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High-Solids Anaerobic Digestion of the Organic Fraction of Municipal Solid Waste State of the Art, Outlook in Florida, and Enhancing Methane Yields from Lignocellulosic Wastes

Gregory Richard Hinds
University of South Florida, ghinds@mail.usf.edu

High-Solids Anaerobic Digestion of the Organic Fraction of Municipal Solid Waste

State of the Art, Outlook in Florida, and Enhancing Methane Yields from Lignocellulosic Wastes

by

Gregory Richard Hinds

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Department of Civil and Environmental Engineering
College of Engineering
University of South Florida

Major Professor: Sarina Ergas, Ph.D.
Daniel Yeh, Ph.D.
Qiong Zhang, Ph.D.

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DEDICATION

To all of the people who have inspired and encouraged me along the path I've taken thus far in life, I dedicate this work to you. My family – my sisters Kelsey and Trisha Hinds, my mother Kelli Prichard, my father Rich Hinds, and of course all of my aunts, uncles, and cousins – you have been the foundation upon which I have built myself and I am forever grateful for your love and support. My friends – your encouragement and belief in me has always helped me to believe in myself. My teachers – you have dedicated your lives to teaching and inspiring the generations of the future and you have played a pivotal role in my development. My classmates and colleagues – you made this experience an enjoyable one filled with memories, you made the tough times bearable, pushed me when I needed to be pushed, and taught me a great deal. Lastly, all of my ancestors – the unfathomable amount of effort that all of those that came before me have put forth has provided me with the opportunity to live this incredibly fortunate life.

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ABSTRACT

Anaerobic digestion (AD) is a biotechnology that employs natural microbial metabolism under oxygen-free conditions to stabilize organic waste. AD has been shown to be the most environmentally sustainable technology for treating the organic fraction of municipal solid waste (OFMSW), as it allows for the recovery of energy and nutrients from the waste. AD of OFMSW also saves landfill space and reduces leachate generation and fugitive methane emissions from landfills. High-solids AD (HS-AD) technologies (those designed to process feedstocks with >15% total solids content) have been shown to yield additional benefits when compared with liquid AD (L-AD) for treating OFMSW, including reduced parasitic energy demands, reactor volume requirements, water usage, and excess leachate generation. These factors paired with increasingly stringent environmentally-driven legislation have resulted in the steady development of HS-AD technologies in Europe since the 1990's and the recent advancement of HS-AD in the United States. However, HS-AD implementation in the US is hindered by the low cost of landfilling and a general lack of regulatory drivers encouraging organics separation and recycling. The goal of this research was to contribute to accelerating the implementation and improving the efficiency of HS-AD technologies. The specific objectives were to: (i) assess the state of the art of HS-AD in Europe and the US and investigate trends in development; (ii) conduct a case study assessment of the outlook for implementation of HS-AD in the state of Florida; and (iii) investigate the potential to enhance methane (CH_4) yields in HS-AD of lignocellulosic wastes through bioaugmentation with pulp and paper mill anaerobic sludge.

Information sources for the assessment of the state of HS-AD in Europe and the US included "grey" and published literature and discussions with consultants and technology vendors. In Europe as of 2014 there were 244 full-scale AD facilities for processing OFMSW with a total capacity of almost 8 million tons per year (TPY), approximately 89% of capacity was "stand-alone" (systems treating *only* OFMSW), 62% was HS-AD, and 70% installed since 2009 was HS-AD. In the US, as many as 181 AD facilities are now processing OFMSW with an approximate total capacity of 780,000 TPY. Only 24% of the total capacity is currently stand-alone HS-AD with the remaining capacity being stand-alone L-AD (28%) or L-AD codigestion (48%) at wastewater treatment plants or on-farm systems. Development trends in the US are mirroring those in the EU, however, with stand-alone capacity steadily increasing and HS-AD capacity increasing particularly rapidly relative to L-AD for OFMSW processing. The number of full-scale HS-AD facilities in the US has increased from one in 2011 to eight in 2015 and another 19 systems are expected to be operational by 2017. There are at least nine vendors of HS-AD technologies in the US, including four with facilities currently in operation and another four with projects in the planning, permitting, or construction phases. Landfill bans and taxation, mandated source-separation of OFMSW, and policies incentivizing recycling and renewable energy generation are critical factors driving the development and implementation of HS-AD.

The case study of HS-AD implementation in Florida incorporated information from industry and data from the Florida Department of Environmental Protection. There is high demand for organics recycling in Florida, with numerous counties generating several hundred thousand TPY of OFMSW and lacking organics recycling infrastructure. HS-AD implementation could increase the statewide recycling rate by as much as 13% and contribute significantly to the reaching the state's recycling goal of 75% by 2020. Furthermore, up to 7,000 and 3,500 TPY of bioavailable nitrogen and phosphorus, respectively, and up to 500 MW of energy could be recovered through HS-AD of OFMSW in the state. Based on current energy conversion efficiencies, 500 MW of energy translates to either 175 MW of electricity (approximately 660,000 metric tons of CO_2 equivalents offsets per year) and 200 MW of heat or nearly 80 million diesel gallon equivalents of vehicle fuel. However, because of the low cost of both landfilling and energy in the state and the lack of markets for compost and renewable energy certificates, legislative action is needed to improve the economic feasibility of HS-AD. Accordingly, a number of policy recommendations were formulated, including banning disposal of OFMSW to landfills and mandating source-separation of OFMSW by all generation sources.

Two phases of side-by-side bench-scale batch HS-AD experiments were carried out to investigate the potential to enhance CH_4 yield from lignocellulosic waste in HS-AD through bioaugmentation with pulp and paper mill anaerobic sludge. In the first phase, the average CH_4 yield from yard waste inoculated with pulp and paper sludge reached 100.2 ± 2.4 L CH_4/kg VS, a 73% enhancement compared with the average CH_4 yield achieved through inoculation with domestic wastewater anaerobic sludge (58.1 ± 1.2 L CH_4/kg VS). In the second phase, CH_4 yield from yard waste inoculated with digestate from digesters originally inoculated with pulp and paper sludge was 68% greater than the CH_4 yield achieved through inoculation of yard waste with digestate from digesters originally inoculated with domestic wastewater sludge (36.5 ± 0.2 L CH_4/kg VS versus 21.7 ± 0.4 L CH_4/kg VS). The enhancement in CH_4 yield achieved in this study is comparable to enhancements achieved through lignocellulosic pretreatment methods. However, this strategy incurs significantly less additional environmental and economic costs when compared with pretreatment, suggesting that it could serve as an alternative to pretreatment and improve the overall sustainability of HS-AD processes.

CHAPTER 1: INTRODUCTION AND OBJECTIVES

1.1. Introduction – Striving Toward Sustainability

It has become increasingly evident in the 20th and 21st centuries that collective human activity is having a detrimental impact on the environment. Rapidly decreasing biodiversity (Ceballos et al., 2015), accelerating eutrophication of inland and coastal waterways (Peñuelas et al., 2012), and the ever-changing climate (IPCC, 2014) are just a few of dozens of concerning examples that verify that industrialized human civilization is taking a toll on the natural world. The realization that anthropogenic influence on the environment is resulting in accelerated loss of life, both human and other, and threatening the very vitality of future generations has catalyzed a persistent and far-reaching movement toward sustainability. Sustainability is defined as the ability to meet the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987). We are far from achieving sustainability in this sense, but with unprecedented efforts to improve the sustainability of our society underway and gaining momentum, the future is bright (WWI, 2013; UNEP, 2015). These efforts range in scale from individual, to community-wide, to nation-wide, to global, each fueling the sustainability movement and playing an important role in this defining era of human history.

The *Reduce, Reuse, and Recycle* campaign, as outlined in the UN Sustainable Development Goals (SDG 12.5), is one example of an effort that has reached global proportions (UN, 2015). Wasteful consumerism and insufficient waste management results in the pollution of the environment with waste, depletion of natural resources, and the imperilment of human health across the globe (WCED, 1987; Cairns and Lackey, 1992; Tchobanoglous, 1993; Oakley et al., 2005). Reducing, reusing, and recycling waste can lessen these impacts (Bogner et al., 2007). Strategies have been developed for reducing waste generation rates, but there will always be byproducts regardless of the effectiveness of waste reduction efforts. Recognizing that these byproducts are resources and developing methods to recover their intrinsic value and reintegrate them safely back into the cycles of nature is therefore, an essential component of the *Reduce, Reuse, and Recycle* effort. The recovery of resources from the organic fraction of municipal solid waste (OFMSW), which includes food waste, yard waste, and wood waste, has been a central focus of recent reuse and recycling efforts (EPA, 2015; UNEP, 2015).

Backyard composting and mulching, grasscycling, xeriscaping, salvage food stores, food banks, community kitchen programs, and the use of food waste as animal feed are some examples of OFMSW reduction efforts. However, the optimization of centralized recycling methods for OFMSW, which makes up nearly half of waste by mass on a global basis, is imperative for reducing the environmental impacts of MSW management (World Bank, 2012). Recovery of the nutrients present in OFMSW is an invaluable endeavor when taking into account the environmental impacts of inorganic fertilizer production, the observed depletion of global mineral nutrient reservoirs, and the contribution of nutrients from OFMSW to eutrophication of aquatic ecosystems worldwide (Tchobanoglous et al., 1993; World Bank, 2012). Recovery of energy from OFMSW is also extremely important when considering the immeasurable environmental and economic costs of anthropogenic-greenhouse gas (GHG)-induced climate change, the quantity of GHG emissions that result from the biological degradation of OFMSW, and the high demand for renewable, carbon-neutral energy in the unstable fossil-fuel dominated global energy grid (EPA, 2009; Chum et al., 2011).

Aerobic composting is one of the most common methods of organics recycling in the US, but this technology only enables the recovery of nutrients from the waste. Other technologies, such as bioreactor and traditional landfill gas-to-energy (LFGTE), incineration waste-to-energy (WtE), and advanced thermal treatment (ATT) technologies (e.g. gasification and pyrolysis), recover energy from OFMSW, but are not conducive to nutrient recovery. Anaerobic digestion (AD), on the other hand, enables the efficient recovery of both energy and nutrients from OFMSW. Accordingly, several life cycle assessments (LCAs) comparing various OFMSW management technologies and studies aimed at developing OFMSW management hierarchies have shown a preference toward AD with respect to overall environmental impacts (Haight, 2005; Edelmann et al., 2005; Sundqvist, 2005; Kim and Kim, 2010; CIWMB, 2009; Morris et al., 2011; Bernstad and la Cour Jansen, 2012; WTERT, 2014). A modern OFMSW management hierarchy based on these results is shown in Figure 1.1. The pyramid is inverted to demonstrate that the most preferred management methods should also become the most common.

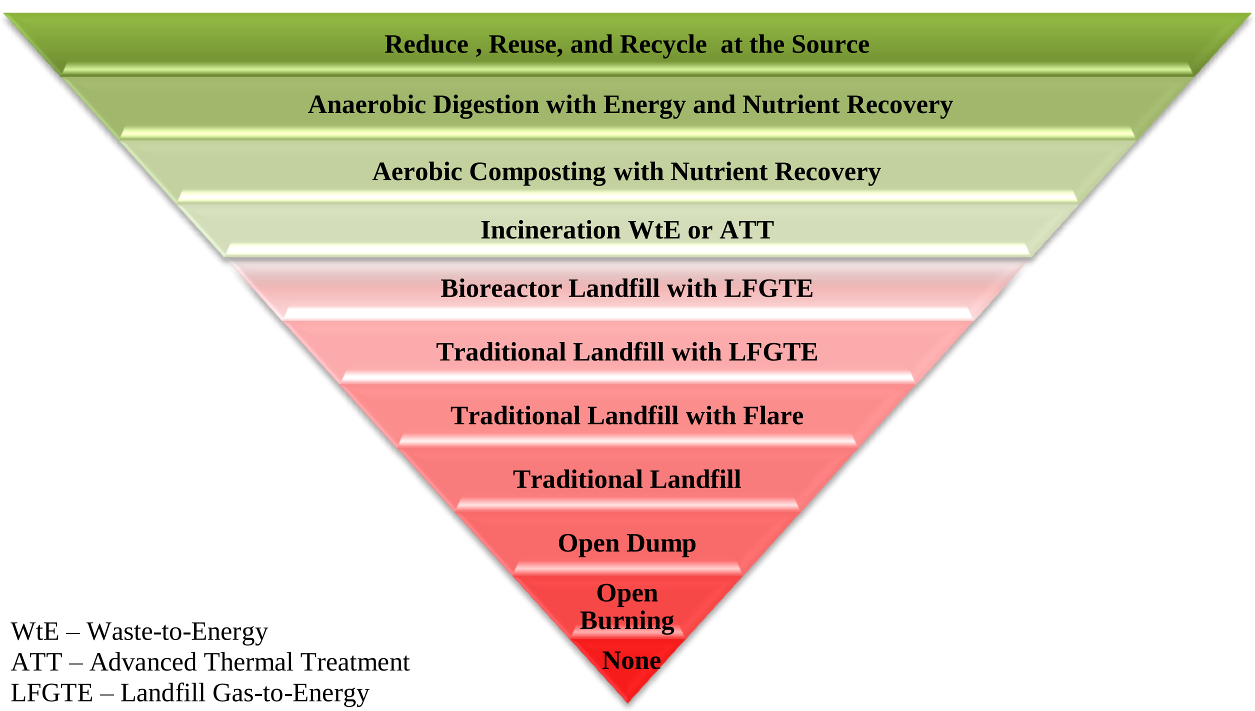

Figure 1.1: Hierarchy of OFMSW management strategies.

1.2. Background – Development of HS-AD

AD is the naturally occurring decomposition of organic materials by microorganisms under oxygen-free conditions. As the anaerobic microorganisms consume the organic material, they emit biogas – a gas mixture composed of methane (CH_4) and carbon dioxide (CO_2), usually at ratios ranging from 1:1 (50 % CH_4) to 3:1 (75% CH_4), and trace concentrations of hydrogen gas (H_2), hydrogen sulfide (H_2S), nitrogen gas (N_2), and water vapor (Chum et al., 2011). The combination of AD and aerobic digestion (microbial decomposition of organic matter in the presence of oxygen) is nature's way of recycling carbon and nutrients in biogeochemical cycles. Civilizations have used the ubiquitous presence of anaerobic microorganisms for centuries to generate fuel from organic materials for use in cooking, heating, and lighting (Khanal, 2008). AD in these household and community level contexts is still a common practice in many parts of the world, especially Asia, Africa, and Latin America, as it enables the generation of a valuable fuel (biogas) from organic household waste, humans waste, livestock waste, and crop residues (Khanal, 2008; Chum et al., 2011).

In municipal and industrial waste management applications, AD technologies are traditionally implemented for treating and recovering energy from high-strength wastewaters and organic sludges (Khanal, 2008). Thus, large-scale AD is most often applied as a low solids technology referred to as liquid AD (L-AD) (generally less than 15% total solids [TS]). It wasn't until the late 1980's and early 1990's that high-solids anaerobic digestion (HS-AD) technologies (those designed to operate with a TS content > 15%) were developed in Europe following increased landfill taxation, banning of organics disposal into landfills, and mandated source-separation of organic waste (De Baere and Mattheeuws, 2014). Since then, HS-AD for processing OFMSW has developed more rapidly in Europe than any other alternative OFMSW management technology (De Baere and Mattheeuws, 2014). A simple schematic of HS-AD for the recovery of resources from organic waste is shown in Figure 1.2 (Zupančič and Grilc, 2012). In some cases, OFMSW, especially the food waste fraction, has been integrated into L-AD systems at municipal or industrial wastewater treatment plants (Rapport et al., 2008). However, in stand-alone systems designed specifically for processing OFMSW, HS-AD technologies have been largely preferred over L-AD technologies because of the many advantages they offer (Table 1.1.) and the ease of pairing them with aerobic composting operations.

In Europe, approximately 70% of the installed capacity for AD since 2009 has been HS-AD, and in the Netherlands and Belgium approximately 80% of all composting operations incorporate AD as a primary treatment technology (De Baere and Mattheeuws, 2014). In the US, however, HS-AD development has been stifled by the low cost of waste disposal in landfills and the lack of legislation incentivizing alternative OFMSW management and recycling (Rapport et al., 22008; van Haaren et al., 2010; Li et al., 2011). Only a fraction of US states have landfill diversion goals or organics disposal bans and source-separation of organic waste is only required in a few locations (Goldstein, 2014; EREF, 2015a). Nevertheless, the first commercial HS-AD facility was constructed in the US in 2012 when the University of Wisconsin, Oshkosh took the leap toward sustainable onsite OFMSW management. Since then, legislative incentives have increased in the US, resulting in increased development of HS-AD projects and a growing number of HS-AD technology vendors doing business across the country (EREF, 2015a). The trend of increased legislative incentive is expected to continue to accelerate and HS-AD is projected to emerge as a leading OFMSW recycling technology (De Baere and Mattheeuws, 2014; RWI, 2013; EREF, 2015a).

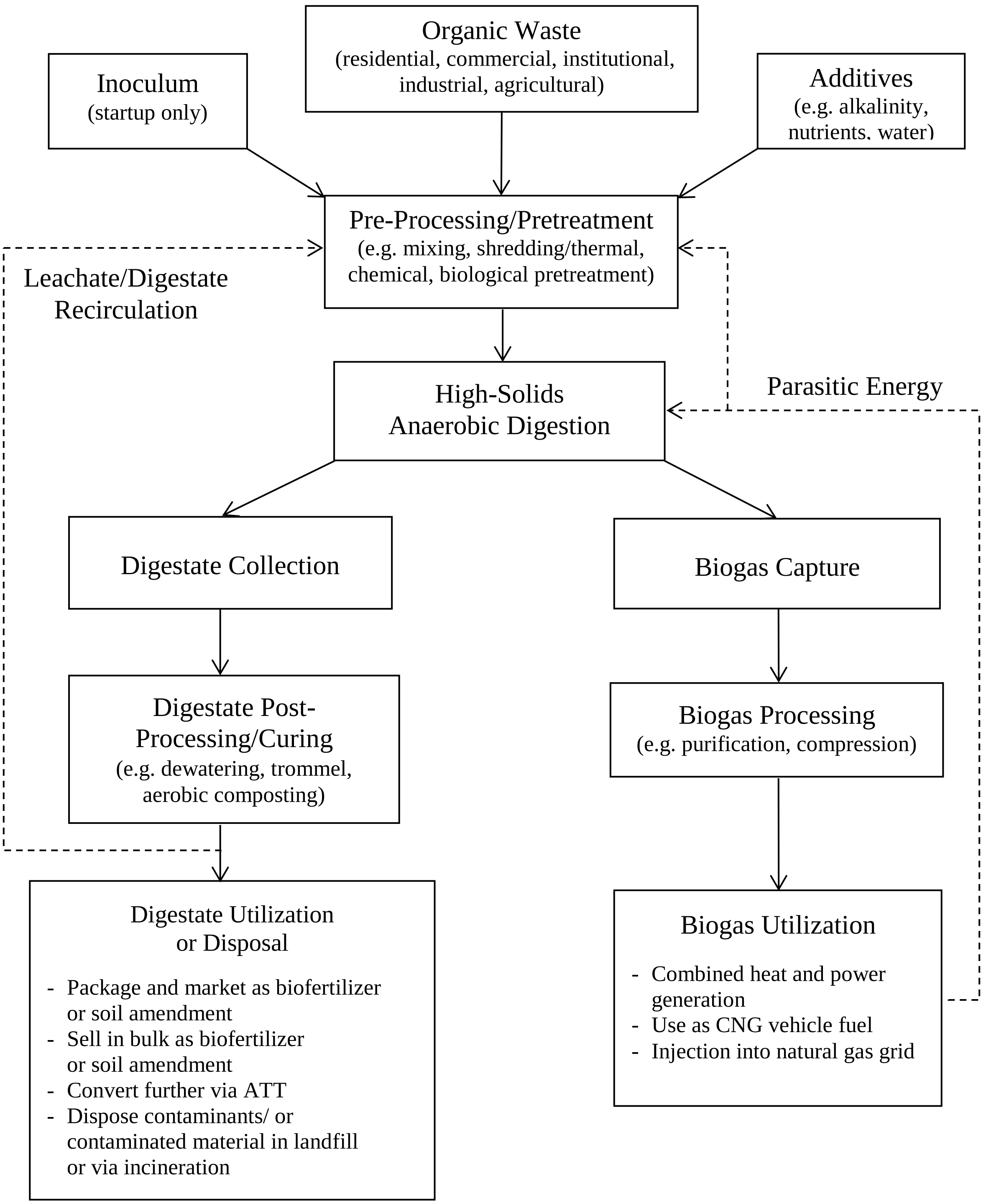

Figure 1.2: Schematic of HS-AD for the recovery of resources from organic waste (adapted from Zupančič and Grilc, 2012 CC BY 3.0 License © The Authors).

Table 1.1: Benefits of AD and advantages and disadvantages of HS-AD vs. L-AD.

Benefits of AD	**Summary**	**Reference(s)**
Enables Energy Recovery	AD is an energy positive process. The production of biogas containing CH_4 enables direct combustion for heating, lighting, cooking, conversion to electricity in combustion engines, production of compressed natural gas for use in vehicles, or injection into the natural gas grid.	Owens and Chynoweth, 1993; Tchobanoglous et al., 2003; Khanal, 2008; Li et al., 2011; Kothari et al., 2014
Enables Nutrient Recovery	Valuable nutrients, especially N and P, are present in high concentrations in the liquid/solid byproducts of AD and can be recovered through post-processing (e.g. trommel and composting/curing of the digestate).	Owens and Chynoweth, 1993; Khanal, 2008; Li et al., 2011; Kothari et al., 2014
Mass/Volume Reduction	Up to 50% substrate mass and volume reduction can be achieved through AD. Because anaerobic microorganisms are slower growing than aerobic, less excess biomass is produced in AD.	Tchobanoglous et al., 2003; Li et al., 2011; Kothari et al., 2014
Destruction of Pathogens	Long term exposure to high temperatures in a microbiologically competitive anaerobic environment ensures reliable pathogen destruction/inactivation.	Wilkie, 2005; Khanal, 2008
Reduced GHG Emissions	AD significantly reduces GHG emissions through capturing and converting CH_4 which otherwise would have been emitted through degradation of organic waste in uncontrolled environments or as fugitive emissions in landfills; additional offsets can be achieved through offsetting fossil-fuel derived electricity consumption.	Owens and Chynoweth, 1993; Tchobanoglous et al., 2003; Edelmann et al., 2005; Li et al., 2011; Kothari et al., 2014
Reduced Odors	AD in enclosed reactors with biogas capture yields little odor.	Wilkie, 2005; Khanal, 2008
Advantages of HS-AD vs. L-AD		
Reduced Energy Consumption	Less energy used for heating and internal mixing yields lower parasitic energy losses and higher overall energy efficiency.	Li et al., 2011; Kothari et al., 2014.
Reduced Water Use	Zero or minimal water addition is required in HS-AD, percolate is often recirculated, and minimal excess percolate production results in reduced side-stream treatment costs.	Li et al., 2011; Kothari et al., 2014.
Reduced Reactor Size	The reduced moisture content and capacity for HS-AD systems to handle greater organic loading rates yield lower required reactor volumes for given loading/biogas yield rates.	Guendouz et al., 2010; Li et al., 2011; Kothari et al., 2014.
Reduced Post-Processing	The compost-like digestate byproduct of HS-AD requires only minor post-processing (trommel/sieve and composting/curing) whereas L-AD byproduct first requires dewatering.	Li et al., 2011; Kothari et al., 2014.
No Waste Stratification	In L-AD stratification of FOG and fibrous materials can create operational challenges. This does not occur in HS-AD systems.	Guendouz et al., 2010.
Disadvantages of HS-AD vs. L-AD		
More Inoculum Required	Lower moisture content can yield reduced microbe-substrate contact resulting in greater inoculation requirements.	Li et al., 2011.
Reduced Homogeneity	Lower moisture content reduces mixing capabilities and homogeneity of digester contents yielding spatial variations in process efficiency.	Kothari et al., 2014.
Longer Retention Times	Although retention times in HS-AD systems are often similar to those of liquid systems (~20 days), up to three times longer retention times are needed in HS-AD in some cases due to slower mass transport.	Li et al., 2011; Kothari et al., 2014.

1.3. Research Motivation and Scope

The underlying motivation behind this thesis research is to contribute to improving the sustainability of OFMSW management. For each passing day that OFMSW is disposed of or incinerated, substantial opportunity to offset environmental impacts and public health costs through OFMSW recycling is lost. This thesis aims to contribute to accelerating the implementation of HS-AD for OFMSW recycling by investigating and reporting the state of the art, conducting a case-study assessing the potential to implement HS-AD in the state of Florida, and exploring potential strategies for enhancing energy recovery efficiency in HS-AD of OFMSW. This thesis focuses solely on HS-AD of OFMSW and does not address in detail L-AD of OFMSW or HS-AD of other possible feedstocks, such as energy crops or livestock waste.

1.3.1. Identifying and Understanding Trends in HS-AD Implementation

Accelerating the integration of HS-AD technologies into existing waste management regimes requires comprehensive knowledge of the state of the art of HS-AD and a fundamental understanding of the environmental, economic, and political factors affecting the integration of HS-AD technologies into waste management plans (UNEP, 2009). Development of HS-AD in Europe is well-documented (De Baere and Mattheeuws, 2014) and is summarized in this thesis (Section 2.3.2.). However, a detailed assessment of the beginnings of HS-AD implementation in the US was previously lacking. Thus, an investigation of existing HS-AD facilities in the US and projects in the planning, permitting, and construction phases was undertaken (Section 2.3.3.). The findings were organized into a detailed database of all ongoing HS-AD projects, with information ranging from project costs and funding sources to technology type and system capacity. The results were used to elucidate trends in development and provide an overview of available technologies. Environmental, economic, and policy considerations associated with HS-AD implementation are also reported (Sections 2.2.3. – 2.2.5.).

1.3.2. Accelerating HS-AD Implementation

Prior to this study, no large-scale case studies had been carried out to quantify the environmental gains attainable through implementation of HS-AD, identify locations where implementation is most suitable, and evaluate the economics such that a pathway for successful integration of the technology could be defined. Thus, a detailed case study of the outlook for implementation of HS-AD in the state of Florida was conducted and is reported in this thesis (Chapter 3). Florida is an appealing state for conducting an assessment of this sort because of the large population, high energy demand, and high OFMSW generation rates in the state. Additionally, Florida has a food waste recycling rate of only 7%, a statewide recycling goal of 75% by 2020, and the warm climate in Florida is economically advantageous for AD because high ambient temperatures reduce the amount of heat energy needed to maintain internal operating temperatures (Tchobanoglous et al., 2003; FDEP, 2015).

1.3.3. Improving HS-AD Process Efficiency

A main factor affecting the economics of HS-AD is the local energy market (Rapport et al., 2008). Assuming there is a high local demand for energy, maximizing process efficiency with respect to net energy recovery is vital to the economic sustainability of HS-AD. Net energy recovery is a function primarily of the biomethanation efficiency in a given HS-AD system (methane yield per unit time per unit waste loaded to the system), the parasitic energy demand of the system (energy demand of system operations), and energy conversion efficiency (methane conversion to electricity, heat, and/or vehicle fuel) (Kothari et al., 2014). Energy conversion efficiency is dependent on the efficiency of external technologies (e.g. combined heat and power units and biogas scrubbing and compressing systems). Reducing parasitic energy demand requires the development of innovative HS-AD systems that minimize energy requirements without sacrificing biomethanation efficiency, a task that many HS-AD vendors are pursuing (Li et al., 2011). Biomethanation efficiency, on the other hand, can be improved through a number of operational strategies.

Prior HS-AD studies have investigated the optimization of operating parameters, codigestion strategies, and pretreatment methods for maximizing biomethanation efficiency (Kothari et al., 2014; Brown and Li, 2013; Chen et al., 2014; Zheng et al., 2014; Yang et al., 2015). Only a few studies have been conducted, however, to explore the possibility of utilizing an alternative inoculum during HS-AD process startup as opposed to the conventional approach of utilizing wastewater anaerobic sludge (Lopes et al., 2013; Mussoline et al., 2013). The novel strategy, known as bioaugmentation, aims to identify an easily attainable and abundantly available inoculum containing microorganisms adapted to degrade wastes commonly present in OFMSW more efficiently than the microorganisms present in conventional inoculum sources. For example, *Clostridium cellulovorans*, which originate in wood chips and produce enzymes that facilitate delignification in lignocellulosic materials such as yard waste and agricultural crop residues (Tamaru et al., 2010), have been hypothesized to be prevalent in pulp and paper mill anaerobic sludge (Mussoline et al., 2013). Due to the lack of research in this area, experiments aiming to assess the effectiveness of bioaugmentation utilizing pulp and paper mill anaerobic sludge for enhancing methane yields from yard waste were designed and conducted and are described in this thesis (Chapter 4).

1.4. Research Questions and Objectives

The specific research questions and associated objectives that were developed and addressed in this thesis are as follows:

1. *What is the state of the art of HS-AD?*
   i. Investigate the state of HS-AD in Europe and trends in development.
   ii. Compile a database of HS-AD projects in the US and report trends in development.
   iii. Provide an overview of HS-AD technologies currently available in the US.
2. *What is the outlook for implementation of HS-AD in the state of Florida?*
   i. Identify locations where HS-AD implementation would be most suitable in Florida based on OFMSW generation and recycling rates and existing MSW infrastructure.
   ii. Quantify the economic and environmental incentive for HS-AD implementation in Florida and identify key barriers.
   iii. Provide policy recommendations and outline possible strategies for improving the economic competitiveness of HS-AD in Florida.
3. *Is bioaugmentation using pulp and paper mill anaerobic sludge a viable method for improving methane yields from lignocellulosic wastes in HS-AD?*
   i. Study the effects of bioaugmentation of yard waste with pulp and paper mill anaerobic sludge on methane yields in batch HS-AD and determine whether enhancements in methane yields can be sustained through digestate recirculation.
   ii. Investigate the mechanisms by which bioaugmentation with pulp and paper mill anaerobic sludge enhances methane yields in HS-AD of yard waste.
   iii. Study the effects of bioaugmentation with pulp and paper mill anaerobic sludge in HS-AD codigestion applications.

CHAPTER 2: LITERATURE REVIEW AND INDUSTRY SURVEY

2.1. Introduction to Anaerobic Digestion

AD, simply defined, is a process in which organic matter is metabolized by microorganisms in an environment free of oxygen (Khanal, 2008). Along with aerobic biodegradation, AD is nature's way of recycling carbon, nutrients, and other constituents present in organic material back into the cycles of life. Anaerobic microorganisms are ubiquitous life forms, some of the oldest on Earth, present nearly everywhere on the planet from the bottom of oceans, to deserts, to deep within the Earth's crust (DOE, 2013). Just as human beings consume organic material and oxygen and produce energy, biomass (cells), and carbon dioxide, anaerobic heterotrophic microorganisms consume organic material and produce biomass, biogas, and heat. In the context of engineered AD systems in waste management, the metabolic processes of anaerobic microorganisms are leveraged to recover energy from organic waste in the form of methane and stabilize the waste to produce a valuable nutrient-rich soil amendment, or biofertilizer (Rapport et al., 2008). On the surface, AD seems like a simple natural process, but when investigated on the micro-scale it is revealed that AD is a complex phenomenon involving a consortium of interdependent microorganisms. Because of the complex nature of AD, the efficiency of the engineered AD processes depends on many parameters. In this chapter, a review of literature pertaining to the fundamentals of AD, with a particular focus on HS-AD, and other literature relevant to the Research Questions described in Chapter 1 is presented. Additionally, MSW management and the role of HS-AD in OFMSW management are introduced in greater detail and the current state of the art of HS-AD of OFMSW is reported (*Research Question 1*).

2.1.1. L-AD and HS-AD Distinction and Alternative Terminology

The distinction between L-AD and HS-AD is the TS content of the feedstock. L-AD systems are generally designed to process feedstock with less than 15% TS and HS-AD systems are generally designed to process feedstock with 15% TS content or greater (Li et al., 2011; De Baere and Mattheeuws, 2014). The TS content of 15% by mass is most often cited as the cutoff line between L-AD and HS-AD, although there is a lack of consistency in literature. For example: Kothari et al. (2014) stated that the cutoff point was 22% TS and then went on to separate AD into low (<15%), medium (15-20%), and high (>20%) solids categories; Tchobanoglous et al. (2003) also separated AD into three categories, but with L-AD constituting AD with less than 10% TS; Ward et al. (2008) defined L-AD as less than 16% TS; and Karagiannidis and Perkoulidis (2009) defined L-AD as 10-25% TS and HS-AD as 30-40% TS.

The terminology used for L-AD and HS-AD is also inconsistent in literature and in industry. L-AD is often referred to as wet-AD, or simply as AD, and there are numerous interchangeable terms for HS-AD. In addition to the term "high-solids" for denoting AD with high TS content, the terms "solid-state", "dry", and "solid-substrate" are all used as equivalent terms. Similarly, the terms "fermentation" and "anaerobic composting" have often been used in the place of "anaerobic digestion". This creates a minimum of twelve equivalent ways to refer to HS-AD, all of which are used in literature. For example, HS-AD has instead been called dry AD (DAD) (De Baere and Mattheeuws, 2014), solid-state AD (SS-AD) (Li et al., 2011), anaerobic composting (Schievano et al., 2010), dry fermentation (Ghanem et al., 2001), dry anaerobic fermentation (Kumar et al., 2010), solid-state fermentation (Pandey, 2003), high-solids anaerobic fermentation (Molnar and Bartha, 1988), and solid-substrate anaerobic digestion (Poggi-Varaldo et al., 1997). In some cases, there are slight differences in the definitions of these alternative HS-

AD terms. For example, *fermentation* terminology has been used specifically for describing simple, single-stage, batch HS-AD systems (EPEM, 2015) and has also been used to describe biochemical conversion systems that aim to generate hydrogen as opposed to methane (Lin et al., 2013). In other cases, researchers have used the same acronyms to denote different things. For example, Heo et al. (2004) use SSAD to denote single-stage AD. In general, specific researchers and technology vendors in industry stick to their preferred terminology, but the inconsistency between them creates challenges with respect to reviewing literature. For this reason, a literature review methodology was developed to ensure a comprehensive review could be conducted (Appendix C). The reasoning behind the selection of HS-AD as the preferred term for this thesis can be summarized as follows and is described in greater detail in Appendix C: HS-AD is the most accurate term and the least likely to be confused with other acronyms or meanings.

2.1.2. Process Microbiology

AD involves four general groups of microorganisms: fermentative, acidogenic, acetogenic, and methanogenic (Khanal, 2008). These organisms can be further divided into additional subgroups, such as solubilizing and hydrolyzing fermenters and CO_2 reducing and acetoclastic methanogens. In many cases, hydrolytic microorganisms are referred to as a separate group of microorganisms (Li et al., 2011; EREF, 2015a). The metabolic progression of AD involves four primary steps (Figure 2.1): hydrolysis, acidogenesis, acetogenisis, and methanogenesis (Adekunle and Okolie, 2015). However, two additional steps, or pathways, are sometimes cited as independent and critical components of the process – solubilization and anaerobic oxidation (Massé and Droste, 2000; Khanal, 2008) – and fermentation is sometimes cited as a separate metabolic step in the place of acidogenesis (Li et al., 2011). A detailed overview of the microbiology, including the main groups of microorganisms and the fundamental substrates and digestion pathways, is shown in Figure 2.1.

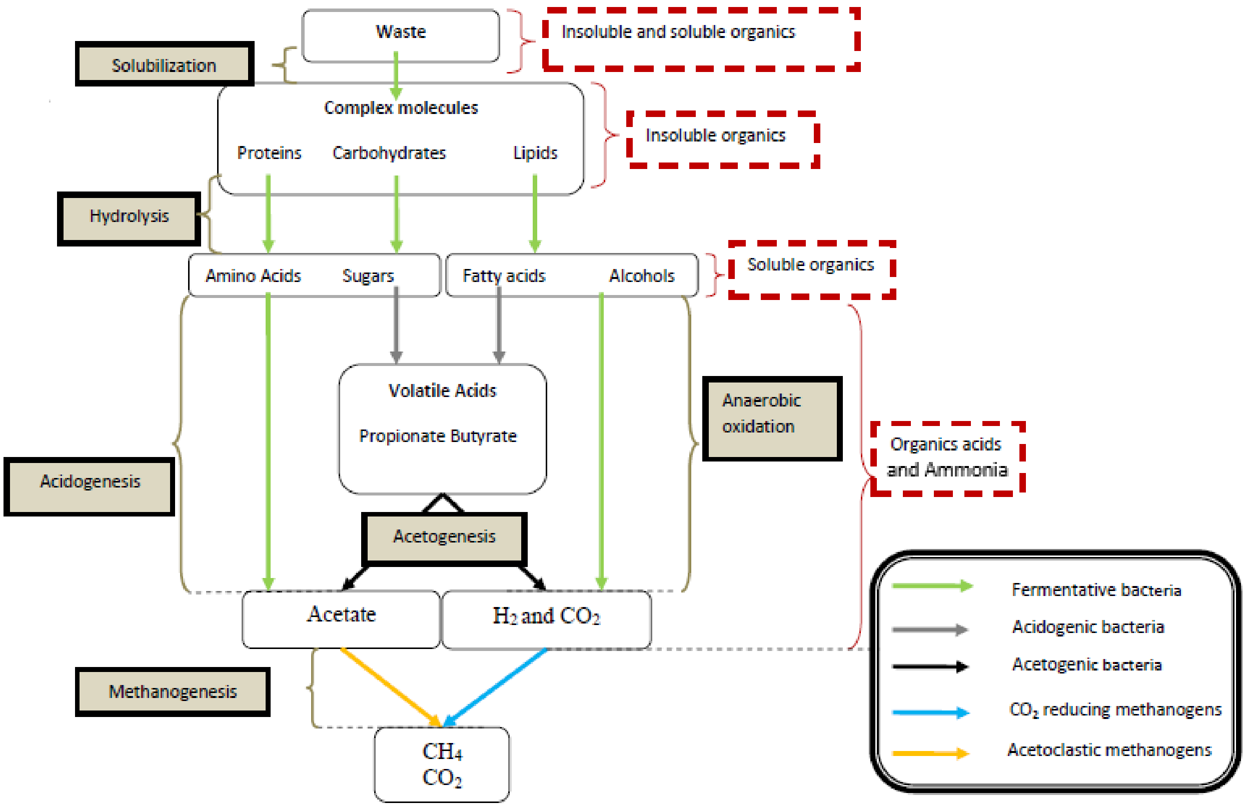

Figure 2.1: Schematic of the metabolic pathways of the AD process (originally adapted from Massé and Droste, 2000, further adapted from Kinyua, 2013 and reused with permission).

The four major microbial processes, or metabolic phases, of AD can be summarized as follows (Khanal, 2008; Li et al., 2011; Adekunle and Okolie, 2015):

- *Hydrolysis* – conversion of insoluble organics and complex molecules, such as lipids, proteins, carbohydrates, polysaccharides, and nucleic acids, into amino acids, monosaccharides, fatty acids, alcohols and other simple organics suitable to serve as energy and/or carbon sources for subsequent groups of microorganisms. Hydrolysis is carried out by strict anaerobes and facultative bacteria, certain species of which secrete extracellular enzymes that aid in breaking down complex molecules. Hydrolysis is generally considered the rate limiting step in HS-AD, especially in cases where lignocellulosic wastes are a primary feedstock (Veeken and Hamelers, 1999).

- *Acidogenesis* – conversion of monomers, simple sugars, amino acids, and fatty acids into short-chain volatile organic acids, such as butyric, propionic, and acetic acids, and hydrogen and carbon dioxide gases. Acidogenesis is carried out by obligate anaerobic and facultative bacteria.
- *Acetogenisis* – conversion of volatile fatty acids (VFAs) to acetate and hydrogen and carbon dioxide gases, which are direct substrates for methanogenesis. Acetogenesis is carried out by strict anaerobes.
- *Methanogenesis* – conversion of acetate, hydrogen, and carbon dioxide into methane and carbon dioxide. Methanogenesis is performed by strict obligate anaerobic archaea and is considered the rate limiting step in most L-AD applications.

On an operational basis, the AD process is often separated into two primary phases: the acid phase (hydrolysis, acidogenesis, and acetogenesis) and the gas phase (methanogenesis) (Adekunle and Okolie, 2015; Deublein and Steihauser, 2008). This convention parallels the method commonly used in two-stage commercial AD systems of facilitating hydrolysis, acidogenesis, and acetogenesis in one reactor and methanogenesis (or in some cases acetogenesis and methanogenesis) in a separate reactor (Section 2.3.1.). Additional microbial process that take place in AD include sulfate reduction (respiration) to hydrogen sulfide and ferric iron (Fe^{3+}) reduction (respiration) to ferrous iron (Fe^{2+}) (Madigan et al., 2014).

2.1.2.1. Microbial Relationships

The relationships that exist between the groups of bacteria active in AD are complex and each interaction is essential to the wellbeing of the community. Of the many relationships and dynamics at work in the process, the syntrophic relationship between methanogens and acetogens is the most critical (Khanal, 2008). As acetogens metabolize VFAs and generate hydrogen gas, the partial pressure of hydrogen gas in the system increases and can become inhibitory to the acetogens themselves (Madigan et al., 2014). It is the conversion of the hydrogen gas to methane by the hydrogenotrophic methanogens that ensures that the partial pressure of hydrogen remains below inhibitory levels. In this way, the efficiency of acetogens in converting volatile acids to methanogenic substrates is dependent upon the efficiency of the methanogens in converting hydrogen gas to methane. Likewise, the rate of methanogenesis is dependent upon the rate of acetogenesis, which supplies the methanogens with a significant fraction of their energy and carbon sources. Hence, the relationship is mutually dependent and syntrophic (Madigan et al., 2014). The relationship between fermentative bacteria and methanogens is similar. Methanogens depend on fermentative bacteria to provide them with their energy and carbon sources. However, if the fermentative bacteria in an AD system produce VFAs more rapidly than the methanogens can consume them, the system pH can plummet and the pH sensitive methanogens can be inhibited, resulting in greater accumulation of VFAs and an eventual total acidification and failure of the system (Amani et al., 2010). When this occurs, the fermentative bacteria can also no longer thrive. Hence, the fermentative bacteria and the methanogens in an AD system are mutually dependent. Inhibition and system acidification is discussed further in Section 2.1.4.

2.1.3. Process Stoichiometry

Each of the main phases making up the AD process has its own general stoichiometric relationships. These relationships are helpful for conducting carbon and nutrient mass balances and for understanding the dynamics of alkalinity and other parameters of concern (i.e. VFAs and ammonia). Equations 2.1 and 2.2 present a simplified stoichiometry of fermentation, in which organic matter, with an assumed empirical formula of $C_5H_7O_2N$, is converted into acetic acid/acetate (CH_3COOH/CH_3COO^-) (Haandel & Lubbe, 2007).

$$C_5H_7O_2N + 3\ H_2O \rightarrow 2.5\ CH_3COOH + NH_3 \qquad (Eq\ 2.1)$$
$$C_5H_7O_2N + 3\ H_2O \rightarrow 2.5\ CH_3COO^- + 1.5\ H^+ + NH_4^+ \qquad (Eq\ 2.2)$$

Notice that in the fermentation phase, ammonia is produced and protons are released. Ammonia is not an energy source for microorganisms active in the AD process and is consumed only in the synthesis of new cells (as a nitrogen source) by the microbial populations present in the process (Kayhanian, 1994; Mussoline et al., 2013). Thus, the majority of the ammonia that is produced remains present in the system and accumulates at a rate that is proportional to the rate of fermentation (hydrolysis/acidogenesis). Note also that as protons are released, alkalinity is depleted. Equation 2.3 shows the representative stoichiometric reaction of the conversion of acetic acid to methane and carbon dioxide (Haandel & Lubbe, 2007).

$$2.5\ CH_3COO^- + 2.5\ H^+ \rightarrow 2.5\ CO_2 + 2.5\ CH_4 \qquad (Eq\ 2.3)$$

Other methanogenic reactions include the conversion of ethanol to acetic acid and methane (Equation 2.4) and the conversion of carbon dioxide and hydrogen gas to methane and water (Equation 2.5).

$$2\ C_2H_5OH + CO_2 \rightarrow CH_4 + 2\ CH_3COOH \qquad (Eq\ 2.4)$$
$$CO_2 + 4\ H_2 \rightarrow CH_4 + 2\ H_2O \qquad (Eq\ 2.5)$$

Theoretically, greater methane production can be achieved from the reduction of carbon dioxide than from the reaction shown in Equation 2.3, though in AD the majority of the methane comes from the acetic acid reaction carried out by acetoclastic methanogens because hydrogen is typically limiting in anaerobic systems (Amani et al., 2010). For every mole of acetic acid consumed in the methanogenesis phase of AD, a mole of protons are consumed (Eq. 2.3). As shown in Equation 2.2, 1.5 moles of protons are produced per 2.5 moles of acetic acid. Hence, when all processes are in balance, the overall AD process is an alkalinity producing process (more protons consumed than produced). This can be seen in Equation 2.6, showing the stoichiometric representation of the overall AD process of microbial biomass. In this reaction, alkalinity as bicarbonate (HCO_3^-) is produced at a one-to-one molar ratio with the biomass being digested (Haandel & Lubbe, 2007).

$$C_5H_7O_2N + 4\ H_2O \rightarrow HCO_3^- + 1.5\ CO_2 + 2.5\ CH_4 + NH_4^+ \qquad \text{(Eq 2.6)}$$

In accordance with the above stoichiometry, a well-functioning AD process will produce methane, CO_2, alkalinity, and ammonia. Based on this observation, these parameters can be used as a qualitative measure for the overall health or functionality of an AD system. For example, in batch AD, the greater the process efficiency, the more organic matter is degraded and the more alkalinity, ammonia, and methane is generated. However, due to the complex biochemical nature of AD, changes in alkalinity and ammonia concentrations can be dynamic, leaving methane generation as the best measure of AD process efficiency (Khanal, 2008).

2.1.4. Operating Parameters

The major operational parameters monitored in AD can be separated into physical and chemical categories (Table 2.1). Physical parameters include TS content, substrate to inoculum ratio (S/I ratio), substrate to substrate ratios in codigestion, temperature, retention time, organic loading rate (OLR), VS reduction, and methane generation rate. Chemical parameters include carbon to nitrogen ratio (C/N ratio), pH and alkalinity, VFA concentration, and concentrations of micronutrients and inhibitory compounds such as free NH_3 and H_2S. Each parameter is interrelated and consequently, changes in any one parameter are accompanied by changes in other parameters.

Table 2.1: Summary of AD operating parameters, optimal ranges, and inhibitory concentrations.

Parameter	Optimal Ranges and Inhibitory Concentrations	Reference(s)
Total Solids Content	AD is often split into two general categories based on total solids content, HS-AD (greater than 15% TS) and L-AD (less than 15% TS), although TS content can vary widely in different AD applications (3% - 40% TS). CH_4 yields in HS-AD have been shown to increase proportionally with TS content, but TS content beyond a certain threshold (25-40%) has been shown to be inhibitory.	Rapport et al., 2008; Li et al., 2011; De Baere and Mattheuws, 2014; Kothari et al., 2014
Substrates, Inocula, S/I Ratio, and Ratios in Codigestion	Substrate(s) and inoculum both significantly influence process efficiency and CH_4 yields in HS-AD. Optimal S/I ratios have been shown to range widely (3/1 – 1/7). Ratios of various substrates in codigestion are also critical to process efficiency and have been shown to range widely.	Li et al., 2011; De Baere and Mattheuws, 2014; Deublein and Steinhauser, 2008
Waste Particle Size and Feedstock Porosity	Reduction of waste particle size has been shown to significantly increase CH_4 yields in HS-AD and thus, wastes are often ground to 40 mm particle size or less before digestion. However, in systems with percolate recirculation, the presence of larger particles (greater than 40 mm) in feedstocks is important for feedstock ‘structure’ and porosity and for overall digestion efficiency. Thus, particles retained in the trommel process are often recirculated.	Sharma 1988; Izumi et al., 2010; Veeken, 2014
Temperature	Common operating ranges of temperature in AD are 35-40 °C (mesophilic) and 50-55 °C (thermophilic), although AD microorganisms are capable of surviving temperatures ranging from 0-82°C. Thermophilic digestion has been shown to generate greater biogas yields, but mesophilic digestion has been traditionally considered more stable.	Amani et al., 2010; De Baere and Mattheuws, 2014
Retention Time	Retention time, defined in practice as the time that the waste material undergoes digestion, is a function of feedstock biodegradability and system operating conditions, and is selected to optimize the economics of the system. Retention times in HS-AD systems generally range from 10-30 days.	Tchobanoglous et al., 2003; Amani et al., 2010; Kothari et al., 2014; PIS, 2008
Organic Loading Rate (OLR)	OLR is the mass of organic matter loaded to an AD system per unit volume per unit time. Biogas yields are directly affected by OLR and if the OLR of system is too high, inhibition will occur. The maximum recommended OLR for mesophilic and thermophilic AD systems are 5 and 8 kg VS/m^3-d.	Vandenburgh and Ellis, 2002; Zupančič and Grilc, 2012; Chen et al., 2014
Volatile Solids Reduction and Methane Yield	VS reduction and methane yield, most commonly expressed in % by mass and L CH_4/kg VS, respectively, are the most direct measures of digester performance and degree of digestion. Both decrease logarithmically with time and depend on feedstock biodegradability and system operating parameters. Maximum VS reduction achievable in AD is often cited as 60%, though this value is exceeded in some cases.	Burton and Turner, 2003; Kaparaju and Rintala, 2005; Li et al., 2011
Carbon to Nitrogen (C/N) Ratio	C/N Ratio is a critical parameter in AD. Optimal values generally range from 20/1 to 30/1, although this varies depending on the bioavailability of N and C in the feedstock(s) being degraded. When the C/N ratio is too low, ammonia inhibition can result, and when it is too high, nitrogen can become limiting and slow biogas production or VFAs can accumulate, leading to inhibition.	Li et al., 2011; Kothari et al., 2014; Yang et al., 2015
pH and Alkalinity	Sufficient alkalinity is essential for efficient digestion. Alkalinity greater than 1,000 mg/L as calcium carbonate ($CaCO_3$) and pH greater than 6.5 are recommended to ensure that methanogenic populations are not inhibited.	Tchobanoglous et al., 2003; Amani et al., 2010
Volatile Fatty Acids (VFAs)	VFAs are a crucial intermediate of AD, which can accumulate at high OLRs and cause inhibition. Total VFA concentrations as acetic acid greater than 10,000 mg/L are generally considered inhibitory to methanogenesis.	Khanal, 2008; Amani et al., 2010
Ammonia (NH_3), Hydrogen Sulfide (H_2S), and Micronutrients	Ammonia is naturally produced in AD and important for cell synthesis, but becomes inhibitory, especially to methanogens, at concentrations around 1,500 - 1,700 mg/L. H_2S is produced when sulfates are present in the feedstock(s) being digested, is odorous, corrosive, degrades biogas quality and can become inhibitory at concentrations greater than 200 mg/L. Additionally, certain micronutrients are required in trace amounts for AD, some of which can become inhibitory beyond certain concentrations.	Gerardi, 2003 Chen et al., 2008 Amani et al., 2010 Yenigün and Demirel, 2013

2.1.4.1. Total Solids Content

HS-AD systems are AD systems commonly selected for processing organic solid wastes such as OFMSW and agricultural crop residues (Li et al., 2011; De Baere and Mattheeuws, 2014). The feedstock to these systems is a “stackable” porous mixture of waste, which can be moved with screw augers, high-power pumps, conveyer belts, or with machinery such as a front-end loader (Rapport et al., 2008; Li et al., 2011; Kothari et al., 2014). L-AD systems are commonly applied for processing domestic wastewater sludge, industrial wastewater, and livestock manures (Tchobanoglous et al., 2003; De Baere and Mattheeuws, 2014). The feedstock to these systems can be easily conveyed to the reactors using liquid or slurry pumps (Rapport et al., 2008). The operational TS content varies from less than 1% to more than 40% and can also change within multi-stage systems or single-stage systems as feedstock changes or with changes in season. Some systems are designed to operate in specific ranges of TS and water or percolate is added to feedstock to adjust TS content when necessary (Figure 1.2) (Li et al., 2011).

As previously mentioned, OFMSW, especially the food waste fraction, is in some cases added to existing L-AD systems treating domestic wastewater to enhance biogas yields and electricity generation at wastewater treatment plants (Rapport et al., 2008; Zupančič et al., 2008). This strategy is beneficial in that food waste can be loaded at high rates without risk of overloading because of the high alkalinity present in the wastewater sludge (Bolzonella et al., 2006a; Zupančič et al., 2008). Stand-alone L-AD facilities constructed specifically for processing OFMSW also exist (EREF, 2015a). However, research exploring the effects of TS content on methane yields from OFMSW suggests that methane yields generally increase proportionally with increases in TS content up to a certain threshold where the OLR becomes too high and inhibition occurs and/or mass transfer limitations between substrates and microbes reduce methane production rates. Chen et al. (2014) found significant increases in methane production from food and yard waste as TS was increased from 5% to 10% and from 10% to 15%, but saw reduction after increasing TS content from 15% to 20% and again from 20% to 25%. Similarly, Fernandez et al. (2008) reported a 17% decrease in methane yield when increasing TS content from 20% to 30% in HS-AD of OFMSW. Based on these studies, it would appear that the optimal TS content for AD of OFMSW is somewhere between 15% and 20%. However, this is not the case for many full-scale HS-AD systems, which consider operation to be optimal at TS contents up to 40% (Section 2.3.3.) (Rapport et al., 2008; Li et al., 2011).

2.1.4.2. Substrates, Inocula, and Codigestion

In microbiology, substrate refers to any molecules involved in metabolic processes. In HS-AD, however, substrate typically refers to a single material which may be digested (e.g. food waste) (Li et al., 2011). Any organic material can be a suitable substrate for AD, although the physical and chemical characteristics of each potential substrate make some substrates more valuable and appropriate than others for processing via HS-AD. Numerous protocols and assays have been developed for assessing the value of various substrates with respect to energy recovery potential in AD (Owen et al., 1979; Owens and Chynoweth, 1993; Angelidaki et al., 2009) and a wide range of potential substrates have been tested (Gunaseelan, 1997). Table 2.2 lists the specific methane yields achieved in select studies focusing on common OFMSW-derived substrates used in HS-AD and common HS-AD codigestion strategies (AD of more than one substrate).

In some cases, the term *feedstock* is used in the place of *substrate* to refer to materials that can potentially be processed via HS-AD (Kothari et al., 2014). In this thesis, the term feedstock is used to refer specifically to material loaded into a digester, which is often a mixture of substrate(s) and inoculum. Inoculum is the term used to refer to any microbiologically active material that is mixed with substrate to increase the density of anaerobic microorganisms present, thereby accelerating the acclimation of the HS-AD process and improving overall process efficiency (Deublein and Steinhauser, 2008). An ideal inoculum is one that is readily available, biologically active, relatively concentrated, and acclimated to the environmental conditions that it is going to be introduced to (Boulanger et al., 2012; Brown and Li, 2013). The most commonly used inocula for HS-AD system startup are anaerobic sludges from domestic wastewater AD systems (Deublein and Steinhauser, 2008). After initial startup of HS-AD systems, digestate (the digested material that removed from HS-AD systems) and/or percolate (the liquid that leaches from wastes while they are being digested) are typically used to inoculate fresh waste material (Rapport et al., 2008).

Forster-Carneiro et al. (2007) investigated the performance of six inoculum sources for HS-AD of source-separated OFMSW (SS-OFMSW) at equal substrate to inocula (S/I) ratios (by VS). Anaerobic wastewater sludge was found to be the best with respect to methane yield and VS reduction and cattle manure was found to be the worst. Boulanger et al. (2012) found that methane yields increased proportionally with decreases in S/I ratio (increases in inoculum addition) during AD of comingled MSW. Several other studies measuring the effects of varying inoculum sources and S/I ratios have been conducted (Sans et al., 1995; Brown and Li, 2013; Mussoline et al., 2013), providing verification that increases in inocula leads to increases in methane yields. These studies also revealed that optimal inocula and S/I ratios are dependent on substrate(s) and design parameters, such as operating temperature and TS content. It should also be noted that in full scale systems there is a practical limit to how much inoculum can be used.

Table 2.2: Specific methane yields from common substrates in HS-AD and in codigestion.

Substrate(s)	Notes	Methane Yield ($L\ CH_4/kg\ VS_{added}$)	Reference(s)
Mechanically Separated OFMSW	Samples from Sumter County, FL; ground to 1.53 mm	186-222	Owens and Chynoweth, 1993
Yard Waste	Turf grass and Laurel Oak leaves and branches at a 1:1:1: ratio (by VS); ground to 1.53 mm	143	
Leaves	Laurel Oak; ground to 1.53 mm	123	
Branches	Laurel Oak; ground to 1.53 mm	134	
Grass	Turf Grass; ground to 1.53 mm	209	
Office Paper	Mixed; ground to 1.53 mm	369	
Newspaper	Spruce w/ print; ground to 1.53 mm	203	
Magazine Paper	Ground to 1.53 mm	100	
Black Alder	Samples described as "woody biomass"; all samples ground to 0.8 mm; CH_4 yields became negligible after 120 days	240	Jerger et al., 1982
Cottonwood		220	
Eucalyptus		14	
Hybrid Poplar		320	
Loblolly Pine		63	
Sycamore		320	
Grass	Bermuda grass	137-226	Sharma et al., 1998
Food Waste	Source-separated, multiple commercial sources; ground to unspecified size	440	Zhang et al., 2007
Food Waste	Source-separated from the University of Cadiz restaurant; ground to2mm	530	Forster-Carneiro et al., 2007
Fruit and Veg. Waste	Source-separated from food markets, shredded to "small" size	420	Bouallagui et al., 2005
Food Waste and Waste Activated Sludge	90% food waste and 10% waste activated sludge (by VS)	489	Heo et al., 2004
Food Waste and Green (Yard) Waste	Food waste from a student was blended, green waste (mostly grass and leaves) from Zhejiang University sieved to 5mm	160-320 (varying FW/YW ratios) 176-272 (varying TS content)	Chen et al., 2014
Food Waste and Yard Waste	Food waste from hopper feeding L-AD system (Quasar, Ohio) and yard waste from OARDC Wooster campus (leaves and branches), milled to 5mm	0-120 (varying S/I ratio and FW/YW ratio)	Brown and Lee, 2013

In a review of the literature on codigestion in AD, Mata-Alverez et al. (2014) concluded that optimal substrate to substrate ratios in codigestion are also dependent on the substrates being digested and on operating parameters. Brown and Li (2013) varied both S/I ratio and food waste/yard waste ratios in HS-AD of OFMSW (19-30% TS), with S/I ratios of 1/1, 2/1, and 3/1, and food waste/yard waste ratios of 0/100, 10/90, and 20/80. Maximum methane yields were achieved at an S/I ratio of 1/1 and a 20/80 food waste/yard waste fraction. Chen et al. (2014) also investigated various food waste/yard waste ratios in HS-AD (15% TS with S/I ratio of 1/1 by VS) and found that specific methane yields increased proportionately with increases in food waste/yard waste ratios but that the optimal ratio, with respect to methane production *rate*, was 40/60 (by VS), with 90% of methane yield observed within 24.5 days of digestion. These studies demonstrated that at high food waste/yard waste ratios and/or high S/I ratios, digesters can be overloaded resulting in decreasing pH and inhibition of methane production (see section 2.1.4.6. for additional information on overloading).

2.1.4.3. Particle Size and Feedstock Porosity

Reduction of substrate particle size, or comminution, has been shown by numerous researchers to significant increase substrate biodegradability and methane yields from solid wastes in AD systems (Sharma et al., 1988; Delgenes et al., 2002; Kaparaju et al., 2002; Bruni et al., 2010; Izumi et al., 2010; Kreuger et al., 2011). Reducing the particle size increases the specific surface area of a substrate which leads to increased hydrolysis rates and enhanced biogas production rates, especially when dealing with substrates for which hydrolysis is the rate limiting step (i.e. lignocellulosic wastes) (Sharma et al., 1988; Izumi et al., 2010; Veeken, 2014). This leads to enhanced energy recovery efficiencies and also reduces reactor volume requirements by reducing pore space (Gollakota and Meher, 1988; Moorhead and Nordstedt, 1993). As a result of these findings, many HS-AD technologies incorporate some form of size reduction (e.g. grinder) before loading wastes to reactors (see Section 2.3.3.). Often times, substrate particle size is reduced to 40 mm maximum particle size or less (Veeken, 2014). However, in HS-AD systems that incorporate percolate recirculation, the presence of large particles (greater than 40 mm particle size) is essential for maintaining adequate porosity of the waste mixture such that mass transfer efficiency through percolate recirculation isn't impeded (Veeken, 2014). Thus, these systems often add large particles from incoming waste and/or recycle large particles from digestate or compost retained in the trommel process to incoming wastes to improve the "structure" of feedstock and guarantee sufficient permeability (Veeken, 2014). For example, the Venlo (The Netherlands) Attero 2-stage AD facility, digestate is composted and then sieved over 40 mm and 15 mm, the fraction that is less than 15 mm is sold as compost and the remaining fractions are mixed with incoming waste as structure material (Veeken, 2014). It is worth noting that at this facility the fraction of degradable organic matter in the digestate is considered an important parameter for reaching sufficient temperatures during aerobic curing (Veeken, 2015).

It is generally believed that the smaller the average particle size of a substrate, the greater the performance in AD with respect to methane yields, as demonstrated by Sharma et al. (1988). However, in a recent study conducted by Izumi et al. (2010), it was demonstrated that in the case of AD of food waste, over-reduction of particle size can lead to VFA accumulation and reduced methane yields. In the study, a 22% increase in methane yields was achieved through reducing the average particle size of food waste from 100 mm to 2 mm and an additional 28% increase in methane yields were observed when reducing the average particle size to 0.7 mm. However, when the particle size was reduced to 0.4 mm, VFA concentrations increased to 5,600 mg/L resulting in significantly lower methane yields. The authors conclude that with highly degradable wastes, such as food waste, an optimal particle size exists that maximizes methane yields without risking inhibition. Optimization of particle size, especially in percolate-recirculating HS-AD systems, is a parameter that deserves further investigation at various scales (Veeken, 2015).

2.1.4.4. Temperature

Temperature is a significant parameter in all biochemical reactions. The two common operational temperature ranges of commercial AD systems are 35-40 °C (mesophilic) and 50-55

°C (thermophilic), although AD microorganisms are capable of surviving temperatures ranging from 0-82°C (Amani et al., 2010). Mesophilic digesters are currently more common in HS-AD applications because they are considered more stable (less sensitive to toxicants, process fluctuations, and temperature variations) and require less energy input for heating than thermophilic digesters (De Baere and Mattheeuws, 2014). However, it has been established that thermophilic digestion improves digestion efficiency (del Real Olvera and Lopez-Lopez, 2012) and in recent years, thermophilic HS-AD has been proven reliable in numerous full-scale systems, yielding 30-50% increases in biogas production rates compared with rates observed in mesophilic digestion (De Baere and Mattheeuws, 2014). Thus, thermophilic digestion is becoming the preferred mode of operation, especially in HS-AD, where heating plays a smaller role in the net energy balance (as compared to L-AD) and in warm climates where less energy input is required for heating to thermophilic ranges (Amani et al., 2010; De Baere and Mattheeuws, 2014). In addition to increased digestion rate, thermophilic digestion provides the advantage of improved pathogen reduction compared with mesophilic digestion (Li et al., 2011).

2.1.4.5. Retention Time

Retention time in L-AD is separated into two components, hydraulic retention time (HRT, or liquid residence time) and solids retention time (SRT, or cell residence time) (Amani et al., 2010). It is important in L-AD systems to decouple the SRT (average time that the microorganisms are present in the reactor) from the HRT because of the slow growth rates of anaerobic microorganisms and the demand for treating large volumes of water quickly and economically (Kato et al., 1994). In HS-AD systems, however, retention time is generally expressed as a single value (i.e. retention time = HRT = SRT) (Cecci et al., 1988). In industry for both batch and continuous single-stage systems, the retention time is simply defined as the average amount of time that the solid waste material is digested and there is no need for decoupling microbial residence times from waste residence times (Kothari et al., 2014). For multi-stage HS-AD systems, other retention time conventions may be used (e.g. residence time in stage-one, residence time in stage-two) (Veeken, 2014). For single-stage batch HS-AD systems with percolate recirculation, SRT and HRT can be decoupled with the SRT describing the time that the solid waste material is digested and the HRT describing the average amount of time that the percolate remains in the system. However, the volume of percolate in these systems is regulated such that it does not change significantly over time and therefore, the HRT is equal to the SRT, which is approximately equal to the system volume divided by the volumetric throughput of feedstock (ZWE, 2015). In cases of percolate depletion, water can be added to incoming feedstock to replenish percolate in the system (Veeken, 2014). In cases when percolate is accumulating, excess percolate can be removed from the percolate storage tank and added to compost (evaporation occurs) (ZWE, 2015). However, the HRT of percolate in these systems will change according to the following equation if the volume of percolate in the system is not adequately regulated (see Appendix C.3. for the control volume and derivation):

$$\Delta HRT_{percolate} = V_{system}/[(Q_{in} \times SG_f \times MC_f) - (Q_{out} \times SG_d \times MC_d) + q_{in} - q_{out}] \qquad \text{(Eq. 2.7)}$$

where $\Delta HRT_{percolate}$ is the change in average retention time of percolate in the system (days), V_{system} is the volume of the system including the digesters and the percolate storage and recirculation system (m^3), Q_{in} is volumetric loading rate of the feedstock (m^3/d), SG_f is the specific gravity of the feedstock (density relative to the density of water), MC_f is the moisture content of the feedstock (% by mass), Q_{out} is the volumetric removal rate of digestate, SG_d is the specific gravity of the digestate, MC_d is the moisture content of the digestate (% by mass), q_{in} is the volumetric loading rate of dilution water (m^3/day), and q_{out} is the volumetric removal rate of percolate from the percolate storage tank (m^3/day). It is worth noting here that optimization of percolate recirculation rates and variation of rates over digestion cycles is an area of research that warrants further investigation, as described by Veeken and Hamelers (2000).

Because long retention times are required to hydrolyze slowly degrading fraction of the feedstock, AD systems are normally designed to optimize economics (Cecci et al., 1988). Increasing retention time increases reactor volume requirements for a given volumetric loading rate. Due to the exponential nature of biogas generation in AD, systems are often designed such that the operational retention time optimizes energy recovery efficiency. For example, a batch system can be designed so that digestion ends when the biogas production rate decreases below a certain percentage of the maximum (Rapport et al., 2008). Thermophilic systems generally require shorter retention times than mesophilic systems and L-AD systems require shorter retention times than HS-AD systems (Tchobanoglous et al., 2003; Rapport et al., 2008). Retention times in full-scale HS-AD systems range from 10 days to 30 days, depending on the system and the feedstock being processed (PIS, 2008; Kothari et al., 2014), which is comparable to retention times in L-AD systems. Retention times for various HS-AD systems are discussed further in Section 2.3.

2.1.4.6. Organic Loading Rate

OLR is expressed in units of mass of VS or COD loaded to the digester per unit volume per unit time (e.g. kg VS/m^3-d), and is calculated as follows:

$$OLR = \%VS \ x \ Q_{in} \ x \ \delta / V \qquad (Eq. 2.8)$$

where %VS is the percent volatile solids fraction of the feedstock by weight, δ is the density of the feedstock (kg/m^3), and V is the volume of the digester (m^3). It should be emphasized that %VS, density, moisture content, and all other physical and chemical parameters vary with time during the digestion process.

OLR is an important operating parameter, especially in continuous systems, as it is representative of the raw substrate available to microorganisms and is directly related to production rate of intermediate compounds (VFAs, H_2, etc.), biogas generation, and overall performance, cost, and stability (Chen et al., 2014). When OLR is very low and increases, biogas generation rate increases. However, beyond a certain threshold, biogas generation plummets with increases in OLR as a result of inhibition due to accumulation of intermediates such as VFAs and H_2 (Vandenburgh and Ellis, 2002), which is discussed below. Some studies suggest HS-AD systems have greater capacity to handle higher OLR as compared to L-AD systems (Schievano et al., 2010) due to reduced rate of mass transfer of toxicants (Vandevivere et al., 2002). Maximum organic loading rate varies significantly depending on feedstock biodegradability, but is typically cited to be 5 kg VS/m^3-d for mesophilic AD and 8 kg VS/m^3-d for thermophilic AD (Zupančič and Grilc, 2012). Schievano et al. (2010) suggest that in HS-AD, the *putrescibility* (short-term digestibility) is a particularly critical indicator of potentially inhibitory OLRs and that with highly putrescible waste (e.g. food waste) inhibitory OLR's can be lower than typically cited values.

2.1.4.7. Volatile Solids Reduction and Methane Yield

VS reduction and methane yield are the most direct measures of digester performance and the degree of digestion in AD systems. VS reduction is calculated on a percent by mass basis as follows:

$$VS\ Reduction\ (\%) = (VS_i - VS_f) / VS_i \qquad (Eq.\ 2.9)$$

where VS_f is the final mass of VS present in the digestate (kg) and VS_i is the initial mass of VS present in the feedstock (kg).

Methane yield can be expressed in terms of volumetric gas flow rate (m^3 CH_4/d) or rate of gas production (m^3 CH_4/m^3reactor-d) but is best expressed as specific methane yield, which is the volume of methane produced normalized by the mass of VS or COD loaded to the digester (L CH_4/kg VS or m^3 CH_4/kg COD). Reporting methane yields as opposed to biogas yields is preferred because methane yield is a more direct measure of energy recovery potential. Specific methane yield is calculated as follows:

$$\textit{Specific Methane Yield} = V_{CH4} / VS_i \quad \text{(Eq. 2.10)}$$

where V_{CH4} is the cumulative volume of CH_4 generated from a known mass of feedstock (L).

Expressing methane yields on a per mass VS basis is preferred over expressing methane yields on a per mass COD basis, especially in HS-AD, because COD concentrations measured from a solid material depend on the solubility of the COD in the material, which can be significantly altered during digestion (Khanal, 2008). Both VS reduction and methane yield are a function of feedstock biodegradability and operating parameters. The maximum VS destruction in full-scale AD systems is generally cited as 60%, but decreases depending on feedstock composition (presence of recalcitrant compounds) (Kaparaju and Rintala, 2005). Some laboratory experiments have achieved up to 90% VS reduction in HS-AD of food waste (Cho et al., 1995).

2.1.4.8. Carbon to Nitrogen Ratio

C/N ratio is one of the most critical parameters pertaining to AD performance and is defined as:

$$C/N = C_{TC} / C_{TN} \quad \text{(Eq. 2.11)}$$

where C_{TC} is the concentration of total carbon present in a feedstock (mg/L) and C_{TN} is the concentration of total nitrogen present in a feedstock (mg/L).

C/N ratios in the range of 20/1and 30/1 are considered good, with ~25/1 being optimal (Li et al., 2011; Kothari et al., 2014). C/N ratios below this range contain too much nitrogen relative to the amount of carbon present resulting in high ammonia release and increased likelihood of ammonia inhibition (Yang et al., 2015). C/N ratios above this range can lead to total uptake of available N by methanogens, resulting in biogas decreases due to insufficient N availability and/or excessive VFA production due to inhibition of methanogens from a lack of N for cell synthesis (Yang et al., 2015). The bioavailability of carbon and nitrogen compounds present in a given substrate affects optimal C/N ratio. Optimal C/N ratio may also be a function of other operating parameters, such as pH and temperature (Yang et al., 2015). Codigestion strategies are often employed to balance C/N ratios in HS-AD of OFMSW. For example, mixed food wastes have C/N ratios ranging from 10/1 – 20/1 (Brown and Li, 2013; Chen et al, 2014) and yard wastes can have C/N ratios greater than 50/1 (Yang et al., 2015). Therefore, codigestion of a mixture of food and yard waste can result in a favorable C/N ratio (Rapport et al., 2008; Li et al., 2011; Chen et al., 2014).

2.1.4.9. pH and Alkalinity

Alkalinity is a measure of the buffering capacity of a system to changes in pH. The alkalinity of an AD system is critical to ensure that the pH remains above inhibitory levels as acids are being produced (Amani et al., 2010). Alkalinity is generally expressed in terms of equivalents per liter (eq/L) or as an equivalent concentration as calcium carbonate (mg/L as $CaCO_3$). Alkalinity is mathematically defined as:

$$Alkalinity\ (eq/L) = [HCO_3^-] + 2[CO_3^{2-}] + [OH^-] - [H^+] \qquad (Eq.\ 2.12)$$

where $[HCO_3^-]$ is the molar concentration of bicarbonate ions in a solution (eq/L), $[CO_3^{2-}]$ is the molar concentration of carbonate ions in a solution (eq/L), $[OH^-]$ is the molar concentration of hydroxide ions in a solution (eq/L), and $[H^+]$ is the molar concentration of hydrogen ions in a solution (eq/L). In laboratory setting, it can be calculated as follows:

$$Alkalinity = 0.1 \times 50{,}000\ \Delta V_{HCL} / V_{sample} \qquad (Eq.\ 2.13)$$

where ΔV_{HCL} is the change in volume (mL) of 0.1 normal (N) hydrochloric acid (HCl) solution required to titrate a given volume of sample from its equilibrium pH to a pH value of 4.3 and V_{sample} is the volume of sample that the HCl solution was added to. The target pH of the titration is 4.3 because that is the theoretical pH at which all of the carbonate alkalinity (HCO_3^- and CO_3^{2}) is consumed (converted to hydrochloric acid, H_2CO_3, through reaction with hydrogen ions) (Crittenden et al., 2012).

As discussed previously, the symbiotic relationship between fermenters and methanogens aids in maintaining a healthy pH and alkalinity balance, with methanogens consuming acids produced by the fermenters and producing alkalinity (see Section 2.1.3). A pH greater than 6.5 and alkalinity greater than 1,000 mg/L as $CaCO_3$ is recommended to ensure that methanogenic populations are not inhibited (Fabián and Gourdon, 1999; Tchobanoglous et al., 2003; del Real Olvera and Lopez-Lopez, 2012). For two-stage AD, the optimal pH range for the acid phase is 5.2-6.3 and the optimal pH range for the gas phase is 6.7-7.5 (Deublein and Steinhauser, 2008). However, high pH values being reported as a concern with respect to AD process efficiency are rare except for in select cases where increased pH has been reported to contribute to free ammonia inhibition (Chen et al., 2008).

It is worth noting that the alkalinity and pH in a given AD system are affected by concentrations of CO_2 in the headspace of the digester (in the biogas). As the CO_2 concentrations in the headspace increase, the partial pressure of CO_2 increases and concentrations of CO_2 as carbonic acid (H_2CO_3) in solution increase, resulting in reduced pH. As alkalinity is depleted, the pH will begin to plummet and digester failure will follow. Ideally, enough alkalinity will be provided by the substrate(s) and inocula and alkalinity levels will remain sufficient throughout the digestion process. However, in the case that alkalinity becomes depleted (e.g. when fermentation is significantly outpacing methanogenesis and VFAs are accumulating), additional alkalinity sources can be added to prevent acidification of a digester. Alkalinity sources that are commonly used include lime (calcium hydroxide), soda ash (sodium carbonate or sodium bicarbonate), and limestone (calcium carbonate) (Tchobanoglous, 2003). A less conventional, but naturally generated source of alkalinity that may be especially suitable for use as an alkalinity source in AD due to its slow dissolution rate is oyster shells (Sengupta et al., 2007). Additionally, oyster shells are a waste material, which, if utilized to provide alkalinity to an AD system would have a newfound beneficial reuse. The effectiveness of this potential strategy, however, is not well-studied, but is discussed briefly in Section 4.5 and in Appendix Section E.2.

2.1.4.10. Volatile Fatty Acids

VFAs are the most important and descriptive intermediate chemicals produced in the AD process (Li et al., 2011). As shown in Figure 2.1, VFAs are naturally produced by acidogens and acetogens and then consumed by methanogens in the AD process (Adekunle and Okolie, 2015). Accumulation of VFAs can therefore be viewed as a measure of the differences in the rates of VFA production by acidogens/acetogens and consumption by methanogens. Because hydrolysis is the preceding metabolic step of the AD process, VFA accumulation is an indicator that methanogenesis is the rate-limiting step rather than hydrolysis. Reductions in VFA concentrations, on the other hand, indicate that methanogenesis is occurring at a more rapid pace than acidogenesis and acetogenesis, an indication that hydrolysis is the rate-limiting step. In the case of batch AD processes, decreases in VFA concentrations can also be an indicator that bioavailable raw substrate (e.g. proteins, carbohydrates, lipids) has been metabolized and the AD process is coming to an end (Mussoline et al., 2013). Total VFA concentrations are typically expressed in terms of acetic acid equivalents because acetic acid is the most common VFA produced in natural systems (Tchobanoglous et al., 2003). VFA concentrations greater than 10,000 mg/L as acetic acid are generally considered inhibitory to methanogenesis (Amani et al., 2010). However, methanogens have been acclimated to survive in high-VFA environments and certain populations are prone to inhibition at lower concentrations (Khanal, 2008).

2.1.4.11. Ammonia, Hydrogen Sulfide, and Other Inhibitors

Ammonia accumulation is another concern with respect to inhibition in AD systems (Chen et al., 2008). Ammonia inhibition of methanogens can lead to an accumulation of VFAs, which compounds the inhibition (Chen et al., 2008). The principle compound of concern is free ammonia, which increases with increasing total ammonia nitrogen (TAN) concentration, temperature, and pH (Yenigün and Demirel, 2013); however, TAN concentration values are most commonly measured and reported in the literature than free ammonia. Reported threshold ammonia inhibition concentrations vary widely, and depend on feedstock and system characteristics (Yenigün and Demirel, 2013). Chen et al. (2008) reported that TAN concentrations greater than 1,500 – 1,700 mg/L are considered inhibitory. Kayhanian et al. (1994) found inhibition began at a TAN concentration of 1,000 mg/L for HS-AD of OFMSW and found optimal performance between 600-800 mg/L. It is generally believed that TAN concentrations greater than 0 mg/L but less than 200 mg/L are beneficial for the AD process because ammonia is an important nutrient for cell synthesis (Liu and Sung, 2002). Ammonia inhibition is a particular concern in the digestion of manures and is less common in AD of OFMSW (Yenigün and Demirel, 2013).

H_2S inhibition can be just as catastrophic as VFA and ammonia inhibition. The presence of sulfate (SO_4^{2-}) in feedstocks and/or inocula results in the production of sulfides (H_2S and HS^-) in AD, which can accumulate and become toxic to the acetoclastic methanogens (Chen et al., 2008). At neutral pH, H_2S concentrations greater than 200 mg/L have been reported as inhibitory (Gerardi, 2003). Measuring input COD to sulfate ratios (COD/ SO_4^{2-}) is an effective strategy for protecting AD systems against sulfide inhibition (Chen et al., 2008). Input COD/ SO_4^{2-} ratios greater than 2.7 are unlikely to result in problems with sulfide inhibition (Chen et al., 2008). Another concern is that production of H_2S by sulfur reducing bacteria (SRBs) is more thermodynamically favorable than the production of methane by CO_2 reducing methanogens (Tchobanoglous et al., 2003; Chen et al., 2008). Thus, SRBs will outcompete CO_2 reducing methanogens for available H_2 and for every two moles of H_2S generated one less mole of CH_4 will be generated.

A number of other substances have been shown to inhibit AD, including various salts, organic compounds, and metals (Chen et al., 2008; Zupančič and Grilc, 2012; Deublein and Steinhauser, 2008). Optimal, moderately inhibitory, and inhibitory concentrations of common inorganic salts are shown in Table 2.3 (Zupančič and Grilc, 2012). High salinity inhibition is more commonly cited in HS-AD than L-AD because the use of percolate recirculation in HS-AD can result in steadily increasing salt concentrations. Several trace metals are essential for AD, especially for methanogenesis, but these elements can also be inhibitory at high concentrations (Deublein and Steinhauser, 2008). A list of trace metals, their minimum required concentrations, and their inhibitory concentrations are shown in Table 2.4 (Deublein and Steinhauser, 2008).

Table 2.3: Optimal, moderately inhibitory, and inhibitory concentrations of inorganic salts in AD (adapted from Zupančič and Grilc, 2012).

Inorganic Salt	**Optimal Concentration** *(mg/L)*	**Moderate Inhibition** *(mg/L)*	**Severe Inhibition** *(mg/L)*
Sodium (Na)	100-200	3500-5500	16000
Potassium (K)	200-400	3500-4500	12000
Calcium (Ca)	100-200	2500-4500	8000
Magnesium (Mg)	75-150	1000-1500	3000

Table 2.4: Micronutrients requirements in AD and potentially inhibitory concentrations (adapted from Deublein and Steinhauser, 2008).

Micronutrient	Minimum Required Concentration *(mg/L)*	Inhibition Concentration as Ion *(mg/L)*	Inhibitory Concentration as Carbonate *(mg/L)*
Chromium (Cr)	0.005-50	28-300	530
Iron (Fe)	1-10	N/A	1750
Nickel (Ni)	0.005-0.5	10-300	N/A
Copper (Cu)	> 0	5-300	170
Zinc (Zn)	> 0	3-400	160
Lead (Pb)	0.02-200	8-340	N/A
Cobalt (Co)	0.06	N/A	N/A
Molybdenum (Mo)	0.05	N/A	N/A
Selenium (Se)	0.008	N/A	N/A
Manganese (Mn)	0.005-50	1500	N/A

2.2. MSW Management and HS-AD in OFMSW Management

In this section, MSW and OFMSW are defined, MSW and OFMSW generation rates and management technologies are introduced in greater detail, and literature relevant to Research Questions 1, *What is the state of the art of HS-AD?*, and 2, *What is the outlook for implementation of HS-AD in the state of Florida?*, is reviewed, including studies pertaining to environmental, economic, and policy considerations of HS-AD implementation.

2.2.1. MSW, OFMSW, and Generation Rates

MSW can be defined as all substances and items that are discarded into trash cans and dumpsters from residential (households, apartments, etc.), commercial (businesses, hotels, restaurants, etc.), institutional (hospitals, schools, etc.), and industrial (packaging plants, processing plants, etc.) sources. MSW includes "product packaging, newspapers, office and classroom paper, bottles and cans, boxes, wood pallets, food, grass clippings, clothing, furniture, appliances, automobile tires, consumer electronics, and lead-acid batteries" (EPA, 2015a). Solid wastes that are not considered MSW include agricultural (livestock wastes and crop residues), process wastes from industrial sources (i.e. byproducts), construction and demolition debris

(C&D), and biosolids (sewage sludge) (EPA, 2015a). OFMSW is implicitly defined by the World Bank (2012) and US EPA (2015a) to include yard wastes (grass, leaves, needles, and tree and brush trimmings, including those removed from streets) food wastes, and wood wastes. According to these definitions, paper and paperboard wastes are not OFMSW.

As of 2012, the world's cities generated approximately 1.2 billion tons of MSW per year, 46% of which was organic (World Bank, 2012). The total cost of managing this waste was estimated at around $205B and is expected to increase to $375B by 2025 with generation rates reaching 2.2 billion tons per year (World Bank, 2012). OFMSW accounts for around 34% of all MSW generated in the US (Figure 2.2), with food waste (14.6%) and yard trimmings (13.5%) together accounting for more than 28% (EPA, 2015a). In the US in 2013, approximately 250 million tons of MSW, 34 million tons of yard waste, and 37 million tons of food waste were generated (EPA, 2015a). Of all the waste that is discarded in the US, a substantial fraction is organic (> 38%), highlighting the need for OFMSW recovery technologies. The approximate categorized percentage by mass of waste generated in the US and of waste disposed in the US in 2013 (167 million tons of MSW) are shown side-by-side in Figure 2.2 (EPA, 2015a). A breakdown of annual MSW generation, recovery, and discards in the US from 1960 to 2013 is shown in Figure 2.3 (EPA, 2015a). This figure demonstrates that waste generation is beginning to level off and that the recovery of OFMSW (the "composting componenet of recycling") has slowly but steadily increased since the early 1990's. When considering the quantities of OFMSW generated, the amount of nutrients that go into the growth of food and plants, and the energy consumed in the production and transportation of food, it becomes clear that recovery of resources from these wastes is of great importance.

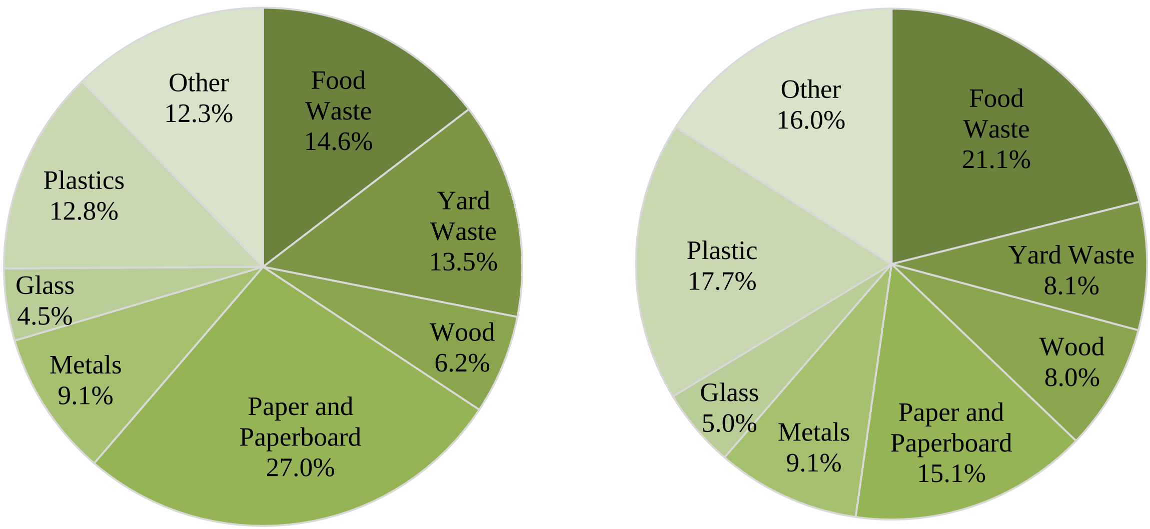

Figure 2.2: Categorized composition of MSW generated (left) and disposed (right) in the US in 2013 (adapted from EPA, 2015a).

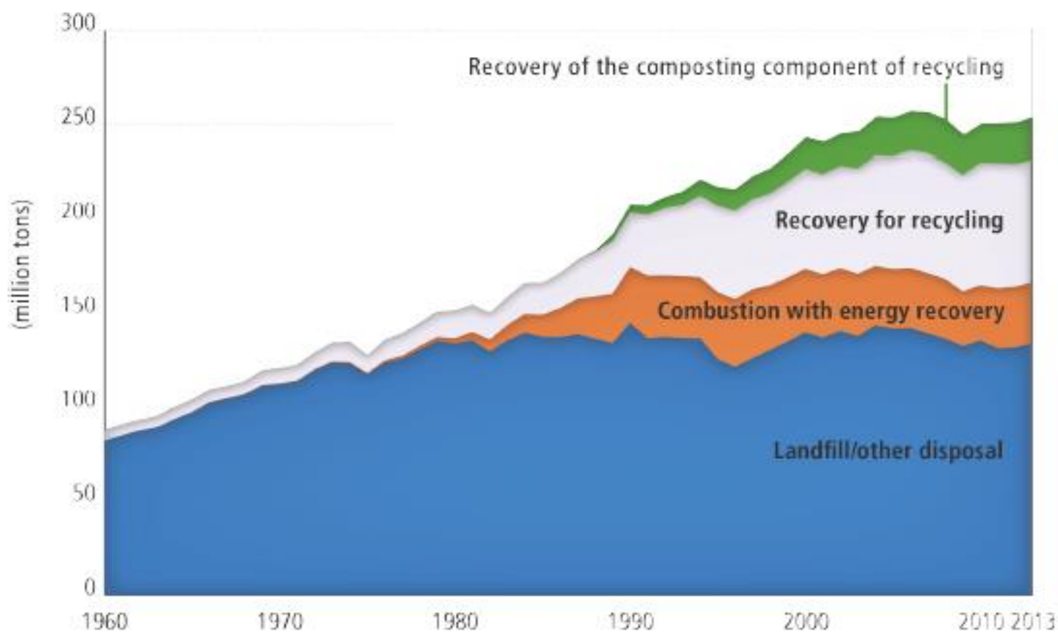

Figure 2.3: MSW generation, recovery, and discards per year from 1960 to 2013 (EPA, 2015a).

2.2.2. MSW and OFMSW Management Technologies

In the majority of locations around the world OFMSW is comingled with the rest of the waste stream (World Bank, 2012). Therefore, it is typically managed with the rest of the waste stream. The most prominent MSW management strategies worldwide are disposal in landfills, incineration, disposal in open dumps, and aerobic composting (World Bank, 2012). In addition to these conventional MSW management technologies, MSW management in industrialized countries is often improved through implementation of Material Recovery Facilities (MRFs), which separate various fractions of MSW from the waste stream and aid in increasing recycling rates (World Bank, 2012). Slightly more advanced Mechanical Biological Treatment (MBT) systems, which integrate MRF separation technologies with biological technologies, such as AD and composting, have also become increasingly common (Bogner et al., 2007).

Other MSW management technologies such as thermal conversion technologies (pyrolysis, gasification, and plasma arc gasification), chemical conversion technologies (e.g. transesterification, hydrogenation, Fischer-Tropsch), and hybrid technologies (combinations of thermal and chemical technologies) are becoming more common for recovering energy from MSW, OFMSW (especially yard waste), and other lignocellulosic biomass (Bogner et al., 2007; Chum et al., 2011). As of 2013, there were 67 AD, 48 gasification, 19 plasma gasification, and 16 pyrolysis companies handling MSW worldwide (EREF, 2013). Table 2.5 shows the evolution of the use of MSW management technologies in the US from 1960 to 2013 in percent by mass of total waste generated per year (EPA, 2015a). A brief description of each technology follows.

Table 2.5: Percent by mass of waste managed via common technologies in the US from 1960 to 2013, including total percentage recovered and discarded (adapted from EPA, 2015a).

Management Method	Percent of Total Generation						
	1960	**1970**	**1980**	**1990**	**2000**	**2009**	**2013**
Total Material Recovery	*6.4%*	*6.6%*	*9.6%*	*16.0%*	*28.5%*	*33.8%*	*34.3%*
Recovered for Recycling[1]	6.4%	6.6%	9.6%	14.0%	21.8%	25.3%	25.5%
Recovered for Composting[2]	0-1%	0-1%	0-1%	2.0%	6.7%	8.5%	8.8%
Total Material Discards	*93.6%*	*93.4%*	*90.4%*	*84.0%*	*71.5%*	*66.2%*	*65.7%*
Incineration WtE[3]	0.0%	0.3%	1.8%	14.2%	13.9%	11.9%	12.9%
Disposal[4]	93.6%	93.1%	88.6%	69.8%	56.6%	54.4%	52.8%

[1]Includes source separated recyclables and those recovered in Material Recycling Facilities

[2]Includes yard trimmings, food waste, and other organic material; does not include backyard composting

[3]Includes comingled MSW, wood, and tires (29.5, 0.5, and 2.6 million tons in 2013, respectively)

[4]Includes landfilling and incineration without energy recovery

2.2.2.1. Landfills

Landfills range in complexity, but typically involve the use of heavy machinery to compact and cover MSW in daily cells (Tchobanoglous et al., 1993). Landfills normally have many “lifts” (levels) to maximize the waste disposal capacity of a given land area (Tchobanoglous et al., 1993). In most industrialized countries, landfills are equipped with internal piping for the collection of biogas that is produced as the organic material in the landfill undergoes AD within the covered daily cells (van Haaren et al., 2010). This allows for energy recovery from landfills (LFGTE), but is usually inefficient due to poor biogas quality and high fugitive biogas emissions (biogas escaping from the landfills) (World Bank, 2012). Landfills in developing countries, on the other hand, are difficult to sustain economically, rarely have biogas capture, and although they are a step in the right direction from open dumps, they often have substantial environmental and public health impacts (Oakley, 2005). The diversion of OFMSW from landfills may reduce energy recovery rates from landfills that are LFGTE equipped, but provides benefits that outweigh these reductions, including reduced fugitive GHG emissions, reduced leachate generation, and shortened landfill closure periods (Tchobanoglous, 1993; World Bank, 2012). According to the EPA, “the promotion of LFG energy is not in conflict with the promotion of organic waste diversion” (EPA, 2015d).

2.2.2.2. Incineration

Incineration is the combustion of waste material. In industrialized countries, incineration is a highly technical and mechanized process, which incorporates electricity generation, emissions control, and regulated ash management (Tchobanoglous, 1993; World Bank, 2012). This form of incineration is often referred to as WtE, Energy from Waste (EfW), or mass burn (EREF, 2013). Electricity is generated by using the heat from the combustion of the waste to convert water to steam and then passing the steam through steam turbines (Chum et al., 2011). Emissions are monitored and controlled using a range of air pollution control technologies and are often regulated locally and/or federally (World Bank, 2012). Ash produced in the process is disposed in specialized landfills or reused in industrial processes (e.g. concrete production) (Chum et al., 2011). Incineration in developing countries is often low-tech and unregulated yielding substantial environmental and public health impacts (Oakley, 2005; World Bank, 2012). Incineration of MSW without energy recovery, but still with regulated emissions control, is also carried out in developed countries in some cases as a low cost method for substantially reducing waste volume and saving landfill space (World Bank, 2012). Food waste and yard wastes both have relatively high moisture contents and low calorific values (energy as heat generated per unit mass incinerated) when compared to other feedstocks such as plastics, and therefore, are less suitable feedstocks for WtE (Franjo et al., 1992; Owens and Chynoweth, 1993).

2.2.2.3. Open Dumps

Open dumps are the most common form of MSW (and OFMSW) waste management in developing countries (Oakley, 2005; World Bank, 2012). They are classified as controlled, semi-controlled, and uncontrolled, with associated environmental pollution decreasing with increases in the level of control at a specific site. Runoff from open dumps contaminates surface waters and aquifers and the prevalence of disease transmitting vectors increases significantly where open dumps are present (Oakley, 2005; World Bank, 2012). Poorly maintained landfills and open dumps in developing countries will often catch fire and burn continuously for years (Oakley, 2005). "Scavenging" in open dumps is a common practice in developing countries, which improves recycling rates, but also increases public health and safety risks by exposing individuals directly to poor air quality (from open burning), hazardous chemicals, pathogens, and pathogen vectors (Oakley, 2005). Open dumps are the least preferred waste management strategy (Figure 1.1) and are the most threatening to environmental and public health.

2.2.2.4. Separation of OFMSW, MRFs, and MBT Plants

OFMSW is generally comingled and managed with the rest of the MSW stream, as previously mentioned, except in certain locations where cultural, economic, or legislative factors encourage source-separation of OFMSW and other recyclable fractions (World Bank, 2012; EPA, 2015a). In industrialized countries where source-separation is lacking, technologies such as MRFs (also sometimes referred to as transfer stations) and MBT have become common for separating various fractions of MSW waste from comingled waste streams to increase resource recovery (World Bank, 2012; EPA, 2015a). In these facilities, recyclables such as metals, plastics, and glass are mechanically separated, sometimes in combination with manual separation, using various technologies such as conveyor belts, industrial magnets, eddy current, trommel screens, shredders, and sometimes water for separation of materials by density/buoyancy (Tchobanoglous et al., 1993). MRFs generally have minimal onsite processing, dealing only with the separation and preparation of materials for subsequent collection for recycling or disposal (of the nonrecyclables) (World Bank, 2012; EPA, 2015a). After all of the recyclables are removed from comingled MSW, only mechanically- separated OFMSW (MS-OFMSW) is left and it is typically transported and landfilled (Bogner et al., 2007). MBT plants, on the other hand, are equipped with similar separation technologies as MRFs, but also have onsite biological treatment, such as composting or AD, for processing MS-OFMSW (Tchobanoglous et al., 1993; Bogner et al., 2007). After composting or digestion, the material is stabilized but still must be transported and landfilled because it is contaminated and not suitable for use as a fertilizer or soil amendment (Lens et al., 2004).

2.2.2.5. Aerobic Composting

Aerobic composting is the facilitated biodegradation of OFMSW under aerobic conditions (in the presence of oxygen). On the backyard scale, it is becoming increasingly common as environmental and economic (often stemming from legislation) incentives have become more prevalent (e.g. awareness of the environmental impact of OFMSW management, increased cost of trash collection and disposal, and increased cost of fertilizers and soil amendments) (van Haaren et al., 2010; EPA, 2015a). Small composting bins or piles are most often used by backyard gardeners and aeration is either done through mixing with a shovel or through the use of small in-vessel composting bins designed for easy turning.

On the municipal level, composting has also become more common in recent years for reasons similar to those cited in the case of backyard composting. A wide variety of composting strategies are used, including windrows, in-vessel systems, aerated static piles, and vermicomposting, with windrow composting being the most common (van Haaren et al., 2010; EPA, 2015a). Large-scale aerobic composting is sometimes viewed as a competing technology with HS-AD with respect to management of OFMSW. However, as demonstrated in the development of HS-AD in Europe, aerobic composting and HS-AD are complementary technologies (De Baere and Mattheeuws, 2014). HS-AD reduces waste volumes and composting time requirements and composting enhances the quality of digestate from HS-AD for marketing as an organic fertilizer or soil amendment (Rapport et al., 2008; Li et al., 2011). These synergistic characteristics of composting and HS-AD are reflected in the prevalence of implementation of HS-AD at existing composting operations. For example, approximately 80% of composting operations in the Netherlands and Belgium have incorporated AD, the majority of which is HS-AD, as a primary treatment technology (De Baere and Mattheeuws, 2014).

2.2.2.6. Thermal and Chemical Conversion Technologies

Pyrolysis and Gasification are two relatively well-developed thermal technologies that are becoming increasingly prevalent on the commercial-scale. These technologies, often referred to as ATT technologies, are somewhat in competition with HS-AD with respect to management of lignocellulosic wastes such as yard wastes and agricultural crop residues (Deublein and Steihauser, 2008). Pyrolysis is defined as the thermal decomposition of biomass under anaerobic conditions resulting in the production of charcoal (or bio-char), pyrolysis oil (or bio-oil), and syngas (or "synthetic gas" composed of hydrogen, carbon monoxide and CO_2) (Chum et al., 2011). Commercial pyrolysis is carried out at 450 ºC to 550 ºC and produces 70-80% pyrolysis oil, which can be further converted or combusted to produce heat and power (Chum et al., 2011).

Gasification is an aerobic process involving controlled aeration under very high temperatures (>700 ºC) (Chum et al., 2011). The result of the process is the direct conversion of most of the material to syngas, which can then be used to produce electricity via gas turbines, boilers and steam turbines, or fuel cells. The syngas can also be used to produce heat and power in suitably designed CHP units, as a fuel in place of diesel in adapted internal combustion engines (Chum et al., 2011), or it can be further converted to hydrogen or methanol (e.g. via Fischer-Tropsch process). Gasification is most often applied for conversion of woody materials, but can also be used for non-woody wastes (Yokoyama and Matsumura, 2008). Compared to combustion, gasification is a more efficient energy recovery process (Kirkels and Verbong, 2011). Plasma arc gasification is a variation of gasification that uses an external (plasma) heating source to produce a higher quality syngas and is theoretically more efficient than traditional gasification, but is difficult to scale up (EREF, 2013). There are also hybrid versions of ATTs, in which gasification and pyrolysis are used in series in a two-stage process or used in combination with fermentation for ethanol production and/or chemical conversion technologies (described below). It's worth noting that these technologies require low moisture content feedstocks, with gasification typically requiring less than 10% moisture and pyrolysis typically requiring less than 20% moisture (EREF, 2013). Therefore, food wastes and other wet wastes are generally not a suitable feedstock or require drying.

Other technologies that are used for recovering energy from OFMSW, especially the yard and wood waste fraction, include fermentation for ethanol production and chemical technologies such as transesterification or hydrogenation for biodiesel production (Chum et al., 2011). In terms of theoretical output/input energy ratio, biogas production via AD is considered to be a most efficient method for recovering energy from lignocellulosic biomass when compared to other biochemical, chemical, and thermal conversion technologies, with the maximum biogas volume (output) generated from biomass input with an equivalent energy of one mega-Joule (MJ) having an average thermal value (output) of 28.8 MJ (a 28.8/1 ratio) (Deublein and Steihauser, 2008). However, as a result of the historically limited biodegradation of lignocellulosic materials achieved in HS-AD (see section 2.4.), innovative recycling configurations for these materials, which integrate several different processes to maximize resource recovery are being explored. Sawatdeenarunat et al. (2015), for example, proposed unique integrated recycling strategies for lignocellulosic materials that merge numerous technologies including AD, algae production, enzymatic saccharification, thermal conversion, and transesterification. Pan et al., (2015) proposed similar integrated solutions for lignocellulosic materials and proposed strategies for adapting waste management to fit into the framework of a "circular economy system" and solving the contradictory relationship between "greening and growth".

2.2.2.7. Tools for Improving MSW Management

The US EPA has dedicated considerable efforts to developing programs and tools for states, counties, and communities to facilitate waste reduction and improving the sustainability of MSW management. One example of a successful program that has been promoted and developed in part by the EPA is the "pay-as-you-throw" (PAYT) program, which encourages waste reduction from residential, commercial, institutional, and industrial sectors by charging for MSW collection on a per-weight or per-volume basis instead of charging a flat rate collection fee like conventional MSW programs (EPA, 1997). Over 2,000 communities across the country had PAYT programs as early as 1997, resulting in an average reported reduction in waste generation of 25-35% (EPA, 1997). The EPA provides several case studies of successful PAYT programs, a list of references pertaining to reducing waste generation rates and improving MSW management, and offers a helpline for MSW management officials interested in PAYT (http://www.epa.gov/wastes/conserve/tools/payt/index.htm). In addition to the PAYT program, the EPA provides tools to help MSW managers and decide what programs and models are best for their community (i.e. SMART BET), to enable efficient evaluation of waste management strategies with respect to GHG emissions (i.e. WARM model), to guide permitting processes for food waste AD projects (HWMA, 2013), and to aid individuals and corporations in waste reduction (EPA, 1992; EPA, 2012a).

The EPA also developed, in collaboration with the US Department of Agriculture and Department of Energy, a Biogas Opportunities Roadmap (USDA/EPA/DOE, 2014), which outlines challenges associated with biogas projects and strategies for overcoming them. In the document, the importance of developing biogas projects is summarized under five headings: provide a renewable source of energy, cut methane emissions, protect the environment, enhance resilient communities, and boost the economy. Although the Roadmap is tailored toward AD in agriculture, it reaffirms the dedication of US agencies to promote biogas projects, including HS-AD. A number of other tools have been developed by local, state, and international agencies. UNEP, for example, has released several volumes of Integrated Solid Waste Management Plan Training Manuals that are available on their web page (UNEP, 2009).

2.2.3. Environmental Considerations of HS-AD in OFMSW Management

OFMSW management contributes significantly to several environmental impact categories of critical concern, including climate change and eutrophication. An estimated 1-5% of total GHG emissions in the US result from waste degradation and waste management practices (EPA, 2009) and on a global scale, degradation of waste in open dumps and landfills accounts for 10-12% of methane emissions (World Bank, 2012). Additionally, nutrient loading from leachate from waste is considered a significant point source of nutrients contributing to eutrophication and negatively impacting wastewater treatment plants to which landfill leachates are commonly discharged (Ansari and Gill, 2014; Townsend et al., 2015). Several LCAs have been conducted comparing the environmental impacts of various OFMSW management methods and although there are many inconsistencies in the studies, the results have shown a strong preference toward AD with respect to overall environmental impacts (Haight, 2005; Edelmann et al., 2005; Sundqvist, 2005; Kim and Kim, 2010; CIWMB, 2009; Zaman, 2009; Morris et al., 2011; Levis and Barlaz, 2011; Bernstad and la Cour Jansen, 2012). A 2011 review of the LCA literature compared the impacts of AD, landfilling with flaring, landfilling with LFGTE, incineration WtE, aerobic composting, and home composting. Table 2.6, adapted from the review, summarizes the results of the LCA studies with respect to climate change impacts.

Table 2.6: Climate change data from LCA literature review (adapted from Morris et al., 2011).

Management Method	Studies Reviewed	Minimum	Maximum	Median	Mean
		(Metric Tons CO_2 Equivalents/ Metric Ton Organic Waste)			
Anaerobic Digestion	5	- 0.74	- 0.06	- 0.14	- 0.25
Aerobic Composting	30	- 0.76	0.22	0.04	- 0.07
Subset[1]	11	- 0.76	0.06	- 0.20	- 0.21
Subset[2]	9	- 0.26	0.06	- 0.04	- 0.09
Incineration Waste-to-Energy	9	- 0.24	0.63	- 0.02	0.02
Subset[3]	8	- 0.24	0.02	- 0.03	- 0.06
Home Aerobic Composting	8	- 0.69	0.29	0.14	0.05
Landfill with Gas-to-Energy	9	- 0.31	1.00	0.11	0.16
Subset[4]	7	- 0.31	0.24	- 0.10	- 0.01
Landfill with Gas Flaring	2	- 0.06	- 0.05	- 0.06	- 0.06

[1]Excludes studies that did not include C sequestration in soils or substitution of compost for synthetic fertilizer

[2]Same as Subset 1, but with low outliers excluded

[3]Excludes high outlier

[4]Excludes high outlier

As indicated most clearly in the column on the far right of Table 2.6, the average climate change impact of processing one metric ton of OFMSW, as compared to its degradation in nature, is far less for AD than for any other waste management technology. The mean value of - 0.25 metric tons of carbon dioxide equivalents per ton of OFMSW waste processed suggests that on average, a quarter of a metric ton of carbon dioxide equivalents, primarily in the form of carbon dioxide and methane (approximately 25 times greater global warming potential than carbon dioxide), is offset for each metric ton of OFMSW processed. Comparatively, landfilling and incineration result in added climate change impacts, on average, when compared to natural degradation. This is primarily a result of fugitive methane emissions in the case of landfilling and carbon dioxide and nitrous oxide emissions (approximately 310 times greater global warming potential than carbon dioxide) in the case of incineration (Morris et al., 2011). Table 2.7, also adapted from the review, ranks the technologies with respect to other select impact categories.

Table 2.7: Non-climate impacts of management technologies (adapted from Morris et al., 2011).

Management Method	Human Carcinogenicity	Eco-toxicity	Acidification	Eutrophication[1]	Ground Level Smog
	Average Ranking[2]				
Anaerobic Digestion	1.5 (2)	1.3 (3)	1.0 (2)	3.0 (1)	1.0 (2)
Aerobic Composting	1.2 (5)	1.3 (6)	1.8 (6)	1.6 (5)	2.0 (2)
Incineration Waste-to-Energy	2.4 (5)	2.3 (6)	2.2 (6)	2.3 (4)	1.7 (3)
Home Aerobic Composting	NR	1.0 (1)	1.5 (2)	1.5 (2)	1.0 (1)
Landfill with Gas-to-Energy	1.8 (5)	2.0 (5)	1.8 (5)	2.0 (3)	1.5 (2)
Landfill with Gas Flaring	2.5 (2)	3.5 (2)	2.5 (2)	3.5 (2)	NR

[1]Eutrophication ranking assumes liquid anaerobic digestion without nutrient removal from leachate

[2]The lowest possible ranking is 1.0, higher rankings are relative to the lowest ranking

Numbers in parentheses (#) indicate the number of studies which included that impact category

NR = Not Ranked because that category was not taken into account in any study

The average life cycle impact of AD for each of the categories shown in Table 2.7 is either the lowest or nearly the lowest when compared with other leading OFMSW management technologies. The one exception is the impact of anaerobic digestion on eutrophication. The table indicates that AD is among the worst options with regard to its impact on eutrophication. This is misleading, however, because the one study which addressed the impacts of AD on eutrophication assumed L-AD without nutrient recovery from the wastewater (leachate/percolate) generated in the process (Morris et al., 2011). In this respect, HS-AD is environmentally superior to L-AD for the processing of OFMSW (Table 1.1) due to minimal excess leachate generation and low impact onsite management (recirculation, addition to compost, and evaporation) at most HS-AD facilities (Rapport et al., 2008; Li et al., 2011).

Furthermore, HS-AD paired with composting and enables the efficient recovery of nutrients from SS-OFMSW and results in reduced nutrient loading to wastewater treatment facilities and aquatic ecosystems. (De Baere and Mattheeuws, 2014). Nutrient recovery also offsets the environmental impacts associated with phosphate mining and inorganic nitrogen production via the Haber-Bosch process (GMI, 2014). In an LCA study conducted by Edelmann et al. (2005), energy generation and nutrient recovery through AD of OFMSW and combinations of AD and composting were shown to yield significant environmental impact offsets when compared with energy generation through incineration or nutrient recovery through composting alone. In a study comparing emissions to soil from composting fresh OFMSW versus composting digested OFMSW, emissions of volatile organic compounds (VOCs) and ammonia were found to decrease by a factor of 195 (from 588 grams per ton to 3 grams per ton) and 1.6 (from 159 grams per ton to 98 grams per ton), respectively (De Baere, 1999).

A final consideration that is relevant to the environmental aspects of HS-AD of OFMSW is comparison of the environmental sustainability of AD relative to ATTs such as gasification and pyrolysis. Only one LCA study that directly compared AD with these technologies could be found (Zaman, 2009), the results of which indicate that ATT offers potential to reduce life cycle impacts of MSW management relative to incineration WtE and landfilling, but still incurs significantly greater impacts that AD. However, a 2014 article reviewing 250 case studies on LCAs comparing thermal technologies to incineration WtE and landfills pointed out "critical inconsistencies" in the studies and concluded that more comprehensive analyses are necessary (Astrup et al., 2014). The study further concluded that the impacts of alternative thermal technologies are generally comparable to traditional incineration WtE, depending on the effectiveness of air pollution control systems (Astrup et al., 2014); supporting the findings of Zaman (2009) that AD likely provides additional environmental benefits compared with ATTs.

In conclusion, it is fairly well-established that AD, in general, reliably offers greater potential to enhance the environmental sustainability of OFMSW management than other leading technologies. Furthermore, because of the benefits that HS-AD offers over L-AD with regard to water consumption, wastewater generation, and energy requirements (parasitic energy demand), it can be reasonably deduced that HS-AD is the most environmentally friendly OFMSW management method, especially when feedstocks are of high quality (uncontaminated organics) and HS-AD is used in combination with composting for final curing to improve the quality of the digestate as a soil amendment for nutrient recovery. However, there are a number of inconsistencies in methods, assumptions, and results in LCA studies and research is needed that specifically compares HS-AD with other MSW management technologies. According to a recent study developed by the US EPA (2012b), an examination of sensitivities and "break-even" points relative to costs and environmental aspects should be researched in the near future and should consider key parameters, such as feedstock composition, energy conversion efficiency, recovery of materials for recycling, beneficial offsets for end-product alternatives, distance to liquid fuel market, and market prices for energy products.

2.2.4. Economic Considerations of HS-AD in OFMSW Management

Similar to the state of evaluating the environmental impacts of various MSW management technologies, the economic/cost-benefits of MSW management technologies, especially advanced technologies such as ATTs and AD, have yet to be proven or even fully evaluated (EREF, 2013). According to a business analysis of AD in the US conducted by Renewable Waste Intelligence (RWI, 2013), the prospect of HS-AD becoming increasingly economical due to changes in the solid waste management industry is promising. It has been suggested that process instability or lack of reliability of HS-AD technologies has been a primary factor affecting its adoption (Kothari et al., 2014). However, decades of successful demonstration of HS-AD technologies in Europe is providing investors and industry leaders with confidence that this is no longer the case (RWI, 2013; De Baere and Mattheeuws, 2014).

Critical factors associated with the economics of HS-AD in MSW management include local tipping fees, local energy market, and quality and consistency of feedstock (RWI, 2013). A primary economic factor associated with capital costs of various projects is the capacity of the system. With greater capacity, comes greater project cost but a reduced cost per ton of capacity (Rapport et al., 2008). Other variable economic factors include permitting, engineering planning and design, construction, labor, and insurance. Accordingly, costs estimates for various MSW management technologies and projects vary substantially. According to the World Bank (2012), the costs of landfilling, incineration, composting, and AD in high income countries per ton of waste processed range from $40-100, $70-200, $35-90, and $65-150, respectively. These values include the sale of electricity, but exclude the sale of compost and digestate. According to a business analysis of AD in the US conducted by Renewable Waste Intelligence (RWI, 2013), AD projects can cost up to $600 per ton of annual processing capacity and operating costs can range from $40-150 per ton of waste processed. Table 2.8, adapted from PIS (2008), shows estimated capital and operations and maintenance (O&M) costs of various management technologies, processing capacities, and estimated time requirements from planning to commission.

Table 2.8: Estimated capital and O&M costs for a select processing capacity or capacity range and estimated time requirements from planning to commission of MSW management technologies (adapted from PIS, 2008).

Technology	Plant Capacity *(ton/day)*	Capital Cost *($/ton)*	O&M Cost *($/ton)*	Time to Commission *(months)*
Landfill	500	5-15	10-30	9-18
Incineration	1,300	30-180	80-120	54-96
Pyrolysis	70-270	16-90	80-150	12-30
Gasification	900	15-170	80-150	12-30
Composting (In-Vessel)	500	50-80	30-60	9-15
Anaerobic Digestion	300	20-80	60-100	12-24

A detailed cost analysis specifically for HS-AD that was conducted by Rogoff and Clark (2014) is the most comprehensive analysis found to date on the economics of HS-AD in the US. In the anlaysis, the authors estimated the capital investment required for a 5,000 ton per year (TPY) capacity HS-AD system (Table 2.9) based on Zero Waste Energy's (ZWE) SmartFerm design. This estimate was used along with variables and assumed values shown in Table 2.10 to develop a *Pro Forma* model for estimating required tipping fees for various HS-AD projects and assessing economic feasibility. Required tipping fees were estimated for four different scenarios, two with electiricty production and two without, and two with 5,000 TPY capacity and two with 10,000. The results are shown in Table 2.11. The study lists a biogas/power generation of 203 kWh/ton of feedstock processed.

Table 2.9: Estimated capital cost of a 5,000 TPY capacity HS-AD facility based on ZWE SmartFerm technology (adapted from Rogoff and Clark, 2014).

Item	**Cost ($)**
Digester Components (Leachate collection slab, gas collection bag, heating elements, gas piping, etc.)	1,000,000
Building Superstructure	575,000
Engine Generator Set	200,000
Improved Base for Foundation	200,000
Mixing Platform	100,000
Biofilters for Odor Control	100,000
Waste Storage Pad	50,000
Electrical Interconnection	75,000
Design, Permitting Support and Fees	50,000
Contingency	100,000
Total:	***2,450,000***

Table 2.10: General assumptions for *Pro Forma* model for HS-AD (Rogoff and Clark, 2014).

Variable	**Value**	**Comments**
Base Year	2014	Costs estimates were made in current 2014 dollars and escalated based on inflation factor shown below
Inflation Rate – Annual Escalation (for Energy, Labor, and Waste Collection)	2%	Based on recent Federal Reserve Board guidance
Organic Waste Received (Tons Per Year)	5,000	Based on model developed by ZWE
Annual Operating Costs ($)	3% of Capital	Estimated from information provided by AD developers
Annual Capital Repair and Replacement	1% of Capital	Estimated from information provided by AD developers
Financing Cost	3.35% interstate rate over 20 years	Agency borrowing costs
Tipping Fees	$35 per ton	Assumed
Energy Sales Prices	$0.1044 per kilowatt hour (kWh)	Assumed retail purchase price
Sale of Digestate	$0.00 per ton	Wholesale compost rates

Table 2.11: Results of *Pro Forma* model showing the tipping fees for four different scenarios required for the economic sustainability of an HS-AD project (Rogoff and Clark, 2014).

Scenario	Plant Capacity	Electricity Production	Tipping Fee Required
1	5,000 TPY	None	\$45.92 - \$53.16
2	5,000 TPY	203 kWh/ton @ \$0.1044/kWh	\$8.76 - \$31.97
3	10,000 TPY	None	\$40.73 - \$48.53
4	10,000 TPY	203 kWh/ton @ \$0.1044/kWh	\$3.57 - \$27.34

Based on conclusions from Rogoff and Clark (2014), observations from the development of HS-AD in Europe (De Baere and Mattheeuws, 2014), business analyses of HS-AD in the US (PIS, 2008, Rapport et al., 2008; RWI, 2013), and feasibility studies for HS-AD implementation in the US (RIS, 2005; FIE, 2009) a number of factors, either singularly or in combination, are critical for the economic sustainability and competitiveness of HS-AD, including:

- high local electricity costs, high onsite or nearby electricity demand and/or economic incentive for utility companies to purchase the renewable bioenergy;
- significant centralized sources of source-separated organic wastes, such as from food processing/packaging plants, hospitals, schools, prisons, or other institutional facilities with large cafeterias, or from large agricultural operations with crop residues;
- limited land suitable for composting and/or landfilling and/or lack of conventional WtE;
- markets for the residual compost;
- public/private partnerships, for example: between municipalities, waste management companies and haulers, utility companies, and local organizations;
- grants for funding renewable energy projects and/or recycling projects;
- regulatory drivers, such as a bans on organics disposal in landfills, regulated source-separation of OFMSW, renewable energy incentives, air quality regulations increasing the costs of composting and/or WtE operations, incentives for nutrient recovery/compost use.

The first commercial-scale HS-AD system constructed in the US at the University of Wisconsin, Oshkosh is an example of the development of partnerships and leveraging of funding sources to realize a vision of sustainable OFMSW management. Implementation of the 10,000 ton per year (TPY) capacity facility was accomplished through the collaborations of multiple entities, including UW Oshkosh, UW Oshkosh Foundation, Inc., City of Oshkosh, Zillges Materials, Inc., and Sanimax (UW Oshkosh, 2015). The $5 million project was funded by a combination of loans and grants, including a $3.7M Midwestern Disaster Area Revenue Bond issued by the City of Oshkosh (lender Wells Fargo Securities, LLC) and grant funding from the State of Wisconsin ($232,587), the U.S. Department of Energy ($500,000), and the U.S. Treasury Section 1603 ($1.1 million) (UW Oshkosh, 2015).

2.2.5. Policy Considerations of HS-AD in OFMSW Management

As highlighted above, several regulatory drivers have been identified as important to the competitiveness of HS-AD. Regulatory frameworks, policies, and incentives encourage investment in HS-AD through reducing capital and O&M costs of HS-AD, alleviating tax burdens, and creating market value for the environmental benefits of HS-AD (GMI, 2014). Exemplifying existing policy that incentivizes HS-AD implementation is useful for identifying locations where HS-AD implementation is most practical and for outlining legislative strategies for improving the feasibility of HS-AD in a given location. Thus, examples of such policies are provided below.

2.2.5.1. Source-Separation, Landfill Diversion, and Landfill Bans

Legislation encouraging source-separation, waste diversion from landfills, and banning the landfilling of OFMSW is tremendously important to the economic sustainability of HS-AD (Bolzonella et al., 2006b; EREF, 2015a). In the US, 20 states have bans on landfilling yard wastes and encouraging source-separation, mulching (informal composting for use in landscaping), and/or composting, five states have bans on landfilling food waste (CA, CT, MA, RI, and VT), and seven states have landfill diversion targets (CA, CT, DE, FL, MA, MI, and NY) (EPA, 2015a; EREF, 2015a). Furthermore, 209 communities across 16 states offer curbside food waste collection, covering approximately 2.3% of US households (2.7 million) (EPA, 2015a). Figure 2.5 is photograph taken in Calistoga, California in May of 2015 showing the three-bin system adopted by many municipalities to encourage source-separation, blue bins are for recyclables, green bins are for yard waste and food waste, and brown bins (the smallest of the three) are for trash. CalRecycle outlined how source-separation and organics recovery can be accomplished economically in CalRecycle (2002). However, only select jurisdictions have enacted legislation requiring source-separation of food and yard waste, as seen widely in Europe.

Figure 2.5: Three-bin system, as seen in May, 2015 in Calistoga, CA, adopted in municipalities across the US to encourage source-separation of MSW. Green is for food and yard waste, blue is for recyclables, and brown (the smallest of the three) is for trash disposal.

In 2013, New York City Council followed in the footsteps of San Francisco, Austin, Portland and Seattle by passing legislation requiring commercial separation of food waste (BioCycle, 2013). States with laws requiring certain organizations and businesses to source-separate food and yard waste include Massachusetts, Vermont, Rhode Island, Connecticut, and

California (GMI, 2014). More recently, precedence for requiring the provision of residential collection services for SS-OFMSW has been established via county-wide legislation in Alameda County, CA, and Hennepin County, MN. Of recently passed legislation, the most groundbreaking are those of Seattle, WA and the state of Vermont. A city-wide Seattle law, which was officially enacted on January 1st, 2015, requires all residents to separate both food and yard waste and includes $1 fines to single-household residents and $50 fines to multifamily residents who have more than 10% (visually) organic waste in the trash bins (Yepsen, 2015). Similarly, Vermont's Universal Recycling Law will require all residents to separate both food and yard waste by 2020 with expectations that HS-AD and composting operations will become increasingly abundant (GMI, 2014).

2.2.5.2. Renewable Energy and Recycling Incentives

There are many renewable energy incentives in the EU (Redman, 2010), including an active carbon market for the trading of carbon credits, The Renewables Obligation, The Climate Change Levy, Feed-in Tariffs, and Renewable Transport Fuel Obligation and Excise Duty Reductions. The carbon credit system was initiated by the signing of the Kyoto Protocol and creates an economic value for GHG emission offsets on a per metric ton of CO_2 equivalents basis ($MTCO_2$), allowing for trading of representative credits around the world (MH-Carbon, 2013). The credits represent offsets in GHG emissions resulting from offsetting fossil-fuel based energy with renewable energy or conversion of GHGs to inert gases (e.g. NOx to N_2) or gases with lower global warming potential (e.g. CH_4 to CO_2) (MH-Carbon, 2013). There are two primary types of markets in the carbon market: voluntary and compliance markets (Westerman, 2008). Compliance markets are mainly a result of the cap-and-trade system established in the Kyoto Protocol, whereas voluntary markets are a result of businesses and organizations seeking to improve their environmental sustainability (e.g. achieve carbon neutrality) (Westerman, 2008; MH-Carbon, 2013). The Renewables Obligation policy encourages the generation and use of renewable energy incentives through a credit system similar to that of carbon credits (Ares, 2012). The Climate Change Levy imposes additional costs (tax) to non-domestic consumers of non-renewable energy (UNESCAP, 2013). Feed-in Tariffs, referred to as "clean-energy cash back", guarantee minimum payment per kWh of renewable energy (Redman et al., 2010; Ares, 2012). The Renewable Transport Fuel Obligation and Excise Duty Reductions require road-fuel suppliers that do not incorporate biofuels to pay penalties, award credits for the use of biofuels, and reduce the costs of biofuels (Hood, 2014). These regulatory frameworks were developed over many years and with great controversy.

In the US, the Renewable Portfolio Standard (RPS) program is the most well developed nationwide program incentivizing the generation of renewable energy (Holt and Bird, 2005). The RPS program is a regulatory mandate designed to encourage state and local governments to increase production and/or indirect consumption of renewable energy (Holt and Bird, 2005). To aid states in meeting their RPSs and facilitate the indirect consumption of renewable energy by creating a means for trading renewable energy, Renewable Energy Certificates (RECs) were created (Holt and Bird, 2005). The EPA refers to RECs as the "currency" for renewable energy markets and defines them as a component of all renewable electricity products which is representative of the environmental benefits of renewable electricity generation (EPA, 2008). For every 1,000 kWh, or one megawatt hour (MWh), of renewable electricity generated, one marketable REC is generated (EPA, 2008). Each REC that is generated is stamped with information, including the date when the REC was created, the location it was created, and the type of renewable electricity that generated the REC (resource type) (EPA, 2008). When a buyer purchases a REC (for whatever the market price may be, depending on supply and demand), they then own the right to claim the environmental benefits associated with one MWh of renewable electricity generation (e.g. GHG offsets) and as soon as a REC buyer makes an environmental claim based on a REC, the REC is "retired" and no longer has monetary value (EPA, 2008).

The REC trading system is similar to the carbon credit system. The US has not ratified the Kyoto Protocol, and therefore, there is no compliance market for carbon credits in the US (Westerman et al., 2008). However, voluntary markets for carbon credits do exist in the US and are expanding, with companies such as AgraGate, Chicago Climate Exchange, and the Environmental Credit actively involved in carbon credit trading (Westerman et al., 2008). With potential overlap between REC markets and carbon credit markets and REC tracking systems still in their infancy, there is some concern over the possibility of "double counting" GHG offsets (Westerman et al., 2008). Some rules exist to prevent this from occurring and in general, carbon credits can be sold only for the destruction of methane (combustion either through flaring or through heat/power generation) and not for the offsets associated with renewable energy generation, whereas RECs can be generated and sold only for heat/power generation.

With the spreading of the sustainability movement, increasing environmental awareness, and increases in legislation pertaining to environmental conservation, many individuals, companies, organizations, and state and federal agencies in the US are interested in or required to meet certain environmental standards and goals (Holt and Bird, 2005). This trend is reflected in the recent and projected growth of REC markets and carbon markets in the US and an increasing number of states with state-wide programs for incentivizing renewable energy generation and recycling (EPA, 2013a; DOE, 2015a; DSIRE, 2015). However, only 29 states currently have RPS, with another eight states having voluntary renewable energy targets, and REC markets and tracking systems are still relatively small and lacking integration (NREL, 2014; Holt and Bird, 2005). In general, current prices for RECs and carbon credits vary considerably between compliance and voluntary markets and also vary based on region and resource type, with compliance markets generally having higher trading prices than voluntary, the eastern US having higher trading prices than the mid-west or western US, and solar yielding the highest REC trading prices followed by biomass and wind (Holt and Bird, 2005; DOE, 2015b).

Regarding recycling incentives, 25 states in the US now offer some type of tax incentive or credit to promote material recycling and reuse (EPA, 2015b) and numerous other programs (non-tax incentive based) have been developed and adopted by state governments to boost recycling rates (Sparks, 1998). Most of the programs offer exemptions or credits on sales, income, or property taxes for the purchase of recycling or pollution control machinery including equipment designed for collecting, processing, treating, of separating MSW, or converting it into a useful product (Sparks, 1998). Many of the incentives are focused on the recycling industry as a whole, offering tax credits based on number of jobs created, the amount of material processed/handled, or on the amount of capital invested (Sparks, 1998). Some programs set limits on minimum recycled content of recycled products (Sparks, 1998). Programs offering tax exemptions on the construction and renovation of recycling facilities and programs offering income tax credits to individuals who purchase products made from recycled materials also exist (EPA, 2015b). State-wide recycling programs can, however, be expensive and information on the effectiveness of the programs in increasing recycling rates isn't well-documented (Sparks, 1998).

2.2.5.3. Air Quality and Nutrient Management

According to a feasibility study for garden waste management in the Sacramento, CA area conducted by RIS International Ltd. (RIS, 2005), recent air quality regulations for windrow composting projects are increasing costs of composting in California, further balancing the economics of HS-AD versus composting and contributing to the enhanced feasibility of HS-AD. Similarly, increasingly stringent air quality regulations could have negative impacts on the economics of WtE technologies, as seen with the Clean Air Act and the Clean Power Plan increasing costs in the fossil fuel industry (Tchobanoglous and Kreith, 2002; EPA, 2015c).

Eutrophication is now a well-recognized human induced problem worldwide and thus, nutrient recovery and integrated nutrient management programs are becoming increasingly abundant. In a review of AD policies and incentives prepared by the Global Methane Initiate (GMI, 2013), 16 out of 30 countries reviewed had nutrient management policies in place. In the US, for example, wastewater treatment plant effluent nutrient concentrations are strictly regulated and the EPA's National Pretreatment Program requires pretreatment of high strength wastewaters (e.g. landfill leachate)including nutrient reduction before discharging to publicly owned treatment works (EPA, 2011). Policies in the US that directly encourage the use of compost for the sake of recovering nutrients could not be found, although it is reasonable to assume that such policies do exist and will continue to become more prevalent. A recent proposal by the United Nations Educational, Scientific, and Cultural Organization (UNESCO, 2015) to "green the nutrient economy and reduce ocean hypoxia through policy, regulatory, and economic instruments to promote nutrient efficiency and recovery" reaffirms this assumption.

2.3. State of the Art of HS-AD of OFMSW

In this section, classifications of AD and HS-AD systems are presented and trends in the development of AD of OFMSW in Europe and the US are reported. A detailed list of existing and planned HS-AD facilities and an overview of available HS-AD technologies in the US are also provided.

2.3.1. HS-AD System Classifications

HS-AD technologies are most often classified according to three key characteristics: loading conditions (continuous or batch), number of stages (single-stage or multi-stage), operating temperatures (mesophilic or thermophilic) (Rapport et al., 2008). In addition to these classifications, HS-AD systems are often classified by feedstock (whether they are processing SS-OFMSW, MS-OFMSW, or mixed MSW) and whether they are processing a single substrate (e.g. OFMSW) or are codigesting (e.g. OFMSW with biosolids) (De Baere and Mattheeuws, 2014). Figure 2.6 illustrates the many possible AD system "types" based on these classifications and Table 2.12 summarizes their advantages and disadvantages. Tracking trends in development of HS-AD with respect to the relative prevalence of these AD system types can be helpful for understanding industry preferences and identifying appropriate technologies.

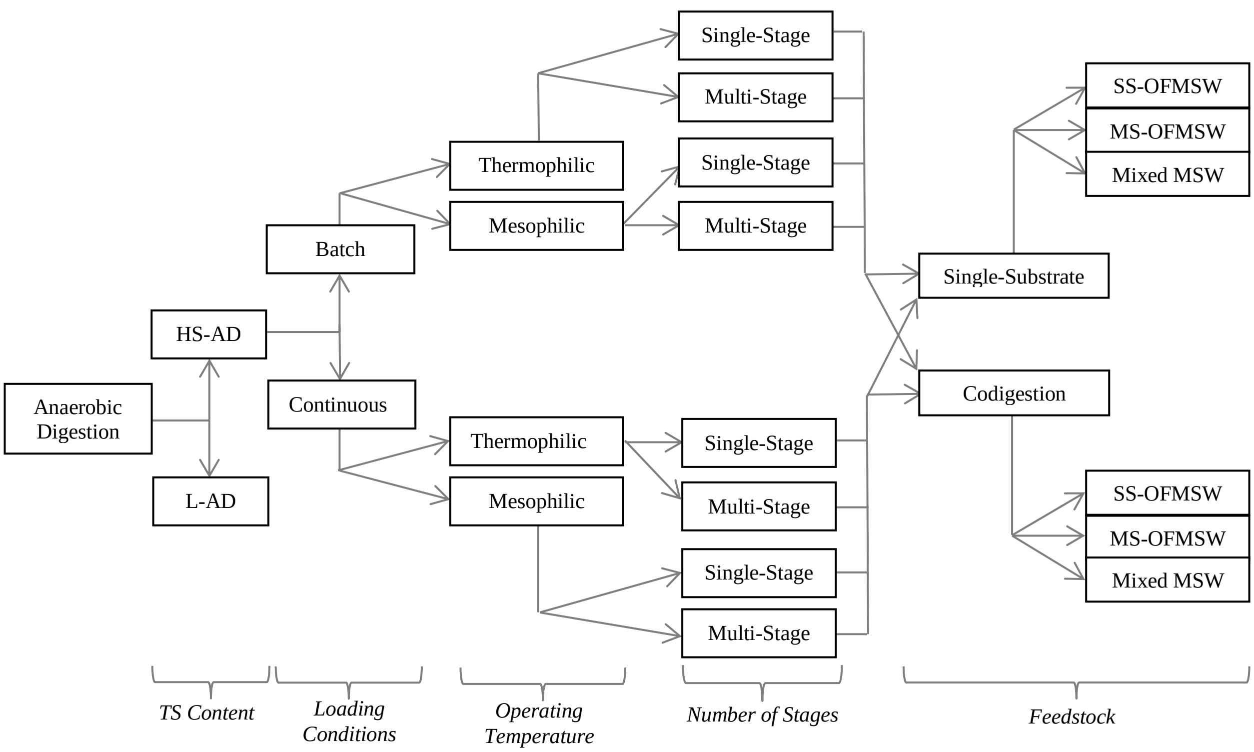

Figure 2.6: Possible AD system "types" based on predominant system classifications.

Table 2.12: Technical, biological, and environmental/economic advantages and disadvantages of AD technologies for OFMSW by classification (adapted from Rapport et al., 2008).

System	Criteria	Advantages	Disadvantages
Batch vs. Continuous	Technical	Simplifies material handling; reduced pre-processing/treatment requirements	Compaction within digester can reduce percolation and percolate recirculation capabilities
	Biological	Separation of hydrolysis and methanogenesis; higher rate and extent of digestion than landfill bioreactors	Variable biogas production with time; reduced process control
	Economic and Environmental	Low capital cost; low O&M costs; lower overall impact	Less complete degradation
Multi-stage vs. Single-stage	Technical	More operationally flexible	Complex design and materials handling
	Biological	Can tolerate high loading rates and fluctuations in loading rates	Can be difficult to achieve true separation of phases in digesters
	Economic and Environmental	Can yield higher digestion efficiencies	Increases capital and O&M costs
Thermophilic vs. Mesophilic	Technical	Requires minimal change in design (heat transfer systems)	Requires more heat transfer equipment
	Biological	Improves digestion efficiency; improves pathogen destruction	Greater risk of process inhibition with thermophilic systems
	Economic and Environmental	Improves bioenergy production rate and marketability of compost	Thermophilic systems require greater heat input
Codigestion vs. Single Substrate	Technical	Requires no change in design	Requires increased preprocessing
	Biological	Enables optimization of environmental conditions which can improve bioconversion rates	Greater potential for variation in feedstock characteristics and shock inhibition
	Economic and Environmental	Can yield significant enhancements in bioenergy generation	Can increase the economic and environmental costs of waste collection
Source Separated OFMSW vs. Mixed MSW	Technical	Collection is simple with mixed MSW and feedstock contamination is of little importance	Collection schemes for SS-OFMSW can be challenging; minor contamination (e.g. glass) can pose a problem for digestate reuse
	Biological	Source separation reduces variation in feedstock characteristics and yields more consistent conditions and performance	Processing mixed MSW poses threats of contamination with strong inhibitory compounds
	Economic and Environmental	Less energy is needed for mixed MSW collection; less energy is needed for processing source separated waste and more energy and nutrients are recovered	Processing mixed MSW increasing energy input requirements and reduces bioenergy yields and nutrient recovery potential

Continuous HS-AD systems are loaded daily, with fresh material going in one end and digested material coming out the other. These systems are generally configured as large plug-flow type reactors. Batch, systems normally consist of multiple "garage" or "shipping container" type reactors that are loaded, sealed, and left to digest for a specified amount of time until being unloaded (Rapport et al., 2008). Single-stage systems use a single reactor for the entire AD process, whereas multi-stage systems use two or more reactors with varying environmental conditions and retention times to separately optimize different phases of the AD process (e.g. hydrolysis and acidogenesis in one reactor and acetogenesis and methanogenesis in a subsequent reactor) (Deublein and Steinhauser, 2008). Multistage systems also sometimes feature both HS-AD and L-AD (e.g. hydrolysis, acidogenesis, and acetogenesis via HS-AD and methanogenesis via L-AD) (Deublein and Steinhauser, 2008). Mesophilic AD systems have operating temperatures ranging from 35-40 °C, whereas thermophilic systems have operating temperature ranging from 50-55 °C. Some multi-stage systems have stages with varying temperatures (e.g. mesophilic first-stage and thermophilic second stage) (Lin et al., 2013).

2.3.2. HS-AD Development in Europe

De Baere and Mattheeuws (2014) provided a comprehensive review of trends in the development of AD of OFMSW in Europe. Information from this review is summarized in Table 2.13. As of 2014, there were 244 full-scale AD plants for processing OFMSW with a total capacity of approximately 8 million TPY, 62% of installed AD in Europe was HS-AD and the remaining 38% was L-AD. HS-AD is preferred over L-AD for processing OFMSW due to their economic and environmental advantages, and this trend is expected to continue in the future (De Baere and Mattheeuws, 2014). The majority of AD systems in Europe as of 2014 were continuous systems; however, batch systems have been increasing in popularity since 2009 due to their simplicity and low cost (De Baere and Mattheeuws, 2014). Single-stage systems made up approximately 93% of AD capacity in 2014, with only 7% being multi-stage (two-stage). Implementation of multi-stage systems has been continuously declining because their benefits do not justify their higher capital and operating costs (De Baere and Mattheeuws, 2014). Mesophilic digestion accounted for 67% of AD in Europe in 2014, but thermophilic digestion is becoming increasingly common and is expected to surpass mesophilic digestion as it is now considered mature and has been shown to yield net economic benefits (De Baere and Mattheeuws, 2014).

Table 2.13: Characterization of AD of OFMSW in Europe (De Baere and Mattheeuws, 2014).

Classification	% of Installed Capacity	Trends	Expected Future Trends
Total Solids Content	62% HS-AD, 38% L-AD	HS-AD systems have been consistently preferred over L-AD systems for processing OFMSW for more than 20 years, over 70% installed since 2009 has been HS-AD.	HS-AD will continue to increase in prevalence due to the economic and environmental advantages it offers compared to L-AD.
Loading Conditions	> 50% Continuous	Continuous systems have traditionally dominated the industry, but batch systems have been catching on quickly since 2009.	Batch systems are expected to continue to increase in popularity due to their simplicity and low cost.
Number of Stages	93% Single-Stage, 7% Two-Stage	Multi-stage systems have been continuously in decline since the 1990's.	No immediate changes in this trend are expected due to the higher investment and operating costs that accompany multi-stage systems.
Operating Temperature	67% Mesophilic, 33% Thermophilic	Thermophilic digestion has been becoming increasingly common in the last decade.	Thermophilic capacity is expected to surpass mesophilic capacity because thermophilic systems are now well-proven and yield net economic benefits in most cases.
Codigestion	89% Single-Substrate, 11% Codigestion	The trend has been almost unanimously from codigestion to single substrate digestion, as "dedicated" systems tailored for OFMSW processing have been designed and implemented; however, in recent years there has been a slight rise in codigestion.	Laboratory research and the agro-industrial sector have demonstrated the potential economic advantages of codigestion and thus, it may become increasingly common.
Feedstock	55% Source-Separated, 45% Mixed MSW	Increases in capacity for processing source separated waste have been in direct proportion to increases in legislation regulating the source separation of OFMSW.	It is expected that source separation regulations will continue to increase and therefore, digestion of source separated OFMSW will continue to increase.

With respect to feedstock, single substrate digestion (OFMSW) accounted for 89% of AD in 2014, with codigestion (e.g. OFMSW with wastewater biosolids or livestock wastes) representing only 11% of installed capacity (De Baere and Mattheeuws, 2014). The longstanding trend has been from codigestion to single substrate digestion, as "stand-alone" systems tailored to process OFMSW have become increasingly common. More recently, there has been a slight increase in codigestion, as facilities in the agro-industrial sector have demonstrated the potential economic advantages of codigestion (De Baere and Mattheeuws, 2014). With respect to source-separation, 55% of European AD systems in 2014 were processing SS-OFMSW while 45% were processing mixed MSW. Increases in capacity for processing SS-OFMSW have been in direct proportion to promulgation of regulations on source-separation of OFMSW in commercial, institutional, and residential settings (De Baere and Mattheeuws, 2014).

2.3.3. HS-AD in the United States

In the United States, several pilot-scale and/or demonstration-scale HS-AD projects were constructed prior to 2002, as described by Rapport et al. (2008). The first full-scale demonstration HS-AD system in the US was constructed in Clinton, NC in 2002 (Greer, 2011). The 3,380 TPY facility employs an HS-AD technology (now marketed by Orbit Energy, Inc.) developed by the US Department of Energy National Renewable Energy Laboratory (Greer, 2011). The first commercial HS-AD system in the US was constructed in 2011 at the University of Wisconsin, Oshkosh and began operation in 2012 (UW Oshkosh, 2015), as described in Section 2.2.4. Currently, eight full-scale HS-AD facilities are operating in the US, with a total capacity of 189,600 TPY. Another 19 or more HS-AD projects are in the planning, permitting, or construction phases (Table 2.17). The majority of the existing and planned facilities are located in California and are or will utilize the SmartFerm technology marketed by Zero Waste Energy, LLC (ZWE, US affiliate of the German company, Eggersmann Group), including the largest HS-AD facility in the country (90,000 TPY in San Jose, CA). However, several other vendors have established themselves in the North American HS-AD market (Table 2.14) and several other states have implemented or are planning to implement HS-AD (Table 2.17). Figure 2.7 shows the number of HS-AD facilities in the US versus time since 2011 and projected out to 2017. Figure 2.8 displays the locations of existing and planned HS-AD facilities in the US.

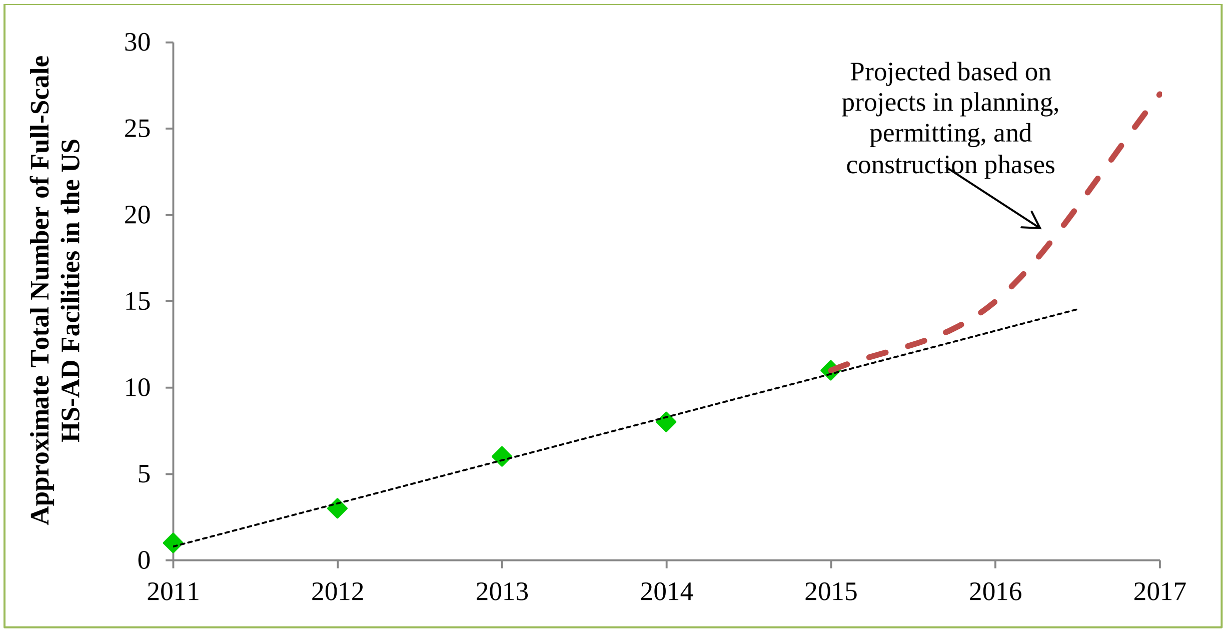

Figure 2.7: Total number of HS-AD facilities in the US versus time, 2011 to 2017 (projected).

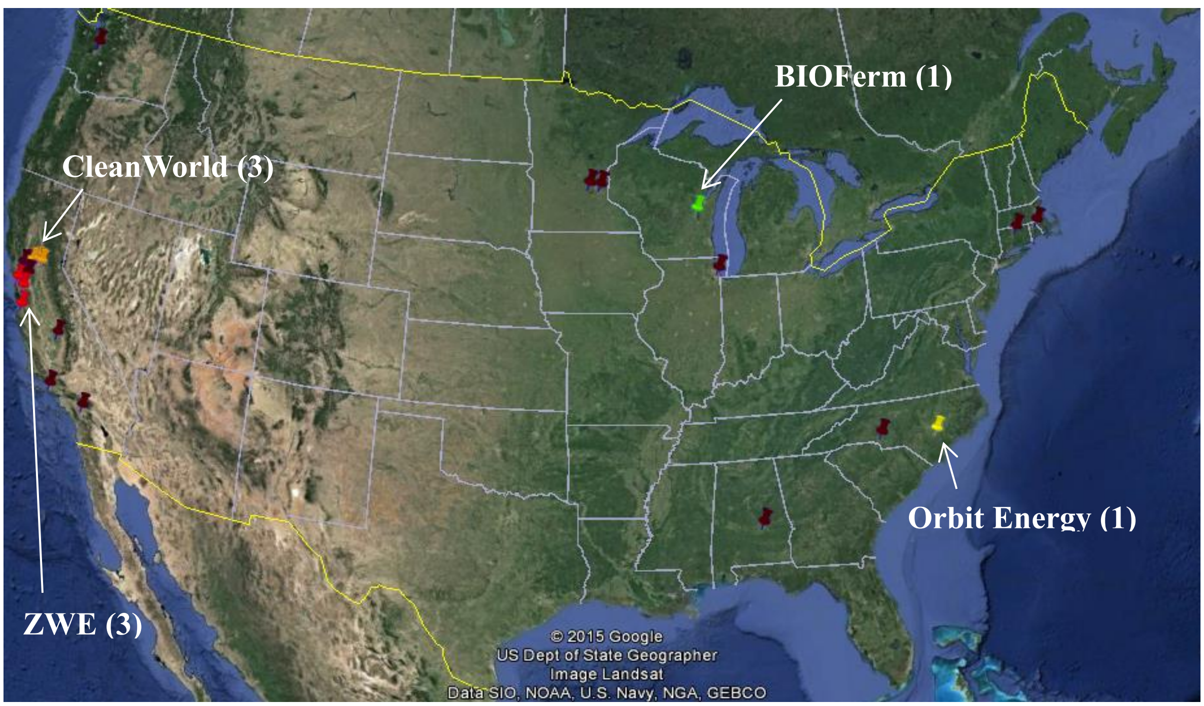

Figure 2.8: Locations of existing and planned HS-AD facilities in the US.

The primary characteristics of the technologies offered by HS-AD vendors in the US are summarized in Table 2.15. Brief descriptions of the systems are provided here. Schematics of system configurations, photographs, and model images of systems are provided in Appendix C.4.

- *Zero Waste Energy (ZWE):* The SmartFerm/KompoFerm technology offered by ZWE was developed in Germany and is referred to as a “dry AD” technology. The thermophilic batch, single-stage technology can process feedstocks with up to 50% TS content. The typical retention time is 21 days, there is no pre-processing required, feedstock is added and digestate is removed via a front end loader. No water addition is necessary and percolate generated during digestion is collected, stored underground, and continuously recirculated through the system. Digestate is processed via a trommel screen, composted (either windrow or in-vessel, composting time can range widely), and then marketed. Any excess percolate generated in the system is added to compost. The system is equipped with one or more biofilters for odor and air emissions control and is reported to have less than 20% parasitic energy demand. The system capacity can range from 4,000 TPY to more than 100,000 TPY by increasing the number of digesters operating in parallel (ZWE, 2014; ZWE, 2015).
- *CleanWorld:* The CleanWorld US Patented technology is referred to as a “high-solids AD” technology. The thermophilic, continuous, three-stage technology includes a “hydrolysis reactor, biogasification reactor, and biostabilization reactor”. Typical total retention times are 20-30 days. Incoming waste is ground to less than 50 mm and stripped of plastics/packaging and water is added to wastes with greater than ~10% TS content. The resulting mixture (solids in an aqueous solution) is moved through system via conventional pumps. The system generates significant quantities of excess percolate, a portion of which may be recycled through system, but the remainder of which requires treatment or, in some cases may be used directly as liquid fertilizer. Solid digestate from the system likely requires trommel/contaminant removal and final aerobic curing. These systems range in capacity from 8,000 to +70,000 TPY (Zhang, 2013; CleanWorld, 2015a; CleanWorld, 2015b).

- *Orbit Energy:* The US Patented Orbit Energy technology developed by the US Department of Energy National Renewable Energy Laboratory is referred to as a “high-solids AD” technology. The thermophilic, continuous, single-stage technology can process feedstocks with up to 45% TS content. The technology employs a proprietary microbial community that can handle high organic loading rates resulting in “short” retention times (unspecified) and helps yield a “small” system footprint (unspecified) and low (8%) parasitic energy demand. Lime is added to feedstocks for pH adjustment. The system is enclosed (indoor) and generates zero excess percolate as digestate is dried with waste heat from CHP units. Digestate from the system is marketed to composting companies or as compost. These systems range in capacity from 4,380 to 91,000 TPY (Greer, 2011; Orbit Energy, 2015).
- *BIOFerm Energy Systems:* BIOFerm supplies two HS-AD type technologies, BIOFerm Dry Fermentation technology and EUCO technology, both developed in Germany. BIOFerm’s “Dry Fermentation” technology is a mesophilic, batch, single-stage technology, similar to ZWE’s SmartFerm technology, and can process feedstocks with up to 25-35% TS content. Typical retention times in these systems are around 28 days. No preprocessing of feedstock is required and the technology uses percolate recirculation *or* fresh waste inoculation. Biofilters are used for odor and air emissions control. The parasitic energy demand is reported to be around 5-10% and system capacity ranges from 8,000 TPY to 70,000 TPY or more. BIOFerm’s EUCO technology is referred to as a “high-solids AD” technology. It is a continuous, horizontal plug-flow unit and is typically the first stage of a two-stage system (often paired with other wet BIOFerm technologies) (BIOFerm, 2014a; BIOFerm; 2014b).
- *Organic Waste Systems (OWS):* OWS’s DRANCO technology was developed in Belgium and is an acronym for “Dry Anaerobic Composting”. These single stage systems are

vertically configured, gravity driven plug-flow units. They can operate either as mesophilic or thermophilic systems. Incoming waste materials are mixed with digestate at up to a 1/6 ratio, passed through a grinder yielding a 40 mm maximum particle size, then pumped to the top of the reactor without addition of water. The average total retention time in these systems is 20 days, with pass-through times ranging from 2-4 days. Digestate is dewatered and composted. Excess percolate is sometimes generated and may be added to compost or treated separately. These systems range in capacity from 3,300 to 100,000 TPY (De Baere, 2012).

- *Harvest Power:* Harvest Power's GICON technology, developed in Germany, is referred to as a "high-solids AD" technology. The batch, two-stage, thermophilic, systems separate the "acid" phase from the "gas" phase and have a total retention time as low as 14 days. No water addition is necessary with these systems, though acceptable feedstock TS content is unspecified. The first stage of the system is "garage" type digestion under HS-AD conditions with percolate recirculation and the second stage is a conventional L-AD "tank" type reactor. The capacity range of these systems is not specified (Harvest Power, 2014).
- *Eisenmann Corporation:* Eisenmann's BIOGAS Green Waste (BIOGAS-GW) technology, developed in Germany, is considered a "dry" AD technology. The technology is a thermophilic, continuous, single-stage, horizontal plug-flow system, but limited supplemental information could be found (Eisenmann, 2014).
- *EcoCorp*: EcoCorp's "dry AD" technology is capable of processing feedstocks with up to 35-40% TS content. In this continuous, thermophilic, single-stage, system, waste is deposited onto a conveyor belt and fed to a screw shredder which produces a feedstock of 40 mm maximum particle size. Positive displacement pumps (similar to those used in the concrete industry) are used to move the feedstock to an equalization/buffer tank where fresh waste is

mixed with digestate at ratios ranging from 1/1 to 1/10. The retention time is approximately 20 days (not explicitly specified). Digestate from these systems is dewatered, the percolate ("filtrate") is partially recirculated through the system and partially treated via centrifugation and aerobic treatment, and the solid fraction is composted for approximately 10 days and marketed. The system is completely enclosed, air emissions are treated via a biofilter, and the parasitic energy demand is expected to be around 20% demand (not explicitly specified). The capacity of these systems can range from 20,000 to 100,000 TPY (EcoCorp, 2015).

- *Turning Earth:* The Aikan technology, developed in Denmark, is a "dry AD", batch, thermophilic, two-stage technology, with "hydrolysis" occurring in a garage type modules under HS-AD conditions and "methane production" occurring in a convention L-AD "tank" type reactor. Percolate is recirculated from the stage-two digester to the stage-one modules until the methane production rate decreases below a target value. The stage-one garage type modules are converted from anaerobic to aerobic systems after approximately 21 days and operated as in-vessel composters for approximately 14 days for digestate curing before removing the solid material and marketing it as compost. Excess percolate is evaporated during the in-vessel composting stage. Emissions are treated via a biofilter (Aikan, 2015).

The current status and trends in the development of AD of OFMSW in the US are provided in Table 2.16. Existing HS-AD facilities in the US are summarized in Table 2.17, along with project-specific details. Well-documented projects in the planning, permitting, or construction phase are also included in this table. According to a recent report by the Environmental Research and Education Foundation (EREF, 2015a), there are currently 181 AD facilities in the US processing OFMSW, with a total OFMSW throughput of 780,000 TPY. Of these facilities, 81 are wastewater treatment plant digesters accepting some food waste or FOG (fats, oils, and greases), with a total throughput of 226,000 TPY (29%), 75 are on-farm digesters accepting food and/or yard waste, with a total throughput of 140,000 TPY (18%), and 25 are stand-alone facilities (designed specifically for processing OFMSW) with a total capacity of 406,000 TPY (52%). It follows that approximately 47% of existing stand-alone capacity for AD of OFMSW is HS-AD (189,600 TPY of 406,000 TPY). However, if all planned AD facilities for OFMSW come online, by 2017 HS-AD will be the dominant AD technology type for processing OFMSW in the US, which parallels trends in Europe. With respect to the prevalence of HS-AD systems by other classification categories, 61% of capacity (on a TPY basis) is of the batch variety, 63% is of the single-stage variety, and 95% is of the thermophilic variety.

Table 2.14: Primary vendors of HS-AD technologies in the US.

Vendor Name	Main Office Location	Founding Year	Primary Partnerships	# of Facilities in Operation in the US	# of Facilities in Development in the US
Zero Waste Energy, LLC	California	2009	Eggersmann Group, Bulk Handling Systems, Environmental Solutions Group	≥ 3	≥ 7
CleanWorld Corporation (formerly CleanWorld Partners, LLC)	California	2009	UC Davis, Synergex	≥ 3	≥ 1
Orbit Energy, Inc.	North Carolina	2002	McGill Environmental	≥ 1	≥ 5
BIOFerm Energy Systems	Wisconsin	2007	Viessmann Group, Schmack Biogas	≥ 1	≥ 1
Organic Waste Systems, Inc.	Belgium (subsidiary in Ohio)	1988	NR	≥ 0	≥ 1
Harvest Power, Inc.	Massachusetts	2008	GICON Bioenergie GmbH	≥ 0	≥ 1
Eisenmann Corporation	Germany (subsidiary in Illinois)	1977	NR	≥ 0	≥ 2
Turning Earth, LLC./Aikan North America, Inc.	Denmark (subsidiary in Georgia)	2009	Solum Group, Aikan A/S	≥ 0	≥ 1
EcoCorp, Inc.	Maryland	2000	NR	≥ 0	≥ 0

Note: NR = Not Reported; ≥ 0 indicates that zero facilities were identified, but that it is possible that some exist

Table 2.15: Primary characteristics of HS-AD technologies available in the US.

Vendor Name	Operating Temperature	TS Content	Loading Conditions	Number of Stages	Retention Time	Parasitic Energy Demand
Zero Waste Energy, LLC	Thermophilic	< 50%	Batch	1	21 days	20%
CleanWorld Corporation (formerly CleanWorld Partners, LLC)	Thermophilic	~10%	Continuous	3	20-30 days	NR
Orbit Energy, Inc.	Thermophilic	< 45%	Continuous	1	"short"	8%
BIOFerm Energy Systems	Mesophilic	25-35%	Batch	1	28 days	5-10%
Organic Waste Systems, Inc.	Thermophilic or Mesophilic	< 50%	Continuous	1	20 days	NR
Harvest Power, Inc.	Thermophilic	NR	Batch	2	≥ 14 days	NR
Eisenmann Corporation	Thermophilic	NR	Continuous	1	NR	NR
Turning Earth, LLC.	Thermophilic	NR	Batch	2	21 days	NR
EcoCorp, Inc.	Thermophilic	35-40%	Continuous	1	20 days	20%

Note: NR = Not Reported; information reported here was derived from the sources cited in the above technology descriptions

Table 2.16: Characterization of AD of OFMSW in the US.

Classification	Current Status	Expected Future Trends
Total Solids Content	Since 2011, the fraction of stand-alone capacity for AD of OFMSW has increased from nearly 0% to around 48% (189,600 TPY of 406,000 TPY).	HS-AD will become the dominant form of AD of OFMSW by 2017 due to the economic and environmental advantages it offers over L-AD.
Loading Conditions	Approximately 61% of HS-AD capacity is currently of the batch variety (116,200 TPY of 189,600 TPY).	14 of the 27 HS-AD systems expected to be in operation by 2017 will be batch systems; no clear trend exists in this respect.
Number of Stages	Around 63% of HS-AD capacity is currently of the single-stage variety (119,600 TPY of 189,600 TPY).	Only 6 of the 27 HS-AD systems expected to be in operation by 2017 will be multi-stage, suggesting that single-stage systems are generally preferred, likely due to their simplicity and low cost.
Operating Temperature	Thermophilic digestion represents the vast majority (>95%) of existing capacity for HS-AD of OFMSW.	Thermophilic digestion is expected to remain the dominant digestion type due the increased efficiency it offers and demonstrated stability.
Codigestion	Currently, 47% of capacity for AD of OFMSW is codigestion, with 29% being at wastewater treatment plants and 18% being at farms.	The stand-alone capacity for AD of OFMSW is expected to quadruple to 2.5 million tons by 2017 (EREF, 2015a) surpassing codigestion as the dominant form.
Feedstock	Limited information exists on whether existing facilities are processing mixed, mechanically separated, or source-separated OFMSW.	Increases in mandates on source-separating OFMSW and studies indicating significant economic advantages associated with processing SS-OFMSW over MS-OFMSW suggest that processing source-separated feedstock will be the dominant form of AD of OFMSW.

Table 2.17: Existing HS-AD facilities and planned HS-AD projects in the US in chronological order.

Location	Company/ Design	Funding Sources	Start-up	Capacity	Cost	Gas Utilization	Digestate Utilization	Source(s)
Clinton, NC	Orbit Energy	NR	2002	3,400 TPY	NR	CHP: unspecified	Marketed as Class A compost	Orbit Energy, 2015
Oshkosh, WI	BioFerm Energy Systems	Midwestern Disaster Area Revenue Bond, State of Wisconsin grant, U.S. Department of Energy grant, the U.S. Treasury Section 1603 grant	2012	10,000 TPY	$5 M	CHP: 370 kW, surplus electricity sold to Wisconsin Public Service	Sold as soil amendment	UW Oshkosh, 2015
Sacramento, CA (Am. River Packaging)	CleanWorld Corporation	Five Star Bank, Central Valley Community Bank, the California Energy Commission, CalRecycle, Synergex	2012	40,000 TPY	NR	CNG: up to 700,000 diesel gallon equiv./yr	NR	CleanWorld, 2012; CleanWorld, 2015b
Monterey, CA	Zero Waste Energy	NR	2013	5,000 TPY	NR	CHP: 100 kW, surplus electricity sold to neighboring wastewater treatment plant	2,200 TPY sold to local farmers	ZWE, 2013a
San Jose, CA	Zero Waste Energy	NR	2013	90,000 TPY	NR	CHP: 1.6 MW, surplus electricity sold to neighboring wastewater treatment plant	NR	ZWE, 2013a
Davis, CA	CleanWorld Corporation	First Northern Bank, CalRecycle, the U.S. Department of Energy	2013	20,000 TPY	NR	Microturbines: 640 kW	Liquid fertilizer, onsite composting is expected soon	CleanWorld, 2015b
South San Francisco, CA	Zero Waste Energy	NR	2014	11,200 TPY	NR	CNG: 120,000 diesel gallon equiv./yr	NR	ZWE, 2013a.
Sacramento, CA	CleanWorld Corporation	NR	2014	10,000 TPY, Expanding to 40,000	NR	NR	NR	CalRecycle, 2014a; CleanWorld, 2015b
Perris, CA	Eisenmann Corporation	NR	2015 (projected), under construction	80,000 TPY, may expand to 300,000	NR	CNG: up to 1,000,000 diesel gallon equiv./yr	NR	CalRecycle, 2014a; Eisenmann, 2012; Eisenmann, 2014
Chicago, IL	Eisenmann Corporation	Illinois Department of Commerce and Economic Opportunity (DCEO); Partnership with The Plant	2015 (projected)	5,000 TPY, may expand to 11,000	NR	CHP: 200 kW	NR	Eisenmann, 2012; Eisenmann, 2014
Tulare, CA	Harvest Power	California Energy Commission grant; Partnership with Colony Energy Partners	2015 (projected), permitting	Up to 182,500 TPY	$25-30M (projected)	CNG: up to 2,800,000 diesel gallon equiv./yr + CHP	NR	CalRecycle, 2014a; Fletcher, 2015

Table 2.17 (Continued)

Location	Company/ Design	Funding Sources	Start-up	Capacity	Cost	Gas Utilization	Digestate Utilization	Source(s)
Montgomery, AL	Zero Waste Energy	Partnership with IREP (Infinitus Renewable Energy Park) and the City of Montgomery	Phase 1 (MRF) began in 2015	12,500 TPY	NR	CNG: 130,000-150,000 diesel gallon equiv./yr	NR	ZWE, 2013a
Vacaville, CA	Organic Waste Systems	California Energy Commission grant; California Alternative Energy and Advanced Transportation Financing Authority	2016 (projected), permitting	65,000 TPY	$26 M (projected)	CNG: 1,100,000 diesel gallon equiv./yr (projected)	40,000 TPY liquid and solid fertilizer	CalRecycle, 2014a; Ruiz, 2014; OWS, 2015
Hartford, CT	Turning Earth	NR	2016 (projected)	50,000 TPY	$20M (projected)	CHP: 1.4 MW	40,000 cubic yards of compost to be marketed	Turning Earth, 2014
Johnston, RI	Orbit Energy	Partnership with National Grid and Blue Sphere Corporation	2016 (projected)	91,250 TPY	$18.9M	CHP: 3.2 MW	13,000 -14,600 TPY of compost to be marketed	Faulkner, 2015; Orbit Energy 2015
Des Moines, WA	Orbit Energy	Partnership with Puget Sound Energy	2016 (projected)	NR	NR	CHP: 4.5 MW	NR	Orbit Energy, 2015
Charlotte, NC	Orbit Energy	Partnership with Duke Energy and Blue Sphere Corporation	2017 (projected)	NR	NR	CHP: 4.8 MW	NR	Orbit Energy, 2015
Napa, CA	Zero Waste Energy	California Energy Commission grant; City of Napa; Napa Recycling and Waste Services, LLC.	2017 (projected)	25,000 TPY	NR	CNG: 330,000 diesel gallon equiv./yr	20,447 TPY of compost to sell to local farmers	ZWE, 2013a
Oxnard, CA	Zero Waste Energy	Partnership with Agromin, Inc.	2017 (projected)	20,000 TPY	NR	NR	NR	ZWE, 2013a
San Leandro, CA	Zero Waste Energy	NR	2017 (projected)	20,000 TPY	NR	NR	NR	ZWE, 2013a
Contra Costa County, CA	Zero Waste Energy	NR	2017 (projected)	20,000 TPY	NR	NR	NR	ZWE, 2013a
Delano, MN	Zero Waste Energy	NR	2017 (projected)	40,000 TPY	NR	NR	NR	ZWE, 2013a
Minneapolis, MN	Zero Waste Energy	NR	2017 (projected)	30,000 TPY	NR	NR	NR	ZWE, 2013a

Note: not included in this list are two planned Orbit Energy projects, one planned BioFerm Energy Systems project, and one planned CleanWorld Corporation project for which minimal project-specific information could be found.

The development of HS-AD can be sufficiently demonstrated with a simple timeline (Figure 2.9) and summarized in a few sentences. The development of L-AD technologies to stabilize sludges at wastewater treatment plants and enable energy recovery/renewable energy generation occurred steadily through the mid-20th century and L-AD facilities were widely implemented by the 1970's. Certain facilities began to add fats, oils, and greases and source-separated food waste to enhance energy generation rates. This is an ongoing practice. However, OFMSW landfill bans, landfill taxation, and renewable energy incentives in the EU increased sharply in the 1980's, resulting in high demand for alternative OFMSW treatment technologies and spurring the development of HS-AD systems. As legislation continued to increase and source-separation became common, HS-AD became the primary form of OFMSW digestion in the EU. Around this time, the US began to follow in the footsteps of the EU with the introduction of legislation encouraging OFMSW diversion, recycling, and renewable energy generation. Now, with legislation steadily increasing, trends in HS-AD development are mirroring those of the EU, more HS-AD vendors are doing business in the US, implementation is accelerating, and HS-AD capacity is projected to soon surpass L-AD capacity for processing OFMSW.

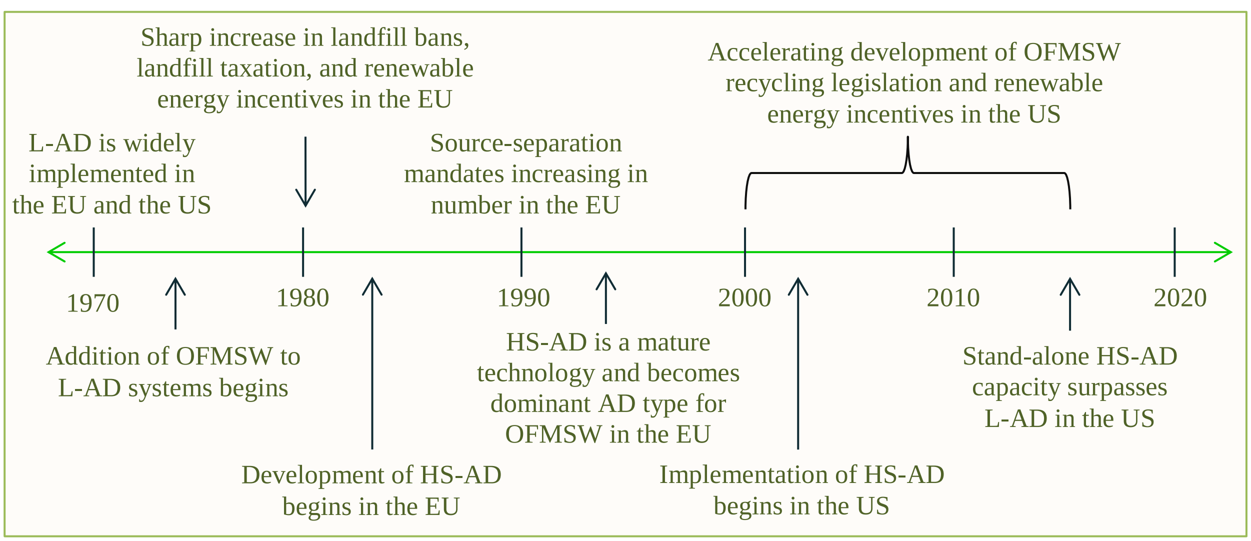

Figure 2.9: Timeline summarizing the development of HS-AD in Europe and the US.

2.4. Enhancing the Biodegradability of Lignocellulosic Wastes in HS-AD

In this section, lignocellulosic waste is defined in greater detail, the role of yard waste in HS-AD is described, and literature relevant to Research Question 3, *Is bioaugmentation using pulp and paper mill anaerobic sludge a viable method for improving methane yields from lignocellulosic wastes in HS-AD?*, is reviewed, including studies on enhancing the biodegradability of lignocellulosic wastes through pretreatment and bioaugmentation strategies.

2.4.1. Lignocellulosic Waste in HS-AD and the Lignocellulosic Challenge

Lignocellulosic waste can be broadly defined as any waste material that contains large quantities of lignin, cellulose, and hemicellulose, including yard waste, wood waste, agricultural plant residues, and any other plant-based waste materials (biomass) (Zheng et al., 2014; Yang et al., 2015). Integration of the lignocellulosic fraction of OFMSW, especially yard waste, with food waste in HS-AD is an essential part of the successful operation of full-scale HS-AD systems processing OFMSW (Rapport et al., 2008; De Baere and Mattheeuws, 2014). The addition of yard waste balances the C/N ratio of the feedstock (as described in Section 2.1.4.8.) and in systems that incorporate percolate recirculation, the addition of yard waste improves the porosity and structure of the waste mixture, which improves mass transfer and digestion efficiency (as described in Section 2.1.4.3.) (Rapport et al., 2008; Li et al., 2011; Yang et al., 2015). Furthermore, the abundance of yard waste generated globally and the widespread pre-existing strategies for segregating yard waste from the rest of the waste stream makes it an excellent candidate for HS-AD. In the US in 2013, approximately 60% of collected yard waste was recycled via composting and mulching (EPA, 2015a). However, these recycling strategies waste the inherent energy in the material as respiration heat. Incorporating this material into HS-AD (and then using it as compost) allows for the recovery of this energy and reduction of the environmental impacts of recycling the material by reducing net GHG emissions and required composting times (Edelman, 2005).

Although the economic and environmental benefits of incorporating lignocellulosic wastes into HS-AD processes are significant, if the biodegradability of the materials could be improved, the benefits would be amplified. The inability to achieve significant degradation of lignocellulosic materials in HS-AD results in low biogas yields from these materials and recovery of only a small fraction of their intrinsic energy (Yang et al., 2015). For example, the average reported methane yield from woody biomass in HS-AD is only around 10% of its theoretical maximum (Jerger and Dolenc, 1982). The cellulose and hemicellulose in lignocellulosic materials are fermentable carbohydrates, but lignin is highly recalcitrant (Zheng et al., 2014). The low biodegradability of lignocellulosic materials is attributed to both the recalcitrance of lignin and the association of cellulose and hemicellulose with lignin, which acts as a barrier to the microbial population that performs hydrolytic conversion of cellulose (Tong et al., 1990; Yang et al., 2015). The challenge of enhancing the biodegradability of lignocellulosic waste in HS-AD has been a main focus of many researchers in the past decade, as revealed by the number of review papers recently published which cover the topic (Mosier et al., 2005; Hendricks and Zeeman, 2009; Zheng et al., 2014; Sawatdeenarunat et al., 2015; Yang et al., 2015). Most of these studies have focused on pretreatment methods, which aim to alter the physical and/or chemical characteristics of lignocellulosic materials for improving biodegradability. However, each pretreatment process incurs additional environmental and economic costs to some extent, and as a result, a number of recent studies have turned their attention to bioaugmentation as a potential alternative to pretreatment.

2.4.2. Pretreatment Methods

Physical, chemical, and biological pretreatment methods have been explored for improving the biodegradability of lignocellulosic waste in HS-AD with varying results (Table 2.18). The mechanisms for improving biodegradability of these materials include reduction of cellulose crystallinity, increasing accessible surface area, disintegration of cellulose, hemicellulose, and lignin complexes, and removal/alteration of lignin (reduction of degree of polymerization) (Mosier et al., 2005; Yang et al., 2015). Physical pretreatments include comminution (particle size reduction), steam-explosion, liquid hot water, extrusion, and irradiation. Chemical pretreatments include alkaline, acid, catalyzed steam-explosion, wet oxidation, oxidative pretreatment with peroxides, and ionic liquid pretreatment. Biological pretreatments include fungal, microbial, enzymatic, and ensilaging (Zheng et al., 2014).

Substrates tested in these studies include agricultural residues (such as wheat straw, rice straw, maize, corn stalks, rape straw, potato pulp, oil palm branches, barley straw, sugarcane bagasse, sunflower stalks, rice stalks, oat straw, clover, bagasse, coconut fiber, hemp, ensiled hay, rapeseed, sugar beet leaves, grape pomace, and greenhouse residues), forest residues (mirabilis leaves and other fallen leaves), hardwoods (birch, willow, and Japanese cedar), softwoods (bamboo, spruce, and pine), grasses (dump grass, grass hay, bulrush, Miscanthu, Miscanthus, seaweed, hybrid grass, switchgrass, cordgrass, wheatgrass, and water hyacinth), mixed yard wastes (mixed fractions of grass, tree and brush trimmings, garden trimmings, leaves, needles, and shredded woody wastes), and MSW (paper tube residuals and pulp and paper sludge).

Table 2.18: Summary of pretreatment experiments for AD of lignocellulosic wastes (adapted from Zheng et al., 2014).

Method	Substrates	Conditions	Results	References
Biological				
Fungal	Agricultural Residues, Hardwood, Yard Wastes	Aerobic; various fungi used; 28-37 °C; 12 days - 8 weeks	15-500% increase in CH_4 yield	Take et al., 2006; Amirta et al., 2006; Muthangya et al., 2009; Mackulak et al., 2012; Zhao et al., 2014
Microbial Consortium	Agricultural Residues	Aerobic; complex microbial agents; 20-55 °C; 12 hours - 20 days	25-96.6% increase in CH_4 yield	Lu et al., 2009; Bruni et al., 2010; Bai et al., 2010; Zhang et al., 2011; Zhong et al., 2011
Enzymatic	Agricultural Residues, Grasses, MSW	Aerobic; laccase, cellulase, hemicellulase, etc., and mixtures; 37 °C; 4-24 hours	0-34% increase of CH_4 yield	Gerhardt et al., 2007; Romano et al., 2009; Bruni et al., 2010; Lin et al., 2010; Zieminski et al.,2012
Ensilaging	Agricultural Residues	Anaerobic; bacteria, enzymes, yeast, fungi; ~37 °C for 7 weeks	15% increase in CH_4 yield; decrease in some cases	Vervaeren et al., 2010
Physical				
Comminution	Agricultural Residues, Forest Residues, Grasses, OFMSW	Particle size: 0.003-30 mm	Up to 30% increase in CH_4 yield; no effect or decrease in some cases	Delgenes et al., 2002; Kaparaju et al., 2002; Bruni et al., 2010; Kreuger et al., 2011
Steam-explosion	Agricultural Residues, Hardwood, Softwood, Grasses, OFMSW	160-260 °C; 0.7-4.8 MPa; seconds - few minutes	Up to 40% increase in biogas yield; no effect or decrease in some cases	Kobayashi et al., 2004; Bruni et al., 2010; Teghammar et al., 2010
Liquid hot water	Agricultural Residues, Grasses, OFMSW	100-230 °C; 0.1-2.8 MPa; few minutes - hours	7-220% increase in CH_4 yield	Chandra et al., 2012a; Chandra et al., 2012b; O-Thong et al., 2012
Extrusion	Agricultural Residues, Grasses, OFMSW	60-90 °C; 0.45-3.5 MPa; 4-12 minutes	8-70% increase in CH_4 yield	Bruckner et al., 2007; Hjorth et al., 2011
Irradiation	Agricultural Residues, Grasses, OFMSW	115-300 °C; few minutes - hours	4-28% increase in CH_4 yield	Cesaro et al,. 2012; Jackowiak et al., 2011a; Jackowiak et al., 2011b
Chemical				
Alkaline	Agricultural Residues, Forest Residues, Hardwood, Softwood, Grasses, OFMSW	NaOH, $Ca(OH)_2$, CaO, KOH, NH_3; 15-170 °C; 1 hr - 10 days	3-230% increase in CH_4 yield; no effect of decrease in some cases	Wang and Wang, 2010; Zhu et al., 2010; Mirahmadi et al., 2010; Liew et al., 2011; Chandra et al., 2012a & 2012b
Acid	Agricultural Residues, Grasses, MSW	H_2SO_4, HCL, HNO_3, H_3PO_4, acetic & maleic acid; 25-170 °C; few minutes - 30 days	20-200% increase in CH_4yield; decrease in some cases	Xiao and Clarkson, 1997; Antonopoulou et al., 2010; Monlau et al., 2013
Catalyzed steam-explosion	Agricultural Residues, MSW	H_2SO_4, SO_2, H_3PO_4, NaOH; 155-220 °C; 5 minutes - 1 hour	18-107% increase in CH_4 yield	Bruni et al., 2010; Teghammer et al., 2010; Kreuger et al., 2011
Wet oxidation	Agricultural Residues, Hardwood, Grasses, MSW, OFMSW	180-220 °C; 0-1.2 MPa; few minutes	34-136% increase in CH_4 yield	Fox et al., 2004; Petersson et al., 2007; Uellendahl et al., 2008
Oxidative with peroxides	Agricultural Residues, Grasses, MSW, OFMSW	H_2O_2 + NaOH; 25-220 °C; few minutes - 7 days	33-120% increase in CH_4 yield; decrease in some cases	Teghammer et al., 2010; Michalska et al., 2012; Song et al., 2012
Ionic liquid	Agricultural Residues, Grasses, Softwood	N-methylmorpholine, N-oxide monohydrate, methylimidazolium chloride; 90-130 °C; 1-15 hours	16%-1200% compared to untreated material	Jeihanipour et al., 2010; Teghammer et al., 2012; Gai et al., 2013; Purwandari et al., 2013

In a comprehensive review of pretreatment studies, Zheng et al. (2014) concluded that very few if any pretreatment methods have conclusively demonstrated to consistently provide enough enhancement of methane yields to outweigh the additional environmental and economic costs they incur. The authors also point out that numerous challenges arise when attempting to integrate proposed pretreatment methods with full-scale AD operations. For example, methods such as irradiation become very expensive and difficult when handling large volumes of waste and methods such as ionic liquid pose challenges associated with regeneration and potential toxicological and inhibitory effects (Zheng et al., 2014). Based on the findings in the review, the following criteria were developed for ideal pretreatment methods: (i) avoid formation of inhibitory compounds (e.g. furfural and phenolic compounds); (ii) require minimal and inexpensive chemicals or water; (iii) avoid costly pretreatment reactors; (iv) require minimal size reduction; (v) require low energy input (yield net positive energy balance); and (vi) avoid the need for waste disposal. Several other reviews (Mosier et al., 2005; Hendricks and Zeeman, 2009; Sawatdeenarunat et al., 2015; Yang et al., 2015) reached similar conclusions as Zheng et al. (2014), stating that the majority of pretreatment processes are not practical at full-scale and that further research is needed to identify and optimize pretreatments that meet similar criteria.

Of each of the pretreatment studies reviewed and summarized in Table 2.18, the methods which best align with the previously listed criteria for optimal pretreatment methods are biological pretreatment methods. Acid and thermal pretreatments have been shown to produce inhibitory compounds such as furfural, thermal pretreatments require high energy inputs, steam-dependent methods require high water inputs, and chemical methods require large quantities of chemicals and often require pretreatment reactors and waste disposal (Mosier et al., 2005; Zheng et al., 2014). Moreover, of the biological methods explored, fungal pretreatment appears to best align with the criteria, ensilaging, enzymatic, and microbial consortium methods all require stringent preparatory work (e.g. extracting enzymes or preparing consortiums) (Vervaeren et al., 2010; Bruni et al., 2010). Fungal pretreatment requires only the addition of common fungi to static lignocellulosic waste piles and next to ionic liquid pretreatment, fungal pretreatment has been shown to yield the most significant enhancements in CH_4 production of all of the pretreatment methods reviewed, as shown in Table 2.18.

In a study on fungal pretreatment of mixed yard waste conducted by Zhao et al. (2014), 30 days of aerobic fungal pretreatment (yard waste mixed with *Ceriporiopsis subvermispora*, a white-rot fungus, and stored in glass bottles open to the ambient air) yielded an enhancement in CH_4 production of 154% in 40 days of HS-AD when compared to non-pretreated yard waste (44.6 L CH_4/kg VS versus 21.6 L CH_4/kg VS). Solids content during pretreatment was also shown to have a significant effect on the enhancement achieved during digestion, with the enhancement decreasing from 154% to 85% (methane yield decreasing from 44.6 L CH_4/kg VS to 32.6 L CH_4/kg VS) when the solids content during pretreatment was reduced from 40% to 35%. The study demonstrated that introducing a commercially available fungal population to yard waste can substantially improve material degradation and energy recovery; however, some degree of process control would be required.

2.4.3. Bioaugmentation Methods

Bioaugmentation is defined as the addition of specialized microorganisms to a microbial community for the enhancement of the community’s capacity to degrade certain compounds or respond to process fluctuations thereby improving treatment (Agarwal, 2005; Tale et al., 2015). Application of bioaugmentation in AD has been explored for improving digester recovery after inhibition (O'Flaherty et al., 1999; O'Flaherty and Colleran, 1999; Tale et al., 2015) and for improving methane yields from animal manure (Angelidaki and Ahring, 2000), distillery wastewater (Savant and Ranade, 2004), lipid-rich wastes (Cirne et al., 2006), sewage sludge mixed with pig manure (Bagi et al.,2007), and lignocellulosic wastes (Yue et al., 2013; Peng et al., 2014). The primary aim of bioaugmentation for improving biodegradability of lignocellulosic waste is to introduce hydrolytic bacteria capable of penetrating lignin, cellulose, and hemicellulose complexes and efficiently hydrolyzing cellulose and hemicellulose. Specific hydrolytic bacteria have been shown to produce specific extracellular enzymes for breaking down specific complex substrates, including proteins, cellulose, hemicellulose, starch, fats, and pectin, (Schnurer and Jarvis, 2009). In the case of hydrolytic cellulose conversion, cellulolytic bacteria such as *Cellulomonas*, *Clostridium*, *Bacillus*, *Thermomonospora*, *Ruminococcus*, *Baceriodes*, *Erwinia*, *Acetovibrio*, *Microbispora*, and *Streptomyces*, produce extracellular cellulase enzymes (Lo et al., 2009). Peng et al. (2014) used *Clostridium cellulolyticum* directly for bioaugmentation of wheat straw in AD, resulting in a 13% enhancement in methane yield. However, in most bioaugmentation studies an alternative inoculum, such as rumen material, is utilized to introduce microbial cultures.

Rumen bacteria are microbial cultures digestion systems of ruminant animals that have the ability to break down plant materials. Scanning electron and atomic force microscopy methods have shown that rumen cultures employ unique mechanisms for degrading lignin, cellulose, and hemicellulose (lignocellulosics), such as rapid production of extracellular substances (e.g. cellulosome and fibrate), adhesion of cellulolytic species to fibers (enhances cellulolytic conversion), and tunneling into fibers (increases flow of nutrients and enzymes to and from lignocellulosics) (Yue et al., 2013). A number of studies have investigated the possibility of leveraging this capability to enhance the biodegradability of lignocellulosic biomass in AD. Hu and Yu (2005) showed that high VS destruction (up to 70%) and lignin destruction (up to 30%) could be achieved with short retention times (~10 days) in AD of corn stover using rumen bacteria as an inoculum source; however, very low quality biogas was generated (< 19% CH_4). The authors suggested that the poor biogas quality may have been a result of overloading of the digesters or low methanogenic activity leading to VFA accumulation. Total VFA concentrations peaked at around 6,000 mg/L (< 10,000 mg/L). Ammonia concentrations were not measured in the study and could have been above inhibitory levels. Similarly, Lopes et al. (2004) showed that increased hydrolysis rates and biogas generation could be achieved through the use of rumen cultures as inoculum in HS-AD of OFMSW, with increases being proportional to the quantity of inoculum added. However, biogas generated in this study was also of low quality (< 43% CH_4). VFA and ammonia concentrations were not measured in this study and the authors made no attempt to explain the low biogas quality.

An alternative source of microorganisms that is has recently been identified as potentially promising for bioaugmentation is anaerobic sludge generated in the treatment of pulp and paper mill waste. This source would not only contain microorganisms acclimated to a high-lignin environment, but is also a waste product typically requiring disposal, which, if capable of enhancing biogas production, would have a newfound beneficial reuse. Mussoline et al. (2013) investigated the possibility of enhancing methane production from rice straw with pulp and paper mill anaerobic sludge. When testing this hypothesis, the theoretical maximum specific methane yields from rice straw of 330 L CH_4/kg VS was surpassed in 92 days of digestion. The specific methane yield of 340 L CH_4/kg VS achieved in this study was 47-74% higher than in similar studies using conventional inocula (e.g. wastewater anaerobic sludge) and was comparable to methane yields achieved in various pretreatment studies, leading to the conclusion that this strategy could be a viable alternative to pretreatment for improving AD of these agricultural residues. This is the only study of its kind and the positive results obtained warrant further exploration of this application and other similar applications, such as AD of OFMSW.

2.4.3.1. Anaerobic Treatment of Pulp and Paper Mill Waste

The anaerobic treatment process of pulp and paper mill waste deserves some attention to provide background information relevant to the use of pulp and paper mill anaerobic sludge as a source of microbes for bioaugmentation in HS-AD. Pulp and paper production is a growing industry and a key component of economies around the world (Mensink, 2007). Anaerobic treatment of wastewater generated in pulp and paper mills is still in its infancy, but is becoming more common as a result of increasingly stringent environmental regulations and economic strain from global completion in the industry (Meyer and Edwards, 2014). Of the ~5,000 pulp and paper mills currently in operation around the world, approximately 400 now have anaerobic wastewater treatment installations, an increase from just over 100 in the year 2000 (Meyer and Edwards, 2014). Installation of onsite anaerobic treatment is expected to continue to increase, by as much as 60% by 2020 (Frost and Sullivan, 2013).

To account for the slow growth rates of anaerobic microorganisms and the demand for treating large volumes of water quickly and economically, a number high-rate reactors that decouple SRTs and HRTs have been developed over the years (Kato et al., 1994). The most successful of these high-rate technologies are upflow anaerobic sludge blanket (UASB) systems, especially advanced versions such as expanded granular sludge bed (EGSB) (Kato et al., 1994). EGSB reactors can handle significantly higher loading rates than other anaerobic treatment technologies because granular sludge displays superior specific methanogenic activity and superior settling characteristics, which allows for extreme decoupling of SRTs and HRTs

(Holshoff Pol et al., 2004). Micromorphology studies have revealed that the unprecedented specific methanogenic activities of granular sludges are the result of close linkages between colonies of acetogenic bacteria and micro-colonies of hydrogenotrophic methanogens allowing for efficient interspecies hydrogen transfer (Holshoff Pol et al., 2004).

Common EGSB reactors in pulp and paper mills include the internal circulation (IC) reactor BIOPAQ IC marketed by Paques, the Biobed EGSB reactor from Biothane, the R2S reactor from Voith, and the external circulation sludge bed reactor from HydroThane (Meyer and Edwards, 2014). The granular anaerobic sludge generated in these reactors has been shown to contain significantly higher fractions of lignin and significantly lower fractions of cellulose than primary (untreated) pulp and paper mill sludge (lignin and cellulose fractions of 36-50 and 19-27% of TS, respectively, in anaerobic sludge and 20-24 and 36-45% of TS, respectively, in primary sludge) (Migneault et al., 2011; Zorpas et al., 2011). This suggests that cellulose is significantly degraded in the anaerobic treatment process (the increase in lignin fraction, by %TS, is likely a result of TS reduction in the digestion process). Furthermore, granular anaerobic sludge generated in these reactors has been shown to contain significantly higher fractions of lignin and cellulose than anaerobic sludge generated in the treatment of primary and waste activated sludge at wastewater treatment plants (less than 0.1 and around 1% of TS, respectively) (Migneault et al., 2011; Zorpas et al., 2011), reaffirming that pulp and paper mill anaerobic sludge is highly adapted to lignocellulosic environments.

In a review of research on AD of pulp and paper waste streams, Meyers and Edwards (2014) concluded that adaptation of anaerobic microorganisms to lignocellulosic material may progress on a time scale of years, and thus, long-term and large scale experiments with pulp and paper mill sludge should be conducted to explore and exploit this phenomenon.

CHAPTER 3: OUTLOOK FOR HS-AD IN FLORIDA

3.1. Introduction

Diversion of the OFMSW from landfills extends the lives of landfills and reduces leachate generation, fugitive methane emissions, and landfill aftercare periods (Kothari et al., 2014). Utilization of OFMSW as a substrate in AD improves the efficiency of energy recovery from the waste when compared with landfill with LFGTE and incineration WtE and enables nutrient recovery in cases when the OFSMW is source-separated (Li et al., 2011). With respect to climate change impacts and cumulative environmental impacts, AD has been shown to be the most sustainable OFMSW management alternative compared with aerobic composting, landfilling with LFGTE, incineration WtE, and ATT technologies (Haight, 2005; Edelmann et al., 2005; Sundqvist, 2005; CIWMB, 2009; Zaman, 2009; Kim and Kim, 2010; Morris et al., 2011; Bernstad and la Cour Jansen, 2012). Furthermore, HS-AD technologies offer additional benefits over traditional L-AD technologies when dealing with OFMSW, including reduced parasitic energy demands, reactor volume requirements, water usage, and excess leachate generation (Li et al., 2011). These findings have translated to observable trends in the MSW management industry in Europe, where AD has developed faster than any other OFMSW treatment technology developed since the 1990's (De Baere and Mattheeuws, 2014). In Europe as of 2014, 244 AD plants processing OFMSW existed with a total capacity of approximately 8 million tons per year (TPY), 62% of the capacity was HS-AD, and approximately 70% of the capacity installed since 2009 was HS-AD (De Baere and Mattheeuws, 2014).

AD of OFMSW has developed slowly in the US relative to in Europe because of the low cost of landfilling and lack of regulatory drivers encouraging organics recycling. However, recent developments in the US with regard to policy development and associated increases in AD of OFMSW (Table 3.1) have paralleled those observed in the EU. Landfilling bans and mandated source-separation of OFMSW have become increasingly common as states across the country strive to improve the overall sustainability of MSW management (EREF, 2015a; EPA, 2015a). These policies have led to an exponential increase in capacity for AD of OFMSW, with stand-alone HS-AD facilities growing in number particularly fast (EREF, 2015a; EPA, 2015a).

Table 3.1: Summary of recent developments regarding improved MSW management in the US.

Category	Status and Observations
Policy Promoting OFMSW Recycling	▪ 20 states now have bans on landfilling yard waste ▪ 5 states now have bans on landfilling food waste (CA, CT, MA, RI, and VT) ▪ 7 states now have landfill diversion targets (CA, CT, DE, FL, MA, MI, and NY) ▪ 209 communities across 16 states offer curbside food waste collection ▪ Certain cities (e.g. San Francisco, CA and Seattle, WA) and states (VT and CT) now have laws *requiring* source-separation of OFMSW ▪ Numerous states, cities, corporations, etc., now have sustainability statements or an equivalent that include reducing waste and increasing recycling rates
AD of OFMSW	▪ 181 AD facilities in the US are now processing OFMSW ▪ 156 are codigestion (wastewater treatment or on-farm systems accepting some OFMSW) ▪ 25 are stand-alone facilities developed specifically for OFMSW processing ▪ 52% of capacity is currently stand-alone and this fraction is projected to increase
HS-AD Development	▪ 47% of existing stand-alone capacity is HS-AD ▪ 7 of the 8 existing HS-AD facilities were constructed within the past 4 years (since 2011) ▪ 19 or more HS-AD facilities are projected to come online by 2017, making HS-AD the dominant AD type for processing OFMSW

Note: Above data acquired from BioCycle (2013), GMI (2014), Yepsen (2015), EREF (2015a), and EPA (2015a)

The overall goal of this study was to evaluate the potential for HS-AD implementation in Florida. Florida is an appealing state for conducting an assessment of this sort because of the large population, high energy demands, high OFMSW generation rates, a statewide recycling goal of 75% by 2020, and a current food waste recycling rate of only around 7% (FDEP, 2015a). The warm climate in Florida is economically advantageous for AD because high ambient temperatures reduce the amount of heat energy needed to maintain internal operating temperatures (Tchobanoglous et al., 2003). Furthermore, Florida MSW management subject matter experts have recently expressed preference toward AD over alternative biological technologies (e.g. composting), as depicted by Yasar and Celik (2016) using the Advanced Hierarchy Process. The specific objectives of this study were to:

1. Identify locations where HS-AD implementation would be most suitable in Florida based on OFMSW generation and recycling rates and existing MSW infrastructure;
2. Quantify the current economic and environment incentives for HS-AD implementation in Florida and identify key barriers;
3. Provide policy recommendations and outline possible strategies for improving the economic competitiveness of HS-AD in Florida.

3.2. Methodology

This assessment was conducted using information on OFMSW generation, disposal, and recycling rates, existing OFMSW recycling infrastructure, and existing policies relevant to OFMSW recycling in the state of Florida obtained from the Florida Department of Environment Protection (FDEP, 2011; FDEP, 2013; FDEP, 2015a; FDEP, 2015b) and various other sources (Kessler, 2009; Dieleman, 2015). Energy and nutrient recovery attainable through HS-AD implementation was estimated from assumed values obtained from "grey" and published literature and reported values from industry (Table 3.3). Greenhouse gas (GHG) offsets were estimated based on calculated electricity production potential (Table 3.4), approximate GHG offsets achievable per unit electricity produced via HS-AD (SGC, 2012), and documented GHG emissions per unit electricity generated via the existing electricity grid in Florida (EPA, 2013b). Policy recommendations and possible strategies for improving the competitiveness of HS-AD were derived from "grey" and published literature and industry sources (RIS, 2005; PIS, 2008, Rapport et al., 2008; FIE, 2009; Rogoff and Clark, 2014; CalRecycle, 2014b; EPA, 2015d).

3.3. Results and Discussion

3.3.1. State of OFMSW Recycling in Florida

According to the most recent FDEP Solid Waste Management annual report (2015), 34.4 million tons of MSW were collected in Florida in 2014, nearly 20% of which was OFMSW (2.2 million tons of food waste and 3.7 million tons of yard waste). In 2008, the Florida Legislature enacted House Bill 7135, establishing a new statewide recycling goal of 75% to be achieved by 2020. Recycling OFMSW is a particularly important undertaking when considering both the GHG emissions related to OFMSW biodegradation and the potential for nutrient and energy recovery from OFMSW (Kessler, 2009). As of 2014, statewide food waste and yard waste recycling rates in Florida were 7% and 51%, respectively (FDEP, 2015a). Considering the current recycling rates and the relative fractions of food waste and yard waste generated in Florida (of total MSW generation) of 7% and 12%, respectively, Florida's statewide recycling rate could be increased by as much as 13% (from 50% to 63%) through OFMSW recycling. Figure 3.1 shows the categorized composition and management of MSW in Florida in 2014. Figure 3.2 displays the counties of Florida, categorized by population and 2013 recycling rates.

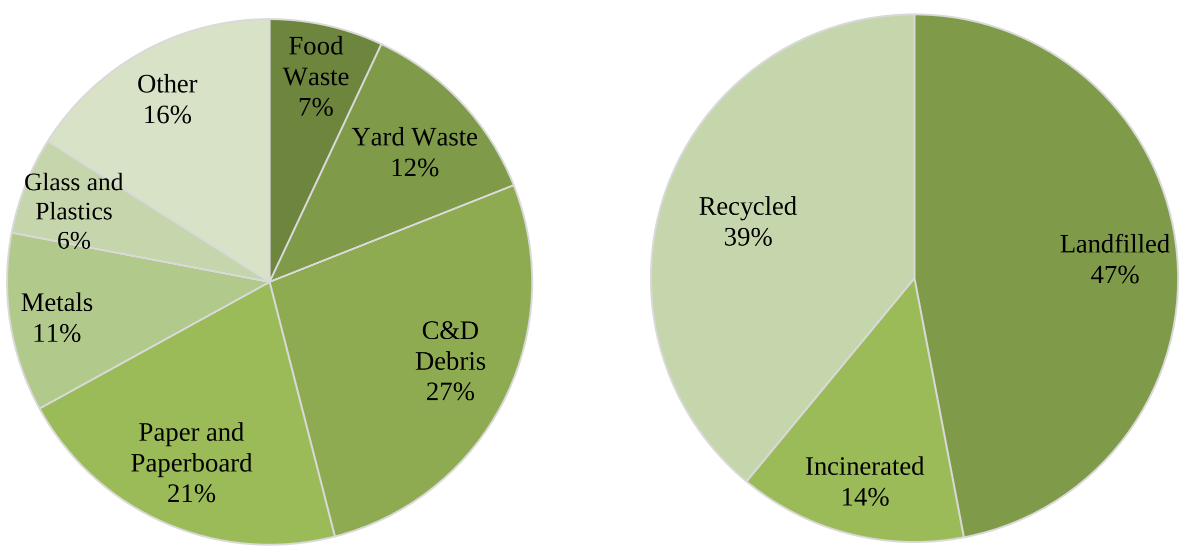

Figure 3.1: 2014 composition and management of MSW in Florida (adapted from FDEP, 2015).

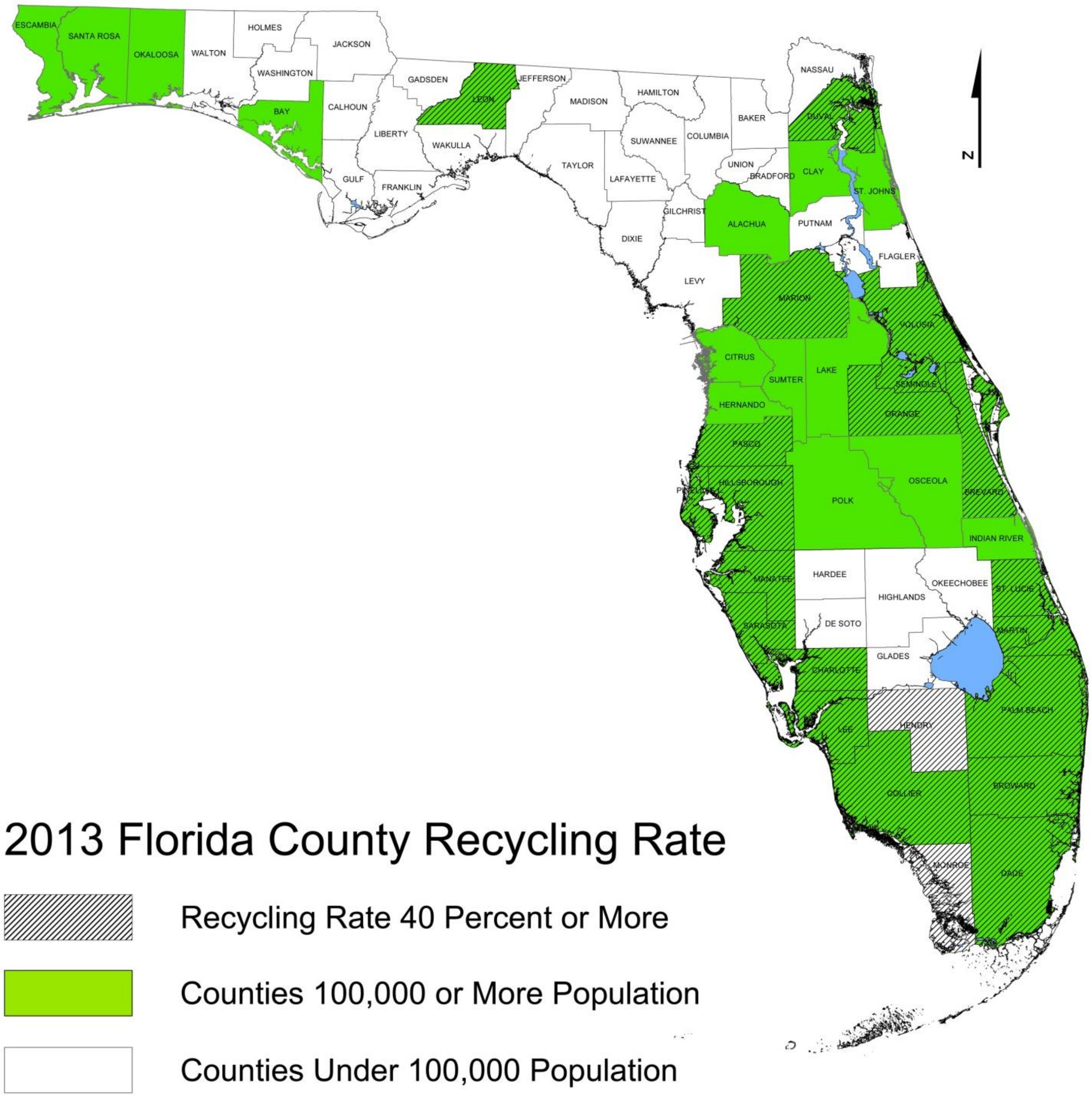

Figure 3.2: Florida counties classified by population and recycling rate as of 2013 (Price, 2015).

3.3.1.1. Florida's Definitions of OFMSW Recycling

The FDEP defines recycling as "any process by which solid waste, or materials that would otherwise become solid waste, are collected, separated, or processed and reused or returned to use in the form of raw materials or products" (Florida Statute 403.703). Florida Statute 403.7032 states that "solid waste used for the production of renewable energy" qualifies as recycling to be counted toward the 75% recycling goal. The FDEP defines renewable energy as "electrical energy produced from a method that uses one or more of the following fuels or energy sources: hydrogen produced from sources other than fossil fuels, *biomass*, solar energy, geothermal energy, wind energy, ocean energy, and hydroelectric power". Biomass is defined by the FDEP as "a power source that is comprised of, but not limited to, combustible residues or gases from forest products manufacturing, waste, byproducts, or products from agricultural and orchard crops, waste or coproducts from livestock and poultry operations, waste or byproducts from food processing, urban wood waste, municipal solid waste, municipal liquid waste treatment operations, and landfill gas" (Florida Statue 366.91). From these definitions, it follows that incineration WtE and LFGTE both count as recycling along with traditional recycling forms (metals, plastics, glass), composting, AD, and bioenergy generation via ATT. This renewable energy is factored into overall recycling rates, with every MWh of energy generated from waste counting as one ton of waste recycled (or 1.25 tons for counties with high traditional recycling rates) (FDEP, 2015a). It should be noted that the FDEP's definition of recycling conflicts with the EPA's. In the EPA's most recent MSW report (EPA, 2015a), incineration and landfilling are both, in all cases, counted as disposal and not counted as recycling.

3.3.1.2. Existing OFMSW Recycling Infrastructure

The majority of yard waste recycling in Florida is accomplished through separate collection of yard waste and management at specified yard waste processing centers where the material is shredded and distributed for use as mulch (garden/landscape bedding), process fuel, or alternative daily landfill cover. According to the FDEP's Source Separated Organics Processing Facility Database, there are currently 273 facilities permitted to process yard waste, all but 56 of which are permitted to recycle the yard waste (FDEP, 2015b). HS-AD can potentially be pared with this type of yard waste recycling as an initial recycling step for energy recovery, as outlined by Sawatdeenarunat et al. (2015). All other existing infrastructure for OFMSW recycling (L-AD, compost, bioenergy, WtE, and LFGTE facilities) that could be identified in Florida was mapped using Google Earth (Figure 3.3).

Composting is one of the most common technologies for OFMSW recycling in the US (EPA, 2015a). The FDEP encourages local governments to provide public education on composting and develop organics source-separation and composting programs (Florida Statute 403.706). Additionally, Florida Statute 403.714 proclaims that state agencies are responsible for the development of compost markets and "are *required* to procure compost products when they can be substituted for, and cost no more than, regular soil amendment products". However, composting in Florida has been slow to develop due to a lack of markets for compost products (Kessler, 2009). In 2008, only four permitted composting facilities existed in the state and the only significant forms of food waste recycling were recovery via a network of collection services, food banks, and soup kitchens and animal feed production from preconsumer food waste (one facility) (Kessler, 2009). However, with the help of the Florida Organics Recycling Center for Excellence (FORCE), the enactment of the 75% recycling goal bill in 2008, and revised regulation allowing for the combined composting of yard waste and food waste, the number of active permitted composting facilities increased to 24 by 2012, with 10 registered to accept both food and yard waste (Kessler, 2009; Zimms and Ver Eecke, 2012).

Currently, there are 14 active permitted "source-separated organics composting" facilities listed in the FDEP database (FDEP, 2015b). Of these facilities, 13 are permitted to accept yard waste, 12 are permitted to accept "vegetative waste", and 11 are permitted to process "pre-consumer vegetative waste" (FDEP, 2015b). The only facility that is not permitted to compost vegetative waste or preconsumer vegetative waste (My World Nursery) was the only facility that was confirmed to *not* be actively composting. The definitions of "vegetative" and "pre-consumer vegetative" waste could not be found but are assumed to include fruit and vegetable waste. Based on these numbers and this assumption, the total number of permitted composting facilities has decreased since 2012, but the number of facilities processing both food and yard wastes has increased. Several additional facilities that claim to be actively composting were identified through web searches and inquiries with industry professionals, including: George B. Wittmer Associates, Inc. facility (Nassau County), Okeechobee Landfill, JFE-Brighton Regional Composting facility (Brighton Seminole Indian Reservation), and MW Horticulture Recycling (two locations in Lee County) (Wittmer, 2015; WM, 2015; McGill, 2015; MWHR, 2015).

The Reedy Creek Improvement District (RCID) composing facility is the only permitted facility not permitted to accept yard waste. The RCID facility, though initially a yard and food waste composting facility and still permitted as a "source-separated organics composting" facility, is now the state's first and only AD system operated for processing OFMSW. The system, which began operation in 2012, is an L-AD system collocated with the District's wastewater treatment plant (WWTP). This allows for low-cost transfer of biosolids from the WWTP to the L-AD system and centrate (leachate/percolate) from the L-AD system to the WWTP after nitrogen (N) and phosphorous (P) removal (Sorensen, 2014). Although the system is of the L-AD variety, it sets precedence for AD of OFMSW in Florida. The system has a processing capacity of 130,000 tons per year (TPY), processes source-separated food waste fats, oils, and greases (FOG) from nearby industrial, commercial, and institutional (IC&I) sources and biosolids from the WWTP. The system produces 3.2 MW of electrical energy, 2.2 MW of recoverable heat via a CAT combined heat and power (CHP) engine-generator system, and approximately 6,600 TPY of granular fertilizer product that meets EPA AA standards for a product containing biosolids (Sorensen, 2014). The facility is also listed by the FDEP as a permitted “Bioenergy” facility (FDEP, 2011).

Several other bioenergy projects (wood-fired power and ATT) have been considered in Florida in recent years, with 22 separate permitted projects listed by the FDEP (2011). Of the 22 permitted bioenergy projects, however, only a few have come to fruition: Gainesville Renewable Energy Center (100 MW wood-fired power), INEOS New Plant Bioenergy (hybrid gasification-fermentation 8 million gallons per year [MGY] ethanol production), and Brooksville Central Power and Lime (70 MW wood-fired power). The FDEP (2011) lists four of the 22 permits as cancelled or withdrawn, but web searches reveal that numerous other projects have been canceled. For example, the Saint Lucie Plasma Gasification project and the Verenium Ethanol project in Highland County were cancelled due to economic challenges (Blandford, 2012; Lane, 2012) and the Adage wood-fired power plant was cancelled due to challenges with public opposition (Sheehan et al., 2011). The high capital cost, technical complexity, and lack of well-established economic/cost-benefit data of bioenergy projects relative to alternative technologies such as incineration WtE are key hurdles that must be overcome for further development of ATT projects (EREF, 2013).

Florida has a longstanding reputation for WtE development, with 12 of the 80 WtE plants in operation in the US (as of 2013) located in the state (FDEP, 2013; Wheelabrator, 2015; EPA, 2015a). This reputation is being upheld, with some controversy and opposition, as the state’s 13th WtE facility is being constructed in West Palm Beach – the first WtE plant to be constructed in the US in more than 20 years (Williams, 2015). Florida is also among leading states in terms of LFGTE, with 20 landfills currently equipped for LFGTE, 18 of which produce electricity and four of which do direct use (two produce electricity and do direct use) (Dieleman, 2015). Another three landfills have plans to implement LFGTE systems (Central County LF, North Dade LF, Saint Cloud City LF) and another 13 are considered as candidate landfills for future LFGTE projects (Dieleman, 2015). As previously mentioned, this form of “recycling” does not fall within traditional definitions, and by many accounts, reduces incentive for waste reduction and traditional recycling.

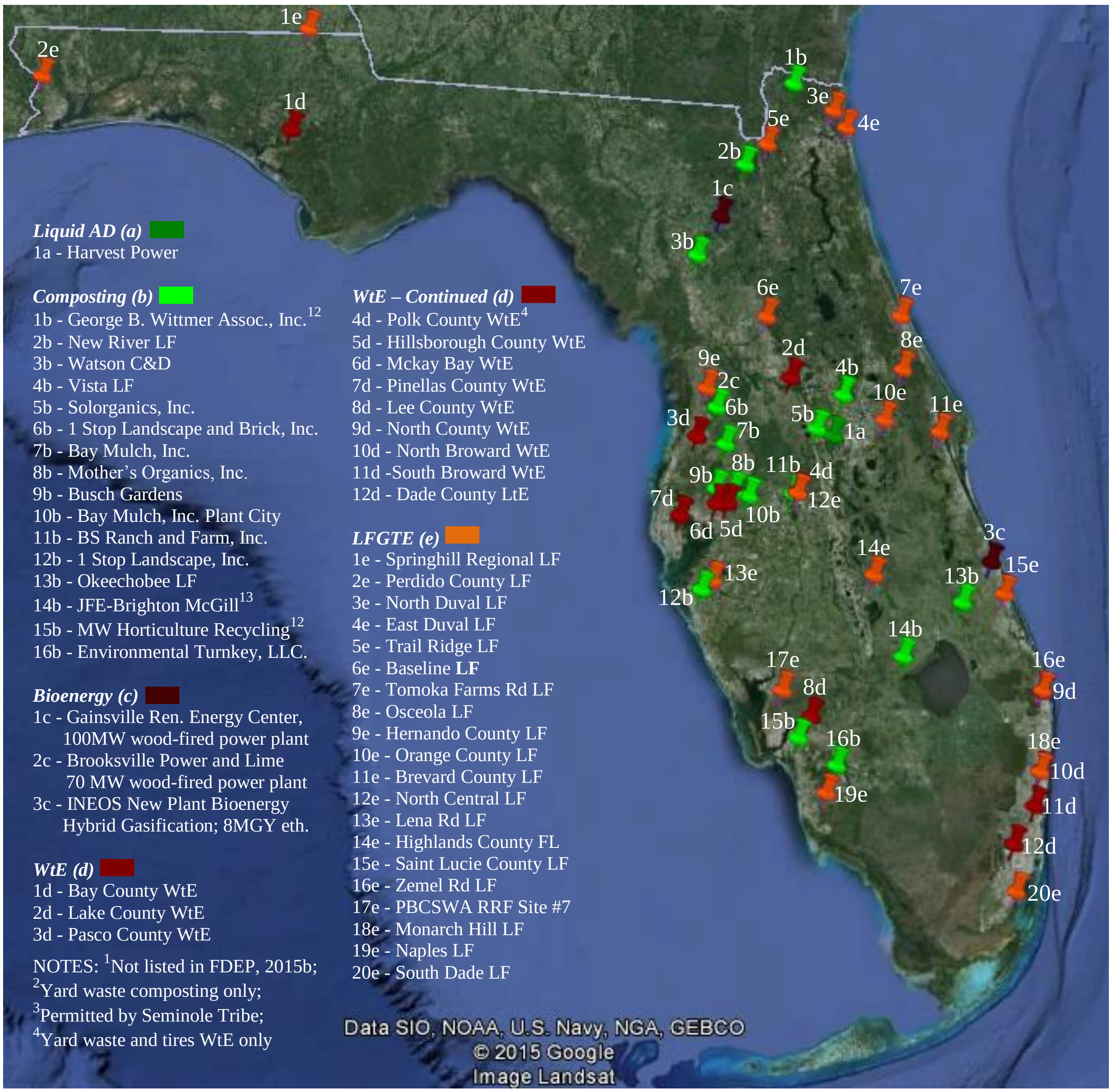

Figure 3.3: OFMSW recycling facilities in Florida, excluding yard waste processing centers.

3.3.1.3. Florida Counties Where HS-AD is Recommended

The FDEP reports generation and recycling rates on a per-county basis. For this reason, the counties of Florida were assessed in this study for their relative suitability for HS-AD implementation. It is well-established that availability of large quantities of pure feedstock (e.g. minimally contaminated food waste and yard waste) in close proximity to HS-AD system location is one of the most critical factors affecting the economic feasibility of HS-AD (Rapport et al., 2008; Rogoff and Clark, 2014). In counties with small populations and low population density, it is unlikely that sufficient quantities of OFMSW (>5,000 TPY) could be aggregated at a single site in an economically feasible manner. For this reason, all counties with less than 100,000 people were disregarded in this assessment. Table 3.2 displays food waste and yard waste generation and recovery rates in the remaining 34 counties (FDEP, 2015a).

Miami-Dade, Broward, Palm Beach, Hillsborough, Orange, Pinellas, Duval, and Lee are the top eight most populated counties in Florida and consistently rank in the top nine with respect to OFMSW generation (total amount) and disposal (unrecycled amount), as shown in Table 3.2. Alachua County, the home county of the University of Florida and the 23rd most populated county in the state, ranks 10th in terms of OFMSW generation and 7th in terms of unrecycled OFMSW. It is these nine counties that are most suitable for HS-AD implementation. Of these counties, all had reached 40% overall recycling rates by 2013 except Alachua County (Figure 3.1) and each has a unique mix of OFMSW recycling infrastructure, but none have significant capacity for recycling food waste. Each of the counties, except Hillsborough and Pinellas, have at least one landfill with LFGTE and each, except Alachua, Orange, and Duval, have at least one WtE facility. However, only two of these counties have an existing bioenergy plant (Alachua and Orange) and few have composting facilities (Alachua, Orange, Hillsborough, and Lee).

Table 3.2: Yard waste and food waste generation and recycling in 2014 in Florida counties with populations greater than 100,000, ranked in descending order by population (FDEP, 2015a).

County	Yard Waste			Food Waste			Potential Feedstock			
	Generated	Recycled	Recycling Rate	Generated	Recycled	Recycling Rate	Total		Unrecycled	
							Amount	**Rank**	Amount	**Rank**
Miami-Dade	517	363	70%	178	0	0%	695	**1**	332	**2**
Broward	190	29	15%	193	5.9	3%	383	**4**	348	**1**
Palm Beach	208	48	23%	120	0.7	1%	328	**7**	279	**4**
Hillsborough	187	118	63%	117	4.4	4%	304	**8**	182	**8**
Orange	103	68	66%	324	75	23%	427	**2**	284	**3**
Pinellas	248	165	67%	159	3.6	2%	407	**3**	238	**6**
Duval	271	104	38%	99	1.2	1%	370	**5**	265	**5**
Lee	158	104	66%	121	0.7	1%	279	**9**	174	**9**
Polk	62	42	68%	74	2.6	4%	136	**15**	91	**13**
Brevard	322	283	88%	40	0.9	2%	362	**6**	78	**14**
Volusia	68	46	68%	83	2.0	2%	151	**13**	103	**12**
Pasco	46	19	41%	38	0.9	2%	84	**19**	64	**17**
Seminole	73	45	62%	41	1.6	4%	114	**17**	67	**16**
Sarasota	77	65	84%	41	4.1	10%	118	**16**	49	**22**
Manatee	103	29	28%	44	0	0%	147	**14**	118	**10**
Marion	48	23	48%	21	0.9	4%	69	**22**	45	**23**
Collier	128	127	99%	42	1.2	3%	170	**11**	42	**24**
Lake	33	1	3%	19	0.3	2%	52	**25**	51	**21**
Escambia	41	10	24%	24	0	0%	65	**23**	55	**19**
Osceola	18	0	0%	16	0	0%	34	**29**	34	**26**
St. Lucie	20	3	15%	12	0.6	5%	32	**31**	28	**27**
Leon	46	15	33%	27	2.3	9%	73	**20**	56	**18**
Alachua	198	19	10%	24	0.3	1%	222	**10**	203	**7**
St. Johns	54	16	30%	17	0.1	1%	71	**21**	55	**20**
Clay	24	24	100%	20	0.3	2%	44	**28**	20	**31**
Okaloosa	29	18	62%	16	0.5	3%	45	**27**	27	**28**
Hernando	26	11	42%	6.5	0.6	9%	33	**30**	21	**30**
Bay	24	5	21%	23	0.6	3%	47	**26**	41	**25**
Charlotte	40	35	88%	17	0.7	4%	57	**24**	21	**29**
Santa Rosa	15	4	27%	6.8	0.3	4%	22	**32**	18	**32**
Martin	107	4	4%	48	42	88%	155	**12**	109	**11**
Indian River	84	25	30%	19	0.7	4%	103	**18**	77	**15**
Citrus	11	10	91%	11	0.6	5%	22	**32**	11	**34**
Sumter	9	0	0%	6.7	0.4	6%	16	**34**	15	**33**

Note: Values are expressed in thousands of tons per year (excluding percentages and ranks); "Unrecycled" values were calculated by subtracting amount recycled from amount generated; WtE and LFGTE recycling credits are *not* included in these numbers, so the "Unrecycled" quantities are representative of the amount disposed via incineration and landfilling

3.3.1.4. Locations and Funding Sources for HS-AD Demonstration

There are numerous prospective locations that could serve as suitable sites for a full-scale HS-AD demonstration project. For example, the University of South Florida generates large quantities of OFMSW and is surrounded by IC&I sources of additional OFMSW (several hospitals, grocery stores, and elementary schools), the majority of which is transported and processed at the Hillsborough County Resource Recovery Facility (incineration WtE with separate yard waste processing). This could serve as an excellent centralized site for an educational demonstration facility. Additionally, HS-AD can be synergistically paired with most existing MSW management infrastructure, including material recovery facilities, landfills with LFGTE, composting facilities, and most bioenergy facilities. HS-AD can be easily paired with composting operations to enable energy recovery, reduce waste volume, and increase total facility throughput/capacity (De Baere and Mattheeuws, 2014; Kraemer and Gamble, 2014). This is apparent in recent developments in the Netherlands and Belgium, where approximately 80% of all composting operations have incorporated AD as a primary treatment technology (De Baere and Mattheeuws, 2014). Specific candidate composting operations could include the Okeechobee Landfill site, which has the capacity to process 30,000 TPY of source-separated OFMSW, and the Vista Landfill site in Orlando County, which is permitted to process 45,000 TPY of OFSMW and was processing approximately 22,000 TPY as of 2012 (Zimms and Ver Eecke, 2012).

Landfills equipped with LFGTE are also appealing for an HS-AD demonstration project. The advantage of this strategy is that biogas from HS-AD systems at landfill sites can be tied into existing LFGTE infrastructure to reduce the capital costs of an HS-AD project, improve energy recovery efficiency at landfills, simplify collection schemes for HS-AD, and reduce waste volume/enable low-cost disposal of digestate (in cases where feedstocks are mixed MSW or mechanically-separated OFMSW) (Rapport et al., 2008; Zaman, 2009; Li et al., 2011). There are at least three existing HS-AD systems in California, for example, that are located at or adjacent to landfills (Monterey, San Jose, and Davis). The development of bioenergy facilities is somewhat in competition with HS-AD, because both technologies partly depend on yard waste as a feedstock. However, as outlined by Sawatdeenarunat et al. (2015) and Pan et al. (2015), yard waste digested via AD can still be used as a feedstock for bioconversion (thermal and/or chemical).

With regard to potential project funding sources, the INEOS bioenergy plant in Indian River County (Figure 3.3) was partially funded by the US Department of Energy, as was the HS-AD system that began operation in 2012 at the University of Wisconsin, Oshkosh. Furthermore, the US EPA and the USDA have existing and forthcoming programs for funding biogas projects (USDA/EPA/DOE, 2014). Florida based grant programs also exist for funding MSW management, recycling, and renewable energy projects. For example, the FDEP has an Innovative Recycling/Waste Reduction Grant Program, a Florida Recycling Loan Program, and a Small County Consolidated Solid Waste Grant Program, and the Florida Department of Agriculture and Consumer Services offers funding through their Research and Development Bioenergy Grant Program. Other HS-AD project funding sources can come from private industry, as demonstrated in numerous HS-AD projects across the country. The partnerships developed for the Harvest Power L-AD facility (between the owner, the technology vendor, and the utility company – who agreed to purchase the energy generated by the facility) is an example of a necessary partnership for economically sustainable AD of OFMSW (Rapport et al., 2008).

3.3.2. Quantified Incentives for HS-AD Implementation

The environmental incentives for HS-AD implementation, with respect to nutrient and energy recovery potential and GHG offset potential, and the associated economic incentives, were estimated using the 2014 statewide food and yard waste generation rates of 2.2 million tons and 3.7 million tons, respectively (FDEP, 2015a), and the assumed values listed in Table 3.3.

Table 3.3: Assumed values for quantifying the environmental and economic incentive for implementation of HS-AD for OFMSW recycling in Florida.

Parameter(s)	Assumed Value(s)	Reference(s)
VS Content	Food Waste = 15% by wet weight Yard Waste = 60% by wet weight	Kothari et al., 2014
Average Biogas Yield	Food Waste = 0.5 m^3/kg VS Yard Waste = 0.3 m^3/kg VS	Kothari et al., 2014
Biogas Quality	60% CH_4	Kothari et al., 2014
Energy Equivalence of CH_4 and Diesel Fuel	9.7 kWh/m^3 CH_4 9.8 kWh/L Diesel	SGC, 2012
Combined Heat and Power Conversion Efficiency	35% to electricity ~40% to heat	SGC, 2012
CNG Conversion Efficiency	67%	ZWE, 2013b
GHG Offsets from Substituting Fossil Fuel with Biogas	100% - 120%	SGC, 2012
Value of Carbon Credits	Voluntary Market Rate = \$4.90/MTCO$_2$E Total Value (including ecological offsets) = \$664/MTCO$_2$E	ICROA, 2014
Mass Destruction in HS-AD	40%	BIOFerm, 2014a
Bioavailable Nutrient Content of HS-AD Digestate	N = 1% by dry weight P = 0.5% by dry weight	Hartz, 2009
Agricultural Value of Nutrients	N = \$0.26/kg N P = \$0.14/kg P	WERF, 2011

The total energy recoverable from food waste and yard waste generated in Florida comes out to over 500 MW (4,000 GWh/year) after applying the values listed in Table 3.3 as shown in Table 3.4. If the CH_4 generated in the HS-AD systems were to be used in combined heat and power (CHP) units, this translates to an annual electricity generation potential of approximately 175 MW (1,500 GWh/year), with a significant portion of the remaining energy (~ 40%) being recoverable heat for maintaining internal temperatures of HS-AD systems and for district heating. Or, if the CH_4 were to be converted to compressed natural gas (CNG), nearly 80 million diesel gallon equivalents (DGE) of CNG could be produced. To put this context, Florida currently generates a total of 246,000 GWh/year of electricity, 5,000 GWh/year of which is renewable (EIA, 2015a), and consumes 688 million DGE of CNG per year as vehicle fuel (EIA, 2015b). Thus, through HS-AD, either around 0.6% of Florida's electricity demand could be fulfilled, increasing statewide renewable electricity generation by roughly 30%, or around 11.5%

of CNG vehicle fuel demand could be fulfilled (Table 3.4). Assuming the recovered energy would be used to produce electricity and that the parasitic energy demand of an HS-AD system is 20%, the excess electricity produced (1,200 GWh/year) could generate more than \$120M in revenue annually (at \$0.10/kWh) not including revenues from GHG offsets or nutrient recovery.

Table: 3.4: Approximate energy recovery potential through HS-AD of OFMSW in Florida.

	Yard Waste	Food Waste	Total
Assumed Generation Rate (short tons/year) =	3,700,000	2,200,000	5,900,000
Assumed Volatile Solids Fraction (% by wet weight) =	0.60	0.15	
Assumed Biogas Generation (m^3/kg VS) =	0.30	0.50	
Total Energy Content (GWh/year) =	3,520	870	**4,390**
Total Electricity Generation Potential (GWh/year) =	1,230	300	**1,530**
Total Electricity Generation in Florida (GWh/year) =			246,200
Fraction of Florida Electricity Demand Fulfilled =	0.5%	0.1%	**0.6%**
OR:			
CNG Generation (DGE/year) =	63,400,000	15,700,000	**79,100,000**
Total CNG Consumption in Florida (DGE/year) =			688,000,000
Fraction of Florida CNG Demand Fulfilled =	9.2%	2.3%	**11.5%**

Note: Assumes 9.7 kWh-m^{-3} CH_4, 9.8 kWh-L^{-1}diesel, 35% electrical conversion efficiency, and 67% CNG conversion efficiency; mass conversion factor = 907 kg per short ton

According to the US EPA (EPA, 2013b), Florida's electricity-based GHG emissions in 2013 accumulated to 103.4 million metric tons of carbon dioxide equivalence, which translates to approximately 430 metric tons per GWh of electricity produced. Applying an assumed 100% reduction in GHG emissions resulting from substituting energy from the existing energy grid with biogas-derived energy (SGC, 2012) to the estimated electricity production potential through HS-AD of OFSMW of 1,535 GWh/year, an estimated GHG offset potential of 660,000 metric TPY of CO_2 equivalents ($MTCO_2E$/year) is obtained. If these offsets were sold as carbon credits in the voluntary market, an additional \$3.2M worth of annual revenue could be generated (at \$4.90$MTCO_2$/year). When taking into account the economic value of the ecological benefits associated with GHG offsets, the value increases to over \$400M per year (at \$664$MTCO_2$/year). Assuming 40% mass reduction in the HS-AD process, approximately 3.5 million TPY of high quality soil amendment can be generated, equating to 7,000 TPY and 3,500 TPY of recoverable N and P, respectively, or $2.1M per year worth of fertilizer offsets (at $0.26/kg N and $0.14/kg P) (Table 3.5). These estimates are easily scalable by inputting alternative annual food waste and yard waste processing values for any scale of interest (e.g. 5,000 TPY facility with approximately 50/50 food waste/yard waste processing, 100,000 TPY facility, county-wide, 75% state-wide OFMSW recycling, and so on) to provide estimates of potential economic and environmental benefits achievable through HS-AD implementation (see Appendix D).

Table 3.5: Nitrogen and phosphorous recovery potential through HS-AD of OFMSW in Florida.

	Nitrogen	**Phosphorous**
Assumed Digestate Generation Rate (short tons/year) =	3,540,000	3,540,000
Assumed Total Solids Content (%) =	20%	20%
Assumed Available Fraction (%) =	1.0%	0.5%
Nutrient Recovery Potential (short tons/year) =	**7,080**	**3,540**

Note: Assumes 40% mass reduction in HS-AD; mass conversion factor = 907 kg per short ton

3.3.3. Economics – The Key Barrier to HS-AD Implementation

For potential environmental and economic advantages achievable through HS-AD to be realized, barriers to HS-AD implementation must be identified and overcome. The single greatest barrier to HS-AD implementation is the relative cost of managing OFMSW via HS-AD as compared to other leading alternatives (e.g. landfill, incineration, and composting). SCS Engineers recently conducted an economic analysis aiming to estimate tipping fees required to ensure economically sustainable HS-AD operation (Rogoff and Clark, 2014). In the analysis, total capital cost requirements were estimated (including design, permitting, materials, equipment, and construction), operations and maintenance costs were estimated, inflation rates and financing costs were incorporated, revenue from compost sales and GHG offsets were neglected, and required tipping fees (break-even) were calculated for four different scenarios (Table 3.6).

Table 3.6: Approximate break-even tipping fees for four different HS-AD project scenarios (Rogoff and Clark, 2014).

Scenario	Plant Capacity	Electricity Production	Tipping Fee Required
1	5,000 TPY	None	\$45.92 - \$53.16
2	5,000 TPY	203 kWh/ton @ \$0.1044/kWh	\$8.76 - \$31.97
3	10,000 TPY	None	\$40.73 - \$48.53
4	10,000 TPY	203 kWh/ton @ \$0.1044/kWh	\$3.57 - \$27.34

To provide context to the results of Rogoff and Clark's model results, general economic data for various MSW management technologies, adapted from PIS (2008), are shown in Table 3.7. In contrast to the values shown, the World Bank (2012) reports that the costs of landfilling, incineration, composting, and AD in high income countries per ton of waste processed range from \$40-100, \$70-200, \$35-90, and \$65-150, respectively. According to these values, and reflected in MSW management practices, landfilling is the lowest cost management option, composting is sometimes comparable, and all other options are significantly more expensive.

In the US, average nationwide landfill tipping fees in 2013 were \$49.78 per ton, down slightly from \$49.99 per ton in 2012 (EPA, 2015a). In Florida in 2013, the average landfill tipping fee was \$43.65 and the lowest rate in the state was \$25.50 (CEP, 2014). From the results of the Rogoff and Clark (2014) *Pro Forma* economic model, it can be concluded that HS-AD becomes economically competitive with increases in processing capacity and that energy sales are a critical factor. The authors mention that when incorporating revenues from GHG offsets (e.g. carbon credits or renewable energy certificates [RECs]), HS-AD projects could reliably yield short pay-back periods and provide returns on investments for developers. The authors further concluded that several factors would have to converge for HS-AD to be economically feasible in Florida, including: high quantities of quality feedstock, high power costs, utility economic incentives, markets for compost, markets for carbon credits and/or RECs, and bans on organics disposal in landfills. This conclusion parallels those drawn in multiple economic analyses of this kind (RIS, 2005; PIS, 2008, Rapport et al., 2008; FIE, 2009; RWI, 2013; Rogoff and Clark, 2014).

Table 3.7: Estimated capital and O&M costs for a select processing capacity or capacity range of MSW management technologies (adapted from PIS, 2008).

Technology	Plant Capacity *(ton/day)*	Capital Cost *($/ton)*	O&M Cost *($/ton)*
Landfill	500	5-15	10-30
Incineration	1,300	30-180	80-120
Pyrolysis	70-270	16-90	80-150
Gasification	900	15-170	80-150
Composting (In-Vessel)	500	50-80	30-60
Anaerobic Digestion	300	20-80	60-100

3.3.4. Policy Recommendations and Strategies for Improving Economics

Several policy-related, market-related, and design-related factors could potentially tip the scale toward economically viable HS-AD. Optimizing system and process designs and operations is the low-hanging fruit, so-to-speak, with respect to optimizing HS-AD economics. Certain HS-AD technologies have been shown to have lower parasitic energy demands than others (see Section 2.3.3.), for example, and certain technologies have been shown to generate higher biogas yields from a given feedstock than others. However, no single technology has emerged as superior relative to others with respect to overall system performance.

Codigestion strategies are another means of improving HS-AD economics. Certain codigestion strategies, such as codigestion of food waste and yard waste at certain ratios, have been shown to improve environmental conditions (e.g. C/N ratio and feedstock porosity) and enhance system performance (see Section 2.1.4.2.). Other codigestion strategies, such as the incorporation of biosolids as a co-substrate, can provide enhanced revenue in the form of increased tipping fees. Biosolids management in Florida is an increasingly expensive endeavor with relatively limited capacity for L-AD of biosolids, land application regulations becoming increasingly stringent, and the costs of biosolids disposal in landfills being very high (Forbes Jr., 2011). Lastly, pretreatment or bioaugmentation strategies can effectively improve the biodegradability of lignocellulosic wastes (e.g. yard waste and agriculture plant residues), providing significant enhancement in energy recovery (see Section 2.4. and Chapter 4).

Market-related factors include markets for compost and carbon credits and/or RECs and energy markets (energy costs and demand for renewables). Increases in energy costs would have proportionally positive effects on the economics of HS-AD and have proportionally negative effects on the economics of energy consuming management technologies (composting and landfill without LFGTE), resulting in improved competitiveness of HS-AD. Energy costs and demand for renewables are influenced by policy, as are compost and carbon credit/REC markets. Policy has the potential to influence, or even dictate, numerous other key factors such as demand for alternative OFMSW management infrastructure (i.e. landfill bans) and quality of feedstock for HS-AD (i.e. source-separation). Policy recommendations (and justifications) for catalyzing improved OFMSW management are as follows:

- Ban the landfilling of OFMSW (yard waste and food waste) in all landfills including those with LFGTE. Diverting OFMSW results in reduced fugitive methane emissions and reduced landfill leachate generation, as previously described. As described by Yasar and Celik (2016), landfilling waste is fundamentally in conflict with the basic principles of sustainability. According to the EPA, "the promotion of LFG energy is not in conflict with the promotion of organic waste diversion" (EPA, 2015d).
- Mandate source-separation of the OFMSW by all generation sources (residential, commercial, industrial, and institutional). According to a study on full-scale HS-AD of OFMSW in Europe (Bolzonella et al., 2006b), energy recovery efficiency increases by a factor of three in systems processing source-separated OFMSW over systems processing

mechanically-separated OFMSW. According to Nate Morris, cofounder, CEO, and director of Rubicon Global, every city in the US should have mandated source-separation of OFMSW within the next decade.

- Implement pay-as-you-throw (PAYT) policies, recycling programs, and other similar progressive MSW management programs to increase incentive for waste reduction and recycling and to facilitate the transformation of the existing disposal-based MSW framework to a recovery-based system. California's Extended Producer Responsibility (EPR) policy, for example, makes "producers" responsible for end-of-life product disposal costs (CalRecycle, 2014b).
- Adopt policies that create incentive for recycling both nutrients and energy from OFMSW (AD) as opposed to recycling only energy (LFGTE, incineration WtE, ATT) or only nutrients (composting). In other words, adopt policies that account for the environmental impacts and offsets of various recycling methods, such that there is incentive to, for example, recycle paper, plastic, and glass via conventional recovery methods rather than incinerating the material. According to Mitch Kessler, President of Kessler Consulting , Inc. and immediate past President of SWANA Florida Chapter, "counting Waste-To-Energy (WTE) as recycling is not, in my opinion, accurate or appropriate, and is not helping the public or private sector to advance waste reduction or recycling. The industry needs innovative and progressive waste reduction and recycling policies and programs to truly increase recycling rates."
- Establish a Florida Renewable Portfolio Standard to enhance incentives for renewable energy generation and growth of REC markets. The majority of states (29) now have Renewable Portfolio Standards and another eight states having voluntary renewable energy targets (NREL, 2014).

3.4. Conclusions and Future Directions

HS-AD is the most environmentally friendly OFMSW management technology, as it enables the efficient recovery of energy and nutrients, minimizes emissions (when compared with composting, incineration WtE, AAT, and landfill with LFGTE), and reduces water requirements and leachate generation (when compared with L-AD). However, implementation of HS-AD in the US has only recently begun due to the lack of legislative incentives and the low cost of landfilling. With environmentally-driven legislation rapidly increasing in prevalence across the country, HS-AD is now being considered by many private companies and city, county, and state governments and the number of HS-AD facilities in the US is growing exponentially. From one small facility in 2011, the number of full-scale HS-AD facilities has grown to eight in 2015 and is projected to reach 27 by 2017.

The objective of this study was to evaluate the potential for HS-AD implementation in Florida. It was determined based on current OFMSW (yard waste and food waste) generation and recycling rates in the state, that there is high demand for the implementation of OFMSW recycling infrastructure in the state. It was also determined that each of the eight most populated countries in the state – Miami-Dade, Broward, Palm Beach, Hillsborough, Orange, Pinellas, Duval, and Lee – consistently rank in the top nine counties in the state with respect to OFMSW disposal/availability for use as feedstock in HS-AD. A ninth county that was identified as particularly promising for HS-AD implementation was Alachua County, the home county of the University of Florida and the 23rd most populated county in the state. It was estimated, based on the current recycling rates of food waste and yard waste of 7% and 51%, respectively, and the relative fractions (of total MSW generated) of food waste and yard waste of 7% and 12%, respectively, that Florida's statewide recycling rate could be increased by nearly 13% (from 50% to 63%) if 100% OFMSW recycling rates could be achieved.

Environmental and economic incentives for implementing HS-AD for OFMSW management in the state were also estimated, assuming 100% of OFMSW were to be processed via HS-AD (although this could be adjusted for any scale/processing capacity). Based on 2014 food waste and yard waste generation, approximately 500 MW (4,000 GWh-year^{-1}) of energy could be recovered from OFMSW via HS-AD annually, equating to approximately 175 MW (1,500 GWh/year) of electricity (around 0.6% of Florida's electricity demand) and 325 MW of usable heat energy if the methane were to be used in CHP units, or equating to nearly 80 million DGEs of CNG (around 11.5% of Florida's CNG vehicle fuel demand) if the methane were to be converted to CNG. Additionally, more than 7,000 tons of nitrogen and 3,500 tons of phosphorous could be recovered annually and at least 660,000 metric tons of GHG emissions (as carbon dioxide equivalents) could be offset.

Unfortunately, the current political and economic climate in Florida is not conducive to economically feasible HS-AD in the state and thus, for the potential benefits associated with HS-AD implementation to be realized, development and enactment of certain policies is necessary. As seen in Europe and California, banning organics disposal, both yard waste and food waste, in all landfills, mandating source-separation of OFMSW by all generation sources, and creating incentives for renewable energy generation are the policy actions that have the greatest influence on the rate of development of HS-AD capacity for OFMSW recycling. Other recommendations include the development of PAYT programs, recycling programs, Extended Producer Responsibility (EPR) type policy, and other progressive policies for incentivizing waste reduction and recycling and creating economic value for the environmental costs of various recycling options. A final recommendation was to establish a Florida Renewable Portfolio Standard to enhance incentives for renewable energy generation and growth of REC markets. Overall, the future of HS-AD in Florida, and all around the country, is promising. Numerous grant and loan programs exist for project funding, public-private partnerships are becoming the norm in the recycling industry, and waste management frameworks are steadily transforming to recovery-based as opposed to the traditional disposal-based systems.

With regard to future work on this topic, in a recent study published by the Environmental Research and Education Foundation (EREF, 2015b), "wastesheds" were defined based on regions with shared MSW management infrastructure to evaluate the potential to implement bioenergy technologies in the state of North Carolina. Future assessment of the outlook for HS-AD implementation or efforts for identifying suitable locations for implementation should employ the method of defining wastesheds. Additional research that should be carried out includes comprehensive LCA studies aiming to identify optimal integrated recycling approaches for specific waste streams in specific contexts, studies aiming to develop/identify effective strategies for facilitating source separation and optimizing organics collection, and research on optimizing HS-AD system and process design and codigestion strategies.

CHAPTER 4: ENHANCED METHANE YIELDS FROM YARD WASTE IN HS-AD

4.1. Introduction

Anaerobic digestion (AD) is considered to be the most environmentally sustainable strategy for managing the organic fraction of municipal solid waste (OFMSW), as it allows for the recovery of energy (in the form of biomethane) and nutrients from the waste (Edelmann et al., 2005, Zaman, 2009; Morris et al., 2011). High-solids AD (HS-AD) has been shown to yield additional benefits when compared with liquid AD (L-AD), including reduced parasitic energy losses, reactor volume requirements, water usage, and excess leachate generation (Li et al., 2011). These findings have resulted in the rapid development of HS-AD technologies in Europe and recent advancement in the United States (De Baere and Mattheeuws, 2014; EREF, 2015). One of the pressing challenges associated with HS-AD is the low degradability of lignocellulosic wastes, such as yard wastes (Li et al., 2011). The lignin in these wastes is highly recalcitrant and the association of cellulose and hemicellulose with the lignin acts as a barrier to the microbial populations that perform hydrolytic conversion of cellulose (Tong et al., 1990; Zheng et al., 2014). Thus, HS-AD of lignocellulosic waste requires pre-treatment or long retention times to achieve sufficient degradation, which reduces the environmental and economic sustainability of the process (Li et al., 2011).

A number of studies have demonstrated that physical, chemical, and/or biological pre-treatment can increase methane yields from lignocellulosic wastes (Vervaeren et al., 2010; Bruni et al., 2010; Kreuger et al., 2011; Purwandari et al., 2013; Zhao et al., 2014). However, recent reviews of pretreatment strategies have concluded the increased methane production rates do not justify the environmental and economic costs incurred from these processes in most cases (Mosier et al., 2005; Hendricks and Zeeman, 2009; Zheng et al., 2014; Yang et al., 2015). A potential low cost, low impact (with respect to additional energy or chemical inputs or waste generation) strategy for enhancing methane yields from lignocellulosic wastes is inoculation of substrates with microbial populations that have a greater capacity to hydrolyze lignocellulosic compounds (i.e. bioaugmentation). For example, ruminant bacteria (e.g. from cattle) have been shown to employ unique mechanisms for degrading lignin, cellulose, and hemicellulose (lignocellulosics), including rapid production of extracellular substances (e.g. cellulosomes and fibrates), adhesion of cellulolytic species to fibers (enhances cellulolytic conversion), and tunneling into fibers (increases flow of nutrients and enzymes to and from lignocellulosics) (Yue et al., 2013). Bioaugmentation of HS-AD systems with ruminant bacteria (e.g. from cattle) has been shown to increase hydrolysis rates, volatile solids (VS) destruction, and biogas generation rates, but has yielded biogas with low methane content due to limited methanogenic populations (Lopes et al., 2004; Hu and Yu, 2005).

An alternative source of microbes that has recently been identified as potentially promising is granular anaerobic sludge generated in the treatment of waste from pulp and paper mills (P&P sludge) (Mussoline et al., 2013). P&P sludge is a waste material that contains microbial populations that are acclimated to a lignin-rich waste stream and likely contains hydrolytic communities capable of degrading lignocellulosics (Mussoline et al., 2013; Meyer and Edwards, 2014). *Clostridium cellulovorans*, for example, originate in wood chips and produce enzymes, such as cellulosome complexes, which aid in delignification (Tamaru et al., 2010). Reported concentrations of cellulose in P&P sludge versus primary (untreated) pulp and paper mill sludge (19-27% of TS versus 36-45% of TS, respectively), reaffirms that this is in fact the case (Migneault et al., 2011; Zorpas et al., 2011). In a study investigated the possibility of enhancing methane production from rice straw in HS-AD by adding P&P sludge, the theoretical maximum specific methane yield from rice straw was reached in 92 days of digestion using a substrate to inoculum (S/I) ratio of 1/2 on a wet weight basis (Mussoline et al., 2013). The specific methane yield of 340 L CH_4/kg VS achieved in this study was 47-74% higher than in similar studies using conventional inocula (e.g. domestic wastewater anaerobic sludge) and was comparable to the methane yield achieved in studies employing various pretreatment methods, leading to the conclusion that this strategy could be a viable alternative to pretreatment for improving AD of agricultural residues.

The overall goal of this research was to investigate the potential to improve the sustainability of HS-AD of OFMSW by using this novel bioaugmentation strategy as an alternative to pretreatment. The specific objective was to study the effects of this strategy in HS-AD of yard waste. Methane yields from yard waste inoculated with P&P sludge were compared directly to methane yields from yard waste inoculated with wastewater anaerobic sludge and were also compared to methane yields reported in other yard waste HS-AD studies. The enhancement in methane yield was compared to enhancements achieved in yard waste pretreatment HS-AD studies and a second round of experiments was conducted to investigate potential to sustain enhancements in methane yields through digestate recirculation, a common operational practice in full-scale HS-AD systems (Li et al., 2011).

4.2. Materials and Methods

Mixed yard waste (containing branches, leaves, and needles, tree trimmings, shrub trimmings, and other mixed yard debris) was obtained from the University of South Florida campus (Figure 4.2). The waste was shredded using a commercial yard waste shredder approximately one week prior to sample collection. Upon collection, the sample was sieved to a maximum particle size of 3 mm to improve homogeneity. In full-scale HS-AD, grinding of waste to 40mm particle size or less is common (De Baere, 2012). P&P sludge from a mill in Matane, Canada was provided by Tembec, a Canadian based manufacturer of forest products. The mesophilic (35ºC) anaerobic reactor at the Matane mill treats raw wastewater from the mill with a total suspended solids content near 200 ppm and a hydraulic retention time of 3.8 hours and generates a granular sludge. Wastewater anaerobic sludge (a conventional HS-AD inoculum source) was obtained from Howard F. Curren Advanced Wastewater Treatment Facility (HFCAWTF) in Tampa, Florida. HFCAWTF digests a mixture of primary and waste activated sludge under mesophilic conditions with an SRT of 21 days and generates a flocculent sludge. The inocula and substrate were stored at room temperature during experiment setup.

4.2.1. Experimental Setup

Two phases of batch HS-AD experiments were carried out in series. Anaerobic digesters were set up in triplicate in 250-mL glass bottles (Figure 4.2), sealed with metal crimp caps and silicone septums, and placed in a thermostatically-controlled room maintained at 35 ± 2 ºC. Figure 4.1 shows the compositions of the digesters assembled for the experiments. Phase 1 compared the performance of digesters containing yard waste inoculated with P&P sludge (Phase 1 bioaugmented digesters) to the performance of digesters containing yard waste inoculated with wastewater anaerobic sludge (Phase 1 control digesters). Phase 2 compared the performance of digesters containing yard waste inoculated with digestate from Phase 1 bioaugmented digesters (Phase 2 bioaugmented digesters) to the performance of digesters containing yard waste inoculated with digestate from Phase 1 control digesters (Phase 2 control digesters). Four additional digesters were prepared for both the bioaugmented digesters and control digesters during the setup of Phase 1 of batch HS-AD for intermediate chemical analysis at the end of weeks 1, 3, 6, and 9. Blank digesters (containing only inocula) were prepared to correct for methane yields from inocula in the bioaugmented and control digesters. The TS content in the digesters was set at 20%, a common TS content for HS-AD (Li et al., 2011). The S/I ratio was set at 1/1 on a wet weight basis, a common relative concentration of inoculum (50% by wet weight) for efficient start-up in batch HS-AD (Martin et al., 2003; Rapport et al., 2008; Li et al., 2011; Brown and Li, 2013; Chen et al., 2014).

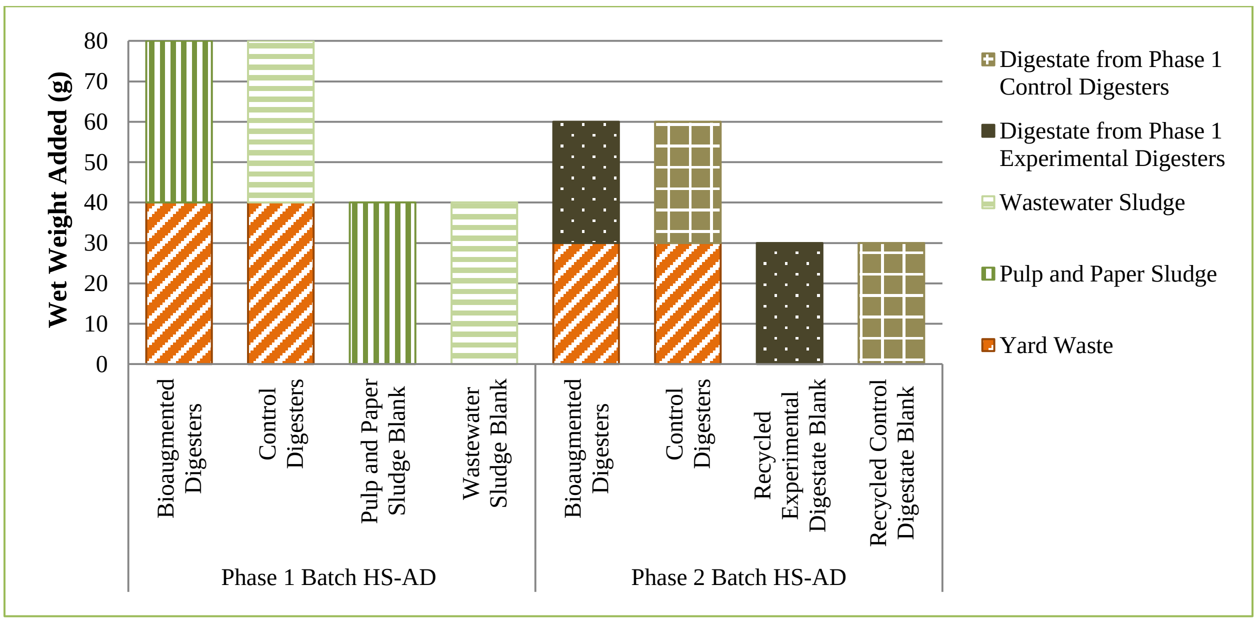

Figure 4.1: Phase 1 and Phase 2 batch HS-AD digester compositions by wet weight.

4.2.2. Analytical Methods

Biogas generation and quality was measured from each digester daily during early stages of digestion and less frequently (2-4 times weekly) as biogas generation rates decreased. Biogas was measured using a 50 mL frictionless syringe with a metal luer lock tip (Cadence Science Inc, 5157) equipped with a 25-gauge needle (BD PrecisionGlide 305125) according to previously described procedures (Figure 4.2) (Jerger et al., 1982; Owens and Chynoweth, 1993). Biogas quality (approximate methane content) was determined by dissolving the carbon dioxide portion of a 20 mL biogas sample into a 3 N NaOH barrier solution and measuring the resulting liquid displacement (Figure 4.2), as described by Wang et al. (in review). Before each biogas measurement, the digesters were shaken vigorously for approximately five seconds to dislodge any gas bubbles from the substrate.

Figure 4.2: Yard waste used as substrate, experiment setup, and biogas monitoring.

TS and VS were measured according to *Standard Methods* (2540) (APHA, 2012). For chemical analyses, samples were diluted with deionized water at a 1/2 ratio (mass of sample to volume of deionized water), mixed vigorously for three minutes, then centrifuged to obtain a representative liquid fraction, as outlined by EPA Method 9045D (EPA, 2004). This supernatant was used to measure pH and concentrations of alkalinity (as $CaCO_3$), total volatile fatty acids (VFA – as acetic acid), chemical oxygen demand (COD), total ammonia nitrogen (TAN), total nitrogen (TN), and total phosphorous (TP) according to *Standard Methods* (APHA, 2012). VFA concentrations were measured by the esterification method (10240) using Hach TNT plusTM 872 test kits. COD concentrations were measured using Orbeco-Hellige high-range COD kits (5200B). TAN concentrations were measured using Hach high-range TAN kits (10031). TN concentrations were measured using Hach high-range TN kits (10072). TP concentrations were measured using Hach low-range TP kits (8190). Remaining ash from the volatilization of inoculum samples and digestate samples was diluted and preserved with 1% nitric acid for digestion (72 hours at 50ºC) and elemental analysis (Na, K, Ca, Mg, Fe, Cu, Cr, Ni, Zn, Pb, Co, Mo, Se, and Mn) using a Thermo-Scientific induced coupled plasma mass spectrometer (ICP-MS). Undigested yard waste and digestate from the bioaugmented digesters and control digesters from the first phase of batch HS-AD were analyzed for lignin, cellulose, and hemicellulose content at the North Carolina State University Environmental Engineering Laboratory via the high-performance liquid chromatography (HPLC) method described by Davis (1998).

4.2.3. Data Analysis

Specific methane yields from yard waste were calculated by first subtracting the total methane yield produced in the blank digesters from the total methane yield produced in the bioaugmented and control digesters to obtain the volume of methane originating specifically from the yard waste. The resulting volume was then adjusted to an equivalent volume at STP. Finally, the adjusted total methane volume (L) was divided by the mass of VS (kg) of yard waste loaded to each digester. Percent enhancement in methane yield was calculated as the percent difference in specific methane yields from bioaugmented digesters and control digesters. Detailed equations are shown in Appendix E.1. Statistical significance was determined by analysis of variance (ANOVA, $\alpha = 0.05$) using Microsoft Excel with $p_{critical} = 0.05$.

4.3. Results and Discussion

4.3.1. Biogas Production and Quality

The use of P&P sludge as an inoculum compared with the use of wastewater sludge generated a significant enhancement in methane yield in Phase 1 (73%, $p = 1.03E-3$) and Phase 2 (68%, $p = 5.15E-7$) (Figure 4.5). The specific methane yields achieved in the Phase 1 bioaugmented and control digesters were 100 ± 2 L CH_4/kg VS and 58 ± 1 L CH_4/kg, respectively (Figure 4.3). The specific methane yields achieved in the Phase 2 bioaugmented and control digesters were 34 ± 0 L CH_4/kg VS and 21 ± 0 L CH_4/kg VS, respectively (Figure 4.4). The average quality of biogas produced in Phase 1 in both the experimental and control digesters was 57 ± 2% and the average quality of biogas in Phase 2 in both the experimental and control digesters was 59 ± 1%. In both phases, biogas quality remained relatively consistent in all digesters (Figure 4.6).

Although the percent enhancement achieved through inoculation with fresh pulp and paper sludge (73%) and the percent enhancement achieved through inoculation with digestate (68%) were comparable, the specific methane yields achieved through inoculation with fresh sludge were more than twice the value of the yields achieved through inoculation with digestate. In general, it is difficult to compare specific methane yields of mixed yard waste from one study to another, because the exact contents (plant species, branches versus leaves, etc.) are rarely reported, specific sample characteristics (e.g. elemental composition, presence of lignocellulosics) vary significantly from study to study, and yard waste composition varies widely with geographic location and season, especially in temperate climates. Jerger et al. (1982) determined that the BMPs for six different species of woody biomass with 0.8 mm maximum particle size, an S/I ratio of 1/1 by VS (inoculation with wastewater sludge), and addition of N and P. Specific methane yields achieved over 120 days of digestion ranged from 14 L CH_4/kg VS (Eucalyptus) to 320 L CH_4/kg VS (Hybrid Poplar and Sycamore) and VS destruction ranged from less than 1% (Eucalyptus) to nearly 57% (Sycamore). In a study conducted by Owens and Chynoweth (1993), the ultimate BMP of leaves (1.53 mm maximum particle size) from Laurel Oak, a native and common tree species in Florida, was determined to be 123 ± 5 L CH_4/kg VS after approximately 80 days of digestion. Zhao et al. (2014) reported specific methane yields from mixed yard trimmings collected in Wooster, Ohio (12.7 mm maximum particle size) to be as low as 17.6 L CH_4/kg VS after 40 days of HS-AD. In contrast, Brown and Li (2013) observed specific methane yields in the range of 30-50 L CH_4/kg VS from yard trimmings (5 mm maximum particle size) collected in Wooster, Ohio as they varied S/I ratio from 3/1 to 1/1.

The specific methane yields achieved in both Phases 1 and 2 of this study fall within expected ranges based on the studies described above. The significant decrease in specific methane yields observed between the first and second phase of batch HS-AD was likely in part a result of differences in the recalcitrance of yard waste samples that were digested in the first and second phases. Other factors that may have contributed include reduced mixing frequency in the Phase 2 relative to Phase 1 (biogas production rates were lower in Phase 2 and therefore biogas was measured less often and shaking occurred less often) and differences in concentrations of micronutrients and salts (see Section 4.3.2.).

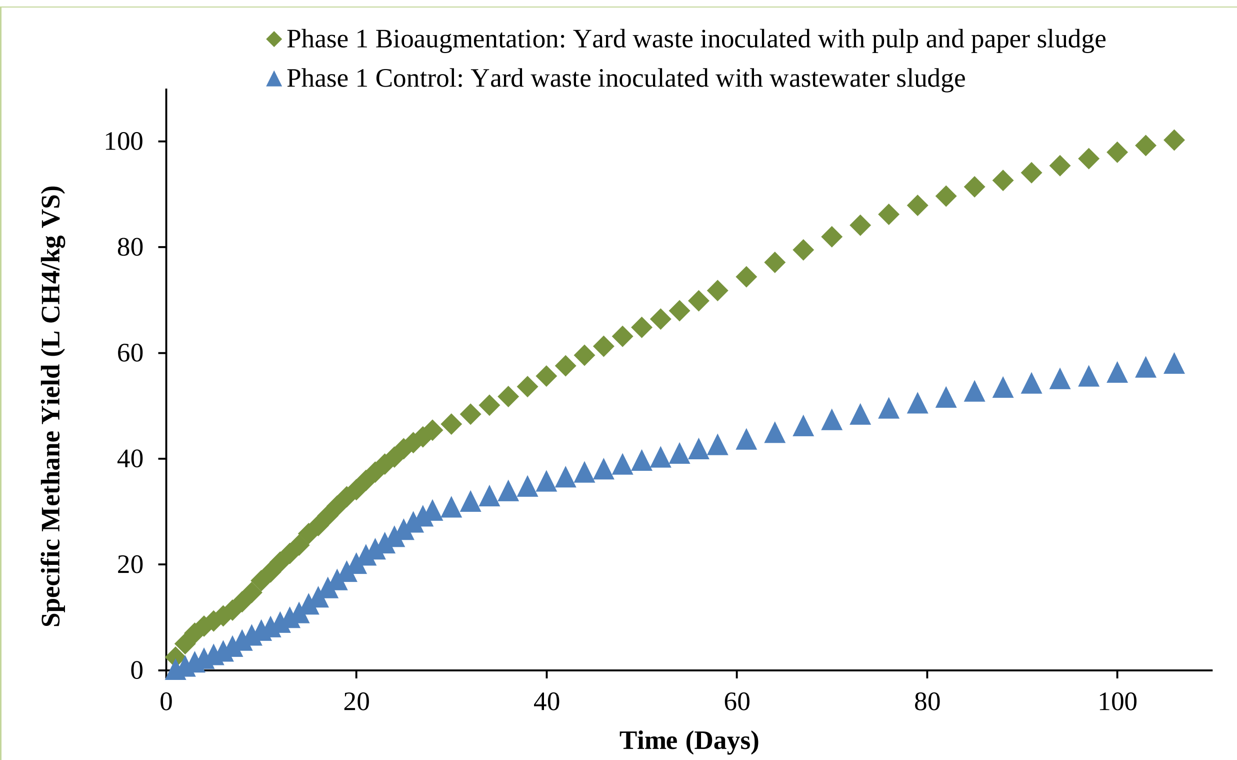

Figure 4.3: Specific methane yields observed in Phase 1 of batch HS-AD over 106 days.

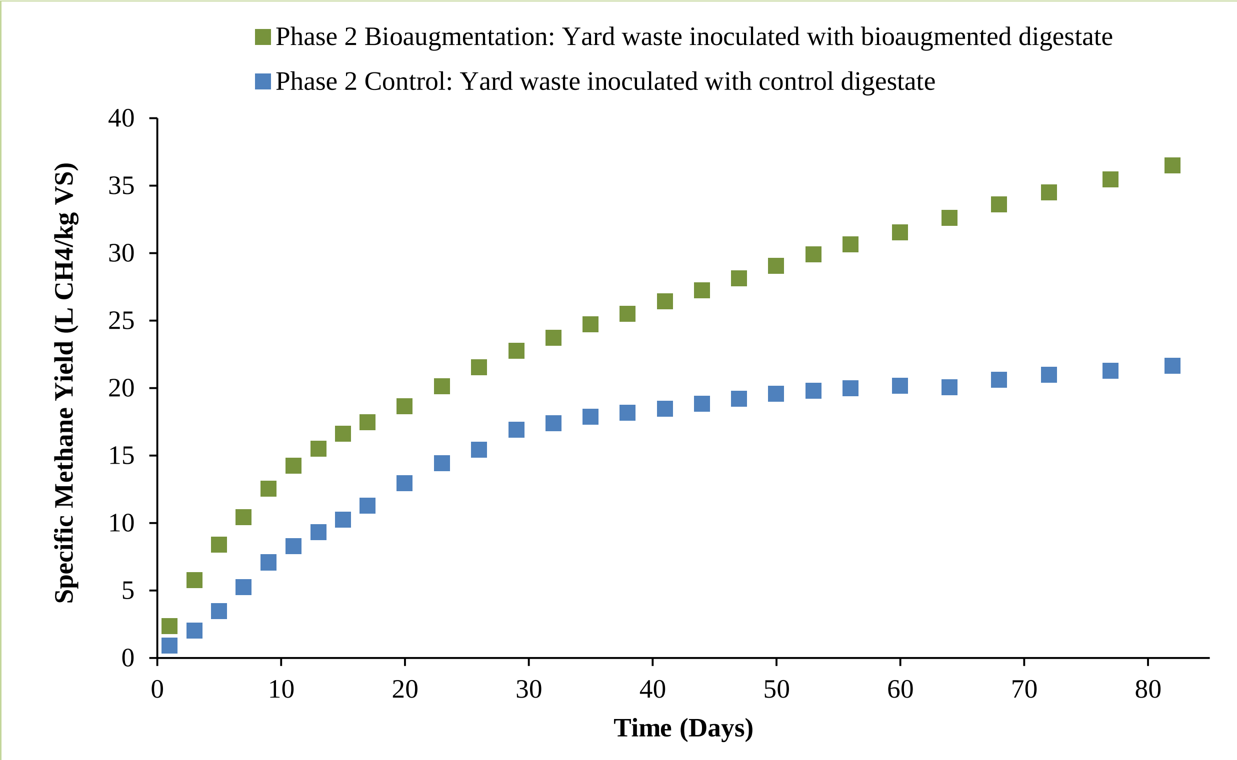

Figure 4.4: Specific methane yields observed in Phase 2 of batch HS-AD over 82 days.

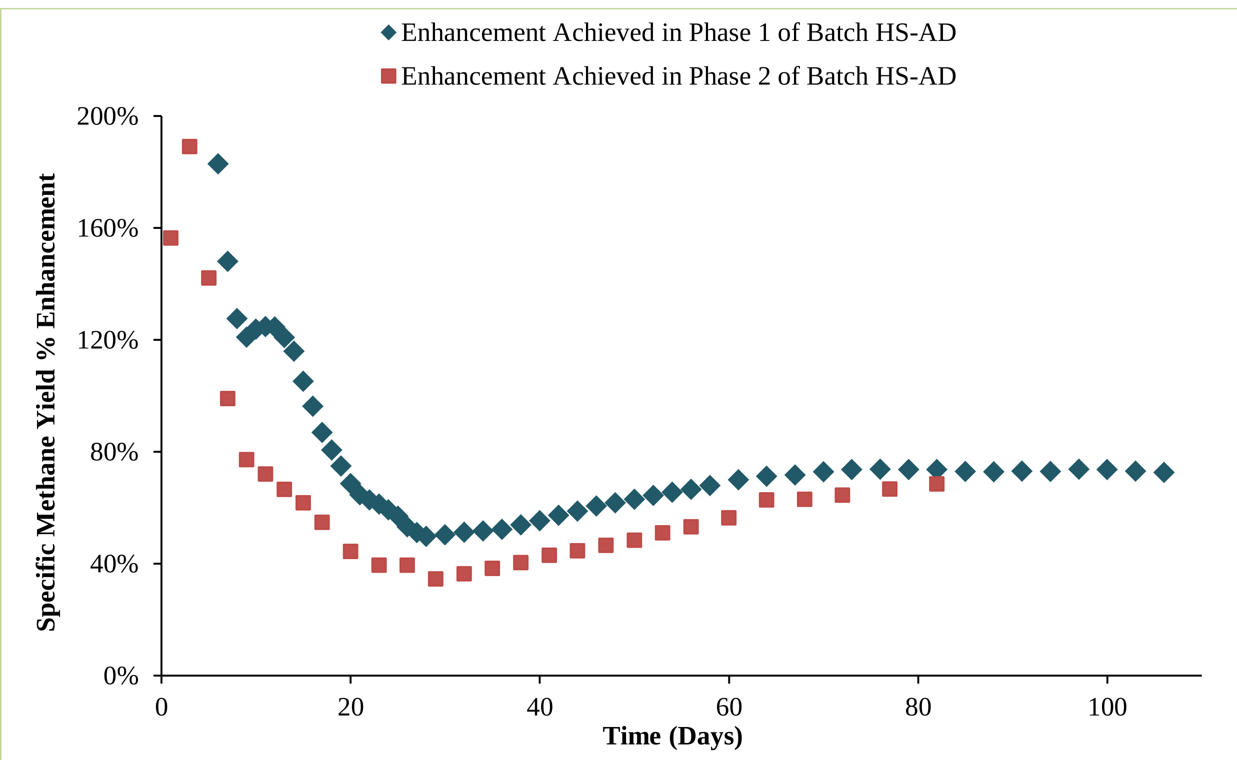

Figure 4.5: Percent enhancement in methane yield achieved in Phases 1 and 2 of batch HS-AD.

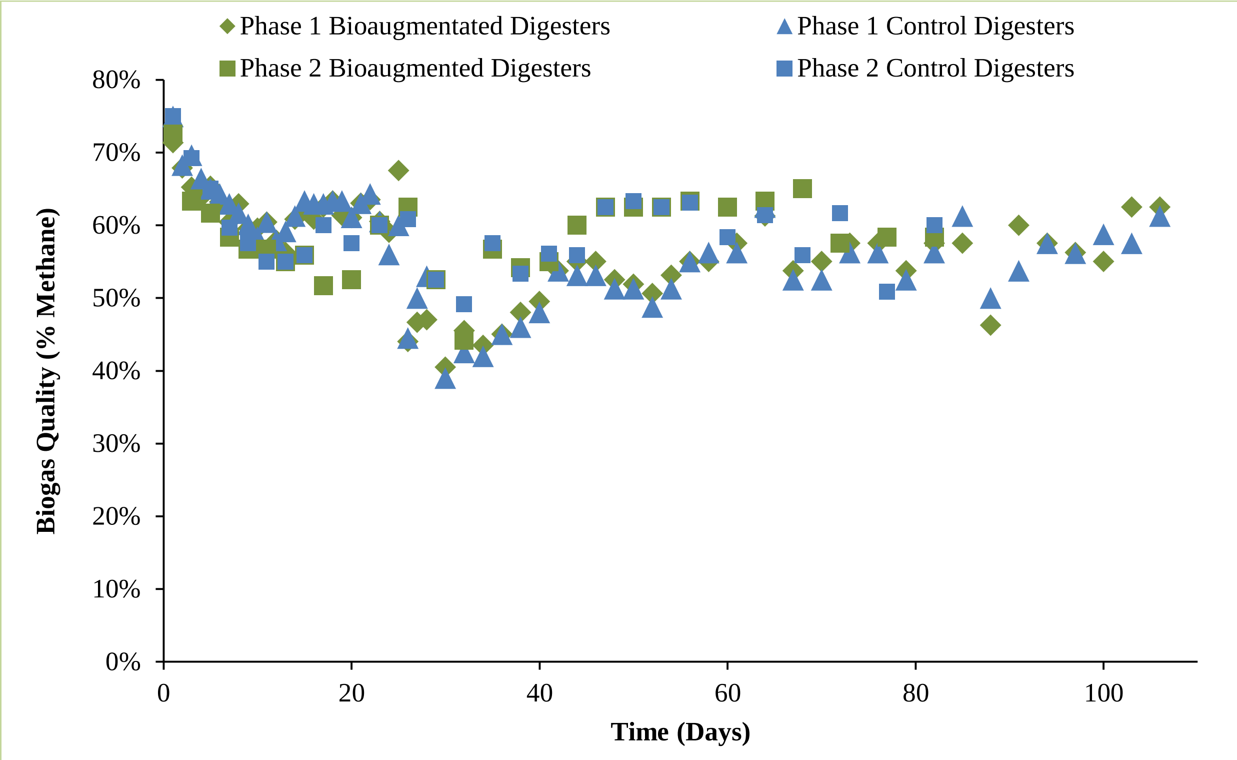

Figure 4.6: Biogas quality observed over the course of Phases 1 and 2 of batch HS-AD.

4.3.2. Substrate and Inocula Characterization

The TS content (by wet weight), VS content (by wet weight), and alkalinity of the yard waste samples and inocula used in Phases 1 and 2 of batch HS-AD are shown in Table 4.1. The elemental characterization of the inocula is shown in Table 4.2. The elements that were selected for measurement were those that have been shown to play important roles in HS-AD. For example, minimum concentrations of certain micronutrients have been reported as essential for methanogenesis (e.g. molybdenum, selenium, and cobalt), others have been reported as essential for acetogenesis (e.g. copper and zinc), and inhibitory concentrations of certain salts (e.g. sodium, potassium, calcium, and magnesium) have been documented (Deublein and Steinhauser, 2008). Minimum required concentrations and inhibitory concentrations (where applicable) as reported by Deublein and Steinhauser (2008) and Zupančič and Grilc (2012) are shown alongside the results of the elemental analysis (Table 4.2).

Table 4.1: Substrate and inocula alkalinity, total solids content, and volatile solids content.

	Phase 1 Substrate and Inocula			Phase 2 Substrate and Inocula		
	Pulp and Paper Sludge	**Wastewater Sludge**	**Yard Waste for Phase 1**	**Phase 1 Bioaugmented Digestate**	**Phase 1 Control Digestate**	**Yard Waste for Phase 2**
Alkalinity (mg/L as $CaCo_3$)	2,100	580	50	400	140	25
TS (% of wet weight)	10.0 ± 0.2	0.6 ± 0.0	50.8 ± 3.4	18.5 ± 0.1	23.7 ± 0.3	64.2 ± 0.5
VS (% of wet weight)	8.4 ± 0.1	0.4 ± 0.0	46.4 ± 2.9	16.6 ± 0.1	21.7 ± 0.2	60.1 ± 0.4

The P&P sludge provided more alkalinity and was more concentrated than the wastewater sludge. Both of these factors may have contributed to differences observed in methane yields. The relatively low concentrations of alkalinity in the wastewater sludge meant that the control digesters had less of a buffer to pH change, which resulted in a temporary decrease in pH in Phase 1 control digesters and apparent transient inhibition of methanogenesis early in the study, as discussed in Section 4.3.3. The concentrated/granular nature of the microbes in the P&P sludge versus the dilute and flocculent nature of the wastewater sludge, as highlighted by the high VS value (8.4% of wet weight) in the P&P sludge versus the low TS (0.4% by wet weight) in the wastewater sludge, likely resulted in faster process start-up and an initial advantage in the bioaugmented digesters in the early stages of digestion. The S/I ratio is considered a major parameter affecting specific methane yields in batch AD (Angelidaki et al., 2009). Thus, inoculating with an S/I ratio on a wet weight basis and comparing inocula of a significantly different concentration (in terms of VS content) could be criticized as a flaw in this experiment. However, considering the average regeneration time of AD microorganisms (1-3 days for acidogenic and acetogenic microbes and 5-16 days for methanogenic microbes), it is reasonable to assume that the effect of this aspect of the experiment setup on cumulative methane yields over the 106 day and 82 day digestion periods was relatively minimal (methanogenic populations increased by a minimum exponential factor of 5) (Deublein and Steinhauser, 2008).

Table 4.2: Elemental characterization of inocula, minimum required concentrations, and inhibitory concentrations.

Element	**Minimum Required Concentration**	**Inhibitory Concentration**	**Pulp and Paper Sludge**	**Wastewater Sludge**	**Phase 1 Bioaugmented Digestate**	**Phase 1 Control Digestate**
Na (mg/L)	100-200	3500-5500	1180 ± 40	898 ± 21	171 ± 25	84.7 ± 17.4
K (mg/L)	200-400	3500-4500	382 ± 4	126 ± 3	180 ± 13	178 ± 3
Ca (mg/L)	100-200	2500-4500	828 ± 20	1230 ± 80	**2370 ± 140**	**2560 ± 70**
Mg (mg/L)	75-150	1000-1500	84.3 ± 0.7	87.8 ± 4.5	121 ± 10	146 ± 2
Cr (mg/L)	0.005-50	28-300	0.515 ± 0.005	0.271 ± 0.021	0.128 ± 0.027	0.024 ± 0.012
Fe (mg/L)	1.0-10	1750	49.7 ± 3.5	61.8 ± 4.5	17.6 ± 2.6	13.2 ± 1.6
Ni (mg/L)	0.005-0.5	10-300	0.758 ± 0.038	0.232 ± 0.016	1.03 ± 0.066	0.235 ± 0.006
Cu (mg/L)	> 0	150-300	8.93 ± 0.03	9.01 ± 0.50	2.56 ± 0.13	1.16 ± 0.12
Zn (mg/L)	> 0	3-400	**113 ± 0**	**7.46 ± 0.48**	**16.8 ± 1.0**	**6.28 ± 0.21**
Pb (mg/L)	0.02-200	8-340	0.327 ± 0.004	0.606 ± 0.021	0.065 ± 0.018	*0.024 ± 0.012*
Co (mg/L)	0.06	N/A	0.588 ± 0.030	*0.027 ± 0.002*	0.272 ± 0.017	*0.016 ± 0.000*
Mo (mg/L)	0.05	N/A	0.475 ± 0.024	0.242 ± 0.012	0.165 ± 0.013	*0.015 ± 0.006*
Se (mg/L)	0.008	N/A	0.029 ± 0.010	0.125 ± 0.009	0.016 ± 0.007	0.019 ± 0.009
Mn (mg/L)	0.005-50	1500	7.68 ± 1.87	0.973 ± 0.064	5.16 ± 0.18	5.31 ± 0.578

NOTE: All values expressed as average of samples run in triplicate plus or minus standard deviation; potentially inhibitory concentrations are shown in bold; potentially limiting concentrations are shown in italics

The P&P sludge and Phase 1 bioaugmented digestate satisfied all minimum required elemental concentrations and only reached potentially inhibitory concentrations of zinc and calcium (Phase 1 bioaugmented digestate only). It should be noted that the inhibitory concentrations of zinc of 3-400 mg/L are expressed as ionic concentrations and the inhibitory concentration of zinc as carbonate is reported as 160 mg/L, which is greater than the concentrations present in any of the four inocula. As for calcium, the high concentrations present in both digestate inocula may have contributed to the lower methane yields observed in Phase 2 of HS-AD. The concentration of cobalt in both the wastewater sludge and the Phase 1 control digestate was found to be potentially limiting and the concentration of molybdenum in the Phase 1 control digestate was also found to be potentially limiting. These factors may have contributed to the observed enhancement in methane yields in Phases 1 and 2 of HS-AD in the bioaugmented digesters relative to the control digesters.

The concentrations of heavy metals in the Phase 1 bioaugmented and control digestate reported in Table 4.2 can also be compared to maximum allowable heavy metal concentrations for compost. According to WRAP (2010), for safe application of compost/digestate in agriculture, chromium, copper, nickel, lead, and zinc concentrations must be less than 100, 200, 50, 200, and 400 mg/L. The measured concentrations of these elements in the digestate samples were far below these limits and European and US regulatory limits as documented by Brinton (2000). This does not come as a surprise, as metals concentrations rarely reach dangerous values in digestate from HS-AD of OFMSW (Drennan and DiStefano, 2010). Rather, it is the odor and phytotoxicity of these digestate that are common concerns and require aerobic curing to eliminate (Drennan and DiStefano, 2010).

4.3.3. Chemical Analysis

Trends observed in the evolution of chemical characteristics in Phase 1 bioaugmented and control digesters (Figure 4.7) correspond with observations in methane yields. As shown in Figure 4.6 (A), the VFA concentrations in the Phase 1 control digesters decreased over the first three weeks of digestion and again toward the end of the digestion cycle, suggesting that hydrolysis was predominantly the rate limiting step in these digesters, as has been shown previously in HS-AD of lignocellulosic materials by Veeken and Hamelers (1999). Conversely, a steady increase in VFA concentration was observed in the bioaugmented digesters, suggesting that methanogenesis rather than hydrolysis was the rate limiting step during the majority of the digestion cycle and providing further support for the hypothesis that the hydrolytic communities in the P&P sludge possess a superior ability to hydrolyze lignocellulosics. Similarly, the continuous increase in TAN concentrations (Figure 4.6 (B)) in the bioaugmented digesters and relatively constant TAN concentrations in the control digesters is consistent with observations in methane yields in that both indicate that more substrate was hydrolyzed/fermented in the bioaugmented digesters than in the control digesters (Kayhanian, 1994).

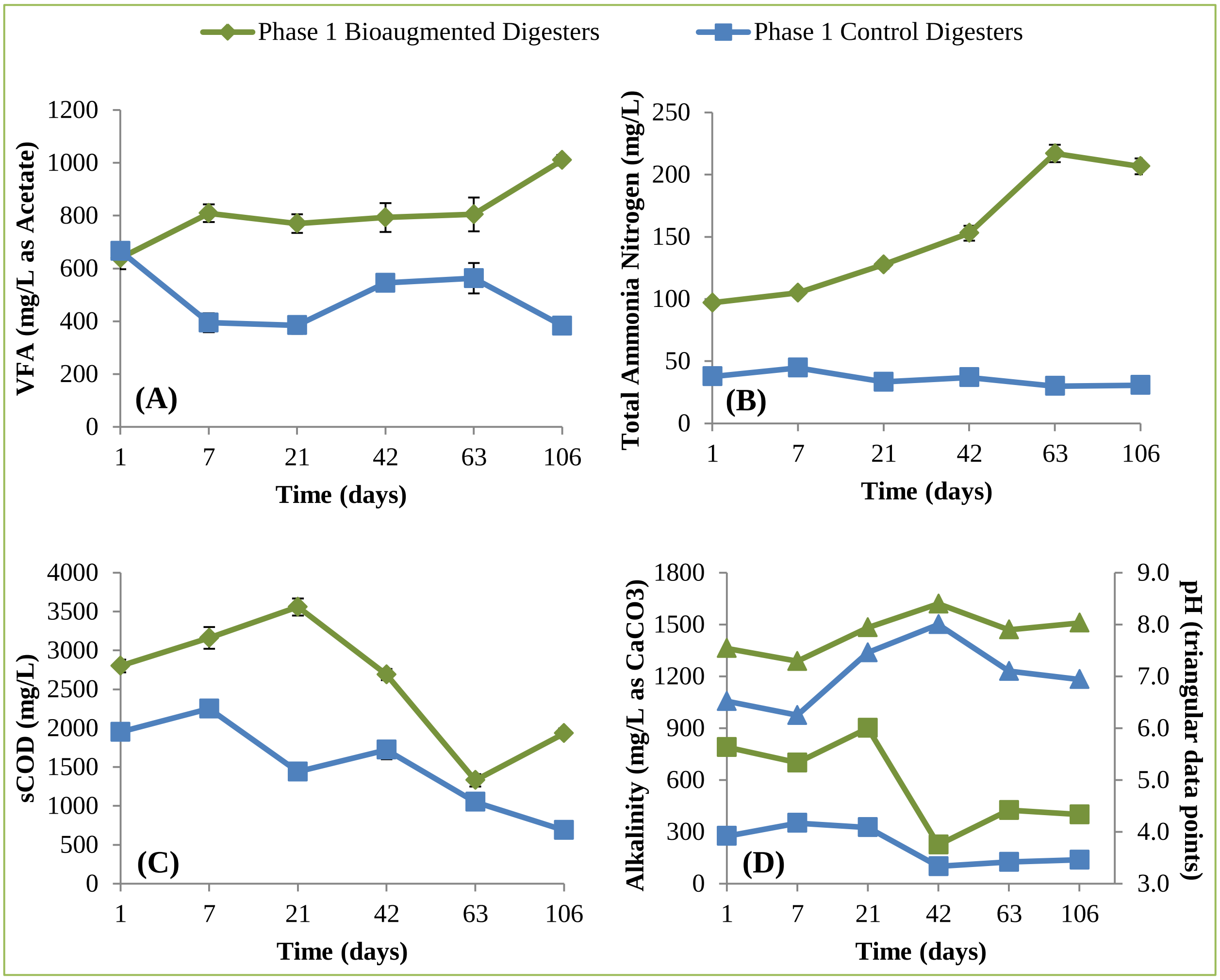

Figure 4.7: Evolution of VFA, TAN, sCOD, and Alkalinity concentrations and pH in Phase 1 of HS-AD over 106 days.

Observations in sCOD concentrations (Figure 4.6 (C)) were as expected, with an initial increase in both sets of digesters as hydrolysis increased the solubility of the COD present in the substrate, followed by decreases as methanogens converted chemical sCOD (VFAs) to methane. Again, the longer period of sCOD increase observed in the bioaugmented digesters relative to the control digesters suggests that hydrolysis occurred to a greater degree in the bioaugmented digesters than in the control digesters. However, observation in sCOD relative to VFA concentrations did not align (Figure 4.7 (A) and (C)). If the majority of sCOD exists as VFAs, then changes in sCOD concentrations would parallel changes in VFA concentrations, but this was not the case. In AD of lignocellulosic wastes, the majority of the complex organic matter being metabolized is carbohydrates (as opposed to proteins or lipids). It follows that the majority of the non-VFA soluble organics present in AD of yard waste would be simple sugars (products of the hydrolytic conversion of carbohydrates, Figure 2.1). It can then be concluded that an observed increase in sCOD concentrations paired with an observed decrease in VFA concentrations is indicative of increasing concentrations of simple sugars. Or in other words, it suggests that carbohydrates are being hydrolyzed to simple sugars more rapidly than simple sugars are being converted by acidogens/acetogens to VFAs, H_2, and CO_2 while the rate of VFA production by acidogens/acetogens is less than the rate of VFA consumption by methanogens. This was the case in the bioaugmented digesters between day 7 and day 21 but not the case in the control digesters during this time period, another indicator that hydrolysis in the bioaugmented digesters was accelerated. Similarly, an observed decrease in sCOD concentrations paired with an observed increase in VFA concentrations, e.g. in the control digesters between day 1 and 7 and in the bioaugmented digesters between day 21 and 63, is indicative of concentrations of simple sugars decreasing more rapidly than VFA concentrations are increasing (greater production of H_2 and CO_2).

An inverse correlation can be seen between alkalinity concentrations and VFA concentration in both control and bioaugmented digesters, as the formation of VFAs temporarily consumed alkalinity and the consumption of VFAs by methanogens produced alkalinity. The significant VFA concentrations remaining in both the bioaugmented and control digesters at the end of the 106 day study suggest that additional methane formation would have occurred had the study been extended. Alkalinity was relatively low in both digesters (Figure 4.6 (D)), but was never depleted below 100 mg/L. Accordingly, the pH stayed within a healthy range in both sets of digesters throughout the digestion period (Figure 4.6 (D)), with the exception of the day seven value of 6.3 observed in the control digesters. This drop in pH below 6.5, the pH value below which methanogenesis inhibition may occur (Fabián and Gourdon, 1999; del Real Olvera and Lopez-Lopez, 2012), is reflected in the methane yield curve for the Phase 1 control digesters shown in Figure 3 (a slight dip in the curve between days 7-14).

It should also be noted that no other forms of inhibition associated with these chemical parameters (e.g. ammonia or VFA) were a concern in this study. VFA concentrations greater than 10,000 mg/L as acetic acid are generally considered inhibitory to methanogenesis (Amani et al., 2010), which is far greater than the concentrations observed in this study. Typical inhibitory values of TAN concentrations are generally reported as being greater than 1,500 – 1,700 mg/L (Chen et al., 2008), again far greater than concentrations observed in this study. Generally, TAN concentrations less than 200 mg/L but greater than 0 mg/L are considered beneficial for AD process efficiency because ammonia nitrogen is an important nutrient for cell synthesis (Liu and Sung, 2002). However, in a study by Kayhanian et al. (1994) TAN concentrations between 600-

800 mg/L TAN were shown to yield optimal performance. In this study, TAN values remained in the “beneficial” range in both the bioaugmented and control digesters throughout the study, but were well below “optimal” ranges cited by Kayhanian et al. (1994).

4.3.4. Lignocellulosics Analysis

The lignin, cellulose, and hemicellulose contents (fraction of dry sample) in the digestate from the Phase 1 bioaugmentation digesters were 42.4 ± 0.4%, 10.8 ± 0.4%, and 8.0 ± 0.3%, respectively (Figure 4.8). The lignin, cellulose, and hemicellulose contents in the digestate from the Phase 1 control digesters were 43.0 ± 0.2%, 12.6 ± 0.4%, and 9.3 ± 0.4%, respectively (Figure 4.8). The differences in lignin, cellulose, and hemicellulose contents in the bioaugmented digestate compared with the control digestate were not statistically significant ($p = 0.206$, $p = 0.0518$, and $p = 0.0624$, respectively). However, the average cellulose and hemicellulose contents detected in the bioaugmented digestate were 16.0% and 16.1% less, respectively, than the average contents detected in the control digestate and the p-values for these parameters were very close to $p_{critical}$. It is expected that the microbial populations that would dominate in an anaerobic system treating lignocellulosic-rich pulp and paper mill waste would be species that can effectively hydrolyze lignocellulosic compounds – in the same way that the microbial populations that are abundant in the guts of ruminant animals are those that have adapted to efficiently hydrolyze lignocellulosics through unique modes of action (as described previously). Conversely, wastewater sludge contains a very low fraction of lignocellulosics and therefore, is highly unlikely to contain microbial populations fit for the task of hydrolyzing lignocellulosic compounds (Migneault et al., 2011; Zorpas et al., 2011). These observations are consistent with the observed increase in reduction in cellulose and hemicellulose contents in the Phase 1 bioaugmented digesters as compared with the control digestates.

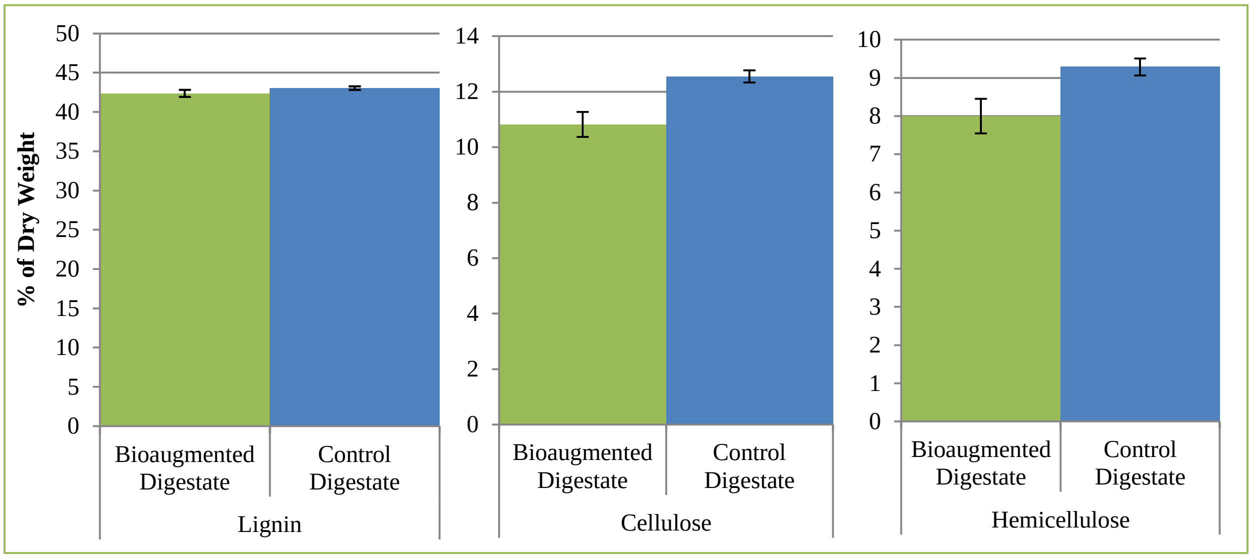

Figure 4.8: Lignin, cellulose, and hemicellulose content in digestate from Phase 1 bioaugmented and control digesters.

It should be noted that the lignin values observed in this study are relatively high and the cellulose and hemicellulose values are relatively low as compared with values reported in other studies. For example, Zhao et al. (2010) reported lignin, cellulose, and hemicellulose contents in fresh yard waste to be 32.9 ± 0.2%, 30.8 ± 0.5%, and 15.9 ± 0.3%, respectively, and reported cellulose and hemicellulose destructions to be approximately 16.7% and 5.7%, respectively. This translates to final cellulose and hemicellulose fraction of 25.7% and 15.0%, respectively. Jerger et al. (1982) showed that the lignin, cellulose, and hemicellulose contents of six different woody biomass species ranged from 24.7-34.5%, 35.2-44.7%, 24.9-39.8%, respectively, but did not report destruction. It is possible that some aerobic degradation of cellulose and hemicellulose took place while the shredded yard waste was sitting in a heap before sample collection, leading to relatively low initial values. Unfortunately, in this study, the lignin, cellulose, and hemicellulose content of the raw yard waste sample were not measured and therefore, this conjecture cannot be verified and destruction cannot be calculated.

4.3.5. Volatile Solids and Mass Destruction

Volatile solids destruction in bioaugmented and control digesters were not statistically significant in Phase 1 of HS-AD ($p = 0.370$) or in Phase 2 of HS-AD ($p = 0.389$). The initial and final Phase 1 VS values and the initial Phase 2 VS values are reported in Table 4.1. The final Phase 2 VS values in the bioaugmented and control digesters were 18.4 ± 0.4% and 17.4 ± 0.4%, respectively. The multiphase nature of the feedstock and digestate in this experiment, e.g. samples containing saturated solids and percolate (free liquid), resulted in relatively high standard deviations in TS and VS measurements, which compounded when calculating VS destruction leading to unreliable results (hence the high p-values). However, the initial weight of feedstock added and final weight of digestate removed were also measured and provided a more reliable means to compare destruction (Figure 4.9, see mass balance in Appendix E.1). Greater mass destruction was observed in bioaugmented digesters than in control digesters in both Phases of HS-AD, which corresponds to the differences observed in methane yields. The reduced difference between mass destruction in the bioaugmented and control digesters in Phase 2 relative to Phase 1 also corresponds to differences observed in methane yield enhancements.

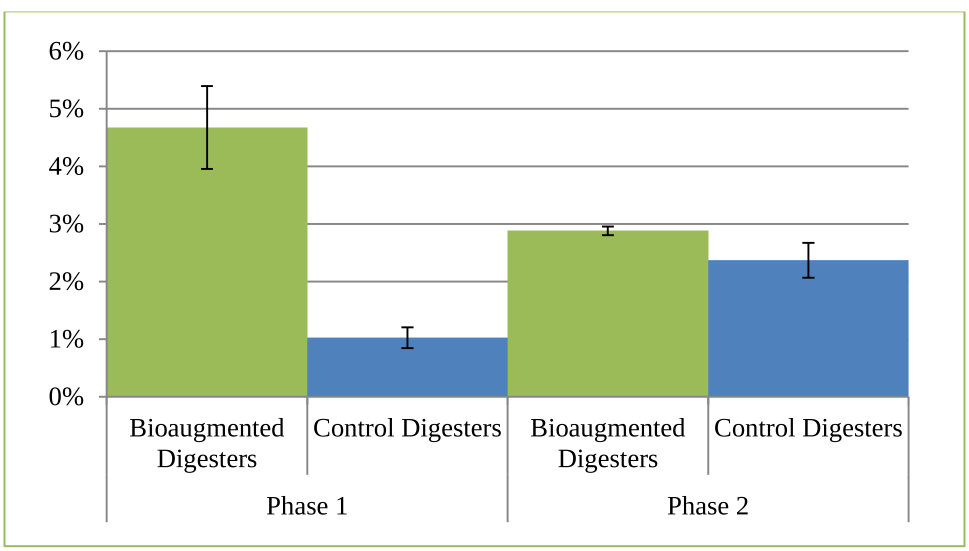

Figure 4.9: Percent total mass destruction in Phases 1 and 2 bioaugmented and control digesters.

4.3.6. Comparison to Rumen Bioaugmentation and Fungal Pretreatment

Bioaugmentation with P&P sludge for enhancing methane yield from lignocellulosic wastes compares most closely with bioaugmentation using rumen cultures – both inoculum sources can be obtained with relative ease, don't require cultivation, and likely possess particularly active hydrolytic microbial populations. This bioaugmentation strategy, however, yielded high quality biogas throughout both phases of digestion, whereas bioaugmentation with rumen cultures has repeatedly fallen short with this regard due to limited methanogenic conversion, as previously mentioned (Lopes et al., 2004; Hu and Yu, 2005).

With respect to pretreatment, bioaugmentation compares most closely with fungal pretreatment – a passive, low energy, low resource strategy and one of the most effective (Zheng et al., 2014). Low impact pretreatment methods, such as fungal pretreatment, could potentially yield net benefits where the benefits from other strategies such as thermal and chemical pretreatments are outweighed by the additional environmental and economic costs they incur (Zheng et al., 2014). A recent fungal pretreatment study of HS-AD of yard waste achieved 85 – 154% enhancement in methane yields (with a max yield of 44.6 L CH_4/ kg VS) via 30 days of aerobic fungal pretreatment under varying moisture contents (Zhao et al., 2014). The enhancement achieved in this study (approximately half of that achieved via fungal pretreatment) was substantial considering that the only measure taken to enhance the methane yield was the use of an alternative inoculum.

The limiting factor associated with the use of pulp and paper sludge for inoculation upon HS-AD process start-up is the proximity of the nearest pulp and paper mill with high-rate AD to a given HS-AD system. However, pulp and paper production is a growing industry and anaerobic treatment systems in pulp and paper mills are becoming increasingly common (Meyer and

Edwards, 2014). The number of onsite anaerobic treatment systems in pulp and paper mills around the world has increased from just over 100 in the year 2000 to more than 400 in 2014 and is expected to increase by 60% by 2020 (Frost and Sullivan, 2013; Meyer and Edwards, 2014).

4.3.7. Preliminary Codigestion and Pilot-Scale Experiments

The breadth of the bioaugmentation research was extended to address to additional research objectives: (i) study the effects of biosolids addition in codigestion of yard waste and food waste, (ii) study the effects of P&P bioaugmentation in codigestion of yard waste, food waste, and biosolids, and (iii) study the effects of scale on methane yield enhancements from yard waste through bioaugmentation with P&P sludge. A pilot-scale HS-AD system was designed and constructed and a preliminary pilot-scale experiment was conducted. The pilot system and the preliminary experiment are described in Appendix E.2 along with information regarding preliminary bench-scale experiments (those that preceded this bioaugmentation study). A preliminary bench-scale codigestion study that was conducted after the conclusion of this bioaugmentation study is described briefly here and in greater detail in Appendix E.2. The materials and methods used for the study were identical to those described in Section 4.2, except additional substrates were added and different masses were selected based on volume limitations (250 mL glass bottles). Substrate to substrate ratios were selected such that the digesters would not be overloaded and the S/I ratio was changed to 1/1.5 from 1/1 to further reduce the OLR. However, within five days of digestion experimental digesters that were inoculated with wastewater anaerobic sludge were showing signs of methanogenic inhibition. The digesters may have rebounded given enough time, but instead, crushed oyster shells were added, bringing in a forth research objective: (iv) to study the effects of oyster shell addition on the recovery of overloaded batch high-solids anaerobic digesters.

In a recent study exploring possible solid-phase buffers for decentralized biological denitrification, oyster shells were shown to be a promising alkalinity source because they are a waste material composed almost entirely of calcium carbonate (Sengupta et al., 2007). The study showed that the rate of dissolution of oyster shells is relatively slow, indicating that the waste material could provide long-term buffering capacity against system acidification. In this study, oyster shell addition led to rapid system recovery after acidification was observed. Additionally, bioaugmentation with P&P sludge led to improved system stability and a net enhancement in methane yield of 15% and biosolids addition resulted in accelerated system recovery after oyster shell addition and higher specific methane yields through the majority of the study (Figure 4.10).

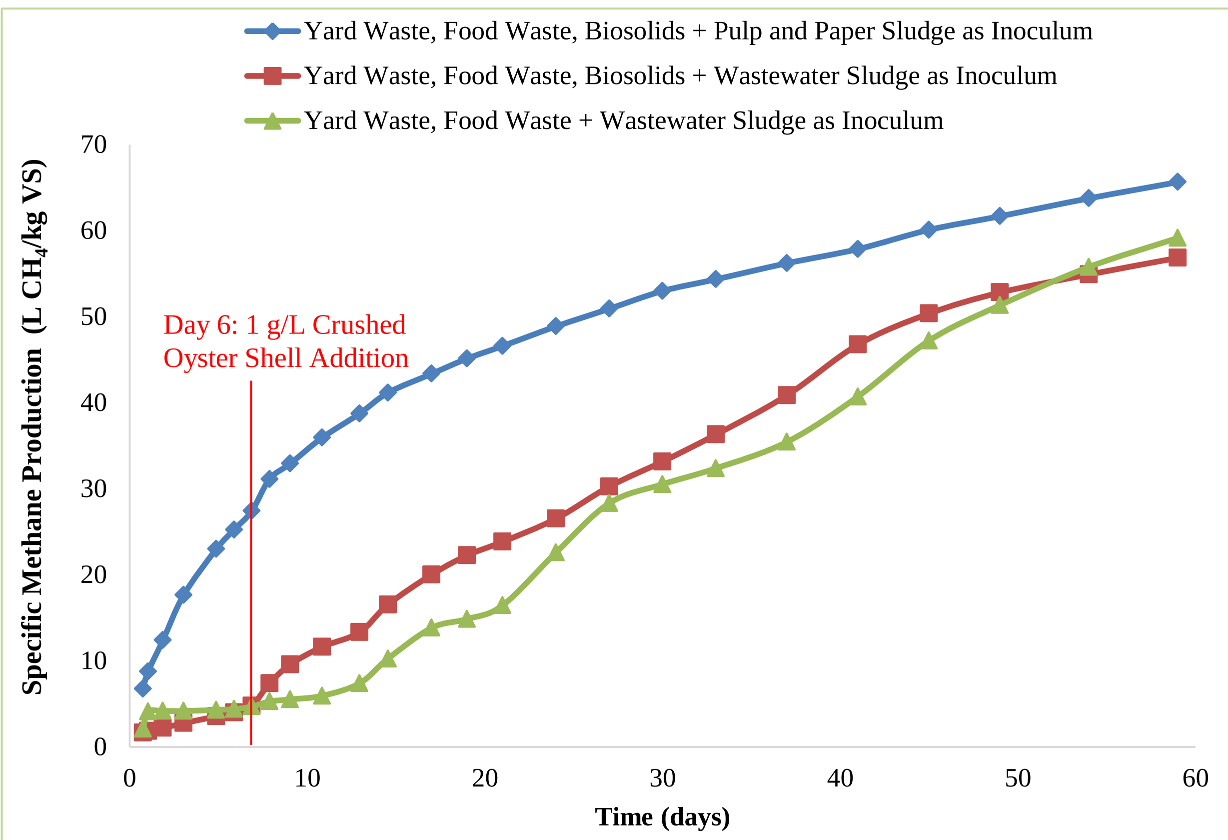

Figure 4.10: Specific methane yields observed in preliminary codigestion experiment.

4.4. Conclusions

A significant enhancement in methane yield from yard waste in HS-AD was achieved via bioaugmentation with P&P sludge as compared to methane yields achieved via inoculation with a conventional inoculum. Chemical data support the hypothesis that the observed enhancement was a result of the hydrolytic communities in the P&P sludge possessing a superior ability to hydrolyze lignocellulosics. The observed enhancement in methane yield was also sustained in a subsequent phase of batch HS-AD via inoculation with digestate from the first phase of digestion, suggesting that this method may have potential to yield prolonged benefits with respect to process efficiency and net energy recovery. The enhancements achieved in this study (68-73%) are comparable to enhancements reported in various pretreatment studies, but the minimal impact of this strategy with respect to overall operational costs and environmental impacts make it an attractive alternative to pretreatment. A preliminary study investigating the effects of undigested biosolids addition as a third substrate along with food and yard waste suggests that incorporating biosolids could improve system stability and methane yields while also providing an added source of revenue in the form of increased tipping fees (undigested biosolids disposal is generally more costly than food waste or yard waste disposal). In the preliminary study, bioaugmentation with P&P sludge in codigestion under HS-AD conditions was also shown to improve system stability and enhance methane yields. A final consideration that emerged from the preliminary study was the possibility of utilizing shellfish waste as an alkalinity source in HS-AD of OFMSW to enable higher OLRs without risk of system acidification. Additional research, as outlined in Chapter 5 of this thesis, is necessary to expand on the findings of these studies.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

Source-separation and recycling of OFMSW is likely to continue to become the norm in cities around the world and HS-AD is expected to become the primary centralized recycling technology for OFMSW because of the many environmental and economic advantages it offers. HS-AD efficiently recovers energy from OFMSW and is easily paired with composting to enable the recovery of nutrients. In the process, GHG emissions that would result from uncontrolled or partially controlled degradation of OFMSW are avoided. GHG emissions are also offset by the substitution of fossil-fuel derived energy with biomethane, which can be used for heating, electricity generation, and/or vehicle fuel. Furthermore, diversion of OFMSW from landfills to HS-AD facilities reduces eutrophication impacts on the environment or additional energy and chemical inputs needed for removing N and P from leachate streams at wastewater treatment facilities. The recovery and use of nutrients as fertilizer also reduces the impacts of inorganic fertilizer production on the nitrogen cycle (Haber-Bosch process) and depletion of mineral nutrient reservoirs. However, trends in the development of HS-AD in Europe and more recently in the US reveal that the optimization of HS-AD technologies, expansion of regulatory drivers, and development of public-private partnerships are necessary for accelerating the transition.

In this thesis, published and grey literature was reviewed, HS-AD facilities in California were toured, interviews were conducted with MSW management professionals, and bench-scale HS-AD experiments carried out in the laboratory. The overall goals were to contribute to (i) accelerating the implementation of HS-AD in Florida and elsewhere by reporting the state of the art of HS-AD in Europe and the US and by conducting a case study of the outlook of HS-AD implementation in Florida and (ii) to contribute to improving HS-AD process efficiency by conducting laboratory-scale experiments aiming to enhance methane yields from yard waste and exploring codigestion strategies. The following provides a summary of the major conclusions for each research question posed in this thesis.

1. *What is the state of the art of HS-AD?*

AD of OFMSW in Europe, especially HS-AD, has come of age more so than any other alternative treatment technology developed since the 1990's. As of 2014, there were 244 full-scale AD facilities treating OFMSW in Europe, with a total capacity of approximately 8 million tons per year (TPY); 89% of capacity was "stand-alone" (systems treating *only* OFMSW), 62% was HS-AD, and 70% installed since 2009 was HS-AD. Approximately 55% of capacity in Europe treats source-separated substrates as opposed to mixed or mechanically-separated substrates. Source separation of OFMSW is expected to continue to emerge as the industry standard (with the help of mandated source-separation), because it substantially improves energy and nutrient recovery efficiency in HS-AD systems.

Trends in HS-AD of OFMSW in the US have paralleled those in the EU. There are currently 181 AD facilities treating OFMSW, with a total capacity of approximately 780,000 TPY, 52% of capacity is stand-alone (25 facilities), and 24% is HS-AD (8 facilities), with the remainder being stand-alone L-AD or L-AD codigestion at wastewater treatment plants or on-farm systems. However, the number of HS-AD facilities is growing exponentially, from one in 2011 to eight in 2015. It is projected that HS-AD will be the dominant form of AD of OFMSW by 2017, with at least another 19 full-scale HS-AD systems expected to come online. In general, batch, thermophilic, single-stage systems have been the dominant HS-AD system types being developed in the US. However, continuous and multi-stage systems are also available. There are at least nine vendors of HS-AD technologies in the US, four of which have facilities in operation and another four have projects in the planning, permitting, or construction phases. Each technology offers certain advantages and no single technology has emerged as dominant in industry at this time. HS-AD is economically competitive with composting or alternative conversion technologies such as WtE and ATT. However, it is unlikely that AD can compete with the low cost of landfilling without significant legislative aid. The primary factors that govern the economic sustainability of HS-AD projects are local waste disposal tipping fees, the quantity and quality of substrate available within close proximity of prospective HS-AD locations, the local markets for energy and compost, and legislative incentives with regard to renewable energy generation (e.g. RPS) and alternative OFMSW management (e.g. landfill bans and source-separation requirements/incentives). A final critical factor affecting the feasibility of HS-AD is the development of public-private partnerships for project

2. *What is the outlook for implementation of HS-AD in the state of Florida?*

In Florida, there is high demand for organics recycling and a lack of organics recycling infrastructure. Based on the analysis carried out in this thesis, the statewide recycling rate could be increased by as much as 13% through HS-AD implementation. Nutrient recovery could reach up to 7,000 and 3,500 TPY of bioavailable nitrogen and phosphorus, respectively. Approximately 500 MW of energy could be generated from this waste stream, which translates to either 175 MW of electricity (approximately 660,000 metric tons of CO_2 equivalents per year) and 325 MW of heat or nearly 80 million diesel gallon equivalents of compressed natural gas. Miami-Dade, Broward, Palm Beach, Hillsborough, Orange, Pinellas, Duval, Lee, and Alachua counties have the highest demand for OFMSW recycling infrastructure development and are the most feasible counties for HS-AD implementation. However, the low costs of energy and landfilling in Florida, lack of legislation incentivizing organics recycling, and lack of markets for compost and RECs make the economics of HS-AD particularly challenging. Without mandated source-separation of OFMSW and OFMSW disposal bans, HS-AD implementation is only economically feasible where significant quantities of high-quality substrate are available and partnerships can be formed for the provision of substrate and sale of energy. Universities or existing MSW facilities such as composting plants or landfills equipped with LFGTE equipment should be targeted as the most promising sites for a successful demonstration project.

3. *Is bioaugmentation using pulp and paper mill anaerobic sludge a viable method for improving methane yields from lignocellulosic wastes in HS-AD?*

A significant enhancement in methane yield from yard waste in HS-AD was achieved via bioaugmentation with P&P sludge as compared to methane yields achieved via inoculation with a conventional inoculum. Chemical data supports the hypothesis that the observed enhancement was a result of the hydrolytic communities in the P&P sludge possessing a superior ability to hydrolyze lignocellulosics. The observed enhancement in methane yield was also sustained in a subsequent phase of batch HS-AD via inoculation with digestate from the first phase of digestion, suggesting that this method may have potential to yield prolonged benefits with respect to process efficiency and net energy recovery. The enhancements achieved in this study (68-73%) are comparable to enhancements reported in various pretreatment studies, but the minimal impact of this strategy with respect to overall operational costs and environmental impacts make it an attractive alternative to pretreatment.

In addition to the bioaugmentation studies, a number of preliminary codigestion studies were performed. These studies showed promise with respect to the potential for addition of biosolids, addition of oyster shells, and/or bioaugmentation with pulp and paper sludge in codigestion to boost revenues and improve process stability and performance, and therefore warrant further investigation. Specific research recommendations associated with these and other relevant research topics include:

1. Further investigate the mechanisms by which bioaugmentation with pulp and paper anaerobic sludge enhances CH_4 yield in HS-AD of yard waste using electron and atomic force microscopy, microbiological assays, and cellulase enzyme additions.
2. Investigate the effects of varying substrate to inocula ratios on CH_4 yields and enhancement achievable through bioaugmentation with pulp and paper mill anaerobic sludge.
3. Further explore codigestion strategies in HS-AD, including the effects of biosolids addition, oyster shells addition, varying substrate/substrate ratios, and bioaugmentation with pulp and paper mill anaerobic sludge.
4. Investigate the usefulness of microaeration in HS-AD codigestion for reducing concentrations of hydrogen sulfide in biogas and enhancing methane yields.
5. Conduct experiments aiming to optimize particle size and percolate recirculation (rates and variation of rates over digestion cycles) in percolate-recirculating HS-AD.
6. Explore the effects of temperature (thermophilic) and scale (pilot-scale and full-scale) on the effectiveness of the above-described strategies.
7. Conduct LCA studies directly comparing HS-AD to L-AD, comparing the environmental impacts of bioaugmentation and pretreatment strategies, considering key parameters, such as feedstock composition, energy conversion efficiency, beneficial offsets for end-product alternatives, and distance to substrate sources and end-product markets, and examining sensitivities and "break-even" points of economic costs and environmental benefits.

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APPENDIX A: GLOSSARY

Anaerobic Digestion: The biochemical decomposition of organic matter in the absence of oxygen resulting in material stabilization (destruction of volatile solids) and production of biogas; in combination with aerobic digestion (microbial decomposition of organic matter in the presence of oxygen), anaerobic digestion is nature's way of recycling carbon and nutrients back into the cycles of life.

Anaerobic Digester: An engineered system designed to facilitate the anaerobic digestion process by creating a controlled oxygen-free environment; implemented broadly for converting various organic wastes and materials into usable energy in the form of biogas and nutrient rich organic fertilizer.

Biogas: A gas mixture produced in the anaerobic digestion process which primarily contains methane (CH_4) and carbon dioxide (CO_2), usually at ratios ranging from 1:1 (50 % CH_4) to 3:1 (75% CH_4), with trace concentrations of hydrogen gas (H_2), hydrogen sulfide (H_2S), nitrogen gas (N_2), and water vapor; biogas is directly combusted for use as a cooking fuel or for heating or lighting, converted to compressed natural gas and used as a vehicle fuel, or used in gas turbines, steam turbines, or combined heat and power units to produce heat and electricity.

Biomass: Any organic material of biological origin, for example: bacterial cells and all plants materials, including fruits, vegetables, grass, trees, bushes, etc.

Codigestion: Anaerobic digestion of more than one substrate simultaneously.

Centrate: Liquid portion effluent from centrifugation.

Comingled: Several waste substances mixed together at the source, for example, organic and non-organic wastes mixed together in individual residences to produce comingled municipal solid waste.

Digestate: The solid fraction of the byproduct resulting from the high-solids anaerobic digestate process.

Feedstock: Feedstock can refer to a single organic material to be processed via anaerobic digestion (synonymous with substrate), but in this paper refers specifically to the mixture of materials (substrate(s) and inoculum) being loaded to an anaerobic digester.

Fugitive emissions: Greenhouse gases that escape from a waste management facility into the atmosphere.

High-Solids Anaerobic Digestion: Anaerobic digestion process with total solids content greater than 15%.

Inoculum: The microbiologically active material being mixed with the substrate to increase the population density of anaerobic microorganisms present, thereby accelerating the start-up period of the AD process and overall process efficiency.

Leachate: Leachate in this paper refers specifically to the liquid which percolates from landfills, but can also refer to the liquid which percolates from solid waste material in high-solids anaerobic digestion systems (synonymous with percolate).

Liquid Anaerobic Digestion: Anaerobic digestion process with total solids content less than 15%.

Parasitic Energy: The energy consumed in a net-energy producing process such as incineration or anaerobic digestion.

Percolate: The liquid that percolates from solid waste material in high-solids anaerobic digestion systems; may also be referred to as leachate.

Pyrolysis: Anaerobic thermal conversion (450 ºC – 550 ºC) of biomass to bio-char and bio-oil.

Semi-dry Anaerobic Digestion: Anaerobic digestion process with total solids content between 10-15%.

Substrate: An organic material to be processed via anaerobic digestion; or, a primary or intermediate chemical compound associated with the metabolism of microorganisms active in anaerobic digestion.

Supernatant: The liquid portion of digestate produced via liquid anaerobic digestion; may also be referred to as centrate for systems which centrifuge effluents or as leachate when discussed in the context of high-solids anaerobic digestion.

Gasification: Controlled aerobic thermal conversion (>770 ºC) of biomass to syngas.

APPENDIX B: LIST OF ACRONYMS

AD – Anaerobic Digestion or Anaerobic Digester

ATT – Advanced Thermal Treatment

BMP – Biochemical Methane Potential

C/N – Carbon to Nitrogen Ratio

COD – Chemical Oxygen Demand

EfW – Energy-from-Waste

EU – European Union

GHG – Greenhouse Gas

HS-AD – High-Solids Anaerobic Digestion

L-AD – Liquid Anaerobic Digestion

LCA – Life Cycle Assessment

LCCA – Life Cycle Cost Analysis

MS-OFMSW – Mechanically Separated Organic Fraction of Municipal Solid Waste

MSW – Municipal Solid Waste

N – Nitrogen

OFMSW – Organic Fraction of Municipal Solid Waste

OLR – Organic Loading Rate

O&M – Operation and Maintenance

P – Phosphorous

REC – Renewable Energy Certificate or Renewable Energy Credit

RPS – Renewable Portfolio Standard

S/I – Substrate to Inoculum Ratio

SS-OFMSW – Source-Separated Organic Fraction of Municipal Solid Waste

TPY – Tones per Year

TS – Total Solids

US – United States

VFA – Volatile Fatty Acids

VS – Volatile Solids

WtE – Waste-to-Energy

APPENDIX C: CHAPTER 2 ADDITIONAL INFORMATION

C.1. Literature Review Methodology

The method utilized for conducting literature reviews for this research is described here. Because of the inconsistencies in terminology in the field of high-solids anaerobic digestion, searches for each topic reviewed (e.g. typical operating parameters, inhibition, codigestion, pretreatment, etc.) were carried out by combining key words associated with the topic (e.g. ammonia inhibition, VFA inhibition, etc., for the topic of inhibition) with each of the three terminology "roots" – (1) anaerobic digestion, (2) fermentation, and (3) anaerobic composting – and with each of the four terminology "prefixes" – (1) high-solids, (2) solid-state, (3) dry, and (4) solid-substrate – in every possible combination. Therefore, for each topic, numerous key words were searched in combination with each of the 12 different possible combination of the above listed "roots" and "prefixes" (e.g. ammonia inhibition in high-solids anaerobic digestion, VFA inhibition in high-solids anaerobic digestion, ammonia inhibition in high-solids fermentation, VFA inhibition in high-solids fermentation, ammonia inhibition in high-solids anaerobic composting, VFA inhibition in high-solids anaerobic composting, ammonia inhibition in solid-state anaerobic digestion, VFA inhibition in solid-state anaerobic digestion, and so on). Google Scholar was used as a search engine and the primary database of peer-reviewed literature from which articles were obtained was Science Direct. The vast majorities of articles that were reviewed were recent (published during or after 2010), however, some older article were also reviewed (published as early as the 1970's). A tremendous amount of research has been done on this topic, so the review conveyed in the document is *not* completely comprehensive in most cases. Rather, of all the literature reviewed, only the literature that was either considered particular important to the field or instrumental to the research conducted for this thesis was reported in this document.

C.2. Creating Consistency in HS-AD Terminology

There is demand to create consistency in terminology in many fields of science. In the field of HS-AD, as it was referred to in this document, creating consistency will result in more efficient communication of research conducted in the field and less confusion overall. HS-AD was selected for this document because it is the most accurate and strait forward of all possible terms. It is, simply, AD occurring with a high-solids content (relative to traditional liquid AD). All other terms fall short in one fashion or another. *Fermentation* as an alternative term is not accurate because AD involves both anaerobic fermentation and anaerobic *respiration* (CO_2 reduction to CH_4 via hydrogenotrophic methanogenesis). *Anaerobic composting* as an alternative term is fine with respect to the accuracy of the term, but can create confusion (e.g. if *composting* is used, but not preceded by anaerobic or aerobic). *Composting* is the well-established term used for aerobic biological conversion of organic matter and therefore, the use of the term *anaerobic composting* should be avoided. It is clear then, that AD should be the one and only term used to describe the biochemical conversion of organic matter in the absence of free oxygen. Next, the use of solid-state AD (SS-AD) or solid-substrate AD can easily be confused with single-stage AD or steady-state AD (continuous AD which reaches consistent operating conditions), and thus falls short relative to HS-AD. Finally, the use of *dry AD* as opposed to HS-AD falls short with respect to the accuracy of the term, as it indicates that the process is dry – meaning no water –

when in fact there is a lot of water present (up to 85 percent by mass). It can then be concluded that HS-AD is the most accurate and appropriate term for this field.

C.3. Control Volume for Deriving $\Delta HRT_{percolate}$

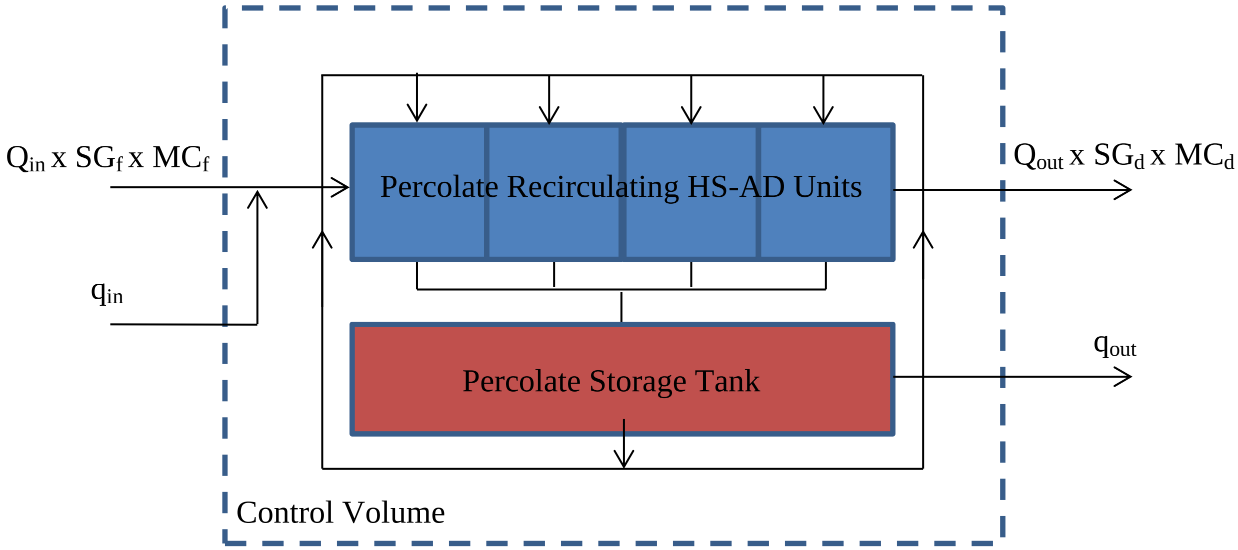


Figure C.1: Control volume for deriving $\Delta HRT_{percolate}$ of a percolate recirculating system.

$$\Delta HRT_{percolate} = V_{system}/\Delta Q_{percolate} = V_{system}/(Liquid\ Flowrate_{in} - Liquid\ Flowrate_{out})$$

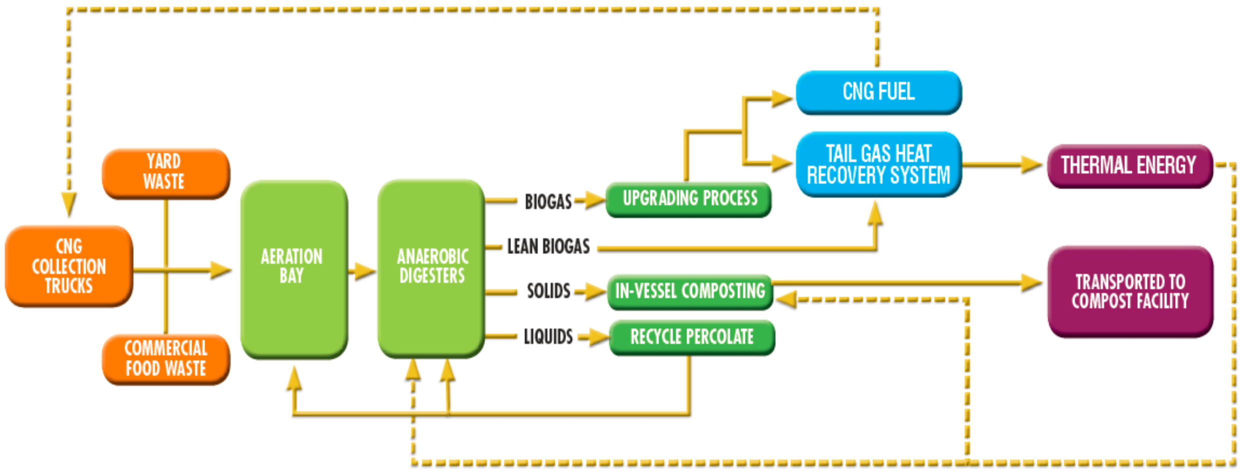
$$\Rightarrow \Delta HRT_{percolate} = V_{system}/[(Q_{in}\ x\ SG_f\ x\ MC_f + q_{in}) - (Q_{out}\ x\ SG_d\ x\ MC_d + q_{out})]$$

$$\Rightarrow \Delta HRT_{percolate} = V_{system}/[(Q_{in}\ x\ SG_f\ x\ MC_f) - (Q_{out}\ x\ SG_d\ x\ MC_d) + q_{in} - q_{out}]$$

Nomenclature is defined in Section 2.1.4.5. If Liquid Flowrate$_{in}$ is greater than Liquid Flowrate$_{out}$, then accumulation will occur and $HRT_{percolate}$ will increase. If Liquid Flowrate$_{in}$ is less than Liquid Flowrate$_{out}$, then percolate volume will decrease and $HRT_{percolate}$ will decrease. However, if Q_{in} x SG_f x MC_f is equal to Q_{out} x SG_d x MC_d, then the volume of percolate in the percolate storage tank will not change over time, hence, no water addition or percolate removal is necessary ($q_{in} = 0$; $q_{out} = 0$) and there is no change in $HRT_{percolate}$. If Q_{in} x SG_f x MC_f is greater than Q_{out} x SG_d x MC_d, then percolate will accumulate and excess percolate must be removed

(q_{out} > 0; q_{in} = 0). If Q_{in} x SG_f x MC_f is less than Q_{out} x SG_d x MC_d, then percolate volume will decrease and water addition will be necessary (q_{in} > 0; q_{out} = 0).

C.4. Images of HS-AD Systems Available in the United States



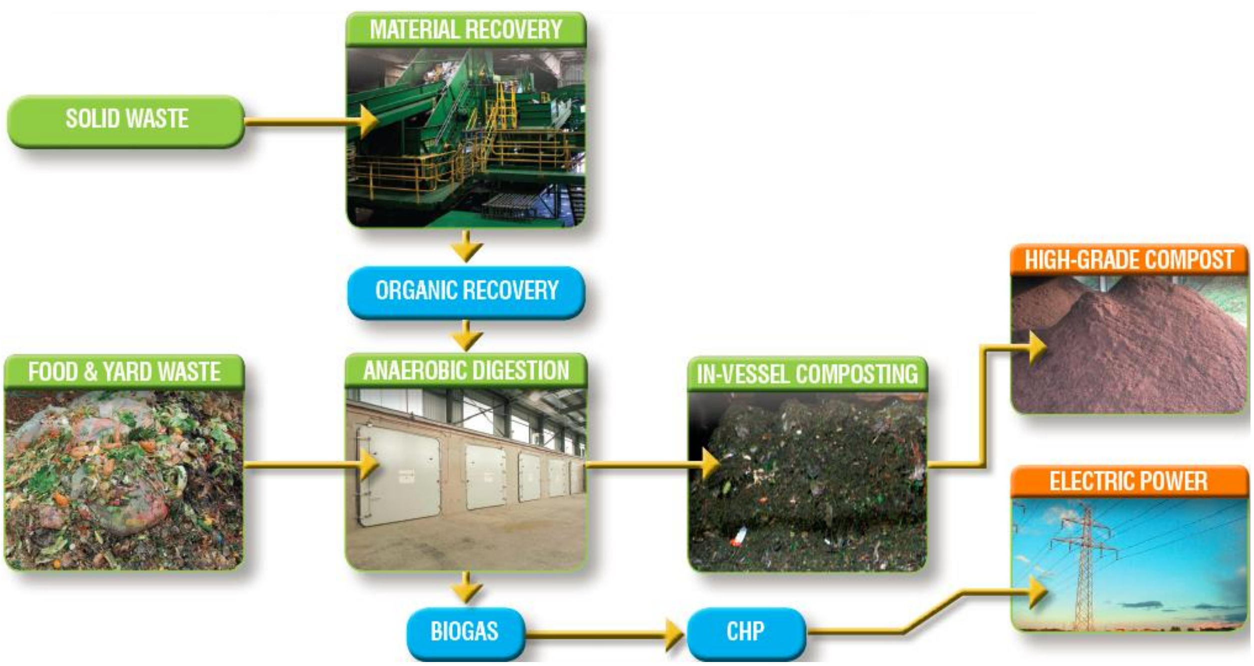


Figure C.2: Zero Waste Energy SMARTFERM HS-AD system schematics. South San Francisco facility process (top) and San Jose facility process (bottom) (from vendor website: http://zerowasteenergy.com/).

Figure C.3: Zero Waste Energy Monterey, CA Facility model (from ZWE, 2013b) and photographs from field visit in May, 2015. Middle row from the left: fresh yard waste, fresh food waste, digester units, CNG unit (and emergency flare, biofilter, and controls building); bottom row from the left: fresh digestate, trommel screen, post-trommel digestate composting, and final compost product.

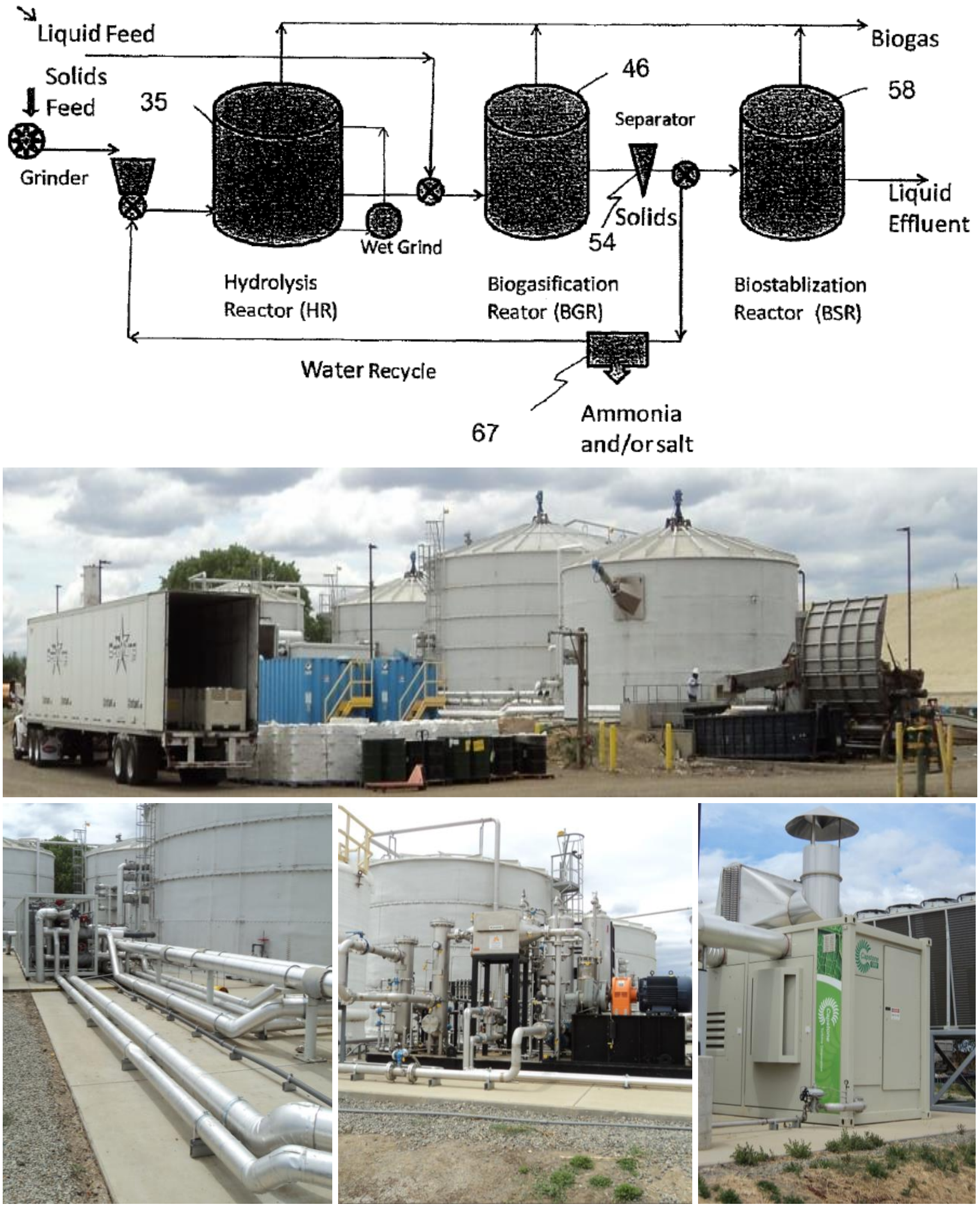


Figure C.4: CleanWorld HS-AD system schematic and UC, Davis facility photographs from field visit in May, 2015. Process schematic (top) (from Zhang, 2013) and UC, Davis facility view from the entrance (middle) and piping, pumps, and micro turbine set (bottom, left to right).

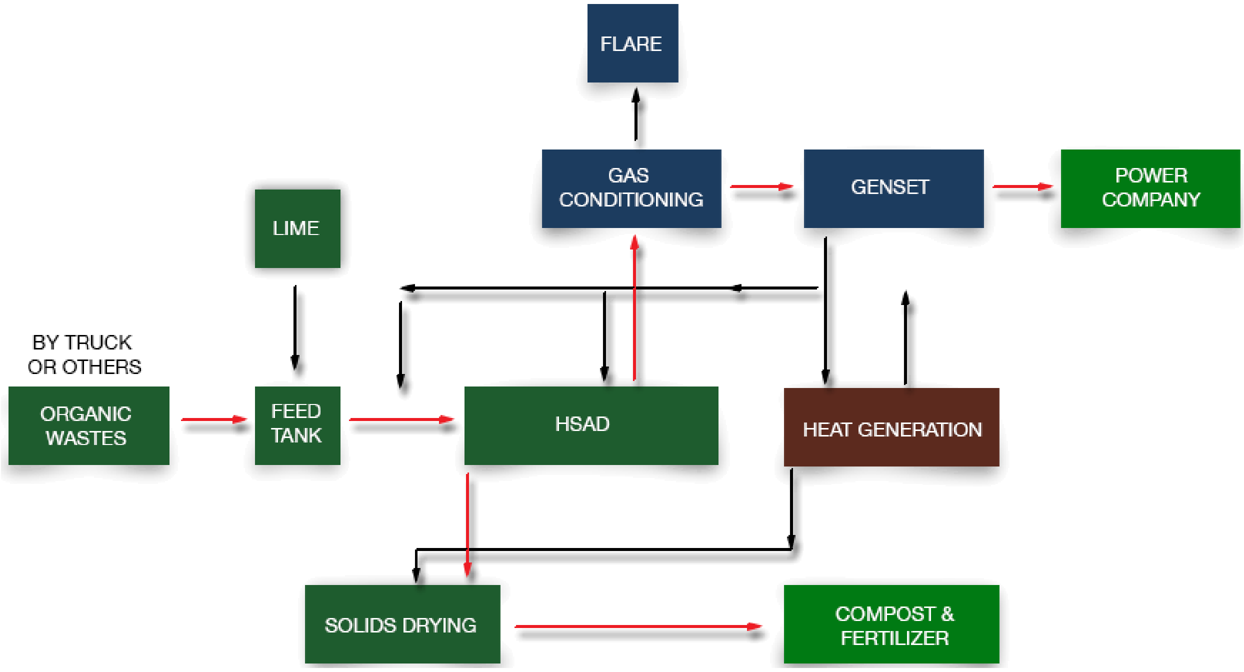


Figure C.5: Orbit Energy HS-AD system schematic (from vendor website: http://www.orbitenergyinc.com/)

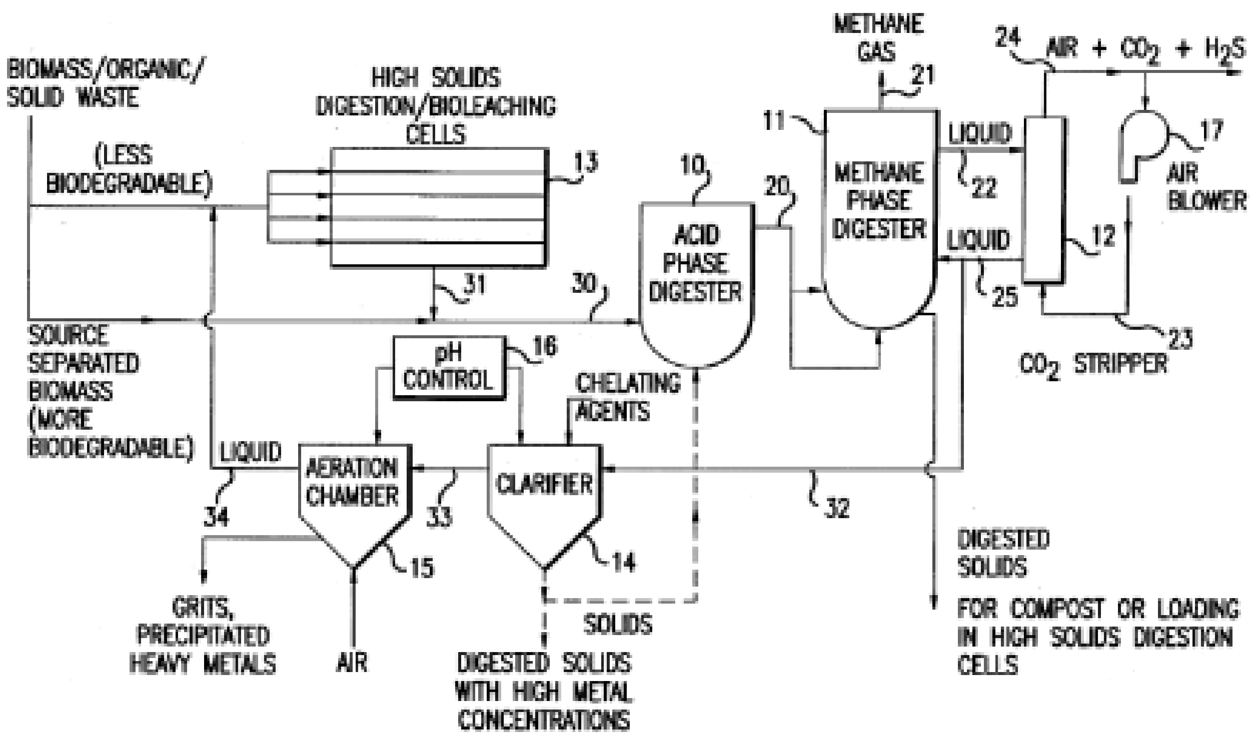


Figure C.6: Turning Earth/Aikan North America HS-AD system schematic (from Google Patents: http://www.google.com/patents/US5500123).

Figure C.7: Turning Earth Central Connecticut's Three-Phase Integrated High Solids Dry Fermentation In-Vessel Composting facility model, Southington, CT (from Turning Earth, 2014, used with permission).

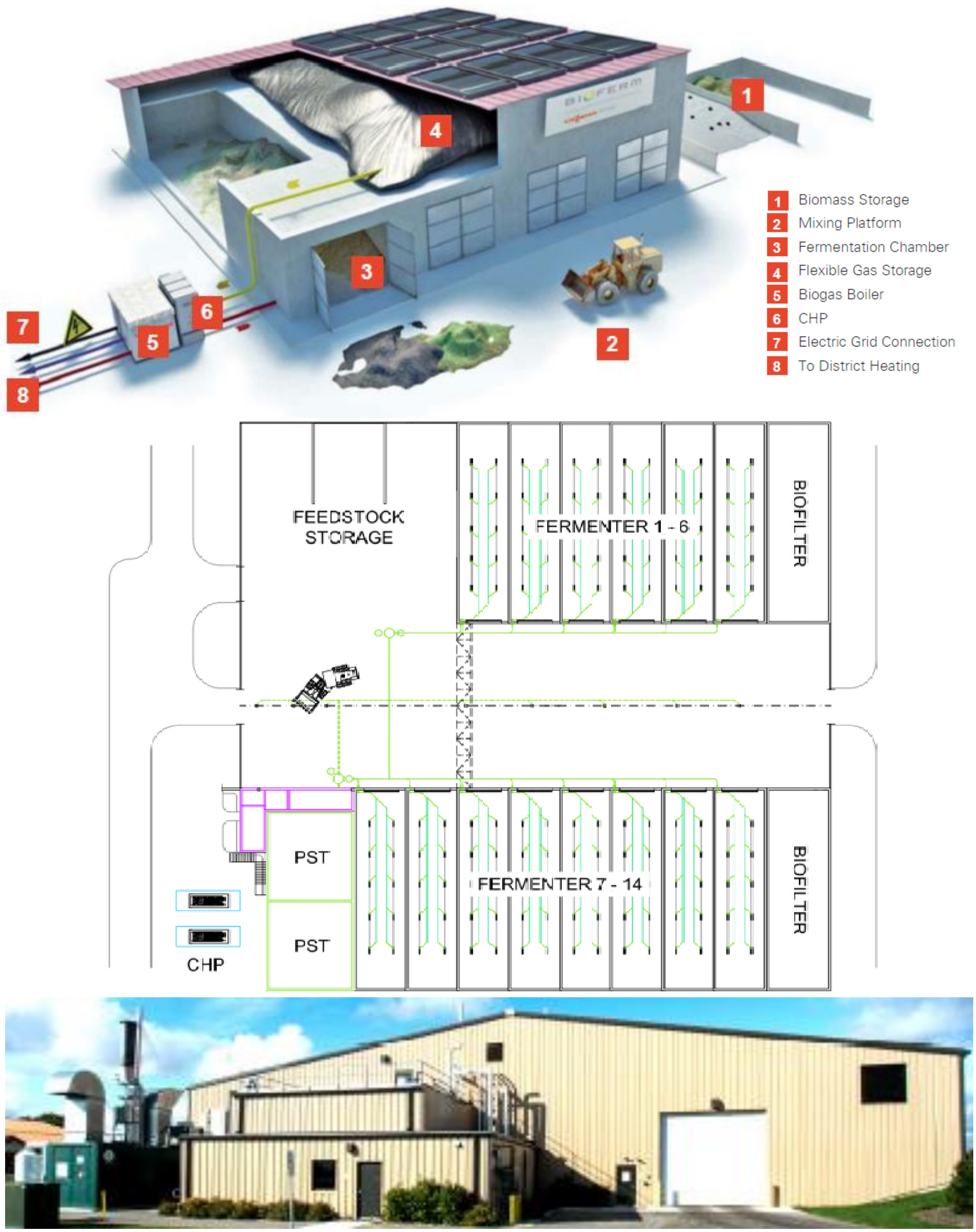


Figure C.8: BIOFerm Dry Fermentation system model (top), site plan (middle), and photographs from the UW, Oshkosh BIOFerm system (bottom) (from BIOFerm, 2014a, used with permission).

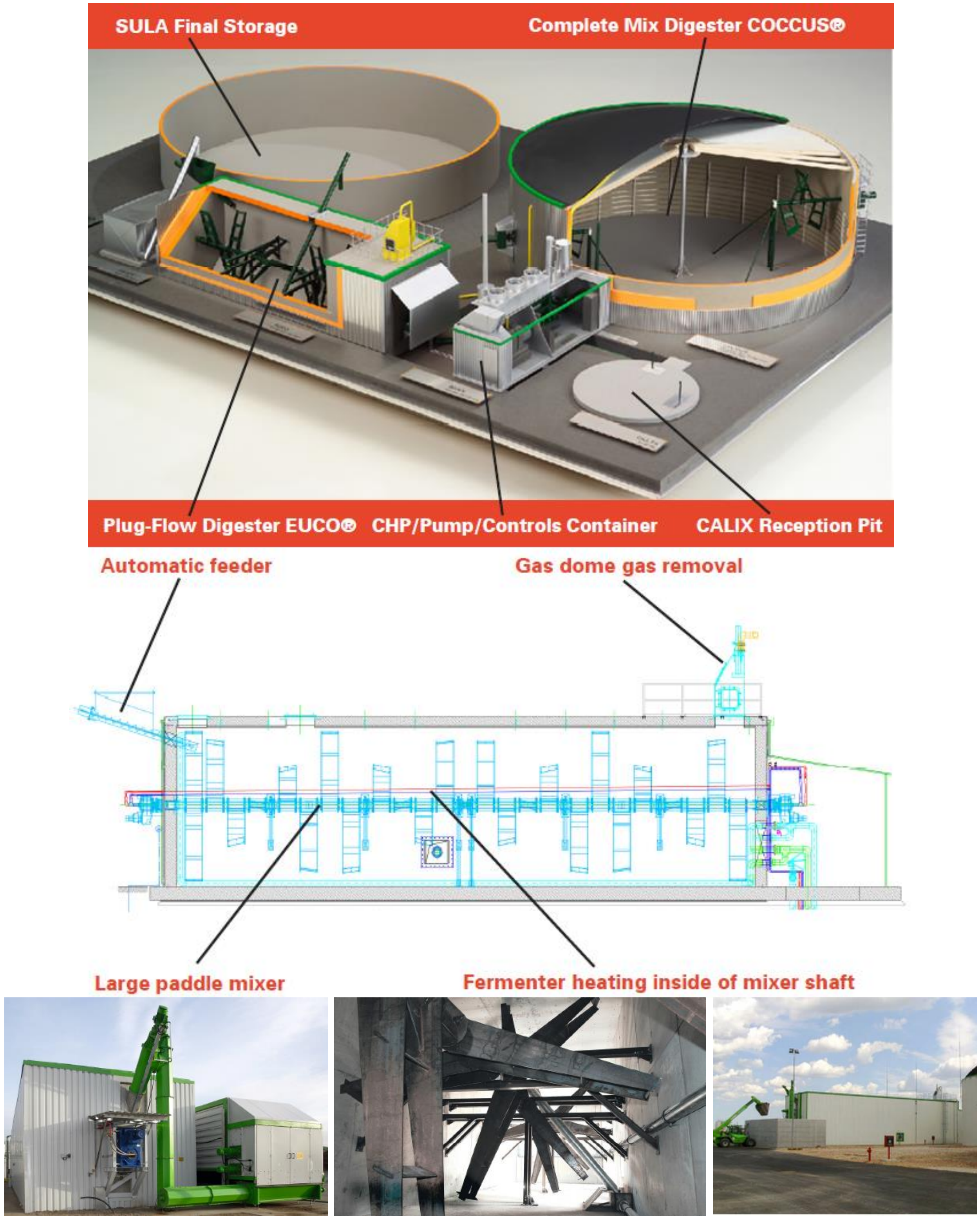


Figure C.9: BIOFerm EUCO HS-AD system model (top), section view (middle), and photographs (bottom) (from BIOFerm, 2014b, used with permission).

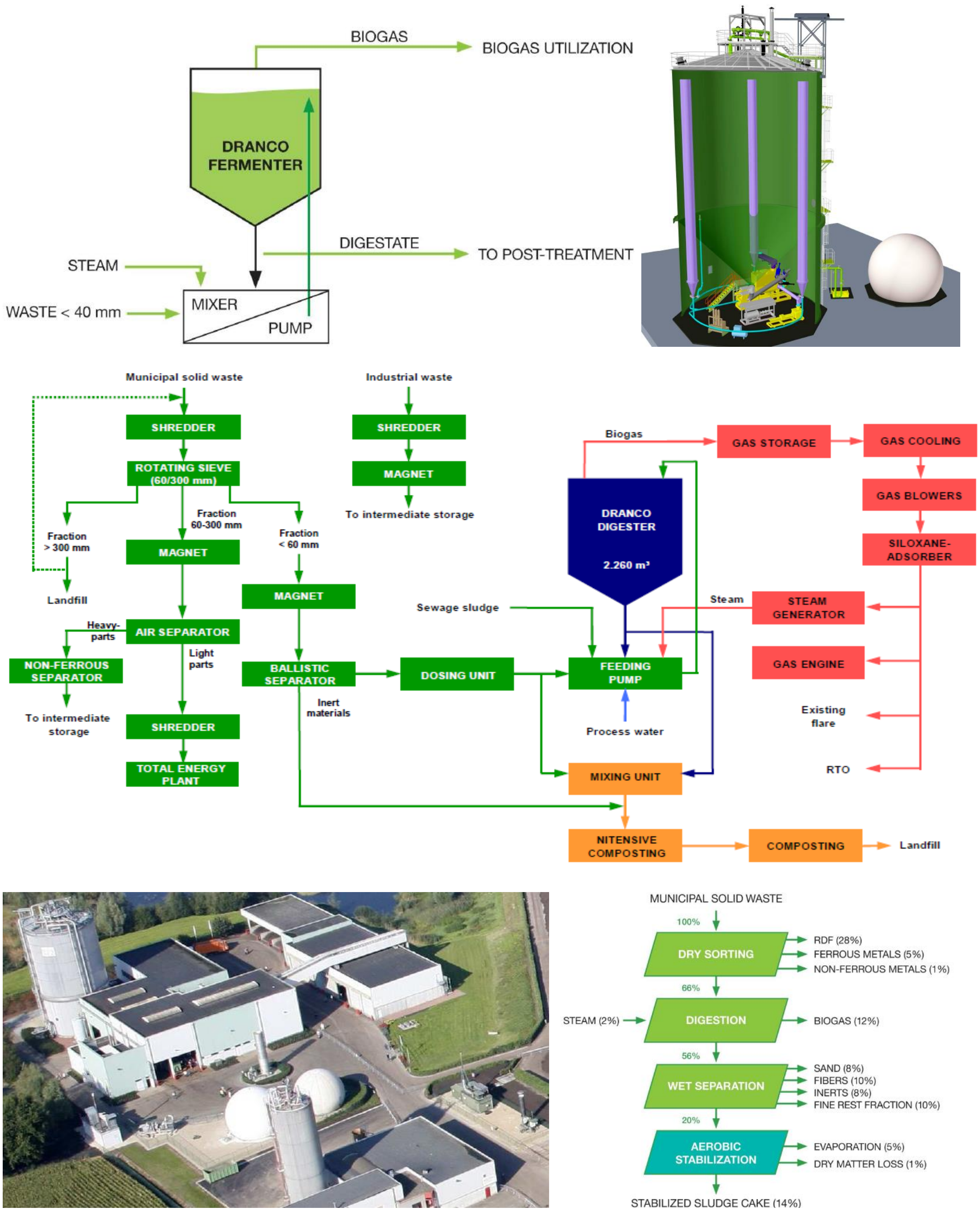


Figure C.10: Organic Waste Systems DRANCO system simple process schematic (top left), model (top right), Pohlsche Heide with partial steam digestion process schematic (middle), Brecht I and II facilities (bottom left), and Sordisep process schematic (bottom right) (from De Baere, 2012, used with permission).

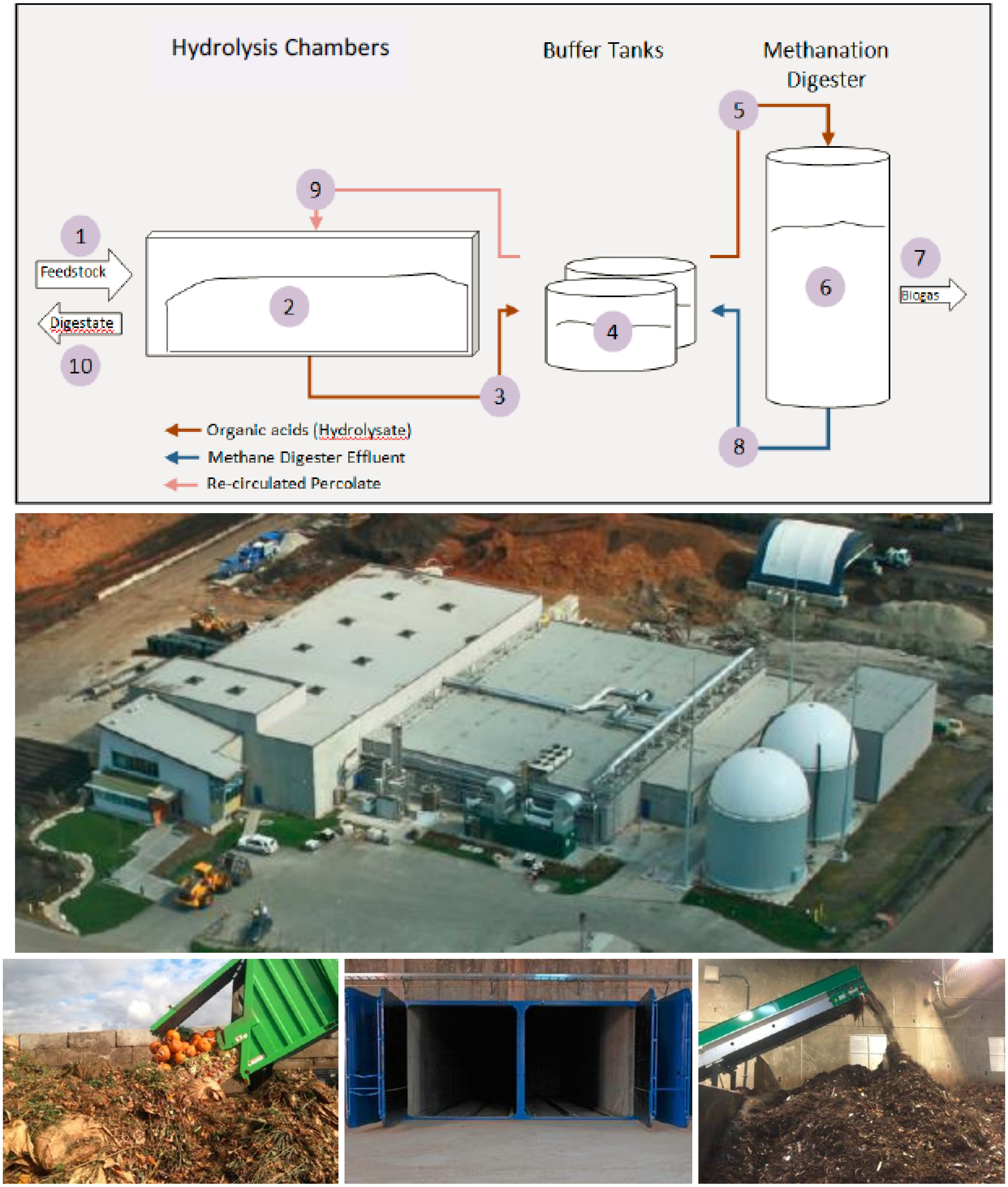
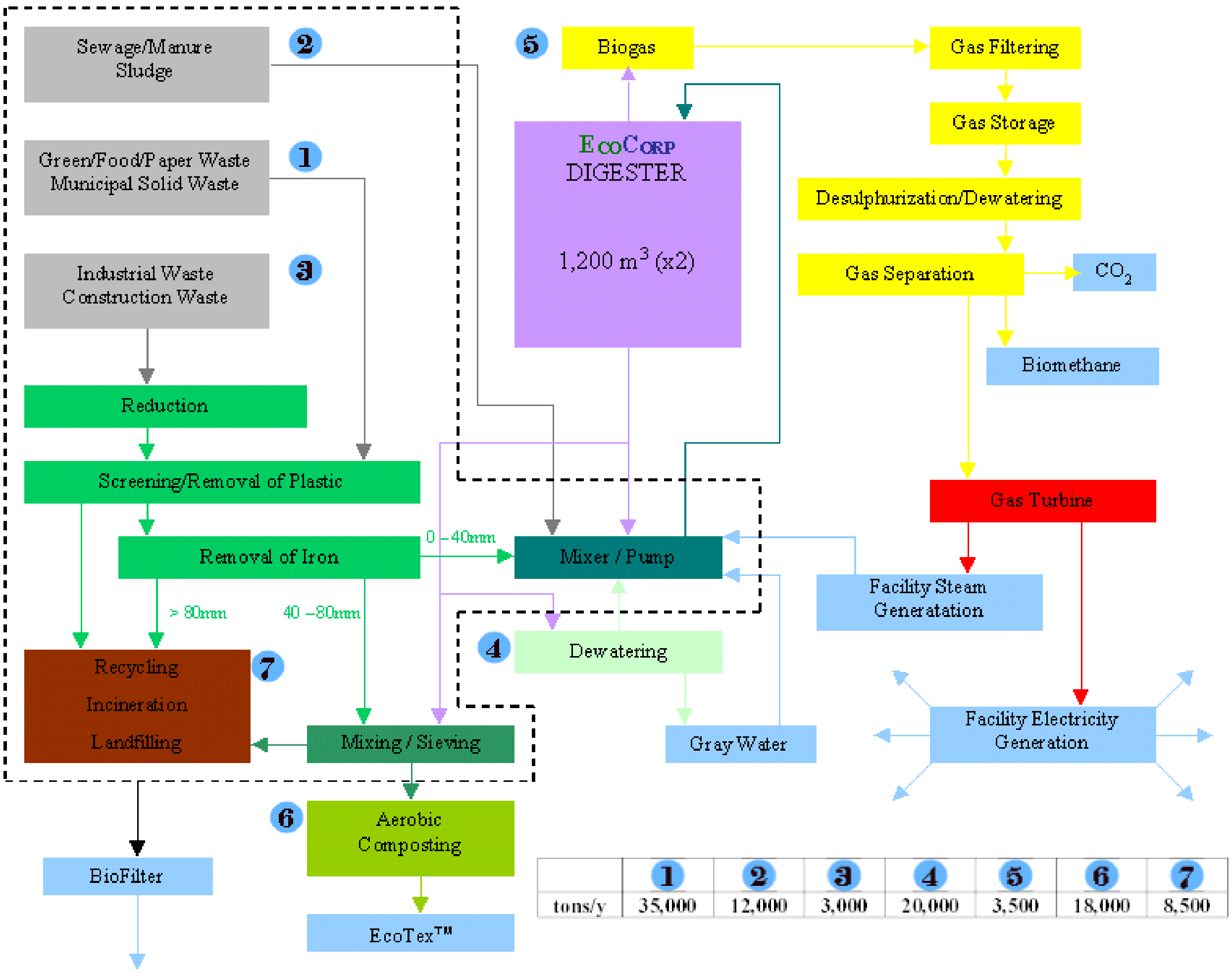


Figure C.11: Harvest Power HS-AD system schematic (top), Metro Vancouver facility (middle), fresh feedstock (bottom left), digestion "tunnels" (bottom center), and digestate (bottom right) (from Harvest Power, 2014).



	1	2	3	4	5	6	7
tons/y	35,000	12,000	3,000	20,000	3,500	18,000	8,500

Figure C.12: EcoCorp HS-AD process schematic (from vendor website: http://www.ecocorp.com/Other_Pages/Flowchart.html, used with permission).

APPENDIX D: CHAPTER 3 ADDITIONAL INFORMATION

The equations used to quantify the energy recovery potential achievable through HS-AD implementation for OFMSW recycling in Florida are displayed in a generalized manner such that project specific data can be inputted to provide estimates for future analyses of this kind. Annual energy recovery potential achievable through HS-AD of substrate *i* (ERP_{Si}) is calculated as:

$$ERP_{Si}\ (kWh/year) = 9.7 \cdot M_{Si} \cdot VS_{Si} \cdot ABY_{Si} \cdot C_{CH4} \qquad (Eq.\ D1)$$

where *9.7* is the conversion factor from m^3 of CH_4 to kWh of energy (9.7 kWh/m^3 CH_4) (SGC, 2012), M_{Si} is the mass loading rate of substrate *i* (kg/yr), VS_{Si} is the volatile solids content of substrate *i* by wet weight, ABY_{Si} is the average biogas yield from substrate *i* (m^3 biogas/kg), and C_{CH4} is the methane content of the biogas (% by volume). The total annual energy recovery potential achievable through HS-AD of *n* substrates (TERP) is calculated as:

$$TERP\ (kWh/year) = \sum_{i=1}^{n} ERP_i \qquad (Eq.\ D2)$$

This total energy recovery potential can then be translated to electricity generation potential and recoverable heat by multiplying by the electrical efficiency of an electric generator (e.g. 35%) and the thermal efficiency of a combined heat and power system (~40%) (SGC, 2012). Or, the approximate CNG production potential (*CNGPP*) can be calculated as:

$$CNGPP\ (DGE/year) = 1{,}010 \cdot 3.79 \cdot \eta_C \cdot TERP \qquad (Eq.\ D3)$$

where *1,010* is the conversion factor from kWh/m^3 CH_4 to kWh/L diesel, *3.79* is the conversion factor from L to gallons, and η_C is the efficiency of conversion of biogas to compressed natural gas (purification and compression) (67%) (ZWE, 2013b).

APPENDIX E: CHAPTER 4 ADDITIONAL INFORMATION

E.1. Yard Waste Bioaugmentation Study – Additional Information

In this study, specific methane yields at STP were calculated as:

$$Specific\ Methane\ Yield\ (L\ CH_4/kg\ VS) = \frac{(V_{CH4,\ Di} - V_{CH4,\ Bi})}{M_{YW} \cdot \%VS_{YW}} \cdot \frac{P_1 T_2}{T_1 P_2} \qquad (Eq.\ E1)$$

where $V_{CH4,\ Di}$ (L) is the cumulative volume of methane generated in a bioaugmented digester, $V_{CH4,\ Bi}$ (L) is the cumulative volume of methane generated in a blank digester i, M_{YW} (kg) is the mass of yard waste added to each bioaugmented and control digester (0.04 kg for Phase 1 of batch HS-AD and 0.03 kg for Phase 2), $\%VS_{YW}$ is the percent by total mass fraction of the yard waste sample that is VS, P_1 is the pressure in the thermostatically controlled room, which was assumed to be approximately 1 atm, P_2 is standard pressure (1 atm), T_1 is the temperature in the thermostatically controlled room (35ºC = 310K), and T_2 is standard temperature (0ºC = 275K).

Percent enhancement in methane yields were calculated as:

$$\%\ Enhancement = \frac{Specific\ Methane\ Yield_1 - Specific\ Methane\ Yield_2}{Specific\ Methane\ Yield_2} \cdot 100\% \qquad (Eq.\ E2)$$

where $Specific\ Methane\ Yield_1$ is the specific methane yield obtained from bioaugmented digesters (digesters inoculated with P&P sludge or digestate from digesters inoculated with P&P sludge) and $Specific\ Methane\ Yield_2$ is the specific methane yield obtained from control digesters (digesters inoculated with wastewater anaerobic sludge or digestate from digesters inoculated with wastewater anaerobic sludge).

A final point that should be made from the yard waste bioaugmentation study is that transitioning from reading biogas on a daily basis in Phase 1 of batch HS-AD (day 1-28) to reading biogas every other day created a noticeable elbow, or change in slope, the specific methane curves of both bioaugmented and control digesters (Figure 4.3). As described in Section 4.2, each time biogas was read, the digesters were shaken vigorously for approximately five seconds before taking readings to dislodge any gas bubbled from the substrate. It can then be concluded, because there are no other possible explanations, that the shaking of the digesters was having a significant effect on overall process efficiency. On an unrelated note, Figure E.1 displays the mass balance from both Phases of HS-AD, which came out with reasonable errors.

Table E.1: Mass balance for Phases 1 and 2 of HS-AD.

	Phase 1		Phase 2	
	Bioaugmented Digesters	Control Digesters	Bioaugmented Digesters	Control Digesters
Initial Mass/Mass In (g)	**122.00**	**102.50**	**124.00**	**132.00**
Average Final Mass (g)	116.30	101.45	120.43	128.87
Mass of Cumulative CH_4 (g)	1.43	0.72	0.56	0.38
Mass of Cumulative CO_2 (g)	4.76	2.39	1.75	1.19
Total Mass Out (g)	**122.49**	**104.55**	**122.74**	**130.45**
% Error in Mass Balance	0.40%	2.00%	1.02%	1.17%

E.2. Preliminary Bioaugmentation, Codigestion, and Pilot Experiments

Preliminary bench-scale bioaugmentation and codigestion experiments were carried out in the fall of 2014 to become familiar with HS-AD experimentation and chemical analysis methods. Challenges were encountered from which lessons were learned. However, the data from the experiments were considered preliminary due to high human error resulting from these challenges. The experiments are described here in chronological fashion to provide future researchers with an understanding of how and why the experiments have evolved. Note that these experiments were inspired by and based largely on the work of Mussoline et al. (2013).

The first bench-scale experiment (B.S. #0) was conducted using 1.9-L plastic bottles with tabulation located at the bottom of the bottles. The idea was that larger volume experiments would be helpful because of the heterogeneity of yard waste. For this study, yard waste was obtained from the Falkenburg yard waste processing facility because that yard waste was thought to be more representative than simply pulling a yard waste sample from the side of the street or from USF botanical gardens. The yard waste was sieved to approximately ½ inch maximum particle size to improve sample homogeneity. For this study, only two digesters were set up because it was expected that problems would occur and that it would therefore not be worthwhile to set the digesters up in triplicate. Both digesters were first loaded with expanded clay to prevent clogging of the tabulation. The idea was that tubes would be attached on one end to the tabulation and on the other end to IV bags such that the percolate from the digesters could be captured and manually recirculated on a daily basis. Biogas was to be collected in gas bags for this experiment and measured periodically via liquid displacement. 500 grams of yard waste, 250 grams of food waste, and 250 grams of biosolids were loaded to both digesters, one was inoculated with 500 grams P&P sludge, and the other was inoculated with 500 grams of wastewater sludge (S/I ratio of 2/1). The ratios were selected based on what was considered representative based on relative generation rates in Florida. Blanks were set up, as usual, to correct for biogas yields from the inocula. Figure E.1 shows the experimental setup.

Figure E.1: Bench-scale experiment 0 setup.

The primary challenge encountered during this study can be summarized as follows: once daily percolate recirculation led to the development of liquid pathways through which the percolate would quickly flow resulting in minimal material wetting and substrate/microorganism/ nutrient/ contact times. Constant and well-distributed recirculation is required to avoid this problem, as seen in full-scale systems. The second bench-scale (B.S. #1) experiment turned form codigestion of yard waste, food waste, and biosolids to only dealing with yard waste. This decision was made to simplify the experiment design such that one research question could be isolated: (i) What are the effects of bioaugmentation with P&P sludge on methane yields from yard waste in HS-AD? This experiment was conducted identically to the experiment described in Chapter 4 of this thesis with the exceptions of the biogas reading method, only one intermediate was set up, P&P sludge was obtained from a mill in the Netherlands, and a third set of digesters was setup and inoculated with a 50/50 mixture of P&P sludge and wastewater sludge. In this experiment, inverted burettes were used to create a "simple" methane measuring apparatus. However, the main challenge encountered in this experiment was leakage of biogas during methane measurements, resulting in large standard deviation in methane yields. Figure E.2 shows the experiment and methane measurement setup. Figure E.3 displays the methane yields observed from the three digester sets.

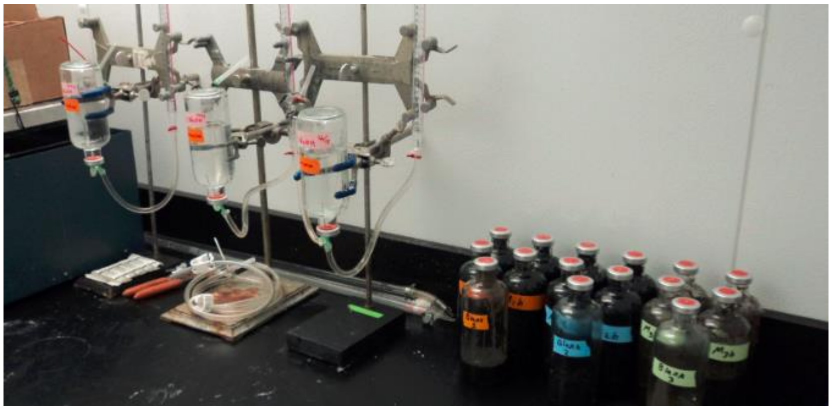


Figure E.2: Bench-scale experiment 1 setup, including methane measuring apparatus.

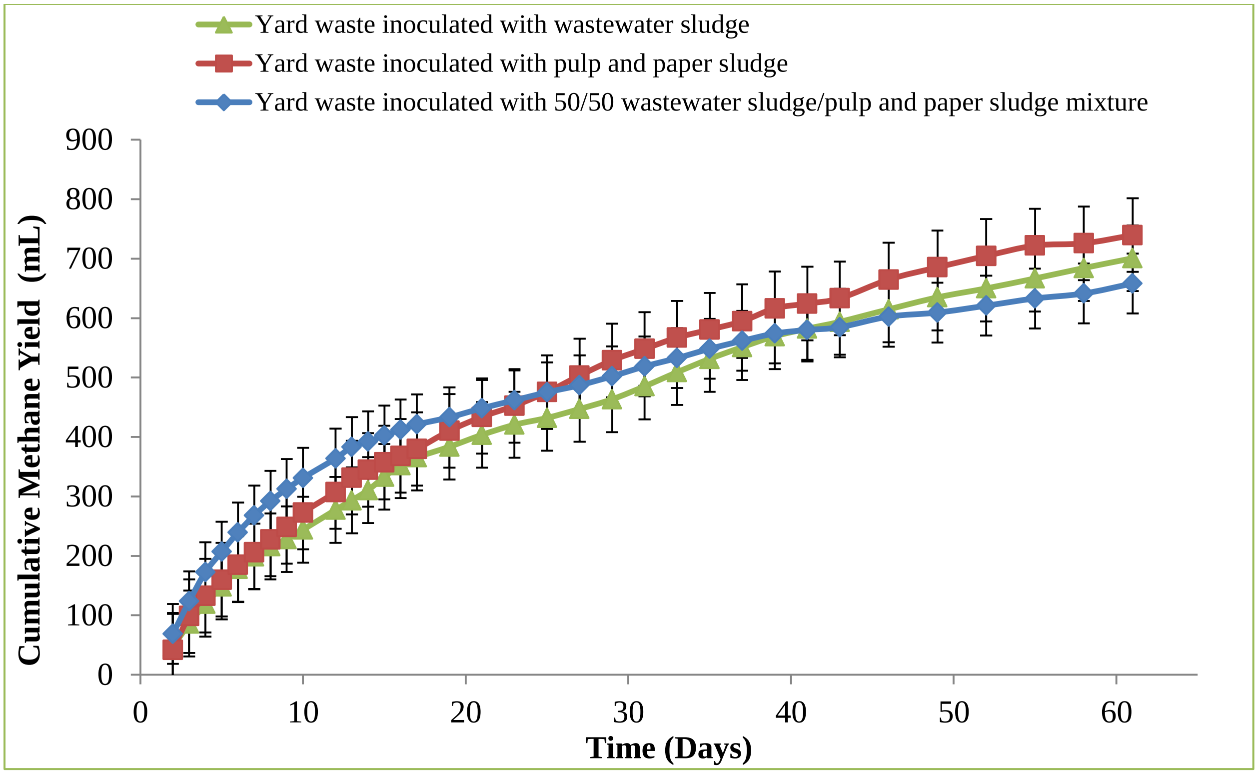


Figure E.3: Cumulative methane yields observed in bench-scale experiment 1.

For the second trial of the bioaugmentation study (B.S. #2), more intermediates digesters were prepared for chemical analysis and the set of digesters with mixed inocula was eliminated to further simplify the experiment. With the standard deviations of methane yields reduced to acceptable values (as reported in Chapter 4), attention was turned back to codigestion to address to two additional research objectives: (i) study the effects of biosolids addition in codigestion of yard waste and food waste, and (ii) study the effects of P&P bioaugmentation in codigestion of yard waste, food waste, and biosolids.

The materials and methods for the next experiment (B.S. #3) were identical to those described in Section 4.2, except additional substrates were added and different masses were selected based on volume limitations (250 mL glass bottles). Substrate to substrate ratios were selected such that the digesters would not be overloaded and the S/I ratio was changed to 1/1.5 from 1/1 to further reduce the OLR. However, within five days of digestion experimental digesters that were inoculated with wastewater anaerobic sludge were showing signs of methanogenic inhibition. This was likely a result of the low alkalinity present in wastewater anaerobic sludge relative to the P&P sludge and perhaps a more active acidogenic population and less active methanogenic population in the wastewater sludge relative to the P&P sludge. The digesters may have rebounded given enough time, but instead, crushed oyster shells were added, bringing in a third research objective: (iii) to study the effects of oyster shell addition on the recovery of overloaded batch high-solids anaerobic digesters.

Yard waste was again obtained from the University of South Florida Campus, P&P sludge was again obtained from the Tembec Pulp and Paper Mill in Metane, Canada, wastewater sludge was again obtained from the HFCA Wastewater Treatment Facility in Tampa, Florida, undigested dewatered biosolids were obtained from the Falkenburg Advanced Wastewater Treatment Plant in Tampa, Florida, and a synthetic food waste was prepared according to formula described in Appendix D (Table E.2). Three different experimental digesters and two sets of blanks were set up in triplicate in this study, the compositions of which are shown in Table E.3. Three additional duplicates of experimental digesters were again set up for intermediate chemical analysis. Figure 4.10 (in Section 4.3.7) shows the specific methane yields observed in the study and Figure E.4 shows the percent enhancement in methane yield observed from bioaugmentation with P&P sludge and from biosolids addition.

Overall, bioaugmentation with P&P sludge led to improved system stability and a net enhancement in methane yield of 15%. Oyster shell addition led to rapid system recovery, and biosolids addition resulted in a slightly more rapid system recovery after oyster shell addition and slightly higher specific methane yields through the majority of the study. This study is being reproduced with oyster shell addition during experimental setup and with reduced OLRs.

Table E.2: Generalized food waste composition and synthetic food waste formula.

Generalized Food Waste Composition		Synthetic Food Waste Formula	
Substance	% by weight	Substance	% by weight
Fruits and Vegetables	60	Apple	10
		Banana Peel	12.5
		Orange Peel	7.5
		Zucchini Squash	10
		Romaine Lettuce	10
		Carrot	5
		Onion	5
Dairy	10	Egg Shell	2
		Cheese	8
Bread/Rice/Grains	15	Multigrain Bread	15
Meats	15	Pork Sausage	15
Total	100	Total	100

Note: Generalized food waste composition based on VALORGAS (2010)

Table E.3: Contents of the three sets of experimental digesters and two sets of blanks tested in the preliminary codigestion study.

	D1	D2	D3	B1	B2
Yard Waste (g)	40	40	40	0	0
Food Waste (g)	5	5	5	0	0
Biosolids (g)	15	15	0	0	0
Wastewater Sludge (g)	0	90	67.5	0	90
Paper Mill Sludge (g)	90	0	0	90	0
Total mass	150	150	112.5	90	90

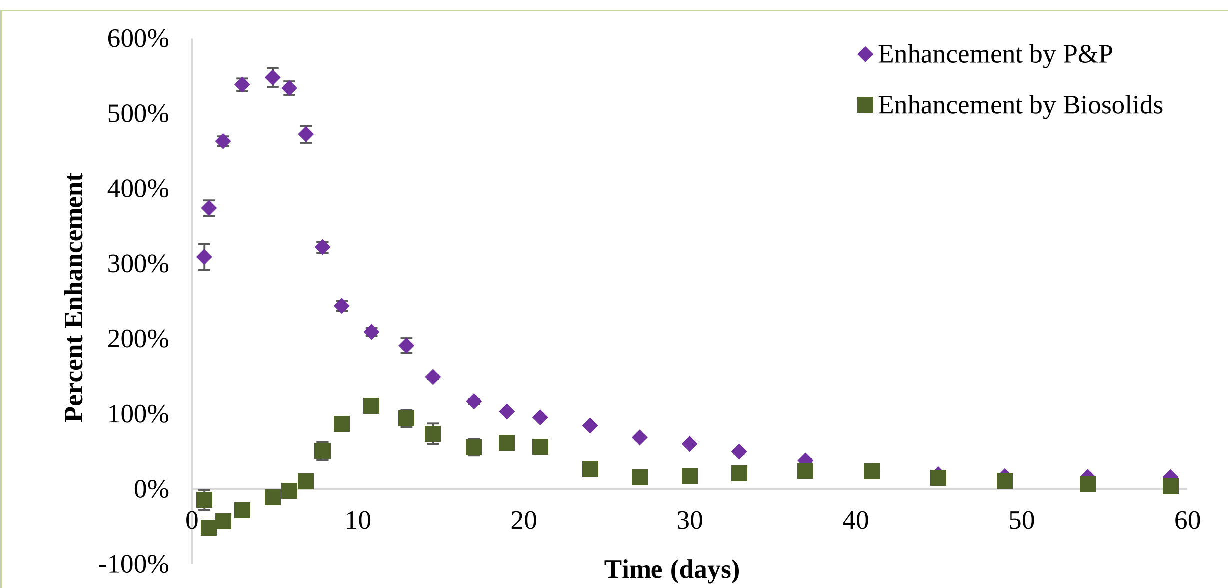
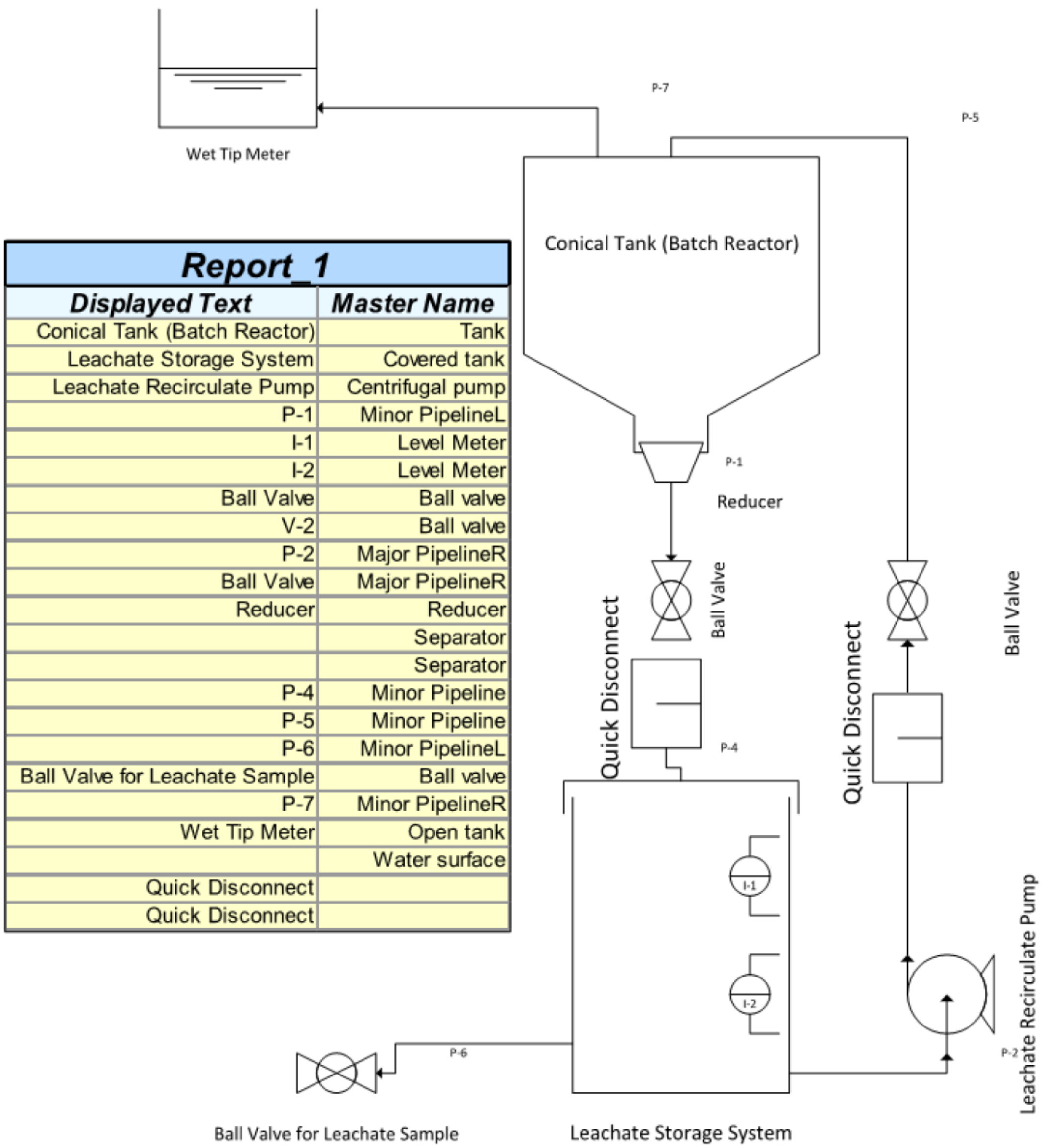


Figure E.4: Enhancements in methane yield observed from bioaugmentation with P&P sludge and from biosolids addition in high-solids anaerobic codigestion.

An additional line of experiments are planned to be conducted at the pilot-scale using a 10-gallon percolate recirculating HS-AD system that was designed by George Dick. Figure E.5 shows the process flow diagram and parts list of the pilot-scale system and Figure E.6 shows the fully constructed system. One preliminary study was conducted using yard waste inoculated with wastewater sludge, during which 16 days of biogas data was collected (Figure E.7) before challenges were encountered with gas leakage and biogas measurement via wet-tip meter.



Report_1

Displayed Text	Master Name
Conical Tank (Batch Reactor)	Tank
Leachate Storage System	Covered tank
Leachate Recirculate Pump	Centrifugal pump
P-1	Minor PipelineL
I-1	Level Meter
I-2	Level Meter
Ball Valve	Ball valve
V-2	Ball valve
P-2	Major PipelineR
Ball Valve	Major PipelineR
Reducer	Reducer
	Separator
	Separator
P-4	Minor Pipeline
P-5	Minor Pipeline
P-6	Minor PipelineL
Ball Valve for Leachate Sample	Ball valve
P-7	Minor PipelineR
Wet Tip Meter	Open tank
	Water surface
Quick Disconnect	
Quick Disconnect	

Figure E.5: Pilot-scale HS-AD system process flow diagram and parts list.

Figure E.6: Photograph of fully-constructed 10-gallon pilot-scale HS-AD system.

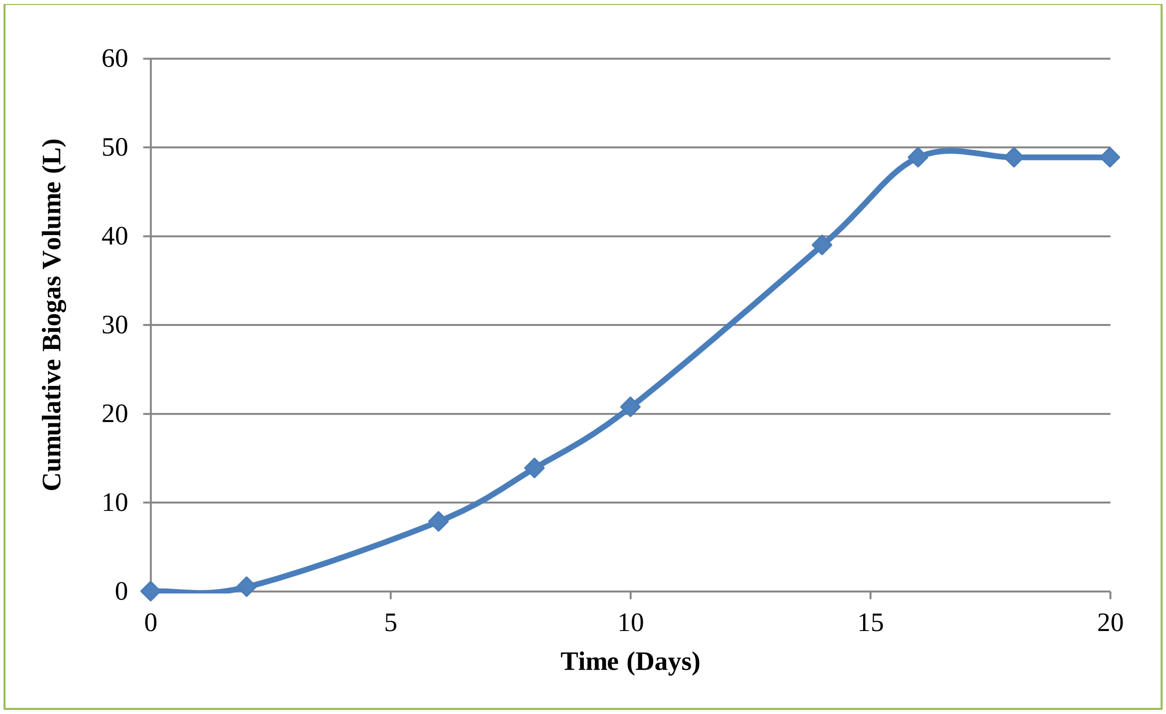


Figure E.7: Cumulative biogas data from preliminary pilot-scale HS-AD experiment.

The challenges with the pilot system continued through a second and third round of pilot-scale experiments. In the second round, biogas leaks were again suspected, leading to a thorough resealing effort before the beginning of another identical trial run. In the third round, it was verified that all leaks had been sealed and biogas production began within 24 hours of sealing the digester. However, problems were again encountered with biogas measuring via wet-tip meter. Biogas production led to accumulation of pressure in the system of gas would not flow through the wet-tip meter. The majority of the biogas that was produced over the first several days of the experiment was unintentionally discharged during troubleshooting and exploration of potential solutions. Efforts are ongoing to develop sound operational techniques such that future experiments will yield reliable data.

APPENDIX F: COPYRIGHT PERMISSIONS

The copyright permission from Maureen Kinyua for Figure 2.1 is displayed below.

Gregory Hinds <ghinds@mail.usf.edu>
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Oct 9 (10 days ago)

Hello Maureen,

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I was advised to request permission via email and paste the email record into my Appendix for copyright purposes.

Thank you,

- Greg

Kinyua, Maureen <MKinyua@hrsd.com>
to me

Oct 13 (6 days ago)

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Maureen Njoki Kinyua, Ph.D.
HRSD Research Intern
Columbia University Postdoctoral Research Scientist

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to info

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Hello,

My name is Greg Hinds. I am writing my Master's thesis on the subject of high-solids anaerobic digestion in the United States and was hoping to include in the Appendix of my thesis the image of the Turning Earth Central Connecticut facility model accessible via the link below. Please let me know if I am granted permission to do so.

http://turningearthllc.com/turning-earth-central-connecticut/

Thank you,

- Greg Hinds

Gregory R Hinds, EI
Master of Science Student in Environmental Engineering
Department of Civil and Environmental Engineering
University of South Florida
4202 E Fowler Ave ENC 118
Tampa, FL 33620 USA

Andrew C. Kessler <andrew.kessler@turningearthllc.com>
to Amy, me

1:41 PM (8 minutes ago)

Great. Authorization per the below granted. Pls send up final thesis. Thank you!

Sent from my iPhone

...

Andrew C. Kessler
(917) 251-5662 cell

On Oct 19, 2015, at 12:05 PM, Gregory Hinds <ghinds@mail.usf.edu> wrote:

> Yes, this is acceptable to me. I will make the changes now.
>
> Thank you Andrew!
>
> On Mon, Oct 19, 2015 at 10:54 AM, Andrew C. Kessler <andrew.kessler@turningearthllc.com> wrote:
>
> > Greg: We would ask that citation read as follows:
> >
> > The caption under the figures in the Appendix should read "**Figure C.6:** Turning Earth Central Connecticut's Three-Phase Integrated High Solids Dry Fermentation In-Vessel Composting Facility Model, Southington, CT (from Turning Earth, 2014)."
> >
> > The reference that the caption is referring to should be cited as follows:
> >
> > "Turning Earth (Turning Earth, LLC.). 2014. Turning Earth Central Connecticut. http://turningearthllc.com/turning-earth-central-connecticut/."
> >
> > Please confirm this is acceptable to you. Upon receipt of that email, I will respond with final authorization from TE.

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Hello Whitney,

I am in the final stages of preparing my thesis for submission and I would like to request permission from BIOFerm to use several images from the two attached pdf's (accessed via the BIOFerm website) in the Appendix.

Thank you,

- Greg Hinds

Gregory R Hinds, EI
Master of Science Student in Environmental Engineering
Department of Civil and Environmental Engineering
University of South Florida
4202 E Fowler Ave ENC 118
Tampa, FL 33620 USA

Whitney Beadle
to me

Oct 9 (10 days ago)

Hi Greg,
Absolutely—not a problem and thanks for asking.
Regards,
Whitney

Whitney Beadle
Marketing Communications
beaw@BIOFermEnergy.com
www.BIOFermEnergy.com

BIOFERM™ ENERGY SYSTEMS
VIESMANN Group

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Gregory Hinds <ghinds@mail.usf.edu>
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Hello Norma,

I am in the final stages of preparing my thesis for submission and I would like to request permission from OWS to use several images from the attached pdf in the Appendix. I am including process diagrams, models, etc. from several vendors of high solids AD/dry fermentation technologies in the United States.

...

Thank you,

- Greg Hinds

Gregory R Hinds, EI
Master of Science Student in Environmental Engineering
Department of Civil and Environmental Engineering
University of South Florida

Norma McDonald
to me

Oct 12 (7 days ago)

Gregory,

Yes, you have my permission.

Regards,
Norma

Norma McDonald
North American Sales Manager

OWS, Inc.
7155 Five Mile Road
Cincinnati, OH 45230

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andyhoover@ecocorp.com <andyhoover@ecocorp.com>
to me

Oct 15 (4 days ago)

Greg, which diagram exactly was it that you wanted to include? We are fine with you using our process as a source as long as you reference Ecocorp as the source of it. Also, we may have more information within our engineering drawings that you can use depending on the level of detail that you are trying to achieve.

Andy Hoover
Vice President, Ecocorp

Hello,

My name is Greg Hinds. I am writing my Master's thesis on the subject of high-solids anaerobic digestion in the United States and was hoping to include in the Appendix of my thesis the image of the EcoCorp HSAD process diagram. Please let me know if I am granted permission to do so.

Thank you,

- Greg Hinds

Gregory R Hinds, EI
Master of Science Student in Environmental Engineering
Department of Civil and Environmental Engineering
University of South Florida

ABOUT THE AUTHOR

Greg Hinds grew up in Plumas County in the northern Sierra Nevada Mountains of California and completed his B.S. degree in civil engineering at California State University, Chico with concentrations in waste management engineering and water resources engineering. As a graduate student, his primary concentrations were in organic solid waste management and sustainable rural community development. Greg Hinds believes that this is the most exciting and defining era of human history and hopes to contribute to improving sustainability and preserving the environment in any way that he can.