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Life-cycle assessment of treating slaughterhouse waste using anaerobic digestion systems

Shunli Wang [a, b], Kamalakanta Sahoo [c, d, *], Umakanta Jena [e], Hongmin Dong [a, b], Richard Bergman [c], Troy Runge [d]

[a] Institute of Environment and Sustainable Development in Agriculture, Chinese Academy of Agricultural Sciences, Beijing, 100081, China
[b] Key Laboratory of Energy Conservation and Waste Treatment of Agricultural Structures, Ministry of Agriculture and Rural Affairs, Beijing, 100081, China
[c] Forest Products Laboratory, United States Forest Service, Madison, WI, USA
[d] Great Lakes Bioenergy Research Center and Biological Systems Engineering, University of Wisconsin-Madison, Madison, WI, USA
[e] Department of Chemical and Materials Engineering, New Mexico State University, P.O. Box 30001, MSC 3805, Las Cruces, NM, USA

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ABSTRACT

The meat industry in the US generates considerable amount of slaughterhouse waste (SHW), which is typically converted into more useable products through the rendering process. Although rendering generates sellable fat and meal commodities, it has large environmental impacts because it is energy intensive. Anaerobic digestion (AD) is a promising technology for treating SHW and reducing environmental impact through biogas production to generate heat or electricity, as well as by enabling nutrient recovery and pathogen reduction. This study compared the life-cycle energy use and global warming impact of treating SHW with traditional rendering with AD to produce heat and electricity. The study also considered the co-digestion of SHW with the organic fraction of municipal solid waste (OFMSW) and sewage sludge. A cradle-to-grave life-cycle assessment (LCA) method was used to quantify the energy use and greenhouse gas (GHG) emissions of these systems for treating SHW. We compared three scenarios: (1) AD of SHW, (2) Co-AD of SHW with OFMSW, and (3) Co-AD of SHW with sewage sludge to reference systems of simple rendering and composting. The study findings revealed that the total cradle-to-gate energy use and GHG emissions by treating SHW with AD and co-AD were 0.5–6.7 GJ/1000 kg-SHW and 400–834 kg-CO_2-eq/1000 kg-SHW, whereas for the rendering control scenarios total cradle-to-gate energy use and GHG emissions were 1.9 GJ/1000 kg-SHW and 96.4 kg-CO_2-eq/1000 kg-SHW, respectively. However, considering all the benefits of treating SHW with co-AD, including the displacement of fossil fuel and electricity and nitrogen fertilizers generated as system outputs, these systems perform better than the rendering process. Compared to the reference systems, the GHG reduction potential of treating SHW with co-AD varied between 426.8 and 524.0 kg-CO_2-eq/1000 kg-SHW. Among all input parameters, methane (CH_4) leak from the AD system, and nitrogen fertilizer displacement were the most sensitive parameters affecting the results. By implementing AD of SHW in meat industries in the southeast US, the energy production and GHG emissions reduction potential were estimated to be 22–29 $\times$ 10^6 GJ and 1.6–2.0 $\times$ 10^9 kg-CO_2-eq per year, respectively. The results indicate that the AD of SHW can substantially reduce GHG emissions of the US meat industry as well as produce bioenergy to provide energy security in the US.

1. Introduction

In 2018, meat producers in the US produced approximately 46.5 million metric tons (MT) of ready-to-eat meats, and an approximately equivalent amount of slaughterhouse waste (SHW) was discharged (USDA, 2019a; USDA, 2019b). SHW contains high amounts of protein and lipids and is thermally treated in a rendering plant. Rendering products are usually byproduct meals, fertilizers, and raw materials of the oleochemical industry for sale in the market (Ockerman and Hansen, 2000). In 2007, the US rendering industry produced 8.6 million MT of rendering products,

* Corresponding author. Forest Products Laboratory, United States Forest Service, Madison, WI, USA.
E-mail address: kamalakanta.sahoo@usda.gov (K. Sahoo).

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Nomenclature

AD	Anaerobic digestion
CH_4	Methane
CHP	Combined heat and power
CO_2	Carbon dioxide
COD	Chemical oxygen demand
GHG	Greenhouse gas
GWP	Global warming potential
HRT	Hydraulic retention time
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life-cycle assessment
LCI	Life-cycle inventory
MT	Metric ton
N_2O	Nitrous oxide
OFMSW	Organic fraction of municipal solid waste
OLR	Organic loading rate
REAP	Rural Energy for America Program
RFS	Renewable fuel standard
RIN	Renewable index number
RPS	Renewable portfolio standard
SHW	Slaughterhouse waste
TS	Total solids
VS	Volatile solids

including 53% fats and greases and 47% protein meals (Nationalrenders, 2020). Rendering is a thermal process of cooking SHW under conditions of high temperature and/or pressure. The rendering process consumes large amounts of electricity and fuel (Gooding and Meeker, 2016), which might lead to adverse environmental impacts.

Anaerobic digestion (AD) is considered a promising alternative for treating SHW for the production of bioenergy (such as biogas or biomethane) and useful byproducts (inorganic nutrients) while reducing greenhouse gas (GHG) emissions (Moukazis et al., 2018; Ortner et al., 2014; Shirzad et al., 2019). High biomethane potential of SHW [0.250–0.512 m^3 kg^{-1} volatile solids (VS)] and nutrient recovery were reported during the anaerobic treatment of waste generated from a poultry processing plant (Yoon et al., 2014). Protein-rich poultry blood waste and lipid-rich swine SHW can produce 0.651 m^3 and 0.999 m^3 methane (CH_4) per kg VS of feedstock, respectively (Wang et al., 2018a; Ning et al., 2018), at different scales of AD. Due to its high nitrogen content, SHW is often co-digested with other carbon-rich wastes, including the organic fraction of municipal solid waste (OFMSW) and sewage sludge (Wang et al., 2018b) to balance the carbon-to-nitrogen ratio. Luste and Luostarinen (2010) reported that approximately 0.4 m^3 CH_4 per kg VS of feedstock can be obtained by anaerobically co-digesting animal byproducts and sewage sludge under the mesophilic condition. Salehiyoun et al. (2020) reported that 0.550 m^3 CH_4 per kg VS of feedstock could be achieved when treating a mixture of SHW and sewage sludge in a continuously run digester under an organic loading rate (OLR) of 1.5 kg VS m^3 d^{-1}. Similarly, 1.85 m^3 CH_4 m^{-3} d^{-1} was achieved when a mixture of SHW and OFMSW was treated in a laboratory-scale anaerobic digester under the mesophilic condition (Cuetos et al., 2008). Furthermore, Rodríguez-Abalde et al. (2017) anaerobically co-digested SHW, pig slurry, and glycerine in a semi-continuous digester and produced 0.64 m^3 CH_4 m^{-3} d^{-1} under an OLR of 3.2 kg chemical oxygen demand (COD) m^{-3} d^{-1}. Because CH_4 is one of the cleanest energy sources and a potent GHG, biomethane produced from the AD of SHW can provide high-quality bioenergy and considerably reduce GHG emissions from waste generated by the meat industry (Sahoo and Mani, 2019; Shirzad et al., 2019).

At present, the US is ranked second among all counties contributing approximately 15% [6677 million MT carbon dioxide equivalents (CO_2-eq) in 2018 (USEPA, 2020a)] to the total global GHG emissions (USEPA, 2019). To reduce GHG emissions, twenty-three states in the US have enacted future targets of GHG reduction (C2ES, 2019a). Twenty-nine states in the US have adopted a renewable portfolio standard (RPS) that requires a percentage of a utility's electricity to be derived from renewable energy sources (C2ES, 2019b). For example, California issued a goal to reduce 40% of the GHG emissions of 1990 levels by 2030 (CLI, 2016). Furthermore, in the US, industry and electricity sectors have been identified as the major GHG emission sources, accounting for 22% and 27% of the total GHG emissions, respectively (USEPA, 2020a). In addition, to reduce GHG emissions from transportation fuel, the US mandated [through the Renewable Fuel Standard (RFS)] to produce 60 billion liters of advanced biofuel by 2022; however, the actual production has remained far behind the required RFS-mandated volume (USEPA, 2020b). To meet the RPS mandate, advanced biofuel, including biomethane production, requires far greater growth (Sahoo and Mani, 2019).

Over the last several years, the actual production of advanced biofuel has increased multifold, especially biomethane that accounts for more than 80% of the total production of advanced biofuel (Masum et al., 2019; USEIA, 2017). A high quantity of SHW in the US could be anaerobically converted to biomethane to meet the RFS volume requirements, foster rapid GHG emission reduction targets, and secure the increasing energy demand in the US (Sahoo and Mani, 2019; Wang et al., 2018b). AD of SHW can be a source of biomethane, which can be converted to different types of energy (liquid fuel, gas, or electricity) and supplied to various industrial and municipal sectors (Wang et al., 2018b). Studies on treating SHW to produce fuel and electricity have been limited, and its life-cycle assessment (LCA), which is critical for the sustainable growth of the meat industry and reducing the carbon footprint, has not been explored.

LCA is an important tool for evaluating the potential environmental impact associated with a product or service throughout its life cycle. LCA studies generally include three stages: raw material acquisition, product manufacturing, and product use and disposal (ISO 14044, 2006; ISO 14040, 2006; Ramirez et al., 2012). LCA has been used to comprehensively evaluate the environmental impact of AD systems that treat various organic wastes/materials, including animal manure (Ebner et al., 2015), wastewater plant sewage sludge (Li et al., 2017; Sanchez, 2020), food waste, and energy crops (Sahoo and Mani, 2019). Some studies have assessed the life-cycle sustainability of a slaughter (Valente et al., 2020), hybrid life-cycle environmental impact of beef processing in the US (Li et al., 2020), and biogas energy recovery of AD treating SHW in a slaughterhouse (Vilvert et al., 2020). In addition, Gooding and Meeker (2016) calculated and compared the energy consumption and GHG emissions directly by treating different animal byproducts. However, the LCA of AD systems treating SHW in the US (Shirzad et al., 2019) has not yet been investigated. Furthermore, LCA studies on treating SHW by AD and other alternatives (such as rendering) can provide critical information that may help decision-makers adopt an environmentally superior option to treat SHW in meat processing plants.

Therefore, the major objectives of this study were to (1) estimate the mass and energy balance of treating SHW in AD (scenario 1) and co-AD with OFMSW (scenario 2) or sewage sludge (scenario 3) to produce biogas, which was used to generate heat and electricity; (2) quantify cradle-to-grave GHG emissions from the above-mentioned scenarios and compare them with reference systems. In scenario 1, diluted SHW was treated in AD and compared with the rendering of SHW (reference system). In scenario 2, SHW and OFMSW were co-digested and compared with the reference systems, such as the rendering of SHW and composting of OFMSW. In scenario 3, SHW and sewage sludge were co-digested and compared with reference systems, such as the rendering of SHW and composting of sewage sludge. In addition, a sensitivity analysis was conducted to analyze the impact of variations in the critical input factors on the results. Furthermore, energy production and GHG emissions from the proposed co-AD systems treating SHW in the southeast US (the major meat producing region in the US) were analyzed and discussed.

2. Materials and methods

2.1. Goal and scope definition

This study conducted a cradle-to-grave LCA for treating SHW by AD or co-digestion (Co-AD) to produce biogas (and further used to generate power) according to the International Organization for Standardization (ISO) guidelines (ISO 14040, 2006; ISO 14044, 2006).

The goal of this study was to estimate the energetic performance and to quantify the global warming impacts of treating SHW through AD and compare it with alternative treatments such as rendering. The functional unit in this study was to treat 1000 kg (or Mg) of SHW. Fig. 1 shows the three scenarios of AD systems and the corresponding reference systems. The scenarios include (1) AD of SHW, (2) co-AD of SHW and OFMSW, and (3) co-AD of SHW and sewage sludge. All three scenarios produced biogas that was combusted to generate power (heat and electricity). The reference systems included rendering of SHW and composting of OFMSW and sewage sludge. The boundary of the AD system and the reference system included all direct inputs and outputs, energy consumption/generation, and GHG emissions, including non-biogenic CO_2, CH_4, and nitrous oxide (N_2O) (Ebner et al., 2015; Sahoo and Mani, 2019). The average distance between slaughterhouses and local rendering plants is approximately 156 km in the US (Lopez et al., 2010). Therefore, the distance between the slaughterhouse and AD plant/composting plant was assumed to be the same in this study because AD plants are generally located close to the local wastewater treatment plants in the US treating sewage sludge (Shen et al., 2015). As data availability regarding transport and consumption of rendering products is limited in the literature, their information and associated impacts were excluded from the system boundary of the study. Therefore, for an equivalent comparison, transport of feedstock and AD digestate and consumption of AD digestate, rendering products, and compost were also excluded from the system boundary of the study. GHG emissions from infrastructure, maintenance, and decommissioning of AD plants were excluded from the calculation as they were reported to be <1% of total GHG emissions (Ebner et al., 2015; Sahoo and Mani, 2019). Similarly, for an equivalent comparison, GHG emissions from rendering and composting plants were also excluded. The geographical scope of this study was specific to the southeast US, where a large amount of SHW is produced (Wang et al., 2018b), and all the proposed systems of this study were applied in this region.

2.2. Life-cycle inventory

2.2.1. Feedstock characteristics

The characteristics of the feedstock used in the analysis are presented in Table 1. The total solids (TS) and VS in SHW were 28.0–31.8% and 91.9–95.4%, respectively. These values are similar to those reported by Wang et al. (2018b), who studied the production of biomethane from SHW, especially animal viscera. Protein and fat are the major constituents of SHW, accounting for 74.9–94.3% of TS, and have higher CH_4 potential than that of other organic wastes such as sewage sludge or animal manure (Deublein and Steinhauser, 2008). Either OFMSW (TS = 6.4%) or sewage sludge (TS = 14.9%) were mixed with SHW to reduce the TS content of the mixture (10.1–23.4%), which helped in pumping the mixture feedstock to the AD digester.

2.2.2. AD systems

The AD systems modeled include grinding, storage, equalization, anaerobic digesters, biogas storage, biogas cleaning, and combined heat and power (CHP) units (Fig. 1). In scenario 1, parameters of anaerobic digester operation were adopted from the results of Yoon et al. (2014), who studied the AD of poultry SHW in the batch digester and yielded 0.512 CH_4 m^3 kg^{-1} VS (Table 2). In scenario 2, parameters of anaerobic digester operation were adopted from the results of Cuetos et al. (2008), who used a semi-continuous anaerobic digester to treat a mixture of SHW and OFMSW and yielded 0.44 CH_4 m^3 kg^{-1} VS (Table 2). In scenario 3, parameters of anaerobic digester operation were adopted from the results of Borowski and Kubacki (2015), who operated a semi-continuous anaerobic digester treating the SHW and sewage sludge and yielded 0.46 m^3 CH_4 kg^{-1} VS (Table 2). The operation parameters and capacity of other units were calculated based on processing (1) 1000 kg-SHW kg d^{-1}; (2) 1000 kg-SHW d^{-1} and 5000 kg OFMSW d^{-1}; and (3) 1000 kg-SHW d^{-1} and 1000 kg sewage sludge d^{-1} in AD for scenarios 1, 2, and 3, respectively (Table 3).

The results of biomethane production from small-scale AD using anaerobic digesters can be used in the proposed large-scale anaerobic digester system with a negligible difference in methane production (Bishop et al., 2009). Empirical equations used to estimate the mass and energy balances and GHG emissions are presented in the Supplementary Material (Tables S1, S2, and S3). GHG emissions of the system were calculated according to the direct fossil fuel and electricity consumption for unit operation, CH_4 leak from all the units, digestate storage (including compost), nutrient displacement, and displacement of heat and electricity from burning biomethane in the CHP unit (Table S4).

2.2.3. Rendering system

The rendering system included conveyors, grinders, cookers, press, centrifuge, settling tank, and product storage (Fig. 1c). The system was similar to those used in previous studies (Lopez et al., 2010; Meeker and Hamilton, 2006; Ockerman and Hansen, 2000). The major cooking process chosen in this study was continuous dry rendering (a dominant process in the US) at 115 °C (388 K) and 1 atm (1.0×10^5 Pa) for 60 min (Lopez et al., 2010; Meeker and Hamilton, 2006). Data on unit capacity were mainly collected and calculated based on previous studies and manufacturer specifications (Table S1). A total of 1000 kg-SHW d^{-1} was treated. Details of the unit capacity, mass, and energy flow of the rendering system are described in the Supplementary Material (Tables S1, S2, and S3). GHG emissions of the system were calculated according to the direct fossil fuel and electricity consumption for unit operation and displacement of rendering products (fat and byproduct meal; Table S4). The fat and byproduct meal were proposed to be displaced by palm oil and soybean meal, respectively, because of

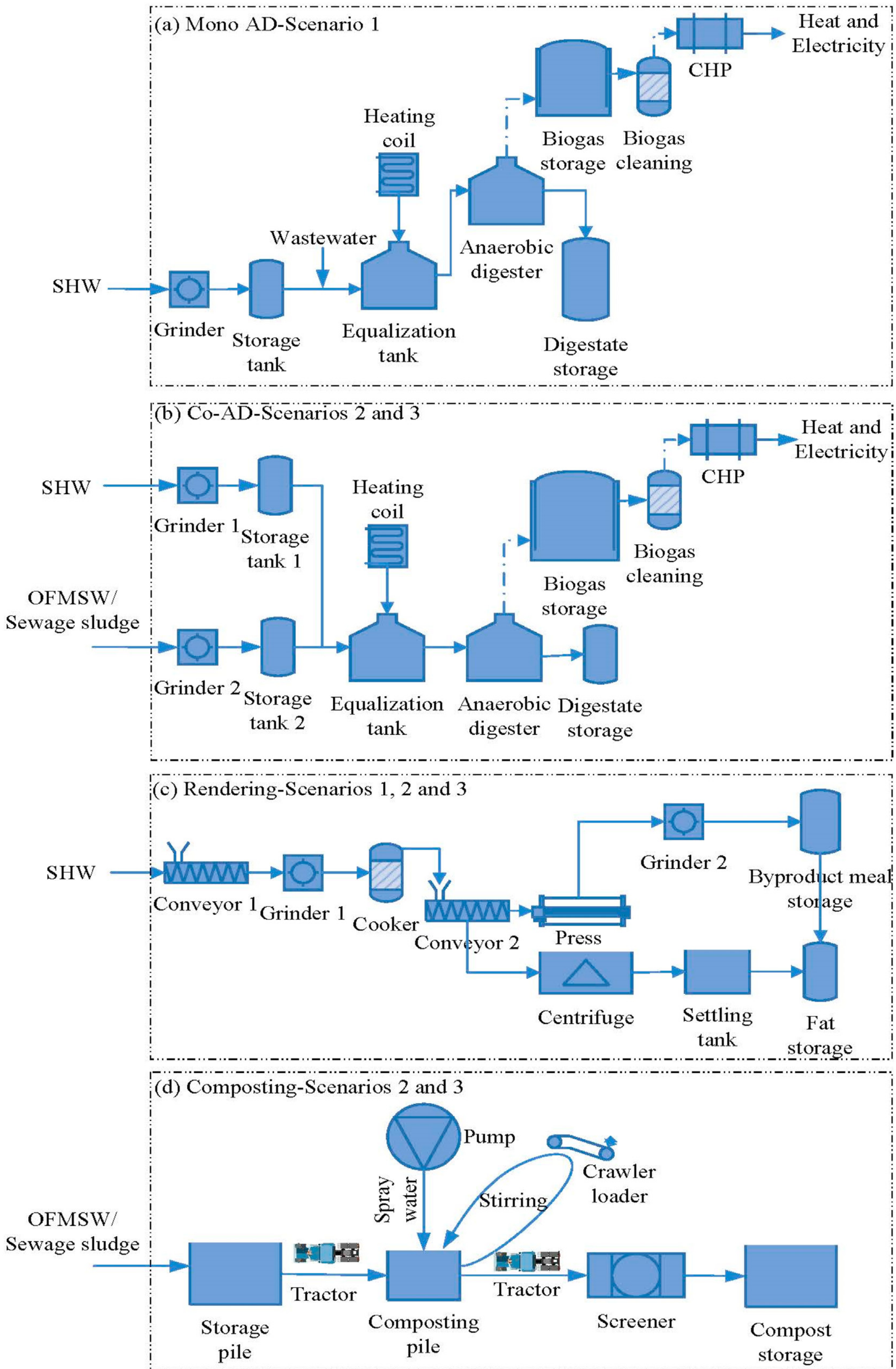

Fig. 1. System boundary of the study. In scenario 1, (a) mono-AD was compared with (c) rendering (reference system); in scenarios 2 and 3, (b) co-AD was compared with (c) rendering and (d) composting (reference system). SHW: Slaughterhouse waste; OFMSW: Organic fraction of municipal solid waste.

Table 1
Characteristics of SHW, OMFSW, and sewage sludge[a].

	Scenario 1[b]	Scenario 2[c]			Scenario 3[d]			SHW[e]
	SHW1	SHW2	OFMSW	SHW2 + OFMSW (1:5)	SHW3	Sewage sludge	SHW3 + Sewage sludge (1:1)	
TS[a] (%)	28.0	28.3	6.4	10.1	31.8	14.9	23.4	29.4 ± 2.1
VS[a] (%)	95.4	91.9	93.8	92.9	93.2	83.4	90.1	93.5 ± 1.8
Protein[a] (%)	45.9	38.9	8.3	22.7	57.1	46.1	53.6	47.3 ± 9.2
Fat[a] (%)	29.0	40.5	0.7	19.4	37.2	NA	25.4	35.6 ± 5.9
Carbohydrate/fiber[a] (%)	0.7	12.5	84.8	50.8	NA[f]	NA[f]	NA[f]	NA[f]

[a] TS: total solids; VS: volatile solids; SHW: slaughterhouse waste; OFMSW: organic fraction of municipal solid waste; VS includes protein, fat, and carbohydrate/fiber (on a dry basis). Carbohydrate is the result of VS subtracted by protein and fat, and the characteristics of the mixture were calculated from SHW and OFMSW at a ratio of 1:5 (fresh basis).
[b] SHW1 is intestine residues of poultry (Yoon et al., 2014).
[c] SHW2 is stomach and intestines of poultry (Cuetos et al., 2008).
[d] Characteristics of SHW were calculated from a mixture of meat tissue, intestines, post-flotation sludge, and bristles of pig (SHW3) (Borowski and Kubacki, 2015). The protein of sewage sludge was calculated from the total nitrogen and multiplying by 6.25. The characteristics of the mixture were calculated from SHW and sewage sludge at a ratio of 1:1 (fresh base).
[e] Mean ± standard deviation, indicating values of scenarios 1, 2, and 3.
[f] NA: not available.

Table 2
AD of SHW and co-substrate used in this study.[a].

Feedstock	Co-substrate	Temp. (°C)	OLR (kg VS m^{-3} d^{-1})	HRT (d)	Methane yield (Nm^3 kg^{-1} VS)	References
Intestine residues of poultry	NA	38	0.04	90	0.512	Yoon et al. (2014)
Stomach and intestines of poultry	OFMSW	34	3.7	25	0.44	Cuetos et al. (2008)
Mixed slaughterhouse waste of pig[b]	Sewage sludge	35	4.1	51	0.46	Borowski and Kubacki. (2015)

[a] NA: not available; OFMSW: organic fraction of municipal solid waste; OLR: organic loading rate; HRT: hydraulic retention time; VS: volatile solids; Nm^3: cubic meter at standard temperature and pressure (0 °C and 1 atm).
[b] 50% intestine, 21% meat tissue, 21% post-flotation sludge, and 8% bristle.

similar product composition (Ramirez et al., 2012; Table S5).

2.2.4. Composting system

Composting has been widely used to treat OFMSW and sewage sludge owing to its benefits of volume and pathogen reduction, as well as the production of organic fertilizer when treating organic waste (Cooperband, 2002; Onwosi et al., 2017). OFMSW and sewage sludge were modeled to be co-digested in the AD systems with SHW in scenarios 2 and 3, or composted in the reference system, as shown in Fig. 1d. The composting systems treating these wastes needed to be included with the rendering of SHW in the reference system for an equivalent comparison with the co-AD system. The composting system included storage, composting, stirring and water spraying, and compost screening (Fig. 1d). The data for the

Table 3
Mass balances (Feedstock input and products output) of SHW treatment systems.

Descriptions		Values
Scenario-1: Mono-AD of SHW		
Mass in (kg d^{-1})[a]		1000
Mass out (kg d^{-1})	Methane	105
	Digestate	79,702
Scenario-2: Co-AD of SHW and OFMSW		
Mass in (kg d^{-1})		6000
Mass out (kg d^{-1})	Methane	176
	Digestate	5501
Scenario-3: Co-AD of SHW and sewage sludge		
Mass in (kg d^{-1})		2000
Mass out (kg d^{-1})	Methane	136
	Digestate	1615
Reference system: Rendering of SHW		
Mass in (kg d^{-1})		1000
Mass out (kg d^{-1})	Byproduct meal	281
	Fat	32
Reference system: Composting of OFMSW		
Mass in (kg d^{-1})		5000
Mass out (kg d^{-1})		800
Reference system: Composting of sewage sludge		
Mass in (kg d^{-1})		1000
Mass out (kg d^{-1})		373

[a] SHW is diluted at 80 fold.

composting process were based on the information from a local composting plant treating animal waste, forestry residue, and food waste in Athens, Georgia (See Supplementary Material, Tables S1, S2, and S3). This information was obtained from personal communication with the plant manager (Wang, 2018). In scenarios 2 and 3, 5000 kg OFMSW d^{-1} and 1000 kg sewage sludge d^{-1} were treated. Details of unit capacity, mass, and energy flow of the composting system are described in the Supplementary Material (Tables S1, S2, and S3). GHG emissions of the system were calculated according to the direct fossil fuel and electricity consumption for unit operation, N_2O emissions during the composting process, and displacement of compost nitrogen nutrients (Table S4). Assuming that the composting process includes aerobic decomposition, CH_4 emissions from the proposed composting process were set at zero in this study (Sanchez et al., 2015).

2.3. Life-cycle impact (LCI) assessment

This study focused on one of the critical environmental impact categories of global warming (GHG emissions) and LCI assessment, which included the aggregation of LCI flows to the functional unit of SHW. The reference flow for the functional unit of the SHW was estimated using the mass and energy balance in combination with the LCI for each unit operation, as described in Fig. 1. Global warming impacts were quantified according to the direct and indirect GHG emissions (Ebner et al., 2015; Sahoo and Mani, 2019) from the systems in the three scenarios. Biogenic CO_2 was not included in the calculation of GHG emissions (Ebner et al., 2015). GHG emissions data (Tables S6, S7, and S8) for fossil fuel, including values of 0.151, 0.071, and 0.091 kg-CO_2-eq for 1 MJ hard coal, natural gas, and oil, respectively and 0.455 kg-CO_2-eq per kWh for electricity, were obtained from published studies (Dones et al., 2004; NREL, 2012). According to the Intergovernmental Panel on Climate Change (IPCC) reports, the 100-year global warming potential (GWP) of CO_2, CH_4, and N_2O (Table S9) are 1, 28, and 265, respectively (Ebner et al., 2015; Myhre et al., 2013). CH_4 leak from AD systems was assumed to be between 0.74% and 19.07% of the total CH_4 produced in the AD, as shown in Table S10 (Dumont et al., 2013; Sahoo and Mani, 2019). N_2O emissions from the composting process were 0.16 kg N_2O–N/1000 kg OFMSW and 0.32 kg N_2O–N/1000 kg sewage sludge (Sanchez et al., 2015). In addition, this study included the GHG emissions reduction associated with the displacement of natural gas by biomethane, synthetic nitrogen fertilizer (2.74–12.79 kg-CO_2-eq kg^{-1} N) (Kool et al., 2012; NREL, 2012) by organic nitrogen fertilizer in AD digestate and compost, palm oil (2.16–19.7 kg-CO_2-eq kg^{-1} product) (Ramirez et al., 2012) by fat, and soybean meal (0.73–0.90 kg-CO_2-eq kg^{-1} product) by byproduct meal (Dalgaard et al., 2008), depending on the different modeling methods and plantation practices. Energy production was estimated as the difference between the direct energy output and input of the systems in the three scenarios. In all systems, the input energy included fossil fuel and grid electricity, and the output energy included heat and electricity generated from biomethane combustion in the CHP unit.

2.4. Sensitivity analysis

Sensitivity analysis was conducted to evaluate the impact of uncertainties on critical input factors on GHG emissions from the proposed systems in the three scenarios. In this study, the evaluated input factors included CH_4 leak, nitrogen fertilizer displacement, and rendering product (fat and byproduct meal) displacement. The range of values for the uncertainty factors was obtained from previous studies (Tables S10 and S11).

3. Results and discussion

3.1. Mass balance

The mass flow of each system is presented in Table 3. Details including the mass flow of each unit operation in the studied systems are presented in Table S2. In the three scenarios, 2.9–10.5% of feedstock was converted to CH_4 and the rest remained in the form of CO_2 and digestate in the AD systems. In particular, large amounts of digestate were produced in scenario 1 due to the high dilution of feedstock. In the rendering system, 1000 kg of SHW was converted to 281 kg byproduct meal and 32 kg animal fat, of which 15 kg of animal fat (Table S8) was combusted to heat the cooker.

Byproduct meal, containing high protein (55%) and fat (10%) (Ockerman and Hansen, 2000), is used as animal food or organic fertilizer. Animal fat can also be added to pet food to improve palatability or converted to biofuel (Lopez et al., 2010; Ockerman and Hansen, 2000). In an industrial composting process, 5000 kg OFMSW and 1000 kg sewage sludge can be converted to 800 kg and 373 kg mature compost, respectively. These composts are typically applied to crops as organic fertilizers or soil amendments (Onwosi et al., 2017).

3.2. Energy balance

The energy balance in AD, rendering, and composting systems are presented in Table 4. Details including the electricity and fuel energy flow of each unit in all systems are also presented in Table S3. In all the systems, the energy used in the unit operation was supplied by grid electricity, fossil fuel (including coal, natural gas, and oil), and animal fat. Sewage sludge and OFMSW were added with SHW to improve bioenergy production due to their high biomethane/bioenergy potential (Ayoub et al., 2016; Cakir et al., 2018). The net energy consumption of the AD system was 1,588, −7,603, and −6037 MJ/1000 kg-SHW in scenarios 1, 2, and 3, respectively. The higher energy consumption of the AD system in scenario 1 was caused by the high dilution (80-fold) of SHW to avoid inhibition of ammonia generated as a result of protein degradation (Yenigun et al., 2013). Moreover, energy flows varied depending on the system studied. In the AD systems, biogas cleaning consumed the most electricity (313–518 MJ/1000 kg-SHW). The CHP unit generated 2340–3917 MJ of electricity and 2758–4619 MJ heat per 1000 kg-SHW. In the rendering system, the net energy consumption was 1883 MJ/1000 kg-SHW (6023 MJ/1000 kg rendering product), of which 4.3% and 95.7% were in the form of electricity and fossil liquid/gas fuel, respectively. These values are close to the results of a previous study conducted in the US, in which approximately 540 MJ electricity/1000 kg product and 5400 MJ fuel/1000 kg product were consumed in the rendering process (Lopez et al., 2010). A part of the animal fat product (15 kg/1000 kg-SHW) was burned to generate heat (580 MJ/1000 kg-SHW) for the cooker and reduce the use of fossil fuels.

Therefore, the method used to calculate energy consumption in this study can be employed to predict and optimize energy consumption of the newly built rendering system. In the composting system of OFMSW or sewage sludge, the net energy consumption was 570.6 MJ/1000 kg feedstock, of which 398.2 and 172.4 MJ were for fossil fuel and electricity, respectively. These results are similar to those reported by previous studies [for example, Zhang and Matsuto (2011)]. When treating 1000 kg-SHW, 1,588, −7,603, and −6037 MJ of net energy was consumed in the AD system of scenarios 1, 2, and 3, corresponding to 1,883, 4,734, and 2455 MJ in the reference system, respectively. Compared to the reference system, the co-AD of SHW with OFMSW or sewage sludge produced large amounts of biomethane that was used as bioenergy, and its

Table 4
Energy flow of anaerobic digestion and rendering of slaughterhouse waste and composting system.[a].

Descriptions	Pre-treatment	Process	Post-treatment	Total
Scenario-1: Mono-AD of SHW[b]				
In (MJ d^{-1})	6372.0	0	313.2	6685.2
Out (MJ d^{-1})	0	0	5097.6	5097.6
Net consumption (In−Out) (MJ d^{-1})	6372.0	0	−4784.4	1587.6
Scenario-2: Co-AD of SHW and OFMSW[b]				
In (MJ d^{-1})	414.0	0	518.4	932.4
Out (MJ d^{-1})	0	0	8535.6	8535.6
Net consumption (In−Out) (MJ d^{-1})	414.0	0	−8017.2	−7603.2
Scenario-3: Co-AD of SHW and sewage sludge[b]				
In (MJ d^{-1})	144.0	0	403.2	547.2
Out (MJ d^{-1})	0	0	6584.4	6584.4
Net consumption (In−Out) (MJ d^{-1})	144.0	0	−6181.2	−6037.2
Reference-1: Rendering of SHW[b]				
In (MJ d^{-1})	10.8	1800.0	72.0	1882.8
Out (MJ d^{-1})	0	0	0	0
Net consumption (MJ d^{-1})	10.8	1800.0	72.0	1882.8
Reference-2: Composting of OFMSW[b]				
In (MJ d^{-1})	0	2703.6	147.6	2851.2
Out (MJ d^{-1})	0	0	0	0
Net consumption (In−Out) (MJ d^{-1})	0	2703.6	147.6	2851.2
Reference-3: Composting of sewage sludge[b]				
In (MJ d^{-1})	0	540.0	28.8	568.8
Out (MJ d^{-1})	0	0.0	0.0	0.0
Net consumption (In−Out) (MJ d^{-1})	0	540	28.8	568.8

[a] In scenarios 1, 2, and 3, the pretreatment included a grinder, storage tank, and equalization tank; the process included an anaerobic digester; the posttreatment included biogas storage, digestate storage, biogas cleaning, and CHP. In reference 1, the pretreatment included conveyor and grinder; the process included a cooker; the posttreatment included conveyor, press, and grinder centrifuge. In references 2 and 3, the pretreatment included feedstock storage; the process included composting pile; the posttreatment included a screener.

[b] The mass flow were 1000 kg-SHW d^{-1} with 80-fold dilution for scenario 1, (1000 kg-SHW + 5000 kg OFMSW) d^{-1} for scenario 2, (1000 kg-SHW + 1000 kg sewage sludge) d^{-1} for scenario 3, 1000 kg-SHW d^{-1} for reference 1, 5000 kg OFMSW d^{-1} for reference 2, and 1000 kg sewage sludge d^{-1} for reference 3. Scenario 1 included reference system 1. Scenario 2 included reference systems 1 and 2. Scenario 3 included reference systems 1 and 3.

application could significantly improve GHG reduction in the meat industry.

3.3. GHG emissions

Fig. 2 shows GHG emissions from treating SHW with AD in the three scenarios and a comparison with the reference systems currently practiced by the US meat industry. GHG emissions from fossil fuel, electricity, gas leak/emission (CH_4 and N_2O) are direct emissions. The results show that AD systems in scenarios 1, 2, and 3 had net GHG emissions (cradle-to-grave) of 200.0, −426.8, and −524.0 kg-CO_2-eq/1000 kg-SHW, respectively. The net GHG emissions of the reference systems for scenarios 1, 2, and 3 were −309.1, 296.2, and −152.5 kg-CO_2-eq/1000 kg-SHW, respectively.

In scenario 1, direct GHG emissions from the mono-AD system (834.2 kg-CO_2-eq/1000 kg-SHW) were more than six times higher than those of its reference system, that is, rendering (106.7 kg-CO_2-eq/1000 kg-SHW). Large amounts of emissions in the mono-AD system were due to the use of fossil fuels during the preheating of SHW. Therefore, preheating of feedstock and CH_4 leak were the major GHG emission sources in the mono-AD system of scenario 1, accounting for 63.9% and 31.3% of direct GHG emissions, respectively. The primary direct emission of the rendering process was from the cooker, accounting for 90.4% of the direct emissions. The GHG emissions reduction potential of the mono-AD system of 634.2 kg-CO_2-eq/1000 kg-SHW due to the displacement of heat, electricity, and fertilizer was approximately 1.5 times that of its reference system, including the displacement of rendering products.

Compared to the reference system, the mono-AD system was not environmentally beneficial because of the positive net GHG emissions (200.0 kg-CO_2-eq/1000 kg-SHW). In scenario 2, co-AD had lower direct GHG emissions (538.3 kg-CO_2-eq/1000 kg-SHW) than the emissions from reference systems of rendering the SHW and composting the OFMSW (728.5 kg-CO_2-eq/1000 kg-SHW). Gas leaks/emissions (CH_4 or N_2O) were the major direct emissions, accounting for 81.1% of the emissions in the co-AD system and 51.4% in the reference system. Moreover, the co-AD system of SHW produced heat, electricity, and nitrogen-rich digestate, which displaced fossil resources and nitrogen fertilizer. Moreover, the total GHG displaced (965.1 kg-CO_2-eq/1000 kg-SHW) was two times that of the reference system. In scenario 3, co-AD produced 25% higher direct GHG emissions (400.5 kg-CO_2-eq/1000 kg-SHW) than emissions from the reference system (306.0 kg-CO_2-eq/1000 kg-SHW). Additionally, emissions from CH_4 gas leak or N_2O were the primary direct emissions, accounting for 91.6% of the emissions in the co-AD system and 48.9% in the reference system. The total GHG displaced (965.1 kg-CO_2-eq/1000 kg-SHW) was approximately two times that of the reference system. Scenario 3 had the best GHG profile of all the systems evaluated because of GHG displacement. Co-AD of SHW and OFMSW or sewage sludge had a significantly higher GHG reduction than the reference system. Nevertheless, there are still many means to improve GHG reduction of the AD system. The CH_4 leak was the major GHG emission source in the co-AD systems, accounting for 81.1%–84.1% of the direct emissions. Particularly, up to 11% of total CH_4 could be emitted from the digestate storage when the container is not covered (Dumont et al., 2013) (Table S10). Therefore, the feedstock should be stored only for a short duration and the digestate storage container should be covered to avoid excessive CH_4 emissions from the entire AD system (Dumont et al., 2013). Furthermore, the capacity and efficiency of the CHP engine should be improved to reduce CH_4 leaks. By minimizing CH_4 leaks, GHG emissions from the AD system can be significantly reduced; thus, the AD system has more environmental benefits than those of the rendering and composting system when

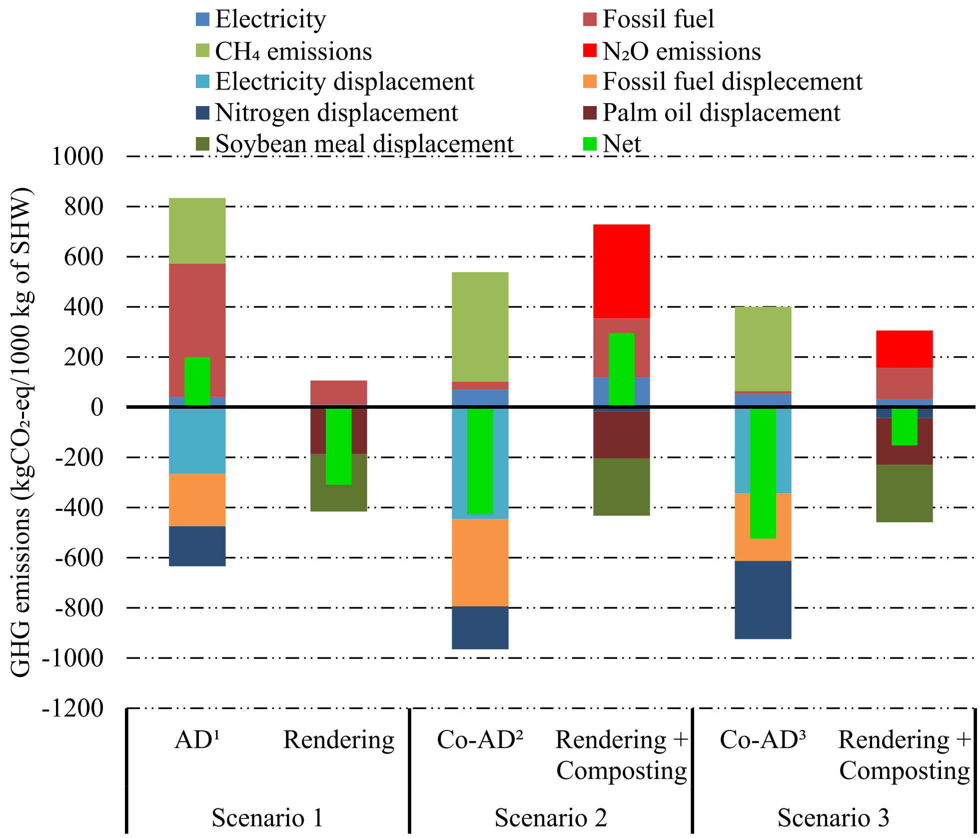

Fig. 2. Greenhouse gas (GHG) emissions from scenarios 1, 2, and 3 along with reference systems ([1]AD of SHW, [2]Co-AD of SHW with OFMSW, and [3]Co-AD of SHW with sewage sludge).

treating SHW. N_2O emissions from composting were the major emission sources in the reference systems, accounting for 48.9–51.4% of the direct emissions. In the composting process, ammonia is converted to nitrate through nitrification and then to nitrite and N_2O through denitrification. Most of the N_2O (>95%) is emitted during a later period of composting when active carbon is used. Good management of the composting process (for example, intermittent feeding of fresh feedstock and a certain frequency of pile turning) can reduce N_2O emissions (Chen et al., 2015; He et al., 2001).

GHG emissions reduction due to thermal energy (heat) and electricity displacement (by burning biomethane) were −794.2 and −612.8 kg-CO_2-eq/1000 kg-SHW, accounting for 82.3% and 66.3% of the total displacement in the co-AD systems of scenarios 2 and 3, respectively. This indicates that increasing the biomethane production by co-digestion can significantly contribute to GHG reduction in the AD system.

3.4. Sensitivity analysis

A sensitivity analysis of the three scenarios was conducted to evaluate the impact of uncertainty of the data used on the LCA results. The CH_4 leak of AD systems, nitrogen fertilizer displacement, and rendering product displacement were the three major factors impacting GHG emission calculations in each scenario. Their impacts are shown in Fig. 3 and discussed in the following sections.

3.4.1. CH_4 leak from AD systems

Because of the imperfect sealing of operational units in AD systems, CH_4 could leak from feedstock storage, digester, digestate storage, biogas upgrading, and CHP operation. Fig. 3a shows the impact of CH_4 leak (0.74–19%) on GHG emissions of the AD system in the three scenarios. Because of the strong GWP effects of CH_4, uncertainty in estimating CH_4 leak from the AD system has a large impact on the GHG emission results. In scenario 1, the GHG emissions from the AD system were always higher than those from the reference system because of the high energy consumed for preheating the feedstock for the AD digester. In scenarios 2 and 3, when the total CH_4 leak was 0.74%, GHG emissions from the AD system were −911.5 and −898.0 kg-CO_2-eq/1000 kg-SHW, respectively, which were considerably lower than that of the reference system, including the rendering and composting system. However, when the total leak was 19.1%, the GHG emissions from the AD systems was comparable to that of the reference system in scenario 3.

3.4.2. Nitrogen fertilizer displacement

A large amount of digestate was produced from the AD system and was likely used as a nitrogen fertilizer. Fig. 3b shows the impact of nitrogen fertilizer displacement on GHG emissions in all the systems in the three scenarios. In scenario 1, GHG emissions from the AD system could be reduced using high nitrogen fertilizer displacement; however, they were still significantly higher than those of the reference system (rendering). In scenarios 2 and 3, the nitrogen fertilizer displacement significantly impacted GHG emissions from all the systems. The nitrogen fertilizer displacement led to maximum reductions of 537.4–725.1 kg-CO_2-eq/1000 kg-SHW in the AD system compared to 180.1 kg-CO_2-eq/1000 kg-SHW in the reference system (rendering and composting). With minimum nitrogen displacement, GHG emission reduction from the AD system

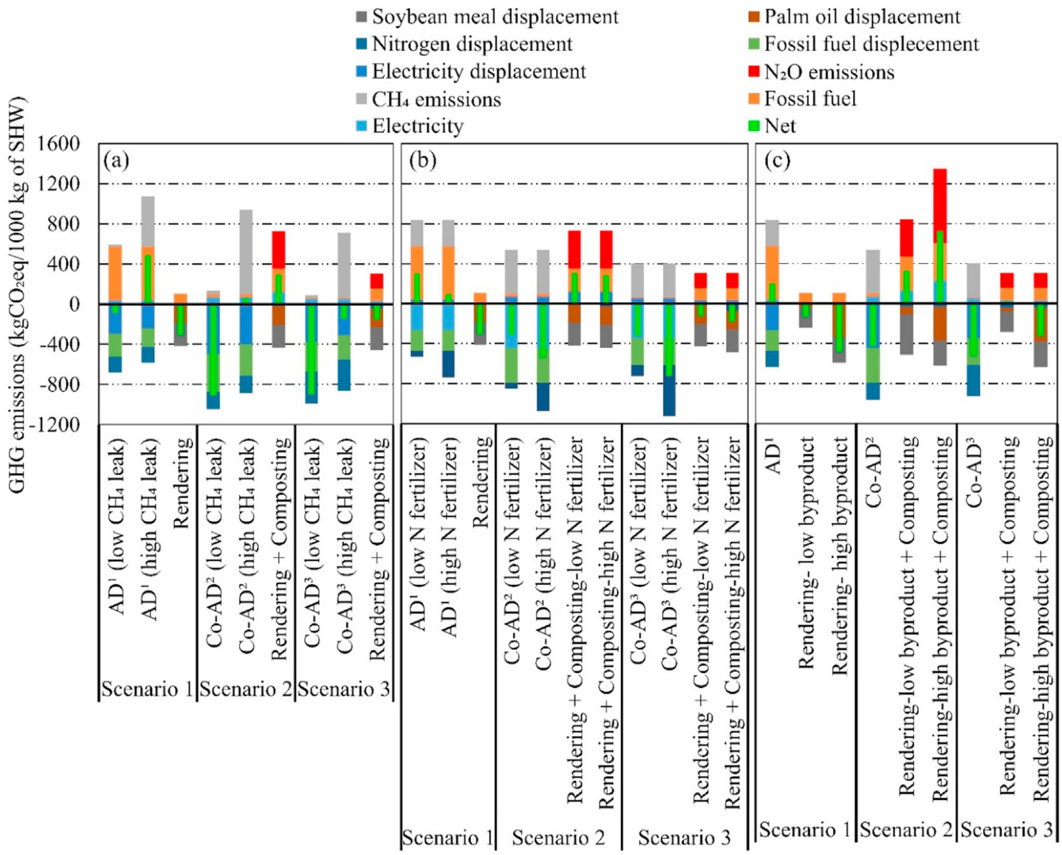

Fig. 3. Sensitivity analysis of factors impacting GHG emissions. (a) CH_4 leak, AD-low CH_4 leak, and AD-high CH_4 leak were calculated using GHG emissions considering low and high CH_4 leak from the AD system; (b) Nitrogen fertilizer displacement; (c) rendering product displacement ([1]AD of SHW, [2]Co-AD of SHW with OFMSW, [3]Co-AD of SHW with sewage sludge).

was still considerably higher than that of the reference system.

3.4.3. Rendering product displacement

Fat and byproduct meal can be used to displace palm oil and soybean meal, respectively, owing to their similar characteristics and industrial usage (Table S5) (Ramirez et al., 2012). Fig. 3c shows the impacts of rendering product displacement on GHG emissions from the AD systems of the three scenarios. The uncertainty of palm oil and soybean meal displacement can cause a large variance in GHG emission calculations from the rendering system (−134.3 to −484 kg-CO_2-eq/1000 kg-SHW). In scenario 1, the rendering of SHW reduced the cradle-to-grave GHG emissions compared to mono-AD (200 kg-CO_2-eq/1000 kg-SHW). However, the co-AD scenarios reduced the GHG emissions (−427 and −524 kg-CO_2-eq/1000 kg-SHW) compared to the rendering of SHW and composting of OFMSW and sewage sludge. In scenarios 2 and 3, the cradle-to-grave GHG emissions of reference system were 329−727 and −327−22 kg-CO_2-eq/1000 kg-SHW, respectively, which were considerably higher than those of co-ADs.

3.5. Implication of co-AD of SHW with OFMSW and sewage sludge

Co-AD of SHW with OFMSW or sewage sludge has been suggested to achieve maximum biomethane production and GHG reduction. These wastes can be easily collected from local municipal solid waste facilities and wastewater treatment plants and can be used for biomethane production (Shen et al., 2015). Several wastewater treatment plants in the US are equipped with AD facilities to convert sewage sludge and co-substrate to biomethane (Shen et al., 2015).

In scenarios 2 and 3, AD of SHW could provide large amounts of bioenergy locally with many meat processing plants or slaughterhouses (for example, the southeast US). Table 5 summarizes the GHG emissions from AD of animal manure, OFMSW, and sewage sludge reported in different studies. The reduction in emissions of SHW co-digestion found in this study was generally higher than those of other feedstocks, which could be due to differences in feedstock characteristics, digester operation, study scope, among others. Therefore, using co-AD to treat SHW can have a relatively higher benefit of GHG reduction than other common organic wastes used in AD.

In the US, more than 18 million tons of SHW are produced annually by the meat industry (Wang et al., 2018b). AD of organic wastes for bioenergy recovery is still underdeveloped because of the technical challenges, unpredictable energy market, and lack of public awareness in the US. The federal and state governments have enacted policies, including the Renewable Index Number (RIN) and the Rural Energy for America Program (REAP), to stimulate the development of AD and other renewable energy projects. Owing to these policies and increased public attention on sustainable development, the AD of SHW is being accepted and developed by the meat production and rendering industries (Wang et al., 2018b).

Fig. 4a−d shows energy generation and GHG emissions from the anaerobic treatment of SHW (viscera and meat tissue) in the

Table 5
GHG emissions from the anaerobic digestion of different feedstocks in LCA studies.[a].

Feedstock	GHG emissions (kg-CO_2-eq/1000 kg feedstock)	Digester	Scope	Reference
Manure + food waste	−35.7	CSTR, 41 °C, HRT = 28 days	Transport, gases leak, land application, displaced electricity, and inorganic fertilizer	Ebner et al. (2015)
Pig manure	359	Covered anaerobic lagoon, HRT = 162 days	Liquid and solid manure storage, anaerobic digestion, digestate and liquid effluent treatment, composting, field application, avoided electricity and inorganic fertilizer	Ramírez-Islas et al. (2020)
OFMSW	28.6	Dry anaerobic digestion, residence time = 21 days	CHP, digestate application, landfill, transportation, electricity, and natural gas	Nordahl et al. (2020)
OFMSW	−64.7	CSTR, 35 °C	Pre-treatment, use of biogas, treatment, and use of digestate	Evangelisti et al. (2014)
Sewage sludge	−23.9	Assumed 61.2 kg per day	Feedstock supply, anaerobic digester infrastructure, biogas production and utilization, dewatering, transport&storage, and agricultural spreading	Singh et al. (2020)
Sewage sludge	40	Mixed digester tank, 36 °C, HRT = 12−20 days	Displacement, transport, plant emissions, heating, land, and chemicals	Mills et al. (2012)
SHW	2.5	38 °C, HRT = 90 days	CH_4 emissions, fossil fuel consumption, electricity consumption, electricity displacement, fossil fuel displacement, nitrogen displacement	This study
SHW + OFMSW	−71.1	34 °C, HRT = 25 days		
SHW + sewage sludge	−262	35 °C, HRT = 51 days		

[a] GHG: greenhouse gases; LCA: life-cycle assessment; CSTR: continuous stirred tank reactor; HRT: hydraulic retention time; OFMSW: organic fraction of municipal solid waste; SHW: slaughterhouse waste.

southeast US, the geographical scope of the study targets, and the highest SHW production region. Annual total energy production and GHG reduction were estimated to be 22.4−28.5 × 10^6 GJ and 1599.6−1963.8 × 10^6 kg-CO_2-eq, respectively, by anaerobic treatment of SHW in this region. Georgia, Alabama, and Arkansas were the top three SHW dischargers owing to their high broiler meat production (Table S12). Reportedly, there are 11−50 AD systems that are currently operational and 300−443 AD systems that can be potentially developed in the future in each state (ABC, 2020). The primary feedstock for the potential AD systems is manure and sludge from wastewater treatment plants, and if SHW is added as a feedstock, the development of AD in this region can be considerably improved.

The economic benefits of AD operation can also be attained by selling carbon credits in the global market ($13.6/1000 kg-$CO_2$-eq). The benefits can be up to $26.7 million in this region, including $4.5 million for Georgia, $3.9 million for Alabama, and $3.5 million for Arkansas (Fig. 4e and f). Carbon prices have gradually increased in recent years (CARB, 2020), which can potentially improve the economy of using AD to treat SHW. In addition, constructing new AD systems would generate large investments (that is, $45 billion in capital deployment) and jobs (that is, 374,000 during short-term construction and 25,000 permanent jobs) in the US (ABC, 2020). Therefore, the advantages of bioenergy, including the benefit to the environment and economy, could significantly promote the sustainability of the meat industry, improve energy security, and accomplish GHG reduction target in the US at local and national levels.

4. Limitation of this study

The scope of the study does not include the transport of feedstock and products (including AD digestate, compost, and rendering products), land application of AD digestate and compost, and consumption of rendering products. Moreover, this study did not include different types of supply chains of SHW, which could substantially impact environmental emissions (Yang, 2017). Nitrogen-rich digestate and compost could potentially emit a large amount of N_2O gas from land application. Accurate quantification of N_2O emissions should be addressed in the future. Other factors could also impact the GHG emission calculations from the systems in the three scenarios, such as allocation of fuel used for electricity and heat, anaerobic co-digestion operation mode (that is, feedstock ratio and OLR). In addition, there are still some inaccuracies and uncertainties in this LCA study. An exergy-based method is suggested for a comprehensive analysis of bioenergy production from the AD of SHW (Rosen, 2018).

5. Conclusions and future prospects

Typically, SHW is rendered to produce various animal products, including meals. However, because rendering SHW is a very energy-intensive process, these products have large environmental impacts. Anaerobic co-digestion (mixing OFMSW or sewage to SHW) are potential alternatives and better options to treat SHW as well as to produce biofuels such as biomethane, which can be burned to produce renewable electricity. This study used SHW in a cradle-to-gate LCA approach to quantify the energy use and GHG emissions to treat SHW using AD (which generates biogas and renewable electricity) and rendering (a business-as-usual case). Further, the results were used to estimate the future GHG reduction potential in the southern US by adopting the AD treatment of SHW. This study analyzed various SHW treatments to determine the system with (possible) the highest positive energy flows while reducing GHG emissions. Two scenarios were identified as the best performers, anaerobic co-digestion of SHW with OFMSW or sewage sludge, compared to the reference systems of rendering and composting. These two co-AD systems produced 6037−7603 MJ of energy/1000 kg-SHW while reducing 426.8−524.0 kg-CO_2-eq/1000 kg-SHW. The other systems consumed energy and did not show significant GHG reduction potential compared to the two co-AD systems.

A sensitivity analysis was conducted to evaluate the impact of CH_4 leak from the AD systems and nutrient displacement on the GHG emissions of the systems. There were large uncertainties in these two parameters, which led to a large variance in the calculated GHG emissions. To improve the environmental performance of the AD systems evaluated in this study, minimizing CH_4 leaks along the supply chain should be a priority, given its large impact on the GHG emission profile.

This study indicates that the sustainability of the meat industry could be improved using co-AD of SHW with other organic wastes

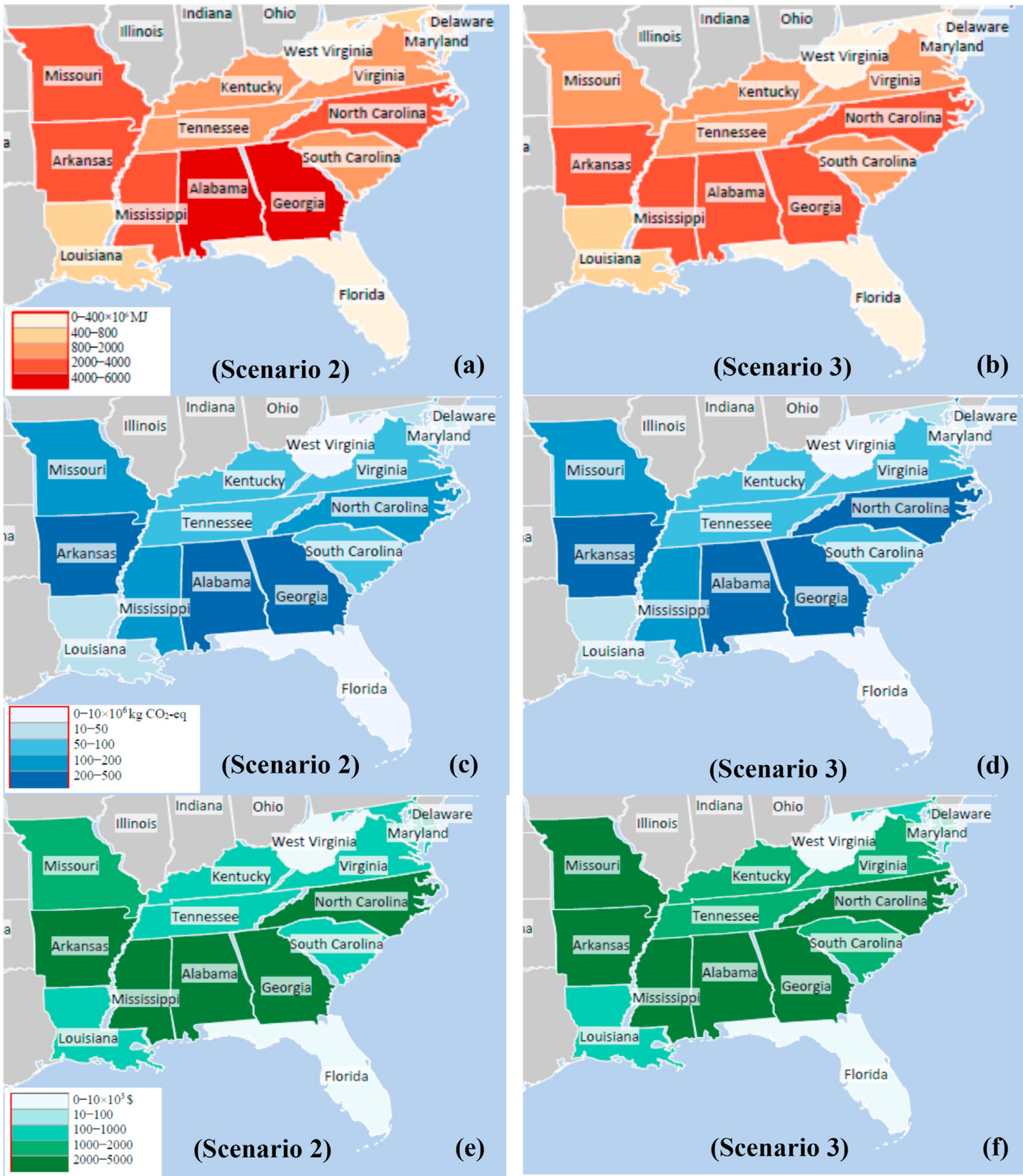

Fig. 4. Annual potential of energy production (a and b), GHG reduction (c and d), and carbon price (e and f) of treating SHW with co-AD system (scenarios 2 and 3) in the southeast US.

such as OFMSW and sewage sludge. This will reduce GHG emissions while producing a large amount of biomethane as a renewable energy source. Considering these two properties, co-locating these co-AD systems in the highest SHW regions, such as the southeastern US, can improve their environmental performance.

More studies on the co-digestion of SHW and other organic wastes at different scales should be conducted. Other types of SHW, including blood waste and feathers, need to be included in future LCA studies to comprehensively compare AD with rendering, composting, or landfilling. In addition, the composting operation should be optimized to alleviate N_2O gas emissions because of its high GWP value. The exergy-based method is suggested to comprehensively evaluate AD of SHW. Such studies can provide various means by which slaughterhouses and rendering plants can treat SHW in an eco-friendly manner to enhance the environment and energy sustainability of the meat industry.

CRediT authorship contribution statement

Shunli Wang: Conceptualization, Investigation, Data curation, Methodology, Writing - original draft. **Kamalakanta Sahoo:** Conceptualization, Data curation, Visualization, Software, Writing - review & editing. **Umakanta Jena:** Writing - review & editing. **Hongmin Dong:** Supervision, Writing - review & editing. **Richard Bergman:** Resources, Methodology, Writing - review & editing. **Troy Runge:** Formal analysis, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.126038.

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