



Life cycle greenhouse gas (GHG) impacts of a novel process for converting food waste to ethanol and co-products



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HIGHLIGHTS

- Co-fermentation using SSF at ambient temperature has potential as an ethanol pathway.
- Bio-refinery GHG emissions are similar to corn and MSW ethanol production processes.
- Net production GHG impact is negative with inclusion of waste disposal avoidance.
- Food waste diversion from landfills is the largest contributor to GHG benefits.

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ABSTRACT

Waste-to-ethanol conversion is a promising technology to provide renewable transportation fuel while mitigating feedstock risks and land use conflicts. It also has the potential to reduce environmental impacts from waste management such as greenhouse gas (GHG) emissions that contribute to climate change. This paper analyzes the life cycle GHG emissions associated with a novel process for the conversion of food processing waste into ethanol (EtOH) and the co-products of compost and animal feed. Data are based on a pilot plant co-fermenting retail food waste with a sugary industrial wastewater, using a simultaneous saccharification and fermentation (SSF) process at room temperature with a grinding pre-treatment. The process produced 295 L EtOH/dry t feedstock. Lifecycle GHG emissions associated with the ethanol production process were 1458 gCO₂e/L EtOH. When the impact of avoided landfill emissions from diverting food waste to use as feedstock are considered, the process results in net negative GHG emissions and approximately 500% improvement relative to corn ethanol or gasoline production. This finding illustrates how feedstock and alternative waste disposal options have important implications in life cycle GHG results for waste-to-energy pathways.

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1. Introduction

Renewable transportation fuels have the potential to mitigate climate change and contribute toward energy independence and security. However, current fuels based on sugar or starch energy

Abbreviations: ABV, alcohol by volume; CO₂, carbon dioxide; CH₄, methane; DDGS, dried distillers grains and solubles; EtOH, ethanol; FFS, Feed/Fuel Slurry; GHG, greenhouse gas; LCA, Life Cycle Assessment; LFG, landfill gas; t, metric tonne; MSW, municipal solid waste; N₂O, nitrous oxide; SSF, simultaneous saccharification and fermentation; ww, wastewater; WWT, wastewater treatment.

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crops face significant challenges in terms of economics, availability of feedstock, land use conflict and life cycle greenhouse gas (GHG) emissions. Using waste as a feedstock offers an alternative that avoids many of these problems while also addressing the growing challenge of waste management.

Food scraps account for 21% of waste currently reaching landfills in the United States [1]. In a landfill, food scraps decompose rapidly to produce methane, often before landfill gas (LFG) recovery systems are in place [2]. Landfills accounted for approximately 16% of total U.S. anthropogenic methane emissions in 2010 [1]. Alternatively, food waste can be broken down to simple carbohydrates and converted to ethanol in a bio-fermentation process. Using waste as a feedstock for ethanol production provides the service of waste disposal and has the potential to generate revenue

to ethanol producers in the form of “tipping fees,” which along with other valuable co-products can contribute to bio-refinery profitability.

Industrial (e.g. food processors) and retail (e.g. food preparation) wastes offer significant potential as a feedstock source because they can be source separated and are often a disposal burden to the generator. Fruit juice and cannery industrial waste have been reported as potential biofuel feedstocks [3,4]. Food scraps, which are generally more complex lignocellulosic materials, also have the potential for conversion to ethanol. However, these substrates require the breakdown of starch, cellulosic or hemicellulosic materials into monomeric sugars to enable fermentation. One method of achieving this is simultaneous saccharification and fermentation (SSF) in which enzymatic hydrolysis is performed together with fermentation; this offers the benefit of reduced inhibition of enzymatic activity by saccharification end products, as well as reduced investment costs [5,6]. Although, empirical studies have demonstrated the potential to create ethanol from food scraps using SSF [7–10], commercial-scale bioethanol plants utilizing food scraps do not yet exist. However, a National Renewable Energy Laboratory (NREL) model for lignocellulosic conversion based upon the SSF process has been used to analyze municipal solid waste (MSW) to ethanol conversion potential [11,12]. Implementation of SSF can vary, but most processes are optimized to include an acid or thermal pretreatment and operate at elevated temperatures. Furthermore, commercial models are usually on the scale of 40–80 million gallons of ethanol/year and often include some form of cogeneration to utilize waste heat [11].

Co-fermentation of feedstocks has received limited attention in the literature. Bellmer and Atieh [13] and Dwidar et al. [14] suggest that co-fermentation of beverage waste feedstock with other waste streams can improve pH, provide nutrients, and minimize diffusion of oxygen that might inhibit fermentation. Other studies have reported synergies when sugar- or starch-rich diluents were co-fermented with cellulosic feedstock (e.g., presaccharified wheat with wheat straw [15] or furfural residue with corn kernels [16]).

This study analyzes a pilot fermentation plant where lignocellulosic food scraps are combined with a sugar rich diluent. The food scraps are ground without any other pretreatment and simultaneously co-fermented with diluent, at ambient temperature. The process produces ethanol as well as compost and animal feed co-products; the business model also encompasses revenue for the service of waste disposal. Furthermore, fermentation and dehydration are conducted at separate facilities. This distributed model minimizes the infrastructure and regulatory requirements at smaller fermentation facilities located close to waste streams, while taking advantage of economies of scale by conducting dehydration at a centralized hub.

The objective of this study is to estimate and analyze the GHG impacts of this novel process. Pilot plant (1/15th scale) fermentation data are combined with small-scale commercial distillation data to create a model of the full ethanol production process. This model is used to assess the life cycle GHG impacts and to evaluate the potential of the process as an alternative fuel pathway. The results are compared to those of corn ethanol and conventional gasoline. This study is unique in the literature in that it analyzes a process that produces ethanol from industrial food waste, whereas existing literature analyzes processes for the conversion of MSW to ethanol [12,17,18]. Comparison of our results to these studies highlights the significant impact of waste feedstock composition which is discussed. Conclusions presented here are intended to contribute to knowledge in the areas of bioethanol production, waste management, and related policy.

2. Methods

2.1. Conversion process modeling

The process and system boundaries are shown in Fig. 1. The bio-refinery process is modeled using primary data from the pilot fermentation plant and a commercial dehydration plant and supplemented with data from the literature (represented by shaded blocks) where primary data were not available.

A mass balance was performed for a control run at a pilot scale fermentation plant (10 wet t/day) operated by Epiphygy LLC. The control run consisted of 4.7 wet t of feedstock: 2.3 wet t lignocellulosic feedstock, consisting of food scrap waste from a supermarket chain and 2.4 wet t of diluted fruit syrup food processing waste as a diluent. The source-separated feedstock was transported from the waste generators in totes on trucks. Upon receipt, the food scraps were ground without any other pretreatment and mixed with the diluent. The mixture was combined with cellulose and starch biocatalysts and antimicrobial agents and simultaneously fermented with *Saccharomyces cerevisiae* at ambient temperature. The resulting ferment slurry contained a dilute concentration of ethanol, residual solids, and yeast grown during fermentation. The solids were separated using an 80 μm filter and fed into a composting process, which is accelerated by the grinding and fermentation. The volume and ethanol content of the filtered ferment, and mass of compost produced were measured. These processes are represented by steps 1.1–1.4 of Fig. 1. In step 1.5 a portion of the dilute ferment is concentrated to create a Feed/Fuel Slurry (FFS) with 15% ABV. This is done to reduce transport weight as much as possible without requiring additional costs and regulatory burden associated with transport of flammable liquids. This process is modeled based on literature pertaining to small-scale ethanol distillation, assuming 0.22% ABV in the stillage [19]. Stillage wastewater volume, which is calculated by mass balance, was modeled to be processed onsite in a wastewater treatment (WWT) facility.

The FFS is transported to a regional facility where it is distilled to 96.5% (ABV) and dehydrated using a molecular sieve to anhydrous ethanol. Dissolved solids and solids that were not removed by the filtering process at the fermentation plant, are separated and dried to create an animal feed product similar to dried distillers grains and solubles (DDGS). Wastewater is treated in an onsite WWT facility. The ethanol dehydration process is estimated to be 96.5% efficient.

2.2. Life Cycle Assessment (LCA) methods

2.2.1. Goal and scope

The objective of the analysis is to evaluate this waste-to-ethanol process as an alternative biofuel pathway in terms of GHG emissions. A functional unit of 1 L of ethanol is used which is then converted to a unit of transport energy (1 MJ) for comparison to conventional gasoline (CG).

2.2.2. System boundaries

The bio-refinery system boundaries are shown as bolded lines in Fig. 1. It consists of two phases: fermentation and dehydration. The system boundary is set where the waste is introduced into the system. The food production processes that generate the waste are considered fixed with respect to process, materials, and consumption and thus not included within the boundaries [20,21].

The life cycle impacts include both indirect and direct emissions. These include, the indirect emissions associated with the production, transmission and distribution of electricity used in the process; the direct and upstream emissions from combustion of natural gas during phase 1 and phase 2 distillations; the

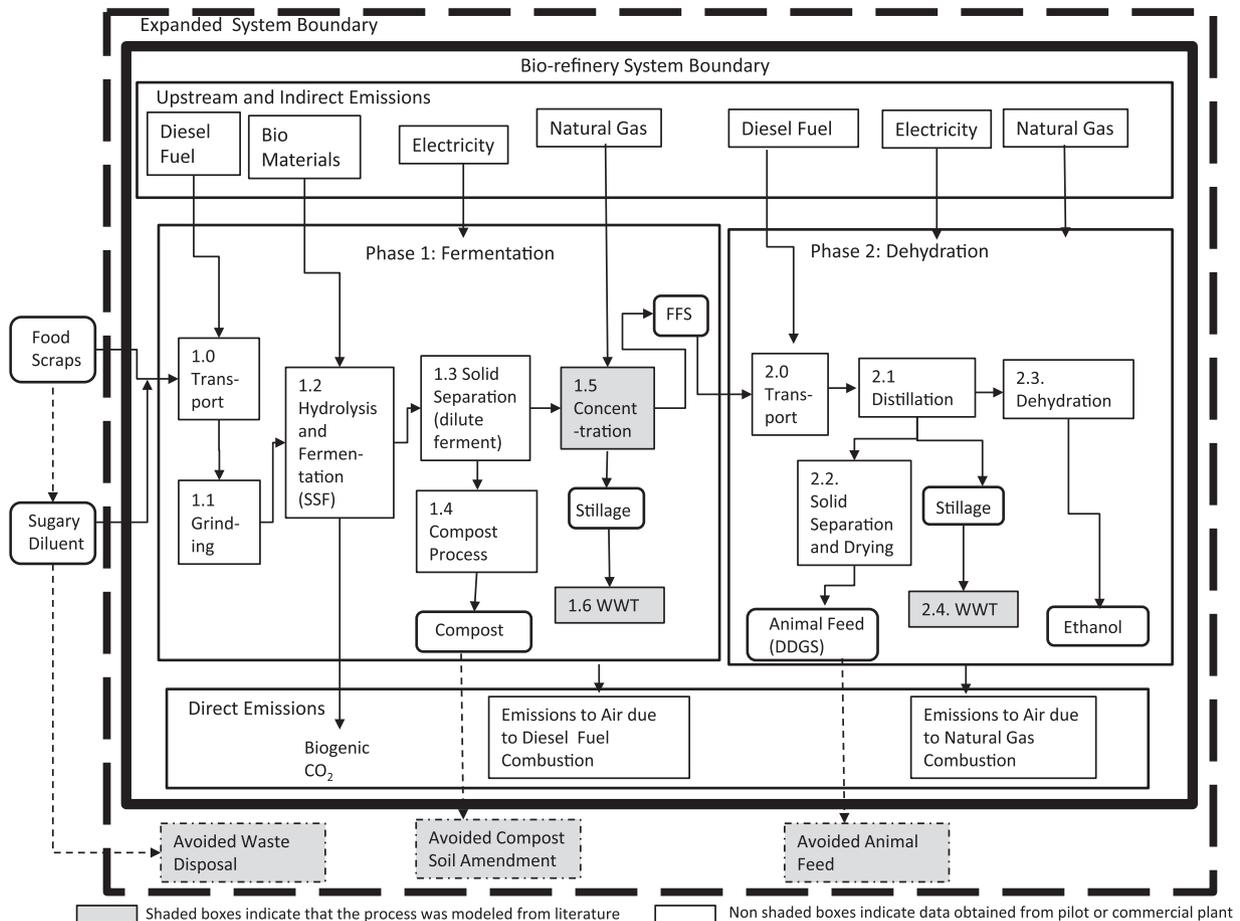


Fig. 1. Ethanol production process and system boundaries. Bolded solid lines indicate the bio-refinery system boundary. Dashed bolded border indicates system expansion to net production process. Dashed arrows and processes indicate production processes for displaced co-products and services. Unit processes in gray are modeled based on the literature. Processes without a background are derived from pilot or commercial data.

production impacts of the material inputs to the process (biocatalysts and nutrients); and the life cycle emissions from diesel fuel required to transport waste material to phase 1 and FFS to phase 2. The impacts associated with the upstream production and construction of the phase 1 and phase 2 plants are not included in this analysis, as they are believed to be negligible per functional unit. Although this is supported by previous studies on corn ethanol, where they represent less than 1% of net GHG emissions [22], verification in a mature, full-scale distributed ethanol system would be worthwhile in the future. Carbon dioxide created during the fermentation process is treated as biogenic and not included in GHG emission inventories [23,24].

2.2.3. Treatment of co-products

A variety of approaches exist for the treatment of co-products in LCA. In accordance with recommended guidelines (ISO14040/44) the system was expanded to model the displacement of competing products by the co-products generated in this process [21]. This is sometimes referred to as consequential LCA.

The bio-refinery process produces two co-products, compost and an animal feed product (analogous to DDGS). Thus the net bio-refinery emissions account for the avoidance of the indirect and direct emissions generated had these co-products been produced through an alternative process. For the compost co-product this consists of the displacement of transportation and processing emissions associated with the alternative production of the compost. The resulting compost co-product is considered to be

equivalent to compost produced by an alternative method and therefore the effects of compost application are considered equivalent and neglected in this analysis.

The treatment of impacts due to DDGS co-products in ethanol production has received much attention as it is shown to have a significant influence on results. A system expansion method is generally considered the most robust and most conservative method of treatment [25]. Accordingly, the emissions are calculated for the nutritionally equivalent quantity of displaced animal feed. These include the indirect and direct emissions associated with the cultivation and production of displaced corn and soy meal. It also includes the net impacts on enteric fermentation due to the relative performance of feed DDGS relative to displaced corn and soy meal [26].

The service of waste disposal generates valuable revenue and is therefore also considered a co-product. Thus the net production emissions of the ethanol include the net bio-refinery emissions and the emissions due to avoided waste disposal. Because the waste feedstock used is diverted from the landfill, the system is expanded to include the avoided emissions associated with transportation of the waste to the landfill, processing of the waste at the landfill, and the net emissions associated with the decay of the waste at the landfill; these emissions are the sum of methane emissions released to the atmosphere, carbon storage within the landfill and avoided grid emissions due to methane captured by the landfill gas recovery system and used to create and displace grid emissions.

2.2.4. Life cycle inventory and impact assessment

Electricity, fuel and materials fluxes are compiled. Emission factors are applied to evaluate these fluxes for the midpoint impact category of global warming potential. The greenhouse gases considered are carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O). GHG emissions are aggregated on a carbon dioxide-equivalent (CO₂e) basis, using the 100-year global warming potential factors for methane and nitrous oxide emissions published by the Intergovernmental Panel on Climate Change [27]. These values are 1 for CO₂, 21 for methane, and 310 for nitrous oxide. Biogenic CO₂ produced in the fermentation process or any of the avoided waste disposal options analyzed is not included in the aggregated impact. (The fluxes and life cycle emission factors are summarized in Table ST3 in Supplementary Materials.)

Emissions due to the transportation of food waste feedstock were calculated using the USLCI v1.6, fuel lifecycle emission factor for freight transport (per t-km) for a single-unit diesel truck, short haul [29]. This includes emissions associated with the provision, transport and distribution of the fuel as well as the combustion of fuel (which is calculated from GREET 1.8 based upon operational data from EPA moves 2010). Calculation of t-km was based upon the weights and distances traveled for the feedstock used in the pilot batch. Deliveries occurred over two days and waste was collected each day via 3 routes from 7 different locations. The routes ranged from 26 to 78 km. A total of 274 km was traveled to transport all of the waste feedstock and an average payload of 0.55 t was calculated (see Supplementary Materials Table ST4). Electricity consumption at phase 1 was estimated based on an inventory of equipment (grinders, augers, pumps and separators), rated or measured current draw, and time of use measurements (see Supplementary Materials). Emissions for electricity consumption were based on the U.S. Average Grid Mix, using the Ecolnvent v2.2 emission factor for electricity, medium voltage, US [28]. Specific biomaterial inputs and quantities were provided by Epiphery LLC and are considered proprietary. However, they were used to calculate the life cycle emissions for biomaterials using factors obtained from the GREET model [26] and the Ecolnvent 2.2 database [28] for the individual biomaterials. Phase 1 natural gas consumption was calculated for the concentration of the dilute ferment using 5 MJ/L anhydrous ethanol [19]. Natural gas emissions from concentration were calculated to account for provision and combustion of the natural gas in an average industrial boiler operating at 85% efficiency using USLCI v1.6 life cycle emissions data [29]. Wastewater treatment at phase 1 is assumed to be performed at an onsite wastewater treatment plant [22]. An emission factor from Ecolnvent v2.2 for the treatment of potato starch effluent at a class 2 wastewater treatment plant is used [28].

Since a phase 2 plant does not exist at this time it was assumed to be 100 km away from the phase 1 plant with FFS transported by a single-unit, diesel-operated truck. Data on natural gas and electricity consumption on a per liter basis to dehydrate FFS to anhydrous ethanol as well as evaporation energy to produce the animal feed product were provided by Merrick and Company which has been operating a similar plant since 1996 (see Supplementary Materials) [30]. This plant processes brewery waste to ethanol and produces 3 M gallons of ethanol per year. The emissions associated with onsite processing of wastewater generated at the phase 2 distillation plant were based upon the emission factor per unit wastewater treated at a corn ethanol plant [22].

Displaced landfill emissions are based on the EPA WARM model using the category of 'food scraps' [31,32]. An adjustment is made based on the dry mass to determine an equivalent mass of food scraps avoided at 70% moisture content. Additionally, a component-specific decay "K-value" of 0.08 was used representing wet landfill conditions to account for additional moisture. The landfill

is modeled to have the current national average LFG recovery system and national average electricity grid mix.

The EPA WARM model was also used to calculate the avoided transportation and processing emissions related to the compost co-product. These are calculated based on the equivalent mass of feedstock required to create compost by an alternative method, using the conversion of 2.1 t of yard waste to create 1 t of compost [33].

The displacement credit for the animal feed co-product is calculated based on the DDGS displacement from the EBAMM v.1.1 model [22,34,35]. The GHG impact is scaled to account for the quantity of feed co-product produced.

3. Results and discussion

3.1. System modeling results

The process modeled here produced 276 liters of anhydrous ethanol, 160 kg of compost and 428 kg of feed from 4.7 wet t of wet food waste. This is equivalent to of 296 L EtOH/dry t feedstock (or 0.23 g EtOH/g dry solids) (Table 1). The theoretical yield for this process is estimated to be 585 L EtOH/dry t of feedstock (calculation in Supplementary Materials). Therefore the co-fermentation process efficiency is estimated to be 54% of theoretical yield.

Previous studies on the conversion of organic MSW to ethanol have produced a broad range of results. Kalogo et al. [17] reported approximately 121 L EtOH/dry t (85 L/wet t) using an acid hydrolysis Gravity Pressure Vessel pilot process using MSW sluff (consisting of food, paper, and yard waste). In contrast, Schmitt et al. [18] modeled a process based on lab experiments using dilute acid hydrolysis pretreatment followed by SSF at 30 °C on a synthetic MSW feedstock (defined as banana peels, cereal, coffee grounds, canned corn, tomato juice and clean hygiene products) and reported a process yield of 469 L/dry t and 74% of theoretical yield. Thus the yield reported here is within the range of reported values on a mass basis. However, it is worth noting that the composition of the co-fermentation substrate has a lower lignocellulosic content than MSW due to the contribution of the sugary diluent, which we would expect to have a higher conversion efficiency.

Table 1

Summary of process inputs, outputs and yields resulting from pilot plant audit and mass balance.

<i>Process inputs</i>	
Organic waste input (wet mass) (kg)	4718
Food scraps (at 70% moisture) (kg)	2309
Syrup diluent (at 90% moisture) (kg)	2409
Biomaterial inputs (kg)	9.23
Estimated dry mass in (kg)	934
<i>Process outputs</i>	
Phase 1: ferment (5.88% ABV) (liters)	4978
Phase 1 estimate: FFS (15% ABV) (liters) ^a	1904
Phase 2 estimate: anhydrous ethanol (EtOH) yield (kg) – calculated from FFS at 15% ABV assuming 96.5% distillation efficiency	276
Phase 2 estimate: DDGS calculated based by mass closure of solids (kg) ^b	428
Phase 2 estimate: stillage calculated from mass balance of phase 2 distillation (liters)	1628
Phase 1 estimate stillage based upon mass balance of distillation phase (liters)	3046
Yield g EtOH/g TS	0.23
Yield liters EtOH/ dry MT	295
Calculated yield liters EtOH @70% moisture (wet tonne)	207
Estimated theoretical yield liters/dry tonne	544
% Theoretical	54%

^a Calculated based upon modeling of the Concentration process. Assuming 0.22% ethanol in stillage.

^b Dry mass In = DDGS + EtOH mass + CO₂mass loss + compost solids.

Table 2
Life cycle GHG Results (gCO₂e/L EtOH) and comparison to corn ethanol and gasoline (gCO₂e/MJ).

	This study	Corn ethanol ^a
Total bio-refinery emissions (gCO ₂ e/L)	1458	
Displaced landfill emissions (8590)		
Net bio-refinery emissions (gCO ₂ e/FU)	(7132)	1608
Reported HV of ethanol (MJ/l)	21	21
Net production process (gCO ₂ e/MJ)	(340)	77
Ethanol distribution (gCO ₂ e/MJ)	1	1
Net produced and distributed (gCO ₂ e/MJ)	(338)	77
Difference between corn EtOH (gCO ₂ e/MJ)	416	0
% Difference improvement between corn EtOH	–554%	0%
g CO ₂ e per MJ of conventional gasoline (CG) produced, distributed, and combusted.	94	94
Percentage difference to CG	–460%	–17%

^a Farrel et al. (2006).

Furthermore, increased conversion of lignocellulosic material is likely to require more inputs and increase production costs. This process differs from other published methods for lignocellulosic SSF in that it functions at lower operating temperatures (20 °C vs. 37 °C) and involves minimal pretreatment. Because the bio-refinery generates revenue from compost, animal feed and waste disposal (tipping fees) as well as ethanol, it is unclear if maximizing ethanol yield would necessarily maximize profits.

3.2. LCA results and analysis

3.2.1. Comparison to corn ethanol and gasoline

The GHG impacts for the process are compared to those of corn ethanol production using a functional unit of 1 liter of ethanol. In order to compare the results to conventional gasoline (CG) results are converted to a MJ basis to account for the difference in performance between ethanol and gasoline (Table 2). These results show a net carbon-negative production process with 553% improvement in GHG impacts relative to corn ethanol and 460% relative to conventional gasoline. This reduction is almost entirely due to the avoided methane emissions that would be incurred by food waste disposal in a landfill. Without the inclusion of avoided landfill impacts, the net bio-refinery emissions (phase 1 and phase 2) show a 9% improvement over commercial corn ethanol production (including agricultural phase impacts).

3.2.2. Contributonal analysis

Table 3 shows the life cycle contributions of the two production phases and landfill avoidance to total process emissions and compares it to corn ethanol emissions. Phase 1 has a larger contribution (1217 gCO₂e/L EtOH) to emissions than phase 2 (241 gCO₂e/L EtOH). This is driven by electricity use (816 gCO₂e/L EtOH) to operate grinding and separating equipment followed by natural gas use for concentration (241 gCO₂e/L EtOH). Compost co-product production provides a small offsetting credit (54 gCO₂e/L EtOH). Phase 2 accounted for only about 22% of process emissions. Electricity use (285 gCO₂e/L EtOH) is again a major driver followed by natural gas for distillation and drying (146 gCO₂e/L EtOH). However, the large credit for animal feed production (260 gCO₂e/L EtOH) reduced emissions for this phase by nearly 50%. The large contribution of electricity and natural gas consumption to process phase emissions indicates that cogeneration of electricity or heat, as is common in cellulosic ethanol processes, may be an opportunity. The life cycle emissions associated with enzymes and other biomaterial inputs (91 gCO₂e/L EtOH) is relatively small due to the small amount of biomaterials used.

Table 3
Contributonal analysis of life cycle GHG impacts (gCO₂e/L EtOH).

This study	gCO ₂ e/L EtOH
Phase 1 emissions	1271
Transportation of waste	109
Electricity consumption	816
Natural gas consumption	249
Biomaterial inputs	91
WWT	6
Avoided compost co product	(54)
Net emissions phase 1	1217
Phase 2 emissions	501
Transportation of FFS	66
Electricity consumption	285
Natural gas consumption	146
WWT	3
Avoided animal feed co product	(260)
Net emissions phase 2	241
Net biorefinery emissions	1458
Displaced landfill emissions	(8590)
Net production emissions	(7132)

The impact of transportation in this two-phase model was also analyzed. It is found to make a noticeable contribution, representing approximately 12% of the process emissions (phase 1 and phase 2 emissions only). Transportation of the FFS to phase 2 (66 gCO₂e/L EtOH) has only 40% of the emissions impact of transportation of feedstock the phase 1 plant (109 gCO₂e/L EtOH). This is due to the reduction in mass transported due to the removal of moisture and solids as a result of the fermentation, separation and concentration processes at phase 1. Furthermore, when avoided waste disposal emissions are considered, the impact of feedstock transport is more than offset by the avoided transportation of waste feedstock to a landfill (calculated using the WARM model). This result is attributed to the lower impact of single unit trucks with only a few collection stops, as compared to modeling heavy waste collection vehicles that make frequent stops. Nonetheless, transportation and in turn the location of the phase 1 and phase 2 plants do impact GHG emissions and will require optimization with process scale-up.

3.2.3. Comparison to other waste-to-ethanol LCAs

Comparisons of life cycle results to other waste-to-ethanol processes are challenging and considerations have been made to provide a meaningful comparison. First, the phase 1 emissions used in this analysis are based on a pilot plant facility operating at 1/15th its intended capacity. Thus it is considered a worst-case scenario since a full-scale production facility will likely benefit from learning curve and scale economy effects. Additionally, treatment of avoided waste disposal in the previous LCAs was inconsistent and highly influenced by feedstock [12,17,18]. Therefore, results for just the production process are compared first and a discussion of treatment of feedstock is presented in the next section. Lastly, differences in processes, co-products, and analysis objectives have also been considered in comparing results. Kalogo et al. [17] reported on MSW-to-ethanol using a dilute acid hydrolysis and Gravity Pressure Vessel technology. Their results show that the classification process to remove inorganic material has a large contribution to emissions (nearly 40%); results are therefore shown with and without classification. Schmitt et al. [18] used the NREL ASPEN model of a dilute acid SSF process on lignocellulosic material. In this system, residual lignin is combusted to generate electricity, offsetting site usage. Chester and Martin [12] perform an Economic Input/Output LCA also using the NREL model, with the objective of comparing the business-as-usual MSW system in California to one of waste-to-ethanol. They do not include waste collection as they rationalize that it would occur in either case.

Despite significant differences in scale and implementation, the results of this process fall within the range of the other studies (Fig. 2). Thus it may be concluded that the GHG impacts of smaller scale and process optimization tradeoffs are offset by less process inputs and the selection of highly degradable, source separated feedstocks.

3.2.4. Food waste vs. MSW

The studies discussed in the previous section have all utilized organics derived from MSW where this study utilizes industrial and retail food waste [12,17,18]. In addition to the reduction in classification required due to source-separated feedstock, the characteristics of the feedstock can have a significant affect on life cycle GHG results. LFG emissions are a function of the rate of decay of the waste and the potential of the waste to generate methane (Fig. 3). Food scraps have a high potential for methane generation as well as a rapid decay rate. Due to the phased implementation of typical LFG recovery systems, rapid decay of food scraps results in net GHG emissions, even with aggressive LFG recovery [31,32]. (Un-captured methane is indicated by the area between the solid lines and the dashed lines in Fig. 3.) In contrast, when Kalogo et al. [17] considered MSW derived organics containing yard waste and paper waste as well as food scraps and Chester and Martin [12] considering MSW including construction /demolition and paper waste, both found that the net avoided GHG emissions flipped from positive with no LFG recovery system to negative with LFG recovery to electricity generation. Food scrap diversion from a landfill always results in positive avoided emissions regardless of the LFG recovery system reducing the sensitivity of the net impacts to landfill technology.

3.2.5. Sensitivity to avoided waste disposal

While this process utilized feedstock that was diverted from the landfill, alternative disposal pathways were also analyzed to determine the impact on our results. For solid waste the alternatives of landfill or industrial composting were modeled. For the diluent, the alternatives of wastewater treatment and land application were modeled in addition to the base case of avoided landfill. Avoided waste disposal emissions are affected by waste composition, technology, environmental conditions and modeling methodology. Data on waste disposal alternatives is limited and emissions factors based upon specific food waste characteristics were not available. In most cases the general category of food waste is modeled. Four scenarios were analyzed and compared to the base case of

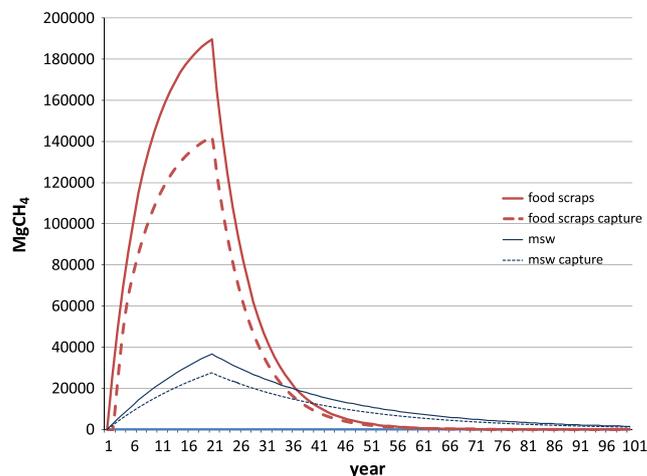


Fig. 3. Methane production rate (m³ CH₄/year). Comparison of methane production (solid lines) and LFG captured (dashed line) for MSW and food scraps over 100 years. Calculated using Land GEM v3.02, based on 1 t of waste. For food scraps: $k = 0.14$, $L_0 = 301$ (m³/dry Mg); MSW: $k = 0.04$, $L_0 = 100$ (m³/dry Mg). Phased-in methane collection: Years 1–2: 0%, Year 3: 50%, Year 4: 70%, Years 5–100: 75% [38,39].

feedstock diverted from landfill and the net emissions of corn ethanol (Fig. 4). Error bars indicate the range of values related to technology and environmental conditions.

Solid waste disposal influenced results more than liquid disposal options. Emissions due to landfilling of waste has the largest magnitude of impact, ranging from 1576 kgCO₂e/wet t food scraps with no LFG recovery to 375 kgCO₂e/wet t food scraps for LFG recovery to electricity. Nevertheless, the net result of diversion of food scraps from a landfill is a significant avoidance in emissions. Thus the net ethanol process remains carbon-negative for all scenarios with landfill avoidance.

When considering diversion of waste from a commercial compost facility, soil carbon storage resulting from the application of the compost is considered along with the emissions incurred due to transportation and processing of the waste per the EPA WARM model [31]. The net application of compost results in the sequestration of –220 kgCO₂e/t waste processed. Therefore diverting waste from the carbon-negative compost process increases net GHG emissions for the waste-to-ethanol process. Scenarios that

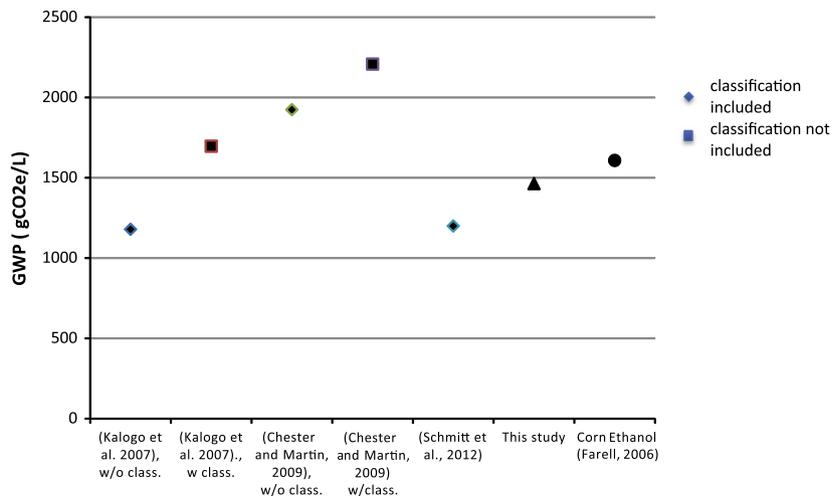


Fig. 2. Comparison of net biorefinery process life cycle GHG results (gCO₂e/L EtOH). This includes the process itself and does not include avoided waste disposal of feedstock or ethanol distribution. This study is compared to MSW to ethanol studies and corn ethanol.

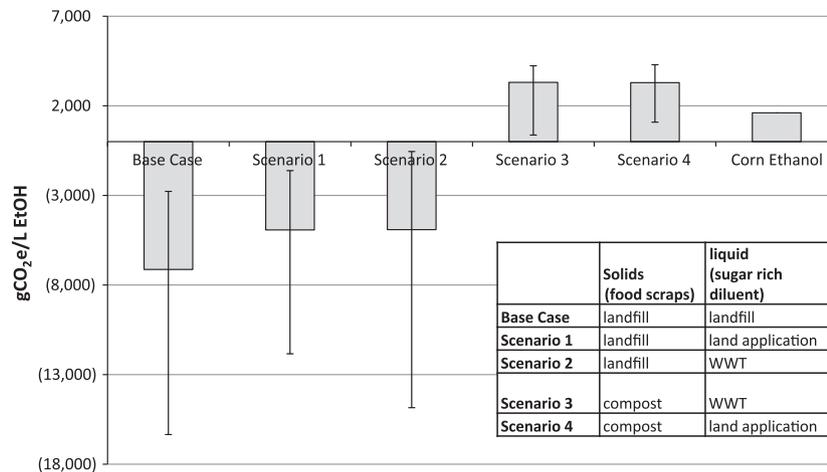


Fig. 4. Sensitivity of results to avoided waste disposal treatment ($\text{gCO}_2\text{e/L EtOH}$). Four scenarios are compared to the base case and to corn ethanol. The base case represents the case reported in this study, where all waste feedstock is diverted from the landfill. The four scenarios consist of waste diverted from either landfill or composting of solids and either wastewater treatment or land application of liquid feedstock and are shown in the inserted table. Error bars indicate the range of results due to technology and environmental conditions.

utilized solid waste diverted from composting resulted in higher net emissions than corn ethanol. However, these results were sensitive to uncertainty in the amount of carbon storage as well as direct CH_4 and N_2O emissions due to the compost process. The amount of carbon storage is affected by soil characteristics and application schedule. A best case carbon storage scenario, resulting in net sequestration of $-331 \text{ kgCO}_2\text{e/kg}$ waste is shown through the lower end of the error bars. Research into CH_4 and N_2O emissions from composting is ongoing and not yet included in the WARM model. While these emissions are considered in the IPCC methodology they are quite small ($4 \text{ g CH}_4/\text{kg}$ wet waste processed) and ($0.3 \text{ g N}_2\text{O}/\text{kg}$ wet waste processed) [23]. A worst case is constructed to include the direct CH_4 and N_2O emissions along with emissions due to fossil fuel used to process the compost, but not including any carbon storage effect. Since in this case, diversion of food waste from the compost process would eliminate these emissions, this represents the lower end of the range shown by the error bars. (Displacement of fossil based fertilizer is not considered in this analysis but could also influence results.) Finally, this study does not take into consideration any difference due to application of compost derived from food waste (as in this study) and compost derived from yard waste as in typical of some commercial compost processes [31].

There is limited U.S. emissions data on wastewater treatment and land application of food processing wastewater. Similar to solid waste, treatment technology, waste characteristics and modeling affect uncertainty in wastewater emissions. A baseline wastewater treatment emission factor of $1.3 \text{ kgCO}_2\text{e}/\text{m}^3$ wastewater (ww) based upon fruit and vegetable processing wastewater was used [36]. However, this process only includes direct CH_4 and N_2O emissions and does not include fossil fuel use or infrastructure, which some studies consider quite large. Therefore data from a range of relevant processes in Ecolnvent v2.2 were used to model uncertainty ranging from $0.51 \text{ kgCO}_2\text{e}/\text{m}^3$ for potato starch wastewater treatment to $83.3 \text{ kgCO}_2\text{e}/\text{m}^3$ ww for treatment of organic wastewater [28]. In all cases wastewater processing results in net emissions, although in some cases quite small, thus avoidance of this process contributes to emission savings for the ethanol production process.

Land application emission factors for food waste were not available. Net emissions due to land application are the result of CO_2 emissions due to spreader fuel consumption as well as the net impact of CH_4 and N_2O emissions, carbon storage and fertilizer

displacement. Similar to composting, the latter impacts are influenced by the soil conditions, waste characteristics and agricultural practices. Land application emissions were modeled to range from $1.21 \text{ kgCO}_2\text{e}/\text{m}^3$ ww due to avoidance of emissions to operate the spreading equipment only to $-8.5 \text{ kgCO}_2\text{e}/\text{kg}$ ww based on studies of manure spreading net impacts [28,37]. Thus diverting waste from compost and land application to the ethanol production process studied herein has the least potential for GHG reduction.

Waste disposal alternatives are driven by many factors including economics, waste characteristics (i.e., solid, liquid, packaged, etc.) as well as market availability. Furthermore, they may vary by type of waste, region and over time for a given waste to energy process. Our results indicate that it is important to understand and model the appropriate waste disposal scenarios to understand the net impact of waste to energy processes.

4. Conclusions

Process yields for co-fermentation of lignocellulosic material with sugar-rich diluent, using SSF with a grinding pretreatment, based upon pilot plant data are reported. Life cycle GHG emissions for the process are comparable to commercial processes studied in the literature. Furthermore when the avoidance of landfill emissions is considered, the process shows a significant improvement over corn ethanol or conventional gasoline with respect to GHG emissions. The results indicate that the use of readily convertible, source-separated commercial or industrial food waste as a feedstock for ethanol offers significant potential for GHG reduction. Furthermore, important to understanding the life cycle impacts of corn ethanol, this study illustrates how feedstock and alternative waste disposal options have important implications in life cycle GHG results for waste-to-energy pathways.

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Appendix A. Supplementary material

Supplementary material associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.apenergy.2014.04.099>.

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