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Beyond recycling: transforming municipal solid waste into sustainable middle distillate fuels

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ABSTRACT: In all US EPA regions except for Region 1 (New England), landfilling is the primary method of disposal of municipal solid waste (MSW). Landfills, which tend to be placed in disadvantaged areas, pose long term management issues due to the generation of landfill gas (LFG) and leachate, as well as groundwater monitoring. Depending on the age of the waste, the LFG can include a substantial amount of methane and carbon dioxide. In 2022, over a sixth of all methane emitted in the US originated from landfills. Leachate is a potential source of pollutants, including contaminants of emerging concern (CECs), even when leachate collection and treatment are used. There is also the potential of landfill failure, including in landfills built and designed according to modern regulations. While the use of recycling can divert MSW from landfills, a portion of waste placed in recycling containers is ultimately deemed non-recyclable MSW, or N-MSW. This N-MSW, which is mostly paper and plastic, is ultimately landfilled. However, waste to energy (WTE) processes have the potential to further reduce the amount of N-MSW sent to landfills. One such process is gasification/Fischer-Tropsch synthesis (FT gasification), in which the N-MSW is used to generate middle distillate (MD) fuels. In this study, a life cycle assessment (LCA) was conducted to calculate the greenhouse gas (GHG) emissions from the production MD fuel via FT gasification for a sample of N-MSW from an MRF in Utah, US. Life cycle GHG emissions from MD fuel from FT gasification were estimated and compared to the emissions from management of this sample under current regional practice. Sensitivity analyses were performed to estimate the effects of MSW transportation distance, allocation method between fuel products, and MD fuel transportation method on life cycle GHG emissions for MD fuel production from MSW.

KEYWORDS: Landfills, life cycle assessment, waste management, Fischer-Tropsch, municipal solid waste

1 INTRODUCTION

In US EPA Region 8, which includes the states of Utah and Colorado, less than a tenth of waste is recycled. In all regions except for Region 1, which is located in New England, most of this non-recyclable MSW (N-MSW) gets landfilled (Mukherjee et al., 2020). Landfills release landfill gas (LFG) and leachate, both of which have the potential for negative consequences. In relatively aged landfills, after the oxygen present has been exhausted, the two main constituents of LFG are carbon dioxide and methane (Vallero and Blight, 2019). Landfills were one of the highest emitters of methane in 2022, with over 5/6 of this methane being from landfills handling the disposal of MSW (EPA, 2024). Landfilled paper and cardboard are together responsible for roughly 40% of this methane (Staley and Barlaz, 2009). Leachate is also generated from landfills (Vaverková, 2019). This leachate has been shown to contain pollutants, including contaminants of emerging concern (CECs), and volatile organic compounds (VOCs). A majority of VOCs in leachate have been shown to have moderate or high groundwater mobility (Roy, 2020). While leachate collection systems and composite liners are mandatory for landfills (Conservation and Act, 1976), it is possible for contaminants from leachate to ultimately be released into the environment (Clarke et al., 2015, Vaverková, 2019). Stormwater runoff from landfills also has the potential to release pollutants at concerning levels (Marques and Hogland, 2001). Communities in which landfills are built tend to be socioeconomically disadvantaged (McKinney and Thomson, 2022).

Landfills also have the potential to harm the environment through failure and release of their contents. While catastrophic failure is more likely in landfills that were built in developing countries without the use of compaction, failures have been observed in landfills built according to modern regulations and practices (Vallero and Blight, 2019). These failures are often caused by gravity failure or an excess in pore pressure (u), which is often ignored in designs. Such excess u can occur due to issues with leachate recirculation, or ice buildup. Excess u at the toe can be especially dangerous (Thiel and Christie, 2005).

Recycling is one alternative to landfilling. Waste that is designated for recycling is initially processed and sorted at a material recovery facility (MRF) (Franchetti, 2009); however, some of this waste is unsuitable for recycling. This non-recyclable MSW, or N-MSW, is generally landfilled (Saha et al., 2023). N-MSW from MRFs is commonly made up of plastic and paper materials (Lee et al., 2023), as is the case for the waste used in the present study and in Saha et al. (2023).

Waste to energy (WTE) is an increasingly prevalent way to reduce landfilled waste. Thermal WTE methods involve heating the waste and include pyrolysis, gasification (small amount of oxidant), and incineration (excess oxidant) (Mukherjee et al., 2020). The difference in the three methods is in the oxidant content. In incineration, the waste is combusted to CO_2 and H_2O. In gasification, the amount of oxidant is limited, such that the waste oxidizes to CO and H_2 instead, which is called syngas (Arena, 2012). In Fischer-Tropsch synthesis (FT), the syngas molecules can be reacted to build the molecules that make up jet fuel, diesel, and gasoline (Suresh et al., 2018).

If FT gasification is used, it is theoretically possible for MSW to be used to satisfy over a third of US Department of Defense needs for jet fuel (Beal et al., 2021). In this study, a life cycle assessment (LCA) on the use of FT gasification to convert a sample of N-MSW from an MRF in Utah, USA into sustainable middle distillate (MD) fuels was performed.

2 BACKGROUND

2.1 *Studies on LCA of FT gasification of MSW*

Suresh et al. (2018) performed an LCA and a technoeconomic analysis (TEA) on the use of FT, alcohol-to-jet (ATJ), and plasma FT gasification for jet fuel from MSW. Conventional management was included as a displaced process, rather than as a separate system. Ranges of values were used for waste properties, process energy requirements, emissions, and costs. Unlike in the present study, Suresh et al. (2018) assumed that the MSW would be made up of all waste not sent for recycling or composting. The LCA by Suresh et al. (2018) found that the life cycle emissions for jet fuel from conventional gasification and FT synthesis (similar to the present study) were 33 g CO_2e, far lower than the 90 g CO_2e/MJ fuel for petroleum fuel. The TEA showed that, without subsidies, the substitution of FT jet fuel from MSW for petroleum jet fuel is unlikely to be cost-effective (Suresh et al., 2018).

An LCA conducted by Argonne National Laboratory (ANL) (Lee et al., 2023) considered three MRF scenarios: 40% paper/60% plastic, 65% paper/35% plastic, and 100% paper. The 100% paper scenario was the closest in composition to the sample of N-MSW in the present study. In Lee et al. (2023) and Suresh et al. (2018), all carbon in paper was treated as biogenic. The study by Lee et al. (2023) found that, without carbon capture in gasification, emissions from the 100% paper scenario were 30 g CO_2e, which is less than for conventional jet fuel. However, for the 60% plastic MRF scenario, the emissions were actually higher than the emissions for conventional fuel and landfilling (Lee et al., 2023).

The International Civil Aviation Organization (ICAO) developed the Carbon Offsetting and Reduction Scheme for International Aviation (CORSIA) Method for LCAs estimating the GHG emissions for sustainable aviation fuels (SAF). Under the CORSIA Method, a sustainable fuel is one for which the life cycle GHG emissions are 8.9 g less than for conventional aviation fuel per MJ of fuel, or 89 g CO_2e/MJ. For fuels produced from FT gasification of MSW, emissions are estimated from the non-biogenic carbon content (Prussi et al., 2021) via the following equation (ICAO, 2024):

$$GHG\ (CO_2e) = NBC * 170.5 + 5.2 \tag{1}$$

In this equation, NBC is the percent of fossil carbon expressed as a decimal. Under the CORSIA Method, the estimated emissions for a 100% biogenic carbon sample would be 5.2 g CO_2e/MJ fuel. Fuels with more than 50% fossil carbon are not considered sustainable (ICAO, 2024).

2.2 *Studies on geotechnical properties of MSW*

Bareither et al. (2012b) performed direct shear tests on MSW samples from two landfills in the US: one in Wisconsin, and another in North Carolina. Some specimens were subject to simulated decay in the laboratory prior to testing. The composite friction angle, ϕ and cohesion, c, were determined at 37° and 20 kPa respectively. In individual tests, values of ϕ and c of 29°-44° and 7 kPa-26 kPa respectively were observed. In addition, Bareither et al. (2012b) briefly reviewed studies on the impact of decomposition on MSW shear strength and found positive and negative effects in different studies. They state that the change in shear strength with age in each landfill depend on the makeup of that landfill.

Bareither et al. (2012b) cites another study by Hossain and Haque (2009) in which the ϕ decreased with increasing decomposition. The two main reasons for this decrease in strength were the loss of fibers that gave the paper its structure and the increase in the relative percentage of plastic in the waste. The placement of cover soil was also simulated by the addition of up to 30% soil. While this resulted in higher ϕ for new and decomposed MSW, a decrease in ϕ with increasing decomposition was still observed (Hossain and Haque, 2009). However, in the tests conducted by Bareither et al. (2012b) the ϕ generally increased with increasing decomposition. This is likely because the MSW tested had proportionally more soil, glass, and inert particles compared to the waste in Hossain and Haque (2009). In general, ϕ and the percentage of soil, glass, and inert material are positively correlated (Bareither et al., 2012b). When processing N-MSW for gasification, such materials are screened out and discarded for landfilling (Saha et al., 2023).

Karimi et al. (2024) performed a literature review on the nature of the relationship between hydraulic conductivity (*k*) and various factors, including decomposition level, unit weight, and overburden pressure. Regardless of whether waste was new or aged (degraded), a negative correlation between overburden and *k* was observed. In new MSW not subject to overburden, *k* values were typically in the magnitude of 10^{-5}cm/s - 10^{-2} cm/s. The range at 600 kPa of stress was substantially lower, at 10^{-7}cm/s - 10^{-5} cm/s. Decomposition generally led to a substantial increase in *k*, from a median on the order of 10^{-5} cm/s for fresh MSW to a median on the order of 10^{-1} cm/s for moderately and severely decomposed MSW. However, the review found that, at times, decomposition can lessen *k*. According to Karimi et al. (2024), one reason for this is that the decomposition process results in smaller particles. Much like in well-graded sand, this results in less free void space in which water can flow. Depending on the characteristics of the waste, it is also possible that the decomposition could result in smaller and fewer voids, which would also lessen *k*.

Bareither et al. (2012a) experimentally determined the coefficient of compression (C_c) in new and aged MSW specimen. All C_c values observed were between 0.22 and 0.28. The highest value was observed for MSW that was retained on a 25 mm sieve (R25); the lowest value was observed for an N-MSW sample that was not sieved. The C_c for the P25 sample was approximately 0.24. The P25 sample had a higher percentage of dirt, glass, and inert particles, while the R25 sample had a higher percentage of paper and plastic. This suggests that MSW with greater percentages of paper and plastic is more compressible. Based on the experimental data and data from prior studies, Bareither et al. (2012a) developed a method to predict C_c based on water content, dry unit weight (γ_d), and amount of degradable waste. The equation is as follows:

$$WCI = w_d * \left(\frac{\gamma_w}{\gamma_d}\right) * \left(\frac{OW}{100 - OW}\right) \tag{2}$$

In this equation, w_d is the water content and γ_w is the unit weight of water. *OW* is the content of decomposable waste, which includes paper, as well as other biodegradable waste. *WCI* is the waste compression index, the log of which is positively correlated with C_c (Bareither et al., 2012a).

3 GOAL AND SCOPE

3.1 *Goal and functional unit*

The goal of this LCA was to calculate the life cycle greenhouse gas (GHG) emissions from the production of one megajoule (MJ) of MD fuel from FT gasification of a sample of N-MSW from an MRF in Utah, USA and to compare the emissions to the baseline scenario. In the baseline scenario, one MJ of MD fuel is produced from petroleum, and the amount of N-MSW needed to produce one MJ of MD fuel is landfilled. A functional unit of one MJ (LHV) of MD fuel combusted was used for this LCA. A similar functional unit was used in Suresh et al. (2018).

3.2 *LCA process and system boundary*

In this LCA, the carbon dioxide equivalent (CO_2e) GHG emissions of greenhouse gases (GHGs) were calculated for two different N-MSW management strategies. The baseline management strategy is the production of conventional MD fuel, as well as the landfilling the waste needed to produce 1 MJ of fuel under the FT gasification scenario. In the FT gasification management strategy, MD fuel is produced using FT gasification of the N-MSW. For this LCA, it was assumed that

MSW, whether it is to be used for SAF manufacturing, was routed through the MRF; therefore, the energy associated with the MRF operation was not considered. A similar assumption was used in Lee et al. (2023).

4 INVENTORY ANALYSIS

4.1 *MSW sample*

The sample of N-MSW for this study was sent from an MRF in Utah, United States. The MRF is the same as the one in Saha et al. (2023). From the MRF, the N-MSW was initially sent to Idaho National Laboratories (INL), where materials that were unsuitable for gasification, such as metals, were separated out of the waste. The N-MSW was separated into five fractions: paper, cardboard, rigid plastic, thin film, and foam. A portion of the waste was left un-fractionated (mixed). The waste was then shredded. This processing is further described in Saha et al. (2023).

The waste was sent to the Georgia Institute of Technology (GT) from INL for physical and chemical analysis. Representative specimens were taken for each waste type and for the mixed waste by sampling the top, middle, and bottom of the drums in which the waste was sent to GT. The N-MSW specimens were subject to proximate analysis (PA) at GT. Ultimate Analysis (UA) was done by Mineral Labs, inc. in Salyersville, KY. The PA and UA results for the sample are shown below in **Tables 1** and **2** respectively.

Table 1. Proximate analysis results

Waste type	Moisture (%)	Volatile (%)	Fixed carbon (%)	Ash (%)
Cardboard	5.9	75.5	7.0	11.6
Paper	5.1	73.3	6.7	14.9
Foam	1.0	93.3	0.8	4.9
Rigid plastic	1.3	95.4	0.7	2.6
Thin film	0.4	96.1	0.9	2.6
Mixed	4.7	78.5	8.8	8.0

Table 2. Ultimate analysis results

Waste type	C (%)	H (%)	N (%)	S (%)	Ash (%)	O (%)
Cardboard	42.2	8.0	0.5	0.4	6.4	42.6
Paper	37.9	7.1	0.5	0.6	12.2	41.8
Foam	80.7	9.0	0.9	0.4	2.3	6.8
Rigid plastic	74.0	10.3	0.4	0.0	4.3	10.9
Thin film	79.2	11.3	0.3	0.4	3.6	5.2
Mixed	42.5	8.3	0.6	0.6	12.6	35.3

The five fractions were combined into two broad categories: "papers," (cardboard and paper), which were assumed to be entirely composed of biogenic carbon, and "plastics" (rigid plastic, thin film, and foam), which were assumed to be entirely composed of fossil carbon. The relative compositions of each fraction within each broad category ("papers" and "plastics") were estimated using composition data from Saha et al. (2023) since samples from the same MRF were studied. "Papers" were estimated at 59.5% cardboard and 40.5% paper, and "plastics" were estimated at 43.6% thin film, 47.3% rigid plastic, and 9.1% foam.

The relative percentages of fractions in each category and the UA results were then used to estimate the average percent carbon for each broad category: 40.5% C in "papers" and 76.9% C in "plastics." The ultimate composition of the "Mixed" MSW shown in Table 2 was assumed. The category percentages were estimated from the UA carbon compositions, rather than PA compositions due to the possibility of synergistic effects in proximate analysis as observed in Gunasee et al. (2016). The percentages of plastic and paper were estimated as follows:

$$\frac{Mixed\_avg - Paper\_avg}{Plastic\_avg - Paper\_avg} * 100\% = PctPlastic \qquad (3)$$

The percent of "papers" was taken as by subtracting the percent plastic from 100%. The percentages were found at 94.4% "papers" and 5.6% "plastics." The carbon percentages in each category, and the relative proportions of "papers" and "plastics" were used to estimate the percent biogenic carbon at 89.9% by mass, and the percent fossil carbon at 10.1% by mass.

4.2 *MSW management via FT gasification*

4.2.1 *Gasification plant*

For this Assessment, FT gasification of the N-MSW was assumed to occur at a plant with a design similar to the hypothetical plant design described in a 2014 report by the National Energy Technology Laboratory (NETL) (Shah et al., 2014).

4.2.2 *Co-products and carbon balance*

In addition to MD fuels, which include diesel and jet fuel (Suresh et al., 2018), FT synthesis also generates naphtha, which can be synthesized into gasoline, and waxes, which can be broken into naphtha and middle distillate fuels via hydrocracking (Botes et al., 2011).

For this Assessment, the carbon balance and fuel yields from the NETL plant design (Shah et al., 2014) were used. The carbon mass percents according to this design are as follows: 11% to naphtha, 30% to diesel, 1% to slag, and 59% to stack gas (Shah et al., 2014). The category of "Diesel" was assumed synonymous with MD fuels. Stack gas was assumed to be entirely CO_2.

4.2.3 *Fuel properties*

The carbon mass percentages and volumetric densities of MD fuel and naphtha were calculated from the NETL plant design (Shah et al., 2014). Because mass energy densities were not included in the report on the plant design, the values from GREET1 for FT diesel from biomass and FT naphtha were used as approximations for this study. These values were 43.25 MJ/kg and 44.38 MJ/kg respectively (Wang et al., 2024). This would correspond to an energy yield of roughly 72.1% diesel and 28.9% naphtha.

4.2.4 *Electricity*

The US national grid mix in the NETL CO2U Database (Skone et al., 2022) was used for emissions estimates from electricity.

4.2.5 *Allocation*

An energy-based allocation method was used to allocate emissions between middle distillate fuels (the product of interest) and gasoline. Energy based allocation was used in two recent LCAs on sustainable fuel from MSW (Lee et al., 2023, Suresh et al., 2018) and is used in the CORSIA Method (ICAO, 2024).

4.2.6 *MSW processing*

N-MSW from the same MRF as the waste in this study was found to have a relatively low moisture content (Smith et al., 2024), so the energy required for drying of the waste was not considered for this Assessment. The waste was assumed to be processed as in Saha et al. (2023), in which waste was milled in a rotary shredder to approximate diameters of 6 mm, 4 mm, and 2 mm. During shredding, an additional screen was used to separate very fine particles, as such particles were likely to be made of materials unfit for gasification, such as glass or sand. Approximately 13% of waste was very fine or unaccounted for after shredding to 6 mm; around 20% was very fine or unaccounted for after shredding to 4 mm or 2 mm (Saha et al., 2023). For this study, 85% of the waste was assumed to remain after processing. The same value was used in a prior LCA (Suresh et al., 2018).

For the FT gasification scenario, the processing facility was assumed to be co-located with gasification/FT. In the NETL plant design, processing/handling requirements were estimated at approximately 8 kwh/ton of coal (Shah et al., 2014). At INL, an average of roughly 14 kwh of electricity was required to shred one ton of MSW from the same MRF as in the present study. The extra processing energy was assumed to displace excess electricity generated by the plant.

4.2.7 *MSW transportation*

The transportation distance for MSW was assumed at 50 miles one-way (100 miles round trip for a truck) for gasification and baseline scenarios.

4.2.8 *Discarded MSW management*

In scenarios in which processing occurs at the FT gasification plant, the N-MSW discarded during processing was assumed to be transported 50 miles to a landfill.

4.2.9 *HHV/LHV during gasification*

Because the NETL design includes heat recovery (Shah et al., 2014), higher heating value (HHV) of the waste was used in the present study instead of the lower heating value (LHV). Because more extensive UA data had been collected than PA and categorization data, the HHV was estimated from the UA results for the mixed MSW in **Table 2** with the following equation from Shi et al. (2016):

$$HHV = (0.350C) + (1.01H) - (0.0826O) \tag{4}$$

Where HHV is in MJ/kg and the elemental letters are the contents as percentages.

4.2.10 *FT gasification*

Efficiency, or percent of energy in the MSW that will be transferred into fuel after gasification and FT synthesis, was estimated at around 44% based on the carbon balance from the NETL plant design (Shah et al., 2014). This would imply that approximately 0.11 kg of processed MSW would be needed to produce 1 MJ of fuel. The hypothetical plant in the NETL design was intended to be capable of operation without external electricity for FT gasification (Shah et al., 2014).

As per the proportions in Table 2 above, the mixed MSW was 42.5% carbon. In the hypothetical gasification plant design from the NETL (Shah et al., 2014), approximately 59% of carbon in the waste was emitted as CO_2 stack gas during gasification/FT. If energy-based allocation were used, this would correspond to an emission of 103 g of total CO_2/MJ MD fuel, of which 10.3 g would be from fossil carbon. For ash/byproduct management, the 0.4 g CO_2e/MJ of fuel used in Suresh et al. (2018) was assumed.

4.2.11 *Excess electricity*

Assuming processing occurred at the plant, approximately 30 kJ of excess electricity would be produced per MJ of fuel. This would correspond to an emissions reduction of 4.1 g CO_2e/MJ fuel assuming the current US grid mix.

4.2.12 *Plant to pump*

The assumed plant to pump emissions in prior studies have varied. For example, a value of 0.38 g CO_2e/MJ fuel was used by Lee et al. (2023), and a value of 0.9 g CO_2e/MJ fuel was assumed in the CORSIA Method (ICAO, 2024). Because the exact distance for fuel transport is not known, the CORSIA value of 0.9 g CO_2e/MJ fuel was assumed.

4.2.13 *Fuel combustion*

Lee et al. (2023) assumed that all the carbon in the MD fuel would be combusted into CO_2; a similar assumption was used for the present study. This corresponds to a CO_2 emission of 71.6 g CO_2 per MJ of fuel during use (combustion). Because 10.1% of the carbon is assumed fossil in origin, this will translate to approximately 7.2 g CO_2/MJ of MD fuel.

4.2.14 *GHG emission calculations*

Given the above assumptions, the well-to-wake emissions for this amount of NMSW were calculated at 15.3 g CO_2e/MJ of fuel. The emissions from 1 MJ of MD fuel are shown below in **Table 4**. Values were rounded to the nearest 0.1 g CO_2e/MJ fuel.

Table 4. GHG emissions for FT gasification scenario

Process	Emissions (g CO_2e/MJ fuel)
MSW transportation (MRF to plant)	1.3
Discarded MSW transportation (plant to landfill)	0.2
Discarded MSW landfilling	-0.9
Stack gas	10.5
Excess electricity	-4.1
Ash/byproduct management	0.4
MD fuel transportation	0.9
MD fuel combustion	7.2
Well-to-wake emissions	15.3

4.3 *Baseline scenario*

4.3.1 *Baseline management practice*

In the Mountain West of the US (Montana, the Dakotas, Wyoming, Utah, and Colorado), only 2% of MSW (2.5% of N-MSW) in the Mountain West is subject to waste to energy (Mukherjee et al., 2020). Because the MRF for this LCA was assumed to be in the same complex as a landfill, it is quite likely that all N-MSW would be landfilled for the baseline case, as is assumed for this LCA.

4.3.2 *WARM inputs*

Emissions from landfilling of the waste were calculated using the US EPA Waste Reduction Model (WARM) in OpenLCA (USEPA, 2023). WARM was also used to estimate landfilling emissions in Suresh et al. (2018). The waste was input as “mixed papers (residential)” and “mixed plastics” with the composition estimated as described above. Dry conditions and the US national average LFG collection rate were assumed.

4.3.3 *Conventional MD fuel*

The total life cycle emissions associated with the combustion and production of diesel according to the NETL CO2U Database (Skone et al., 2022) were used for the value of the emissions for conventional MD fuel. The properties of U.S. Conventional Diesel from GREET1 (Wang et al., 2024) were assumed.

4.3.4 *Conventional management emissions*

The estimated GHG emissions for the conventional management of MSW are shown below in Table 5. An energy-based allocation method was used.

Table 5. Conventional management emissions

Process	Emissions (g CO_2e/MJ fuel)
Landfill gas emissions	4.9
Landfill operation	2.5
Electricity generation	-9.1
Conventional diesel	91.2
Baseline emissions	89.5

4.3.5 *GHG emissions savings*

Overall, the diversion of MSW from landfilling/incineration to FT gasification will result in 89.5 g CO_2e/MJ (Baseline) – 15.3 g CO_2e/MJ (FT gasification) = 74.2 g CO_2e/MJ of emission reductions fuel relative to using conventional MD fuel and managing the waste through landfilling.

5 SENSITIVITY ANALYSES

The relationships between GHG emissions and MSW processing plant location and transportation distance, allocation method, and fuel transportation method were characterized via sensitivity analyses as described below.

5.1 *N-MSW transportation sensitivity analysis*

Assuming processing occurred at the FT gasification plant, the GHG emissions would increase by approximately 0.0288 g CO_2e/MJ fuel for each additional mile of transportation.

5.2 *Allocation method sensitivity analysis*

Gravimetric and volumetric flow rates from the NETL plant design (Shah et al., 2014) were used to estimate allocation ratios for volume and mass. The allocation factors for MD fuel were as follows: 0.72 for energy, 0.73 for mass, and 0.70 for volume. The GHG emissions were calculated when steps upstream of and including gasification/FT were allocated by mass, volume, and energy for the FT gasification and baseline management scenarios. For the FT gasification scenario, the use of mass-based allocation resulted in the highest emissions value, at 15.4 g CO2e/MJ fuel, followed by energy-based allocation at 15.3 g CO2e/MJ fuel and volume-based allocation at 15.2 g CO2e/MJ fuel. For the baseline management strategy, GHG emissions were calculated at 89.5 g CO_2e/MJ fuel for all three allocation methods.

5.3 *Plant to pump sensitivity analysis*

The GHG emissions from truck transport and pipeline transport were compared. The energy intensity from GREET1 was assumed for pipeline transport. Electricity was assumed as the energy source. Fuel economy and capacity values from GREET1 for heavy-heavy duty trucks were assumed (Wang et al., 2024); diesel combustion data from the NETL CO2U database was used, as for MSW transportation (Skone et al., 2022). The GHG emissions for fuel transport were estimated at 0.0044 g CO_2e/MJ fuel per mile of truck transport and 0.0014 g CO_2e/MJ fuel per mile of pipeline transport assuming the use of existing pipelines.

6 RESULTS AND DISCUSSION

6.1 *Effect on GHG emissions*

Based on the results from this LCA, for the current sample, management of the N-MSW sample with FT gasification to MD fuel can result in lower GHG emissions than management with the current practice and conventional MD fuel production. If, as is done in the CORSIA Method (ICAO, 2024), avoided landfilling emissions were added to the GHG values for this LCA, and if excess electricity was not considered, the FT gasification emissions for this Assessment would be 21.2 g CO2e/MJ fuel. This is close to value of 22.4 g CO_2e/MJ fuel that would be obtained using the CORSIA method (ICAO, 2024).

The GHG emissions savings from gasification of MSW would differ depending on waste composition. For example, CO_2e emissions would be far higher if the N-MSW blend had more fossil carbon. In the CORSIA Method, fuels produced from FT gasification of MSW with less than 50% biogenic carbon would have emissions that either do not differ significantly from, or exceed, conventional fuel (ICAO, 2024). Similar results were found in the LCA conducted by Argonne National Laboratory (Lee et al., 2023). If the N-MSW had 40% paper/60% plastic, as in one of the example MRF blends in Lee et al. (2023), the emissions from the FT gasification management case would have been over 116 g CO_2e/MJ of fuel, which is far higher than the 20.9 g CO_2e/MJ fuel for the present sample and higher than the baseline emissions of 89.5 g CO_2e/MJ. Lee et al. (2023) also found emissions for such a composition to be higher than the baseline management.

Because of the relatively high biogenic carbon content in paper and cardboard, the global warming potential of fuel produced from the gasification of these materials would be lower than that of a fuel produced from gasification of plastics or foam. Paper is also far more prone to decomposition than plastic and is thus a greater contributor to methane generation than plastic (Bareither et al., 2012b). The global warming benefits of an FT gasification management strategy would be maximized by designating paper N-MSW for gasification and plastic N-MSW for landfilling.

6.2 *Effects on geotechnical properties of landfills*

If a large quantity of MSW were to be diverted from MRFs to landfills, the composition of landfills will be substantially altered. The effects of this alteration will depend on the types of waste that are sent for gasification. According to Bareither et al. (2012b), landfills with a greater percentage of paper and plastic tend to have lower ϕ; landfills with more soil, glass, and inert materials tend to have higher ϕ. In Saha et al. (2023), very fine particles, which tended to be soil or inert particles, were screened out and designated for landfilling during processing. This would mean that only the soil and inert materials would be sent to landfills, rather than the bulk N-MSW. It is therefore possible that reducing the amount of N-MSW from MRFs that is sent to landfills will result in higher ϕ in future landfills.

However, waste samples with higher plastic contents generally have lower values of ϕ (Bareither et al., 2012b), indicating a potential reduction in shear strength if only paper and cardboard N-MSW are managed via FT gasification. Reducing the amount of paper and cardboard in landfills will also reduce the amount of biodegradable waste (Bareither et al., 2012b). As discussed above, the effect of decomposition on shear strength in each landfill varies with makeup of the waste. Ultimately, the long-term effect of the diversion of paper and cardboard from landfills on shear strength would depend on variables such as the relative amounts of plastics and other waste particles and the amount of cover soil.

The effect of the removal of paper on the k of the soil would also likely vary between landfills; however, it is possible that, as a general trend, the increase in k with the age of the waste would be less pronounced because the waste would have less degradable material. Because particles unfit for gasification tend to be smaller Saha et al. (2023), it is possible that the average particle size of the waste could decrease. This would lead to lower values of k (Karimi et al., 2024).

Based on results by Hossain and Haque (2009), it is possible that an increase in the amount of daily cover soil could mitigate any reduction in shear strength if one is observed. The feasibility and effectiveness of the use of higher strength cover soils should be explored in future studies.

The C_c is related to unit weight, water content, and OW content according to Bareither et al. (2012a). A reduction in paper and cardboard will decrease the OW, which, would have a negative effect on C_c according to Equation (2) assuming γ_d and w_d do not change (Bareither et al., 2012a). A variety of factors can affect the unit weight of MSW, including the types of waste present (Zekkos et al., 2006) and the depth (Zekkos et al., 2005). Future studies should be conducted to characterize the effect of the diversion of N-MSW from landfilling on compressibility.

7 CONCLUSIONS

In this LCA, the effect on global warming potential of the diversion of an N-MSW sample from landfills was quantified. The effects of MSW transportation distance and processing plant location, allocation method, and MD fuel transportation distance and method on GHG emissions were also studied. The results from this Assessment show that the management of paper N-MSW via FT gasification can lead to reductions in GHG emissions. This LCA contributes to the field of geotechnical engineering because it examines the benefits of a potential solution to the problem of landfilling and its associated negative effects. Potential effects of this management strategy on landfill geotechnical characteristics were also discussed.

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