Comparative Life Cycle Assessment of Food Waste Management Systems: An Evaluation of a Novel Anaerobic Digestion Technology

by

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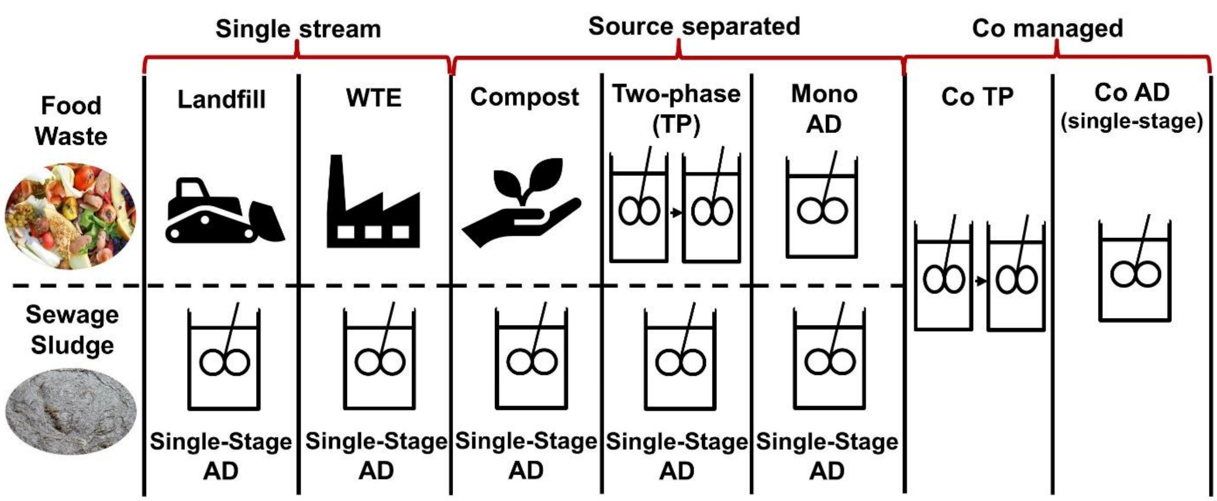
Abstract

Resource recovery has the potential to capture energy and nutrients from municipal organic wastes and convert them to useful products while reducing environmental impacts. We developed a novel two-phase anaerobic dynamic membrane bioreactor (AnDMBR) that can achieve a high overall methane yield at a low hydraulic retention time to increase energy recovery from food waste and sewage sludge. We utilized life cycle assessment to evaluate the environmental impacts of the novel two-phase system in comparison to conventional food waste and sewage sludge management options in the United States. Co-management of food and sewage sludge waste streams through co-digestion was shown to maximize energy recovery and had a net global warming benefit, while minimizing other environmental impacts. The net impacts of anaerobic digestion (AD) systems were most sensitive to the background electric grid mix, as the benefits of displacing grid electricity with electricity produced from biogas decline when fossil fuel sources are replaced with renewable energy sources like solar and wind. Upgrading biogas to renewable natural gas to displace the use of fossil natural gas for other energy requirements that are difficult to decarbonize may sustain the environmental benefits of utilizing AD to produce biogas from municipal organic waste streams in the long-term.



Keywords: life cycle assessment, food waste, organic fraction of municipal solid waste, anaerobic digestion, co-digestion, anaerobic dynamic membrane bioreactor

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List of Abbreviations

anaerobic digestion: (AD)

anaerobic dynamic membrane bioreactor: (AnDMBR)

combined heat and power: (CHP)

global warming potential: (GWP)

greenhouse gas: (GHG)

hydraulic retention time: (HRT)

landfill gas: (LFG)

life cycle assessment: (LCA)

recirculating anaerobic dynamic membrane bioreactor (R-AnDMBR)

renewable natural gas: (RNG)

solids retention time: (SRT)

United States: (U.S.)

volatile fatty acid: (VFA)

volatile solids: (VS)

waste-to-energy: (WTE)

water resource recovery facility: (WRRF)

Introduction

Of the 63 million tons of food waste generated from residential, commercial, and institutional sectors in the United States in 2018, 56% was sent to landfills (USEPA, 2020). Decomposition of organics in municipal landfills are the third-largest source of anthropogenic methane emissions in the United States, accounting for 17% of all anthropogenic emissions (USEPA, 2021a). Landfills also account for about 70% of all greenhouse gas (GHG) emissions from the U.S. waste sector (USEPA, 2021a).

Processes have been developed to capture energy and nutrients from food waste and convert them to useful products while reducing environmental impacts. For example, landfills equipped with gas collection and waste incineration at modern waste-to-energy (WTE) facilities are capable of capturing energy from direct combustion of waste or landfill gas (LFG) to generate electricity and thermal energy (Ma and Liu, 2019; Spokas et al., 2006). About 20% of landfills in the U.S. recover energy from captured LFG (USEPA, 2021b), while WTE accounts for 12% of all food waste disposal (USEPA, 2020). Source separated food waste management systems like composting and anaerobic digestion (AD) recover resources including energy (in the form of biogas) and nutrients (in the form of compost or digester biosolids).

Life-cycle assessment (LCA) is a framework that has been utilized to systematically evaluate the environmental impacts and compare management options for food waste. Several studies have conducted a comparative LCA of different management systems using a system boundary that encompasses the environmental benefits of useful products and energy produced from food waste. Specifically, treatment of food waste with AD has been shown to have a net global warming benefit and lower GHG impacts than landfill, WTE, and composting under typical operating conditions (Eriksson et al., 2015; Evangelisti et al., 2014; J. W. Levis and Barlaz, 2011). However, when adjusting for local treatment requirements, or varying background offset factors, the environmental profile of each treatment system may change. For example, several studies concluded that the environmental benefits of AD for food waste management are mainly due to offset emissions from avoided electricity generation from fossil fuels (Becker et al., 2017; Evangelisti et al., 2014; Hodge et al., 2016; J. W. Levis and Barlaz, 2011; Thyberg and Tonjes, 2017). Therefore, declining use of fossil fuels for electricity production will decrease the environmental benefits of using combined heat and power (CHP) for energy recovery from AD.

Consequently, it is important to evaluate the sensitivity of low-carbon background electricity generation on AD's environmental performance.

Despite AD's superior environmental performance in comparison to other common food waste disposal methods, only 8.3% of food waste was sent to anaerobic digesters in 2018 (USEPA, 2020). Reasons behind the low uptake of this technology for food waste management include concerns about AD stability due to the low buffering capacity of food waste and potential rapid pH fluctuations it can cause, which limits the volumetric loading rate of the system. A stable AD process requires long retention times (20 – 40 days) for the growth of important microbial populations like methanogens (Braguglia et al., 2018; Fonoll et al., 2016; Kong et al., 2016; Mata-Alvarez et al., 2000; Pramanik et al., 2019; Xu et al., 2018). For reactor operations, a long retention time requires more space and higher operating costs that are not practical for small-scale applications.

Co-digestion of sewage sludge with food waste at a water resource recovery facility (WRRF) can improve the overall energy and nutrient recovery from both waste streams. The addition of food waste increases the combined feedstock's C:N ratio to improve biogas production, and additional alkalinity provided by sewage sludge can improve AD stability (Bolzonella et al., 2006; Macintosh et al., 2019; Mata-Alvarez et al., 2014; Shen et al., 2015). Several food waste management LCAs have considered the co-management of food waste with municipal sewage sludge to assess the environmental benefits of co-digestion (Becker et al., 2017; Edwards et al., 2018; Lee et al., 2020; Tong et al., 2019). These studies showed that co-digestion of sewage sludge and food waste had a lower net global warming potential than conventional food waste and sludge treatment options because the synergistic effects of co-digestion improve biogas production, which can offset a greater amount of fossil electricity.

To advance the use of AD for food waste resource recovery, we have developed a novel two-phase anaerobic dynamic membrane bioreactor (AnDMBR) that enhances hydrolysis (increasing biogas yield and volatile solids (VS) destruction) and operates under stable conditions at short hydraulic retention times (HRTs) (< 10 days) (increasing biogas production rate). Both reactor phases promote the formation of a "dynamic membrane" which enhances biomass retention to effectively decouple the solids retention time (SRT) from the HRT. Therefore, the system can operate at a low HRT, avoiding high space requirements, but at SRTs favorable for methanogens growth and substrate degradation (SRT < 20 days). The first-phase

AnDMBR "rumen reactor" was designed to mimic the first stomach of ruminant animals to enhance hydrolysis and acidogenesis for volatile fatty acid (VFA) production (Fonoll et al., 2019). The VFA rich permeate from the first-phase is fed to the second-phase recirculating anaerobic dynamic membrane bioreactor (R-AnDMBR). The second-phase reactor enhances methanogenesis and VFA degradation by recirculating the permeate through the dynamic membrane (Fairley et al., 2019).

Through bench-scale experiments with flow-through reactors and food waste as a substrate, we have achieved a high overall methane yield of 0.5 kg kg^{-1} VS_{fed} also attained by conventional digesters (Nagao et al., 2012), but with a reduction in HRT by over 70%. The two-phase system operated with a total HRT of only 6 days. The decrease in overall HRT from between 20 – 40 days for stable conventional AD systems to 6 days for the two-phase AnDMBR system has the potential to dramatically decrease the space requirements for AD, which could expand the use of AD for food waste management. Using this technology for co-digestion of food waste with sewage sludge at existing WRRFs may also increase energy and nutrient recovery from both waste streams. However, it is unclear what the true environmental impacts and potential benefits of this technological innovation are without evaluating the flows of the system's entire lifecycle and comparing it to other conventional management systems.

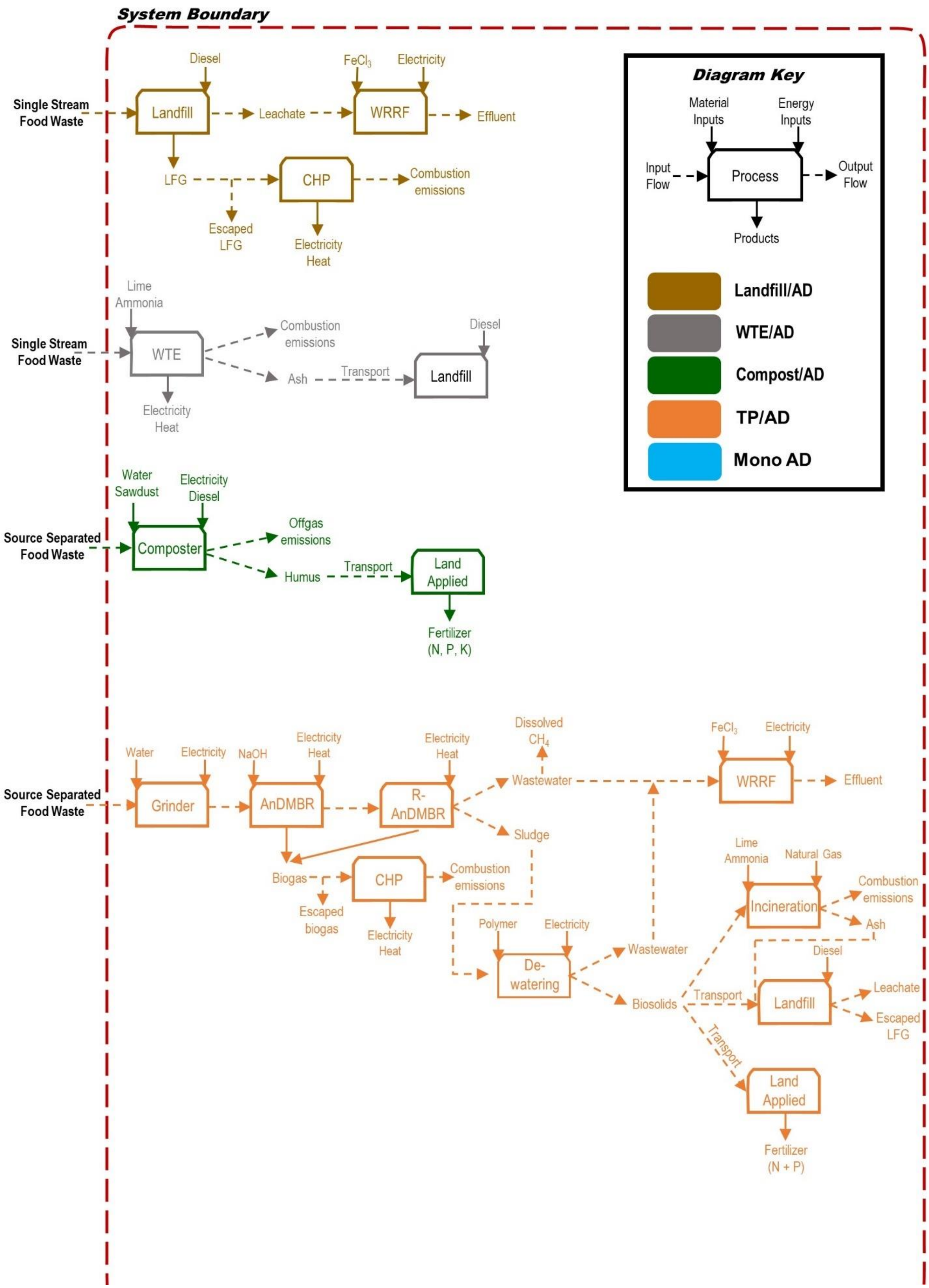
We utilized LCA to evaluate the environmental impacts of the two-phase system in comparison to four typical food waste management systems in the United States. We evaluated both single-stream management systems where food waste is not separated from residual wastes (i.e. landfill and WTE) and source separated management systems where food waste is collected and processed separately from residual wastes (i.e. compost, novel two-phase, and conventional AD). A system expansion to include the treatment of sewage sludge was also performed to consider the co-management of food waste and sewage sludge for the two-phase system and conventional AD. In total, seven waste management systems were evaluated: (1) landfilling of food waste and AD of sludge, (2) WTE of food waste and AD of sludge, (3) composting of food waste and AD of sludge, (4) two-phase of food waste and AD of sludge, (5) mono-digestion of food waste and sludge, (6) co-digestion of food waste and sludge using two-phase, and (7) co-digestion of food waste and sludge using conventional AD. We also evaluated the effects of varying parameters identified as typical for waste management operations including food waste diversion rates for source separated management systems, and the impact of various generation sources for grid electricity. The objective of this paper is to evaluate both current and emerging food waste management systems to demonstrate which technologies may reduce the environmental impacts of waste treatment and increase resource recovery considering a variety of operating conditions. Baseline results depict environmental impacts under typical operating conditions, while sensitivity and uncertainty analysis highlight parameters that most affect the environmental performance of each waste management system.

Materials and methods

Functional unit and system boundary

The functional unit for this study was the treatment of twenty years' worth of food waste and sewage sludge generated by a U.S. municipality of 50,000 people. Per capita food waste generation was assumed to be 0.10 kg VS person^{-1} day^{-1} and sewage sludge generation was 0.07 kg VS person^{-1} day^{-1} (see Table A1). The system boundary included the treatment activities from the gate of the treatment system and ended at the point at which waste was either disposed or converted to a useful product (Figure 1 and Figure A1). Collection and processing of source separated food waste has been shown to have relatively low impacts (Hodge et al., 2016; Thyberg and Tonjes, 2017), and was not considered in this study. Infrastructure impacts were also not considered as construction-phase environmental impacts have been shown to be negligible in comparison to operational-phase impacts for waste management systems (Lee et al., 2020; Rahman et al., 2016).

Useful products generated within each system were considered to replace similar products on the market. For example, electricity generated from combustion processes (i.e. CHP and WTE) was used to meet the energy requirements of the treatment system and excess electricity was assumed to be sold to offset electricity from the 2018 U.S. grid mix. Thermal energy generated from combustion processes (i.e. CHP and WTE) was used to meet the heating requirements of the treatment system and excess heat was assumed to be captured and sold to offset thermal energy generation from fossil natural gas combustion. Biosolids generated from the treatment of digester sludge was modeled to meet U.S. EPA Class B standards (i.e., digester SRT < 15 days at temperatures between 35 - 55°C) (Walker et al., 1994). Land-applied biosolids offset the use of 9.0×10^{-3} kg N fertilizer and 1.2×10^{-2} kg P fertilizer per kg VS (Hospido et al., 2010). Finished compost offset the use of 7.2×10^{-3} kg N fertilizer, 5.0×10^{-3} kg P fertilizer, and 1.0×10^{-2} kg K fertilizer per kg dry compost (Levis and Barlaz, 2013).


System Boundary
Single Stream Food Waste
Diesel
Landfill
Leachate
FeCl3
Electricity
WRRF
Effluent
LFG
CHP
Combustion emissions
Escaped LFG
Electricity Heat
Single Stream Food Waste
Lime Ammonia
WTE
Combustion emissions
Ash
Transport
Diesel
Landfill
Electricity Heat
Source Separated Food Waste
Water Sawdust
Electricity Diesel
Composter
Offgas emissions
Humus
Transport
Land Applied
Fertilizer (N, P, K)
Source Separated Food Waste
Water
Electricity
Grinder
NaOH
Electricity Heat
AnDMBR
Electricity Heat
R-AnDMBR
Dissolved CH4
Wastewater
Sludge
Biogas
Escaped biogas
CHP
Combustion emissions
Electricity Heat
FeCl3
Electricity
WRRF
Effluent
Polymer
Electricity
De-watering
Wastewater
Biosolids
Lime Ammonia
Natural Gas
Incineration
Combustion emissions
Ash
Diesel
Transport
Landfill
Leachate
Escaped LFG
Transport
Land Applied
Fertilizer (N + P)
Diagram Key
Material Inputs
Energy Inputs
Input Flow
Process
Output Flow
Products
Landfill/AD
WTE/AD
Compost/AD
TP/AD
Mono AD

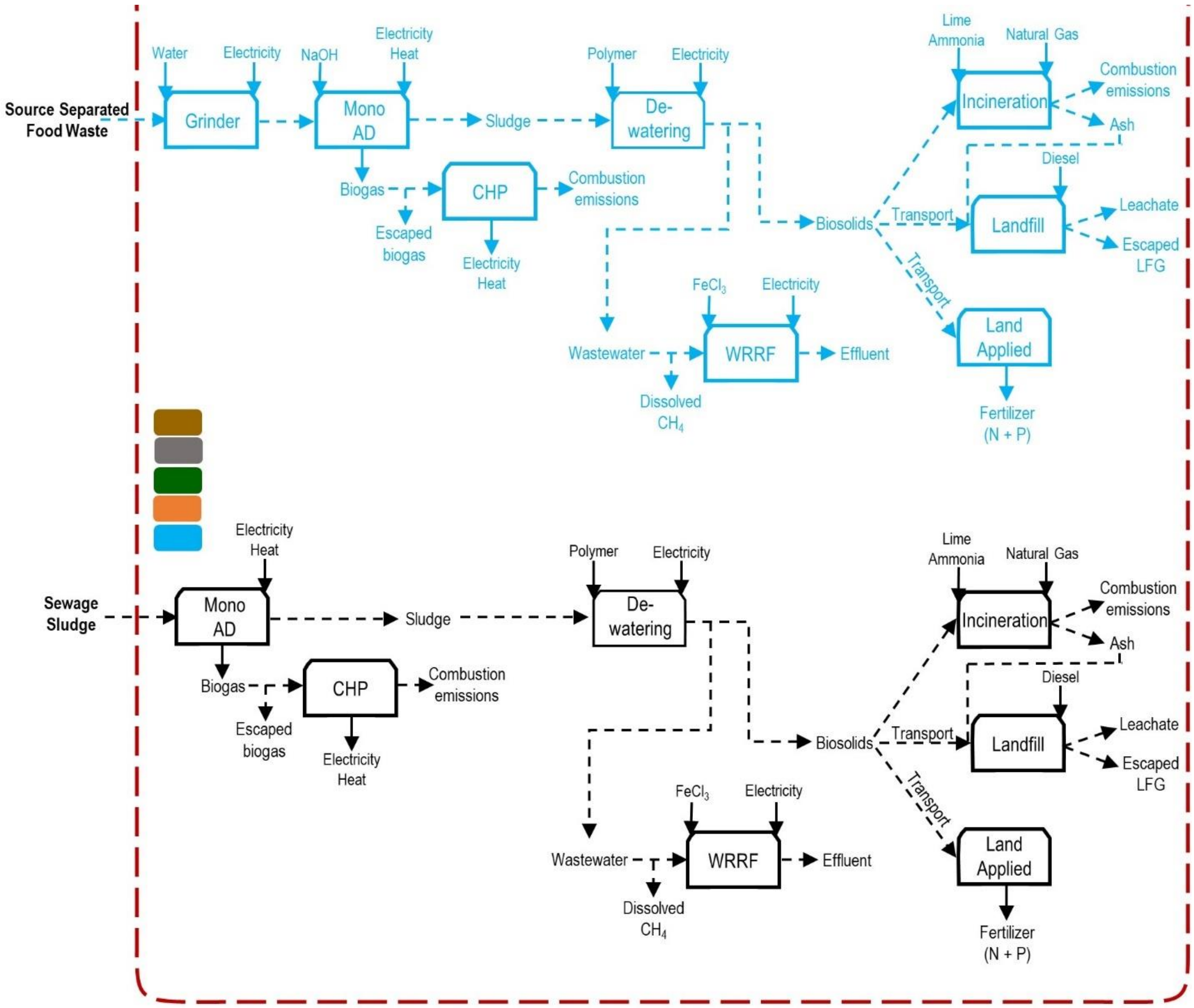


Figure 1. System boundary of single stream (landfill, WTE) and source separated (compost, two-phase digestion, mono-digestion) food waste management systems. Sewage sludge is managed independently of food waste for each waste management system as shown in black. Co-managed systems (Co TP and Co AD) can be found in the Appendix.

Impact assessment

Life-cycle impacts considered for this evaluation were selected based on their relevance as indicators for both global and regional environmental impacts. The five impact categories considered were: 100-year global warming potential (GWP), measured in kg CO_2 eq; photochemical smog formation potential, measured in kg O_3 eq; atmospheric acidification potential, measured in kg SO_2 eq; water eutrophication potential, measured in kg N eq; and respiratory pollutants, measured in PM 2.5 eq. All impacts were calculated using the U.S. EPA Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI), version 2.1. Emissions of biogenic CO_2 were not included in accordance with the IPCC 2006 guidelines for combustion or decay of short-lived biogenic material (IPCC, 2006). Emissions data for grid electricity, fuel use, and other energy and material inputs and products generated within the system boundary were taken from GREET_1_2019 developed by Argonne National Laboratory (UChicago Argonne, 2019) and U.S. LCI (NREL, 2012) (see Table A7).

System descriptions

Each waste management system included the processes for treatment of food waste and sewage sludge. The co-digestion systems (Co AD and Co TP) considered co-management of food waste and sewage sludge while all other food waste management systems considered sewage sludge to be managed separately using conventional mono-digestion. All system models considered biogas and LFG to be utilized on-site with a CHP unit to generate electricity and thermal energy. We selected an average biogas capture efficiency of 96% for AD systems (Liebetrau et al., 2017) and used U.S. average factors from an analysis by Levis & Barlaz for LFG utilization (51.6% of LFG captured for energy recovery, 16.5% flared or oxidized to neutral biogenic CO_2, and the remaining 31.9% escaped to the atmosphere) (Levis and Barlaz, 2014). Digester sludge was processed to form biosolids, which were assumed to be disposed according to aggregate biosolids practices in the United States (51% land applied, 33% to landfill, 16% to incineration (USEPA, n.d.)). All system models considered digester permeate and/or sludge supernatant to be sent to a WRRF with nutrient removal that could achieve a final effluent quality of 8 mg L^{-1} total N and 1 mg L^{-1} total P (details provided in the Appendix). Results for modeling wastewater treatment with enhanced nutrient removal to achieve a final effluent quality of 1 mg L^{-1} total N and 0.1 mg L^{-1} total P are provided in Table A8 and Figure A2.

Food waste was comingled with residual wastes for the single stream food waste management systems (Landfill/AD and WTE/AD). Landfill activities included the energy inputs needed to operate the landfill, capture and utilization of LFG for power and heat generation, treatment of landfill leachate, and emissions from escaped LFG. The landfill methane yield of food waste was considered as 0.4 m^3 CH_4 kg^{-1} dry food waste and methane landfill emissions were considered over a 100-year time horizon to encompass all of the potential emissions from decaying waste (Levis and Barlaz, 2014). It was assumed that 1 kg of food waste would produce 2×10^{-4} m^3 of landfill leachate (Kurniawan and Lo, 2009), which is captured as wastewater and sent to a WRRF. For the WTE system, the material inputs required for air pollution controls and estimates of stack emissions were adapted from the model derived by Harrison et al. (Harrison et al., 2000). Energy released from waste incineration was recovered to generate electricity and thermal energy that was exported to the grid. Bottom ash from incineration was estimated to be 0.018 kg ash kg^{-1} food waste (Hegde and Trabold, 2019). Disposal of bottom ash was modeled as being transported to an ash landfill where it was assumed to be inert.

Source separated system models (Compost/AD, TP/AD, and Mono AD) assumed food waste had been presorted and was free of contaminants. The compost model used was adapted from the model derived by Levis and Barlaz (Levis and Barlaz, 2013). Compost humus was produced by mixing water and sawdust with food waste, and energy inputs were needed to operate the composting facility. Output flows from the compost facility included air emissions from curing offgases (CH_4, N_2O, NH_3, and VOCs). The finished humus was transported to be land applied to offsets the use of mineral fertilizers.

The two-phase mono-digestion system model required food waste to be mixed with water to create a slurry with a 3.0% total solids concentration to achieve an organic loading rate of 40 kg VS m^{-3} d^{-1} for the first-phase digester and 4 kg VS m^{-3} d^{-1} for the second-phase digester. To account for the operational uncertainty of the loading rate for the novel two-phase system, the solids feed rate was varied between 1 to 5% as part of the uncertainty analysis. NaOH was added as a pH buffer to the first-phase digester at a rate of 0.04 kg dry NaOH kg^{-1} food waste. The two-phase biogas methane yield was 0.50 m^3 CH_4 kg^{-1} VS_{fed}. Methane emissions from both fugitive biogas and dissolved methane from the reactor permeate were considered. Reactor permeate was considered wastewater and sent to a WRRF for treatment.

Conventional mono-digestion of food waste was modeled as a simple mesophilic CSTR, where food waste was mixed with water to achieve a 7.5% total solids concentration. The AD model included the use of NaOH to counteract acidic pH fluctuations due to rapid accumulation of VFAs which was added at a rate of 0.04 kg dry NaOH kg^{-1} food waste. The digester was assumed to have a biogas methane yield of 0.40 m^3 CH_4 kg^{-1} VS_{fed} (Holliger et al., 2017). Conventional mono-digestion of sewage sludge was modeled as a simple mesophilic CSTR fed with a 4.0% solids concentration and biogas methane yield of 0.25 m^3 CH_4 kg^{-1} VS_{fed} (Metcalf et al., 1991).

To account for the operational benefits of co-digestion, the two-phase and conventional AD systems were also modeled considering the co-management of food waste and sewage sludge (Co TP and Co AD). Co-digestion was assumed to eliminate the need for water and NaOH, as sewage sludge acts as a sufficient dilutant and pH buffer. Both Co AD and Co TP were also assumed to be fed a higher percent solids feed than their respective mono-digestion systems. Assuming 100% food waste diversion, the baseline systems were modeled as being fed an influent with a 7% solids concentration (55% food waste, 45% sewage sludge by dry mass). Co-digestion was assumed to improve the methane yield to levels achieved for mono-digestion of food waste (Holliger et al., 2017). For Co AD, methane yield was assumed to be 0.40 m^3 CH_4 kg^{-1} VS_{fed}, and for Co TP methane yield was assumed to be 0.50 m^3 CH_4 kg^{-1} VS_{fed}. For the Co TP system, sludge from the first-phase reactor produced from digesting food waste and sewage sludge would not meet current EPA Class B biosolids requirements because of the reactor's low SRT (Walker et al., 1994). Therefore, biosolids produced from Co TP were modeled as being sent only to landfill (67%) and incineration (33%).

Uncertainty and sensitivity analysis

A Monte Carlo analysis (10,000 simulations) was performed to account for data uncertainty within the LCA model (Table 1). The majority of uncertainty parameters were associated with the anaerobic digestion process models. A triangular distribution was assumed for uncertainty parameters with a likely midpoint value, while a uniform distribution was used in the absence of a midpoint estimate.

A sensitivity analysis was conducted to evaluate each system's global warming performance depending on the (i) percentage of food waste diverted from landfill, and (ii) electric grid generation mix. Source separated food waste management systems were evaluated assuming a food waste diversion percentage of 0%, 20%, 40%, 60%, 80%, and 100% with remaining food waste being sent to landfill. Landfill and WTE were not evaluated for varying degrees of food waste diversion as neither system would require food waste to be separated from the residual waste stream. The GHG intensity of electricity consumed and excess electricity generated was varied based on generation type. Generation sources evaluated included coal, natural gas, the U.S. national grid mix (used for baseline modeling), and solar PV. The fuel-cycle GHG emissions for electricity corresponding to each source were 1.05, 0.50, 0.48, and 0 kg CO_2e kWh^{-1} respectively (UChicago Argonne, 2019).

Table 1. Uncertainty parameters used for Monte Carlo analysis.

Uncertainty Parameter	Waste Management System	Baseline Value	Low Value	High Value	Distribution
Digester solids feed (%)	TP	3	1	5	Uniform
	Co TP	7	3	7	
	Mono AD (FW)	7.5	5	10	
Digester methane yield (m^3 CH_4 kg^{-1} VS fed)	TP	0.5	0.4	0.5	Uniform
	Co TP	0.5	0.4	0.5	
	Mono AD (FW)	0.4	0.3	0.45	
	Co AD	0.4	0.3	0.45	
Biogas capture efficiency (%)	All	96	93	99.9	Triangular
Biosolids to land application (%)	TP	51	0	100	Uniform
	Mono AD (FW)	51	0	100	
	Mono AD (SS)	51	0	100	
	Co AD	51	0	100	
Usable heat output (%)	All	100	0	100	Uniform
Distance to landfill (km)	All	32	10	120	Triangular
Distance to land application (km)	TP	32	10	120	Triangular
	Mono AD (FW)	32	10	120	
	Mono AD (SS)	32	10	120	
	Co AD	32	10	120	
	Compost	32	10	120	
LFG utilization (%)	All	51.6	22.2	65.4	Triangular

Results and discussion

Environmental impacts of baseline system models

The co-digestion systems which treat food waste and sewage sludge together maximized resource recovery and had the greatest decrease in GWP impacts in comparison to Landfill/AD (Table 2 and Figure 2). Co AD ranked within the top three lowest net impacts for all environmental emissions considered, and Co TP ranked within the top two lowest net impacts. Co AD had net environmental benefits for GWP and acidification, and Co TP had net environmental benefits for GWP, smog, respiratory, and acidification. The sum of offset emissions from net electricity and thermal energy generation for Co TP and Co AD (-7.45×10^7/-

5.58×10^7 kg CO_2e) were about 1.5 times greater than each system's GWP burdens. Offsetting the use of fossil fuels for electricity and thermal energy generation was the primary benefit for minimizing net environmental burdens for all environmental emissions. Both co-digestion systems had lower net impacts than their respective mono-digestion systems because of the synergistic benefits of co-digesting sewage sludge with food waste. Both TP/AD and Mono AD had higher environmental burdens because of the emissions related to the production of NaOH used for pH control. Likewise, the increase in biogas production from co-digestion resulted in a 28% and 18% increase in net electricity production for Co TP and Co AD, which increased the emissions offset by both co-digestion systems.

Table 2. Baseline environmental impacts of waste management systems. All food waste is assumed to be diverted to each respective management system.

	Waste Management Systems						
	Landfill/ AD	**WTE/ AD**	**Compost/ AD**	**TP/ AD**	**Mono AD**	**Co TP**	**Co AD**
GWP ($kg\ CO_2e$)	6.83E+07	-1.32E+07	3.02E+07	1.67E+07	1.08E+07	-2.43E+07	-1.93E+07
Smog ($kg\ O_3$ eq)	2.31E+05	1.56E+06	9.45E+05	6.63E+05	9.93E+05	-4.12E+05	6.15E+05
Respiratory (kg PM2.5 eq)	4.36E+02	1.57E+04	9.98E+03	7.00E+03	9.45E+03	-1.21E+03	5.88E+03
Acidification ($kg\ SO_2$ eq)	-2.67E+04	7.13E+04	1.74E+05	-8.24E+03	2.05E+04	-1.05E+05	-2.84E+04
Eutrophication (kg N eq)	1.34E+04	3.69E+04	8.48E+04	6.63E+04	3.12E+04	1.37E+04	1.87E+04
	Higher Impact						**Lower Impact**

Landfill/AD had the greatest net GWP impact among all systems studied. Fugitive methane emissions accounted for 1.02×10^8 kg CO_2e or 92% of the GWP burdens. Net energy production displaced only about 38% of the system's GWP burdens. However, Landfill/AD ranked among the top three lowest net impacts for all other environmental emissions. WTE/AD ranked in the top three lowest impacts for GWP with a net benefit (-1.32×10^7 kg CO_2e), but ranked among the highest for all other environmental impacts. Air pollutants associated with direct air emissions from incineration and production emissions for the materials needed for pollution controls (i.e. lime and ammonia) accounted for 55 – 85% of the system's burdens across all impacts. Compost/AD had the second highest GWP impact (3.02×10^7 kg CO_2e) among all systems. Emissions offsets associated with land applying compost humus and AD biosolids only accounted for -2.60×10^6 kg CO_2e, enough to offset only 6% of the system's total GWP burden

(4.29×10^7 kg CO_2e), mirroring other studies that found minimal emission reductions from offsetting the use of mineral fertilizers (Hodge et al., 2016; Thyberg and Tonjes, 2017). Among all systems, land application of biosolids offset less than 5% of each system's total burdens across all environmental impacts. Offgas emissions from the composter contributed between 40 – 88% of Compost/AD burdens for smog, acidification, eutrophication, and respiratory emissions, findings similar to other studies (Becker et al., 2017).

Challenges and opportunities for the two-phase system

The two-phase system's dynamic membrane is able to retain biomass to reduce sludge volume in comparison to conventional AD. This was shown to be beneficial for reducing emissions associated with hauling biosolids and disposing biosolids to landfill or incineration. However, significant emissions were associated with the treatment of nutrients from food waste discharged in the reactor's permeate. For the two-phase system, over 75% of the influent N and P is discharged as permeate. Using model calculations, we estimate the total nutrient load from wastewater produced from reactor permeate and sludge dewatering to be 750 kg N day^{-1}/65 kg P day^{-1} for TP/AD and 950 kg N day^{-1}/110 kg P day^{-1} for Co TP. Conversely, the total nutrient load from wastewater generated from the Landfill/AD system was estimated to be only 15 kg N day^{-1}/ 35 kg P day^{-1}, as the majority of N and P from food waste is retained in the landfill with only a small amount being discharged as leachate.

Wastewater from the two-phase system accounted for over 75% of the eutrophication burdens for the TP/AD and Co TP systems when modeling basic nutrient removal for wastewater treatment (Figure 2). Eutrophication impacts can be attributed to the volume of wastewater that was treated and discharged. Decreasing the solids feed by diluting the feed slurry will generate a higher volume of reactor permeate and consequently increase eutrophication impacts. Conversely, if the two-phase system can operate at a higher loading rate and thus a higher solids feed, the volume of wastewater generated will decrease, decreasing eutrophication impacts. Eutrophication impacts from wastewater treatment can also be mitigated by utilizing enhanced nutrient removal to produce a final effluent with a much lower concentration of N and P per volume of wastewater discharged (see Figure A2). However, based on our wastewater treatment model using MeOH and $Al_2(SO_4)_3$ for enhanced N and P removal, we found that there were tradeoffs with acidification and respiratory impacts. Emissions attributed to the production of MeOH and $Al_2(SO_4)_3$ lead to a three- to ten-fold increase in acidification and respiratory burdens for TP/AD and Co TP. Directly recovering the nutrients in the two-phase permeate by using it as a liquid fertilizer could lead to significant reductions in eutrophication, acidification, and respiratory impacts associated with wastewater treatment and nutrient removal. Reclaimed water for use in irrigation typically only requires filtration and secondary disinfection (USEPA, 2012), and nutrients in the finished reclaimed water could offset the use of fertilizer. Future implementation of the two-phase system should consider opportunities to utilize the permeate as reclaimed water to reduce impacts associated with wastewater nutrient removal and to beneficially recover N and P.

Uncertainty of fugitive methane impacts

Escaped biogas and LFG accounted for 90% or more of fugitive methane, whereas dissolved methane from digester wastewater accounted for 10% or less of total fugitive methane emissions for all systems. Biogas capture efficiency for biogas plants has been reported to range from between 93% to virtually 100% (Liebetrau et al., 2017). Escaped biogas from AD systems directly increases GWP burdens and indirectly decreases GWP benefits from the loss of potential energy generation from biogas combustion. Decreasing biogas capture efficiency from 96% to 93% increased net GWP impacts by over 75% for TP/AD, Mono AD, Co TP, and Co AD. In practice, ensuring a high capture efficiency of biogas from AD systems is of particular importance to maximize resource recovery and mitigate GWP impacts.

Landfill gas capture efficiency is highly dependent on the age of the landfill cell and cell cover type (Hodge et al., 2016; Levis and Barlaz, 2014). The LFG capture efficiency may range from only 35% for an active cell during initial waste burial to 90% for a cell with a geomembrane final cover (Spokas et al., 2006). As food waste is readily degradable, methane emissions from food waste are difficult to capture because food waste is degraded before an effective cover can be applied to the landfill cell. An analysis by Levis & Barlaz concluded that the lifetime utilization of LFG from food waste for energy generation may vary between 22.2 – 65.4% depending on the waste decay rate and landfill regulatory requirements (Levis and Barlaz, 2014). Compared to the baseline Landfill/AD model with an average LFG utilization of 51.6%, a poorly managed landfill with a LFG utilization rate of only 22.2% had a two fold increase in net GWP impacts, whereas a state of the art landfill with a LFG utilization rate of 65.4% decreased net GWP impacts by half.

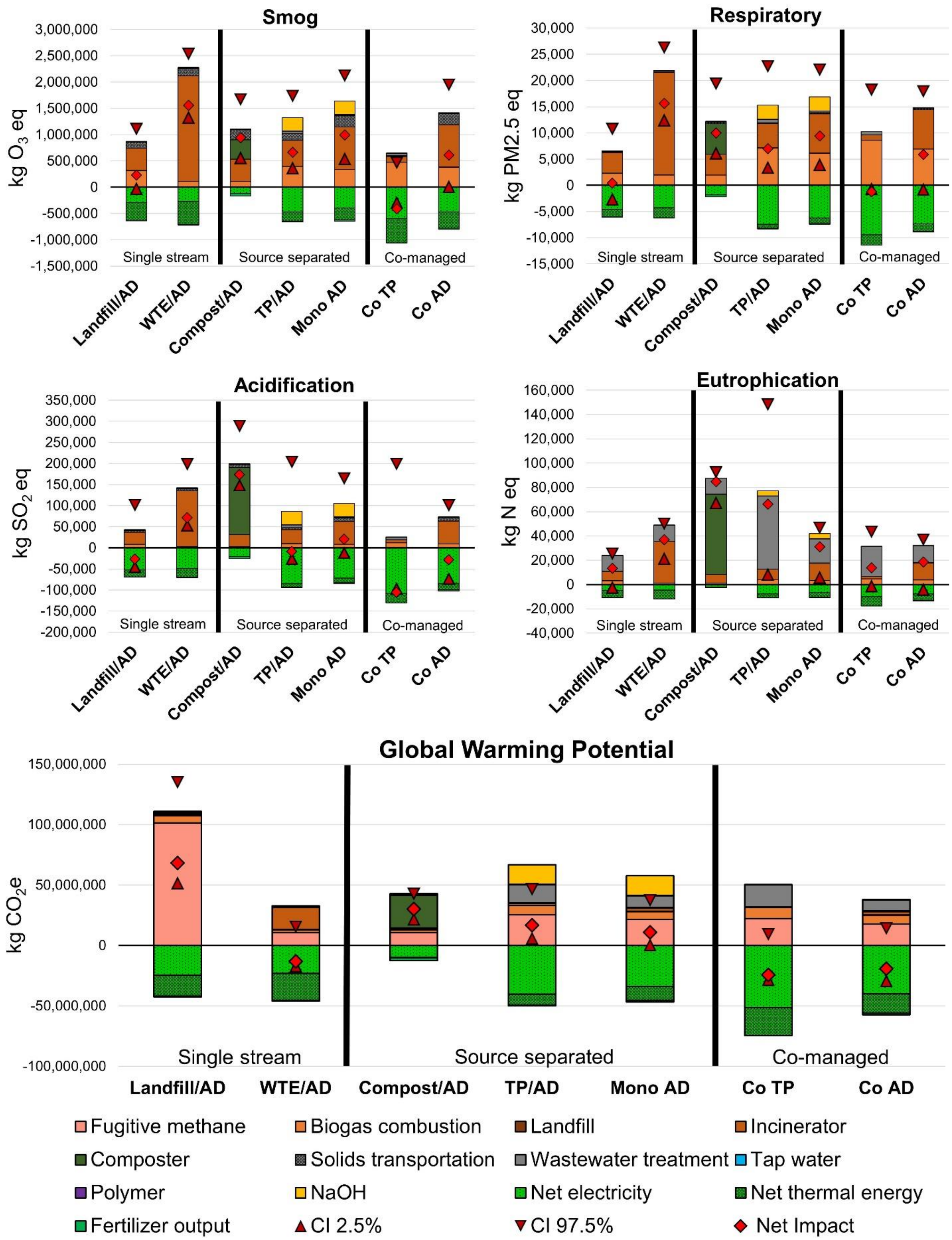


Figure 2. Environmental impacts of waste management systems. Red diamonds represent the baseline net impacts, red triangles indicate the 95% confidence interval calculated from the Monte Carlo analysis.

Limits of source separating food waste

The sensitivity analysis accounted for food waste landfill diversion rates for source separated waste management systems (see Figure 3A and 3B). Compost/AD, TP/AD, and Mono AD were found to have net GWP burdens even when all food waste was diverted from landfill, whereas WTE/AD had a net GWP benefit. Similarly, both co-digestion systems did not achieve the same net GWP benefit as WTE/AD until over 90% of food waste was diverted from landfill. Considering food waste sorting efficiency is reported to be only as high as 80% (Edwards et al., 2018), it is unlikely that over 90% of food waste would be diverted from landfill. This indicates that it *may not* be advantageous from a GWP perspective for municipalities with WTE to separate food waste from the residual waste stream, findings that mirror other studies that have evaluated WTE (Hodge et al., 2016; Thyberg and Tonjes, 2017).

Grid intensity and energy recovery

The grid mix for each system was varied from the baseline 2018 U.S. grid mix to determine the effects of GHG grid intensity on GWP impacts (see Figure 3C and 3D). Converting waste to electricity either through WTE or biogas combustion with CHP becomes less favorable as grid intensity decreases. Both co-digestion systems were most sensitive to grid intensity because they generated the greatest amount of exported electricity from biogas production. Net GWP impacts for Co TP and Co AD decreased by over 60% when the background electricity source was coal, but increased by about 40% when the background electricity source was solar PV. Both co-digestion systems had net GWP burdens when grid intensity fell below about 0.25 kg CO_2e kWh^{-1}. However, both co-digestion systems still had lower net GWP impacts than Landfill/AD and Compost/AD, and about the same GWP impact as WTE/AD when grid intensity was assumed to be zero (i.e. no GHG credit for electricity generation).

As our results show, the benefits of recovering energy from waste primarily in the form of electricity with CHP will decline overtime as the grid transitions away from fossil fuels to renewable sources for electricity generation, which concurs with results of other studies (Becker et al., 2017; Hodge et al., 2016; Thyberg and Tonjes, 2017). Another option to utilize biogas for energy generation is to upgrade biogas to renewable natural gas (RNG) by removing CO_2 and other trace contaminants. RNG is indistinguishable from fossil natural gas so it can be transported via conventional gas pipelines and used in the same manner (IEA, 2020). RNG can be used for energy options that are more difficult to decarbonize than electricity including high-temperature heating and heavy-duty transport (IEA, 2020). Upgrading to RNG to displace the use of fossil natural gas for energy uses other than electricity generation is an option to further improve the environmental benefits of energy generated from the co-digestion systems as the GHG intensity of the electric grid decreases (Ardolino et al., 2018). Landfill material, animal manure, wastewater, and industrial, institutional, and commercial organic waste in the United States is estimated to be about 431 trillion BTUs per year (NREL, 2013), which can displace about 4% of current industrial natural gas use or 9% of residential gas use (USEIA, 2021). AD systems exporting electricity generated from biogas in regions with low GHG intensity electric grids may want to consider instead upgrading biogas to RNG to produce a fuel that can offset more GHG intensive energy uses like heating or heavy-duty transport.

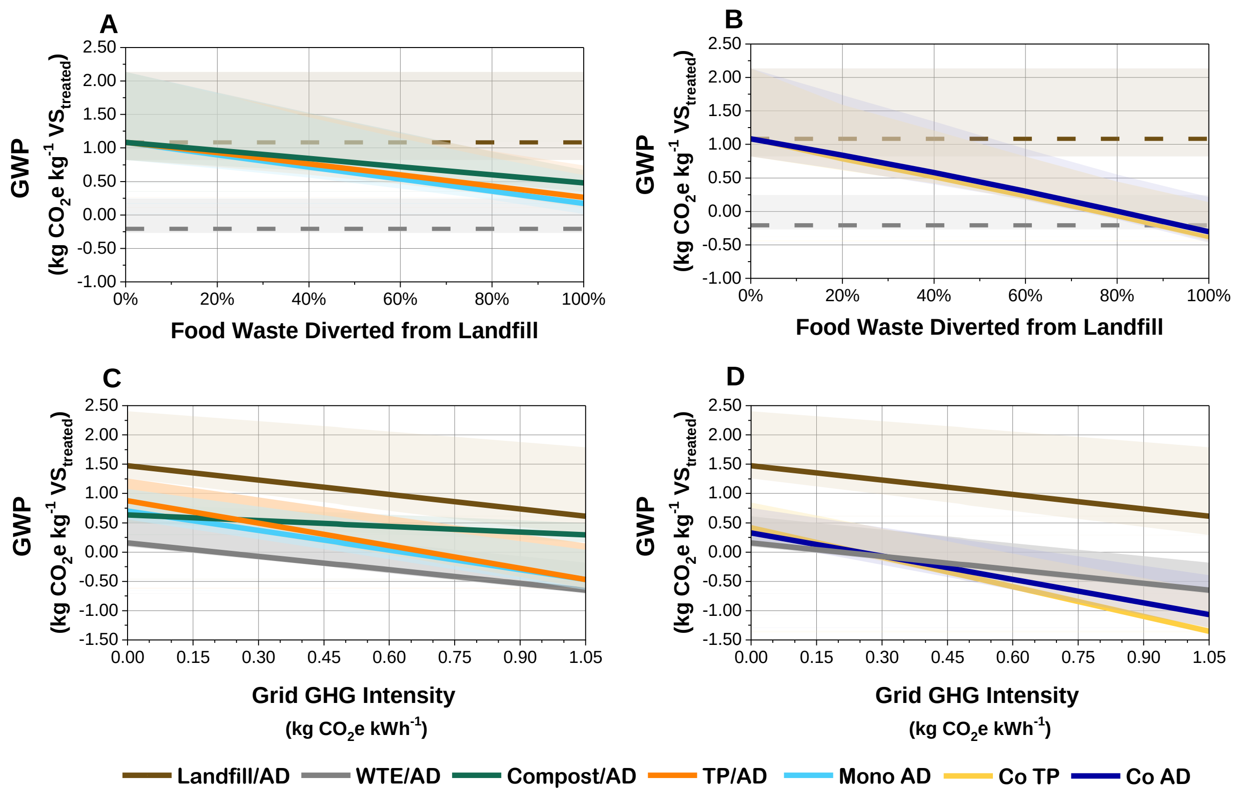


Figure 3. (A) and (B) GWP for single stream/source separated and single stream/co managed waste management systems for varying food waste diversion percentages. Landfill/AD and WTE/AD do not vary food waste diverted and are represented by dashed horizontal lines. (C) and (D) GWP for single stream/source separated and single stream/co managed waste management systems for varying electric grid GHG intensity. The results in Figure A, B, C and D are normalized by kg of VS treated and the shaded color region indicates the 95% confidence interval for each system calculated from the Monte Carlo analysis.

Conclusions

Co-management of food and sewage sludge waste streams through co-digestion maximizes resource recovery in the form of biogas production and reduces the environmental impacts of conventional waste management by offsetting the use of fossil-based energy. Landfill/AD has the greatest global warming impacts relative to all other management systems, even with an optimal landfill gas collection and utilization rate. Accordingly, it is beneficial to always divert food waste from landfill. Compost/AD ranks among the highest burdens for waste management among all environmental impacts because compost humus and AD biosolids used to offset mineral fertilizers cannot overcome the burdens of the system's energy requirements and offgas emissions. It is therefore more beneficial to divert food waste to a management options which generate energy. WTE/AD has lower global warming impacts than source separated food waste management systems, indicating it is not beneficial to divert food waste from WTE when WTE is displacing grid electricity generated from predominantly fossil fuels.

The synergistic benefits of co-digestion to eliminate the need for NaOH for pH control and improve biogas yield significantly reduced the impacts of Co TP and Co AD in comparison to mono-digestion of food waste and sewage sludge. Overall, the novel two-phase system has favorable environmental impacts similar to conventional AD, but with the operational advantage of a much lower HRT. This development may make AD more attractive to waste managers in areas where it has not been favorable in the past. However, increased nutrient loading from waste permeate generated from the two-phase system may lead to increases in eutrophication, acidification, and/or respiratory impacts and create operational challenges for waste managers. Future implementation of the two-phase system should consider coupling the system with low impact nutrient removal technology, or utilizing the high-strength permeate as reclaimed wastewater for irrigation to beneficially reuse, rather than treat and discharge, waste nutrients. The global warming benefits of generating primarily electricity from WTE or biogas using CHP fade as the GHG intensity of the grid decreases. Future investigation should consider upgrading biogas produced from AD systems to RNG to displace other energy uses that are more difficult to decarbonize so that biogas production can retain an offset credit to reduce the environmental impacts of anaerobic digestion technologies.

References

Ardolino, F., Parrillo, F., Arena, U., 2018. Biowaste-to-biomethane or biowaste-to-energy? An LCA study on anaerobic digestion of organic waste. J. Clean. Prod. 174, 462–476. https://doi.org/10.1016/j.jclepro.2017.10.320

Baumann, E.R., Oulman, C.S., 1955. The specific heat of garbage. Public Health Rep. 70, 805.

Becker, A.M., Yu, K., Stadler, L.B., Smith, A.L., 2017. Co-management of domestic wastewater and food waste: A life cycle comparison of alternative food waste diversion strategies. Bioresour. Technol. 223, 131–140. https://doi.org/10.1016/j.biortech.2016.10.031

Bolzonella, D., Battistoni, P., Susini, C., Cecchi, F., 2006. Anaerobic codigestion of waste activated sludge and OFMSW: The experiences of Viareggio and Treviso plants (Italy). Water Sci. Technol. 53, 203–211. https://doi.org/10.2166/wst.2006.251

Braguglia, C.M., Gallipoli, A., Gianico, A., Pagliaccia, P., 2018. Anaerobic bioconversion of food waste into energy : A critical review. Bioresour. Technol. 248, 37–56. https://doi.org/10.1016/j.biortech.2017.06.145

Coskun, C., Oktay, Z., Koksal, T., Birecikli, B., 2020. Co-combustion of municipal dewatered sewage sludge and natural gas in an actual power plant. Energy 211, 118615. https://doi.org/10.1016/J.ENERGY.2020.118615

Edwards, J., Othman, M., Crossin, E., Burn, S., 2018. Life cycle assessment to compare the environmental impact of seven contemporary food waste management systems. Bioresour. Technol. 248, 156–173. https://doi.org/10.1016/j.biortech.2017.06.070

Eriksson, M., Strid, I., Hansson, P.A., 2015. Carbon footprint of food waste management options in the waste hierarchy - A Swedish case study. J. Clean. Prod. 93, 115–125. https://doi.org/10.1016/j.jclepro.2015.01.026

Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2014. Life cycle assessment of energy from waste via anaerobic digestion: A UK case study. Waste Manag. 34, 226–237. https://doi.org/10.1016/j.wasman.2013.09.013

Fairley, T., Fonoll, X., Van Steendam, C., Raskin, L., Skerlos, S.J., 2019. Maximizing Energy Recovery in Anaerobic Membrane Bioreactors (AnMBRs) by Minimizing Permeate Dissolved Methane and Eliminating Biogas Sparging, in: WEFTEC 2019. Chicago, USA.

Fonoll, X., Astals, S., Dosta, J., Mata-Alvarez, J., 2016. Impact of paper and cardboard suppression on OFMSW anaerobic digestion. Waste Manag. 56, 100–105.

https://doi.org/10.1016/j.wasman.2016.05.023

Fonoll, X., Meuwissen, T., Aley, L., Shrestha, S., Raskin, L., 2019. Development of Dynamic Membrane Bioreactor Based on Rumen Physiology for Efficient Hydrolysis of Lignocellulosic Biomass, in: 16th IWA World Conference on Anaerobic Digestion (AD-16). Delft, Netherlands.

Grady, C.P.L., Daigger, G.T., Love, N.G., Filipe, C.D.M., 2011. Biological Wastewater Treatment, Third Edition. Taylor & Francis.

Gunaseelan, V.N., 1997. Anaerobic digestion of biomass for methane production: A review. Biomass and Bioenergy 13, 83–114. https://doi.org/10.1016/S0961-9534(97)00020-2

Harrison, K.W., Dumas, R.D., Barlaz, M.A., Nishtala, S.R., 2000. A Life-Cycle Inventory Model of Municipal Solid Waste Combustion. J. Air Waste Manage. Assoc. 50, 993–1003. https://doi.org/10.1080/10473289.2000.10464135

Hegde, Trabold, 2019. Anaerobic Digestion of Food Waste with Unconventional Co-Substrates for Stable Biogas Production at High Organic Loading Rates. Sustainability 11, 3875. https://doi.org/10.3390/su11143875

Hodge, K.L., Levis, J.W., DeCarolis, J.F., Barlaz, M.A., 2016. Systematic Evaluation of Industrial, Commercial, and Institutional Food Waste Management Strategies in the United States. Environ. Sci. Technol. 50, 8444–8452. https://doi.org/10.1021/acs.est.6b00893

Holliger, C., de Laclos, H.F., Hack, G., 2017. Methane production of full-scale anaerobic digestion plants calculated from substrate’s biomethane potentials compares well with the one measured on-site. Front. Energy Res. 5. https://doi.org/10.3389/fenrg.2017.00012

Hospido, A., Carballa, M., Moreira, M., Omil, F., Lema, J.M., Feijoo, G., 2010. Environmental assessment of anaerobically digested sludge reuse in agriculture: Potential impacts of emerging micropollutants. Water Res. 44, 3225–3233. https://doi.org/10.1016/j.watres.2010.03.004

IEA, 2020. Outlook for biogas and biomethane: Prospects for organic growth. Paris.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Introduction to the 2006 Guidelines.

Kjeldsen, P., Barlaz, M.A., Rooker, A.P., Baun, A., Ledin, A., Christensen, T.H., 2002. Present and long-term composition of MSW landfill leachate: A review. Crit. Rev. Environ. Sci. Technol. https://doi.org/10.1080/10643380290813462

Kong, X., Yu, S., Xu, S., Fang, W., Liu, J., Li, H., 2016. Effect of Fe0 addition on volatile fatty acids evolution on anaerobic digestion at high organic loading rates. Waste Manag. 71, 719–727. https://doi.org/10.1016/j.wasman.2017.03.019

Kurniawan, T.A., Lo, W. hung, 2009. Removal of refractory compounds from stabilized landfill leachate using an integrated H2O2 oxidation and granular activated carbon (GAC) adsorption treatment. Water Res. 43, 4079–4091. https://doi.org/10.1016/j.watres.2009.06.060

Lee, E., Oliveira, D.S.B.L., Oliveira, L.S.B.L., Jimenez, E., Kim, Y., Wang, M., Ergas, S.J., Zhang, Q., 2020. Comparative environmental and economic life cycle assessment of high solids anaerobic co-digestion for biosolids and organic waste management. Water Res. 171, 115443. https://doi.org/10.1016/j.watres.2019.115443

Levis, J., Barlaz, M.A., 2014. Landfill Gas Monte Carlo Model Documentation and Results.

Levis, J., Barlaz, M.A., 2013. Composting Process Model Documentation. Raleigh.

Levis, J., Barlaz, M.A., 2011. Background Information Document for Life-Cycle Inventory Landfill Process Model.

Levis, J.W., Barlaz, M.A., 2011. What Is the Most Environmentally Beneficial Way to Treat Commercial Food Waste? Environ. Sci. Technol 45, 7438–7444. https://doi.org/10.1021/es103556m

Liebetrau, J., Reinelt, T., Agostini, A., Linke, B., Murphy, J.D., 2017. Methane emissions from biogas plants.

Ma, Y., Liu, Y., 2019. Turning food waste to energy and resources towards a great environmental and economic sustainability: An innovative integrated biological approach. Biotechnol. Adv. https://doi.org/10.1016/j.biotechadv.2019.06.013

Macintosh, C., Astals, S., Sembera, C., Ertl, A., Drewes, J.E., Jensen, P.D., Koch, K., 2019. Successful strategies for increasing energy self-sufficiency at Grüneck wastewater treatment plant in Germany by food waste co-digestion and improved aeration. Appl. Energy 242, 797–808. https://doi.org/10.1016/j.apenergy.2019.03.126

Mata-Alvarez, J., Dosta, J., Romero-G?iza, M.S., Fonoll, X., Peces, M., Astals, S., 2014. A critical review on anaerobic co-digestion achievements between 2010 and 2013. Renew. Sustain. Energy Rev. 36. https://doi.org/10.1016/j.rser.2014.04.039

Mata-Alvarez, J., Mace, S., Llabres, P., 2000. Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives. Bioresour. Technol. 74.

Metcalf, L., Eddy, H.P., Tchobanoglous, G., 1991. Wastewater engineering: treatment, disposal, and reuse. McGraw-Hill New York.

Nagao, N., Tajima, N., Kawai, M., Niwa, C., Kurosawa, N., Matsuyama, T., Yusoff, F.M., Toda, T., 2012. Maximum organic loading rate for the single-stage wet anaerobic digestion of food waste. Bioresour. Technol. 118, 210–218. https://doi.org/10.1016/J.BIORTECH.2012.05.045

NREL, 2013. Biogas Potential in the United States.

NREL, 2012. U.S. Life Cycle Inventory Database [WWW Document]. U.S. Life Cycle Invent. Database. URL https://www.lcacommons.gov/

Pramanik, S.K., Suja, F.B., Zain, S.M., Pramanik, B.K., 2019. The anaerobic digestion process of biogas production from food waste: Prospects and constraints. Bioresour. Technol. Reports. https://doi.org/10.1016/j.biteb.2019.100310

Rahman, S.M., Eckelman, M.J., Onnis-Hayden, A., Gu, A.Z., 2016. Life-Cycle Assessment of Advanced Nutrient Removal Technologies for Wastewater Treatment. https://doi.org/10.1021/acs.est.5b05070

Shen, Y., Linville, J.L., Urgun-Demirtas, M., Mintz, M.M., Snyder, S.W., 2015. An overview of biogas production and utilization at full-scale wastewater treatment plants (WWTPs) in the United States: Challenges and opportunities towards energy-neutral WWTPs. Renew. Sustain. Energy Rev. https://doi.org/10.1016/j.rser.2015.04.129

Smith, A.L., Stadler, L.B., Cao, L., Love, N.G., Raskin, L., Skerlos, S.J., 2014. Navigating Wastewater Energy Recovery Strategies: A Life Cycle Comparison of Anaerobic Membrane Bioreactor and Conventional Treatment Systems with Anaerobic Digestion. Environ. Sci. Technol. 48, 5972–5981. https://doi.org/10.1021/ES5006169

Spokas, K., Bogner, J., Chanton, J.P., Morcet, M., Aran, C., Graff, C., Golvan, Y.M. Le, Hebe, I., 2006. Methane mass balance at three landfill sites: What is the efficiency of capture by gas collection systems? Waste Manag. 26, 516–525. https://doi.org/10.1016/j.wasman.2005.07.021

Tauber, J., Parravicini, V., Svardal, K., Krampe, J., 2019. Quantifying methane emissions from anaerobic digesters. https://doi.org/10.2166/wst.2019.415

Thyberg, K.L., Tonjes, D.J., 2017. The environmental impacts of alternative food waste treatment technologies in the U.S. J. Clean. Prod. 158, 101–108. https://doi.org/10.1016/j.jclepro.2017.04.169

Tong, H., Tong, Y.W., Peng, Y.H., 2019. A comparative life cycle assessment on mono- and co-digestion of food waste and sewage sludge, in: Energy Procedia. Elsevier Ltd, pp. 4166–4171. https://doi.org/10.1016/j.egypro.2019.01.814

U.S EPA, 2010. Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories : Solid Waste Disposal, Wastewater Treatment, Ethanol Fermentation, US Environmental Protection Agency.

UChicago Argonne, 2019. GREET-Greenhouse gases, Regulated Emissions, and Energy use in Transportation [WWW Document]. Greet. URL https://doi.org/10.11578/GREET-Net-2020/dc.20200913.1

USEIA, 2021. How much natural gas is consumed in the United States? [WWW Document]. URL https://www.eia.gov/tools/faqs/faq.php?id=50&t=8 (accessed 6.21.21).

USEPA, 2021a. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2019.

USEPA, 2021b. LMOP Landfill and Project Database [WWW Document]. URL https://www.epa.gov/lmop/lmop-landfill-and-project-database (accessed 6.30.21).

USEPA, 2020. 2018 Wasted Food Report. Washington D.C.

USEPA, 2016. Volume-to-Weight Conversion Factors.

USEPA, 2012. 2012 Guidelines for Water Reuse. Washington, D.C.

USEPA, 2011. Opportunities for Combined Heat and Power at Wastewater Treatment Facilities: Market Analysis and Lessons from the Field.

USEPA, 2003. Biosolids Technology Fact Sheet: Use of Incineration for Biosolids Management.

USEPA, n.d. Basic Information about Biosolids [WWW Document]. URL https://www.epa.gov/biosolids/basic-information-about-biosolids (accessed 6.14.21).

van der Lubbe, J., van Haandel, A., 2019. Anaerobic Sewage Treatment: Optimization of Process and Physical Design of Anaerobic and Complementary Processes. IWA Publishing.

Walker, J., Knight, L., Stein, L., 1994. A Plain English Guide to the EPA Part 503 Biosolids Rule.

Xu, F., Li, Yangyang, Ge, X., Yang, L., Li, Yebo, 2018. Anaerobic digestion of food waste – Challenges and opportunities. Bioresour. Technol. 247, 1047–1058. https://doi.org/10.1016/j.biortech.2017.09.020

Appendix

Table A1. Food waste and sewage sludge modeling parameters.

Parameter	Units	Value
U.S. per capita food waste generation [a]	kg FW $person^{-1}$ d^{-1}	0.48
Food waste density [b]	kg m^{-3}	275
Food waste % TS [c]	%	24
Food waste % VS (of TS) [c]	%	91
Food waste % C [c]	%	52
Food waste % N [c]	%	3
Food waste % P [c]	%	0.19
Food waste ash content [c]	%	1.8
Food waste LHV [c]	MJ kg^{-1} dry solids	23.1
Specific heat of food waste solids [d]	MJ kg^{-1} $°C^{-1}$	0.0014
U.S. per capita sewage sludge generation [e]	m^3 SS $person^{-}1$ d^{-1}	2.61E-03
Sewage sludge specific gravity [e]	NA	1.02
Sewage sludge % TS [e]	%	3.8
Sewage sludge % VS (of TS) [e]	%	68
Sewage sludge % N (of TS) [e]	%	4.0
Sewage sludge % P (of TS) [e]	%	1.4
Sewage sludge ash content [f]	%	30
Sewage sludge biomethane potential [g]	m^3 CH_4 kg^{-1} VS	0.403
Sewage sludge LHV [f]	MJ kg^{-1} dry solids	12.8
Specific heat of sewage sludge solids [e]	MJ kg^{-1} $°C^{-1}$	0.0042

a. Estimated using total annual food waste generation from residential, commercial, and institutional sectors in 2018 as reported in the US EPA *2018 Wasted Food Report* (USEPA, 2020)

b. As reported in the US EPA's *Volume-to-Weight Conversion Factor* (USEPA, 2016)

c. Properties as measured by Hedge & Trabold for cafeteria food waste (Hegde and Trabold, 2019)

d. Specific heat of garbage solids (Baumann and Oulman, 1955)

e. As reported in *Wastewater engineering: treatment, disposal, and reuse* (Metcalf et al., 1991)

f. Values obtained from US EPA *Biosolids Technology Fact Sheet* (USEPA, 2003)

g. As reported by Gunaseelan (Gunaseelan, 1997)

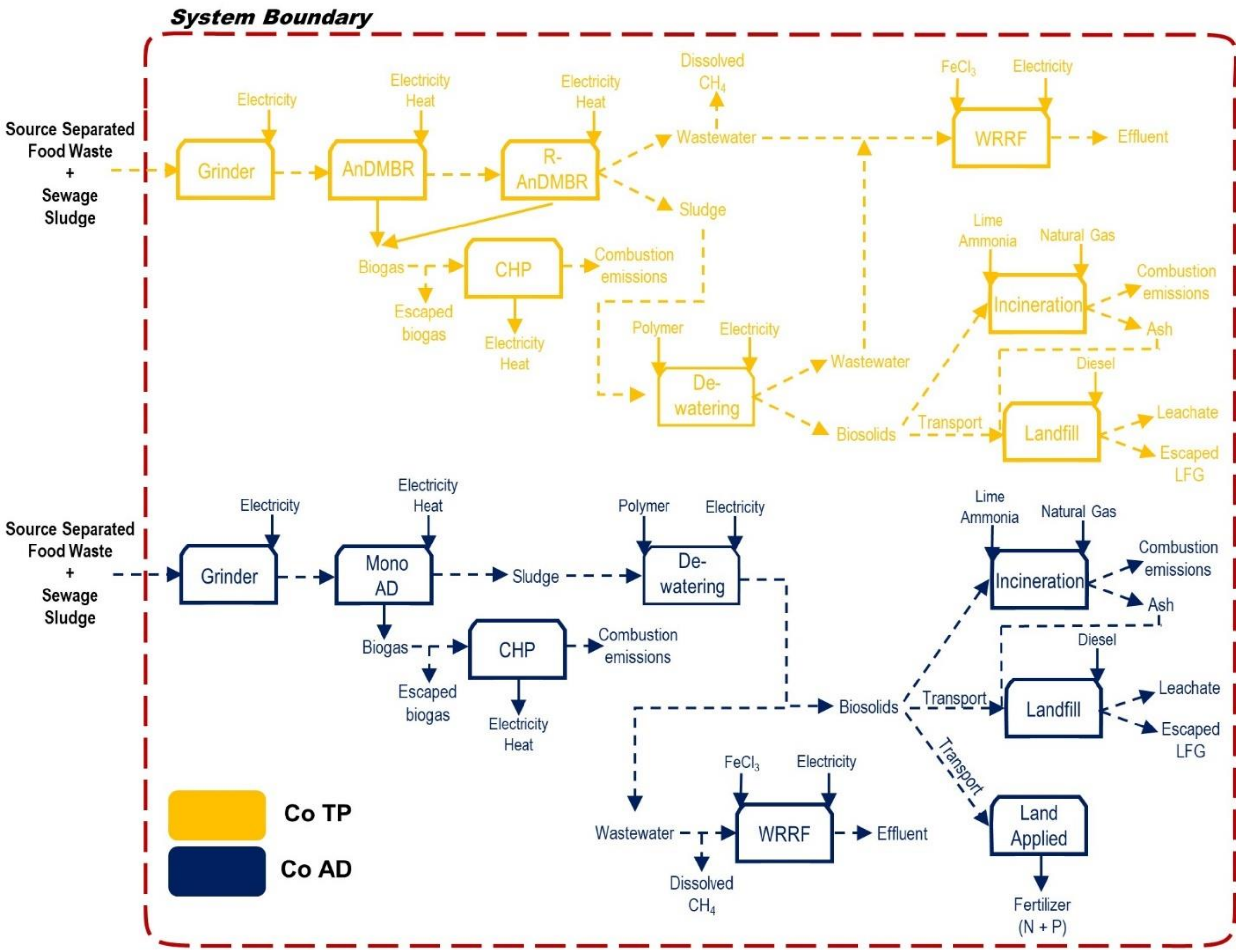


Figure A1. System boundary for co-managed (Co TP and Co AD) waste management systems. Food waste and sewage sludge are co-digested using the two-phase system (Co TP) or conventional AD (Co AD).

Details of Process Models for Waste Management Systems

Anaerobic Digestion Process Models

Table A2. Parameters used for anaerobic digestion process models.

System Model Parameter	Units	AD System	Value
HRT	d	TP (first-phase)	0.5
		TP (second-phase)	5
		Mono AD (FW)	20
		Mono AD (SS)	20
SRT	d	TP (first-phase)	3
		TP (second-phase)	1,000
		Mono AD (FW)	20
		Mono AD (SS)	20
Operating Temperature	°C	TP (first-phase)	39
		TP (second-phase)	20
		Mono AD (FW)	35
		Mono AD (SS)	35
VS Removal	%	TP (first-phase)	-
		TP (second-phase)	68
		Mono AD (FW)	55
		Mono AD (SS)	55
Methane Yield	m^3 CH_4 kg^{-1} VS_{fed}	TP (first-phase)	0.1
		TP (second-phase)	0.4
		Mono AD (FW)	0.4
		Mono AD (SS)	0.25
Biomass nutrient requirement [a]	kg N kg^{-1} VS_{wasted}	All	0.125
Biomass nutrient requirement [a]	kg P kg^{-1} VS_{wasted}	All	0.025

a. Values taken from *Biological Wastewater Treatment*, 3rd Edition.(Grady et al., 2011)

Thermal energy for digester heating is the sum of heat loss through the digester and sludge heating requirements. Heat loss through the digester sides, top, and bottom was calculated using Equation A.1 (Metcalf et al., 1991). Sludge heating requirements calculated using Equation A.2.

$$q_1 = \frac{U}{31.5576} A \Delta T n \tag{A.1}$$

Where:

q_1 = total heat loss (MJ)

U = overall coefficient of heat transfer (W m^{-2} °C)

A = cross-sectional area through which the heat loss is occurring (m^2)

ΔT = temperature drop across the surface in question (°C)

n = time digester is in use (years)

Table A3. Heat transfer coefficients for digester heating.

Heat-transfer coefficient ($W\ m^{-2}\ °C$) [b]	**Value**
Dry earth	0.68
Moist soil	0.85
Air	0.91

b. Values taken from *Wastewater engineering: treatment, disposal, and reuse* (Metcalf et al., 1991)

$$q_2 = MC\Delta T \tag{A.2}$$

Where:

q_2 = total sludge heat requirement (MJ)

M = mass of sludge (kg)

C = specific heat of sludge (MJ kg^{-1} $°C^{-1}$)

ΔT = difference in temperature between influent sludge and digester (influent sludge is assumed to be 15°C)

Sludge Dewatering Process Model

Digester sludge dewatering was modeled based on biosolids' end disposal method. Land applied sludge was dewatered with a gravity belt thickener that was assumed to produce a cake with 22% solids. Sludge sent to landfill or incineration was dewatered with a centrifuge that was assumed to produce a cake with 18% solids. Polymer for dewatering was dosed at a rate of 5E-03 kg kg^{-1} dry solids (Smith et al., 2014). Electrical energy requirements for the gravity belt thickener and centrifuge were calculated using Equation A.3 and Equation A.4 (Smith et al., 2014)

$$GBT_{EE}, \left(\frac{\text{kWh}}{\text{d}}\right) = \frac{\left(422{,}832\ \frac{\text{kWh}}{\text{yr}}\right)(GBT\ inf, \text{MGD})^{0.9248}}{365.25\ \frac{\text{d}}{\text{yr}}} \tag{A.3}$$

$$CENT_{EE}, \left(\frac{\text{kWh}}{\text{d}}\right) = \frac{\left(5{,}024{,}825\ \frac{\text{kWh}}{\text{yr}}\right)(CENT\ inf, \text{MGD}) + 39{,}693\frac{\text{kWh}}{\text{yr}}}{365.25\ \frac{\text{d}}{\text{yr}}} \tag{A.4}$$

Combined Heat and Power (CHP) Process Model

Heat and power generation from biogas and LFG combustion was modeled assuming a reciprocating engine (lean-burn) with a power-to-heat ratio (α) of 0.86 and electric efficiency (η_E) of 32.6%.(USEPA, 2011). Electric and thermal energy outputs were calculated using Equation A.5 and Equation A.6.

$$CHP_{\mathrm{EO}} = \left(HHV_{\mathrm{CH_4}}\right)(\eta_E) \tag{A.5}$$

$$CHP_{\mathrm{HO}} = \frac{\left(HHV_{\mathrm{CH_4}}\right)(\eta_E)}{\alpha} \tag{A.6}$$

Where:

CHP_{EO} = total CHP electric energy output (MJ)

CHP_{HO} = total CHP thermal energy output (MJ)

HHV_{CH4} = high heating value of methane

η_E = CHP electric efficiency (HHV)

α = CHP power-to-heat ratio

Wastewater Treatment Process Model

Dissolved methane in wastewater generated from reactor permeate of the two-phase system and sludge dewatering was calculated using Equation A.7 (van der Lubbe and van Haandel, 2019) The methane oversaturation value for permeate from the two-phase system was 1.3 according to experimental data. Wastewater from sludge dewatering was assumed to have an oversaturation value of 1.07 (Tauber et al., 2019).

$$\left[\mathrm{CH}_{4,dissolved}\right] = \left(\frac{p_{\mathrm{CH_4}}}{K_{H,\mathrm{CH_4}}}\right)(M)\left(MW_{\mathrm{CH_4}}\right)(OS)(1000) \tag{A.7}$$

Where:

[CH_4, *dissolved*] = concentration of dissolved methane (mg L^{-1})

p_{CH4} = partial pressure of methane (atm)

$K_{H,\,CH4}$ = Henry's constant for methane (atm)

M = molarity of solution (mol L^{-1}), assumed to be 55.6 mol L^{-1}

MW = molecular weight of methane (g mol^{-1})

OS = methane oversaturation

Material and energy requirements for nutrient removal (N+P) were estimated using LCI results from Rahman et al. (2016). For basic nutrient removal, $FeCl_3$ is added in the primary clarifier and the University of Cape Town process is used for biological nutrient removal. For enhanced nutrient removal, MeOH is added as a supplementary carbon source for enhanced nitrogen removal and $Al_2(SO_4)_3$ is added for enhanced chemical phosphorous removal. Dosing requirements for $FeCl_3$ and $Al_2(SO_4)_3$ were normalized to total P concentration in the influent and dosing requirements for MeOH were normalized to total N concentration in the influent based on the LCI results presented by Rahman et al.

Table A4. Parameters used for wastewater treatment process model.

Wastewater Treatment Parameter	Units	Basic Nutrient Removal	Enhanced Nutrient Removal
Final N discharge	kg total N m^{-3} WW	8	1
Final P discharge	kg total P m^{-3} WW	1	0.01
Electricity Use	kWh m^{-3} WW	0.35	0.36
$FeCl_3$ Use	kg kg^{-1} P in WW	1.09	3.88
MeOH Use	kg kg^{-1} N in WW	-	0.674
$Al_2(SO_4)_3$ Use	kg kg^{-1} P in WW	-	3.80

Direct emissions of N_2O during biological nutrient removal were calculated using Equation A.8 following USEPA methodology (U.S EPA, 2010).

$$N_2O_{WWTP} = (Q_i)(TKN_i)\left(EF_{\mathrm{N_2O}}\right)\left(\frac{44}{28}\right)(10^{-6}) \quad \text{(A.8)}$$

Where:

N_2O_{WWTP} = N_2O emissions generated from WWTP process (Mg N_2O hr^{-1})

Q_i = Wastewater influent flow rate (m^3 hr^{-1})

EF_{N2O} = N_2O emission factor

= 0.005 g N emitted as N_2O per g TKN

Landfill Process Model

Diesel fuel requirement for landfill operations was 0.19 gal ton^{-1} waste (J. Levis and Barlaz, 2011). Landfill leachate captured and treated as wastewater was assumed to have 740 mg L^{-1} ammonia-N and 6 mg L^{-1} total P (Kjeldsen et al., 2002).

Waste-to-Energy (WTE) & Incineration Process Models

The WTE and incineration process models were based on the life cycle inventory data reported by Harrison et al. (2000). Lime and ammonia material inputs were modeled for use as acid gas and NO_x controls. In addition, the lime added is included in ash production. Material inputs and direct air emissions from food waste and biosolids combustion were taken from data for food waste and miscellaneous combustibles provided by Harrison et al. Food waste combustion was modeled as a waste-to-energy process where power and heat are generated and captured. Power and heat efficiencies for food waste combustion were taken from Hodge et al. (2016). Biosolids incineration did not consider generation of power or capture of excess heat. Food waste was assumed to be combusted with the residual waste stream and would not require auxiliary fuel. Biosolids were assumed to be combusted with natural gas (Coskun et al., 2020).

Table A5. Parameters used for WTE/incineration process models.

Process Parameter	Units	WTE (food waste)	Incineration (biosolids)
Waste LHV	MJ kg^{-1} dry solids	23.1	12.8
Waste % TS	%	24	18
Auxiliary fuel requirements	kg natural gas kg^{-1} waste	-	0.01
WTE net electric efficiency	%	10.3	-
WTE net thermal efficiency	%	37.5	-
Lime use	kg kg^{-1} waste	0.145	0.095
Ammonia use	kg kg^{-1} waste	0.031	0.020
Direct air emissions	kg biomass CO_2 kg^{-1} waste	5.00E-01	1.30E00
	kg SO_2 kg^{-1} waste	1.80E-04	4.70E-04
	kg HCl kg^{-1} waste	8.50E-05	2.20E-04
	kg NO_x kg^{-1} waste	4.20E-04	1.10E-03
	kg CO kg^{-1} waste	2.60E-04	6.80E-04
	kg PM kg^{-1} waste	5.00E-05	1.30E-04
Waste ash	kg ash kg^{-1} waste	0.018	0.300

Compost Process Model

The compost process model and model parameters were based on the model derived by Levis and Barlaz (Levis and Barlaz, 2013). Electricity use is included for grinding and screening food waste and diesel use is included for operations of front-end loaders used to move materials and build piles. The aerobic composting process releases most carbon as biogenic CO_2, but some is also released as CH_4. Most nitrogen is released as N_2 but some is emitted as NH_3 and N_2O. VOC emissions depend on the mass of volatile solids entering curing. The compost model was assumed to use an odor control system and biofilter that can reduce emissions of CH_4, NH_3, and VOCs.

Table A6. Parameters used for compost process model.

Process Parameter	Units	Value
Sawdust use	kg kg^{-1} FW	0.02
Water use	kg kg^{-1} FW	0.32
Electricity use	kWh kg^{-1} FW	4.00E-04
Diesel use	L kg^{-1} FW	1.02E-03
Carbon loss	% of FW C	65.7
CH_4 production	% of C loss	1.7
Nitrogen loss	% of FW N	66
NH_3 production	% of N loss	4.0
N_2O production	% of N loss	0.4
VS loss to Carbon loss	kg VS kg^{-1} C	1.9
VOCs production	kg VOCs kg^{-1} VS	2.38E-04
Biofilter removal efficiency		
CH_4	%	15
NH_3	%	48
N_2O	%	0
VOCs	%	18

Table A7. Emission factors for unit processes.

Process	Units	GWP (kg CO_2e)	Smog Air (kg O_3 eq)	Acidification Air (kg SO_2 eq)	Eutrophication Water (kg N eq)	HH Particulate Air (kg PM2.5 eq)	Source
Water	per kg	1.60E-08	6.67E-10	2.87E-11	1.12E-11	1.53E-12	GREET 1 (2019)
Sodium Hydroxide	per kg	2.08E+00	3.26E-02	4.08E-03	5.43E-04	3.50E-04	GREET 1 (2019)
LDPE*	per kg	1.77E+00	5.48E-02	3.34E-03	9.00E-04	2.21E-04	GREET 1 (2019)
Generic N	per kg	4.58E+00	9.67E-02	5.84E-03	1.62E-03	5.19E-04	GREET 1 (2019)
Generic P	per kg	4.79E+00	7.78E-02	8.41E-03	1.30E-03	7.39E-04	GREET 1 (2019)
Potassium Chloride	per kg	4.68E-01	9.20E-03	1.71E-03	1.55E-04	1.46E-04	GREET 1 (2019)
Ammonia	per kg	1.04E+00	2.98E-02	1.68E-03	4.88E-04	1.08E-04	GREET 1 (2019)
Lime (CaO)	per kg	4.49E-01	7.50E-03	4.79E-04	1.23E-04	1.53E-04	GREET 1 (2019)
MeOH	per kg	4.81E-01	1.20E-02	9.04E-04	1.78E-04	7.79E-05	GREET 1 (2019)
PCl_3†	per kg	1.55E+00	4.59E-02	4.59E-03	7.73E-04	5.26E-04	GREET 1 (2019)
Copper (II) sulfate‡	per kg	1.24E+00	5.47E-02	8.49E-02	9.34E-04	5.31E-03	GREET 1 (2019)
Fuel Combustion for Stationary Applications: Biogas – Stationary Reciprocating Engine	per MJ	1.08E-02	5.56E-04	1.36E-05	5.55E-06	1.01E-05	GREET 1 (2019)
Baseline Conventional and LS Diesel: Well-to-Pump	per MJ	1.72E-02	6.48E-04	3.80E-05	9.40E-06	4.06E-06	GREET 1 (2019)
Fuel Combustion for Stationary Applications: Diesel - Stationary Reciprocating Engine	per MJ	7.34E-02	3.32E-02	1.38E-03	5.73E-04	7.75E-05	GREET 1 (2019)
Fuel Throughput: Natural Gas as Stationary Fuels	per MJ	1.13E-02	6.79E-04	3.75E-05	1.10E-05	1.45E-06	GREET 1 (2019)

* Material substitute for polymer
† Material substitute for $FeCl_3$
‡ Material substitute for $Al_2(SO_4)_3$

Natural Gas Fuel Combustion for Stationary Applications: Small Industrial Boiler	per MJ	5.64E-02	6.65E-04	2.75E-05	1.13E-05	4.39E-06	GREET 1 (2019)
U.S. Mix Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): feedstock	per kWh	4.23E-02	2.10E-03	1.30E-04	3.33E-05	1.53E-05	GREET 1 (2019)
U.S. Mix Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): fuel	per kWh	4.36E-01	3.47E-03	8.77E-04	5.94E-05	7.27E-05	GREET 1 (2019)
Coal-Fired Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): feedstock	per kWh	5.32E-02	2.56E-03	1.62E-04	3.91E-05	3.96E-05	GREET 1 (2019)
Coal-Fired Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): fuel	per kWh	1.00E+00	6.42E-03	2.73E-03	1.10E-04	2.13E-04	GREET 1 (2019)
NG-Fired Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): feedstock	per kWh	7.37E-02	3.80E-03	2.29E-04	6.13E-05	1.01E-05	GREET 1 (2019)
NG-Fired Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): fuel	per kWh	4.25E-01	3.78E-03	1.74E-04	6.42E-05	1.65E-05	GREET 1 (2019)
PV Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): feedstock	per kWh	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	GREET 1 (2019)
PV Power Plant Fuel-Cycle Emissions of Electricity Available at User Sites (wall outlets): fuel	per kWh	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	GREET 1 (2019)
CIDI - Diesel (vehicle operation): Well-to-Wheels	per MJ	7.56E-02	7.44E-04	2.48E-05	1.01E-05	4.53E-06	GREET 1 (2019)
Transport, combination truck, short-haul, diesel powered	per tonne-km	1.10E-01	3.26E-02	1.29E-03	7.70E-05	1.67E-09	U.S. LCI

Table A8. Baseline environmental impacts of waste management systems considering enhanced nutrient removal.

	Waste Management Systems						
	Landfill/ AD	**WTE/ AD**	**Compost/ AD**	**TP/ AD**	**Mono AD**	**Co TP**	**Co AD**
GWP (kg CO_2e)	7.04E+07	-1.10E+07	3.23E+07	2.28E+07	1.41E+07	-1.50E+07	-1.68E+07
Smog (kg O_3 eq)	3.10E+05	1.64E+06	1.02E+06	8.67E+05	1.10E+06	-9.17E+04	6.90E+05
Respiratory (kg PM2.5 eq)	5.47E+03	2.07E+04	1.50E+04	1.76E+04	1.47E+04	1.61E+04	9.00E+03
Acidification (kg SO_2 eq)	5.14E+04	1.49E+05	2.52E+05	1.55E+05	1.01E+05	1.62E+05	1.93E+04
Eutrophication (kg N eq)	2.84E+03	2.64E+04	7.42E+04	1.44E+04	1.50E+04	-3.13E+03	7.41E+03
	Higher Impact						**Lower Impact**

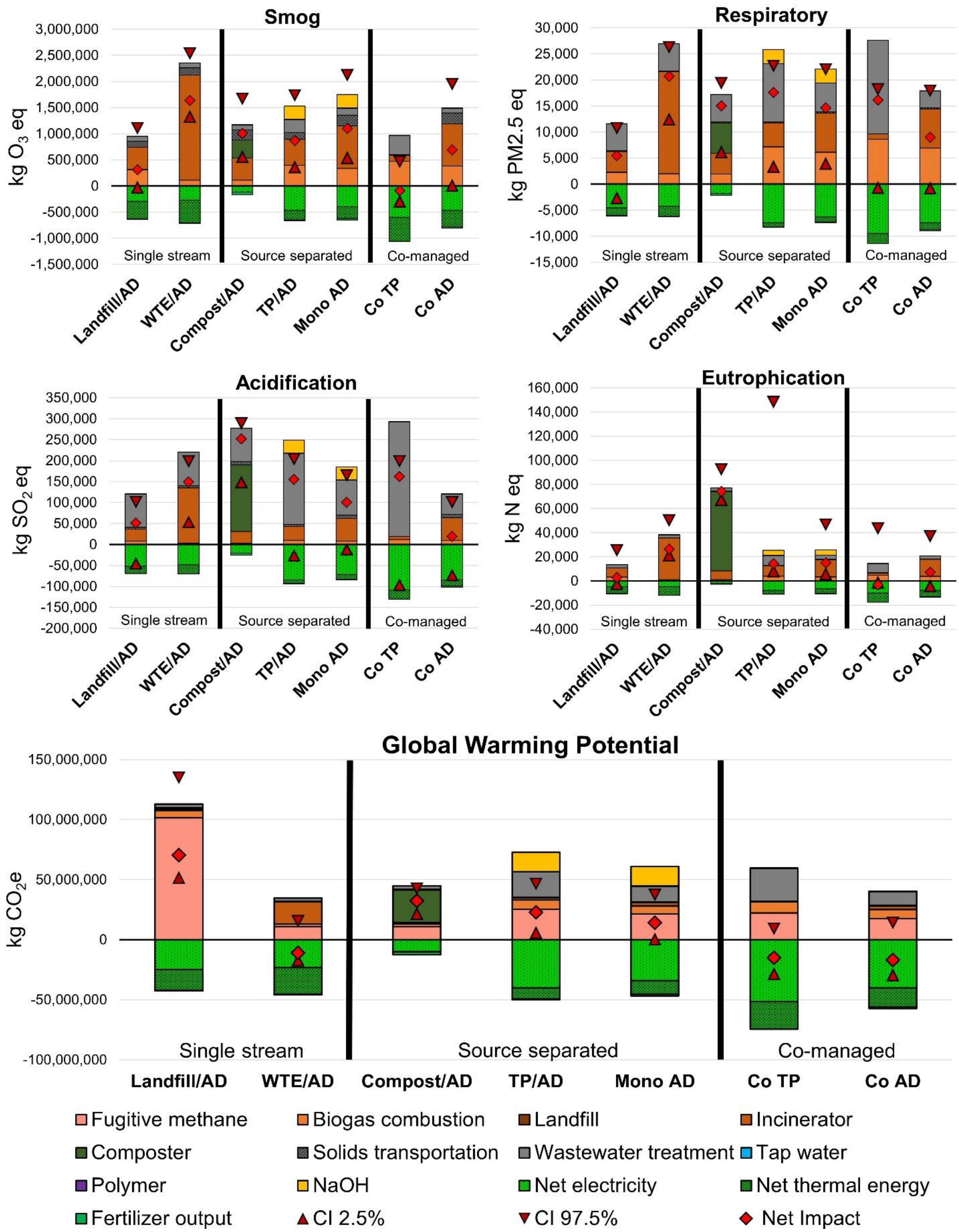


Figure A2. Environmental impacts of waste management systems considering enhanced nutrient removal. Red diamonds represent the baseline net impacts, red triangles indicate the 95% confidence interval calculated from the Monte Carlo analysis.