

Sustainability Research and Training Program

PUBLICATION SRTP-2017-01-01

Sequential Energy and Compost Production from Organic Residues

Sara Pace, Jesus Fernandez Bayo, Ramin Yazdani, Christopher Simmons, Alissa Kendall, Mutasem Fadel and Jean VanderGheynst

Sponsored by



In Collaboration with the BCBCB Consortium



THE AMERICAN UNIVERSITY IN CAIRO
الجامعة الأمريكية بالقاهرة



UC DAVIS
UNIVERSITY OF CALIFORNIA



Suggested Citation:

Pace, S., Bayo, J.F., Yazdani, R., Simons, C., Kendall, A., Fadel, M., VanderGheynst, J. 2017. Sequential Energy and Compost Production from Organic Residues. SRTP-2017-1-1, Sustainability Research and Training Program, University of California, Davis, California. <http://srtp.ucdavis.edu/>

Progress Report: Sequential energy and compost production from organic residues

Contributors: Sara Pace, Jesus Fernandez Bayo, Ramin Yazdani, Christopher Simmons, Alissa Kendall, Mutasem Fadel and Jean VanderGheynst

Reporting period: October 1, 2014-December 31, 2016

Project Abstract: Composting and amendment of compost to soil can improve the sustainability of an ecosystem and community. However, composting processes, especially those treating wastes that are readily degradable, require energy to aerate and water to maintain biological activity, and can emit greenhouse gases (GHG). Some organic residue sources, including many that originate from food processing and handling operations, contain organic matter fractions that can be readily converted to energy through anaerobic digestion. The overall goal of the proposal is to develop tools that allow organic waste management specialists to evaluate digestion and composting conversion scenarios to sustainably convert organic residues to valuable soil amendment. This goal will be met through coordinated research and training efforts at UC Davis and the American University of Beirut that will involve laboratory studies to elucidate the potential for methane and heat production from organic wastes followed by validation studies in the field.

Objective 1. Measure methane production potential from anaerobic digestion of food waste and green waste and generate digestate for composting studies.

Experimental set-up for AD experiments

Studies were conducted to (1) assess which type of feedstock combinations and digestion conditions are ideal for stabilizing organic residues under aerobic and anaerobic conditions, and (2) collect data for the life cycle assessment model. Several parameters were tested to understand their impact on digestion. In both anaerobic (AD) and aerobic digestion (AED) various feedstocks with different C/N ratios were examined as well as total solids content of the process. The selected feedstock and inoculum type were the same for both types of digestion, however, inoculation level and water content varied. The feedstocks included model green waste (GW) and food waste (FW) mixed at different levels to obtain different initial C/N ratios (Table 1). The material composition of the GW was based on the California 2008 Statewide Waste Characterization Study, published by the California Integrated Waste Management Board in 2009. The GW components and mass fractions included leaves (41.0% dry basis), grass (11.4% dry basis), pruning and trimmings (40.4% dry basis), and branches and stumps (7.2% dry basis). The model FW was a variety of dog food that was selected according to its similar composition (carbohydrate, protein, fiber and fat) to that of the food waste from Tadweer. The inoculum comprised a digestate sludge (DS) from the UC Davis anaerobic digester. For both AED and AD experiments, C/N was varied between 17 and 34.

Table 1. Properties of the model food and green wastes

	DM	C total	N total	C/N	Ash	VS	FSP
	%	%	%		%	%	% (g H ₂ O/g dry weight)
Food waste	91.51	45.50	3.69	12.35	5.34	94.66	410
Green waste	92.09	48.8	1.43	34.1	4.70	95.30	472
Digestate	3.5	1.20	0.34	3.57	-	-	-

DM = Dry Matter; FSP = Fiber saturation point

Effect of C/N ratio of the feedstock in the AD rate

Batch anaerobic digesters consisted of 250-mL glass media bottles fitted with modified caps connected to an in-line check valve. Digesters were then loaded with 7.5 g (dry weight) of the corresponding GW/FW mix and 92.5 g of DS (wet basis). Three replicates per feedstock were used. Following preparation, digesters were incubated at 55 °C until methane production had dropped and stabilized (14-15 days). Methane and carbon dioxide content of the biogas produced from each digester was measured via a MicroOxymax respirometry system (Columbus Instruments, Columbus, OH) that was run in anaerobic mode according to the manufacturer's instructions. Total biogas production for each digester was estimated by summing the measured volumes of evolved carbon dioxide and methane, the primary constituents of biogas. Methane quality was calculated as the volumetric fraction of methane in the biogas.

The methane data collected during AD were using to predict the biomethane potential (BMP) using a Modified Gompertz model (Nielfa et al., 2015)

$$\text{BMP} = \gamma \exp \left(- \exp \left(\frac{k(\lambda - t)e^1}{\gamma} + 1 \right) \right) \quad \text{Eq 1}$$

where γ (mlCH₄/g VS) is maximum volume accumulated at an infinite digestion time (t), k is the specific rate constant (ml CH₄/g VS/d) and λ is the lag phase time constant (days). This equation is a modification of the Gompertz model and assumes biogas production is proportional to microbial activity.

Outcomes

The efficacy of the anaerobic digestion process was assessed by monitoring CH₄ and CO₂ evolution and estimating the biomethane potential (BMP). After 14d of incubation the peak in methane production had been achieved and the rate of methane production began to decline for all samples (Figure 1a). At this time, the amount of CH₄ recovered corresponded to more than 85% of CH₄ estimated as BMP for each C/N, meaning that at this time most of the potential CH₄ was already produced. Treatments with a C/N of 34 and C/N of 17 yielded the lowest BMP values (67.46 and 50.45 mL CH₄/g VS, respectively, Figure 1b). Mixtures with C/N ratios of 20 and 27 exhibited an average BMP of 108 and 118 mL CH₄/g VS, respectively. The maximum BMP (171 mL CH₄/g VS) was observed at a C/N ratio of 23. The ANOVA analysis showed

significant differences ($P < 0.001$) between some of the treatments. Specifically, the HSD-Tukey analysis showed that methane production at C/N ratio of 23 was significantly higher than at C/N ratio of 34 ($P = 0.014$) and C/N 17 ($P = 0.006$). Regarding the quality of the biofuel produced (Figure 1b), the best quality was observed for the C/N ratio of 23 (1.96, that is, 66% of the biogas was CH_4) and the lowest quality was observed for the C/N ratio of 34 (1.39, where only 59% of the biogas was CH_4). However, differences in the biogas quality between the different C/N ratios were not significant.

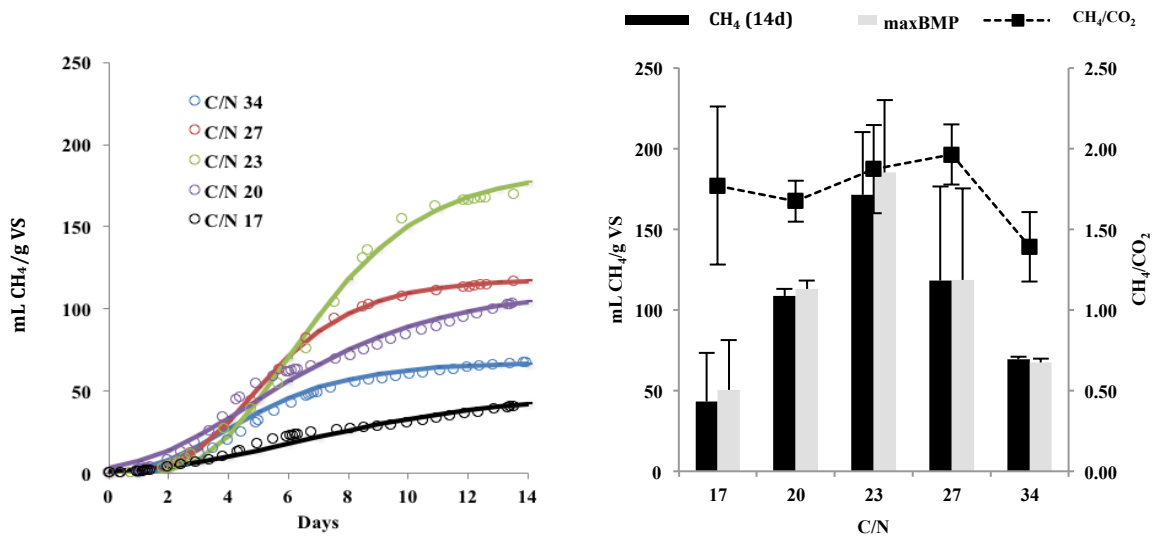


Figure 1. (A) Mean ($n=3$, points) cumulative and predicted (full line) CH_4 evolution at different C/N ratios. (B) Cumulative CH_4 emission after 14 days, estimated maximum biomethane potential (BMP), and biogas quality assessed by the CH_4/CO_2 ratio (right axis). The COD and total soluble N of the liquid digestate provided information on the end stage of anaerobic digestion. The highest COD and N levels were found for the C/N ratio of 17.

The estimated BMP_{max} showed a significant second order polynomial correlation with the C/N ratio ($P=0.009$). The fitted model using least minimum squares ($R^2=0.55$) was:

$$\text{BMP}_{\text{max}} = -710.9 + 67.9(\text{C/N}) - 1.33(\text{C/N})^2 \quad \text{Eq 2}$$

Based on this model the optimal C/N ratio for these feedstocks would be 25.5 producing up to 159 mL $\text{CH}_4/\text{g VS}$.

Moreover, COD and total soluble N showed a significant linear negative correlation with the C/N ratio (Figure 2, $P=0.002$ and $P=0.001$ for N and COD, respectively). The sample with the C/N ratio of 17 showed the highest COD and N in agreement with the low degradation rate observed. During anaerobic digestion, unbalanced C/N ratios can result in high total ammonia nitrogen release and/or high volatile fatty acids (VFA) accumulation that can inhibit the AD process (Yan et al., 2015).

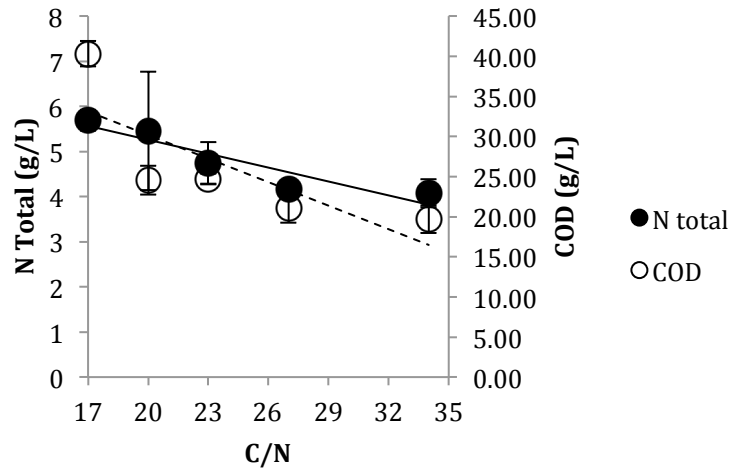


Figure 2. Total N (left axis) and COD (right axis) of the liquid digestates after digestion as a function of initial C/N ratio. Bars represent the standard deviation (n=3) and lines represent the linear relationship between dependent variables and initial C/N ratio.

Objective 2. Measure decomposition rates and greenhouse gas emissions during composting of digestate.

Experimental set-up for composting experiments

The composting experiments were completed to examine both the influence of C/N and moisture on stabilization rate. The C/N ratios used were the same as previously described. The experimental water content target was approximately 60% (low, L), 73% (medium, M) and 85% (high, H) of the Fiber Saturation Point (FSP), which was equivalent to 62%, 69% and 74% moisture content (wet basis). To estimate the FSP, feedstocks were soaked in water for an hour and then the excess water was left for drainage for another hour. The moisture content of the saturated feedstock was then measured by weighing the wet samples before and after drying at 105°C for 24h. The FSP was expressed as the percentage of the mass of water held by a given mass of feedstock (dry weight).

As for the AD system, the inoculum for composting experiments was DS. To confirm that DS was stable, aerobic bioreactors loaded with only DS were run in parallel with the samples and no CER was observed (data not shown). The inoculum levels for the low, medium and high moisture content studies were 5%, 8% and 10% of the initial feedstock (dry mass equivalent), respectively. These differences were attributed to the high moisture content of the DS (3.5% total solids), so inoculum amount was adapted to the target moisture content. When necessary, distilled water was added to the reactors to reach the target moisture content.

A detailed description of the aerobic systems can be found elsewhere (Reddy et al., 2013). Prior to loading the bioreactors, the feedstock, the inoculum and distilled water were

thoroughly mixed to reach the target moisture content. The 250-mL bioreactors (four replicates) were loaded with 6 g (dry weight) of each mixture. To maintain aerobic conditions, reactors were supplied with air at a rate of 20 mL/min and were incubated at 55 °C in a temperature-controlled incubator. As for the AD experiments, incubation was stopped after the maximum respiration peak was achieved and the rate was low and approximately stable (approx. 14-15 days). In the composting system, the respiration rate in terms of CO₂ evolution rate (CER) and cumulative respiration (cCER) were monitored as stabilization indicators. From these values, the maximum cumulative respiration (cCER_{max}) was estimated by fitting cumulative respiration as a function of time to an exponential model as expressed in Eq. 3.

$$cCER = cCER_{max} \times (1 - e^{-kt}) \quad \text{Eq 3}$$

Effect of C/N ratio and moisture in the aerobic digestion process

Treatments experienced variable cumulative respiration depending on the C/N ratio and moisture content (Figure 3). For the same moisture content (60% of the FSP), the lowest cCER was for the treatments with C/N ratios of 17 and 34, whereas the highest respiration was recorded at the C/N of 20. In general, performance of treatments with lower C/N ratio was affected by excess water; microbial activity ceased after 24-48 hours of incubation for mixtures with C/N ratios of 17, 20 and 23 at the highest moisture content (Figure 3(B)). Treatments with a C/N ratio of 17 had no detectable activity at any moisture content while treatments with a C/N ratio of 20 failed at 70% of the FSP. However, the same mixture at 60% of the FSP presented the highest cCER for all samples (1231 mg CO₂/g dry weight). Treatments with GW only (C/N = 34) had a cCER of 489 and 226 mg CO₂/g dry weight at 80% and 60% of the FSP, respectively.

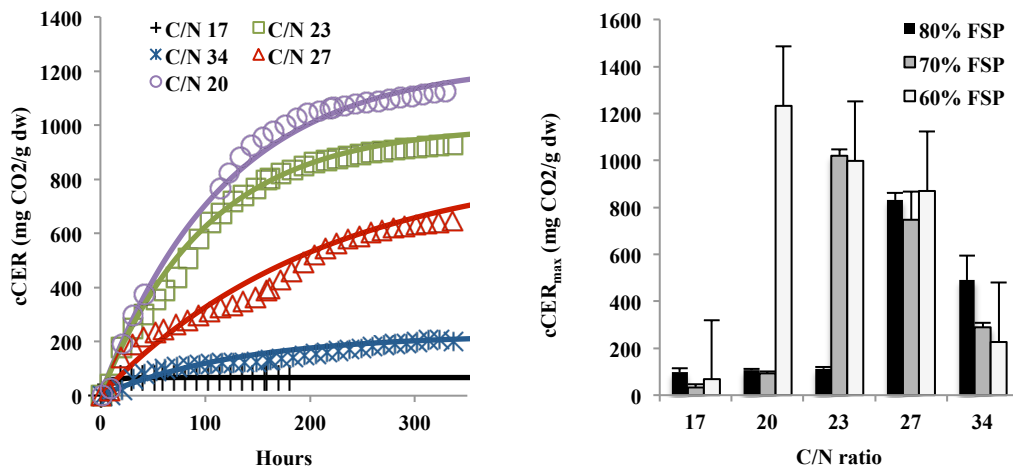


Figure 3. (A) Cumulative CO₂ evolution (cCER) of feedstocks incubated at 60% of the FSP at different C/N ratios. (B) Estimated maximum cCER as a function of C/N ratio.

The initial moisture content in composting is important as it provides mobility to the microorganisms and helps with nitrogen mineralization and polysaccharide hydrolysis. However, too high moisture can fill the small pores leading to limited oxygen transport and generation of

anaerobic conditions (Hubbe et al., 2010). Moreover, at low C/N the hydrolysis of substrates in the food waste can result in the accumulation of liquid further decreasing air-filled porosity. The lower respiration rate for the GW at any moisture content could be attributed to both limited available N and/or higher concentration of recalcitrant compounds such as lignocellulosic material in GW (Zhang et al., 2016). On the other hand, the cellulosic material from the GW acts as bulking material providing a more porous structure for gas diffusion and preventing loss of N (Bernal et al., 2009). The low levels of GW likely explain the failing of the process at the C/N ratio of 17 at any moisture content.

A response-surface analysis of the $cCER_{max}$ as a function of the moisture content (expressed as moisture content, MC, %wet basis) and the C/N ratio showed a significant relationship ($P < 0.001$ and a $R^2 = 0.95$, Eq 4).

$$cCER_{max} = -13765 + 140(C/N) + 398(MC) + 4.92(C/N)(MC) - 8.98(C/N)^2 - 4.02(MC)^2$$

Eq 4

As for the AD, $cCER$ for AED revealed significant first-order and second-order relationships with C/N ratio ($P < 0.0001$ for both), a linear relationship with moisture content ($P = 0.0004$), and an interaction between C/N and moisture content ($P = 0.005$). According to this model, optimal composting conditions would be achieved at C/N of 25.6 and a moisture content of 65% to produce a $cCER_{max}$ of 947 mg CO_2/g dry weight.

Objective 3. Measure compost quality to assess value as a soil amendment. Work performed under this objective was integrated with the research proposed by Prof. Christopher Simmons and described in detail in the project *Developing co-products from anaerobic digestion: Application of composted anaerobic digestate to soil to enhance sustainable agriculture and waste management*.

¹³C-CP-MAS Nuclear Magnetic Resonance spectroscopy analysis of the final products

¹³C-Solid State Cross Polarization Magic Angle Spinning Nuclear Magnetic Resonance (¹³C-CPMAS-NMR) was used to characterize the main functional group of the organic C in the samples. The functional group classification of the organic C was as follows: (i) aliphatic-C bonded to other aliphatic chain-Short Chain (0-27 ppm); (ii) Aliphatic-C bonded to other aliphatic chain-Long Chain (27-47 ppm); (iii) O-CH₃ or N-Alkyl O-alkyl-C di-Oalkyl-C (47-113 ppm); (iv) aromatic-C phenol or phenyl ether-C (113-160 ppm) and (v) Carboxyl-C keto-C (160-210) (Tambone et al., 2015). To estimate the stability of the substrates, the aromaticity index was calculated based on the ratio of the integrated areas of C types as in Equation (5) (Piterina et al., 2009):

$$\%Aromaticity = \frac{\%Aromatic(113-160ppm)}{(\%Aliphatic(0-47ppm) + \%Alkyl(47-113ppm) + \%Aromatic(113-160ppm))} \times 100 \quad \text{Eq 5}$$

The Aliphatic C region (0–45 ppm) is associated with proteins, lipids, and aliphatic branched and short-chain molecules (Piterina et al., 2009; Tambone et al., 2015). The spectra

show two large peaks in this region at 24 ppm (short chain branched aliphatic C) and at 30-33 ppm (the methylenic C in the long chains of aliphatic compounds, such as suberin, cutin, waxes and fatty acids) (Piterina et al., 2009). The O and N-alkyl C (45–110 ppm) region is primarily associated with O-substituted alkyl carbon in carbohydrates (Piterina et al., 2009). The predominant peak was found at 72-74 and 82 ppm, which is associated with O-alkyl-C of atoms C-2, C-3 and C-5 in polysaccharides (cellulose and hemicellulose)(Piterina et al., 2009). This peak, along with the peak at 105 ppm that represents the atoms of the anomeric carbon (C-1) of cellulose (Kogel-Knabner, 2002), indicates that hemicellulose and cellulose are the dominant material at any C/N ratio, before and after digestion. The signals observed at 56 ppm (lignin methoxyl-C) and the region between 130-153 ppm (aromatic C ring) are related to aromatic C from lignins and lignin-derived molecules (Tambone et al., 2015). These signals increased after digestion indicating an accumulation of these recalcitrant compounds. Finally, the peak at 174 ppm (160–200 ppm) includes carboxylic acids primarily associated with organic acids that are free or in esters or amides (Piterina et al., 2009).

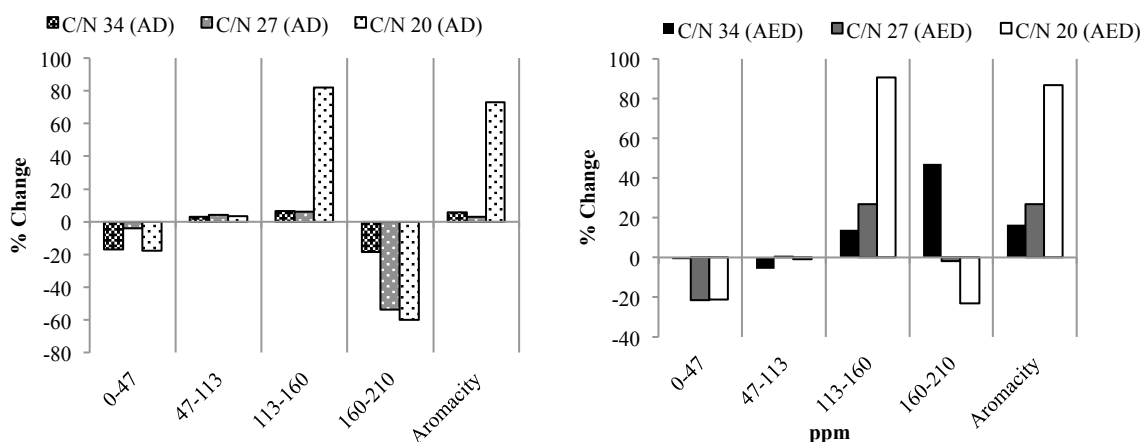


Figure 4. Percent change in total aliphatic-C (0-47 ppm), O-alkyl carbon (47–113 ppm); aromatic carbon (113–160 ppm) and carboxyl groups (160–210 ppm) for the solid anaerobic and aerobic final products compared to the initial feedstock at the C/N ratios of 34, 27 and 20.

Figure 4 presents a comparison of the percentages of the different organic C groups after anaerobic or aerobic digestion relative to the initial feedstock. Total aliphatic chains showed a general decrease after digestion. The opposite trend was observed for aromatic C which increased for all the digested samples. This increase was 90 and 81% for the aerobic and anaerobic digested samples, respectively, at C/N of 20. Moreover, for the same C/N ratio, the composted samples always showed a higher increment than those under anaerobic digestion. On the other hand, the carboxyl groups were reduced in all the samples except for the samples at C/N ratio of 34 from aerobic digestion. In this case, these groups were more reduced after anaerobic digestion compared to aerobic digestion for the same C/N ratio, being 60% lower for the C/N of 20 after anaerobic digestion. The increase in aromatic C and phenols might be related to the degradation of non-aromatic cell wall compounds, which would lead to a relative enrichment in aromatics (Tambone et al., 2013). This is due to the preference of

microorganisms to decompose easily degradable carbon compounds resulting in accumulation of recalcitrant molecules. The dominance of more recalcitrant compounds in the digested samples is confirmed by the increase of aromatic carbon at 56 ppm that corresponds to lignin methoxyl-C (Dignac et al., 2000). The aromaticity values are also known to provide an evaluation of the evolution of humification during composting (Albrecht et al., 2008). The results confirm the higher stability of the digested samples and, as expected, this stability seems to be higher for the composted samples for the same C/N ratio.

Finally, the aromaticity values are known to provide information of the humification during composting. The aromaticity increased for every digested sample. Particularly, for a specific C/N ratio, the aromaticity was higher for the composted material than for the products from AD. This, along with the UV spectra results, seem to confirm that for the same initial feedstock, aerobic digestion yielded more stable and mature products. A higher shift in aromaticity was observed at a C/N of 20. However, this does not indicate that stability is higher as the aromaticity of these samples (15.17 and 14.07 for AED and AD samples, respectively), was similar to that found at the C/N of 34 and 27.

Objective 4. Develop life-cycle assessment (LCA) model for the combined anaerobic digestion and composting conversion system.

An energy and mass balance based LCA model was developed to evaluate the impact of combining anaerobic digestion and composting to convert organic residues to soil amendment on environmental variables: water use, energy use, and greenhouse gas emissions. This was achieved by developing a new method to assess the life cycle impact of water use to aid in tracking water use and determine its impact on scarcity. Furthermore, a mathematical model was created to describe the mass and energy balance of anaerobic digestion followed by composting. Life cycle inventory data were tracked in this model, including associated greenhouse gas emissions, energy use, and water use for the combined system. The total environmental impacts were evaluated using a process-based LCA and incorporated the new water impact assessment method and mathematical model.

Outcomes:

A life cycle impact assessment method for water use is pertinent in water-limited areas and to evaluate freshwater consumption and its potential environmental impacts. Areas with high water stress are typically found in areas with high agricultural practices (Schlosser et al. 2014). The agriculture sector requires high water use for crops and livestock management compared to other sectors (FAO 2016). Additionally, high agricultural yields and practices produce large amounts of agricultural residue that require waste management strategies to dispose organic waste at that scale (Ward et al. 2008). The new mass balance based water use impact assessment (MBB) method that was developed makes it possible for analyzing water use in biological and chemical processes, such as organic waste conversion scenarios where materials have associated moisture content and the potential to generate water through chemical reactions, in both water-limited and water-rich areas. The MBB approach allows practitioners to track potential sources of water generation to serve as water sources within the

system boundary of the entire process to limit external water use, as well as match water quality needs for a system with less stressed water supplies. The method builds upon previously developed methods for inventory and impact assessment based on analysis of the current state of the art in water use methods (Pfister et al. 2009, Boulay et al. 2011a, and Boulay et al. 2011b).

The MBB method follows conservation of mass for tracking water use. Equation 6 represents the conservation of mass within the system and was used for accounting the inputs and outputs of the process.

$$m_{in} - m_{out} + m_{gen} - m_{acc} = 0 \quad (Eq.6)$$

Where m_{in} is the rate of water input into the system and includes moisture content associated with input material, irrigation, and other water inputs; m_{out} represents the rate of water output from the system and includes evapotranspiration, moisture content of materials leaving the process, and any other water outputs; m_{gen} represents the rate of water generation and captures production of water associated with chemical and physical reactions that may occur in the process; and m_{acc} represents the rate of water accumulation in the system. For some systems, m_{acc} would be zero under steady state operation. A generation term was included to allow researchers to track potential sources of water generation to serve as water sources within the system boundary of the entire process. Each water input and output was categorized based on quality and type and to account for possible water degradation (Boulay 2011a). The Boulay 2011a categorizing inventory method was incorporated in this stage to ensure that the quality and type of water generated in the subsystem matches minimum water quality requirements for other parts in the system for proper reuse.

Water stress indicator (WSI) was used as the midpoint indicator and was calculated based on the Pfister 2009 method. Water use types, shown in table 2, followed Boulay 2011a categories for water quality and are assigned quality factors based on scarcity of water for each category. Water categories range from excellent to unusable based on levels of toxicity and microbial contamination related to fecal coliforms.

Table 2: Water quality indicators derived from Boulay et al. 2011a.

Category	Name	Qualitative Description
1	Excellent	Low microbial contamination, low toxicity
2a	Good	Low microbial contamination, medium toxicity
2b	Average	Medium microbial contamination, medium toxicity
2c	Average Toxicity	Low microbial contamination, higher toxicity
2d	Average Microbial Contamination	High microbial contamination, low toxicity
3	Poor	High microbial contamination, medium toxicity
4	Very Poor	High microbial contamination, high toxicity
5	Unusable	Other unusable water

After a water mass balance was completed for the system, WSI was calculated using the Pfister 2009 method. Equation 7 shows the calculation for the midpoint characterization factor of water use:

$$CF = Q_{f,i} WSI \quad (Eq.7)$$

Where CF is the characterization factor reported in units of m^3 stressed water equivalent (sw_e) per m^3 water use; $Q_{f,i}$ is the quality factor for category type i (shown in table 2) based on Boulay 2011a supplemental information at the watershed or country level; and WSI is the water stress index calculated with Pfister 2009 methods and using current watershed/country supply and use values from U.S. Geological Services (USGS), U.S. State water board data, or AQUASTAT, FAO global water information system (FAO 2016; USGS 2016).

By creating a relationship between water use and water stress for separate water quality categories, practitioners can be informed of sources of water that may have lower water quality, but also lower water stress that are still acceptable for different processes in an analyzed system. Combining water use with WSI and a quality factor based on the type of water used gives a more defined midpoint category and information on the impact of water use for endpoint areas of protection.

Model Development:

Anaerobic digestion followed by composting as an organic waste management practice can help reduce the need for external water addition and aeration energy during the composting process because anaerobic digestion can remove the easily degradable carbohydrates present in the waste first. This is likely to lead to reduced total greenhouse gas emissions and production of high value co-products in the form of renewable natural gas from anaerobic digestion, and soil amendment from composting. However, the efficacy of this combined system depends on organic matter composition and the duration of anaerobic digestion prior to composting.

A model was developed to include those variables to determine when this system, shown in Figure 6, may be the most appropriate management practice. The base system included heated in-vessel anaerobic digestion followed by aerated static pile composting. A 500,000 gallon ($1893 m^3$) anaerobic digestion tank comprised of a mixture of food waste and green waste was maintained at mesophilic conditions ($35\text{ }^\circ\text{C}$) inside the reactor. Biogas produced during digestion was collected and purified using downstream processing to generate electricity and reuse waste heat for the digester for combined heat and power (CHP). The waste was digested under high-solids conditions, which requires the organic waste to have 60-90 % moisture content. External water was added to the organic waste to meet minimum moisture content requirements for digestion and biogas production. Composting was maintained at $>40\%$ wet basis moisture content and completed to achieve pathogen reduction and maturation based on temperature and respiration data (CA Regulations Title 14, Division 7, Chapter 3.1). Results were normalized for 1 Mg organic waste.

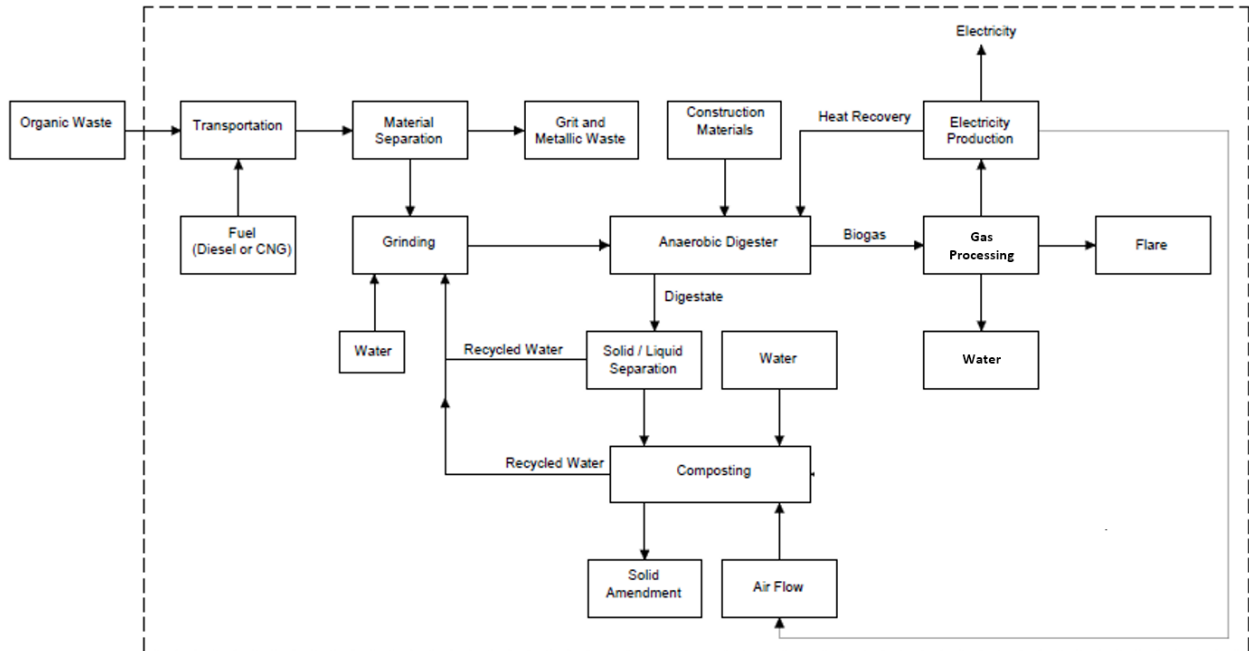


Figure 6: Flow diagram of combined anaerobic digestion and composting with system boundary for life cycle inventory.

The mass and energy flow diagram in Figure 7 shows the flow of organic waste, energy supply and generation, gas flow, and water flows and is incorporated in Figure 6 to inform the life cycle inventory.

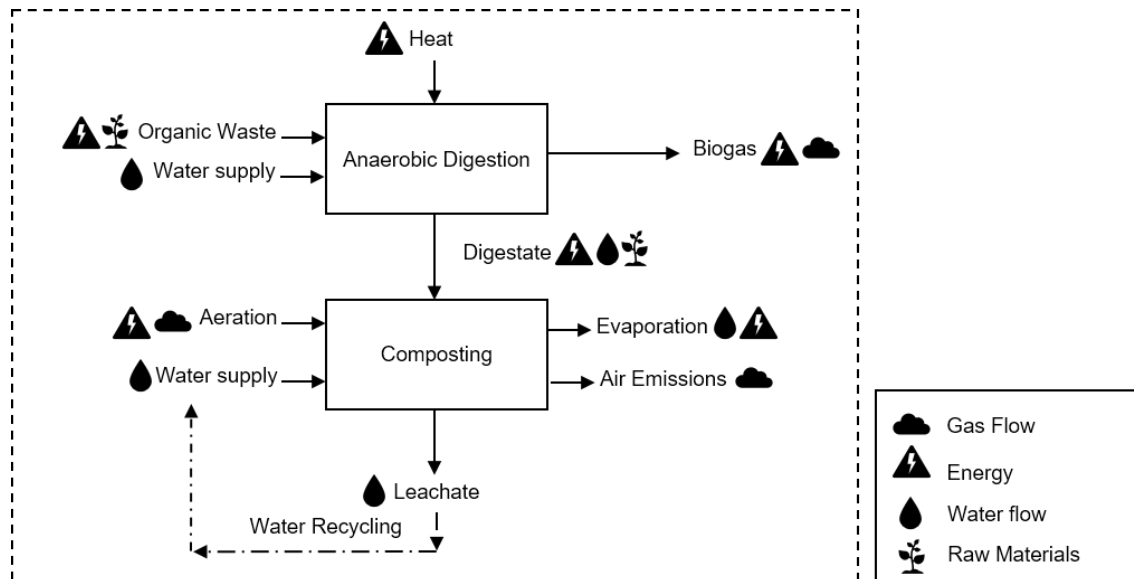


Figure 7: Mass and energy flow diagram for combined anaerobic digestion followed by composting.

The overall governing equation for the system, shown in Equation 8, is the general heat balance equation.

Energy Accumulation = Energy Generation + Energy Input – Energy Output (Eq. 8)

Energy accumulation is dependent on specific heat of the organic waste, shown in Eq. 9.

$$Q_{acc} = \rho_f C_{p,f} V_f \frac{dT_f}{dt} \quad (Eq. 9)$$

where Q_{acc} (W) is the accumulated heat, ρ_f (kg/m³) is the density of the feedstock, $C_{p,f}$ (kJ/kg-K) is the specific heat of the feedstock, and V_f (m³) is the volume of feedstock digested. Organic material specific heat was based on waste composition (fat, carbohydrate, protein, ash, and fiber content), and moisture content (Eqs. 10 and 11), where X_{ash} is the ash mass fraction (kg ash/kg dry solids), $C_{p,ash}$ is specific heat of ash (1.2 kJ/kg-K), X_{carb} is the carbohydrate mass fraction (kg carbohydrate/kg dry solids), $C_{p,carb}$ is specific heat of carbohydrate (1.6 kJ/kg-K), X_{fat} is the fat mass fraction (kg fat/kg dry solids), $C_{p,fat}$ is specific heat of fat (1.926 kJ/kg-K), X_{fiber} is the fiber mass fraction (kg fiber/kg dry solids), $C_{p,fiber}$ is specific heat of fiber (1.9 kJ/kg-K), $X_{protein}$ is the protein mass fraction (kg protein/kg dry solids), $C_{p,protein}$ is the specific heat of protein (2.1 kJ/kg-K), $X_{w,wb}$ is the wet basis moisture content (kg H₂O/kg total weight), and $C_{p,w}$ is the specific heat of water (4.184 kJ/kg-K). Specific heat of feedstock was implicitly time dependent due to potential moisture content changes during the conversion process.

$$C_{p,f} = X_{ash}C_{p,ash} + X_{carb}C_{p,carb} + X_{fat}C_{p,fat} + X_{fiber}C_{p,fiber} + X_{protein}C_{p,protein} \quad (Eq. 10)$$

$$C_{p,fw}(t) = (1 - X_{w,wb}(t))C_{p,f} + X_{w,wb}(t)C_{p,w} \quad (Eq. 11)$$

During the AD process, the total energy added to the system included initial heat to reach mesophilic temperature and conductive heat losses from the reactor walls. Given a closed, sealed system, convective and radiative heat losses were assumed negligible. Carbon mineralization models were used to determine the rate of organic matter decomposition in AD and composting, as well as CO₂ evolution. The total potential mineralizable carbon is proportional to the chemical potential energy of the organic matter. The specific potential mineralizable carbon, C_T (mg CO₂-C/g TS organic waste) contained in the untreated organic waste was assumed to be the same for anaerobic digestion and composting. (Eq. 12) where $C_R(t = 0)$ represents the initial mineralizable carbon in the organic waste at the start of anaerobic digestion (mg CO₂-C/mL organic waste).

$$C_R(t = 0) = C_T \rho_{TS} \quad (Eq. 12)$$

The rate of mineralization during anaerobic digestion was assumed to be very small compared to composting. Using C_T for the combined anaerobic digestion followed by composting system was a conservative estimate of the potential energy contained in the feedstock. The composting process was modeled to help degrade the remaining mineralizable carbon in the digestate, $C_R^*(t)$ (mg CO₂-C/mL organic waste) (Eq. 13).

$$C_R^*(t) = C_R(t = 0) - \left(m_{CH4} \frac{44.01 \text{ g}^{CO_2}}{16.04 \text{ g}^{CH_4}} + m_{CO_2} \right) V_f^{-1} \quad (Eq. 13)$$

The rate of mineralization was derived from the laboratory testing data during anaerobic digestion. Carbon mineralization was determined by the mass of carbon removed from the organic matter during the anaerobic process due to methane and carbon dioxide production (Eqs. 14 and 15)

$$m_{CH_4} = V_{CH_4} \rho_{CH_4} \quad (\text{Eq. 14})$$

$$m_{CO_2} = V_{CO_2} \rho_{CO_2} \quad (\text{Eq. 15})$$

where, m_{CH_4} (kg CH₄) is the mass of methane produced, V_{CH_4} is the volume of methane (m³), ρ_{CH_4} is the density of methane gas (kg/m³), m_{CO_2} (kg CO₂) is the mass of carbon dioxide produced, V_{CO_2} is the volume of carbon dioxide (m³), and ρ_{CO_2} is the density of carbon dioxide gas (kg/m³). Cumulative methane production during anaerobic digestion was modeled using the integral form of U.S. EPA's LandGEM model (Eq. 16),

$$V_{CH_4} = L_o m_f (1 - e^{-k_c t_{AD}}) \quad (\text{Eq. 16})$$

where L_o is the ultimate methane yield (96 m³ CH₄ /Mg) and k_c is the decay rate (0.7 yr⁻¹) for a wet bioreactor, m_f is the initial mass of organic waste (Mg), t_{AD} is the time of digestion (yr), and V_{CH_4} is the cumulative methane collected (m³ CH₄) (Alexander et al., 2005).

Following AD, heat loss was assumed due to evaporative and advective cooling during composting. Heat generation rate followed the model developed in VanderGheynst (1997), which was developed for mixed-culture and multiple substrate aerobic fermentation and assumed heat generation from microorganisms was proportional to the oxygen consumed (Eq. 16) (VanderGheynst et al., 1997). A constant heat generation factor, Q (1.4 x 10⁴ kJ/kg-O₂), was based on the oxygen uptake.

$$Q_{gen} = Q(OUR) \quad (\text{Eq. 17})$$

A carbon mineralization model (Eq. 18) was used to determine the rate of organic matter decomposition and CO₂ evolution, where $\frac{dC_R^*}{dt}$ is the rate of carbon mineralization (mg CO₂-C/mL compost-day) (Aslam et al., 2008).

$$CER = \frac{dC_R^*}{dt} V_{comp} \quad (\text{Eq. 18})$$

In the first order kinetic model, the rate of carbon mineralization is proportional to the amount of potential mineralizable carbon (Eq. 19), C_R^* . The kinetic rate constant, k_i (day⁻¹), is assumed to be dependent on temperature and oxygen concentration (Eq. 20). It followed an Arrhenius relationship with incubation temperature, where T is the temperature of the compost pile (K) and R is the universal gas constant (8.314 J/mol-K). Activation energy, E_a (4.54 x 10⁴ J/mol), and Arrhenius constant, A_i (5.39 x 10⁶ day⁻¹), values were estimated from a previous study (Aslam et al., 2008). The rate constant included a correction factor for oxygen concentration, f_{O_2} (dimensionless), to ensure that mineralization rate decreases as oxygen concentration decreases. For suboptimal conditions, the correction factor for oxygen concentration incorporates the minimum oxygen concentration for oxygen consumption, $K = 2\% \text{ O}_2$ (Eq. 21) (Richard et al., 2006).

$$\frac{dC_R^*}{dt} = -k_i C_R^* \quad (\text{Eq. 19})$$

$$k_i = f_{O_2} A_i e^{(-E_a/RT)} \quad (\text{Eq. 20})$$

$$f_{O_2} = \frac{X_{O_2,out}}{K + X_{O_2,out}} \quad (\text{Eq. 21})$$

Following compilation of the full mathematical model, time for anaerobic digestion of organic matter of varying food waste and green waste mixtures was evaluated from zero to 90 days to determine impacts on composting time, overall energy use, and water use. Varying food and green waste mixtures were represented by specific potential mineralized carbon values, C_T . Composting time was determined by completing composting to maturity and pathogen reduction based on temperature and respiration data (CA Regulations Title 14, Division 7, Chapter 3.1). Methane production was modeled using U.S. EPA LandGEM, shown in Figure 8.

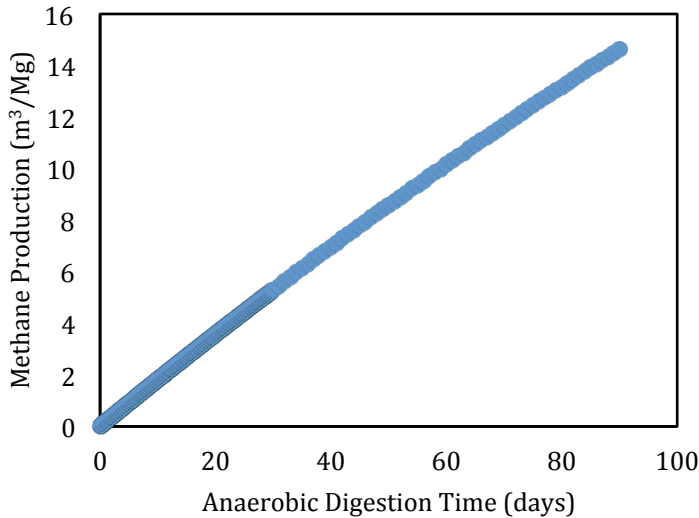


Figure 8: Methane production over time. Modeled using LandGEM

Outcomes for composting time to reach maturation showed composting time was inversely proportional to AD time, as shown in Figure 9, over a C_T range from 200-800 mg CO_2 -C/g TS. Composting time increased with increasing C_T values. However, the feedstock mixture with $C_T = 200$ mg CO_2 -C/g TS did not have enough remaining energy to self-heat and ensure pathogen reduction during composting after 30 days of AD. Aeration energy required for composting and total water use followed a similar trend as composting time. Aeration energy and water use (Figure 9) decreased as the organic waste had longer retention time in the anaerobic digester.

Net energy production was calculated from the energy produced due to methane generation, minus the energy required to run the composting and AD systems. Net energy production exhibited a positive relationship between energy production and AD time; as AD time

increased, so did net energy produced (Figure 9). However, there may be diminishing returns on energy production.

Overall results showed that increased time during anaerobic digestion reduced energy requirements for aeration, overall water use, and required composting time. They also showed increased AD time also increased net energy production from the combined system. However, there is a limit to how long the organic waste should be treated with AD. If left for too long, it can remove too much chemical potential energy from the material prior to composting, so once it undergoes aerobic treatment, the microorganisms do not have enough fuel to generate sufficient energy to self-heat to achieve minimum pathogen reduction standards.

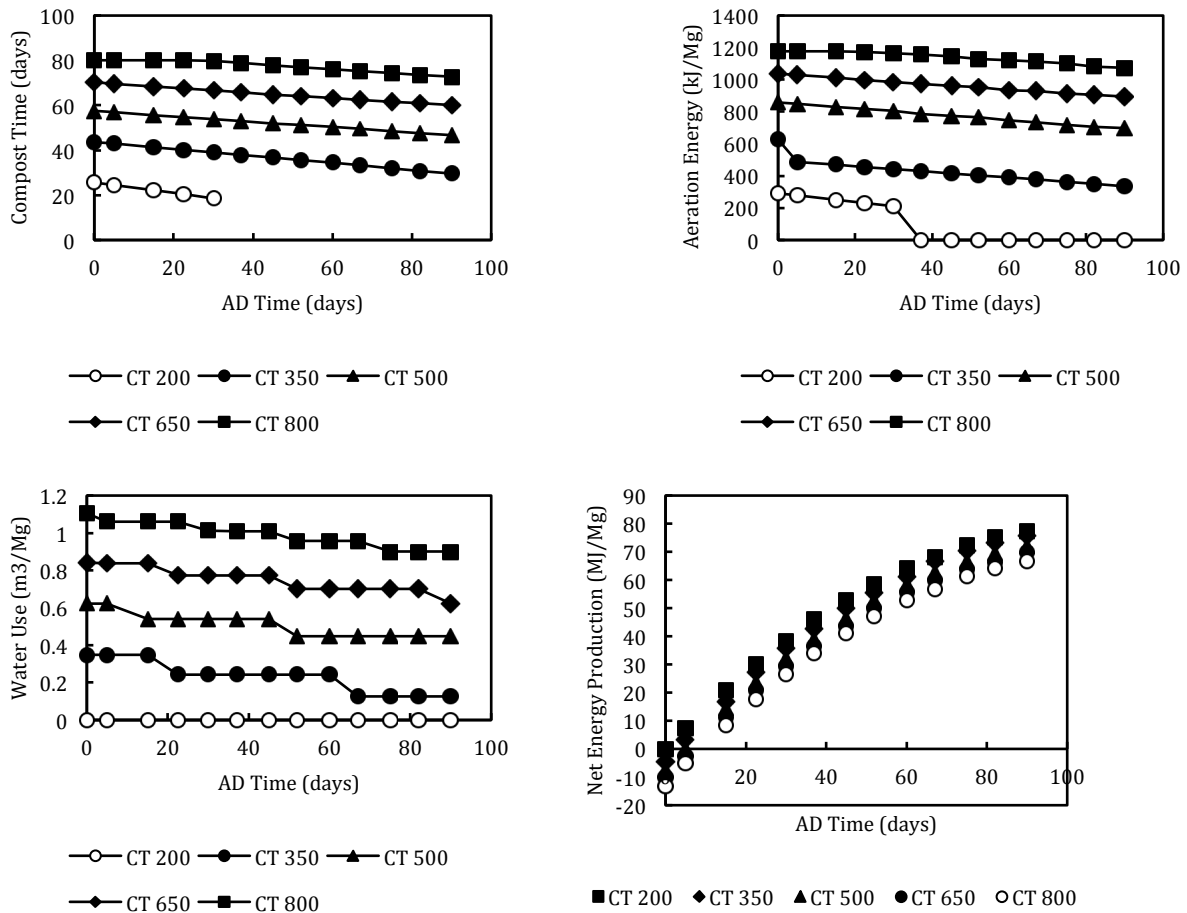


Figure 9: (1) Time for compost to reach maturation (pathogen reduction and respiration), (2) aeration energy (3) water use, and (4) net energy production based on AD time for different C_T values. After AD treatment for 30 days, $C_T = 200$ mg $\text{CO}_2\text{-C/g}$ TS did not have enough energy remaining to self-heat to ensure pathogen reduction.

These outcomes from the mathematical model were incorporated into the LCA model, which also included transportation of organic residues to the waste management facility, operation and maintenance, and equipment inventory, including materials used for the facility and equipment, based on primary data related to the Tadweer facility in Dubai and Yolo County

Central Landfill materials and practices used. The environmental impact categories considered in the analysis included global warming potential due to greenhouse gas emissions, fossil and renewable energy use, and equivalent scarce water use. Using $C_T = 500 \text{ mg CO}_2\text{-C/g TS}$, and AD time equal to 45 days, the total energy use for the system was 431 MJ/Mg organic waste (fossil-based) and 217.3 MJ/Mg organic waste (renewable-based). The high fossil-based energy use was attributed transportation energy requirements for hauling the waste to the waste treatment facility. The total global warming potential for the combined system was 293 kg CO_{2e} /Mg organic waste. This does not include biogenic CO_2 emissions since they are considered carbon neutral and thus do not contribute to global warming potential. The total GWP is also attributed to emissions from refuse vehicles to haul to the waste. Overall equivalent scarce water use for the system was 0.36 m^3 scarce water/Mg organic waste.

Objective 5. Create educational materials for waste management specialists, policy makers, and the academic community engaged in sustainable management of organic wastes for integration into the living laboratory of The Sustainable City in Dubai.

Outcomes:

Post-doctoral scholar Jesus Fernandez-Bayo and PhD student Sara Pace engaged in training activities to improve their skills in Science, Technology, Engineering and Mathematics (STEM) communication and development of educational materials in renewable energy and sustainability. These skills were applied toward creating educational materials for waste management specialists, policy makers, and the academic community engaged in sustainable management of organic wastes.

Two lessons and corresponding hands-on activities related to solarization and life cycle assessment were developed for publication on Teachengineering.org. The LCA lesson focused on decision making related to sustainability practices in everyday life, whereas the solarization lesson focused on engaging elementary school students and teachers how solarization can be applied and understood. The lessons, activities and supporting educational materials are available free of charge at the following links:

https://www.teachengineering.org/lessons/view/ucd_soil_solarization_lesson01

https://www.teachengineering.org/activities/view/ucd_soil_solarization_lesson01_activity1.

The LCA lesson will be published in April 2017.

The educational materials were developed at a sixth-grade level, but they can be adjusted for multiple age ranges and are still relevant for creating base understanding of the material. The lessons and activities have been taught in four different classrooms in California in underserved communities, and at a teacher workshop for sustainability education. The lessons were designed to engage young learners and families and included defined learning outcomes and assessment measures to determine how much students learned about the topic upon completing the lesson and activity. Furthermore, the LCA model was presented at the BioCycle REFOR16 conference to a group of solid waste management specialists and other parties related to treating organic waste and producing renewable forms of fuel. The outcomes

from objectives 1-4 have also been summarized in manuscripts in review or in preparation. Support from the SRTP program has been acknowledged in all presentations and manuscripts.

References

- Albrecht, R., Ziarelli, F., Alarcon-Gutierrez, E., Le Petit, J., Terrom, G., Perissol, C. 2008. (13)C solid-state NMR assessment of decomposition pattern during co-composting of sewage sludge and green wastes. *European Journal of Soil Science*, **59**(3), 445-452.
- Bayart, J.-B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., Koehler, A. 2010. A framework for assessing off-stream freshwater use in LCA. *The International Journal of Life Cycle Assessment*, **15**(5), 439-453.
- Bernal, M.P., Albuquerque, J.A., Moral, R. 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. *Bioresource Technology*, **100**(22), 5444-5453.
- Boulay, A.-M., Bulle, C., Bayart, J.-B., Deschênes, L., Margni, M. 2011. Regional characterization of freshwater use in LCA: modeling direct impacts on human health. *Environmental science & technology*, **45**(20), 8948-8957.
- Dignac, M.F., Derenne, S., Ginestet, P., Bruchet, A., Knicker, H., Largeau, C. 2000. Determination of structure and origin of refractory organic matter in bio-epurated wastewater via spectroscopic methods. Comparison of conventional and ozonation treatments. *Environmental Science & Technology*, **34**(16), 3389-3394.
- Hanafiah, M.M., Xenopoulos, M.A., Pfister, S., Leuven, R.S., Huijbregts, M.A. 2011. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. *Environmental science & technology*, **45**(12), 5272-5278.
- Hubbe, M.A., Nazhad, M., Sanchez, C. 2010. COMPOSTING AS A WAY TO CONVERT CELLULOSIC BIOMASS AND ORGANIC WASTE INTO HIGH-VALUE SOIL AMENDMENTS: A REVIEW. *Bioresources*, **5**(4), 2808-2854.
- i Canals, L.M., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., Cliff, R. 2009. Assessing freshwater use impacts in LCA: Part I—inventory modelling and characterisation factors for the main impact pathways. *The International Journal of Life Cycle Assessment*, **14**(1), 28-42.
- Katase, M., Kubo, C., Ushio, S., Ootsuka, E., Takeuchi, T., Mizukubo, T. 2009. Nematicidal activity of volatile fatty acids generated from wheat bran in reductive soil disinfection. *Japanese Journal of Nematology*, **39**(2), 53-62.
- Kogel-Knabner, I. 2002. The macromolecular organic composition of plant and microbial residues as inputs to soil organic matter. *Soil Biology & Biochemistry*, **34**(2), 139-162.
- Kounina, A., Margni, M., Bayart, J.-B., Boulay, A.-M., Berger, M., Bulle, C., Frischknecht, R., Koehler, A., Milà i Canals, L., Motoshita, M., Núñez, M., Peters, G., Pfister, S., Ridoutt, B., van Zelm, R., Verones, F., Humbert, S. 2013. Review of methods addressing freshwater use in life cycle inventory and impact assessment. *The International Journal of Life Cycle Assessment*, **18**(3), 707-721.
- Motoshita, M., Itsubo, N., Inaba, A. 2011. Development of impact factors on damage to health by infectious diseases caused by domestic water scarcity. *The International Journal of Life Cycle Assessment*, **16**(1), 65-73.
- Nielfa, A., Cano, R., Fdz-Polanco, M. 2015. Theoretical methane production generated by the co-digestion of organic fraction municipal solid waste and biological sludge. *Biotechnology Reports*, **5**(0), 14-21.

- Pfister, S., Koehler, A., Hellweg, S. 2009. Assessing the environmental impacts of freshwater consumption in LCA. *Environmental science & technology*, **43**(11), 4098-4104.
- Piterina, A.V., Barlett, J., Pembroke, J.T. 2009. (13)C-NMR Assessment of the Pattern of Organic Matter Transformation during Domestic Wastewater Treatment by Autothermal Aerobic Digestion (ATAD). *International Journal of Environmental Research and Public Health*, **6**(8), 2288-2306.
- Reddy, A.P., Simmons, C.W., D'Haeseleer, P., Khudyakov, J., Burd, H., Hadi, M., Simmons, B.A., Singer, S.W., Thelen, M.P., VanderGheynst, J.S. 2013. Discovery of Microorganisms and Enzymes Involved in High-Solids Decomposition of Rice Straw Using Metagenomic Analyses. *Plos One*, **8**(10), e77985.
- Tambone, F., Adani, F., Gigliotti, G., Volpe, D., Fabbri, C., Provenzano, M.R. 2013. Organic matter characterization during the anaerobic digestion of different biomasses by means of CPMAS C-13 NMR spectroscopy. *Biomass & Bioenergy*, **48**, 111-120.
- Tambone, F., Terruzzi, L., Scaglia, B., Adani, F. 2015. Composting of the solid fraction of digestate derived from pig slurry: Biological processes and compost properties. *Waste Management*, **35**, 55-61.
- Yan, Z., Song, Z., Li, D., Yuan, Y., Liu, X., Zheng, T. 2015. The effects of initial substrate concentration, C/N ratio, and temperature on solid-state anaerobic digestion from composting rice straw. *Bioresource Technology*, **177**, 266-273.
- Zhang, L.L., Jia, Y.Y., Zhang, X.M., Feng, X.H., Wu, J.J., Wang, L.S., Chen, G.J. 2016. Wheat straw: An inefficient substrate for rapid natural lignocellulosic composting. *Bioresource Technology*, **209**, 402-406.