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Implementation of Large-Scale Anaerobic Digestion of Food Waste at the University of South Florida

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Implementation of Large-Scale Anaerobic Digestion of Food Waste at The University of South Florida

by

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Dedication

To all those who have supported my journey up until this point, I dedicate this work to all of you. Thank you to my family- my grandmother Kanwaljit Kaur, my mother Charan Panesar, my father Harry Panesar and my brother Steven Panesar for being the reason I have become the person I am today and being a strong pillar to help support all of my dreams. To my cousins, for all of the encouragement, support, check-ins and love that I experience from each of you on a daily basis is humbling and inspiring. To my friends- for the understanding, the late nights supporting and the belief you have in everything I try to do inspires me to push harder and further for everything in my life. To all my fellow classmates throughout this journey- thank you for the solidarity, the wisdom and guidance throughout this process. And lastly to my fellow SGEF project team- thank you for the wisdom, the eye-opening experiences of being able to look at problems through different lenses and thoughts, and for the strong dedication and will to truly wish better for this planet and all those in it.

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Table of Contents

Table of Tables

Table of Figures

Abstract

Food waste represents a major sustainability issue in the United States. Food waste represents 18% of landfill space and is the largest contributor of landfill methane emissions. US EPA's Food Recovery Hierarchy recommends alternatives to landfilling food waste, including source reduction, food donation and anaerobic digestion.

The Student Green Energy Fund (SGEF) Food Recovery Project aims to follow suggestions outlined by the US EPA Food Recovery Hierarchy to encourage the University of South Florida (USF) to become a zero-waste campus. A multi-disciplinary approach aims to source reduce wasted food, look for opportunities for source reduction and anaerobically digest any remaining food waste from there. Currently, pilot-scale digesters are being operated to help demonstrate the feasibility of large-scale anaerobic digestion.

This thesis explores different approaches for the anaerobic digestion of food waste to determine alternatives to the pilot-scale approach to help implement large-scale technology. A community partnership with a tea wholesaler was identified, providing opportunities for food waste co-digestion with both tea leaves and compostable sugarcane-based polylactic acid (PLA) plates. The objectives of this research were to: 1. Analyze the co-digestion of food waste with compostable plates and tea leaves, 2. Conduct an initial lifecycle assessment that compares incineration of food waste and anaerobic digestion of food waste for the entirety of USF's food waste, as well as a larger-scale digester that can process USF and surrounding hospital waste.

The results of the first phase of co-digestion showed that food waste digestion on its own will result in souring of the reactor and inhibition of methanogenesis. When food waste is co-digested with either tea leaves or compostable plates, the reactor remains healthy and produces methane, however both digesters saw lag periods of 21 and 30 days with tea leaves and with plates, respectively. The methane yield for tea leaves was 372 ml CH_4 / g VS and for compostable plates was 445 ml CH_4 / g VS. The digestion period was 92 days, at which point the tea leaves methane production stabilized to inoculum control levels, however the compostable plates were still producing more methane and did not reach their full methane potential.

The second phase of tests investigated whether the methane yield of food waste on its own could be improved. This was done through the introduction of an alkalinity source using a combination of sodium bicarbonate and oyster shells, or through a separate digester with an F:M ratio of 0.5, as opposed to an F:M ratio of 1 applied in all other digestion sets. The third phase of tests investigated whether the lag period observed in Phase 1 for tea leaves and compostable plates could be reduced by introducing acclimated inoculum from Phase 1 into the digestion set. Phase 3 also mixed all three substrates together to see the effects of co-digestion of all three substrates. At the time of submission, Phases 2 and 3 were ongoing.

An initial lifecycle assessment was conducted using the US EPA Waste Reduction Model (WARM). This model compares alternative disposal methods for solid waste disposal. The study compared USF’s current process of sending food waste to incineration at McKay Bay Waste to Energy Incineration facility to sending food waste to an on-campus anaerobic digester. Sending food to an anaerobic digester decreased GHG emissions by 8 mton of CO_2 equivalents each year, which is negligible and opposing other literature. Therefore, it is recommended that a more comprehensive lifecycle assessment be carried out.

Chapter 1. Introduction

The large volume of food waste produced in North America represents a major sustainability issue. In 2016, Americans wasted 63 million tons of food (US EPA 2016). A majority of food ends up in landfills, where it produces methane emissions and landfill leachates that contaminate groundwater. Approximately 18% of landfill waste is food waste, and 16% of methane gas emissions are a direct result of landfills (Buzby et al. 2014). Food waste is the main source of methane emissions from municipal solid waste (Trabold and Nair 2018) Food waste also strains other resources. In the US, wasted food accounts for 21% of fresh water use, 19% of fertilizer use, and 18% of crop land use (Trabold and Nair 2018).

Food waste is generated from a variety of sources throughout the food system, ranging from food production to distribution and consumption ("Food loss and food waste" n.d.)The majority of North American food waste is generated at the consumer level, either through businesses such as restaurants, or household waste; however, the underlying issues remain in policy and practice (Evans 2011; Neff et al. 2015). Food waste can be separated into three main categories: 1) avoidable 2) possibly avoidable or 3) unavoidable food waste (Quested et al. 2011). In affluent countries such as the U.S., most food waste falls under the avoidable or possibly avoidable categories. Major contributors to food waste include consumerism in developed countries, coupled with decreased food costs and increased incomes (Trabold and Nair 2018).

Food waste can be reduced and possibly prevented. The US EPA has generated a Food Recovery Hierarchy to help divert food from being in landfills, which has been adopted by The

University of South Florida (USF). (Figure 1.1). This hierarchy lists the most preferred method to the least preferred method from top to bottom; source reduction and feeding hungry people are the most desirable ways to prevent food waste and recover food. Another desirable alternative to landfilling is industrial uses, such as anaerobic digestion (AD) of the food to produce bioenergy and recover nutrients for agricultural applications. AD is a well-established technology for organic waste recovery throughout Europe and has recently seen a surge for treating organic waste in regions of the United States (Baere and Mattheeuws 2015). It is often seen as an economically viable option for food resource recovery due to the fact that the two main biproducts of AD, biogas and nutrient-rich inoculum, can both be sold for energy use and fertilization respectfully.

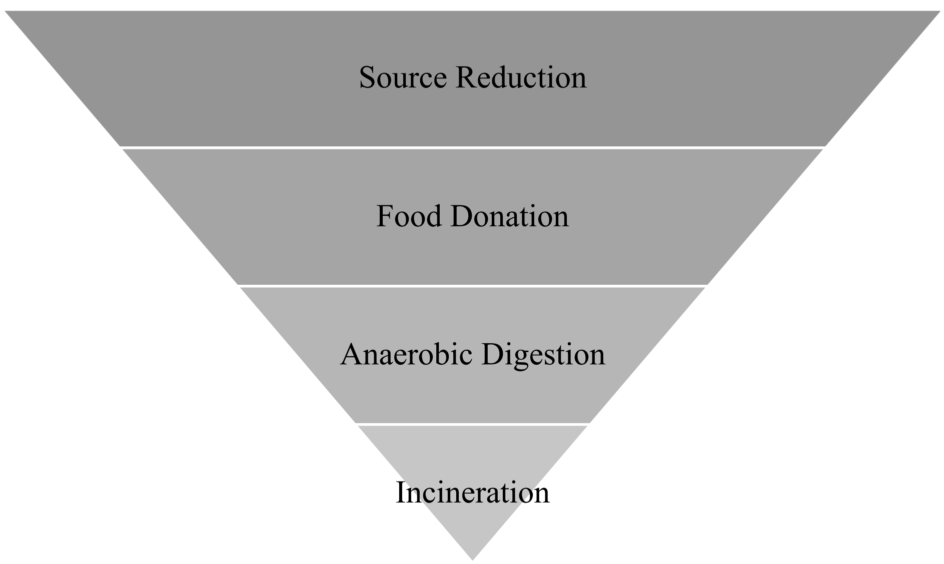

Figure 1.1 Food Recovery Hierarchy adapted for USF (US EPA 2016)

As communities move toward becoming more sustainable, college and university campuses have also increased efforts to be more environmentally friendly, specifically in the food recovery area. Several universities in the United States have already implemented AD to treat their on-campus food waste. A summary of different universities and their lessons learned is shown in Table 1.1.

Table 1.1 Summary of Successful Full-Scale University AD Projects in the US

Site	**Lessons Learned**	**References**
UC Davis Renewable Energy Anaerobic Digester	• 8 000 tons organic waste/year producing roughly 1.4 million kWh/yr • When operating at average capacity, net positive energy generation • Feedstock processing and biogas purification are highest costs • Capital costs roughly 1/3rd of typical AD cost	(Zhang et al. 2017)
University of Wisconsin-Osh Kosh high solids anaerobic digestion	• 6000 tons/yr producing 3 million kWh/yr • Dry anaerobic digestion process • Liquid "percolate" seeps out and is sprayed back onto substrate • Largely food and yard waste • Cannot handle manure • Requires less input energy than wet biodigesters	("Biogas Systems" 2016)
Michigan State University	• 20 000 tons/yr food waste producing 2.8 million kWh • Subject to substrate change which prevents production of useable fertilizer	(Stuever 2013)

USF is also looking into implementation of full-scale AD to handle the entirety of their food waste through the Campus Food Recovery Project. This project uses a multi-disciplinary approach to combat the overall issue of food waste, including preventing food waste through changed behaviors amongst students, donating edible food to food-insecure students, and lastly through anaerobically digesting all remaining food waste. Currently, this project is operating 6 pilot- scale digesters at USF. These digesters, designed by Solar Cities (solarcities.eu) are semi-batch reactors, hold 1 m^3 each, operating at ambient temperatures and currently only load food waste. The main goal during the operation of these digesters is to figure out logistical issues including how long it takes to pick up food waste, how long it takes to mechanically pre-treat the food waste, how often food can be picked up, and where to store food waste before pickup. Many of the lessons learned throughout working with this project helped to motivate the research conducted in this thesis and while it is outside of the scope of this thesis, details on the work are provided in Appendix A.

In the University applications, such as those seen in Table 1.1, waste was taken not just from university dining halls, but from surrounding industries as well. This allowed for larger scale digesters, as well as for the opportunity for co-digestion (Zhang et al. 2014). Co-digestion allows for a more optimal carbon/nitrogen ratio and often results in greater methane production (Zhang et al. 2014). Therefore, it is important to identify other substrates that can potentially be co-digested with food waste to stabilize the system and generate tipping fees. Two co-substrates that may potentially help stabilize food waste AD are spent tea leaves or compostable plates. Tea is the second most common beverage drank after water, and spent tea leaves are a common waste product sent to landfill (Goel et al. 2001). The Tampa area is also home to a fresh tea wholesaler, TBD Café, who is enthusiastic about disposal of their products in a sustainable fashion. One of the main contaminants in food waste are plastic waste from disposable one-use plastics, therefore if there is an opportunity for a source switch that may turn a contaminant into a biodegradable substrate, it can help eliminate labor hours required to separate food waste (Labatut and Pronto 2018).

After determining the best logistical approach for the anaerobic digestion of food waste, it is then important to determine whether full scale AD is the more sustainable choice using a lifecycle assessment. There are many research papers currently comparing anaerobic digestion of food waste versus landfilling at various scales (Edwards et al, 2017, Righi et al, 2013, Lundi et al, 2005), however none focus on the university scale, or compares anaerobic digestion of food waste to incineration, a common practice in Tampa, Florida. Focusing on the university scale is unique in that universities are one of few areas where people live, eat and waste all in the same dense area. This may heavily influence transportation GHG emissions as well as provide energy that can be sent back to the university. As well, this study is not aware of any university-scale LCAs that have occurred for Florida, which has its own unique energy compositions. The lifecycle assessments will help ensure that all choices made for constructing a full-scale digester system are environmentally conscious.

The objectives of this research are:

1. Analyze the co-digestion of food waste with spent tea leaves and compostable plates
2. Conduct a lifecycle assessment of full-scale anaerobic digestion implementation for USF

Chapter 2. Literature Review

2.1 Food Waste and Food Recovery

2.1.1 Food Waste Generation

2.1.1.1 Types of Food Waste

There are many different ways of categorizing and sorting food waste, depending on specific applications. One of the main categories involves whether or not the food was suitable for consumption. This category divides food waste into avoidable, possibly avoidable or unavoidable food waste (Quested et al. 2011). Avoidable food waste is food that could have been eaten but went bad, or excess food that was prepared but wasted. Possibly avoidable food waste is waste that could have not been waste if it was prepared differently, such as using potato peels in cooking. Unavoidable food waste is any inedible food, such as bones or certain fruit skins. In North America, most food waste is either avoidable or possibly avoidable (Trabold et al. 2018).

Another classification for food is in regard to when the food is wasted. Food can be wasted either at the pre-consumer stage or post-consumer stage (Trabold and Nair 2018). Pre-consumer food waste can be produced during different parts of the food chain- including waste generated during farming, distribution, retail stores or restaurants. At farms, food can be wasted due to variations in food demand, which helps a farmer determine whether or not it is economically feasible to harvest their crop. Farmers may overproduce food to account for the potential for bad weather affecting crops (Trabold et al. 2018). In distribution, food is wasted due to issues such as storage, demand needs or food quality. At grocery stores, less aesthetically pleasing food, or food nearing its sell-by date, is often wasted. Issues regarding food storage and lack of proper food storage infrastructure also leads to waste. In restaurants, demand leads to food waste. Over 40% of edible food grown is wasted before it reaches consumers (US EPA

2016). 15-30% of purchased food is wasted at the post-consumer stage, often due to over-buying. (Trabold and Nair 2018).

2.1.1.2 University Scale Food Waste

Universities produce food waste from restaurants and dining halls that are prevalent on campuses. Universities also host events for students, staff and the surrounding community. USF has three different dining halls, one large concert venue, and close to thirty different restaurants. There are also many different catered events, which occur regularly on campus. Universities also represent an interesting demographic due to the fact that there is a mix of on-resident students, off-campus housing, professors and other staff, all of which have different food consumption patterns.

Universities are a unique location for food waste monitoring due to the differences in people who dine at universities. Students living in dormitories may have a majority of their meals at the university, whereas off campus students may still enjoy university dining services, but to a smaller extent. RecyclingWorks Massachusetts estimates that on campus students waste 141.75 lbs./student/year, whereas off-campus students waste 37.8 lbs./student/year of food from on-campus dining ("Food Waste Estimation Guide" 2018).

University dining halls have been the location for different food waste audits. One food waste audit occurred at USF in 2011 at Juniper-Poplar hall, and found that 187 students wasted a total of 56 lbs (Saleh 2011). This results in roughly 109 lbs./student/year, which appears to be within the RecycleWorks Massachusetts range, assuming both on and off-campus students, as well as visitors, dine at Juniper-Polar, with the majority being on-campus students. This audit only performed over one day, during a breakfast and lunch service, and therefore may not provide a representative estimate for food waste. Another food waste audit occurred in

November 2019, also at Juniper Poplar, to monitor food waste as well as determine student behavioral causes for wasting food. The three day audit showed that food waste volumes vary considerably over each day, depending on events in the University.

Other universities have attempted to characterize their waste. The University of Missouri-Kansas City carried out a full characterization of waste and recycling at their residence halls over a three day sampling period to determine what type of waste is disposed. They found that 24.9% of waste was food waste (Johnston 2003).

Another study conducted in Adana, Turkey investigated food waste generated from three on-campus dining halls, and split food waste generated into food waste generated by students, academic staff and administrative staff. Students wasted the most food, at 0.2 pounds of food per day on average. Another interesting note is that amongst all groups, there was a large standard deviation, with student standard deviation being at 0.11 pounds (Ozcicek-Dolekoglu and Var 2019). This indicates that food is not wasted homogenously amongst students, and there are a wide range of values for food waste.

Lastly, The University of British Columbia conducted a full solid waste audit of its campus based on different buildings, including academic buildings, bookstores, food service (divided into meal plans, restaurants and coffee shops) and residences (Felder et al. 2001). The coffee shop, meal plan and restaurant produced 22.5, 16.2 and 24 tons of food waste/year, respectively. These values also represented a large standard deviation (5.5, 3.9 and 28 tons/year respectively), which again can be attributed to large differences per plate and per meal. Compostable food also represented the largest portion of solid waste on campus, at 34% of the total waste. This is slightly higher than what was reported in Kansas City.

2.1.2 Food Waste Management

2.1.2.1 Source Reduction

Source reduction involves not creating food waste in the first place (US EPA 2016). In developed countries, up to 40% of food that is grown is disposed of before reaching a table (Trabold and Nair 2018). In 2016, the US wasted 63 million tons of food (US EPA 2016). Wasted food accounts for 21% of fresh water use, 19% of fertilizer use and 18% of crop land use (Trabold and Nair 2018). The EPA also has many examples of different entities focusing on source reduction to decrease food waste. Quicken Loans Arena reduced food waste at the source by tracking daily food waste in their kitchens and adjusting accordingly to decrease food composted from 3.5 tons to 1.5 tons a month. Hannaford supermarkets, a chain supermarket operating out of North-Eastern US, changed their delivery schedule and infrastructure so that food deliveries occurred daily to prevent food spoiling, rather than guessing what food would be needed over a longer period of time (US EPA 2019).

The University of Texas-Austin also conducted a food waste audit to determine how to decrease post-consumer food waste in the Spring of 2008. They measured the post-consumer food waste from one dining hall during lunch and dinner over 5 days and subtracted inedible food waste from edible food waste. They determined 112 tons of food is wasted per academic year, corresponding to 0.44 pounds per person per plate. This results in $588,659.33 lost in food per year, or $618,609.88 lost in overall resources per year. These results encouraged a social marketing campaign to help reduce food waste, with another food waste audit conducted in Fall 2008. After the campaign, food waste decreased to 81 tons per academic year, or a 32% reduction overall in food waste (US EPA 2019).

2.1.2.2 Food Donation

13.2% of Americans are considered to have income below the poverty line, and 15% of Americans are considered food insecure (US EPA 2016). As mentioned earlier, over 40% of edible food is wasted before it reaches a table. This combination of factors represents a large equity issue surrounding how food is divided in the country. One attempt at reaching a more equitable food system is through food donation. Food donation is funded through a combination of government funding, corporations, private donations as well as food distribution networks (Trabold and Nair 2018). Food donation also represents equity issues, as women are more likely than men to be food insecure (FRAC 2015). People of color also have more difficulty accessing government benefits Supplemental Nutritional Assistance Program (SNAPs). This is due to the lack of grocery stores that accept SNAPs in primarily racial communities. In Leon County, Florida, primarily black communities had zero grocery stores that would accept SNAPs (Rigby et al. 2012).

Food donation can occur in one of four ways (Trabold and Nair 2018). The first method is food gleaning, in which food that famers have harvested but are not going to sell are captured and donated. The second method is perishable food rescue, where perishable food is collected from wholesalers and retail sources. The third method is food rescue, where perishable food is collected from the service industry. The last issue is non-perishable food collection, the method most commonly seen by the public. This is through public outreach such as food drives.

2.1.2.3 Landfill

Landfilling is the least desirable option for food waste management, as per the US EPA Food Recovery Hierarchy(“US EPA Food Recovery Hierarchy” 2016). However, 97% of US food waste ends up in landfills (Trabold and Nair 2018). Landfilling results in more greenhouse gas (GHG) emissions than any other disposal option, both due to the low efficiency of GHG recovery from landfills, as well as the long distance that waste must be transported to reach the landfill. Landfills also produce leachate, which can enter water supplies during heavy rainfall periods.

2.1.2.4 Incineration

Incineration (or thermal waste to energy [WtE]) is another option that municipalities may choose to decrease the volume of municipal solid waste that needs to be landfilled. Incineration involves the combustion of waste materials into heat or energy. Incineration commonly powers either steam turbines or heat exchangers (Pham et al. 2015). 1 kg of solid waste can be converted to 0.51 kg of CO_2 equivalents when incinerated (Trabold and Nair 2018). Incineration is seen as a preferred solution to landfilling because it can decrease waste volumes by 80-85% (Pham et al. 2015). However, energy recovery decreases when incinerators are not designed to the correct MSW conditions (Trabold and Nair 2018). Food waste is often considered a poor incineration feedstock due to its low solids content (Trabold and Nair 2018). Kim et al. (2013) evaluated the incineration of food waste in Korea after drying as a pre-treatment. This incineration process resulted in a global warming potential (GWP) of -315 kg of CO_2 equivalents, resulting in a carbon negative process. Yang et al. (2012) also compared incineration of municipal solid waste and saw a positive global warming potential between 25-207 kg of CO_2 equivalents. Tampa's MSW is treated at the McKay Bay Refuse-To-Energy Facility, which can handle over 360,000 tons per year, and provide enough electricity to power 15,000 homes ("McKay Bay Refuse to Energy Facility" n.d.)

2.1.2.5 Anaerobic Digestion

Anaerobic digestion (AD) is a well-established technology commonly used in Europe and Asia to treat organic waste but is gaining popularity in America. Section 2.2 provides a detailed overview on the microbiology and operational conditions of AD, as well as a more detailed look into the chemistry behind the AD of food waste.

AD of food waste has already been implemented in several universities throughout America. One of the most successful examples is at the Davis campus of the University of California (UC Davis). Ruihong Zhang et al conducted many different experiments to help classify and optimize the AD of Davis' food waste (Zhang et al. 2007; Zhang and Zhang 1999; Zhu et al. 2010). They built a large scale semi-continuous AD designed to take 100% of food waste from the campus, as well as taking food waste from surrounding industries. Biogas produced is then used for heating or cooling, or converted into electricity and sent back into the grid. (Zhang et al, 2017).

Another example of food waste AD on campuses is of high-solids AD at The University of Washington Osh Kosh campus. Osh Kosh utilized high solids AD, commonly used in Europe,, to allow for a less diluted digestate and larger biogas volumes. Osh Kosh works as a semi-continuous reactor with recirculation by collecting liquid effluent and spraying it back to the top of the digester ("Biogas Systems" 2016).

Another example of university-scale AD is at Michigan State University (Stuever 2013) Their system utilizes 20% of the biogas for system heating. The rest of the biogas currently heats part of their campus. Their system mixes cow manure, dining hall food waste and fats, oils and grease (FOGs) from surrounding restaurants (Stuever 2013). A detailed overview of these three universities, as well as some lessons learned is shown in Table 1.1.

2.1.3 Life Cycle Assessments

Lifecycle assessments (LCA) are a powerful tool used to assess the long-term environmental impact of a product or solution over its full life, from materials processing, to manufacturing, to use, and final disposal at the end of its life. Process-based lifecycle assessments follow a "bottoms up" approach, by focusing on processes and information collection for each specific process through measurements and modeling.

Lifecycle assessments are conducted through four main steps 1) goal and scope definition 2) inventory assessment 3) impact analysis and 4) interpretation. Goal and scope definition is mainly described as defining the problem and system boundary of the LCA. It is also the main purpose of the LCA and determines what information the LCA will eventually provide. Choices such as level of detail, system boundary, assumptions and the functional unit are chosen in this step. The functional unit can be used to quantify the function of the LCA and ensure that alternative scenarios are comparable.

The inventory assessment stage involves obtaining data on the various processes that are included in the system boundary. Data needed in this stage include materials, resource use, transportation requirements, assembly processes and disposal methods.

Impact analysis involves quantifying the impact that the chosen alternatives have on various factors such as human health, ecological damage, or resource depletion. LCA software, such as SimaPro, contain method libraries to help quantify the impact. These impacts are characterized by multiplying the inventory data by a characterization factor. Different method libraries include Eco-Indicator 99, ReCiPe or the Tool for the Reduction of Assessment of Chemical and other Environmental Impacts (TRACI). TRACI is often used in North America as it was developed by the US EPA as a strong tool for a North American context. There are also another simplified impact analysis tools designed to eliminate the full inventory analysis. One example is the US EPA Waste Reduction Model (WARM). WARM focuses on different disposal pathways of different municipal solid wastes including food waste, plastic waste and yard trimmings. WARM outputs GHG emissions, energy hours, labor hours, wages and taxes, based on specifications such as volumes of waste source reduced, landfilled, composted or anaerobically digested, as well as location (US EPA 2018).

The final stage of the LCA is interpretation, which is analyzing the results and determining what the results claim about the different alternatives. An important thing to note is that interpretation should be done at every stage of the LCA process as well, to ensure accuracy at every stage of the process. Various LCAs have been conducted for different food waste solutions. Table 2.1 summarizes different LCAs, including their comparisons and main results.

A comprehensive study conducted by Lundie and Peters (2005) looked at four different food waste disposal methods for a year’s worth of household food waste in Sydney, Australia, including in-home and industrial composting, an in-unit food waste processor to a wastewater treatment plant and landfilling. This assessment determined that in-home composting, if maintained properly, has the least impact on the environment. If home-composting was not maintained, then an in-unit food waste processor has the least impact on the environment. The main limitation of this study is it did not focus on AD exclusively as a treatment option, but instead assumed that landfilling created anaerobic conditions which produced methane that would be flared. Another limitation to this study is that it focuses on Sydney, Australia and is not as directly applicable to Tampa, Florida.

Table 2.1 summary of LCAs of different organics that include FW

Author	Comparison	Location	Impact Categories	Relevant Results
(Lundie and Peters 2005)	Insinkerator, in-home C, industrial C, LF	Sydney, Australia	HTP, energy usage, AETP, AP, EP	In-home composting is most effective due to low transportation costs but only if maintained
(Xu et al. 2015)	AD, AD+sewage sludge, LF	Shandong province, China	GWP, ODP, HTP, freshwater EP, marine ecotoxicity	Electricity usage is main impact issue for AD, landfilling is significantly worse in all categories
(Grosso et al. 2012)	Centralized WtE, AD+C	Milan, Italy	HTP, AP, GWP, POCP	Creating an AD+aerobic treatment process significantly reduces GWP by 37%
(Righi et al. 2013)	LF, compost+LF, AD, AD+C	Emilia-Romagna, Italy	GWP, AP, EP, ODP, POCP	Decentralized approaches have significantly lower impacts due to decrease in transportation
(Naroznova et al. 2016)	AD+CHP, AD+electricity, incineration	Denmark	GWP	Current AD technology is more sustainable than incineration unless the incinerator is highly efficient
(Lee et al, 2020)	High solids AD, incineration, landfill, composting	Hillsborough County, FL	GWP, AP, EP,AETP	High solids AD and composting both resulted in negative GWP

HTP= human toxicity potential, AETP= aquatic ecotoxicity potential, TTP=terra toxicity potential AP= acidity potential, GWP=global warming potential, POCP= photochemical ozone creation potential, EP= eutrophication potential, ODP= ozone depletion potential, C= compost, LF=landfill, CHP= combined heat and power

Xu et al. (2015) also conducted an LCA of three different methods of biogas generation using food waste produced in China. They focused on anaerobic co-digestion with sewage sludge, anaerobic digestion of only food waste, and finally sending food waste directly to landfill. Their functional unit was 1 ton of volatile solids (VS) worth of food waste, and conducted the analysis using the ReCiPe database. Their results showed that transporting food waste to landfill, regardless of whether or not there was energy recovery, resulted in the highest environmental impact. The main limitations to this study are that it focuses on the impact in China and is therefore not directly applicable to Tampa. In addition, this study assumed a centralized treatment process, whereas USF's treatment process may be decentralized.

Grosso et al. (2012) compared a centralized incineration facility or introducing a new anaerobic and aerobic treatment facility specifically for food waste, while maintaining the existing incineration facility for all other forms of waste. They also explored multiple biogas utilization methods, such as combined heat and power (CHP), upgrading to biomethane to inject into the natural gas grid, and upgrading to biogas to be used as a vehicle fuel. The functional unit in this study was 504,000 tons of food waste and residual waste. They found that the new facility would have lower impacts in almost all categories compared to an incineration facility, and that GWP would improve by 37%. The main benefit of this study is the comparison between an AD and an incineration facility, which would be the main comparison for USF. Another benefit is that it focuses on urban areas with a denser population, similar to Tampa. Lastly, this study assumes that energy generation will occur using a CHP system. CHP systems are also commonly used in North America for AD. The limitation of this study is that it focuses on Milan, Italy. Another limitation is that it depends on effective source separating of food waste by the consumer, which is a large assumption to make and may take years of community outreach to achieve effective source separation. For USF, this assumption may hold more valid as food waste is cleared by Aramark employees and not by the consumer, and therefore it is easier to implement training to ensure effective source separation. This study also assumed that biogas production is subsidized, making it more economically feasible. This does not seem like a valid assumption to make in an American context as subsidies are very politically dependent and cannot always be relied on for economic feasibility.

Righi et al. (2013) investigated the lifecycle impact of a decentralized system treating the organic fraction of municipal solid waste (OFMSW) and dewatered sewage, with and without aerobic post-treatment, and compared it to landfilling in the Emilia-Romagna region of Italy. The functional unit of this study was 1000 tons of OFMSW and 2000 tons of sewage sludge, which is roughly the volume of waste produced by the chosen community in one year. The analysis was conducted using the CML impact analysis. For all chosen impact categories, the combination of anaerobic digestion with an aerobic post-treatment showed the least impact. The main benefit of this study is that it focused on a decentralized system, which is the main benefits of building an AD directly at USF. The limitations of this study are that it focuses on Italy, and therefore is not applicable to a North American context. As well, this study focused on co-digestion with food and sewage sludge in a continuous reactor, whereas the USF digester may operate under different conditions.

Franchetti (2013) looked at both the environmental and economic impacts of five different food waste treatment methods for the University of Toledo Food Services in Toledo, Ohio; landfilling, two stage AD, thermophilic acidogenic hydrogenesis, long term AD with trace elements, and single-stage AD. The economic analyses were conducted using internal rate of return (IRR) and the payback period. This analysis determined that thermophilic acidogenic hydrolysis had the least GHG impact, with two stage AD the second least. Landfilling had the greatest GHG impact and also had a negative IRR, making it the least environmentally and economically favorable options. The main benefit of this study is that it focuses on university dining halls, which is similar to the USF Food Recovery Project. It also focuses on a US city, whereas other LCAs focused on Europe. One limitation of this study is that it only focused on GHG emissions and did not look at any other impacts. Another limitation is that it assumed the product life for each scenario was ten years, which may not be accurate for stages such as landfills or anaerobic digesters. Another limitation in this study was the use of economic values from 1997 which were then scaled up using consumer price index (CPI) values for 2010.

Bernstad and La Cour Jansen (2012) conducted a review of food waste LCAs not as a comparison, but rather to determine how standardized the methods for food waste LCAs are and where there may be issues while utilizing the ISO 14040 method for LCAs. They concluded that LCA results vary greatly based on the system boundary as well as assumptions made, and therefore results vary greatly amongst food waste LCAs. The authors proposed coming up with more detailed guidelines for LCAs to help eliminate these biases. Some concerns that they noticed is that in LCAs, food waste pretreatment is often not taken into consideration, despite the large potential energy usage required from pretreatment. Another issue is not taking into account any reject wastes, therefore not fully balancing the system. Another assumption that does not always hold true is that anaerobic digestate may be able to be directly substituted for fertilizer or does not require further treatment to be safe for agricultural applications. Most studies also do not take into account the required treatment process for ash produced through combustion. Finally, storage results in some methane emissions, which are often not taken into account.

2.2 Anaerobic Digestion Overview

2.2.1 Microbiology

AD is a process where organic waste is biodegraded through microbial metabolism in the absence of oxygen. The process utilizes four different processes to break down complex organic matter: hydrolysis, acidogenesis, acetogenesis and methanogenesis, as shown in Figure 2.1

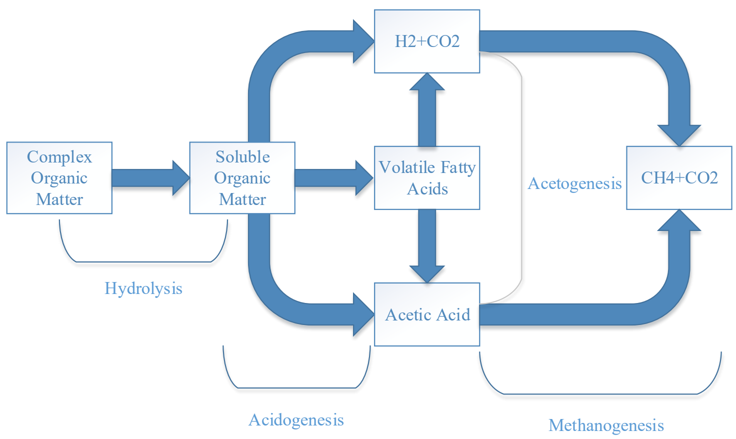

Figure 2.1 Metabolic processes and intermediates for AD. Adapted from (Hinds 2015)

Hydrolysis involves breaking down complex polymers such as proteins, lipids and carbohydrates into smaller, more soluble monomers. Hydrolysis is carried out either by strict anaerobes or facultative bacteria (Hinds et al. 2016). Hydrolysis is often viewed as the rate limiting step in anaerobic digestion (Hinds et al. 2016). One reason for this is that hydrolytic enzymes need to be absorbed onto the surface of solid substrates (Rittmann and McCarty 2001). Due to this, extensive research has been done to improve hydrolytic kinetics, including mechanical grinding, ultrasound, microwave, thermal pre-treatment or biological retreatments. Mechanical grinding can help to decrease particle size, which plays a large effect on hydrolysis rates. In anaerobic digestion, it is recommended that particle size is smaller than 0.6 mm (Meegoda et al. 2018). Thermal hydrolysis is another common pretreatment option in which substrates are first subjected to thermophilic or hyperthermophilic temperatures to speed up hydrolysis, and then anaerobically digested (Rittmann and McCarty 2001).

Acidogenesis involves producing volatile fatty acids (VFAs) from the monomers produced in hydrolysis. Acidogenesis is carried out by obligate anaerobes or facultative bacteria (Hinds et al. 2016). VFAs are the main intermediate product in anaerobic digestion and influences the health of the anaerobic digester, which will be discussed below. VFAs also destroy alkalinity, which can decrease pH and inhibit methanogenesis.

Acetogenesis involves the production of acetate, carbon dioxide and hydrogen gas from the VFAs produced during acidogenesis. Each of these products are directly needed in methanogenesis. Acetogenesis is carried out by strict anaerobes.

Methanogenesis is the final step in anaerobic digestion and results in the production of methane gas from acetate, hydrogen and carbon dioxide. Methanogenesis is carried out by strict obligate anaerobic archaea (Hinds et al. 2016). Methanogens help maintain a healthy pH due to their metabolism of acetate, which increases alkalinity and improves buffering capacity. Methanogenic bacteria are highly unstable and require very specific operating conditions to ensure the continuous production of methane gas. Methanogens are especially susceptible to pH changes, and thrive in a neutral pH, but can survive from a pH of roughly 6.5 to 7.5 (Meegoda et al. 2018). Methanogens are also sensitive to temperature changes, and do not respond positively to temperature shocking (Rittmann and McCarty 2001). They also have the longest doubling time out of all microorganisms in the AD process, and therefore plants are required to account for their slow growth by not overloading digesters until there are sufficient methanogenic archaea.

The process chemistry is balanced on the idea that the main electron acceptor in anaerobic digestion is carbon dioxide. While in practice, acetate fermenting methanogens do not utilize carbon dioxide as the electron acceptor, it still helps to balance the stoichiometric coefficients by assuming carbon dioxide (Rittmann and McCarty 2001). The full for methanogenesis of generalized organic waste is shown in Equation 1

$$C_nH_aO_bN_c + \left(2n + c - b - \frac{9df_s}{20} - \frac{df_e}{4}\right) H_2O \rightarrow \frac{df_e}{8} CH_4 + \left(n - c - \frac{df_s}{5} - \frac{df_e}{8}\right) CO_2 + \frac{df_s}{20} C_5H_7O_2N + \left(c - \frac{df_s}{20}\right) NH_4^+ + \left(c - \frac{df_s}{20}\right) HCO_3^- \qquad \text{Eq 1}$$

Where $d = 4n + a - 2b - 3c$, $f_e$represents the portion of waste organic matter converted for energy, and f_s represents the portion of waste organic matter converted into cells, which is represented by $C_5H_7O_2N$. As an example, the half reactions (Rd and Ra) for acetate, as well as the overall reaction (R) is shown in Equations 2-4 (Rittmann and McCarty 2001)

$$-R_d: \quad \frac{1}{8} CH_3CO^- + \frac{3}{8} H_2O \rightarrow \frac{1}{8} CO_2 + \frac{1}{8} HCO_3^- + H^+ + e^- \qquad \text{Eq 2}$$

$$R_a: \quad \frac{1}{8} CO_2 + H^+ + e^- \rightarrow \frac{1}{8} CH_4 + \frac{1}{4} H_2O \qquad \text{Eq 3}$$

$$R: \quad \frac{1}{8} CH_3COO^- + \frac{1}{8} H_2O \rightarrow \frac{1}{8} CH_4 + \frac{1}{8} HCO_3^- \qquad \text{Eq 4}$$

2.2.2 Full Scale Process

Full scale anaerobic digestion involves a five step process as outlined by the US EPA AG Star Anaerobic Digestion Handbook. This guideline was adapted for USF Food waste collection. An overview of the process is shown in Figure 2.2

Figure 2.2 Overview of the full scale anaerobic digestion process, adapted from Roos et al. (2004)

Substrate collection and pre-treatment involves determining a suitable substrate, as well as storage options for the substrate before it can be fed into the anaerobic digester. For this application, food waste has already been determined as the substrate, with the potential for co-digestion with tea leaves or compostable plates. Pre-treatment in this application is mechanical pre-treatment using a grinder.

Anaerobic digestion is the main metabolic reaction that breaks down organic substrates into the effluent and biogas. Anaerobic digestion can be carried out in a variety of different reactors, as will be outlined in Section 2.1.4. Each type of reactor is suited to specific substrates, land area, labor force and economic conditions.

Effluent storage involves determining the correct storage size for effluent. Storage is dependent on the requirements of the facility, from sizing to seasonal requirements. As an example, if effluent use is only required in the summer months, effluent storage tanks may need to be larger to store effluent in winter to meet summer demands. Effluent use is also dependent on whether effluent meets Federal and State standards for land application. Florida does not have any State-specific reuse laws, but Florida digesters are still required to follow federal effluent reuse standards governed under the Resource Conservation and Recovery Act (RCRA) Subtitle D Requirements for non-hazardous wastes and 40 CFR Part 258 for landfills ("Resource

Conservation and Recovery Act" n.d.). These laws help govern location restrictions, operating criteria, groundwater monitoring, closure and foreclosure care and financial assurance criteria. If effluent is stored in underground storage tanks, it is governed by 40 CFR Parts 280-282 ("Resource Conservation and Recovery Act" n.d.). These laws help determine required standards for underground storage, including technical standards, state approval processes, and rules governing approved underground storage tanks.

Another law that focuses on land application of biosolids is 40 CFR Part 503- Standards for the use and disposal of sewage sludge. This regulation helps to determine general requirements and pollutant limits for land application (Subpart B), surface disposal (Subpart C) limit pathogen concentrations and vector attraction reduction (Subpart D) ("Biosolids Laws and Regulations" 2019). Florida also governs biosolids land application based on Rule Chapter 62-640 Biosolids. This regulation also helps determine requirements for nutrient management (Subsection 500), pathogen reduction and vector attraction reduction (Subsection 600), monitoring (Subsection 650), and requirements to meet Class AA, A and B for land application (Subsection 700) ("Biosolids" 2010).

Gas handling involves transporting biogas from the digester into the gas storage system. In smaller scale systems, this is pressure driven by the natural flow of gas to an area of lower density. In larger scale systems, biogas flows through a storage tank by creating a vacuum, normally through a gas pump or blower. Gas handling also requires many fail-safes, including gas meters, pressure regulators, and condensate drains (Roos et al. 2004). This is due to the inherent risks involved with high pressure gas handling.

The final stage in the process is gas use. Gas refinement standards are different depending on the preferred use of the gas. An overview of different gas applications and their required pre-treatment is provided in Section 2.1.5. Gas combustion systems are governed under the Clean Air Act and under CFR 40 Part 60 rules for air emissions for new stationary devices ("Stationary Gas and Combustion Turbines" 2012). If air emission volumes are below the minimum government threshold, then the system can be exempt from permitting through the Clean Air Act ("Controlling Air Pollution from Stationary Engines" n.d.).

2.2.2 Operating Parameters

All microorganisms involved in anaerobic digestion work together in a symbiotic relationship, and if one microbe is not thriving, then the entire digester can sour. Due to this symbiotic relationship, various operating conditions are required to maintain a healthy digester and ensure each organism is operating as efficiently as possible.

2.2.2.1 Organic Loading Rate and Solids Concentration

Loading rate refers to the volume of influent entering a specifically sized continuous or semi-continuous digester over a unit of time. The organic loading rate (OLR) can be calculated using the following equation:

$$OLR = \frac{S_0 \times Q}{V} \qquad \text{Eq 5}$$

where So refers to the influent solids concentration (kg/m^3 VS or COD), Q refers to the flow rate (m^3/time) and V (m^3) refers to the volume of the reactor. The appropriate organic loading rate is most often dependent on the type of reactor and the substrates used in the digester. The Water and Environment Federation estimates that the average wastewater anaerobic digester operates between 1.6-6.4 $\frac{kg\ VS}{m^3 \times day}$ (Roos et al. 2004). Loading rates should be kept consistent to avoid shocking the bacteria within the digester. Methanogenic bacteria especially are susceptible to loading shocks (Meegoda et al. 2018).

As per Equation 4, loading rates can increase in the event of increased flow rates or increased solids concentrations. For this reason, flow rates may be normalized through holding tanks for larger operations such as wastewater treatment plants (Labatut and Pronto 2018). Solids concentrations must also be kept at appropriate levels to prevent too high of an organic loading rate.

Anaerobic digestion can also occur at various solids concentrations. Depending on total solids (TS) concentration, AD can be low solids (<15% TS), medium solids (15-20% TS) and high solids (>20% TS) (Kothari et al. 2014). Low solids AD requires larger quantities of water but is easier for system stabilization. High solids AD limits added water usage but may be more difficult to maintain a stable digester, due to the need for larger volumes of inoculum, longer retention times and larger potential for VFA accumulation (Kothari et al. 2014).

2.2.2.2 Temperature

Temperature plays a strong role in enzyme activity, microbial growth, methane yield and quality of fertilizer produced. There are three main operating conditions in anaerobic digestion- psychrophilic (10-30 ^oC), mesophilic (30-40 ^oC), and thermophilic (50-60 ^oC). Up to roughly 60 ^oC, increasing temperature will increase the production of methane gas in the reactor (Labatut and Pronto 2018). However, most digesters operate in the mesophilic range. The methanogens which thrive in the mesophilic range are more stable to changes in temperature, resulting in decreased risk of the reactor souring. Thermophilic conditions can also decrease the solubilization of food waste (Labatut and Pronto 2018). Psychrophilic conditions are normally only used in small anaerobic digesters, often seen in homes and small farms. This range is not recommended for large scale applications as the high cost of larger reactors required for psychrophilic conditions is not often seen as economically feasible or profitable in industrial scales. One option to achieve benefits of both mesophilic and thermophilic digesters is through the operation of two-phase anaerobic digestion, where the first phase operates in thermophilic conditions with a shorter retention time, and the second phase operates in mesophilic conditions with a longer retention time.

2.2.2.3 pH and VFA concentration

The pH of a reactor plays an important role in the health of anaerobic microorganisms. Methanogenic archaea thrive under a neutral pH, allowing for ranges between 6.5 to 7.2 (Rittmann and McCarty 2001). Methanogens increase the pH in the reactor, due to the uptake of acetic acid to produce methane. At the same time, acetogenesis and acidogenesis consume alkalinity, due to the production of VFAs. If the pH drops too much, methanogens are inhibited. This will cause an even faster drop in pH which will in turn sour the reactor (Rittmann and McCarty 2001). Methanogens also grow slower than other anaerobes, which runs the risk of the reactor souring during the start-up of an anaerobic digester. One way that plants prevent this is by slowly increasing the organic loading rates in the reactor to facilitate the slow growth of the methanogens. Another prevention technique is to add an alkalinity source which can buffer any pH changes. A VFA concentration of roughly 300 mg/l is also required in the digester to ensure methanogens are fed (Schuyler 2013). VFA concentrations above 1500-2000 mg/l will begin to show methanogenic inhibition (Labatut and Pronto 2018).

2.2.2.4 VFA to Alkalinity Ratio

The VFA:Alkalinity ratio is an effective indicator of the health of the reactor, and whether the pH can be maintained at an appropriate level (Rajagopal et al. 2017). Alkalinity is the measure of the ability of a solution to resist a change in pH and is usually reported in units of $CaCO_3$ equivalents. A sufficient alkalinity level ensures that methanogens do not have to deal with a pH shock, especially during loading when there is a sharp increase in hydrolysis. It is recommended that digesters have an alkalinity of at least 1000 mg/l (Roos et al. 2004). The VFA:Alkalinity ratio should be maintained from 0.1 to 0.35 to ensure that there is sufficient alkalinity to prevent pH shock to the methanogens (Roos et al. 2004). If the VFA:Alkalinity ratio increases above 0.35, then there is probably overloading of the reactor, and the OLR should be decreased.

2.2.2.5 Carbon to Nitrogen Ratio

The Carbon to Nitrogen (C:N) ratio plays an important role in the efficiency of the anaerobic digester. The optimal C:N ratio is between 20:1 to 30:1, due to the fact that a much larger amount of carbon is required in the chemical reactions than nitrogen (Rittmann and McCarty 2001). If the C:N ratio is too low, such as in high-protein waste, the concentration of ammonia increases resulting in ammonia toxicity (Rittmann and McCarty 2001). If the C:N ratio is too high, microbes in the digester do not have access to enough N for cell synthesis, limiting biogas production (Hinds et al. 2017). However, C:N ratio varies greatly between substrates and should be studied specifically to determine appropriate conditions. Food waste has a C:N ratio of roughly between 14:1 to 18:1 (Meegoda et al. 2018), which, when digested alone, can result in free ammonia inhibition. To prevent this from occurring, food waste is often co-digested with other organic matter to help ensure a more accurate C:N ratio. Co-digestion will be further explained in Section 2.1.3.

2.2.2.6 Food to Microorganism Ratio

The food/microorganism (F:M) ratio represents the ratio of volume of the substrate and the inoculum inside the reactor. This is an important parameter in AD because a balanced population of microorganisms can ensure a healthy production of methane from the beginning of the digestion period (Lee et al. 2019). Optimal F:M ratios vary considerably depending on the substrates and inoculums, and literature values have shown anywhere from 3:1 to 1:7 (Hinds et al. 2017). F:M ratio is related to the OLR of a continuous process in that an incorrect OLR results in an incorrect F:M ratio, leading to either too much substrate that microorganisms cannot metabolize, or an inefficient system that can handle more substrate than it is receiving. A higher OLR results in a higher F:M ratio, which can overload the system.

2.2.2.7 Retention Time

Solids retention time (SRT), also referred to as the mean cell residence time (θ_x), refers to the ratio of active biomass in the system to the production or wasting rate of active biomass (Rittmann and McCarty 2001). It is also described as the average amount of time solids remain in the digester. Hydraulic retention time (HRT) refers to the average time that liquids remain in the system. For a continuous stirred tank reactor without recycle, solids and liquids spend an equal time in the reactor and therefore,

$$SRT = HRT = \frac{V}{Q} \qquad \text{Eq 6}$$

The required SRT for an AD varies depending on the reactor type, environmental conditions and substrate. For food waste, single stage mesophilic ADs require longer SRTs, from 10-60 days; however two-stage reactors require only 10-15 days per reactor (Zhang et al. 2014). SRT is determined based on the Monod equation for microbial biomass growth. There must be a minimum SRT in any digester to avoid cell washout. Cell washout occurs when the loading flow rate of the digester is too high and biomass does not have sufficient time to grow and cannot sustain itself in the reactor. The minimum SRT is calculated by:

$$SRT = SF * SRT_{min} \cong \frac{SF}{\mu_{max}} \qquad \text{Eq 7}$$

Where SF represents a safety factor and μ_{max} represents the maximum specific growth rate of the microorganisms. In anaerobic digestion, each microbial group has a different maximum specific growth rate and therefore the reactor must be designed for the slowest growing microorganisms, the methanogens. Digesters SRTs are designed by multiplying SRT_{min} by a safety factor to determine the design SRT for the digester. μ_{max} can be influenced by certain parameters such as temperature. In anaerobic digestion, higher temperatures result in a higher μ_{max}. This can help shorten the SRT and a decreased reactor volume.

2.2.3 Types of Digesters

Anaerobic digesters can also be fed continuously, semi-continuously, or be batch processes, shown in Table 2.2 (Rittmann and McCarty 2001). Anaerobic digesters can also further be classified into how many stages the reactor has. Multi-stage reactors are often implemented to separate hydrolysis, to help facilitate the slow growth of methanogens and prevent reactor souring, and maximize methane yield (Djalma Nunes Ferraz Júnior et al. 2016a).

Table 2.2 Overview of anaerobic digester reactor types (adapted from Debruyn and Hilborn 2007)

Name	**Solids (%)**	**HRT (days)**	**Temperature**
Covered lagoons	n/a	30-40	Psychrophilic
Plug flow reactors (PFR)	11-14	10-25	Mesophilic or thermophilic
Constant stirred tank reactors (CSTRs)	5-10	10-25	Mesophilic or thermophilic
Fixed film	<1	a couple of days	Mesophilic or thermophilic

2.2.4 Biogas

Biogas is one of the main biproducts of anaerobic digestion and represents an exciting opportunity to replace fossil fuels with a renewable energy resource. Biogas is most often composed of 50-70% methane, 30-50% carbon dioxide, and a few trace elements including water vapor, siloxane and hydrogen sulfide (Labatut and Pronto 2018). The main source of energy comes from methane, which can provide roughly 50-55 MJ/kg (Rittmann and McCarty 2001). Carbon dioxide does not contribute to power generation, and the trace can contribute to corrosion of machinery. For these reasons, it is necessary to treat biogas before it is used, depending on its selected end use. An overview of different biogas purification techniques is shown in Table 2.3

Table 2.3 Purification Techniques for different biogas component removals

Contaminant Removed	**Technology Needed**	**Source**
Carbon Dioxide	Membrane Separation	(Rittmann and McCarty 2001)
H_2S	Iron oxide media or activated carbon	(Lema n.d.)
Water Vapor	Condenser or chemical adsorption	(Labatut and Pronto 2018)
Siloxanes	Siloxane absorbing media such as activated carbon	(Soreanu et al. 2011)

The chosen biogas usage determines the needed purification techniques. Most applications require hydrogen sulfide removal due to how corrosive the gas is. If biogas is being used for electricity generation or heating and cooling, it is important to remove water vapor and siloxanes, which can damage generators. Siloxanes are detergents commonly seen in shampoos and detergents, but may end up in FW AD in lower concentrations through food preparation, and can be damaging to generators (Soreanu et al. 2011). Carbon dioxide is the most expensive biproduct to remove and requires the use of membrane technology. Carbon dioxide removal is only necessary to create biomethane or a renewable natural gas and is usually not a recommended end use for biogas unless there is a large enough volume. Creating biomethane or natural gas can be useful for injecting natural gas back into the grid, or powering utility vehicles ("How to make RNG/ Biomethane" n.d.).

In many wastewater treatment plants (WWTPs) in the US, electricity generation is one of the most common uses of biogas. Normally, energy is generated through combined heat and power (CHP) engines, which generate both electric energy through the use of a generator, as well as harvest the excess heat which can then go back into heating the digester and maintaining either thermophilic or mesophilic conditions. Table 2.4 provides an overview of different CHP systems

Table 2.4 Overview of different CHP technology, as adapted from Labatut et al, 2018

CHP Technology	**Capital Cost ($/kW)**	**Operating Cost ($/kW)**	**Efficiency (%)**
Gas Turbine	700-2000	0.006-0.011	22-36
Microturbine	1100-2000	0.008-0.02	25-35
Reciprocating Engine	800-1500	0.008-0.025	22-45
Fuel Cell	1000-5000	0.03-0.04	40-60

2.2.5 Anaerobic Digestion of Food Waste

Food waste is a common AD substrate, due to its high biodegradability. When landfilled, the high biodegradability of food waste results in the largest portion of methane released from municipal solid waste (Labatut and Pronto 2018). Therefore, recovering resources from food waste through anaerobic digestion can result in harnessing methane from a waste product and converting it to energy. In Europe, anaerobic digestion is the most commonly used method for treating organic municipal waste such as food waste (Labatut and Pronto 2018).

The anaerobic digestion of food waste has been extensively researched and applied in large scale applications. It is commonly seen in Europe, where source separation has been strongly pushed since the 1990's (Baere and Mattheeuws 2015). Germany and Spain can treat 2 million and 2.5 million tons per year, respectively (Baere and Mattheeuws 2015).

FW AD is a growing emerging technology in the US, seeing a large spike in the amount of digesters in the early 2000s (US EPA 2019). Food waste AD in the US commonly occurs through three different digestion types- stand-alone digesters, co-digestion with animal waste, or co-digestion at wastewater treatment plants (Labatut and Pronto 2018). According to the latest EPA study, there are 62, 59 and 77 stand alone, co-digested, and wastewater treatment plant food digesters in the US respectively, which process a total of 10 million tons of food waste per year, and produce enough biogas to power 79,000 homes each year. (US EPA 2019). Most single-stage AD are multi-source food waste digesters, working as a for-profit digester that collects various substrates from surrounding industry (US EPA 2019). These digesters mostly tend to take fats, oils and grease, food processing waste from industry, or beverage processing waste from industry (US EPA 2019).

Stand-alone AD of food waste often leads to process instability due to the low buffering capacity of food waste (Labatut and Pronto 2018). This leads to a sharp decrease in pH due to the formation of VFAs, which in turn can inhibit methanogenesis. Proteins in food waste also lead to large nitrogen concentrations in the reactor, well above an ideal C:N ratio. This results in the formation of FA which can inhibit methanogenesis (Banks et al. 2011). Food waste as a substrate is also known for its lower pH, and therefore requires inoculum with a larger buffering capacity (Pavi et al. 2017). Therefore, many studies have attempted to stabilize food waste AD to prevent methanogenic inhibition.

Banks et al. (2011) looked at the long-term operation of source-separated domestic food waste in a 900 m^3 reactor loading on average 2.5 kg VS m^{-3} day^{-1} and an HRT of 80 days operated at thermophilic conditions. These conditions resulted in a methane yield of 402 m^3/ tonne VS and 62.6% methane in the biogas. In this application, food waste was first ground and mixed with recirculated digestate, and then pasteurized at 70 ^{o}C for one hour before entering the thermophilic digester. These operating conditions resulted in no methanogenic inhibition throughout the operation of the reactor.

Pavi et al. (2017) looked at the digestion of organic municipal food waste with fruit and vegetable waste in batch reactors, with a 1:1 and 1:3 mixing ratio compared to both substrates digesting on their own, at an F:M ratio of 1. Their inoculum was already acclimated to food waste and had an alkalinity of 906 mg/l as HCO_3^-. They reported a maximum methane yield at the 1:3 mixing ratio, at 396.6 ml g^{-1} VS, with a C:N ratio of 34.7. This is higher than the usual methane range of 20-30, but also indicates that substrate and inoculum choice play a role in determining the optimum C:N to maximize methane yield. Further studies, including their operating conditions and main results can be seen in Table 2.5

Table 2.5 Summary of different food waste AD experiments with varying operating conditions

Substrate	Operating Conditions	Results	Source
Source-separated dining hall food waste	Two dry parallel CSTR reactors operated at mesophilic and thermophilic conditions	Stepwise increase of OLR helped to increase methane production without any reported process inhibition	(Nguyen et al. 2017)
Food waste from a San Francisco Waste Management company	Batch tests with an OLR of 6.8 and 10 g VS/L operated at thermophilic conditions	Methane yield of 0.44 L/ g VS with methane production failing after 6 days and a C:N of 14.8	(Zhang et al. 2007)
Korean food waste at 12.8% solids	CSTR with HRTs varying from 8-12 days and temp varying from 40 to 55 C	Highest methane yield at achieved at 50 C and 12 day HRT of 223 l kg^{-1} soluble COD	(Kim et al. 2006)
Dining hall food waste	Batch reactors at F:M ratio of 1 with pH adjustment to 7, 8 or 9	pH adjustment to 8 resulted in highest methane yield of 170.1 ml g^{-1} VS, almost 8x higher than the control reactor	(Yang et al. 2015)
Food waste mixture including rice, noodles, bread, vegetables	Batch mesophilic reactors with FW processed either through grinding or bead milling	Particle size of 0.7 mm achieved through bead milling resulted in largest methane production of 320 ml g^{-1} COD	(Izumi et al. 2010)
Cafeteria waste	Semi-continuous reactors operated at 37 C supplemented with trace amounts of Co, Fe, Mo, Ni	Adding trace elements resulted in stable operation of long term anaerobic digestion	(Zhang and Jahng 2012)
Food waste from garbage collection company mixed according to average Japanese food waste	Single stage mesophilic reactors with an OLR varying from 3.7 to 12.9 kg VS m^{-3} day^{-1}	9.2 kg VS m^{-3} day^{-1} OLR resulted in highest methane yield of 455 ml g^{-1} VS	(Nagao et al. 2012)

2.2.6 Co-digestion

Co-digestion is an effective method of maintaining an appropriate C:N ratio in the digester without needing to pay for an added substrate. Co-digestion can also benefit large scale applications as it allows for companies to accept multiple substrates and become more economically feasible. As previously mentioned, an optimum C:N ratio is 30:1, but food waste is only at 18:1 (Meegoda et al. 2018). Therefore, various studies have been conducted to determine opportunities for co-digestion with food waste, shown in Table 2.6.

Table 2.6 Overview of FW co-digestion research conducted

Substrates	**Experimental Setup**	**Results**	**Reference**
FW+ cattle manure	Batch tests of C:N ratios of 15.8,17.1 and 17.8	C:N ratio of 15.8 produced largest volume of biogas	(Zhang et al. 2013)
FW+ Yard Waste	Batch tests with C:N ratios of 16.9 to 32.2	F:M ratio of 1 and C:N ratio of 16.9 lead to greatest methane yield (120 L/ kg VS)	(Brown and Li 2013)
FW + dairy manure	30 day batch reactors	100% food waste without dairy manure resulted in greatest biogas yield	(El-Mashad and Zhang 2010)
FW+ dairy manure	10 and 20 g VS/l loading rate batch reactors	FW +NaOH achieved the highest methane yields (458.4 mL/g VS) but FW+cow manure at 20 g VS/l were highest non-dosed option (310.8 ml/ g VS)	(Li et al. 2010)
FW+ yard waste + waste activated sludge	High solids batch reactors at multiple F:M ratios and semi-continuous reactors at a 28 day SRT	F:M ratio of 1 resulted in highest methane yield for batch processes and including fast and slow alkalinity sources showed improved performance	(Lee et al. 2019)

2.2.7 Anaerobic Digestion of Tea Leaves

Tea leaves have not been as extensively explored for their anaerobic biodegradability as food waste has. To our knowledge, no prior testing has occurred through co-digestion of tea leaves and food waste. One study looked at the AD of tea leaves and reported 480 ml CH_4 /g VS after nutrient addition. Tea leaves provide a novel source of organic matter due to the strong prevalence of tea universally. Tea in the US represents a $6.4 billion industry (Perkins 2019).

2.2.8 Anaerobic Digestion of Compostable Plates

Compostable plates represent another novel area in which anaerobic digestion could possibly be an effective treatment method. Anaerobically digesting single use plates could be effective in helping with contamination, as one-use paper and plastic are often a major source of contamination in food waste (Labatut and Pronto 2018). The main downside of compostable plates is that although they do provide a much-needed carbon source to AD, they have minimal content of N, P and K (Wang et al. 2012). This provides a good opportunity, however, for co-digestion with more nitrogen rich substrates such as food waste.

One common material used to make compostable plates is sugarcane waste. Sugarcane waste can be used to derive a bio-poly lactic acid (PLA) that can be molded into sugarcane plates (Benn and Zitomer 2018). Both sugarcane vinasse and bagasse has already been studied for their anaerobic biodegradability (Ferraz et al. 2016, Fuess et al. 2017, Fuess et al. 2018) and appears to be a promising technology for anaerobic digestion. Fuess et al. (2017) noted a forty day start up period through a two phase anaerobic structured bed reactor but then continued stable operation through the end of the study.

Compostable plates have also been studied for AD but often less extensively than sugarcane biproducts. Yagi et al. (2009) studied the anaerobic biodegradability of PLA under both mesophilic and thermophilic conditions. Their substrate was a PLA powder digested in a stirred batch reactor. On its own, PLA only achieved 10% biodegradability after ten days. Benn and Zitomer (2018) also attempted to biodegrade PLA bioplastics using either thermal or chemical pretreatments. The untreated PLA followed similar results to Yagi et al, in which negligible biodegradability occurred. However, PLA pretreated at 90° C for 48 hours and then chemically pretreated to a pH of 10 produced a methane yield of 86 mL CH_4/ g theoretical oxygen demand. This paper also noted that PLA does not biodegrade under mesophilic conditions and requires thermophilic conditions. One comprehensive study (El-mashad et al. 2012) looked at the co-digestion of sugarcane PLA plates with food waste at an F:M ratio of 2, under thermophilic conditions, in equal volumes by TS. Plates were digested for forty-five days and achieved a final biogas yield of 600 ml / g VS. Biodegradable plates out-performed the food waste-only control, which produced 300 ml biogas/ g VS.

2.2.9 Effect of Acclimated Inoculum on Anaerobic Digestion

Acclimated inoculum is anaerobic digestate with microorganisms that have been exposed to a specific substrate and become better suited at breaking down that substrate. This is due to the enrichment of microorganisms that release enzymes more suited to a substrate and other microorganisms dying out (Hinds et al. 2016). The use of acclimated inoculum has been proven to decrease lag phases at the beginning of digestion and can increase methane yield (Lee et al. 2019). Lee et al (2019) saw a 38% increase in methane yield through the use of acclimated inoculum when conducting high solids anaerobic digestion of food waste, yard waste and waste activated sludge. For the organic fraction of municipal solid waste, which includes a large portion of food waste, anaerobic sludge from a wastewater treatment plant is considered a better inoculum than other sources of anaerobic microbes, such as cattle, corn silage or swine sludge

(Forster-Carneiro et al. 2007). Pre-digested sludge also is considered a better inoculum source for ligno-cellulosic waste than fresh cattle manure (Sharma et al. 1988). Hinds et al. (2016) also looked at the enhancement of biodegradation of ligno-cellulosic waste using anaerobic sludge obtained from a pulp and paper anaerobic digester, and noticed that pulp and paper sludge increased the rate of hydrolysis, often the rate-limiting step in the AD of ligno-cellulosic waste due to the arrangement of cellulose with the lignin.

Chapter 3. Materials and Methods

3.1 Experimental Methods

The main goal of this research was to determine the most effective method to implement full-scale anaerobic digestion at USF. This will be done through co-digestion biochemical methane potential (BMP) assays and conducting a lifecycle assessment to determine the full impact of this system over the life of the digester. One opportunity for increasing methane production has been through the co-digestion of tea leaves and compostable plates with food waste through a partnership with TBD Café at 301 in Riverview, Florida. Co-digestion with food waste, tea leaves and compostable plates has not yet been explored and therefore represents a novel area to stabilize anaerobic digestion. The BMP tests were conducted in three phases, as outlined in Table 3.1

Table 3.1 Biochemical methane potential assays conducted through the combination of food waste (FW), tea leaves (TL) and compostable plates (P) and added alkalinity (alk)

Experimental Phase	**Substrates**	**Mixing Ratio (FW TS :substrate TS)**	**Food/Microoganism Ratio (VS/VS)**	**Alkalinity added**
1	FW	n/a	1	No
	FW+TL	1:1	1	No
	FW+P	1:1	1	No
2	FW	n/a	0.5	No
	FW+alk	n/a	1	5000 mg/l
3	FW+TL+P	2:1:1	1	No
	FW+TL+P	1:1:1	1	No

An initial 2.5% total solids (TS) concentration was chosen to mimic the TS of a substrate after it was ground in an Insinkerator, to allow for a closer comparison of potential larger-scale pretreatment practices. In Phase 1, the substrates were mixed at a food to microorganism (F:M) ratio of 1, a common ratio seen in anaerobic digestion (Lee et al. 2019).

Phase 2 looked at how to increase methane yield of FW, either by adding an alkalinity source to reactors or by decreasing the F:M ratio. Based on the results in Phase 1, it was determined that food waste on its own could not anaerobically digest without souring the reactor. Therefore, 5000 mg/l of alkalinity as $CaCO_3$ was introduced. This value was chosen as the maximum alkalinity recommendations for AD (Schuyler 2013). Alkalinity was introduced using both fast release alkalinity (baking soda) and slow release alkalinity (oyster shells) at a ratio of 0.3, based on results from prior studies in our lab (Lee et al. 2019). Another digestion set at an F:M ratio of 0.5 with no added alkalinity was also studied.

Phase 3 was designed to investigate the effects of mixing FW, TL and P together to determine whether mixing all 3 substrates would increase or decrease methane yield, as well as to decrease the lag period observed in Phase 1. Phase 3 also introduced acclimated inoculum obtained from Phase 1 of digestion, at 35% acclimated inoculum and 75% fresh inoculum obtained from Clearwater Wastewater Treatment Plant. This ratio was chosen due to the availability of acclimated inoculum. The introduction of acclimated inoculum was chosen to see if acclimated inoculum could decrease the long lag periods observed in Phase 1.

At the time of writing this thesis, Phase 1 has been completed. Phases 2 and 3 are ongoing so only preliminary results are included in Appendix E.

3.1.1 Biochemical Methane Potential Assays

Biochemical methane potential (BMP) tests were conducted in 250 ml glass bottles. Each digestion set was set up in three sets of duplicates to allow for chemical analysis at 1 week, 3 weeks and at the end of the BMP test, when biogas production levels in test bottles were at similar levels to the inoculum control. Bottles were filled with substrate and inoculum and then flushed with nitrogen gas for 1 minute to remove any oxygen and ensure anaerobic conditions. The bottles were then sealed using rubber septums and crimped shut using metal crimp caps. All BMP assays included an inoculum-only controls, also done in duplicates, that had the same volume and source of inoculum as the test bottles. The biogas and methane production from the inoculum were subtracted from the other bottles when calculating methane yields. Inoculum was obtained from the Clearwater Wastewater Treatment plant, which had been successfully used as an inoculum in prior experiments in our lab (Lee et al. 2019). Food waste substrate was obtained from Champion's Choice Dining Hall through Aramark Dining. Tea leaves were obtained from TBD Café at 301 in Riverview, Florida. Compostable plates were produced by Monogram Cleaning and Disposables (Rosemont, IL) and were also obtained from TBD Café at 301. Each substrate came from a specific source and required its own form of pretreatment to prepare for the BMP assay, as outlined in Table 3.2

3.1.1 Chemical Analysis

Chemical analysis was carried out on all liquid samples from both the BMP tests and the pilot-scale system (Appendix A). Measurements included TS, volatile solids (VS), alkalinity, pH, ammonia, chemical oxygen demand (COD), and volatile fatty acids (VFA).

Table 3.2 Overview of substrates used in the BMP assays

Substrate	**TS Mg/l**	**VS Mg/l**	**Detailed overview**	**Location Sampled**	**Pretreatment**
Food Waste	25100	22800	Fruit peels including melons, strawberries	Champion's Choice Dining Hall	Ground in Insinkerator with tap water
Tea Leaves	29000	27900	Green tea, passion fruit tea	TBD Café at 301	Mixed with DI water to achieve 2.5% solids
Compostable Plates	95000	93700	Compostable Plates made from sugarcane	Monogram Cleaning and Disposables	Cut into small mm thick strips and mixed with DI water to achieve 2.5% solids
Inoculum	26400	18800	AD Inoculum from a mesophilic digester	Clearwater Wastewater Treatment Plant	N/A

TS and VS were measured using *Standard Methods* 2540 (APHA 2012). Samples were centrifuged for 20 minutes at 9000 rpm for alkalinity, pH, ammonia, COD and VFAs. The centrate obtained was then filtered through a 0.45 µm GE Whatman Glass filter (Pittsburgh, PA). COD and ammonia measurements were diluted using deionized (DI) water based upon expected values from literature and previous lab studies. Alkalinity (as $CaCO_3$) measurements were done using 0.1 N hydrochloric acid (HCl) according to *Standard Methods* 2320 B (APHA, 2012). pH was measured using a Thermo Fisher Scientific 5 Star pH probe (Waltham, MA), which was calibrated at the beginning of each testing day using 4.0, 7.0 and 10.0 pH buffer solutions. COD measurements were done using Hach High Range COD TNT 822 vial tests (Loveland, CO) as per *Standard Methods* 5200 B (APHA, 2012). VFA measurements were done using Hach VFA TNT 872 vial tests (Loveland, CO), following *Standard Methods* 5560 D (APHA, 2012). Ammonia measurements were done using a Timberline Instruments Model TL-2800 Ammonia Analyzer (Boulder, CO) connected to a CETAC ASX-260 Auto-Sampler (Omaha, NE).

Ammonia standards were produced at concentrations of 1.0, 5.0, 10, 15, 20 and 25 mg/l as ammonia.

Biogas production in the BMP assays were measured using a 50 ml Cadence Science frictionless syringe (Cranston RI) fitted with a 25 gauge BD needle (Franklin Lakes, NJ). Volumes were adjusted to STP conditions using the ideal gas law. Biogas production in the pilot digesters were measured using a wet tip gas meter (Nashville, TN). Methane content was measured using *Standard Methods* 6211 C (APHA, 2012).

3.3 Lifecycle Assessment

WARM inputs were determined based on data provided by RecyclingWorks Massachusetts for college and university scale pre- and post-consumer food waste, shown in Table 3.3 (“Food Waste Estimation Guide” 2018). According to RecyclingWorks Massachusetts, on-campus students produce 141.75 pounds of food waste each year in University dining halls, and off campus students produce 37.8 pounds of food waste. This estimate includes both pre-consumer and post-consumer waste and while it is focused on student waste, accounts for other FW sources on campus including employees. These values were then scaled to match the population of on and off campus students at USF Tampa based on available dorms and Spring 2020 enrollment information (“Record-breaking number of students to live on USF’s main campus” n.d.; “USF InfoCenter” n.d.).

WARM allows for model inputs to help personalize outputs to a specific geographic location and disposal methods. Florida was chosen as the state where the disposal methods would occur. Anaerobic digestion was assumed to be wet digestion, to help account for mechanical grinding pre-treatments which includes adding water. Finally, it is assumed that digestate is not cured, which currently requires further testing to ensure compliancy. The WARM model will output GHG emissions, energy usage, labor hours, wages and taxes. This thesis does not focus on the result of economic impacts because the WARM model undervalues the labor hours, wages and taxes required from anaerobic digestion and therefore may not be an accurate model (US EPA 2018).

The base case scenario follows USF's current practice that food waste is currently combusted at the McKay Bay incineration facility, located 7.9 miles away from USF. The alternative scenario is an on-campus AD with negligible transport distance from the source. The functional unit in this model is food waste produced at the University of South Florida over one year, or 1015 mtons of food waste. The food mix is outlined in Table 3.3. The functional unit was determined through data provided by RecyclingWorks Massachusetts for colleges and universities ("Food Waste Estimation Guide" 2018), as well as USF's population in Spring 2020. Future models will validate these assumptions through food waste audits conducted at USF.

Table 3.3 WARM Input parameters for lifecycle assessment

Input	**Value**	**Source**
Population	On campus- 6,304 Off campus- 35,565	("Record-breaking number of students to live on USF's main campus" n.d.; "USF InfoCenter" n.d.)
Distance	7.9 miles to McKay Bay	("McKay Bay Refuse to Energy Facility" n.d.)
Emissions	"Worst Case" scenario of meeting EPA guidelines	(Curren 2019)
Food Mix	50% fruit and vegetable, 25% meat, 25% grain	("Food Waste Estimation Guide" 2018)

WARM follows the Intergovernmental Panel on Climate Change (IPCC) inventory method to evaluate global warming potential and determine sources and sinks of GHG emissions (US EPA 2018). This approach looks at the GHG emissions produced over a 100 year period . Incineration models assume that incineration facilities are utilizing produced energy for electricity generation, which is consistent with McKay Bay's WtE facility. Anaerobic digestion models assume the land application of digestate, as well as fugitive methane emissions from digestates, and methane leaks from the digester.

Figure 3.1 represents the lifecycle that WARM follows when evaluating either incineration (top pathway) or anaerobic digestion (bottom pathway) (US EPA 2018). When modeling incineration, WARM does not account for any GHG emissions that result from landfilling ash produced from incineration. This is a fair assumption to make for this model because ash should no longer have any excess energy that can be converted into GHGs. WARM also does not account for the transportation GHG emissions when transporting digestate to where it will be land applied, in this case USF athletics field. This is also a fair assumption for this model as this evaluation assumes that AD occurs on campus, and any produced digestate will also be used on campus (US EPA 2018).

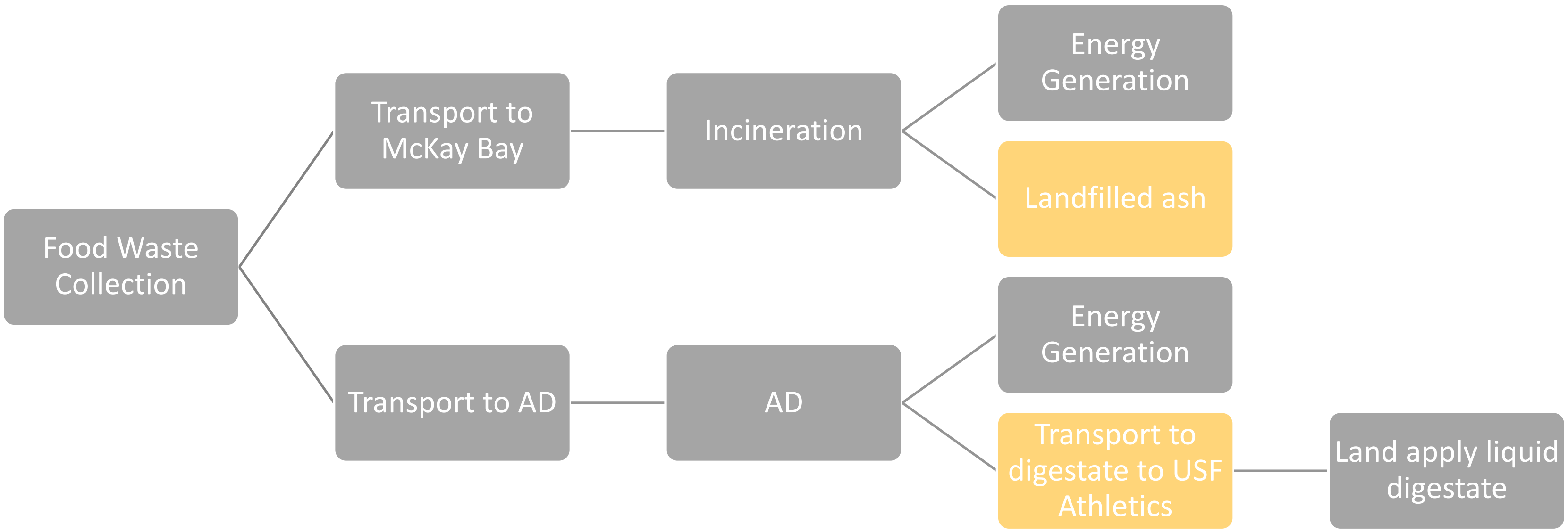

Figure 3.1 Product lifecycle of the base case (top pathway) and alternative for the LCA according to WARM

Steps outlined in gold are not included in WARM's pathway

Chapter 4. Results and Discussion

4.1 BMP Assays

Three different phases of BMP assays were conducted to effectively determine the ability for FW to anaerobically digest either on its own, or through co-digestion with TL or P. Phase 1 was an initial biodegradability study to determine whether TL or P could biodegrade when co-digested with FW, and how it would compare to FW alone. Phase 1 digesters were operated at an F:M ratio of 1 and a TS of 2.5%. Phase 2 looked at ways to increase the methane yield of FW alone, either through simultaneously introducing sodium bicarbonate and oyster shells as an alkalinity source, or by decreasing the F:M ratio to 0.5. Solids percentages remained the same. Phase 3 looked at reducing lag periods seen in Phase 1for TL and P through the introduction of acclimated inoculum, as well as co-digesting all three substrates together to determine the methane yield. Phase 3 was set at an F:M ratio of 1 and a TS of 2.5% as well. Phases 2 and 3 are in operation during the writing of this thesis, and preliminary data is provided in Appendix D.

4.1.1 Biogas Properties

Phase 1 was operated for 92 days. Biogas production, biogas quality, methane production and methane yield for Phase 1 are shown in Figure 4.1. FW only had the lowest biogas volume at 41.2$\pm$24 ml. The highest volume of biogas produced was on day 10, with a cumulative biogas volume of 139 ml. The decrease in biogas volume after day 10 was due to the inoculum producing more biogas than FW. FW+TL saw a biogas volume of 946$\pm$40 ml and FW+P saw a biogas volume of 1,337$\pm$100 ml. It is important to note that biogas volumes for FW+P did not level out after 92 days, and there may be potential for more biogas production.

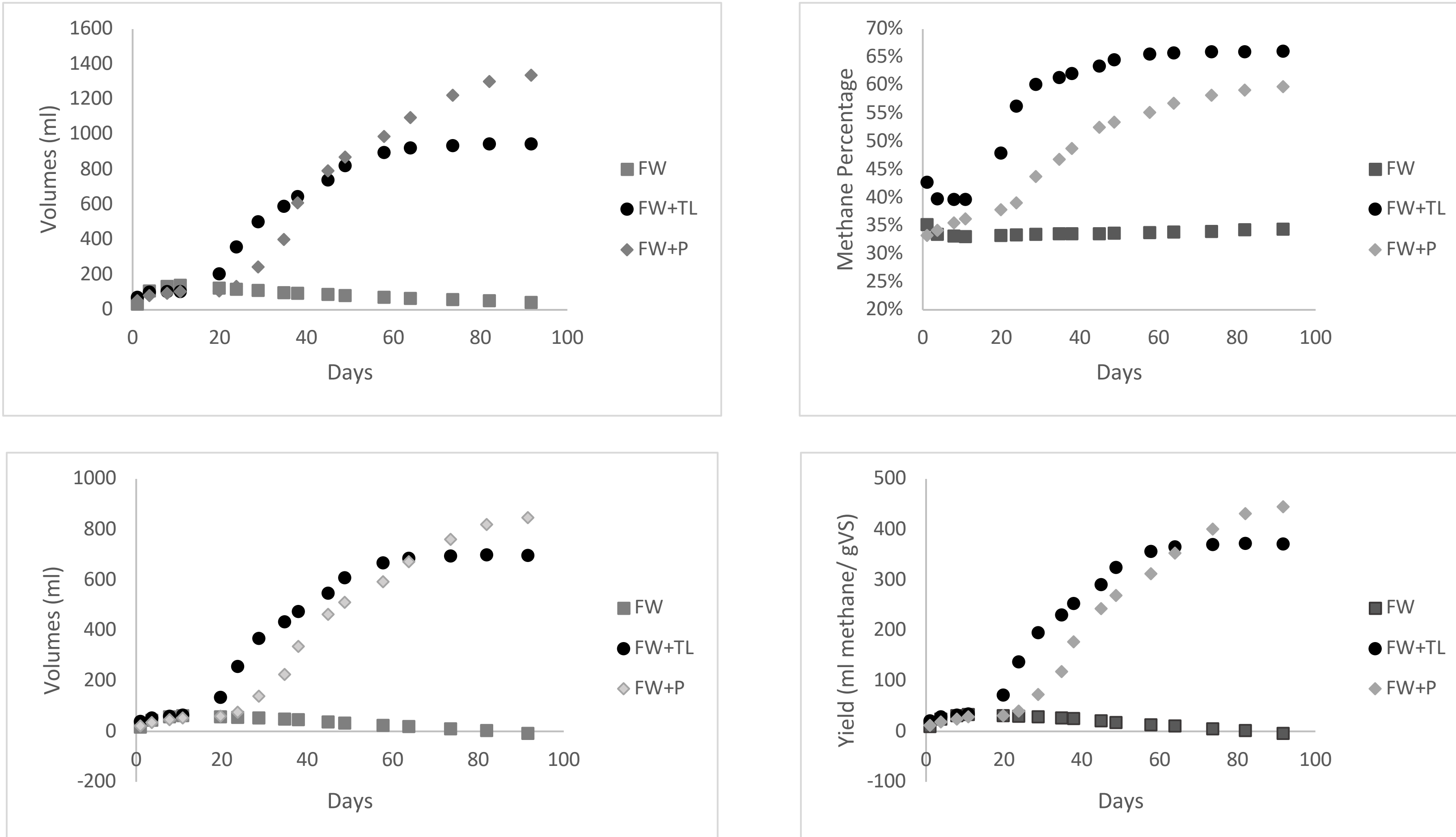

Figure 4.1 a) Cumulative biogas volumes over the digestion period for phase 1 of the BMP assay. B) Biogas quality over the digestion period for Phase 1. C) Cumulative methane volumes for Phase 1. D) Methane yield (normalized to g VS added into the bottles) for Phase 1

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Biogas quality also differed between substrates. FW biogas quality was the lowest, remaining at 34±0.1% methane throughout the entire digestion period. FW+TL had the highest biogas quality, stabilizing at 66±1% methane after 48 days. FW+P continued to see biogas quality rise throughout the 92 day digestion period, reaching 60±1% methane.

Methane volumes were calculated using the biogas quality and volume of the biogas. FW saw the lowest methane volumes, reaching a volume of -7 ml of methane produced throughout the 92 day digestion period. The volume is negative because the inoculum only control produced more methane than FW. The highest methane volume achieved by the FW reactors occurred on day 20, with a volume of 57 ml. FW+TL saw a maximum methane volume on day 82 at 698 ml. After this, methane volumes remained similar to inoculum methane volumes. FW+P saw a methane volume of 846 ml. It is worth noting that methane production for FW+P did not level out to reach volumes similar to the inoculum only control, and therefore methane production may be higher than the 846 ml that was reached on day 92.

Methane yield was calculated by dividing the methane volume by the initial substrate VS in each reactor. FW achieved the lowest methane yield at -3.9 ml CH_4/ g VS. This negative yield was due to the inoculum only control producing more methane than FW. FW+TL achieved a methane yield of 372 ml CH_4/ g VS. FW+P achieved a final methane yield of 445 ml CH_4/ g VS.

FW digesters operated significantly worse than any other digesters, including the inoculum control. This is similar to results reported in literature over long-term operation of FW Zhang et al. (2014) attribute process failures in FW AD to low C:N ratios, low alkalinity or to trace metal elements missing in FW that are present in other substrates. Zhang and Jahng (2012) operated a semi-continuous FW reactor with no added substrates and saw decreased methane productivity after 64 days of operation, with the system souring by day 85. The longer operational time before souring can be attributed to the semi-continuous conditions, which may remove accumulated VFAs from the reactor as opposed to the batch reactors used in this study where VFAs can only exit the system through methanogenic metabolism. One exception to this study is Zhang et al. (2007), who saw successful operation of batch AD of FW and achieved a final methane yield of 440 ml CH_4/ g VS. They observed that 80% of the methane was released within the first ten days of operation. The main difference in their study is they operated under thermophilic conditions.

Co-digestion of FW with TL or P lead to higher methane yields than FW alone. Zhang et al. (2014) recommend co-digestion as a method to increase methane yields by either providing more carbon sources, added alkalinity, or introducing trace elements. El-Mashad and Zhang (2010) and Zhang et al. (2013) both saw increased methane yields through co-digestion than through FW AD on its own.

FW+TL also led to higher methane yields than TL on their own. Goel et al. (2001) operated a two-phase semi-continuous AD and achieved a methane yield of 146 ml CH_4/ g VS, less than half of this study's methane yield of 372 ml CH_4/ g VS. Goel et al. also only saw an increase in methanogenic activity after adding calcium chloride and magnesium chloride. Goel et al did see higher biogas quality, at 73% methane. Comparing both studies still remains difficult since Goel et al. did not specify the exact types of tea leaves used in their study, as well as the fact that their tea leaves were dried to increase solids, whereas this study decreased the solids to match FW conditions.

FW+P had results similar to other studies. Yagi et al. (2009) witnessed a 55 day lag period in the AD of their PLA-derived compostable plates without any other added substrates. This is a longer lag period than this study, indicating that added FW may help in the hydrolysis of P. Yagi et al. (2009) did not see lag periods under thermophilic conditions most likely due to the higher temperature increasing the rate of hydrolysis. El-mashad et al. (2012) co-digested FW + sugarcane plates as well at thermophilic conditions and observed a methane yield of 350 ml CH_4/ g VS. They operated with double the volume of compostable plates by g VS than FW, whereas this study kept both solids volumes equal. Their lower methane yield despite having higher operating temperatures indicates that further research is required to determine the optimal volumes of FW to P to maximize methane yield. Another interesting conclusion drawn from their study is that adding P to FW decreases the methane yield compared to FW alone, which is the opposite result of what was seen in this thesis. Therefore, further research is required to compare both the effects of temperature and different co-digestion ratios to help determine the optimal loading conditions for both FW and P.

4.1.2 Chemical Properties

Chemical properties were measured on 7, 21 and 92 days into digestion. Results for VFA, COD, alkalinity, ammonia and pH are shown in Figures 4.2-4.5.

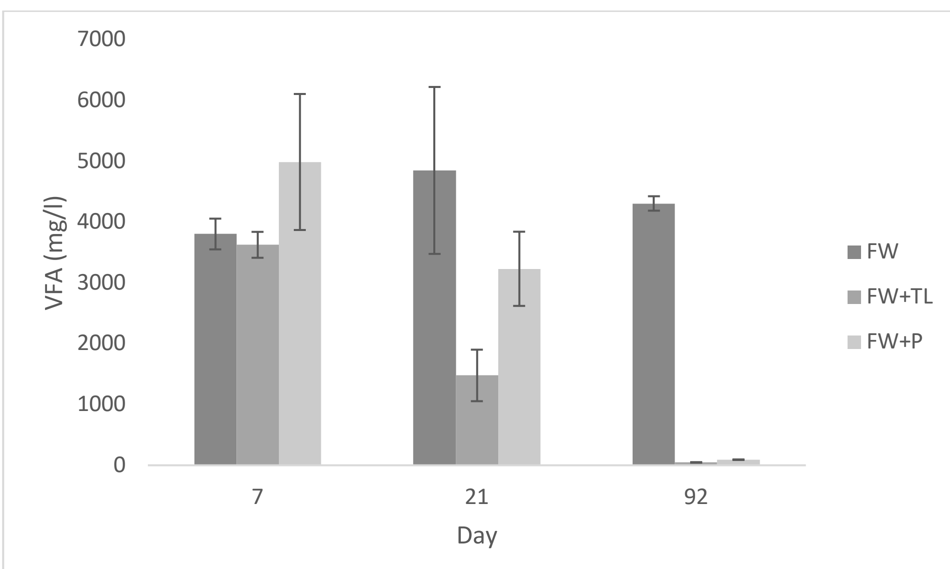

Figure 4.2 VFA levels for Phase 1 of the BMP assay

VFA concentrations decrease for every sample except FW, where there is an increase throughout the digestion period. This increase in VFAs for FW shows that while hydrolysis is metabolising FW to convert into VFAs, methanogens are not metabolising the VFAs to produce methane, corresponding to the minimal methane volume and yield seen in biogas analysis. On Day 7 and Day 21, all reactors have VFA levels higher than the recommended range of 50-300 mg/l (Schuyler 2013). High levels of VFA concentrations correspond to the long lag periods observed by FW+TL and FW+P reactors, which saw lag periods until Day 21 and 30 respectively. Both FW+TL and FW+P saw VFA concentrations significantly decrease by Day 92.

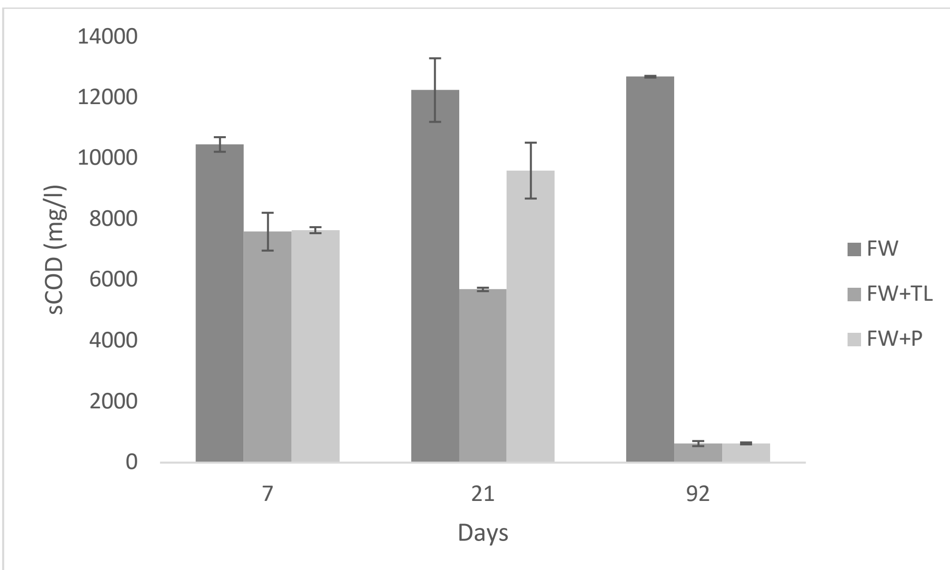

Figure 4.3 sCOD levels for Phase 1 of the BMP assay

sCOD observations also correspond to results seen during biogas analysis. sCOD values in FW increased throughout the entire digestion period. The increase in sCOD for FW shows that hydrolysis was not inhibited in this reactor, but that methanogens did not convert the sCODs to methane gas. sCOD values in the FW+P reactors increased from Day 7 to Day 21, indicating that methanogenesis was inhibited during this period. This corresponds to results seen during biogas analysis, where the lag period in the FW+P reactors lasted 30 days. The increase in sCOD from Day 7 to Day 21 in FW+P is comparable to the increase seen from FW. This indicates that although hydrolysis of food waste was occurring, there was minimal hydrolytic activity on P during the first 21 days of digestion. This is similar to results seen by Yagi et al. (2009) who saw minimal anaerobic degradation of PLA over the first 55 days of digestion. sCOD concentrations in the FW+TL reactors decreased throughout the entire digestion period. This corresponds to methane production seen during biogas analysis, where the lag period for FW+TL ended after 20 days.

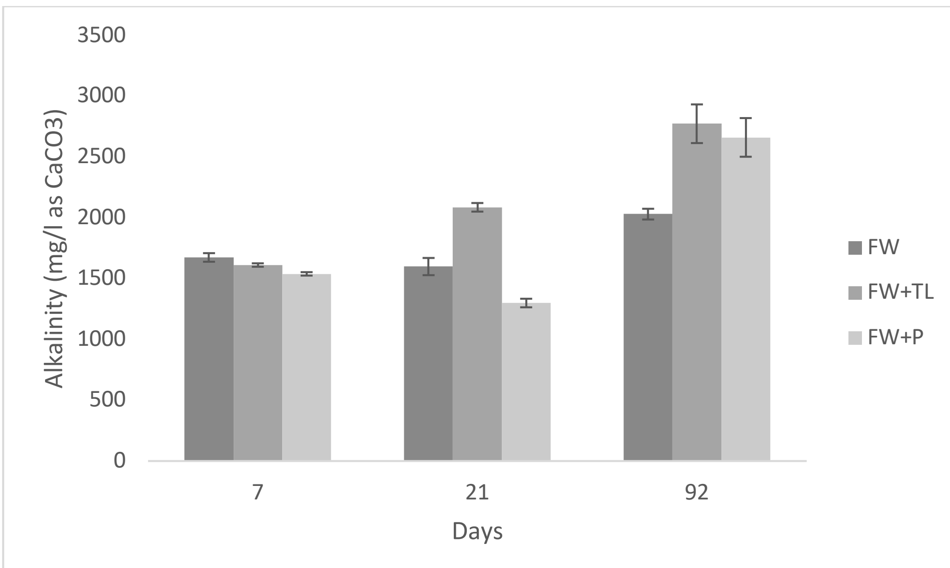

Figure 4.4 Alkalinity levels for Phase 1 of the BMP assay

Alkalinity values for FW and FW+P decreased from Day 7 to Day 21, but all other samples observed a steady increase in alkalinity throughout the entirety of the digestion period. The decrease in alkalinity for FW and FW+P from Day 7 to Day 21 indicates that hydrolysis is consuming alkalinity quicker than methanogenesis can generate alkalinity. This is similar to results observed during biogas analysis, in which FW and FW+P produced minimal methane by Day 21.

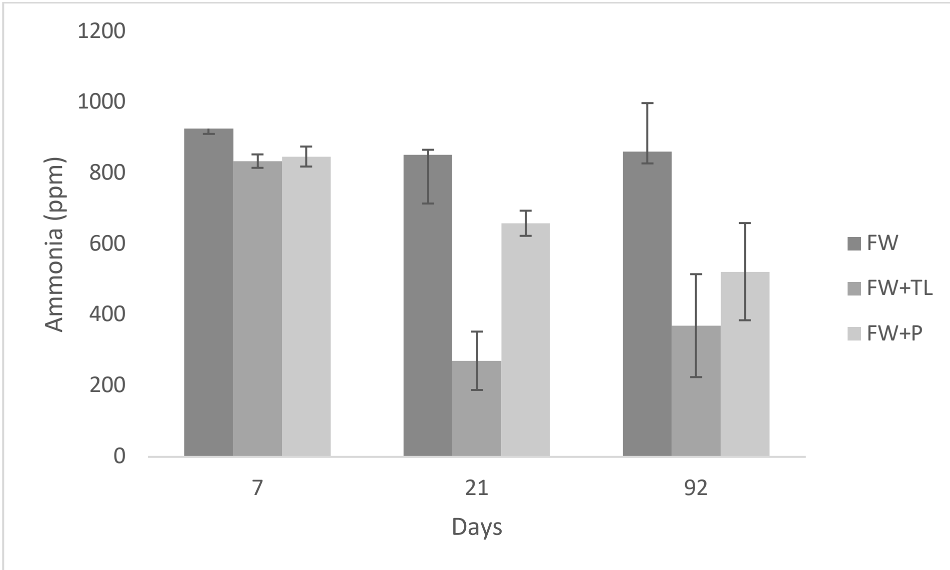

Figure 4.5 Ammonia concentrations for Phase 1 of the BMP assay

On Day 7 of digestion, ammonia levels remained relatively similar between all the digestion sets. Food waste ammonia levels remained constant throughout the entire digestion period. This indicates that the C:N ratio in the digester may be too low, and an additional carbon source is required to ensure healthy digestion. Ammonia is toxic to methanogens, and this higher concentration may indicate that methanogens were exposed to a toxic environment. In the TL digestion set, ammonia concentrations decreased between Day 7 and 21, and then slightly increased on Day 92. For the P digestion set, ammonia concentrations decrease throughout the digestion period.

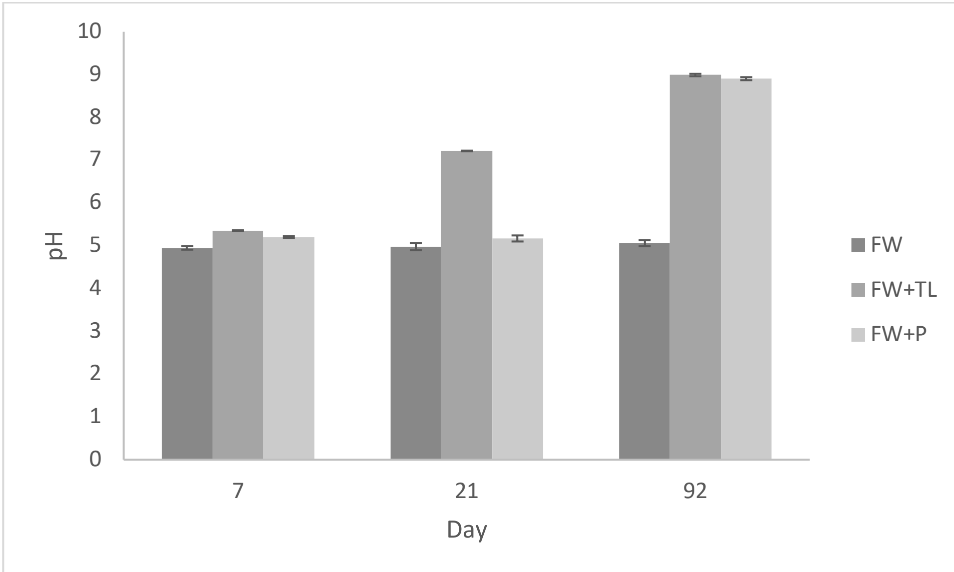

Figure 4.6 Changes in pH in Phase 1 of the BMP assays

As per Figure 4.5, pH on Day 7 was well below a neutral pH for all three digestion sets. This is similar to results seen in the methane production, where methane production remained relatively low throughout the beginning of digestion. On Day 21, FW+TL pH rose to 7.5, which is within the range of healthy pH to encourage methane production. This is also representative of methane productions, as FW+TL saw an increase in methane production after Day 21. By Day 92, FW+TL and FW+P pH rose considerably, whereas FW pH remained low. This also represents trends seen in methane production, where FW did not produce large volumes of methane and ended with a negative methane yield, whereas FW+TL and FW+P saw significantly higher methane yields. Other chemical parameters indicated the health of the digesters. Table 4.1 provides a summary of final digestion health indicators to determine the success of AD for each sample after the 92 day digestion period.

Table 4.1 Final digestion parameters for Phase 1 of the BMP assays

Sample	**VS Reduction (%)**	**Final pH**	**VFA:Alkalinity Ratio**	**Methane Yield (**ml CH_4/ g VS)
FW	40.7±0.1	5.06±0.07	2.12±0.10	-3.90
FW+TL	62.4±3.0	8.99±0.03	0.017±0.001	372
FW+P	60.6±2.5	8.91±0.04	0.035±0.003	445

FW+TL saw the greatest VS reduction from all samples. As noted earlier, methane production for FW+P did not level out to inoculum levels, resulting in potential for more VS reduction over a longer digestion period. While it is recommended to continue operation of BMP assays until methane production can level out, large-scale FW AD rarely operate at HRTs longer than 60 days (Zhang et al. 2014). Therefore, future work should focus on decreasing the 30 day lag period observed by the FW+P so that it may be more realistic to digest in larger applications.

Methanogens operate most effectively at pH of 6.5 to 7.2 (Zhang et al. 2014). FW saw souring of the reactor with the lower pH values, whereas both FW+TL and FW+P had pH values above the values recommended for methanogenesis. FW+TL and FW+P still saw strong methane production despite the higher pH. Future work should focus on looking whether pH control can increase methane production even more. pH values for FW+P was slightly higher than values observed by Yagi et al. (2009), who saw a pH of 8.3 in their thermophilic reactors. Benn and Zitomer (2018) observed a lower pH of 7.29, but their PLA was pretreated either through heating or through chemical pre-treatment using a basic solution.

The VFA:Alkalinity ratio remained below the maximum value of 0.35 for both FW+TL and FW+P (Schuyler 2013). FW was significantly above that value, which corresponds to the souring of the reactor. FW+TL and FW+P were both lower than the minimal value of 0.1 (Schuyler 2013), indicating potential to increase loading within these two reactors without overloading the system and souring the reactors.

4.2 Lifecycle Assessment

Inputs from the materials and methods were inputted into WARM. Appendix B shows model inputs that helped determine the results. Results are shown in Table 4.2

Table 4.2 WARM Outputs for USF Food Waste Treatment

Parameter	**Incineration**	**AD**	**Difference**
GHG Emissions (mton CO_2 equivalents)	-146	-154	-8
Energy Usage (kWh)	-734,000	-457,000	277,000

Anaerobic digestion resulted in a decrease in GHG emissions compared to incineration. However, AD results in a decrease in energy production compared to incineration. Lee et al. (2020) compared AD with incineration using Hillsborough County municipal solid waste values. Lee et al. (2020) observed that AD resulted in a decrease of 1000 kg CO_2 equivalents during AD as opposed to a decrease of only 400 kg CO_2 equivalents during combustion. Lee et al. (2020) provided a more comprehensive LCA using SimaPro to conduct a full-scale study from construction, collection, transportation, emissions and avoided products, whereas this study focuses on FW disposal at different sources. A more comprehensive LCA using an LCA software such as SimaPro is recommended to verify results seen in this thesis.

Chapter 5. Conclusion and Recommendations

Food waste is a considerable challenge to tackle in North America due to the large scale of the problem. The US on its own wastes 63 million tons of food each year. This large volume of waste most often gets landfilled, where the large organic fraction of food waste decomposes and releases as greenhouse gases into the atmosphere. Food waste is the main source of methane emissions from landfills, which already account for 16% of all methane emissions in the country. Food waste is also generated at every point in the food production line, from farms not harvesting food to restaurants generating large values of pre-consumer waste.

Universities are currently trying to tackle the large volumes of food waste produced at their dining halls and establishments by incorporating recommendations based on the US EPA Food Recovery Hierarchy recommendations to combat their food waste. At the University of South Florida, The Campus Food Recovery Program has looked to reduce food waste on campus through a combination of source reduction, food donation and determining the feasibility of large-scale AD of food waste. They have also been looking to find community partners who are committed and willing to donate excess food waste and determine if those extra donations can result in system stability for AD. One of the main supporters of this project is TBD Café @ 301, a wholesale tea supplier operating out of Riverview.

This thesis specifically focuses on the feasibility of large-scale AD of food waste at USF. This feasibility study was done simultaneously with a pilot study conducted through the Campus Food Recovery Program. The specific goals of this research were to 1. Look at the co-digestion of food waste with tea leaves and compostable plates. 2. Conduct a lifecycle assessment of implementing large scale AD at USF.

1. Analyze the co-digestion of food waste with tea leaves and compostable plates

Three phases of batch BMP studies were conducted to look at the biodegradability of FW, either on its own or through co-digestion with tea leaves and compostable plates. Phase 1 looked at the initial biodegradability of the substrates. While food waste digesters soured and inhibited methanogenesis, both the tea leaves and compostable plate co-digested reactors showed great methane yield, at 372 and 447 ml CH_4/ g VS respectively. The food waste digesters showed both VFA accumulation and an incredibly high VFA:Alkalinity ratio.

2. Conduct a lifecycle assessment of large-scale anaerobic digestion

A lifecycle assessment was conducted using the US EPA WARM model for solid waste disposal. The model focused on different end-of-life options including incineration, composting, and anaerobic digestion. A base case of USF's current practice of sending food waste for incineration was compared to sending all of USF dining hall waste to on-site AD. Switching to AD showed a decrease in GHG emissions, but the decrease was minimal at 8.23 tons of CO_2 equivalents. AD also resulted in increased energy usage because incineration can recover more energy than anaerobic digestion, with an increase of 1558 million and 1670 million BTU respectively compared to the base case.

Following this research, there still remains specific research gaps that must be addressed before full-scale AD can be implemented at USF. Recommendations for future testing includes

- Investigate different F:M ratios of the co-digestion of FW+ TL and FW+P to ensure maximum biodegradability

- Investigate AD of FW+TL+P in a semi-continuous reactor to determine the appropriate OLR and biodegradability
- Conduct a full LCA that includes materials and construction, usage, transportation, and end of life based on ISO 14001 for building an on-campus AD to verify data outputted from the WARM model
- Scale up the LCA to account for additional streams of food waste including tea leaves, compostable plates, and surrounding industries

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Appendix A: Acronyms

AD- Anaerobic Digestion
AETP- Aquatic Ecotoxicity Potential
AP- Acidity Potential
C:N- Carbon to Nitrogen
CHP- Combined Heat and Power
CPI- Consumer Price Index
CSTR- Continuous Stirred Tank Reactor
EP- Eutrophication Potential
F:M- Food to Microorganism
FOGs- Fats, Oils and Grease
FW- Food Waste
GWP- Global Warming Potential
HRT- Hydraulic Retention Time
HTP- Hydro Toxicity Potential
IRR- Internal Rate of Return
LCA- Life Cycle Assessment
MO- Microorganism
MSW- Municipal Solid Waste
ODP- Ozone Depletion Potential
OFMSW- Organic Fraction of Municipal Solid Waste
OLR- Organic Loading Rate
P- PLA plates
PLA- polylactic acid
PFR- Plug Flow Reactor
POCP- Photochemical Ozone Creation Potential
RCRA- Resource Conservation and Recovery Act
sCOD- Soluble Chemical Oxygen Demand
SGEF- Student Green Energy Fund
SNAPs- Supplemental Nutritional Assistance Program
SRT- Solids Retention Time
T- Tea leaves
TRACI- Tool for the Reduction of Assessment of Chemical and other environmental Impacts
TS- Total Solids
TTP- Terra Toxicity Potential
UC Davis- University of California Davis
US EPA- United States Environmental Protection Agency
USF- University of South Florida
V- Volume

VFA- Volatile Fatty Acids
VS- Volatile Solids
WARM- Waste Reduction Model
WtE- Waste to Energy
WWTP- wastewater treatment plants

Appendix B: Pilot Study

Appendix B1. Materials and Methods

Six pilot-scale anaerobic digesters were operated at ambient temperature beginning in September 2019. The digesters were purchased through Solar Cities (Tampa, FL) and operation was based off Solar Cities' recommendations. The digesters are each 1 m^3 in size and operate as a semi-batch system. A schematic of the digesters is shown in Figure A1, and a detailed process and instrumentation diagram (P&ID) outlining the digesters is shown in Figure A2. Light green corresponds to the inlet pipe, blue corresponds to the gas outlet, and dark green corresponds to the effluent tank.

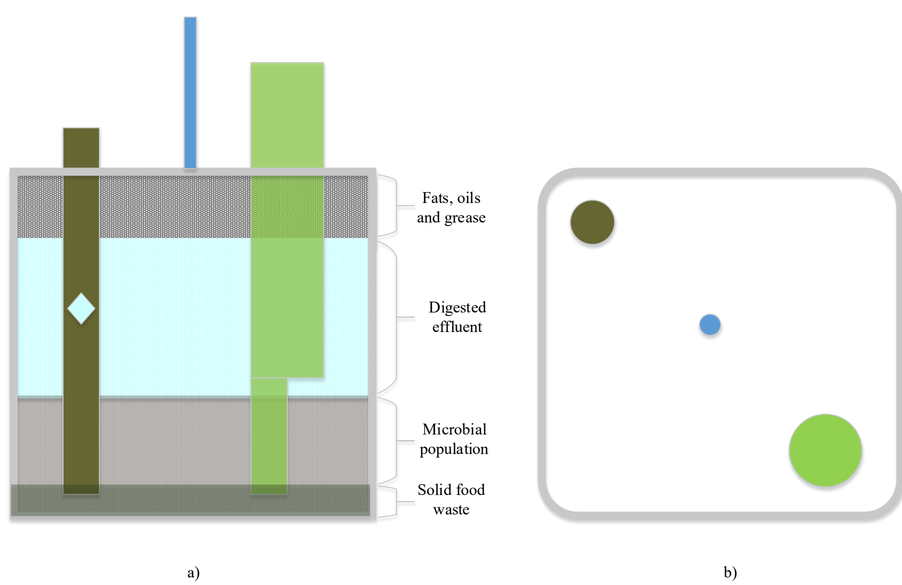


Figure A 1 a) side view of the anaerobic digester b) birds eye view of the anaerobic digester.

Figure A1 a) shows the side view of the digester, include how the different components of the digester settle into the tank. Food waste, being the densest, settles to the bottom of the tank once it is fed in through the influent pipe (light green). An L-shaped hole is cut into the influent pipe to ensure that food waste does not clog at the bottom of the pipe. Food waste travels from one corner of the digester towards the effluent pipe, which is diagonal to the influent (Figure A1 b). During this time, the microbial population breaks the food waste down, and digested effluent floats to the middle of the tank. Any fats, oils, or grease rise to the top of the digester due to their lower density. The effluent pipe has a hole cut in the middle of the pipe to ensure only the most nutrient-rich effluent leaves the tank.

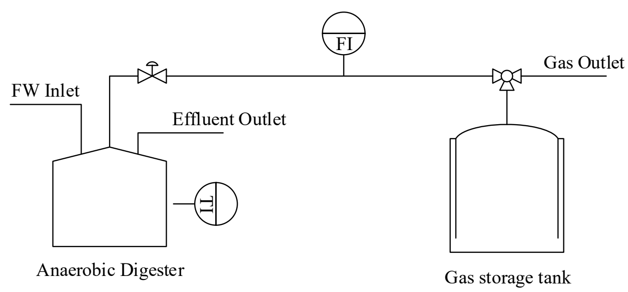


Figure A 2 P&ID of the Solar Cites IBC Tank Biogas digester system

Figure A2 shows a detailed process and instrumentation diagram (P&ID) of the Solar Cities IBC tank, including the gas storage tank. Gas leaves through a gas outlet pipe into the gas storage tank, which either stores the gas in the tank, or allows for another outlet so that the biogas can be tested or further treated down the line.

The digesters were initially inoculated using a mixture of cow manure and inoculum obtained from Rosebud Continuum in July 2019 and given two months to allow for the methanogens in the manure to grow before loading with food waste. Rosebud Continuum is a sustainable farm and education center located in Land O' Lakes, Florida. The inoculum obtained from Rosebud is also from food waste digesters that are fed food waste from a family's home and were considered a strong source of acclimated methanogens. The digesters were considered ready for food waste once the digesters began producing methane, indicating a large enough population of methanogens, which was first seen in August 2019. Food waste was obtained from Champions Choice, a dining hall at the University of South Florida that is operated by Aramark Dining Services. This location was chosen as a pilot location to begin testing collection of pre-consumer food waste at dining halls on recommendation from Aramark Dining Services and due to the strong relationship, the project team had with Aramark. Food waste was collected roughly three times a week (Monday, Wednesday and Fridays) in two 5 gallon buckets. Buckets were then weighed to determine the weight of food waste per bucket, pictures taken of the contents, and then ground using the Insinkerator Evolution Compact Garbage Disposal, ¾ HP (St. Louis, MI) with water, doubling the volume in the buckets. The ground food waste is then put into two digesters per pickup day, with each digester getting a total of two buckets of ground food a week. The hydraulic retention time in the digesters was 184 days, as per the loading suggestion of Solar Cities. Due to the high water content in the ground food (2.5% TS) and long retention times, no alkalinity sources were added to the digesters. Food waste composition varied due to the menus offered by Champions Choice, but mostly contained fruit peels from pineapples or melons, beans, strawberry pits or wilted lettuce. Certain pre-consumer food waste, such as raw meat, bones or pineapple heads, were excluded from this study due to limitations from the Insinkerator. The organic loading rate varied based on the substrate received per week but remained at roughly 0.13 kg VS/m^3/day over the period of operation.

Effluent samples were collected weekly from each digesters to help monitor the health of the digesters. On sampling days, ground food waste samples were also collected. Full chemical analysis was done on each sample.

Appendix B2: Pilot-Scale Digester

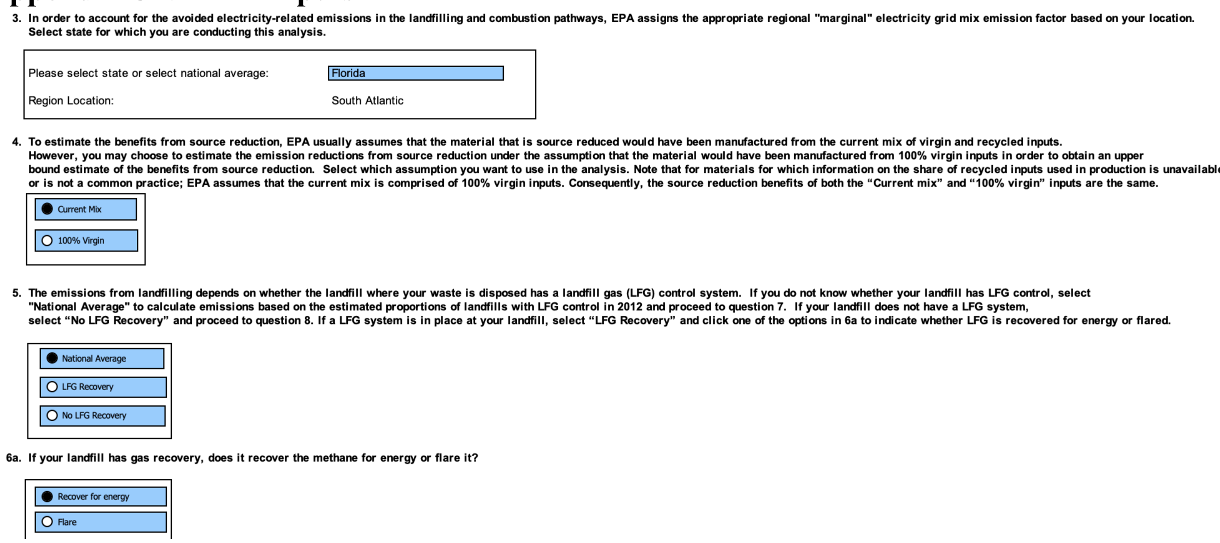
Pilot scale digesters ran from September 2019 to December 2019, and again starting January 2020 after a break due to the holiday season. During this time, loading remained inconsistent due to a multitude of issues such as mechanical issues from food scale grinders, university holidays, and training periods required at the beginning of each semester with all entities involved. Through conversations and lessons learned during this period, it was determined that the most reasonable method of achieving full-scale digestion would be through incorporating FW pre-treatment through mechanical grinding in dining halls, and to transport broken down FW to the digesters through routes that USF Facilities' recycling team already takes to pick up the college's recycling rather than setting up new routes.

Pilot scale digesters operated at ambient temperatures were tracked throughout the operational period for TS, VS, sCOD, VFAs and alkalinity. Biogas production was also tracked in January 2020, however, data appears inconclusive due to biogas leaks in the digester.

The difficulty in operation and maintenance of low-technology AD remains the largest barrier in scaling up this technology for large-scale AD. In January 2020, maintenance was done weekly to repair insulation, seal leaks, and unclog digester inlet pipes. A more autonomous system with better controls and minimal maintenance is recommended for scale up to provide more accurate data.

Appendix C: WARM Model Screenshots

Appendix C1: Main Inputs



3. In order to account for the avoided electricity-related emissions in the landfilling and combustion pathways, EPA assigns the appropriate regional "marginal" electricity grid mix emission factor based on your location. Select state for which you are conducting this analysis.

Please select state or select national average:	Florida
Region Location:	South Atlantic

4. To estimate the benefits from source reduction, EPA usually assumes that the material that is source reduced would have been manufactured from the current mix of virgin and recycled inputs. However, you may choose to estimate the emission reductions from source reduction under the assumption that the material would have been manufactured from 100% virgin inputs in order to obtain an upper bound estimate of the benefits from source reduction. Select which assumption you want to use in the analysis. Note that for materials for which information on the share of recycled inputs used in production is unavailable or is not a common practice; EPA assumes that the current mix is comprised of 100% virgin inputs. Consequently, the source reduction benefits of both the "Current mix" and "100% virgin" inputs are the same.

- ● Current Mix
- ○ 100% Virgin

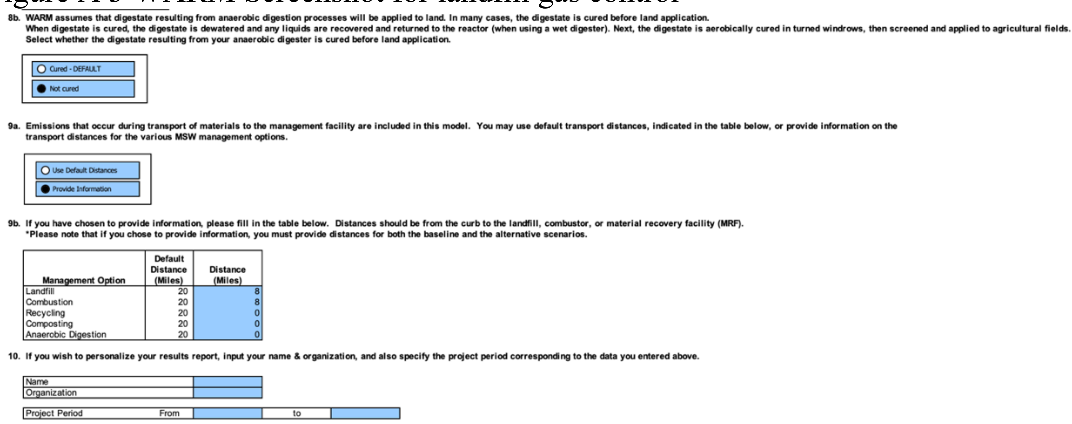
5. The emissions from landfilling depends on whether the landfill where your waste is disposed has a landfill gas (LFG) control system. If you do not know whether your landfill has LFG control, select "National Average" to calculate emissions based on the estimated proportions of landfills with LFG control in 2012 and proceed to question 7. If your landfill does not have a LFG system, select "No LFG Recovery" and proceed to question 8. If a LFG system is in place at your landfill, select "LFG Recovery" and click one of the options in 6a to indicate whether LFG is recovered for energy or flared.

- ● National Average
- ○ LFG Recovery
- ○ No LFG Recovery

6a. If your landfill has gas recovery, does it recover the methane for energy or flare it?

- ● Recover for energy
- ○ Flare

Figure A 3 WARM Screenshot for landfill gas control



8b. WARM assumes that digestate resulting from anaerobic digestion processes will be applied to land. In many cases, the digestate is cured before land application. When digestate is cured, the digestate is dewatered and any liquids are recovered and returned to the reactor (when using a wet digester). Next, the digestate is aerobically cured in turned windrows, then screened and applied to agricultural fields. Select whether the digestate resulting from your anaerobic digester is cured before land application.

- ○ Cured - DEFAULT
- ● Not cured

9a. Emissions that occur during transport of materials to the management facility are included in this model. You may use default transport distances, indicated in the table below, or provide information on the transport distances for the various MSW management options.

- ○ Use Default Distances
- ● Provide Information

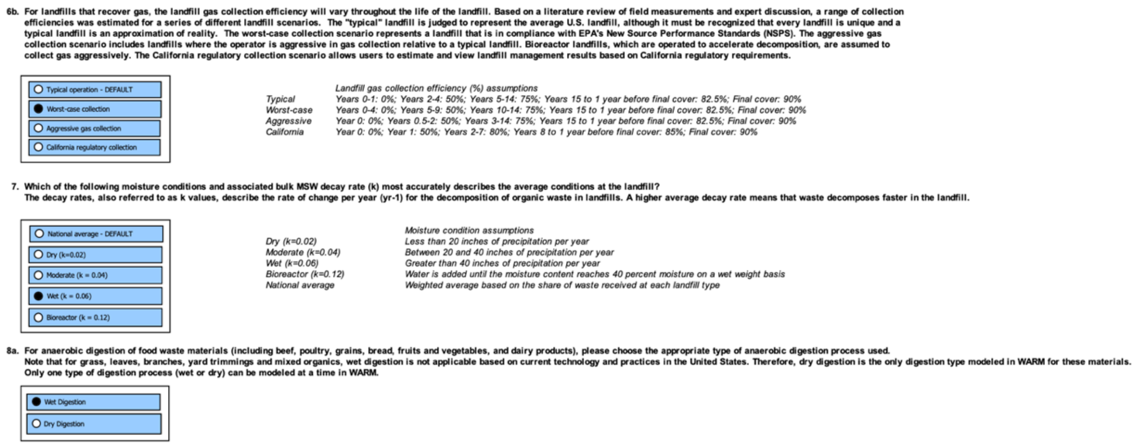
9b. If you have chosen to provide information, please fill in the table below. Distances should be from the curb to the landfill, combustor, or material recovery facility (MRF).
*Please note that if you chose to provide information, you must provide distances for both the baseline and the alternative scenarios.

Management Option	Default Distance (Miles)	Distance (Miles)
Landfill	20	8
Combustion	20	8
Recycling	20	0
Composting	20	0
Anaerobic Digestion	20	0

10. If you wish to personalize your results report, input your name & organization, and also specify the project period corresponding to the data you entered above.

Name			
Organization			
Project Period From		to	

Figure A 4 WARM Screenshot for distance inputs



6b. For landfills that recover gas, the landfill gas collection efficiency will vary throughout the life of the landfill. Based on a literature review of field measurements and expert discussion, a range of collection efficiencies was estimated for a series of different landfill scenarios. The "typical" landfill is judged to represent the average U.S. landfill, although it must be recognized that every landfill is unique and a typical landfill is an approximation of reality. The worst-case collection scenario represents a landfill that is in compliance with EPA's New Source Performance Standards (NSPS). The aggressive gas collection scenario includes landfills where the operator is aggressive in gas collection relative to a typical landfill. Bioreactor landfills, which are operated to accelerate decomposition, are assumed to collect gas aggressively. The California regulatory collection scenario allows users to estimate and view landfill management results based on California regulatory requirements.

- ○ Typical operation - DEFAULT
- ● Worst-case collection
- ○ Aggressive gas collection
- ○ California regulatory collection

	Landfill gas collection efficiency (%) assumptions
Typical	Years 0-1: 0%; Years 2-4: 50%; Years 5-14: 75%; Years 15 to 1 year before final cover: 82.5%; Final cover: 90%
Worst-case	Years 0-4: 0%; Years 5-9: 50%; Years 10-14: 75%; Years 15 to 1 year before final cover: 82.5%; Final cover: 90%
Aggressive	Year 0: 0%; Years 0.5-2: 50%; Years 3-14: 75%; Years 15 to 1 year before final cover: 82.5%; Final cover: 90%
California	Year 0: 0%; Year 1: 50%; Years 2-7: 80%; Years 8 to 1 year before final cover: 85%; Final cover: 90%

7. Which of the following moisture conditions and associated bulk MSW decay rate (k) most accurately describes the average conditions at the landfill?
The decay rates, also referred to as k values, describe the rate of change per year (yr-1) for the decomposition of organic waste in landfills. A higher average decay rate means that waste decomposes faster in the landfill.

- ○ National average - DEFAULT
- ○ Dry (k=0.02)
- ○ Moderate (k = 0.04)
- ● Wet (k = 0.06)
- ○ Bioreactor (k = 0.12)

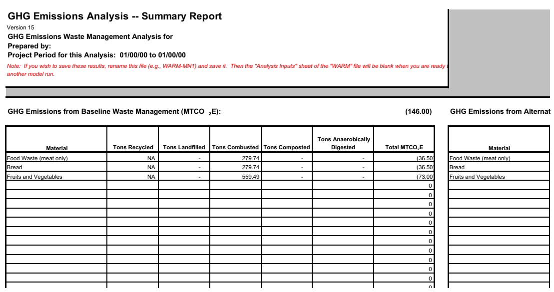
	Moisture condition assumptions
Dry (k=0.02)	Less than 20 inches of precipitation per year
Moderate (k=0.04)	Between 20 and 40 inches of precipitation per year
Wet (k=0.06)	Greater than 40 inches of precipitation per year
Bioreactor (k=0.12)	Water is added until the moisture content reaches 40 percent moisture on a wet weight basis
National average	Weighted average based on the share of waste received at each landfill type

8a. For anaerobic digestion of food waste materials (including beef, poultry, grains, bread, fruits and vegetables, and dairy products), please choose the appropriate type of anaerobic digestion process used.
Note that for grass, leaves, branches, yard trimmings and mixed organics, wet digestion is not applicable based on current technology and practices in the United States. Therefore, dry digestion is the only digestion type modeled in WARM for these materials.
Only one type of digestion process (wet or dry) can be modeled at a time in WARM.

- ● Wet Digestion
- ○ Dry Digestion

Figure A 5 WARM screenshot for anaerobic digestion inputs

Appendix C2: Main Outputs



GHG Emissions Analysis -- Summary Report
Version 15
GHG Emissions Waste Management Analysis for
Prepared by:
Project Period for this Analysis: 01/00/00 to 01/00/00
Note: If you wish to save these results, rename this file (e.g., WARM-MN1) and save it. Then the "Analysis Inputs" sheet of the "WARM" file will be blank when you are ready
another model run.

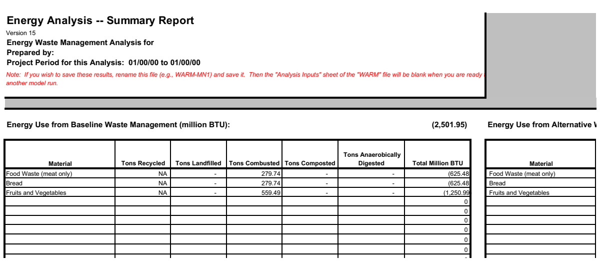
GHG Emissions from Baseline Waste Management ($MTCO_2E$): (146.00)

Material	Tons Recycled	Tons Landfilled	Tons Combusted	Tons Composted	Tons Anaerobically Digested	Total $MTCO_2E$
Food Waste (meat only)	NA	-	279.74	-	-	(36.50
Bread	NA	-	279.74	-	-	(36.50
Fruits and Vegetables	NA	-	559.49	-	-	(73.00
						0
						0
						0
						0
						0
						0
						0
						0
						0
						0
						0

GHG Emissions from Alternat

Material
Food Waste (meat only)
Bread
Fruits and Vegetables

Figure A 6 GHG emissions output from WARM



Energy Analysis -- Summary Report
Version 15
Energy Waste Management Analysis for
Prepared by:
Project Period for this Analysis: 01/00/00 to 01/00/00
Note: If you wish to save these results, rename this file (e.g., WARM-MN1) and save it. Then the "Analysis Inputs" sheet of the "WARM" file will be blank when you are ready
another model run.

Energy Use from Baseline Waste Management (million BTU): (2,501.95)

Material	Tons Recycled	Tons Landfilled	Tons Combusted	Tons Composted	Tons Anaerobically Digested	Total Million BTU
Food Waste (meat only)	NA	-	279.74	-	-	(625.48
Bread	NA	-	279.74	-	-	(625.48
Fruits and Vegetables	NA	-	559.49	-	-	(1,250.99
						0
						0
						0
						0
						0
						0

Energy Use from Alternative V

Material
Food Waste (meat only)
Bread
Fruits and Vegetables

Figure A 7 energy usage output from WARM

Appendix D: BMP Compositions

Table A 1 Bottle Compositions for Phase 1

Mix	Bottle Size	FWTS	Substate2TS	Food1V	Sub VS	Microbe VS	MicrobeV	F:M Ratio	Total V	Headspace
FW only	250	22.783	N/A	80	0	18.8076	100	0.97	180	70
FW+TL	250	22.783	22.783	40	40	18.8076	100	0.97	180	70
FW+P	250	22.783	22.783	40	40	18.8076	100	0.97	180	70
Inoculum	250					18.8076	100	0	100	150

Table A 2 Bottle Compositions for Phase 2

Mix	Bottle Size	FW VS	Alk (mg/l)	Food1V	Food2V	MO VS	MO V	F:M Ratio	Total V	Headspace
FW only	250	2.3		80		1.86	75	1.32	155	95
FW+alk	250	2.3	5000	80	0	1.88	75	1.32	155	95
FW 0.5	250	2.3		40	0	1.88	75	0.65	115	135
Inoculum	250					1.88	75	0	75	175

Table A 3 Bottle Compositions for Phase 3

Mix	Bottle Size	FW VS	TL VS	P VS	FWV	TLV	PVS	MO VS	Mo V	F:M Ratio	Total V	Headspace
FW:TL:P	250	2.3	2.3	2.3	20	20	20	1.56	90	0.98263079	150	100
FW:TL:P 2	250	2.3	2.3	2.3	30	15	15	1.56	90	0.98263079	150	100
Inoculum	250	2.3	2.3	2.3	0	0	0	1.56	90		90	160

Appendix E: Phase 2 and 3 of BMP Assay

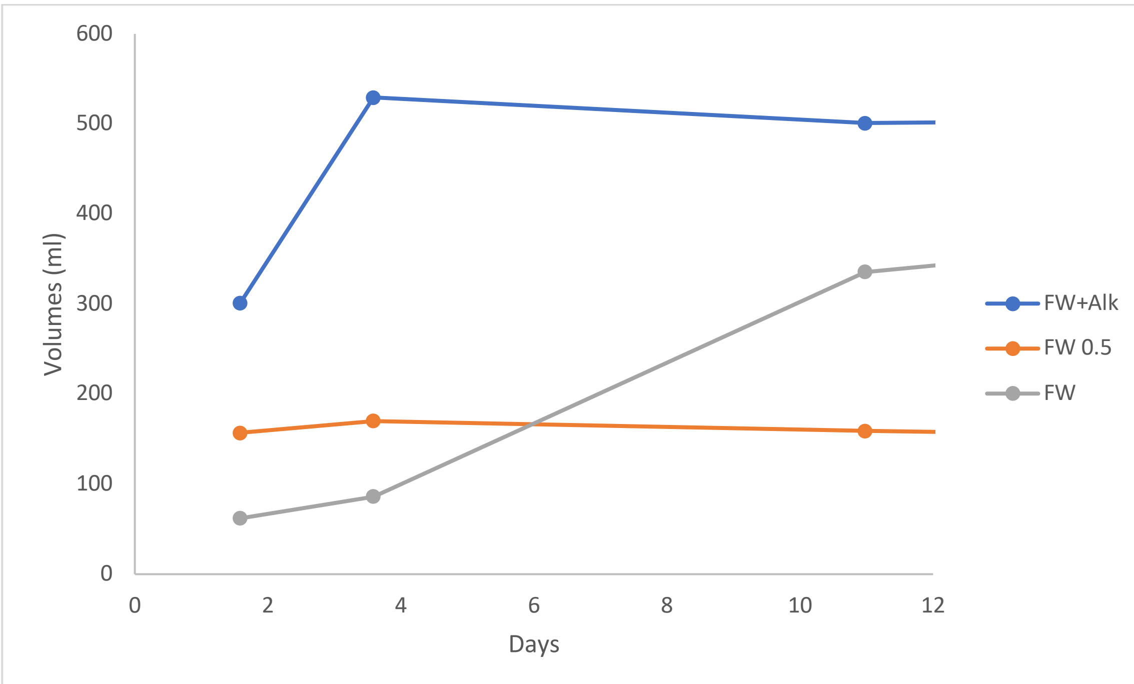


Figure A8 Biogas volumes for Phase 2 of the BMP assay

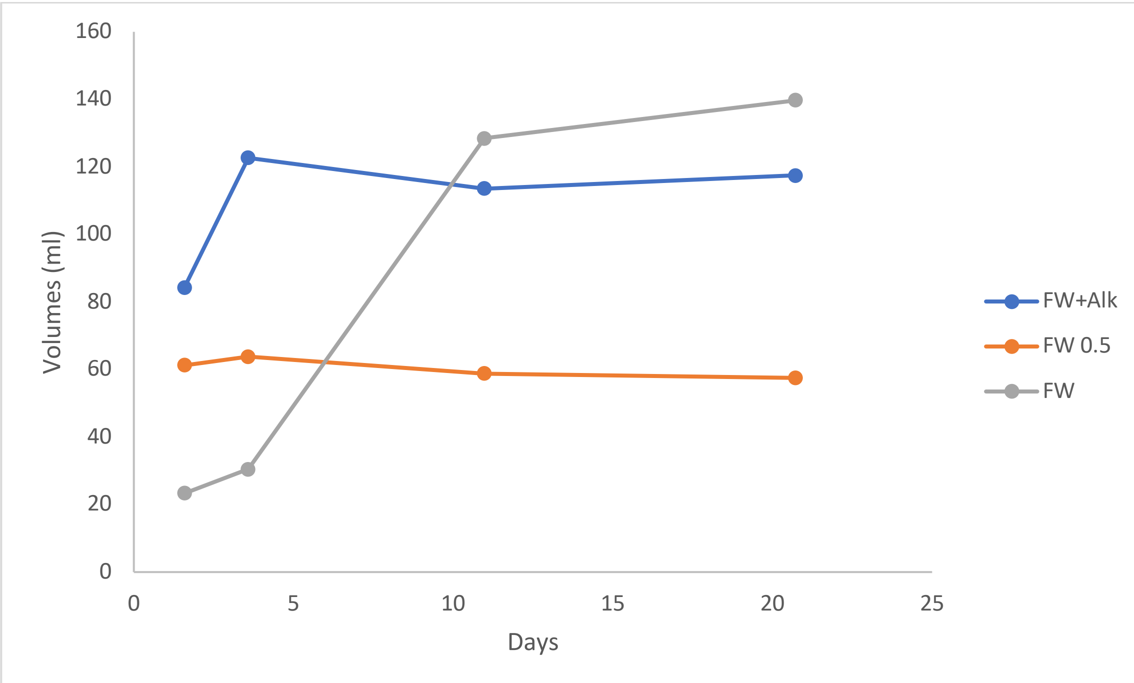


Figure A9 Methane volumes for Phase 2 of the BMP assay

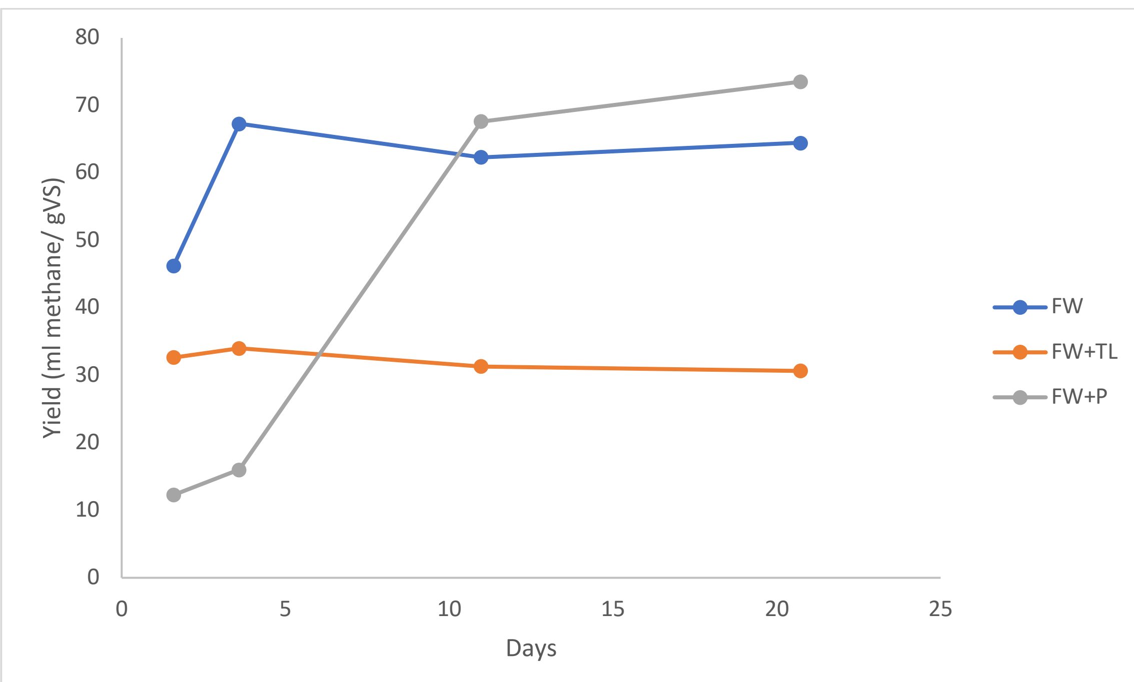


Figure A10 Methane yields for Phase 2 of the BMP assay

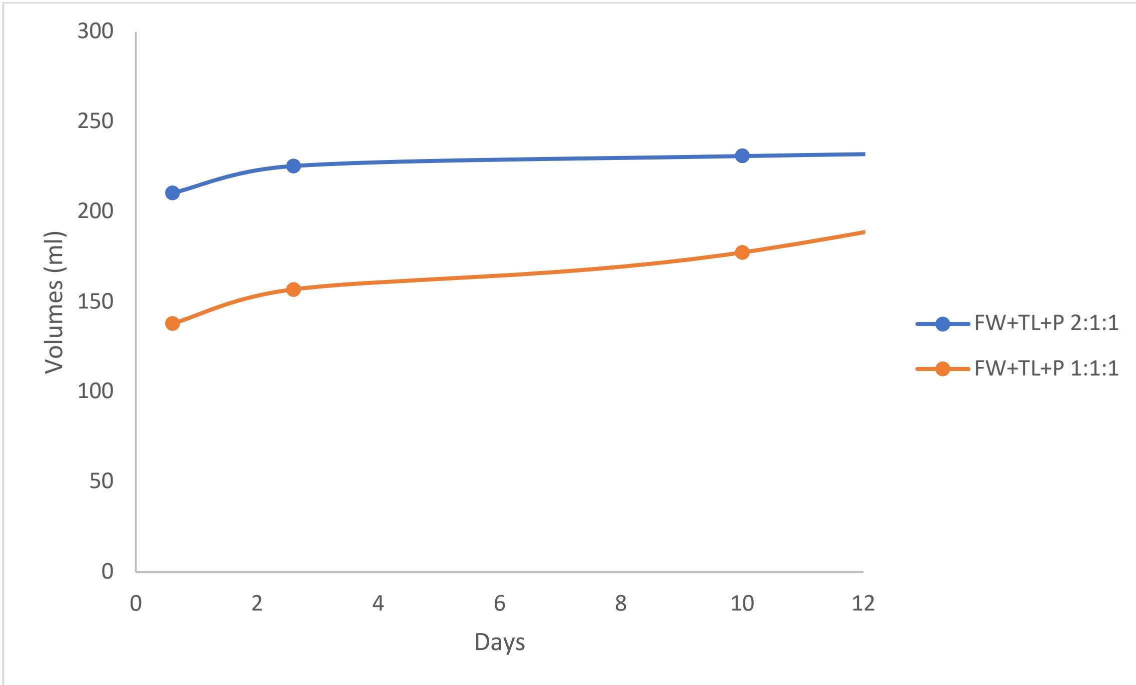


Figure A11 Biogas Volumes for Phase 3 of the BMP Assay

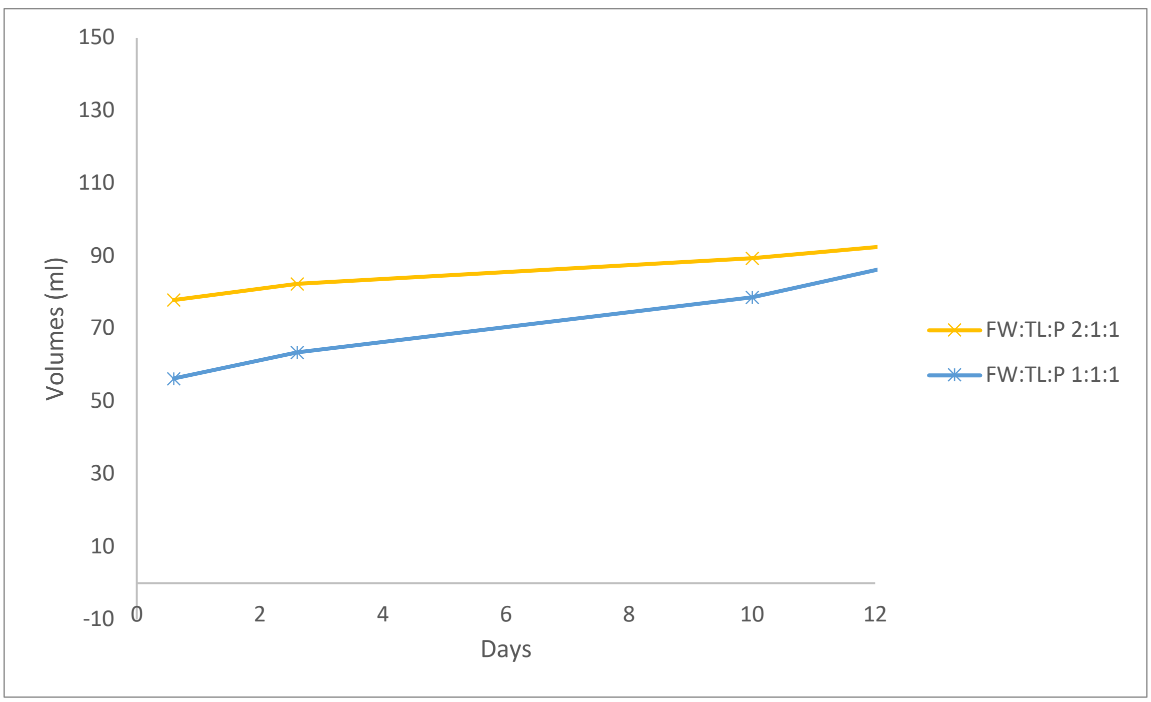


Figure A12 Methane volumes for Phase 3 of the BMP assay

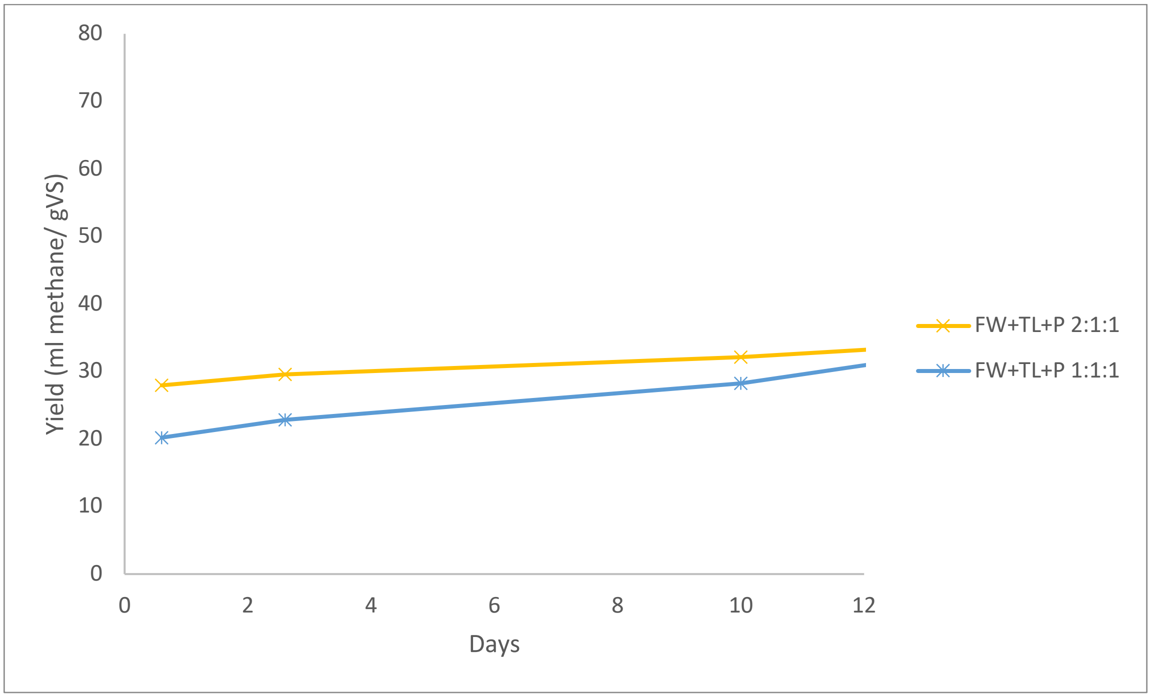


Figure A13 Methane Yields for Phase 3 of the BMP assay