. Improving fuel yields while maintaining sufficient electricity generation to meet the plant's utility needs could reduce the lifecycle GHG emissions of all three pathways.

The results of the TEA show that MSW MD fuels have higher costs of production than conventional MD fuels. The probability of positive NPV is less than 15% for all three pathways.

Based on capital costs and conversion yields, the conventional gasification and FT pathway has the greatest probability of positive NPV (14%) and lowest median MSP ($0.99/L). MSP can be reduced, and NPV increased, by improving conversion efficiencies and the sale of recyclables for all three pathways. The ATJ MD pathway has the lowest net capital costs for the scale considered, and therefore, the least negative median NPV of -247 million USD. The sensitivity analysis shows the significant impact of the perceived risk of the investment on NPV and MSP results. Several policies and corporate agreements exist that can reduce investment risk and, therefore, increase the economic performance of the different pathways.[60] These include, but are not limited to loan guarantees, capital subsidies and offtake agreements.

Finally, the economic analysis in this study is conducted from the perspective of an investor. Therefore, only actual financial streams were accounted for. We can complement this analysis with a societal perspective which values resource streams instead of financial streams. This is done by assuming the social opportunity cost of capital instead of an investor-driven discount rate, by eliminating tax payments from the analysis as they do not constitute a resource stream, and by valuing GHG emissions savings by means of carbon pricing as outlined in the sensitivity analysis. If this societal perspective is taken, the probability of positive NPV increases to 93%, 67% and 92.5% for the FT, Plasma FT, and ATJ MD pathways, respectively.

Associated content

In the Supporting Information (SI) the reader can find 11 supporting tables, 5 supporting figures, and 9 supporting text excerpts. The SI is available as a PDF file.

Acknowledgements

This work was made possible by funding from the US Federal Aviation Administration (FAA) Office of Environment and Energy under Project 1 of the ASCENT Center of Excellence for Alternative Jet Fuels and Environment under grant: 13-C-AJFE-MIT Amendment Nos. 003, 012 and 016). The authors would like to thank Daniel Williams for his FAA project management. The authors would also like to thank Dr. Wallace Tyner and Xin Zhao at Purdue University, and Dr. James Hileman at FAA for their guidance on technical matters. Any views or opinions expressed in this paper are those of the authors and not of the FAA or other ASCENT sponsors.

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Figures

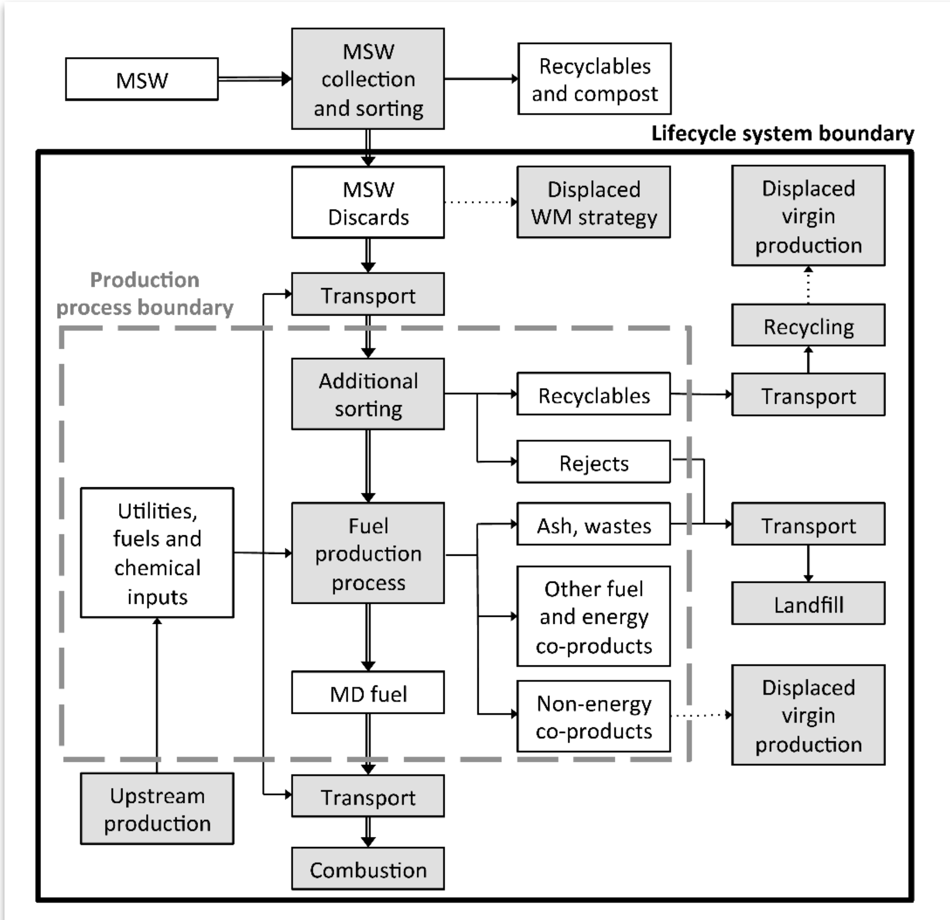

Figure 1. Expanded system boundary of the MSW to MD fuels life cycle (solid line boundary) and fuel production process boundary (dashed line boundary).[a]

[a]Double lined arrows indicate the primary material flow path. Single line arrows indicate secondary input and co-product material flows. Dotted line arrows connect to displaced processes. Processes are indicated with grey background.

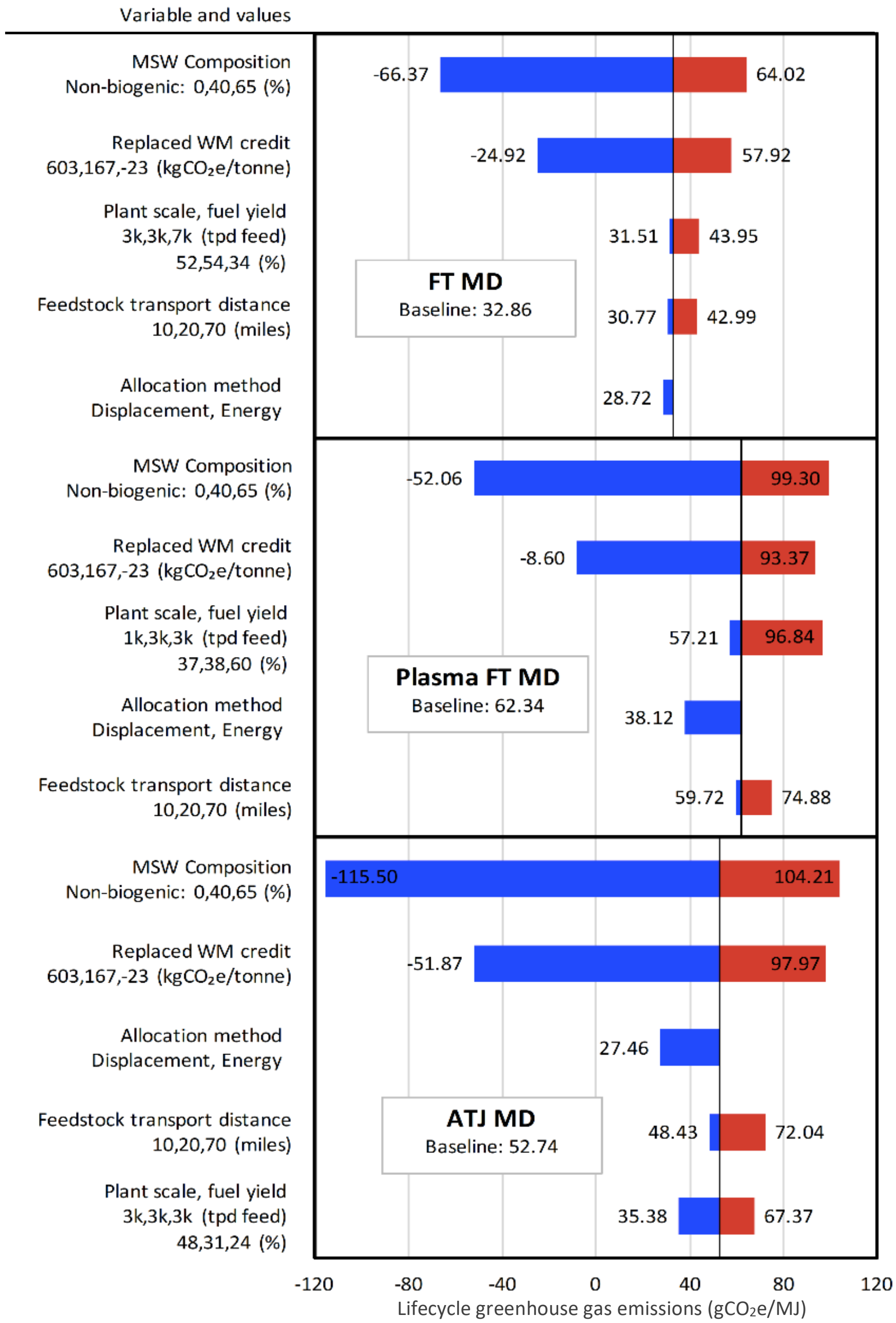

Figure 2. Life cycle GHG emissions sensitivity analysis showing the resultant median values.[b]

[b]The variables and assumptions are listed on the left axis (low, baseline, high).

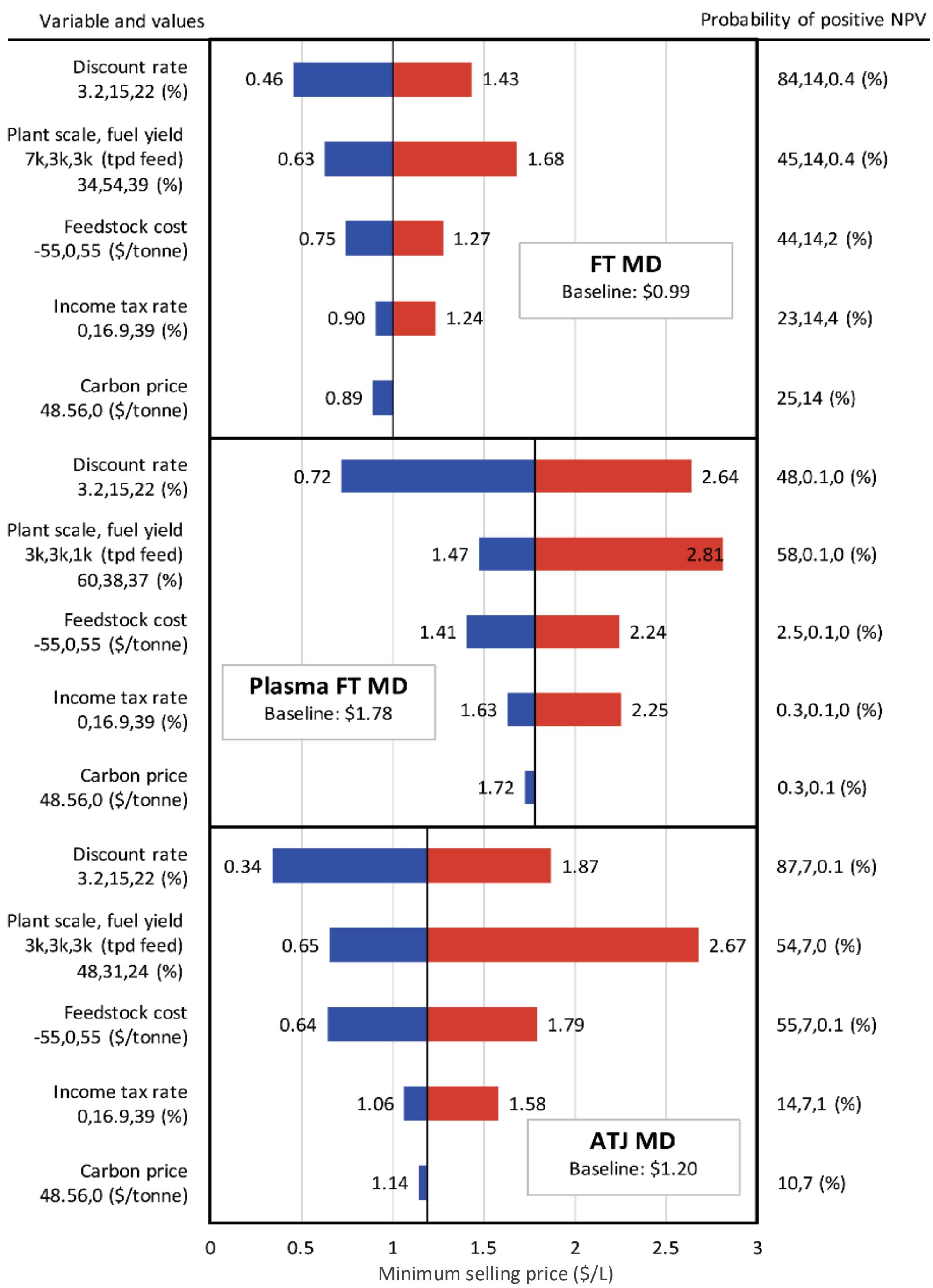

Figure 3. MSP sensitivity analysis showing the resultant median values.[c]

[c]The variables and assumptions are listed on the left axis (low, baseline, high). On the right axis, the probability of positive NPV associated with each case (low, baseline, high) is listed.

Tables

Table 1. Life cycle GHG emissions, MSP and NPV results

	Conversion Pathway	Median	Mean	Std. Dev.
Life cycle GHG emissions (gCO_2e/MJ)	FT MD	32.86	32.89	7.22
	Plasma FT MD	62.34	62.51	9.48
	ATJ MD	52.74	52.88	13.22
Minimum selling price ($/L)	FT MD	0.99	1.00	0.14
	Plasma FT MD	1.78	1.81	0.29
	ATJ MD	1.20	1.22	0.27
Net present value ($B)	FT MD	-0.339	-0.344	0.312
	Plasma FT MD	-0.753	-0.761	0.252
	ATJ MD	-0.247	-0.247	0.163