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Mesophilic Anaerobic Co-Digestion of Municipal Wastewater Sludge and Un-dewatered Grease Trap Waste

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Dedication

To my parents Recep Kadri Yalcinkaya and Zubeyde Yalcinkaya,

and my wife Arzu Yalcinkaya.

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Mesophilic Anaerobic Co-digestion of Municipal Wastewater Sludge and Grease Trap Waste

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Fat, oil, and grease residues, food particles, solids and some kitchen wastewaters are collected in grease traps which are separate from the municipal wastewater stream. Grease traps are emptied periodically and grease trap waste (GTW) is hauled for treatment. This dissertation focuses on anaerobic co-digestion of un-dewatered (raw) GTW with municipal wastewater treatment sludge (MWS) at wastewater treatment plants. In particular, this research focuses on the biochemical methane potential of un-dewatered GTW as well as the stability and performance of anaerobic co-digestion of MWS and un-dewatered GTW.

A set of modified biochemical methane potential tests was performed to determine the methane potential of un-dewatered GTW under mesophilic conditions (35 ^{0}C). Methane potential of un-dewatered GTW in this study was 606 mL CH_4/g VS_{added} which is less than previously reported methane potentials of 845 – 1050 mL CH_4/g VS_{added} for concentrated/dewatered GTW. However, the methane potential of un-dewatered GTW (606 mL CH_4/g VS_{added}) was more than two times greater than the 223 mL CH_4/g VS_{added} reported for MWS digestion alone.

A comprehensive study was performed to determine the stability and performance of anaerobic co-digestion of MWS with un-dewatered GTW as a function of increasing

GTW feed ratios. The performance of two semi-continuously fed anaerobic digesters at 35 ^{0}C was evaluated as a function of increasing GTW feed ratios. Anaerobic co-digestion of MWS with un-dewatered GTW at a 46% GTW feed ratio (on a volatile solids basis) resulted in a 67% increase in methane production and a 26% increase in volatile solids reduction compared to anaerobic digestion of MWS alone. On the other hand, anaerobic co-digestion of un-dewatered GTW resulted in a higher inhibition threshold (46% on VS basis) than that of dewatered GTW. These results indicate that using un-dewatered GTW instead of dewatered GTW can reduce the inhibition risk of anaerobic co-digestion of MWS and GTW.

Recovery of the anaerobic digesters following upset conditions was also evaluated and semi-continuous feed of digester effluent into upset digesters yielded of the biogas production level of the undisrupted digestion. Finally, a mathematical model was used to describe the relationship between methane potential and GTW feed ratio on a VS basis. The results of this research can be used to predict methane production and identify suitable GTW feeding ratios for successful co-digestion of un-dewatered GTW and MWS.

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Chapter 1: Introduction

1.1 Problem Statement

Discharge of fats, oils and grease (FOG) from food service establishments (FSE) is one of the most frequent causes of blockages in municipal wastewater collection systems. The USEPA reported that 48% of sanitary sewer overflows are caused by blockages in the collection system and 47% of line blockages are caused by FOG depositions in the US (EPA, 2004). Overflows in municipal wastewater collection systems can create serious public health and environmental issues attributed to the release of pathogens, nutrients, solids and odor. Discharge of FOG into the wastewater collection systems in amounts that can cause obstruction in the publicly owned treatment works (POTWs) and the collection system is prohibited by Pretreatment Program regulations in 40 CFR 403.5(b)(3) and underscores the potential risk of line blockages (EPA, 2014). Authority to control FOG discharges from FSEs is given to local POTWs authorities. An increasing number of local POTW authorities are implementing management practices including the use of grease traps (EPA, 2007).

Fat, oil, and grease residues, solids such as food particles along with some kitchen wastewater are collected in grease traps separately from the municipal wastewater stream. Grease traps are periodically emptied and the wastes are hauled for treatment. Grease trap residues often are called grease trap waste (GTW) and constitute a new kind of waste stream.

There is a lack of direction by local authorities for management plans on how to dispose or utilize GTW. GTW may be disposed at landfills, or rendered into WWTPs and digested anaerobically, or used for composting, incineration and biodiesel production.

Anaerobic co-digestion (AcoD) of municipal wastewater sludge from primary and secondary sedimentation tanks (MWS) with GTW is an attractive option for resource recovery because of the high methane potential of GTW. Anaerobic co-digestion of GTW at favorable mixture ratios has been reported to increase biogas production, methane content and volatile solids (VS) reduction (Davidsson et al., 2008; Kabouris et al., 2007; Kabouris et al., 2008; Lansing et al., 2010a; Li et al., 2011; Luostarinen et al., 2009; Pastor et al., 2013; Zhu et al., 2011). However, digestion of GTW alone or AcoD of GTW at high GTW feed ratios can cause inhibition and even failure of the anaerobic digestion process (Kabouris et al., 2008; Lansing et al., 2010a; Li et al., 2011; Luostarinen et al., 2009; Zhu et al., 2011). Anaerobic co-digestion of MWS and GTW at favorable mixture ratios can dilute toxic or inhibitory substances and balance critical anaerobic digestion process parameters, such as carbon:nitrogen ratio, alkalinity, ammonia nitrogen, and volatile acids to yield a more stable digestion process (Mata-Alvarez et al., 2011).

Performance data observed in previous studies indicate that AcoD of MWS with GTW is an attractive approach to integrate GTW into resource recovery and increase methane production at existing anaerobic digestion facilities. However, most of the previous studies have focused on concentrated/dewatered GTW. The effect that co-digestion of MWS with un-dewatered GTW has on methane production and system inhibition have not been established. Inhibition of anaerobic digestion systems that have been fed wastes composed of long chain fatty acids (LCFAs), has been reported previously (Angelidaki & Ahring, 1992). Although high concentrations/loading rates of LCFAs have been identified as a source of inhibition in AcoD of MWS with GTW, no research has been conducted with un-dewatered GTW to compare inhibiting GTW loading rates.

Determining the inhibitory and optimum GTW feed ratios is crucial to establish a stable and effective anaerobic digestion process. Unsuitable GTW feed ratios can cause process disturbance and a significant reduction in methane production. A more accurate description of the relationship between GTW feed ratio and AcoD process performance is needed. As of yet, only a limited number of studies have applied mathematical models to describe time dependent cumulative methane potential from anaerobic co-digestion of MWS and GTW and to estimate kinetic parameters such as ultimate cumulative methane production and the first order reaction rate constant (hydrolysis rate) (Donoso-Bravo & Fdz-Polanco, 2013; Kabouris et al., 2007). Non-linear regression of the modified Gompertz equation has been successfully utilized to reproduce the time-dependent methane production from AcoD of MWS and restaurant oil receptacle waste (Li et al., 2011). However, there is no published study that provides a mathematical relationship between GTW feed ratio and methane potential for anaerobic co-digestion systems.

1.2 Research Objectives

The primary objective of this research was to investigate the effect that blending un-dewatered grease trap waste with municipal wastewater sludge in anaerobic co-digestion reactors has on methane potential and system inhibition.

Specific objectives of this research included:

1. Determining the biochemical methane potentials of un-dewatered GTW, MWS and mixtures of un-dewatered GTW and MWS at various ratios.

2. Evaluating the effects of anaerobic co-digestion of MWS and un-dewatered GTW on (i) biogas production, (ii) methane content, and (iii) volatile solids reduction.
3. Examining the stability of anaerobic co-digestion of MWS and un-dewatered GTW at increasing GTW feed ratios under semi-continuous feed and mesophilic conditions (35 ^{0}C).

1.3 SCOPE

The research was conducted in two major phases as follows:

- Phase I: Batch Experiments. The biochemical methane potentials of un-dewatered GTW, MWS and mixtures of un-dewatered GTW and MWS were determined through a series of biochemical methane potential (BMP) tests. The standard BMP test procedure was modified to periodically monitor the process parameters. A mathematical equation describing the relationship between GTW feed ratio (on a VS basis) and methane potential was fit to the experimental data from the current study and previous BMP studies. The kinetic and performance parameter estimates that provided the best fit to the experimental data were used to assist in the interpretation of the batch experimental results.
- Phase II: Semi-continuous Feed Reactor Experiments. The performance of mesophilic (35 ^{0}C) AcoD of MWS and un-dewatered GTW was investigated in two semi-continuous feed anaerobic co-digestion reactor systems. Methane potentials and VS reductions were determined at increasing GTW feed ratios and compared with previously reported performance data for AcoD of MWS and concentrated/dewatered GTW.

Optimum and inhibiting GTW feed ratios on a VS basis were determined. The risk of inhibition of the AcoD process using un-dewatered GTW and dewatered/concentrated GTW as co-substrates was assessed. The effects of digester effluent feeding on the recovery of digesters following upset conditions was studied and compared with previously reported recovery methods.

1.4 Organization

The dissertation includes 5 chapters. Chapter 1 presents the problem statement, research objectives and organization of the thesis.

Chapter 2 provides a comprehensive review on GTW management through AcoD. Fat, oil, and grease deposition mechanisms in wastewater collection lines, performance and upgrades of grease traps, characteristics of GTW, and the biochemistry of AcoD processes were analyzed. In addition, AcoD of grease trap waste including process performance, effects of various reactor configurations and pre-treatment methods, recovery of the digester following upset, and mathematical modeling were examined. Future research needs in GTW management, as well as research needs addressed in this study, were evaluated.

Chapter 3 and 4 describe and examine experimental results. Chapter 3 presents an investigation of the biochemical methane potentials of un-dewatered GTW, MWS and mixtures of un-dewatered GTW and MWS at various ratios through a series of biochemical methane potential (BMP) tests. Chapter 4 focuses on investigating the performance and stability of mesophilic (35 ^{0}C) AcoD of MWS and un-dewatered GTW from a set of semi-continuous feed reactor experiments.

Chapter 5 includes conclusions based on the research findings in this dissertation and provides recommendations for future research.

Chapter 2: Literature Review

Discharge of fats, oils and grease from food service establishments was reported to cause approximately 25% of all municipal wastewater collection system blockages (EPA, 2004). An increasing number of local authorities are requiring food service establishments to install grease traps to reduce the discharge of fat, oil and grease into wastewater collection systems (EPA, 2007). Grease trap residues often are called grease trap waste (GTW) and constitute a new kind of waste stream. There is a lack of direction in grease trap waste management plans of local authorities regarding disposal or utilization. Anaerobic co-digestion of grease trap waste with municipal wastewater sludge at wastewater treatment facilities is becoming an attractive alternative for grease trap waste management because of the high methane potential of GTW. An increase in methane production and reduction in volatile solids concentration were reported in laboratory studies and full-scale anaerobic co-digestion applications. However, inhibition and possible failure of anaerobic digesters also were reported, potentially as a result of the high long chain fatty acid concentrations in grease trap waste. This summary presents a review of the current state and perspectives of grease trap waste management using anaerobic co-digestion. Fat, oil, and grease deposition mechanisms in wastewater collection lines, performance and upgrades of grease traps, anaerobic co-digestion of grease trap waste including process performance, effects of various reactor configurations and pre-treatment methods, recovery from digester upsets, and mathematical modeling were examined. Future research needs also were evaluated to improve grease trap waste management strategies, understanding of fat, oil, and grease deposition phenomena, efficiency of grease traps, and performance of co-digestion of grease trap wastes.

2.1 INTRODUCTION

Anaerobic digestion (AD) is a biological treatment process commonly used for the treatment of municipal wastewater sludge (MWS). Organic materials are degraded and stabilized by bioconversion into biogas consisting primarily of methane and carbon dioxide, and biomass in the absence of oxygen. Relatively low energy requirements, as well as reductions in residual sludge, pathogens, odor as well as global warming impacts are the advantages of anaerobic digestion (Dahab & R.Y., 2002; Whiting & Azapagic, 2014; Wright et al., 2003). In addition, the end products of biogas and digested organic solids can be used as a renewable energy source and in compost, respectively (Belyaeva & Haynes, 2010; Ullrich & M. W., 1950). Anaerobic co-digestion (AcoD) is simultaneous AD of multiple substrates. Anaerobic co-digestion of different substrates often increases the methane content of the biogas produced and yields more biogas production than the sum of the biogas produced when digested separately. In addition, co-digestion of toxic/inhibitory substances such as grease trap waste (GTW) is possible because of microbial activity enhancement and dilution of toxic substances, while digestion of such materials alone can cause inhibition or failure of the entire digestion system (Davidsson et al., 2008; Li et al., 2011; Martin-Gonzalez et al., 2010; Zhu et al., 2011). Therefore, co digestion at favorable sludge constituent ratios can balance critical process parameters in anaerobic digestion such as carbon:nitrogen (C:N) ratio, alkalinity, ammonia and volatile acids and yield a more stable digestion (Mata-Alvarez et al., 2011).

Anaerobic co-digestion of concentrated/dewatered grease trap wastes (GTW) has been reported to increase biogas production, methane content of biogas produced, and volatile solids (VS) reduction. Early studies focused on biodegradability and biogas production potential of concentrated/dewatered GTW (Davidsson et al., 2007; Kabouris et al., 2007; Luostarinen et al., 2009; Suto et al., 2006). More recently, determination of inhibiting GTW feed ratios, mathematical modeling of biogas production and effects of pretreatment, temperature, feeding and reactor configuration on concentrated/dewatered GTW co-digestion were evaluated (Donoso-Bravo & Fdz-Polanco, 2013; Li et al., 2013; Li et al., 2011; Wang et al., 2013). Experiments were conducted mostly in batch, semi-continuous feed and 2-phase reactors (Creamer et al., 2010; Lansing et al., 2010a; Li et al., 2014; Noutsopoulos et al., 2013; Zhu et al., 2011). On the other hand, inhibition and failure of the anaerobic digestion process were reported at high concentrated/dewatered GTW feeding ratios (Davidsson et al., 2008; Girault et al., 2012; Luostarinen et al., 2009; Martinez et al., 2011; Noutsopoulos et al., 2013; Razaviarani et al., 2013; Silvestre et al., 2011; Wan et al., 2011; Wang et al., 2013). Accumulation of long chain fatty acids (LCFAs) resulting from feeding high concentrations/loading rates of GTW were identified as the source of inhibition in anaerobic co-digestion of GTW (Luste et al., 2009; Suto et al., 2006). A few successful full-scale GTW co-digestion implementations were reported; however published data from full-scale implementations are scarce (Bailey, 2007; Ripley et al., 2014). Scarcity of full-scale examples, serious consequences of process failure and lack of information on anaerobic digestion recovery are still major operational concerns. Modeling may help to predict GTW feeding limits that would not cause process inhibition or failure; however, more studies are required to improve available models.

This comprehensive review focuses on the current state and perspectives of GTW management through AcoD. The results of experimental studies of batch and semi-continuous feed reactors were examined and compared in terms of methane potentials, inhibiting GTW feed ratios, and reactor configurations. The effects of various pretreatment methods on process performance, recovery of digesters following upset, and mathematical modeling of the process and methane production were evaluated. The review also examines performance and upgrades of grease traps, and management strategies. The results are discussed throughout the document to evaluate current challenges and future research needs of AcoD of GTW. The research needs which are intended to address in this dissertation are emphasized throughout the literature review.

2.2 THE BASES OF GREASE TRAP WASTE STREAM

2.2.1 Regulations and environmental issues

Discharge of fats, oils and grease (FOG) from food service establishments (FSE) was reported as the most frequent cause of blockage of municipal wastewater collection systems. The USEPA reported that "48% of sanitary sewer overflows caused line blockages and 47% of line blockages are caused by FOG depositions in the US" (EPA, 2004). Sanitary sewer overflows can create serious public health and environmental problems by the release of pathogenic organisms, nutrients, solids and odor. Discharge of FOG into the sewer systems in amounts that can cause obstruction in the publicly owed treatment works (POTWs) and the sewer system is prohibited by Pretreatment Program regulations in 40 CFR 403.5(b)(3) and underscores the potential risk of line blockages (EPA, 2014). Authority to control FOG discharges from FSE is given to local POTWs authorities. An increasing number of local POTW authorities are implementing management practices including the use of grease traps (EPA, 2007).

FOG deposits in the sewer lines were reported to be calcium-based fatty acid salts (soaps), mainly consisting of saturated fatty acids (Keener et al., 2008). He at al. (2011) simulated the formation of FOG deposits in sewer lines under laboratory conditions.

Chemical properties of FOG deposits formed in the lab and FOG deposit samples from sanitary sewer lines were compared in order to evaluate the accuracy of the simulation to the actual FOG deposition phenomenon in sewer lines. Two mechanisms were suggested to explain FOG deposit formation in sewer lines: saponification reaction between free fatty acids and metal ions such as calcium, and aggregation of excess calcium, fatty acids, and debris in the wastewater as a result of DLVO type process (compression of double layer) (He et al., 2011; Iasmin et al., 2014). Although calcium was identified as the dominant metal in FOG deposits, a correlation between calcium concentration in FOG deposits and water hardness was not confirmed (Keener et al., 2008). On the other hand, He et al. (2011) and Williams et al. (2012) reported a positive relationship between water hardness and calcium concentrations in FOG deposits. Corrosion was reported as a source of excessive calcium in concrete sewers that react with free fatty acids to form calcium salts of fatty acids (He et al., 2013; Iasmin et al., 2014). In addition, the possibility of calcium production in sewer lines caused by microbial activity was mentioned (Williams et al., 2012); however, experimental evidence was not provided to validate this hypothesis. The presence of oil was suggested to be required as a transport medium for un-reacted free fatty acids to interact with calcium for the formation of FOG deposits through saponification reactions and as a minor source of free fatty acids (He et al., 2013). Different types of free fatty acids yielded soaps with different adhesive qualities (He et al., 2013). Moreover, flow properties of soaps were limited as a result of adhesive characteristics when they contact with pipe walls (Iasmin et al., 2014). However, more research on the rheology of FOG deposits was suggested to determine adhesive characteristics of FOG deposits.

Characterization of FOG deposits in sanitary sewer lines, and formation mechanisms are scarce in recent publications. Although evidence of FOG deposition in sewer lines caused by GTW discharge has been reported in numerous studies, the formation process has not been clearly defined. The suggested deposition mechanisms, such as saponification of free fatty acids and metal ions, and DLVO type aggregation processes need to be supported with further research. The relationship between hardness in wastewater and calcium concentration in FOG deposits is ambiguous and needs to be clarified. More research is needed to determine the effects of different types of free fatty acids on the saponification reactions. The factors, other than corrosion, that are causing the excess calcium in FOG deposits require identification. Research is required to confirm the contribution of microbial activity to calcium production. The role of oil in the formation of FOG deposits is not very well known. Transformation of fatty acids from unsaturated to saturated forms in the sewer lines is largely unknown. Research on rheology of FOG deposits should provide a better understanding of the flow conditions and adhesive characteristics of the deposits. Effective control of FOG deposits in sanitary sewer lines may be possible based on a better understanding of these mechanisms.

2.2.2 Grease traps and grease trap waste characteristics

Grease traps usually are divided into two major types; internal grease traps located in FSEs and external grease traps located underground outside FSEs. Internal grease traps are relatively small in size and consist of tanks with capacities less than 10 gallons (37.85 L) and shorter hydraulic retention times (HRT), while conventional external grease traps are large tanks with capacities up to 5000 gallons (18927 L) and sufficient retention time to physically separate FOG and solids such as food particles from kitchen wastewaters (EPA, 2004). The term “grease trap” was used to define conventional 2-chamber external grease traps throughout this study. Grease traps do not only separate FOG and solids, but also equalize the highly fluctuating flow rates of FSEs. Sufficient volume, proper retention time, and low turbulence are required in grease traps for effective separation of FOG (EPA, 2004). Different HRT criteria were recommended for sizing grease traps by different authorities. For example; minimum 20 minutes HRT at maximum flow rates was suggested by Hong Kong Environmental Protection Department (HKEPD, 1995). Chu and Ng (2000) and Tchobanoglous et al. (2003) recommended 30 minutes minimum HRT. Fat, oil, and grease removal slightly increased at longer HRTs, while shorter HRTs resulted in drastic decrease in FOG removal (Chu & Ng, 2000). Other parameters such as number of seats at a FSE and open hours are also used in grease trap sizing (Ducoste et al., 2008). Grease traps must be cleaned periodically in order to maintain optimum FOG removal performance. Clean-up frequency depends on the capacity of grease traps and accumulation rate of FOG and solids in traps. For example; the minimum required cleaning frequency is 90 days in San Antonio, TX (City of San Antonio). On the other hand, the City of Austin, TX requires more frequent cleaning schedules, if FOG and solids accumulated in the second/final chamber of the trap is 50% or more of the wetted height of the trap before 90 days (City of Austin, 2006).

Physico-chemical characteristics of four full-scale grease traps with capacities of 1000, 2000, 2000 and 5400 gallons were monitored daily during an 8-week study by Wong et al. (2007). Average effluent pH of 4 to 5.23 and temperature of 28.2 to 39.9 ^{0}C were reported, respectively (Wong et al., 2007). Aziz et al. (2012) investigated characteristics of conventional grease traps in the field. Hydraulic residence times at average flow rates were 8.8 times greater (average of 24 grease traps) than HRTs at maximum flow rates based on the flow rate data collected at 24 FSE at every 15 minutes for over a 24-hour period (Aziz et al., 2012). The significant difference between HRT at maximum flow rate and HRT at minimum flow rate was due to highly intermittent flow rates. Aziz et al. (2012) also reported that in the 1st chamber of a buffet-style FSE's grease trap, the average pH was 6.35, dissolved oxygen was 0.47 mg/L and temperature was 43 ^{0}C, while the pH=4.57, dissolved oxygen = 0.43 mg/L and temperature = 39 ^{0}C in the 2nd chamber. Microbial activity was not measured; however, the possibility of anaerobic microbial processes in the grease traps was suggested by the low pH and low dissolved oxygen. He et al. (2012) described physical conditions in grease traps as anaerobic, acidic, and oxidation reduction potential within the range major electron acceptors of sulfate and ferric iron. Three distinct layers were noted in a grease trap; floatable FOG layer at the top, aqueous middle layer and gravity settled sludge layer at the bottom by Suto et al. (2006).

The entire content of a grease trap, FOG and solids along with some kitchen wastewater, are often called "grease trap waste". The chemical characteristics of GTW are significantly influenced by the type of FSE, the design of the grease trap and cleaning frequency. The composition of water free GTW collected from restaurants in Guangzhou, China, where the water free GTW production rate is 20000 tons/year, was analyzed by Park et al. (2010). Grease trap waste was composed of 51.45% free fatty acids, 0.67% monoglycerides, 9.4% diglycerides, 26.6% triglycerides, 4.4% glycerol, 0.71% water, and 6.77% other components on weight basis (Park et al., 2010). Kabouris et al (2009a) reported GTW contents as 15% carbohydrate, 7% protein, and 78% fats on a VS basis. Suto et al. (2006) reported fat composition of GTW as 48.6% saturated, 15.3% polyunsaturated, and 36.1% monounsaturated, while Kabouris et al. (2009a) reported

37.9% saturated, 7.4% polyunsaturated, 39.5% monounsaturated, and 15.2% trans fat. According to Suto et al. (2006), palmitic acid, linoleic acid, and oleic acid were the major components of the GTW saturated, polyunsaturated, and monounsaturated fats, respectively, whereas Kabouris et al. (2009a) reported palmitic acid, and oleic acid were the major components of the GTW saturated and monounsaturated fats, respectively. Suto et al. (2006) reported significant variations in minimum, maximum, and average concentrations of GTW characteristics. Volatile solids concentration is one of the most variable characteristics of GTW. At haulers' transfer stations, dewatering GTW is common and high VS concentrations such as 25%, 41%, and 64% on a wet weight basis were reported by Luostarinen et al. (2009), Kabouris et al. (2009), and Liu and Buchanan et al. (2011), respectively. However, low VS concentrations such as 2%, 3%, and 3.6% were also reported by Davidsson et al. (2007), Wan et al. (2011), and Suto et al. (2006), respectively. Grease trap waste is an acidic waste stream. The highest and lowest pH reported in the literature were 4.03 and 5.6 by Kabouris et al (2007) and Luste et al. (2009), respectively. Ammonia nitrogen concentrations reported in the literature were less than the inhibiting concentrations for anaerobic digestion processes, 1.5 -3 g NH_4^+-N/L (Tchobanoglous et al., 2003). Calcium (320 ppm) and sodium (220ppm) were stated as the most abundant nutrients excluding carbon and nitrogen in GTW, while iron, nickel, copper, zinc, and selenium were available in ppb levels (Wan et al., 2011).

2.2.3 Performance and upgrades of grease traps

The performance of grease traps in terms of FOG removal efficiency was investigated by researchers. Fat, oil, and grease (4% by volume corn oil/water emulsion) removal efficiencies of an internal passive flow grease traps PFGT, internal mechanical flow grease traps MFGT, and a laboratory-scale external conventional grease trap under weak, medium, and strong emulsion strengths; 21 ^{0}C, and 38 ^{0}C influent temperatures; and maximum, and average flow rates were investigated (Gallimore et al., 2011). A camera was installed before the traps to analyze size distribution of oil droplets. Stronger emulsion production and reduction in FOG removal were reported at the higher temperature as a result of breakage of oil droplets. Increasing emulsion strength and flow rate also decreased FOG removal efficiencies. FOG removal efficiencies of 77% and 88% were observed under all conditions for external conventional grease traps, while FOG removals of PFGT and MFGT were 8% to 46%, and 2% to 48%, respectively. Higher FOG removal efficiencies of external conventional grease trap were attributed to the longer HRT of external conventional grease trap (30 min) than PFGT (30 sec) and MFGT (1 min).

Research studies on upgrading the conventional grease traps focused on either physical modifications on the grease traps or biological product addition, in order to improve FOG removal performance and meet discharge standards. Comparison of studies on grease trap performance evaluation and upgrades are summarized in Table 2.1. Installing tube settlers inside a laboratory-scale grease trap resulted in 8-10 % increase in FOG removal efficiency by increasing oil droplet coalescence and required less space than conventional grease traps at the same capacity (Chu & Ng, 2000). Chu and Ng (200) also reported that performance of COD and FOG removal in a conventional grease trap at inflow concentrations between 400 mg FOG/L and 1600 mg FOG/L at a constant HRT of 30 minutes did not show a considerable change. Therefore, the authors concluded that FOG or COD removal efficiencies of a grease trap do not depend on inflow FOG concentration. (Chu & Ng, 2000). Improvements in chemical oxygen demand (COD) and biochemical oxygen demand (BOD) removals at peak flow rates or short HRTs also were reported and attributed to “the entrapment of semi-hydrophobic COD and BOD to the oil droplets” (Chu & Ng, 2000; Jaruwan et al., 2014). Jaruwan et al. (2014) investigated the effects of filter media arrangements, flow rate and HRT on grease trap performance. Filter media was installed as three separate layers constituting the 30% of the grease trap height. The authors reported that installing 5 cm diameter of mon brick, 1 cm diameter of gravel and 3 cm diameter of mon brick at a ratio of 1:1:2 on volume basis generated the highest efficiency. Jaruwan et al. (2014) also reported that the highest grease trap performance for both filter media installed at 1:1:2 ratio and without filter installation were achieved at 10-hr HRT and 2 L/min flow rate. 87% suspended solids, 70% BOD, and 87% FOG removals were monitored in the filter media installed grease trap under these conditions, (Jaruwan et al., 2014). Installation of effluent filters on 1000, 2000 and 5400 gallons full-scale grease traps resulted in 41% to 57% total suspended solids (TSS) removal and 43% to 52% overall FOG removal; however, the results were not compared with TSS and FOG removal efficiencies from conventional grease traps (Wong et al., 2007). Nisola et al. (2009) installed cell immobilized matrices and air diffusers in order to maintain aerobic biodegradation and prevent biomass washout. Matrices were constructed by immobilizing Pseudomonas aeruginosa D2D3 on rice bran, polyurethane and ceramic beads. Experiments were conducted in a 600 L full-scale grease trap installed at a barbeque restaurant and a 6 L laboratory-scale grease trap under laboratory conditions. Fat, oil, and grease and COD removal efficiencies of both full-scale and laboratory-scale grease traps were determined with and without air diffusers and cell immobilized matrices. Based on FOG depth measurements and mass balance calculations, the authors estimated about 93.6% of FOG removal in the laboratory-scale matrix installed grease trap was attributed to biodegradation. The results observed in laboratory-scale experiments did not show a considerable difference in FOG and COD removal efficiencies between matrix installed and conventional grease traps. However, significant improvements in FOG and COD removals were reported at the full-scale grease trap after installation of cell immobilized matrices. In addition, the performance of full-scale grease trap was more stable after the installation of cell immobilized matrices (Nisola et al., 2009). The authors indicated that increase in FOG and COD removal efficiencies were caused by biodegradation although, no investigation into microbial activity was performed. Chan (2010) installed fine bubble membrane diffusers on the bottom of the 1st chamber of a 2-chamber conventional grease trap and periodically added alum and cationic polymer into the 1st chamber. They reported 93% FOG removal at the maximum flow rate and 30 min HRT (Chan, 2010). However, FOG removal reported by Chan (2010) was 13% to 16% greater than FOG removals of the conventional grease trap reported by Gallimore et al. (2011), increase in operating costs and sludge production were not taken into account. Application of increasing HRT from 20 minutes to 1 hr resulted in 12% more FOG (4% by volume corn oil/water emulsion) removal in a laboratory-scale conventional grease trap, “two-chamber tank with submerged inflow and outflow pipes”, while changes in pipe configurations yielded 5% to 9% more FOG removal than the laboratory-scale conventional grease trap (Aziz et al., 2011). Aziz et al. (2011) also reported 7% increase in FOG removal performance after removing the baffle wall. The authors reported increase in FOG removal efficiencies; however, the experiments were only performed with a 4% corn oil/water emulsion. Therefore, the effects of food solids and fluctuations in flow rate were not taken into account as the authors mentioned (Aziz et al., 2011). The effects of three different inlet configurations on FOG and solids accumulation profiles were analyzed by measuring FOG and solids thicknesses at 8 points in a two chamber full-scale GT by Aziz et al. (2012). Inlet configuration influenced solids accumulation much more than FOG accumulation. Accumulation in the 2nd chamber and channeling were the lowest with the use of distributive inlet compared to straight pipe inlet and no-inlet configurations.

Garry and Sneddon (1999) investigated the effects of enzyme addition on a GTW sample, collected from a restaurant, in a laboratory-scale study. Two types of enzymes were used. One type of enzyme included surfactants, while the other type did not. Decomposition of top FOG layer and increase in turbidity in the samples treated with enzymes were compared to untreated samples. Addition of enzymes with surfactants yielded the highest effluent FOG concentrations (Gary & Sneddon, 1999). Therefore, it can be inferred that detergents and sanitizers in FSE wastewaters also can influence FOG removal performance of grease traps by affecting emulsification characteristics. He et al. (2012) monitored the physico-chemical characteristics in two 1000-gallon, 2-chamber grease traps with and without addition of biological products for one year. He et al. (2012) determined similar or better effluent chemical characteristics in terms of COD, BOD, and FOG concentrations when treated with biological product. The authors also reported that addition of biological product did not increase the FOG concentration in the downstream unlike the common understanding of passing FOG downstream due to the addition of biological products (He et al., 2012). The effects of a blend of nutrients and five Pseudomonas and Bacillus strains on FOG deposit formation and WWTP processes were assessed comparing the characteristics of treated and untreated effluents (Tang et al., 2012), and significant reductions in COD, total nitrogen, total phosphorus, and FOG deposit formation were observed. The authors also reported increase in readily biodegradable COD in the grease trap effluent with biological product addition which can help to increase biological phosphorus removal at WWTPs. In addition, increase in the saturated fatty acids ratio with biological product addition was stated. However, the effluent concentrations of COD, total nitrogen, total phosphorus had an increasing trend after the first addition of biological product. Therefore, more frequent grease trap cleaning schedule was suggested in order to maintain the optimum performance (Tang et al., 2012). The authors reported improvements in effluent characteristics by evaluating treated and untreated effluents, characteristics of influent and effluent were not compared. Therefore, the effects of flow rate and fluctuations in concentration were not taken into account in the results.

Very little information on the characteristics of GTW is known. A comprehensive physico-chemical characterization of GTW is needed. Availabilities of micro-nutrients, macro-nutrients, and trace metals in GTW are needed to be investigated for effective grease trap upgrades. A classification of FSEs such as meat processing facilities, barbeque restaurants, buffet style FSEs, etc. may reduce the deviations in GTW characteristics. Installation of tube settlers, filter media or effluent filters in conventional grease traps may help to meet discharge standards during the peak hours of operation of restaurants when more kitchen wastewater is discharged. In addition, FOG and solids can be collected separately in tube settler installed grease traps. However, operation of modified grease traps may require experienced staff and more frequent maintenance. Furthermore, it is important to determine oil droplet size distribution of GTWs and the ability to coalescence these droplets in order to design more effective grease traps. Cell immobilized matrix installed grease traps showed substantial increase in FOG and COD removal efficiencies and provided more stable removal under fluctuating inflows, clean up and seeding frequencies are must be evaluated. Addition of biological products, such as lipase, can help to hydrolyze FOG components in a short time. However, more research is required to investigate simple ways of biomass immobilization in grease traps. Otherwise, addition of biological product may even yield higher effluent FOG concentrations caused by the breakdown of oil droplets into smaller sizes. A revision of the current regulations on the addition of biological products in grease traps may be required, if successful studies increase. Few studies were found in the literature that addressed upgrading grease traps. Additional research on modified grease traps with simple operating procedures may improve the efficiency of grease traps and provides pretreatment for kitchen wastewaters.

Table 2.1: Comparison of studies on grease trap modifications and observed effects.

Reference	Source of GTW	Experimental setup	Modification/Test	Effects
(Gary & Sneddon, 1999)	GTW from the GT of a fried chicken restaurant.	Lab-scale batch reactors	Two types of enzymes were used: EBR-D-30, and EBR-D-30F with surfactants.	Addition of either enzyme led to decomposition of top FOG layer and increase in turbidity. Addition of the enzyme with surfactants doubled the effluent FOG concentration.
(Chu & Ng, 2000)	Peanut oil/water emulsion	187 +105 +80 L/chamber, 3-chamber, lab-scale GT, HRT: 0 - 70 min	A mixer was installed in the 1st chamber to prepare the sample mixture. A tube settler was installed in the 2nd chamber.	8-10 % increase in FOG removal efficiency. Significant increase in COD removal efficiency at HRTs shorter than 30 min.
			Inflow FOG concentration was increased from 400 mg/L to 1600 mg/L at a constant HRT of 30 min without any modification	No considerable change was observed at either FOG or COD removal efficiencies.
(Wong et al., 2007)	GTW from 4 restaurants in Cookeville, TN	3785 L, 3-chamber, full-scale GT, HRT_{ave}: 6.7 h	An effluent filter was installed	56 ± 14% TSS removal, 52 ± 16% FOG removal
		7571 L, 2-chamber, full-scale GT, HRT_{ave}: 17.5 h		47 ± 14% TSS removal, 47 ± 8% FOG removal
		7571 L, 2-chamber, full-scale GT, HRT_{ave}: 17.5 h		57 ± 11% TSS removal, 50 ± 4% FOG removal
		20441 L, no compartment, full-scale, HRT_{ave}: 19.1 h		41 ± 9%TSS removal, 43 ± 13% FOG removal

Table 2.1: Continue.

Reference	Source of GTW	Experimental setup	Modification/Test	Effects
(Nisola et al., 2009))	GTW from a barbeque restaurant	600 L, 3-chamber, full-scale GT, HRT: 30-109 min	Cell immobilized matrices and air diffusers were installed.	Without matrices: 74.6 ± 27.13% FOG removal, 60.4 ± 31.26% COD removal at average influent concentrations of 463.4 ± 296.86 mg FOG/L and 3307.3 ± 2120.99 mg COD/L
				With Matrices: 92.7 ± 9.06% FOG removal, 85.9 ± 11.99% COD removal at average influent concentrations of 1044.8 ± 537.27 mg FOG/L and 5597.9 ± 3061.65 mg COD/L
	Soybean oil/water emulsion	2 L/chamber, 3-chambers, lab-scale GT, HRT: 30 min		With and without matrices: 90-99.9% FOG removal and >85% COD removal at 200-5000 mg/L influent FOG concentrations.
(Chan, 2010)	GTW from a Chinese fast food restaurant	1000 L/chamber, 2-chamber, full-scale GT, HRT: 30-40 min	Fine bubble membrane diffusers were installed in the 1st chamber. Alum and cationic polymer were periodically fed into the 1st chamber	92.7% ± 1.6% FOG removal, 48.6 ± 0.6% COD removal
(Aziz et al., 2011)	4% by volume corn oil/water emulsion	40 L, 2-chamber, lab-scale GT, HRT: 20 min	no modification	78% FOG removal
			1 hr HRT	80% FOG removal
			Short inlet, no baffle	85% FOG removal
			Flared pipe	83% FOG removal
			Inverted Tee inlet, no baffle	69% FOG removal
			Inverted Tee inlet, dual pipe baffle	87% FOG removal

Table 2.1: Continue.

Reference	Source of GTW	Experimental setup	Modification/Test	Effects
(Gallimore et al., 2011)	4% by volume corn oil/water emulsion	19 L, internal PFGT, full-scale	21 ^{0}C, HRT: 30 sec, Emulsion strength: weak-medium-strong	38 ± 7% - 17 ± 3% - 8 ± 0.3% FOG removal, respectively.
			38 ^{0}C, HRT: 30 sec, Emulsion strength: weak-strong	29 ± 4% - 14 ± 2% FOG removal, respectively.
			21 ^{0}C, HRT: 1 min, Emulsion strength: weak-medium-strong	82 ± 2% - 46 ± 0.2% - 38 ± 2% FOG removal, respectively.
			38 ^{0}C, HRT: 1 min, Emulsion strength: weak-strong	69 ± 7% - 34 ± 2% FOG removal, respectively.
		95 L, internal MFGT, full-scale	21 ^{0}C, HRT: 1 min, Emulsion strength: weak	2 ± 3% FOG removal
			38 ^{0}C, HRT: 1 min, Emulsion strength: weak	2 ± 2% FOG removal
			21 ^{0}C, HRT: 2 min, Emulsion strength: weak-medium	48 ± 1% - 28 ± 0.2% FOG removal, respectively.
			38 ^{0}C, HRT: 2 min, Emulsion strength: weak	35 ± 9% FOG removal
		102 L, 3-chamber, lab-scale conventional GT	21 ^{0}C, HRT: 30 min, Emulsion strength: weak-strong	80 ± 1% - 77 ± 3% FOG removal, respectively.
			21 ^{0}C, HRT: 1 hr, Emulsion strength: weak-strong	88 ± 2% - 84 ± 0.1% FOG removal, respectively.

Table 2.1: Continue.

Reference	Source of GTW	Experimental setup	Modification/Test	Effects
(Aziz et al., 2012)	GTW from a university dining hall.	2-chamber, full-scale GT	Flowrate of 24 FSEs at 15 min intervals over a 24-hour period were monitored.	Longer HRTs at average flowrate and highly intermittent inflows.
			pH-dissolved oxygen-temperature were monitored.	6.35 - 0.47 mg/L - 43 ^{0}C in the 1st chamber.4.57 - 0.43 mg/L - 39 ^{0}C in the 2nd chamber
			FOG and solids accumulation profiles were analyzed at 8 points in the grease trap.	Using distributive inlet led to the least channeling and accumulation in the 2nd chamber compared to straight pipe inlet and no-inlet configurations.
(He et al., 2012)	GTW from a restaurant and retirement center kitchen	3785 L, 2-chamber GTs at both sources.	Biological product (Bacillus strain) addition	Physical characteristics of GT without biological product addition: anaerobic, acidic, and oxidation reduction potential within the range major electron acceptors of sulfate and ferric iron. Similar or slightly better effluent COD, BOD, and FOG concentrations when treated with biological product.
(Tang et al., 2012)	GTW from a university dining hall	7500 L, full-scale GT	No biological product addition	Effluent: 2570 ± 500 mg COD/L, 145 ± 13 mg TN/L, 115 ± 15 mg TP/L, 413 ± 18 mg FA/L
			A blend of nutrients and 5 Pseudomonas and Bacillus strains	Effluent: 1570 ± 470 mg COD/L, 97 ± 18 mg TN/L, 50 ± 22 mg TP/L, 169 ± 49 mg FA/L

Table 2.1: Continue.

Reference	**Source of GTW**	**Experimental setup**	**Modification/Test**	**Effects**
(Jaruwan et al., 2014)	Domestic wastewater from household diswashing activities	60 L, no-compartment, lab-scale GT, HRT:10 hr	5 cm diameter of Mon brick: 1 cm diameter of gravel: 3 cm diameter of Mon brick at a ratio of 1:1:2 on volume basis were installed as filter media.	87% SS, 70% BOD, 87% FOG removal.

Abbreviations: GTW, grease trap waste; GT, grease trap; FOG, fat, oil, and grease; HRT, hydraulic residence time; FA, fatty acids; COD, chemical oxygen demand; TSS, total suspended solids; PFGT, passive flow grease trap; MFGT, mechanical flow grease trap; FSE, food service establishment; BOD, biochemical oxygen demand; TN, total nitrogen; TP, total phosphorus; SS, suspended solids.

2.2.4 Management options

Collected GTW may be disposed at landfills, or rendered into WWTPs or composted, incinerated, biodiesel production or anaerobic co digestion. Landfilling GTW (organic wastes in general) are restricted by environmental legislations and plans, such as The Austin Resource Recovery Master Plan on zero waste going to landfills policy and The European Union Council Directive 1999/31/EC on the landfill of waste, because of the potential of aggravating the effects of climate change (City of Austin, 2011; EU, 1999). Rendering GTW into municipal WWTPs can be inhibitory to microbial growth by reducing oxygen transfer rates. In addition GTW may cause foaming, increase in the growth of filamentous bacteria and washout of biomass resulting from flotation (Chipasa & Mędrzycka, 2006). Composting GTW with materials originated from both domestic dwellings and municipal parks, gardens such as tree wood and bark, prunings from young trees and shrubs, dead and green leaves, and grass clippings and reserves (municipal green waste) yielded more extractable phosphorus and electrical conductivity than composting municipal green waste alone and municipal green waste and poultry manure mixture that can damage plant growth (Belyaeva & Haynes, 2010). Belyaeva and Haynes (2010) also noted that GTW added to compost piles reached the highest temperature (maximum 78 ^{0}C) and increase in temperature was the most rapid. Although land application of GTW may provide organic carbon and reduces nitrogen leaching, spray application of GTW on vegetation and land can cover the plants and land surface with grease and reduce yields and clog pores in the soil (Coker, 2006a; Rashid & Voroney, 2004). Wastewater transformations during anaerobic co-digestion of GTW and swine manure were evaluated in terms of fertilizer value by Lansing et al. (2010b). Anaerobic co-digestion of GTW with swine manure reduced organic content, fecal coliform population and grease content. However, the Nitrogen to Phosphorus (N:P) ratio of the GTW-manure mixture decreased because total nitrogen reduction was more than total phosphorus reduction after anaerobic co-digestion. Therefore, the authors did not suggest land applications of co-digested GTW for phosphorus saturated fields (Lansing et al., 2010b). Diluting anaerobically co-digested GTW in composting process may increase N:P ratio as suggested by Belyaeva and Haynes (2010). Land application of GTW can be a relatively low cost disposal alternative depending on land use. Grease trap waste may be an economic feedstock alternative for biodiesel production; however, the high moisture and free fatty acids contents make transesterification process more complex. Free fatty acids form soaps and produce water during transesterification by alkaline catalyst that prevent separation of glycerin and inhibit transesterification process (Canakci, 2007). A 2-step transesterification process successfully converted GTW into biodiesel in pilot-scale reactors by Canakci and Van Gerpen (2003). Acid catalyst was added before performing alkali catalyst for the transesterification of brown grease that had 40% free fatty acids (Canakci & Van Gerpen, 2003). Lopez, et al. (2014) proposed an integrated disposal method for GTW that included both biodiesel production and anaerobic co-digestion. Grease trap waste was first preheated to 70 ^{0}C overnight to liquefy solid fats. The water portion of GTW was transferred into a column and maintained at 74 ^{0}C for one day to separate more solids and FOG from water portion. The water portion was anaerobically digested in two biochar filled packed bed column reactors in series, after hydrolyzation and pH buffering were performed in another reactor. Methane gas production of 0.35 m^3 methane per kg of COD reduced at a hydraulic retention time of 1 day under mesophilic conditions was achieved. Theoretical energy production from only anaerobic digestion of GTW and co-production of biodiesel and methane from GTW were estimated to be similar in quantity (Lopez et al., 2014). However, energy input to each method was not taken into account, therefore reduction of net energy production from co-production of biodiesel and methane yield from GTW could not be determined. Anaerobic co-digestion of GTW at the existing digestion facilities can also be an attractive disposal method because of the high methane production potential. The effects of anaerobic co-digestion of GTW will be discussed in more detail in the next section.

Anaerobic co-digested GTW may increase pH up to neutral levels and provide a more stable composting process or better quality liquid fertilizer for land applications than composting or land application of GTW directly. However; the low N:P levels also may be aggravated. Therefore, fertilizer value and compost quality of GTW before and after AcoD must be established to develop proper management strategies for GTW and products after anaerobic co-digestion of GTW. A comprehensive comparison of current treatment/disposal methods has not been evaluated to conclude the most feasible disposable method for GTW.

Three management strategies for treatment of GTW were developed based on the size of FSEs by Stoll and Gupta (1997). Manual cleaning of grease traps and collection of GTW in bins were proposed for small FSEs with internal (under counter) grease traps or external small conventional grease traps that collection by a tanker is not possible. Stoll and Gupta (1997) reported this option as the most common management option in Bangkok. However, internal and small conventional grease traps are no longer approved in some U.S cities, e.g. Austin, TX and Fort Worth, TX. The smallest sized grease trap accepted in Austin, TX and Fort Worth, TX are 100 gallon and 500 gallon, respectively (Austin Water Utility; Worth). Direct re-use of GTW in soap manufacture and animal feeding was suggested by Stoll and Gupta (1997) for FSEs that release large amount of GTW daily. This management strategy requires more frequent collection and on-site separation of FOG and solids. In addition, re-using GTW in animal feeding is prohibited in some countries such as China. Periodic collection and either treatment at anaerobic digesters or separation at central receiving containers were suggested as alternative options for medium size FSEs (Stoll & Gupta, 1997). This option is the most practiced GTW management strategy currently used in the US.

GTW periodically is collected by private haulers and generally transferred to WWTPs to feed into anaerobic digesters or processed as low grade lubricant. Not all WWTPs accept GTW into their anaerobic digesters because of the operational challenges and lack of knowledge regarding anaerobic co-digestion of GTW, Therefore, transferring GTW to further WWTPs is common practice. For example, GTW collected in Austin, TX is hauled for 3 hours to Dallas, TX (Liquid Environmental Solutions). The collected GTW is first hauled to transfer stations. Dewatering, usually with chemical addition, is performed at the transfer stations to decrease the volume. The dewatered GTW is transferred to the WWTP. This practice requires a licensed transfer station, dewatering unit, chemical addition, discharge of high BOD wastewater into the municipal wastewater collection system, and creates extra carbon dioxide emission resulting from additional truck trafficking. Anaerobic co-digestion of un-dewatered/raw GTW at nearest WWTP after collection may eliminate these requirements and reduce the inhibitory effects of GTW on the anaerobic co-digestion process. Performance of anaerobic co-digestion of un-dewatered GTW with MSW was evaluated in laboratory-scale digesters and the observed data are presented and discussed in the following chapters. More successful anaerobic co digestion of GTW applications at existing facilities may encourage WWTP operators to feed GTW into existing anaerobic digestion facilities.

2.3 ANAEROBIC CO-DIGESTION OF GTW

2.3.1 Biochemical process

Anaerobic degradation of GTW is achieved via consecutive biochemical breakdown of organic polymers into methane and carbon dioxide gases under anaerobic conditions. Variety of microorganisms work in step-wise metabolic interactions and generally interact and affect the performance of individual species. These associations have been identified generally in four recognizable phases including enzymatic hydrolysis, acidogenesis, acetogenesis and methanogenesis, a specific group of microorganisms dominate the substrate conversion in each phase (Mara & Horan, 2003). Hydrolytic microorganisms excrete extracellular enzymes such as cellulose, amylases, proteases, lipases, etc. to hydrolyze organic polymers into soluble monomers such as glucose, amino acids, glycerol, short-chain fatty acids and LCFAs. Acidogenic fermentation provides subsequent degradation of organic monomers to mainly acetate, propionate, butyrate, lactate, ethanol, carbon dioxide and hydrogen (Miyamoto & FAO, 1997). Products of acidogenesis are converted into acetate, carbon dioxide and hydrogen by obligate hydrogen producing acetogens (OHPA). The second group of acetogens, homoacetogens (HA), produces acetate from hydrogen and carbon dioxide. Homoacetogens play a relatively minor role in the anaerobic digestion process because of the number of homoacetogens in anaerobic digesters is lower than that of the methanogens (Toerien & Hattingh, 1969). The major amount of acetate and hydrogen are produced during acetogenesis. Hydrogen and carbon dioxide, and acetate are converted to methane as a final product by hydrogenotrophic and acetoclastic microorganisms, respectively, during methanogenesis. Approximately 2/3 of the final methane production results from the metabolization of acetate, while 1/3 is the result of oxidation of hydrogen. Acetate, hydrogen and carbon dioxide are the main substrates for methanogenesis; however, formate, methanol, methylamines, and carbon monoxide also can be converted to methane through hydrogenotrophic or disproportionation reactions (Joseph F. Malina & Pohland, 1992). Anaerobic conversion process and microorganisms responsible for each step are summarized in Fig. 24.2 in “Handbook of Water and Wastewater Microbiology” (Mara & Horan, 2003).

Maintaining the balance (syntrophy) between different metabolic groups of microorganisms, which are responsible for biochemical breakdown in each phase, is crucial. Accumulation of the products in one phase may inhibit the organisms in following phase, e.g., slower degradation of substrates in one phase may be a limiting factor for the following phase. The accumulation of fatty acids, substrates of OHPA, inhibits methanogens, while accumulation of hydrogen, substrate of hydrogenotrophic methanogens, inhibits OHPA (Mara & Horan, 2003; Mcinerney et al., 1981). The accumulation of either fatty acids or hydrogen caused by the lack of balance between the OHPA and the hydrogen consuming hydrogenotrophic methanogens may lead to the failure of the anaerobic process. Hydrogen consuming sulfate reducing bacteria and nitrate reducing bacteria also can help to prevent inhibition of acetogens; however, these bacteria can outcompete methanogens for available substrate in the abundance of alternative electron acceptors, such as nitrate and sulfate (Joseph F. Malina & Pohland, 1992; Mara & Horan, 2003; Miyamoto & FAO, 1997). Outcomes of sulfate and nitrate reductions, which are hydrogen sulfide and ammonium gas, respectively, can be toxic for the anaerobic digestion process. In addition, the balance between the hydrolysis-acidogenesis and acetogenesis-methanogenesis must be maintained. If the conversion of substrates is more rapid during hydrolysis-acidogenesis, accumulation of volatile acids may cause acidification of the process, while slower breakdown of complex polymers and β-oxidation of LCFAs may be a limiting factor for acidogens and methanogens, respectively (Weiland, 2010).

Lipids are one of the major organic polymers that can be present at high concentrations in GTW. Hydrolysis of lipids yields glycerol and LCFAs under anaerobic conditions (Hanaki et al., 1981). Glycerol is degraded into volatile fatty acids by acidogens, while LCFAs are degraded into acetate and hydrogen through β-oxidation (Ahmad et al., 2011; Schink, 1997). β-oxidation degrades an LCFA molecule with n+2 carbon atoms to one carboxylic acid molecule with n-2 carbon atoms, one molecule of acetic acid and 2 hydrogen until the complete conversion to acetic acid is completed (Madigan et al., 2009). The degradation rate of LCFAs is in direct proportion to the chain length (Novak & Carlson, 1970). The hydrolysis of lipids was reported to be more rapid than the further degradation of lipids, degradation of LCFAs through β-oxidation after hydrolysis, therefore, high lipid loadings may cause LCFAs accumulation in the digesters (Angelidaki & Ahring, 1995; Hanaki et al., 1981; Novak & Carlson, 1970). The degradation pathway of unsaturated LCFAs is still not completely elaborated, two pathways have been suggested; degradation through β-oxidation after saturation (Heukelekian & Mueller, 1958; Novak & Carlson, 1970), and direct degradation of unsaturated LCFAs through β-oxidation without saturation (Lalman & Bagley, 2000; Lalman & Bagley, 2001). The microbiology of LCFA degradation has not been studied extensively; therefore, specific microorganisms have not been identified. About 14 acetogenic bacteria species, belonging to the families Syntrophomonadanceae and Syntophaceae, have been identified to date that can degrade fatty acids with 4 or more carbon atoms in syntrophy with hydrogenotrophic methanogens and/or other hydrogen using microorganisms such as sulfate reducing bacteria (Jackson et al., 1999; McInerney et al., 2008; Sousa et al., 2007; Wu et al., 2006; Zhao et al., 1993). Seven out of 14 species are able to degrade LCFAs, fatty acids with more than 12 carbon atoms, while only 4 species are able to degrade unsaturated LCFAs (Alves et al., 2009).

2.3.2 Batch feed GTW digestion experiments

Batch experiments have been used widely in anaerobic digestion research in order to obtain fundamental information about the test substrates, especially before performing continuous feed digester experiments. Biochemical methane potential (BMP) test is the most used experimental procedure in batch anaerobic digestion of GTW. Biochemical methane potential tests were developed to estimate biochemical methane potential of various substrates in a simple way. Initial and final concentrations of desired parameters are generally compared to use in assessment of biodegradability of organics or presence of potential toxic/inhibitory substances. Development of BMP test started with the adaptation of Warburg respirometer to anaerobic treatment processes. Later, this technique was combined with anaerobic serum bottles containing inoculum, samples and defined media for the optimum growth of anaerobes (Miller & Wolin, 1974). The syringe method, gas transducer and water displacement method were used to monitor the volumetric biogas generations (Demirer et al., 2000; Nottingham & Hungate, 1969; Shelton & Tiedje, 1984). The first BMP test was developed and demonstrated by using peat samples in 1979 (Owen et al., 1979). The method continues to be used with some modifications today (Hansen et al., 2004).

Lipids have the highest methane potential among the organic biopolymers. Methane potentials of lipids, carbohydrates and proteins were reported as 990, 415 and 634 mL CH_4/g VS_{added}, respectively (Alves et al., 2009). In another study, theoretical methane potentials of lipids, carbohydrates and proteins were calculated as 1014, 415 and 496 mL CH_4/g VS_{added}, respectively (Angelidaki & Sanders, 2004). Methane potentials of lipid rich wastes under mesophilic conditions reported as high as that of pure lipids. Methane potentials of polymer and lime dewatered restaurant GTW were 993 and 878 mL CH_4/g VS_{added}, respectively, after a 120-day BMP test (Kabouris et al., 2007). Methane potential of GTW from meat processing facilities was reported as 900 and 918 mL CH_4/g VS_{added} (Luostarinen et al., 2009; Luste et al., 2009), while methane potential of DAF skimmings from meat processing plant was 872 mL CH_4/g VS_{added} (Girault et al., 2012). Pastor et al. (2013) determined methane potential of restaurant used oil as 748 mL CH_4/g VS_{added}. Methane potential of DAF skimmings from WWTPs was determined between 432 and 529 mL CH_4/g VS_{added} by Silvestre et al. (2011). High fat content of DAF skimmings from meat processing plant may cause higher methane potential than that of DAF skimmings from WWTPs. Methane potentials of major substrates such as MSW, manure and organic fraction of municipal solid waste (OFMSW) were reported less than that of lipid rich wastes. Methane potentials of MSW were between 263 and 470 mL CH_4/g VS_{added} (Davidsson et al., 2008; Donoso-Bravo & Fdz-Polanco, 2013; Kabouris et al., 2007; Luostarinen et al., 2009; Martínez et al., 2012; Pastor et al., 2013; Silvestre et al., 2011; Wang et al., 2013; Zhu et al., 2011), while methane potentials of OFMSW and manure were reported as 298 mL CH_4/g VS_{added} and 290 mL CH_4/g VS_{added}, respectively (Lansing et al., 2010a; Martin-Gonzalez et al., 2010). The reviewed results from batch GTW digestion experiments are presented in Table 2.2. Previously reported methane potential data from anaerobic digestion of major substrate alone and co-digestion with GTW were compared and increases in methane potentials caused by co-digestion of GTW were presented in Table 2.2. Optimum and inhibiting GTW feed ratio on VS basis also were compared in Table 2.2. Anaerobic co-digestion of major wastes, such as MWS and OFMSW, with GTW may be an attractive option for resource recovery because of the high methane potential of GTW.

Table 2.2: Previously reported methane potential data from batch experiments.

Reference	**Substrates (SS + GTW)**	**T**	**V_{active}**	**GTW Loading**		**Methane Yield**		
				Optimum	**Failure**[a]	**w/o GTW**	**Optimum**	**Increase**
		^{0}C	**mL**	**(% VS basis)**		**(mL CH_4/g VS_{added})**		**%**
(Donoso-Bravo & Fdz-Polanco, 2013)	60% PS & 40% WAS + GTW (≈3% VS)[b] from MWWTP	35	60	10	no	350	434	24%
(Young et al., 2013)	42% PS & 58% TWAS + MWWTP primary clarifier skimmings (5% VS)	35	600 - 642	na	na	430	692	61%
(Pastor et al., 2013)	MWS + Restaurant used oil (100% VS)	38	na	na	na	325	748	130%
(Martínez et al., 2012)	30% PS & 70% WAS + GTW from MWWTP (≈11% VS)	36	60	10	no	350	434	24%
(Girault et al., 2012)	WAS + DAF skimmings from meat processing plant (9% VS)	38	na	40[c]	no	483	na	-
(Silvestre et al., 2011)	70% PS & 30% WAS + DAF skimmings from 4 MWWTPs (≈10% VS)	35	500	na	na	322	529	64%
(Zhu et al., 2011)	MWS + Restaurant GTW (14% VS)	35	100	74	81	470	≈800	70%
(Li et al., 2011)	WAS + Restaurant oil receptacle waste (96% VS)	37	190	83	91	117	418	257%
(Martin-Gonzalez et al., 2010)	OFMSW + MWWTP primary clarifier skimmings (27.5% VS)	37	nr	15	35	298	318	7%
(Lansing et al., 2010a)	Swine manure + Restaurant used cooking grease (71% VS)	22 - 26	250k	10[d]	no	8.8[e]	44.5[e]	406%
(Luste et al., 2009)	MWS + GTW from meat processing plant (11% VS)	35	1500	100	no	na	900	-
	MWS + DAF skimmings from meat processing plant (3.5% VS)	35	1500	100	no	na	340	-
(Luostarinen et al., 2009)	MWS and Thickened GTW from meat processing plant (25% VS)	35	60	50[f]	70	263	788	200%

Table 2.2: Continue.

Reference	Substrates (SS + GTW)	T	V_{active}	GTW loading		Methane Yield		
				Optimum	Failure[b]	w/o GTW	Optimum	Increase
		^0C	mL	(% VS basis)		(mL CH_4/g VS_{added})		%
(Davidsson et al., 2008)	50% PS + 50% WAS, and GTW (17% VS)	35	500	60	no	325	681	110%
(Kabouris et al., 2008)	17% PS & 83% TWAS + dewatered restaurant GTW (31% VS)	35	1000	33	41	537[g]	1358[g]	153%
(Kabouris et al., 2007)	PS & TWAS & C + dewatered restaurant GTW (37-41% VS)	35	120	29	no	283	510	80%

Abbreviations: no, not observed; na, not available; MSW, municipal wastewtaer sludge; GTW, grease trap waste; PS, primary sludge; WAS, waste activated sludge; TWAS, thickened waste activated sludge; MWWTP, municipal wastewater sludge; DAF, dissolved air flotation; OFMSW, organic fraction of municipal solid waste; C, secondary sludge cake; V_{active}, liquid volume in digestion bottles.

[a] GTW loading at failure or inhibition

[b] VS concentration of GTW used in the study was shown in parenthesis.

[c] % COD basis.

[d] % volume basis.

[e] L CH_4/d.

[f] % wet weight of GTW

[g] % mL CH_4/ g $VS_{destroyed}$

2.3.3 Continuous feed GTW co-digestion experiments

Continuous feeding reactor experiments usually performed after obtaining fundamental information such as methane potentials and inhibiting GTW feeding ratios from BMP tests (Girault et al., 2012; Kabouris et al., 2008; Luostarinen et al., 2009; Pastor et al., 2013; Silvestre et al., 2011). Continuous feed reactor experiments simulate AcoD process closer to full-scale applications than batch experiments or BMP tests. Continuous feed reactor experiments typically are performed to determine process performance in terms of methane potentials and VS reductions, inhibiting GTW feed ratios, reactor configurations, and effects of temperature and pretreatment on process performance. Substrates are fed continuously or semi-continuously during the experiment. The monitored parameters are evaluated after the process reaches stability. Creamer et al. (2010) considered process was at stability when effluent pH was constant and coefficient of variation of daily biogas production was less than 5% over a continuous 5-day period. Liu and Buchanan (2011) assumed that the process has reached stability when biogas production, pH and alkalinity values over 5 continuous days lay within two standard deviations of the corresponding mean values. Martin-Gonzales et al. (2010) defined stability condition as less than 10% change in volatile fatty acids, biogas yield, and volatile and total solids reductions during a minimum of one HRT.

Laboratory-scale continuous or semi-continuous feed reactor experiments demonstrated that AcoD of GTW with major wastes such as MWS, OFMSW and manure can increase methane production and VS reduction compared to digestion of these major substrates individually. Increase in methane potential from AcoD of MWS and GTW, from restaurants, WWTP DAF units and meat processing plants, under mesophilic (35-38 ^{0}C) or thermophilic (50-55 ^{0}C) conditions was reported between 27% and 318%

(Davidsson et al., 2008; Wang et al., 2013), while increase in VS reduction was reported between 26% and 83% (Kabouris et al., 2008; Wang et al., 2013). Martin-Gonzalez et al. (2010 and 2011) reported 36% and 46% increase in methane potential and up to 4% increase in VS reduction when co-digesting OFMSW with WWTP primary clarifier skimmings. Anaerobic co-digestion of swine manure with GTW yielded 7% increase in methane potential at 20 to 26 ^{0}C operating temperatures (Lansing et al., 2010a). Performance data from continuous feed reactor experiments are presented in Table 2.3. The results reported for all studies indicated an increase in methane potential and VS reduction when co-digesting with GTW.

One of the major operational problems of AcoD of GTW is LCFA accumulation caused by the high concentration of lipids in the GTW. Accumulation of LCFA occurs at high GTW loadings because hydrolysis of LCFA is more rapid than further degradation through β-oxidation as discussed under biochemical process section. Accumulation of LCFA may cause 1) sludge flotation and washout, 2) substrate and product transport limitation, 3) inhibition of methanogens (Alves et al., 2009). Therefore proper GTW feed ratios must be maintained otherwise LCFA accumulation may lead to inhibition or even failure of the AcoD process. Studies have been performed on various major substrates and GTW combinations to determine the optimum and inhibiting GTW feed ratios are presented in Table 2.3. Usually the inhibition threshold occurs just before the optimum feeding ratio in terms of methane production and VS reduction. Optimum GTW feed ratios were varied between 12.5% and 68% on VS basis, while inhibiting GTW feed ratios were between 30% and 100% on VS basis (see Table 2.3). Unsteady characteristics of GTW and the variety of substrates combinations may cause variations in optimum and inhibiting GTW feed ratios. Therefore, pilot-scale experiments are recommended before full-scale applications of AcoD of GTW.

Anaerobic co-digestion of GTW typically is performed under mesophilic or thermophilic conditions. Increase in process temperature generally yields high removal of pathogens and increase in biochemical and enzymatic reactions leading to higher microbial growth rates and more rapid degradation; however, high process temperatures does not change the ultimate methane potential of substrates. Therefore, AD digestion under thermophilic conditions may be more effective at high organic loading rates (OLR) or short HRTs when rapid digestion is needed. On the other hand, increase in process temperature may result in higher LCFA concentrations and higher risk of ammonia inhibition (Angelidaki et al., 2003). Moreover, methanogens were reported to be more sensitive to LCFA toxicity under thermophilic conditions (55 ^{0}C) than under mesophilic conditions (30 and 40 ^{0}C) (Hwu & Lettinga, 1997). Methanogenic diversity detected in digesters was less at thermophilic temperatures compared to digesters operated at mesophilic temperatures (Karakashev et al., 2005). Higher methane potentials and more VS reductions were reported for both AcoD of MWS with GTW and single digestion of MWS under thermophilic conditions (52 ^{0}C) than mesophilic conditions (35 ^{0}C) (Kabouris et al., 2009a). The authors also reported higher nutrient release at 52 ^{0}C temperature than 35 ^{0}C. Suto et al. (2006) reported slightly lower biogas production rates from AcoD of MWS and GTW at 50 ^{0}C than 35 ^{0}C. However, the authors were monitored more LCFA degradation at thermophilic conditions than mesophilic conditions. Methane potentials and VS reductions from both AcoD of OFMSW with GTW and single digestion of OFMSW at 52 ^{0}C were greater than that at 37 ^{0}C (Martin-Gonzalez et al., 2011; Martin-Gonzalez et al., 2010). Anaerobic digestion is also possible in the lower portion of mesophilic range (20-30 ^{0}C), but at higher HRTs. Anaerobic co-digestion of swine manure with used cooking oil at temperatures between 22 and 26 ^{0}C resulted in methane potential of 310 mL CH_4/g VS_{added} (Lansing et al., 2010a). An important change in optimum and inhibiting GTW feeding ratios due to temperature change was not observed in any of the studies reviewed above. According to the results from reviewed studies, it can be concluded that AcoD at thermophilic temperatures is more beneficial, if higher VS reduction and methane production is desired. However, additional energy requirement to increase the temperature from mesophilic to thermophilic levels and higher nutrients in the effluent needs to be considered.

Most of the continuous feed experiments were performed with single phase continuous stirred reactor (CSTR) configuration commonly used at WWTP. However, two-phase CSTR, two-stage CSTR and plug flow reactor configurations also were used in AcoD of GTW (Kabouris et al., 2009b; Lansing et al., 2010a; Li et al., 2014). Mesophilic (35 ^{0}C) and thermophilic (52 ^{0}C) AcoD of MWS with GTW in semi-continuously fed two-phase CSTRs were successfully performed by Kabouris et al. (2009). The waste mixture was first fed into an acid phase reactor operated at 35 ^{0}C temperature, 1-day HRT and 52.5 g VS/L-d OLR. Effluent from the acid phase reactor was fed into two methane phase reactors operated at 12-day HRT, 4.35 g VS/L-d OLR and 35 ^{0}C and 52 ^{0}C temperatures, respectively. The authors associated high degradable COD reductions to increase in microbial activity at the high OLR, although microbial activity was not analyzed. An important decrease in unsaturated fat concentration (43%) was achieved during the acid phase which may have reduced LCFA inhibition in methane phase. Two-phase CSTR configuration may be more reasonable than single-phase, if high OLRs or shorter HRTs are desired. Li et al. (2014) reported 493 mL CH_4/g VS_{added}

methane potential from AcoD of PS with GTW in a semi-continuously fed two-stage CSTR. Two thermophilic digesters in series operated at 1.83 g VS/L-d and 0.46 g VS/L OLRs and 24-day HRTs, respectively. Methane production and VS reduction in the first stage digester were 10.82 L/d and 75.1%, respectively, and 1.91 L/d and 9.45% in the second stage digester, respectively. Multiple stage digester configurations may not result in low effluent concentrations.

Pretreatment prior to anaerobic digestion usually is applied to improve hydrolysis of organic polymers into smaller monomers that can be utilized by bacterial cells. Various pre-treatment methods were investigated including thermal, ultrasound, acid, base and bacterial product addition. Effects of thermal, ultrasound, acid and base addition pretreatments on solubilization and methane potential of GTW from a meat processing facility were evaluated by Luste et al. (2009). Ultrasound and thermal treatments were the most effective pretreatments in terms of solubilization. Application of heat (70 ^{0}C) for 60 minutes increased VS and soluble COD concentrations 40% and 178%, respectively, compared to the VS and soluble COD concentrations of untreated GTW. Application of ultrasound at 24 kHz frequency with 5600±300 kJ/kg TS energy input increased VS and soluble COD concentrations 30% and 188%, respectively, compared to the VS and soluble COD concentrations of untreated GTW. On the other hand, acid and base addition did not enhance solubilization of GTW. Methane potentials of untreated and pretreated GTW were determined through BMP tests under mesophilic conditions (35 ^{0}C). Hydrochloric acid addition at 6M (pH 2-2.5 for 4 hours) increased methane potential of GTW. The authors related lower methane potentials of thermal and ultrasound pretreated GTW to a possible inhibition by hydrolysis byproducts or re-crystallization of hydrolyzed compounds. Luste and Loustarinen (2010) also investigated the effects of thermal pretreatment on AcoD of MWS and a mixture of animal by-products including GTW from a meat processing facility at 35 ^{0}C in semi-continuously fed CSTRs. The results observed indicated that thermal pretreatment at 70 ^{0}C for one hour decreased the particle sizes, increased the concentrations of soluble compounds (VFA, ammonium and hydrogen) and increased methane potential. A surplus net energy production of 0.56 to 0.66 GWh/year was calculated when digesting 41800 ton/year thermally pretreated MWS and animal by-products at 1:7 ratio on volume basis. Donoso-Bravo and Fdz-Polanco (2013) performed a series of BMP tests at 35 ^{0}C to evaluate the effects of a commercial enzyme-lipase addition and dosage on AcoD of MWS and GTW. Increases in methane potentials with enzyme addition were about 130%, 127% and 78% at GTW additions of 2%, 5% and 10% on VS basis, respectively. The optimum enzyme dosage was determined between 0.33% and 0.83% on volume basis. The effects of ultrasound and thermo-chemical pretreatments on methane potential of a MWS and GTW mixture was assessed through a series of BMP test by Li et al. (2013). Application of ultrasound at increasing power inputs between 5300 and 36000 kj/kg TS (20 kHz) and treatment times between 5 min and 40 min did not improve methane potential. However, sodium hydroxide addition up to pH 8, 10 and 12 and stirring the feed mixture at 55 ^{0}C for 1.5 hour before the digestion resulted in higher methane potentials than AcoD of untreated feed. The highest increase in methane potential (approximately 9.9%) with thermo-chemical pretreatment was achieved at pH=10 and temperature of 55 ^{0}C. Moreover, AcoD of thermo-chemically pretreated (at pH=10 and T = 55 ^{0}C) MWS and GTW mixture in semi-continuous feed two-stage laboratory-scale digesters resulted in higher methane potential and VS reduction than that of untreated feed (Li et al., 2014). Thermal, thermo-chemical and enzyme addition pretreatment methods have been reported to increase methane production. However, applicability and cost effectiveness of these methods in full-scale operations have not been evaluated in sufficient details.

Table 2.3: Performance data from semi-continuous feed GTW co-digestion experiment.

Reference	Substrates (MWS + GTW)	T	HRT	GTW loading		OLR	VS Reduction		Methane		Methane Yield			Reactor & Feeding configuration
				Opt	Failure[a]	Opt	w/o GTW	Opt	w/o GTW	Opt	w/o GTW	Opt	Increase	
		^{0}C	days	(% VS basis)		(gVS/L-d)	(%)		(%)		(mL CH_4/gVS$_{added}$)		%	
(Li et al., 2014)	PS + Restaurant GTW (77% VS)[b]	55	24	49	na[c]	1.83	na	60	na	70	na	493-346[d]	-	2-Stage CSTR, Semi-C
(Wang et al., 2013)	TWAS + Restaurant GTW (≈19% VS)	37	20	20	40	2.16	30	55	60	70	180	752	318%	CSTR, Semi-C
(Noutsopoulos et al., 2013)	74% PS & 26% WAS + MWWTP primary clarifier skimmings (64% VS)	35	15	60	60	3.5	52	59	65	70	452	700	55%	CSTR, Semi-C
(Alanya et al., 2013)	50% PS & 50% WAS + MWWTP primary clarifier skimmings (≈28-61% VS)	35	13	64[e]	na	11[f]	51	70	na	na	na	308[g]	na	CSTR, Semi-C
(Pastor et al., 2013)	MWS + Restaurant used oil (100% VS)	38	30	19	na	0.9	na	na	na	65	na	716	-	CSTR, Semi-C
(Razaviarani et al., 2013)	75% PS with scum & 25% TWAS + Restaurant GTW (13-26% VS)	36	20	23	30	1.58	44	56	≈66	≈66	1.1[h]	1.83[h]	66%	CSTR, Semi-C
(Girault et al., 2012)	WAS + DAF skimmings from meat processing plant (9% VS)	36	25	60	80	1.2	29	44	66	69	264	546	107%	CSTR, Semi-C
(Liu & Buchanan, 2011)	75% PS & 25% TWAS + Restaurant GTW (≈64% VS)	37	20	68	na	4.24	na	73	na	66	na	620	na	CSTR, Semi-C
(Wan et al., 2011)	TWAS + Restaurant GTW (3% VS)	37	15	64	75	2.34	40	57	65	67	252	598	137%	CSTR, Semi-C

Table 2.3: Continue.

Reference	Substrates (MWS + GTW)	T	HRT	GTW loading		OLR	VS Reduction		Methane		Methane Yield			Reactor & Feeding configuration
				Opt	Failure[b]	Opt	w/o GTW	Opt	w/o GTW	Opt	w/o GTW	Opt	Increase	
		^{0}C	days	(% VS basis)		(gVS/L-d)	(%)		(%)		(mL CH_4/gVS$_{added}$)		%	
(Silvestre et al., 2011)	70% PS & 30% WAS + DAF skimmings from MWWTP (≈6-14% VS)	35	20	23	37	1.6	36	52	72	70	249	369	48%	CSTR, Semi-C
(Martin-Gonzalez et al., 2011)	OFMSW + MWWTP primary clarifier skimmings (10% VS)	55	14.4	14	na	5	70	73	68	72	360	490	36%	CSTR, Semi-C
(Martin-Gonzalez et al., 2010)	OFMSW + MWWTP primary clarifier skimmings (27.5% VS)	37	14.5	15	na	~4.5	66	65	62	63	240	350	46%	CSTR, Semi-C
(Luste & Luostarinen, 2010)	MWS + GTW from meat processing plant[i]	35	20	12.5	na	2.8	na	na	na	65	na	430	-	CSTR, Semi-C
(Lansing et al., 2010a)	Swine manure + Restaurant used cooking grease (71% VS)	22 - 26	40	2.5[j]	5[j]	na	na	na	70	67	290	310	7%	Plug-flow, Semi-C
(Creamer et al., 2010)	Swine feces + dewatered DAF skimmings from a meat processing plant	54.5	10	50[k]	100[k]	4.7	na	na	na	71	na	470	-	CSTR, Semi-C
(Luostarinen et al., 2009)	SS and Thickened GTW from meat processing plant (25% VS)	35	16	46	55	3.46	52	67	63	62	278	463	67%	CSTR, Semi-C
(Kabouris et al., 2009b)	40%PS + 60%TWAS + polymer dewatered restaurant GTW (≈41% VS)	35	12	48	na	4.35	25	45	66	66	159	473	197%	2-Phase CSTR, Semi-C
		52	12	48	na	4.35	31	51	66	69	197	551	180%	2-Phase CSTR, Semi-C

Table 2.3: Continue.

Reference	Substrates (MWS + GTW)	T	HRT	GTW loading		OLR	VS Reduction		Methane		Methane Yield			Reactor & Feeding configuration
				Opt	Failure[b]	Opt	w/o GTW	Opt	w/o GTW	Opt	w/o GTW	Opt	Increase	
		^{0}C	days	(% VS basis)		(gVS/L-d)	(%)		(%)		(mL CH_4/g VS_{added})		%	
(Davidsson et al., 2008)	50% PS + 50% WAS, and GTW (17% VS)	35	13	30	100	2.4	45	58	65	69	271	344	27%	CSTR, Semi-C
(Kabouris et al., 2008)	17% PS & 83% TWAS + dewatered restaurant GTW (31% VS)	35	13.3	20	na	3.75	35	44	na	na	610[n]	1385[n]	127%	CSTR, Semi-C
(Suto et al., 2006)	40% PS + 60% TWAS + Restaurant GTW (3.6-14% VS)	35	20	35[k]	50[k]	na	na	na	na	na	0.9[m]	1.74[m]	93%	CSTR, Semi-C
		50	20	35[k]	50[k]	na	na	na	na	na	0.88[m]	1.59[m]	81%	CSTR, Semi-C

[a] Opt, optimum; PS, Primary sludge; WAS, waste activated sludge; TWAS, thickened waste activated sludge; GTW, grease trap waste; MWS, municipal wastewater sludge (PS+WAS); OFMSW, organic fraction of municipal solid waste; DAF, dissolved air flotation

[b] GTW loading at failure or inhibition

[c] VS concentration of GTW used in the study was shown in parenthesis.

[d] na: not available

[e] Methane potentials from 1st and 2nd stage digesters, respectively.

[f,g,h] % COD basis, g COD/L-d, mL CH_4/g COD, respectively.

[i] Biogas production rate (m^3/d)

[j] Meat processing by-products composed of digestive tract content & drum sieve waste: DAF skimmings: GTW at 53:34:13 wet weight ratio (13%:7%:16% VS)

[k] % Volume basis

[l] % Weight basis

[n] mL CH_4/g $VS_{destroyed}$

[m] Biogas production rate (L/hr)

2.4 Recovery of anaerobic digestion following process upset

Recovery of upset anaerobic digestion as a result of excessive GTW loadings was evaluated using three methods: (1) addition of base (sodium hydroxide, NaOH), (2) batch digestion and (3) switching the feed from co-digestion to mono-digestion. Addition of 0.2% w/v NaOH on a single day increased the pH from 5.6 to 7.72; however, biogas production did not recover (Wan et al., 2011). After 2 weeks of batch digestion, pH increased from 5.6 to 6.3. Mono-digestion of major substrate in semi-continuous feed configuration was started after 40 days of batch digestion. Biogas production only reached half of the biogas production of the undisrupted digestion when the digestion process was at stability (Wan et al., 2011). Switching the feeding strategy from co-digestion to mono-digestion also used to decrease the inhibiting effects of GTW overdose. As a result of switching the feeding strategy by feeding only the major substrate for 31 days, the pH increased from 6.6 to 6.9, alkalinity increased from 3.64 to 4.29 g $CaCO_3$/L, biogas production and methane content were at levels of the undisrupted digestion values (Wang et al., 2013). Recovery was started (at pH 6.6) immediately after the first signs of digester inhibition and full recovery was maintained in the last study. On the other hand, feeding at excessive GTW loadings was continued (until pH 5.6) even after inhibition started in the first two methods. Therefore, some of the microbial activity may have been permanently damaged and a full recovery was not maintained in the first study. Early determination of inhibition and taking the proper precautions may be crucial in digester recovery. Feeding an upset digester with effluent of a stable digester may dilute toxic content of the digester and provide a microbial population that will result in biodegradation of accumulated compounds. Feeding with effluent of a stable digester may be another alternative for digester recovery.

2.5 MATHEMATICAL MODELING

Achieving a successful GTW anaerobic co-digestion and higher methane production requires careful management of feed ratios. Random GTW feed ratios may cause inhibition and possibly digestion failure. Therefore, accurate modeling of AcoD process is needed to predict methane production and process behavior at different GTW feed ratios. Anaerobic Digestion Model No.1 (ADM1) is one of the most recent and sophisticated models that was developed by Anaerobic Digestion Modeling Task Group of the International Water Association in 2002 (Batstone et al., 2002). Although successful applications of the model with different waste streams are available, the model has not been applied to AcoD of MWS and GTW (Mata-Alvarez et al., 2011). Non-linear regression of the first-order reaction equation was applied to simulate the co-digestion process and estimate the first order rate constants of polymer and lime dewatered GTW COD degradation (Kabouris et al., 2007). Non-linear regression of the first-order reaction equation also applied to evaluate the effects of enzyme addition and dosage on methane production from AcoD of MWS and GTW from a WWTP. (Donoso-Bravo & Fdz-Polanco, 2013). The authors simulated the time dependent methane production and estimated kinetic parameters such as ultimate cumulative methane production and first order reaction rate constant (hydrolysis rate). Linear and non-linear regression models were used jointly to evaluate the performance of AcoD of WAS and restaurant oil receptacle waste (Li et al., 2011). The first order biogas production rate and time to reach steady state were estimated using linear regression of the first-order reaction equation, while length of lag phase, the ultimate cumulative biogas production and maximum biogas production rate were estimated using non-linear regression of modified Gompertz equation. Non-linear regression of the modified Gompertz equation was successfully applied to reproduce the time-dependent methane production from AcoD of MWS and restaurant oil receptacle waste.

2.6 CONCLUSIONS AND FUTURE RESEARCH NEEDS

Anaerobic co-digestion of GTW with MWS at existing anaerobic digesters may be an economically and environmentally sustainable alternative for GTW management. Industrial applications are available; however, published data are scarce for full-scale anaerobic co-digestion. Additional research, especially on full-scale anaerobic digestion, is required to obtain knowledge of the process behavior, assess the feasibility and develop more effective management strategies. Future research dealing with GTW management through co-digestion including FOG deposition mechanisms, upgrade of grease traps, recovery of digesters following upsets and mathematical modeling of GTW co-digestion are summarized below.

Many municipalities in Texas require FSEs to install grease traps and periodically empty the traps (Rainwater, 2004). However, most of the municipalities have no further GTW management strategies. Based on observations in Texas, operators have prejudice against feeding GTW into anaerobic digesters because of the lack of information and serious negative consequences of improper management. This lack of data leads to difficulties in GTW management in Texas. Not all municipalities are accepting GTW and additional treatment and long distance hauling are required. Similar difficulties are probably experienced in other states. Ultimate disposal or resource recovery of GTW needs to be addressed in management policies of municipalities and local solutions should be identified. The economic feasibility of AcoD of un-dewatered GTW with MWS at local anaerobic digesters versus chemically dewatering at local transfer stations and hauling long distances for disposal or resource recovery must be evaluated. A comprehensive comparison of current treatment/disposal methods has not been evaluated to identify the most feasible disposable method for GTW management. Additional research also is needed on the products of GTW anaerobic co-digestion. Fertilizer value, compost quality and dewaterability of GTW after AcoD must be established to develop proper management strategies. Theoretical energy production from AD of GTW and co-production of biodiesel and methane from GTW were estimated to be close to each other (Lopez et al., 2014), further research is needed to support theoretical data.

Research papers dealing with characterization of FOG deposits in wastewater collection lines, and formation mechanisms are scarce. Evidence of FOG deposition in wastewater collection lines attributed to GTW discharges have been reported in all studies, the formation process has not been clearly identified yet. The suggested deposition mechanisms, such as saponification of free fatty acids and metal ions, and DLVO type aggregation processes needs to be supported with further research. The relationship between hardness in wastewater and calcium concentration in FOG deposits is ambiguous and needs to be clarified. Additional research is needed to determine the effects of different types of free fatty acids on the saponification reactions. The factors, other than corrosion, that are causing the excess calcium in FOG deposits need to be identified. Research is required to confirm the contribution of microbial activity on calcium production. The role of oil in the formation of FOG deposits still is not defined very well. Transformation of unsaturated fatty acids to saturated forms in the wastewater collection lines is largely unknown. Additional information on the rheology of FOG deposits will provide a better understanding on the flow conditions and adhesive characteristics of the deposits. Effective control of FOG deposits in wastewater collection lines may be possible by building knowledge on these mechanisms.

The characteristics of GTW are not sufficiently characterized. A comprehensive physico-chemical characterization of GTW is needed. Information on micro-nutrients, macro-nutrients, and trace metals in GTW are needed to effectively upgrade .grease trap performance. A classification of FSEs such as meat processing facilities, barbeque restaurants, buffet style FSEs, and etc. may reduce the deviations in the GTW characteristics. Installation of tube settlers, filter media or effluent filters in conventional grease traps may help to meet discharge standards during the peak restaurant operation hours when more kitchen wastewater is discharged. In addition, FOG and solids can be collected separately in tube settler installed grease traps. However, operation of modified grease traps may require experienced staff and more frequent maintenance. Furthermore, determination of the size of oil droplets and size distribution of GTWs and the coalescence ability is important to design more effective grease traps. Cell immobilized matrix installed grease traps showed substantial increase in FOG and COD removal efficiencies and provided more stable removal under fluctuating inflows; however, the clean up and seeding frequencies need to be studied. Addition of biological products, such as lipase, can help to hydrolyze FOG components in a short time. However, investigations of simple ways of biomass immobilization in grease traps must be established. Addition of biological products may yield higher effluent FOG concentrations resulting from the breakdown of oil droplets into smaller sizes. A revision of the current regulations on the addition of biological products in grease traps may be required. Very few studies on upgrading grease traps are found in the literature. More research on modified grease traps with simple operating procedures may improve the efficiency of grease traps and provide pretreatment for kitchen wastewaters.

The degradation pathway of unsaturated LCFAs and the microbiology of LCFA degradation is not completely understood. Identification of degradation pathways of unsaturated LCFAs and microorganisms responsible for LCFAs degradation may help to improve mathematical modeling of the process and lead to reactor modifications that may yield more efficient degradation. An important decrease in unsaturated fat concentration (43%) was reported during the acid phase of AcoD of MWS and GTW in a laboratory-scale 2-phase CSTR digester configuration (Kabouris et al., 2009b). Multiple phase reactor configurations may help to reduce LCFA inhibition in methane phase; however, additional research is needed to determine, if this result may be achieved with other GTW streams.

2.6.1 Research Needs Addressed in This Study

Most of the studies that have been reported on AcoD of dewatered/concentrated GTW with MWS, OFMSW or manure. Using dewatered/concentrated GTW as a co-substrate in AcoD was reported to yield not only stabilization of FOG, but also resource recovery because of the high methane potential of FOG (Davidsson et al., 2008; Kabouris et al., 2009b; Kabouris et al., 2008; Liu & Buchanan, 2011; Luostarinen et al., 2009; Razaviarani et al., 2013; Wang et al., 2013). However, inhibition and potential failure of the digestion process were reported when fed with lipid rich GTW (Liu & Buchanan, 2011; Wan et al., 2011; Wang et al., 2013). Accumulation of long chain fatty acids (LCFAs) resulting from feeding high concentrations and loading rates of GTW were addressed as the source of inhibition in anaerobic co-digestion of GTW. The effects of co-digestion of un-dewatered GTW on methane production and system inhibition have not been established.

In most of the studies, experiments were ended when the anaerobic digestion became inhibited or failed because researchers often focused on determining optimum GTW feeding ratios in terms of methane production. Therefore, recovery of digester operation following upset condition was not studied in sufficient detail. Additional research is needed to determine effective and rapid methods to recover digesters after failures caused by GTW overdose.

Only a few studies have applied mathematical models to describe time dependent cumulative methane potential from AcoD of MWS and GTW and estimated kinetic parameters that can be used in more sophisticated AcoD models. However, considerable variations in kinetic parameters were observed among these studies likely caused by variations in the feedstock characteristics. There is no mathematical model that describes the relationship between GTW feed ratio and methane potential. Such a mathematical model will be an invaluable resource for the prediction of methane production and determination of optimum GTW feed ratios in full-scale applications.

Chapter 3: Batch GTW Co-Digestion Experiments

ABSTRACT

The performance of anaerobic co-digestion (AcoD) of municipal wastewater sludge (MWS) with un-dewatered grease trap waste (GTW) was assessed using modified biochemical methane potential tests under mesophilic conditions (35 ^{0}C). Methane potentials, process inhibition and chemical behavior of the process were analyzed at different GTW feed ratios. Nonlinear regression analyses of first order reaction and modified Gompertz equations were performed to assist in interpretation of the experimental results. The methane potential of un-dewatered GTW was measured as 606 mL CH_4/g VS_{added}, while the methane potential of MWS was only 223 mL CH_4/g $VS_{added.}$ The results indicated that anaerobic digestion of GTW without dewatering yields a lower methane potential than concentrated/dewatered GTW. However, AcoD of MWS and GTW still yields over two times more methane potential and approximately 10% more VS reduction than digestion of MWS alone. Inhibition of the AcoD process has been reported for highly concentrated/dewatered GTW additions (≥ 70% on VS basis) in previous studies; however, no inhibition was observed at 100% un-dewatered GTW digestion. These results indicate that AcoD of un-dewatered GTW may reduce the inhibition risk compared to AcoD of concentrated/dewatered GTW. In addition, a mathematical equation was developed in this study to describe the relationship between GTW feed ratio (on a VS basis) and methane potential. Experimental data from the current study as well as previous BMP studies were successfully fit to this relationship and allowed estimation of maximum methane potential rate and the initial methane potential lag phase, parameters that provide additional insight into the factors affecting BMP.

3.1 INTRODUCTION

Discharge of fat, oil, and grease from food service establishments (FSE) into the wastewater collection systems in amounts that can cause obstruction in the publicly owned treatment works and the collection system is prohibited by Pretreatment Program regulations in 40 CFR 403.5(b)(3) and underscores the potential risk of line blockages (EPA, 2014; EPA, 2004). Fat, oil, and grease residues, solids such as food particles along with some kitchen wastewater are collected in grease traps that are separate from the municipal wastewater stream. Grease traps are periodically emptied and the wastes are hauled for treatment. Grease trap residues often are called grease trap waste (GTW) and constitute a new kind of waste stream. Landfilling GTW is restricted by environmental legislations and plans, such as The Austin Resource Recovery Master Plan on zero waste going to landfills policy and The European Union Council Directive 1999/31/EC on the landfill of waste, because of the potential of aggravating the effects of climate change (City of Austin, 2011; EU, 1999). Rendering GTW into municipal wastewater treatment plants creates serious problems caused by oxygen transfer limitations and a slow rate of aerobic biodegradation (Chipasa & Mędrzycka, 2006). Incineration and composting are not suitable because of the high moisture content of grease trap waste (Coker, 2006b). Land application of GTW may provide organic carbon and reduces nitrogen leaching; however, spray application of GTW on vegetation and land can cover the plants and land surface with grease and reduce yields and clog pores in the soil (Coker, 2006a; Rashid & Voroney, 2004)

Anaerobic co-digestion (AcoD) of municipal wastewater sludge from primary and secondary sedimentation tanks (MWS) with GTW is an attractive option for resource recovery because of the high methane potential of FOG in GTW (Alves et al., 2009; Angelidaki & Sanders, 2004). Anaerobic co-digestion of GTW at favorable mixture ratios has been reported to increase biogas production and methane content as well as improve volatile solids (VS) reduction (Davidsson et al., 2008; Kabouris et al., 2007; Kabouris et al., 2008; Lansing et al., 2010a; Li et al., 2011; Luostarinen et al., 2009; Pastor et al., 2013; Zhu et al., 2011). On the other hand, digestion of GTW alone or AcoD of GTW at high GTW feed ratios can cause inhibition and even failure of the digestion process (Kabouris et al., 2008; Lansing et al., 2010a; Li et al., 2011; Luostarinen et al., 2009; Zhu et al., 2011). AcoD of MWS and GTW at favorable mixture ratios can dilute toxic or inhibitory substances and balance critical anaerobic digestion process parameters, such as carbon:nitrogen (C:N) ratio, alkalinity, ammonia nitrogen, and volatile acids for a more stable digestion (Mata-Alvarez et al., 2011). The performance data observed in previous studies indicates that AcoD of MWS with GTW is an alternative approach to integrate GTW into resource recovery and increase methane production at existing anaerobic digestion plants. However, most of the previous studies have focused on concentrated/dewatered GTW. The effect that co-digestion of MWS with un-dewatered GTW has on methane production and system inhibition have not been established. Although anaerobic co-digestion of MWS with concentrated/dewatered GTW was reported to cause process inhibition, no research has been conducted with un-dewatered GTW to identify inhibiting GTW concentrations/loading rates, to our knowledge.

Mathematical models have been used to estimate kinetic and performance parameters associated with methane potential for a variety of waste streams and to assist in interpretation of experimental results. To date, only a few of these studies have applied mathematical models to describe time dependent cumulative methane potential from anaerobic co-digestion of MWS and GTW. Non-linear regression of the first-order reaction equation was applied to simulate the co-digestion process and estimate the first order rate constants of polymer and lime dewatered GTW chemical oxygen demand (COD) degradation (Kabouris et al., 2007). Non-linear regression of the first-order reaction equation also was used to evaluate the effects of enzyme addition and dosage on methane production from AcoD of MWS and GTW from a wastewater treatment plant (WWTP) (Donoso-Bravo & Fdz-Polanco, 2013). The authors simulated the time dependent methane production and estimated performance and kinetic parameters including the ultimate cumulative methane production and first order reaction rate constant (hydrolysis rate). Linear and non-linear regression models were used jointly to evaluate the performance of AcoD of waste activated sludge and restaurant oil receptacle waste (Li et al., 2011). The first order biogas production rate, time to reach steady state, length of lag phase, the ultimate cumulative biogas production and maximum biogas production rate were estimated. However, there is no published study that describes the mathematical relationship between GTW feed ratio and methane potential. In addition, parameter estimation using non-linear regression models have not been performed for AcoD of MWS with GTW.

Assessment of the use of un-dewatered GTW as the co-substrate in AcoD is not well documented despite the potential for GTW to increase the quantity of methane generated in anaerobic digesters and reduce the system inhibition. To this end, this study is directed at determining biochemical methane potentials of un-dewatered GTW, MWS and mixtures of un-dewatered GTW and MWS at various ratios. The observed performance and stability results were compared with previously reported inhibiting concentrations and methane potentials of concentrated/dewatered GTW. Regression analyses of non-linear models were performed to assist in interpretation of the experimental results. In addition, in this study a mathematical expression was developed to describe the relationship between GTW feed ratio (on VS basis) and resulting methane potential. Fitting of the observed data to this mathematical expression allowed the estimation of key performance parameters that affect methane potential as a function of GTW feed ratio.

3.2 Materials and Methods

3.2.1 Materials

Municipal wastewater sludge, raw grease trap waste (un-dewatered GTW) and mixtures with different ratios of both MWS and GTW were used as the substrates for the experiment. MWS was obtained from a biosolids management plant in Austin, Texas which treats solids consisting of primary sludge (PS) and waste activated sludge (WAS). The plant receives MWS from two wastewater treatment plants in Austin. The MWS was screened and thickened before a 30-day mesophilic anaerobic digestion. Grease trap waste sample was obtained from the receiving facility of a GTW hauler in Austin, Texas. The hauler collects GTW from grease traps of restaurants, food courts, food processing facilities, etc. grease trap waste sample was collected before dewatering at the receiving facility. The anaerobic digestion inoculum (INO) was obtained from the same biosolids treatment facility where the MWS samples were obtained. Digester effluent before further thickening was collected as INO. Initial tests and incubation were completed within two days after collecting the samples.

3.2.2 Experimental Procedure

Biochemical methane potential (BMP) tests were conducted in 250 mL serum bottles with 120 mL liquid volumes. Digestion was performed in an incubator at 35 ^{0}C for 31 days. The length of incubation period was established based on the EPA's minimum of 15-day hydraulic residence time (HRT) suggestion for complete mixed digesters (EPA, 2011) and the findings of Kabouris et al. (2007) on the ultimate anaerobic biodegradability of MWS and GTW. After a lag phase of 3 to 5 days, a 15 to 20-day HRT under mesophilic (35 ^{0}C) and batch conditions was found sufficient to produce between 90 and 95% of total available methane from GTW and primary sludge (Kabouris et al., 2007). The bottles were shaken manually once a day. Photos of the batch digestion bottles are presented in Appendix A.

Seven different feed combinations were prepared based on GTW feed ratio which is the ratio of VS from GTW to the sum of VS from MWS and GTW (GTW/(MWS+GTW)). Each combination had 4 reactors and 28 reactors were conducted in total. The contents of each combination are shown in Table 3.1. Each combination shown in Table 3.1 was diluted with deionized water up to 1 Liter before being used to fill into the digestion bottles. The first combination consisted of only INO in order to determine the contribution of INO to VS reduction and methane production. The second combination consisted of INO and MWS which was used as the reference (MWS alone digestion). The third combination consisted of INO and GTW in order to evaluate the performance of GTW alone digestion. The last 4 combinations consisted of INO, MWS and increasing amounts of GTW. The ratios of GTW to overall substrate addition (GTW/(GTW+MWS)) in the last 4 combinations were 14%, 24%, 32%, and 39% on a VS basis. Higher organic loading was avoided in order to prevent excessive gas production. Substrate:Inoculum (S:INO) ratio on a VS basis and percent of GTW in substrate were the major factors in determining the feed ratios. The S:INO ratio was selected to target the optimum level of 0.46 as reported by Liu and Buchanan (2011).

Table 3.1: Feed mixture combinations and loadings for each experiment.

Combination[a]	**INO (mL)**	**MWS (mL)**	**GTW (mL)**	**S:INO (VS basis)**	**GTW ratio**[b] **(%-VS basis)**
1	230	0	0	0	0
2	230	56	0	0.38	0
3	230	0	44	0.38	100%
4	230	56	7	0.44	14%
5	230	56	14	0.50	24%
6	230	56	21	0.56	32%
7	230	56	28	0.62	39%

Abbreviations: INO, inoculum; MWS, municipal wastewater sludge; GTW, grease trap waste; S:INO, substrate to inoculum ratio.

[a] Each combination was diluted with deionized water up to 1 L after prescribed volumes of substrates and inoculum were added.

[b] Calculated based on the volatile solids of substrates added, GTW/(GTW+MWS) (see text).

The fundamentals of the BMP test were used in this study (Hansen et al., 2004; Owen et al., 1979). The standard BMP test procedure was modified to periodically monitor the process parameters for each combination (Periodic/10-day testing). All the bottles in each combination were assumed to behave identically, since incubation conditions and feed ratio were maintained the same. Every 10 days one bottle from each combination was opened and all the tests were performed. Biochemical behavior of each combination was observed on day 0, day 10, day 20 and day 31. One bottle from each combination was incubated only for the first 10 days, another bottle was incubated for 20 days and the last two bottles were opened at the end of the incubation period, 31 days.

3.2.3 Data Analysis

A first order reaction equation (Eq. 1) and modified Gompertz equation (Eq. 2) were fitted to experimental methane potential results as a function of incubation time for each GTW feed ratio. Estimated kinetic and performance parameters were used to assist in interpretation of the BMP tests results and can be used as a first estimation of the parameters for more complex anaerobic digestion models such as ADM1 (Batstone et al., 2002). Least squares algorithm, using Matlab® 7.14 and SigmaPlot® 12.0, were used to perform the nonlinear regression analyses. Coefficient of determination (R^2) values were used to evaluate how well Eq. 1 and Eq. 2 fit the experimental data.

Degradation of MWS and GTW were assumed to follow a first order rate (Donoso-Bravo et al., 2010; Kabouris et al., 2007; Li et al., 2011). The ultimate methane potential/final cumulative net methane potential at the end of the incubation period (B_u, mL CH_4/g VS_{added}) and first order rate constants for methane potentials (k, 1/day) were estimated using (Eq. 1) where B is cumulative methane potential (mL CH_4/g VS_{added}) at time t (day).

$$B = B_u(1 - e^{-kt}) \qquad \text{Eq. 1}$$

There are several sigmoidal models, such as the modified Gompertz, Logistic and Tranference models, which have been used for modeling methane potential. The modified Gompertz model was used to estimate the performance parameters in this study because it was reported to be the best model to simulate the anaerobic digestion process (Altas, 2009; Donoso-Bravo & Fdz-Polanco, 2013). The modified Gompertz model (Eq.

2) was used to estimate the ultimate methane potential (B_u, mL CH_4/g VS_{added}), the length of lag phase (λ, h) and maximum methane potential rate (R_m, mL CH_4/g VS_{added}.h).

$$B = B_u exp\left\{-exp\left[\frac{R_m e}{B_u}(\lambda - t) + 1\right]\right\} \qquad \text{Eq. 2}$$

As available models describe anaerobic digestion performance as a function of time, e.g. sigmoidal models, they cannot be used to evaluate the effects of various GTW feed ratios on methane potential. Therefore, a mathematical equation was developed to describe methane potentials as a function of GTW feed ratio on a VS basis. The equation was developed to assist in evaluation of the effects of various GTW feed ratios on methane potential. Various sigmoidal non-linear models were fit to the experimental data, using SimaPlot 12.0®, to evaluate the goodness of fit. The 4-parameter Gompertz equation was determined to be the best describe the observed relationship between GTW addition and methane potential. The equation was modified to relate the equation parameters to the anaerobic digestion process. The equation does not estimate methane potentials at inhibiting GTW additions. A few experimental data points, starting from 0% GTW to inhibiting GTW addition, are required to perform the regression analysis and to evaluate how much the true (experimental) relationship is close to the estimated (modeled) relationship. The developed equation and definition of parameters are presented below.

$$B = B_i + (B_m - B_i) exp\left\{-exp\left[\frac{R_m e}{(B_m - B_i)}(X_o - X) + 1\right]\right\} \quad \text{(Eq. 3)}$$

- B: Methane potential at “X” GTW feed ratio, mL CH_4/g VS_{added}
- B_i: Initial methane potential (at 0% GTW addition) or methane potential of the major substrate at experimental/digester operation conditions, mL CH_4/g VS_{added}
- B_m: Maximum methane potential of substrate mixture at experimental/ digester operation conditions, mL CH_4/g VS_{added}
- R_m: Maximum methane potential rate, mL CH_4/g VS_{added} – % GTW
- e: Euler’s number, 2.7183
- X_o: The initial methane potential lag phase, during which methane potential remains relatively constant at increasing GTW ratios on VS basis, %
- X: GTW ratio on VS basis (GTW/(GTW+MWS)), %

3.2.5 Analytical Methods

Total solids (TS), volatile solids (VS), pH, alkalinity and COD were analyzed in accordance with Standard Methods (APHA et al., 2012). Total solids and VS were measured using the 2540 B and E methods; pH was measured using the 4500 B Electrometric Method; alkalinity was measured using the 2320 B Titration Method and COD was determined using the 5220 D Closed Reflux Colorimetric Method. Samples were centrifuged at 3900 rpm for 15 minutes. Supernatant was used in determination of

COD, NH_3-N, alkalinity and volatile acids (VA) concentrations. Chemetrics® (HR+) mercury free COD vials were used to measure COD concentrations. The test kit is able to measure COD concentrations between 0 and 15 g/L. pH was monitored using a pHmeter and dual junction pH/temperature electrode. Ammonia nitrogen concentration was measured using HACH® nitrogen-ammonia reagent test kit in accordance with the Salicylate Method. The test kit is able to measure NH_3-N concentrations between 0.4 and 50 mg/L. Dilution was applied in COD and NH_3-N measurements. Volatile acid concentration was determined using the Volatile Acid Alkalinity Method described in Operations Manual Anaerobic Sludge Digestion (EPA, 1976).

Gas production was measured 23 times and gas composition was analyzed 16 times during the 31 days of incubation. Gas production was measured daily during the first two weeks and later 2-3 times a week. The fundamentals of the gas collection procedure utilized were adopted from Owen et al. (1979). Produced gas was collected in 30 mL BD® luer lock disposable syringes equipped with 20-gauge needles. Methane concentration was measured with a gas chromatograph equipped with a thermal conductivity detector and two stainless steel Alltech® Hayesep Q80/100 column packed columns. Helium was used as the carrier. The methane potentials were calculated at standard temperature and pressure (P=1 atm and T=273.15 ^{0}K). The average volume of methane produced in the first combination bottles (INO alone digestion) was subtracted to calculate net methane potentials of substrates tested. The net methane potentials were presented at mL CH_4 per gram of VS added as total substrates (VS of MSW+GTW).

3.3 RESULTS AND DISCUSSION

3.3.1 Characteristics of the Materials.

The characteristics of the INO, MWS and GTW samples are shown in Table 3.2. Because INO consisted of digested sludge, the organic components (VS, COD and VA) in the INO had already been degraded. The inoculum consequently had low VS, COD and VA concentrations. Breakdown of proteins and amino acids during the digestion at the wastewater treatment plant might have caused the high alkalinity and NH_3-N concentrations in INO (Tchobanoglous et al., 2003). pH and alkalinity values in INO were high enough to raise the pH of mixtures up to the desired range, pH 6.8-7.2 (Girault et al., 2012). The MWS had a lower pH than the optimum pH range of anaerobic digestion process. However, its alkalinity concentration was within the optimum range of 2000 – 4000 mg/L $CaCO_3$ to compensate for its acidity (WEF, 1996). The organic (VS and COD) content in GTW was slightly higher than that in MWS, 55.94 g VS/L and 12.20 g COD/L, respectively. The organic content of GTW in this study was much lower compared to concentrated/dewatered GTW used in the previous studies (Davidsson et al., 2008; Kabouris et al., 2007; Lansing et al., 2010a; Li et al., 2011; Luostarinen et al., 2009; Pastor et al., 2013; Zhu et al., 2011). The GTW samples contained lower concentrations of alkalinity and pH compared to INO and MWS samples because of its acidic nature. Lipids are one of the major polysaccharides in GTW. When lipids degrade under anaerobic conditions, they yield glycerol and LCFAs (Hanaki et al., 1981). Anaerobic degradation of GTW in grease trap may also cause low pH values. Considering only pH, alkalinity and VA concentrations of the two substrates, MWS and GTW, it can be concluded that proper feed ratios must be maintained for anaerobic co-digestion of MWS and GTW. Otherwise the system can easily become acidic and fail.

Table 3.2: Characteristics of inoculum, municipal wastewater sludge and grease trap waste.

Parameter	INO[a]	MWS	GTW
Total Solids (g/L)	79.46±2.46[b]	83.56±1.04	83.50±1.28
Volatile Solids (g/L)	28.51±0.74	44.05±1.01	55.94±1.18
COD (g/L)[c]	1.32±0.05	11.96±0.08	12.20±0.09
pH	7.67	5.4	4.11
Alkalinity (mg $CaCO_3$/L)[c]	7025	3693.2	137.5
Volatile Fatty Acids (mg/L)[c]	137.5	5267.0	2587.5
Ammonia-N (mg NH_3-N/L)[c]	1524.38±1.58	623.17±2.49	122.04±0.65

[a] Abbreviations: INO, inoculum; MWS, municipal wastewater sludge; GTW, grease trap waste; COD, chemical oxygen demand.

[b] Mean±standard deviation (n≥3).

[c] Measured on the liquid portion of samples after centrifugation.

3.3.2 Methane Potentials and VS Reduction

Gas production and methane content were monitored throughout the entire incubation period. Overall VS reductions along with final net methane potentials are shown in Table 3.3. Approximately 90% of the total methane production was observed during the first 10 days for the combinations with high GTW ratios (32%, 39% and 100% GTW), while it was 83% for the reference (MWS alone digestion). Faster degradation of GTW compared to MWS is in agreement with the previous studies reported in the literature (Kabouris et al., 2007). The maximum lag phase of 3 days was observed at MWS alone digestion as well as at the 14% GTW and 100% GTW digestions. The minimum lag phase of 1 day was observed at the, 24%, 32% and 39% GTW digestions. These results indicate that the proper GTW addition ratio can speed up the digestion process and shorten the duration of the lag phase as compared to the digestion of MWS alone. On the other hand, excessive GTW additions may inhibit the digestion start-up period. It would be expected that co-digestion should improve anaerobic digestion performance due to the availability of various nutrients in co-substrate. For the first combination (INO alone digestion), most of the methane production was observed after the 10th day. Methane production was insignificant during the first 10 days as the inoculum mainly consisted of anaerobic microorganisms and depleted biodegradable materials. Increase in methane production after the 10th day may have originated due to cell lysis.

Increase in net cumulative methane potentials during the incubation period are presented in Figure 3.1. VS reductions were calculated based on the overall VS concentrations including INO, MWS and GTW. Overall VS reduction increased with increasing GTW addition. The highest VS reduction was observed in GTW alone digestion, 39% overall VS reduction. Total final VS concentration at GTW alone digestion was as low as at INO alone digestion. Similarly, total final VS concentrations at 14%, 24%, 32% and 39% were almost same as at MWS alone digestion. The results indicate that almost all of the VS from GTW were degraded because of the rapid degradation of GTW as mentioned earlier. The highest net methane potential of 606 mL CH_4/g VS_{added} was observed at 100% GTW digestion, while the methane potential from the digestion of MWS alone was 223 mL CH_4/g VS_{added}. Methane potentials at 14% and 24% GTW digestions were 214 and 233 mL CH_4/g VS_{added}, respectively, which are not very different than the digestion of MSW alone. Low volumes of GTW additions, 0.7% and 1.4% on a volume basis, had an insignificant impact on methane potentials. However, methane potentials for the 32% and 39% GTW digestions were double (470 and 517 mL CH_4/g VS_{added}, respectively) the levels observed with MWS alone digestion.

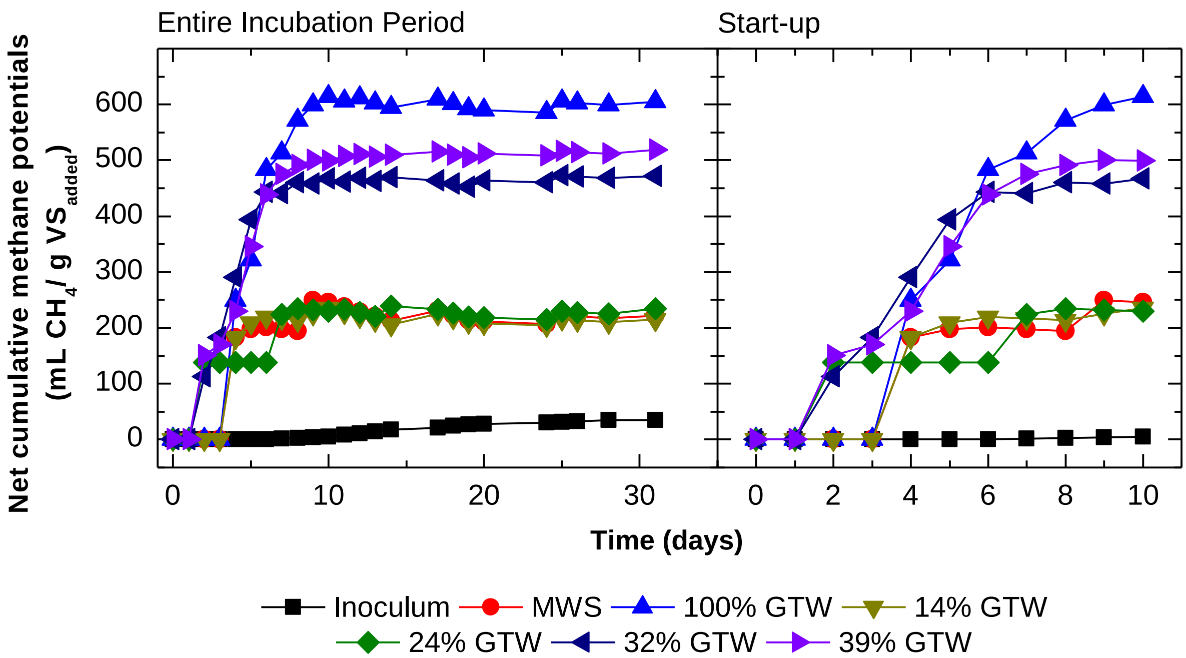

Figure 3.1: Methane potentials vs. time for the entire incubation and associated start-up periods. MWS, municipal wastewater sludge; GTW, grease trap waste.

Table 3.3: Process performances in terms of volatile solids (VS) reductions and methane potentials at increasing grease trap waste (GTW) feed ratios.

Parameter	**Combination**						
	1	**2**	**3**	**4**	**5**	**6**	**7**
GTW (%)	0	0	100	14	24	32	39
Total initial VS (g/L)[a]	6.56	9.02	9.02	9.42	9.81	10.20	10.59
Total final VS (g/L)[b]	5.05±0.28[c]	6.65±0.07	5.47±0.18	6.50±0.14	6.80±0.07	6.72±0.25	6.88±0.11
Overall VS reduction (%)	23	26	39	31	31	34	35
Methane production (mL)	27	93	206	101	118	233	278
Net methane potential (mL CH_4/g VS_{added}) [d]	-	223	606	214	233	470	517

[a] Overall nominal VS concentrations including inoculum, municipal wastewater sludge and GTW at the beginning of the incubation.

[b] Volatile solids concentrations at the end of 31-day incubation.

[c] Mean±standard deviation (n=3).

[d] Calculated at standard temperature and pressure. Methane production from inoculum was subtracted and the results were divided by volatile solids from substrates (see text).

Table 3.4 provides a comparison of methane potentials under mesophilic conditions (35 ^{0}C) observed in the present study to those reported in previous studies (Davidsson et al., 2008; Kabouris et al., 2007; Luostarinen et al., 2009; Zhu et al., 2011). The methane potential of the un-dewatered GTW (3rd combination) in the present study is less than the methane potentials of concentrated/dewatered GTW reported previously. However, the methane potential was more than two times greater than MWS digestion alone. The high water content of GTW could be the reason for the low methane potential of GTW in the present study compared to previous studies. Another important difference observed in the current study is that inhibition was not observed even at 100% GTW digestion. In contrast, inhibition was reported for a 70% GTW feed ratio on a wet weight basis and for a 81% GTW feed ratio on a VS basis in concentrated/dewatered GTW fed co-digestion studies (Luste et al., 2009; Zhu et al., 2011). These results suggest that AcoD of un-dewatered GTW may reduce the risk of inhibition compared to AcoD of concentrated/dewatered GTW. The high volume and alkalinity content of INO and low FOG content of GTW may have prevented process inhibition. Increase in methane potentials with GTW additions may be attributed to higher energy content of GTW and/ or enhancement in microbial activity.

Table 3.4: Comparison of the methane potentials from the present study and previous studies.

Reference	**VS of GTW**[a]	**T**	**V_{active}**	**Time**	**GTW loading**		**Methane Potential**[b]		
					Optimum	**Inhibiting**	**MWS**	**GTW**	**AcoD**
	(%)[c]	**(^{0}C)**	**(mL)**	**(day)**	**(% VS basis)**		**(mL CH_4/g VS_{added})**		
(Present Study, 2014)	5.6	35	120	31	100	no	223	606	470-517
(Zhu et al., 2011)	14	35	100	27	74	81	470	1050	800[d]
(Luostarinen et al., 2009)	25	35	60	50	50[e]	70[e]	263	918	435-788
(Davidsson et al., 2008)	17	35	500	37	60	no	325	845-928	425-681
(Kabouris et al., 2007)	37-41	35	120	120	29	no	291	878-993	406-494

Abbreviations: T, temperature; V_{active}, active volume; MWS, municipal wastewater sludge; GTW, grease trap waste; AcoD, anaerobic co-digestion; no, not observed.

[a] VS concentration of GTW used in the study.

[b] Methane potentials at MWS alone digestion, GTW alone digestion and co-digestion of MWS and GTW.

[c] Converted from mass/volume to percent values by assuming density of the samples is 1000 g/L.

[d] Approximate value

[e] % wet weight of GTW

3.3.3 Periodic/10-Day Testing

Observations from the 10-day tests provided additional insight into the process. Figure 3.2 shows the 10-day test results. The general behavior of the process can be summarized as decreasing trends in VS and VA and increasing trends in NH_3-N and alkalinity. A significant amount of total biodegradable VS were consumed during the first 10 days of incubation. An important decrease in VA also was observed in the first 10 days. Volatile acid utilization rate was slow for the rest of the incubation as compared to the first 10 days. The final VA concentrations were between 31 and 42 mg/L. These observations are in agreement with the drastic increase in methane production during the first 10 days of incubation. A rapid increase was observed in NH_3-N in the first 10 days except in the 1st (INO alone) and 3rd combinations (100% GTW). Increases in NH_3-N concentration continued at a slower rate during the rest of the incubation period. The increase in NH_3-N concentration was the result of the breakdown of proteins and organic nitrogen. Overall, a relatively faster decrease in substrate concentrations and increase in anaerobic digestion products were observed during the first 10 days of incubation period. Slower rates and steady state conditions were observed for the rest of the incubation compared to the first 10 days. Based on these results, it can be concluded that an important portion of digestion occurs in the first 10 days of incubation. Therefore, hydraulic residence times greater than 10 days should be maintained when continuous or semi-continuous feeding conditions are employed. The ammonia nitrogen concentration increased after the 10th day in the first combination. Breakdown of organic nitrogen may have caused an increase in ammonia nitrogen due to cell lysis. Alkalinity also showed an increasing trend with the major increase observed during the first 10 days. This increase may be explained by the mineralization of organic nitrogen into NH_4^+ that occurred during anaerobic digestion (Tchobanoglous et al., 2003). The increase in alkalinity was sufficient to keep the pH within the optimum range for anaerobic digestion.

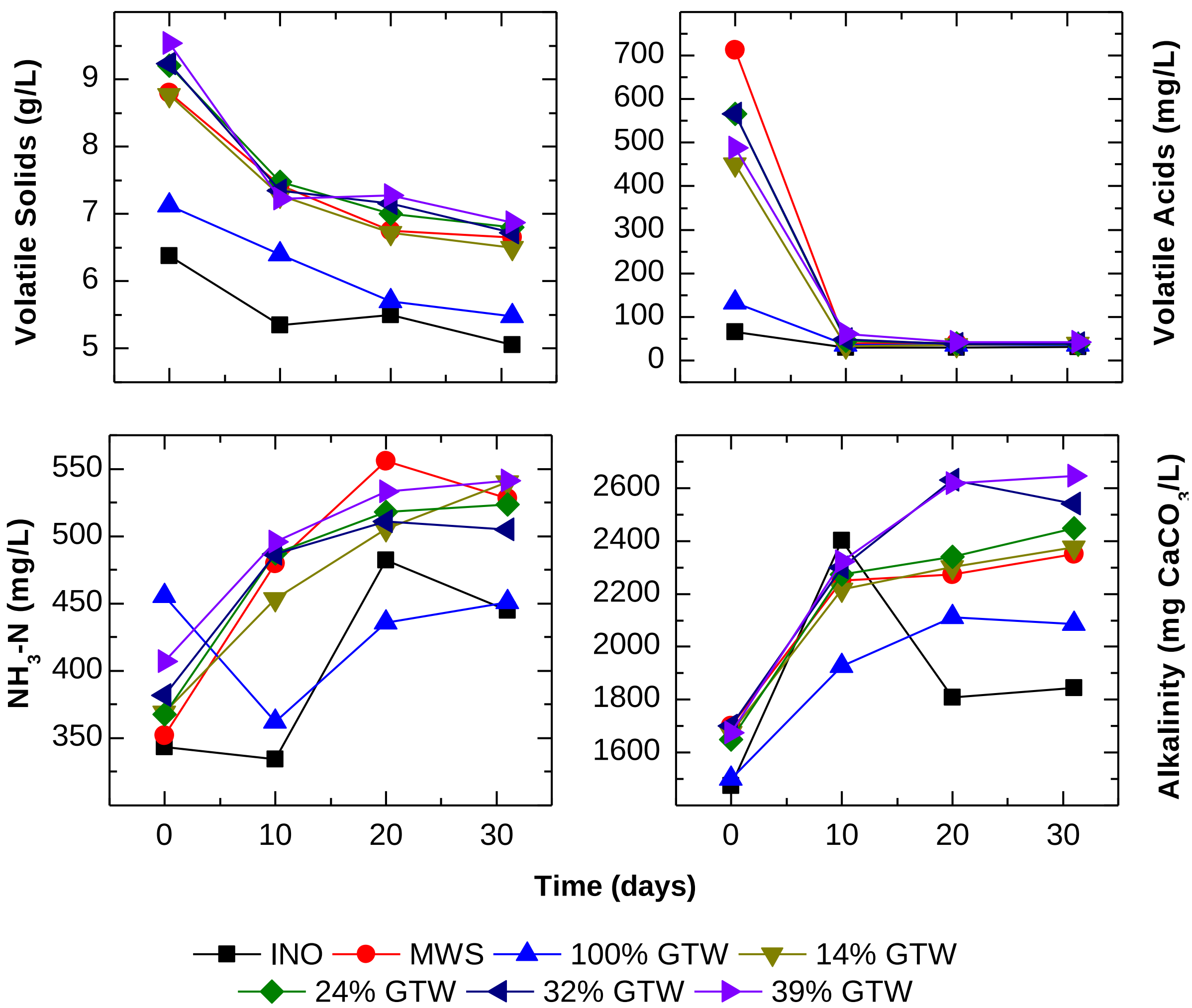

Figure 3.2: Periodic/10-day testing results for each combination. INO, inoculum; MWS, municipal wastewater sludge; GTW, grease trap waste.

3.3.4 Parameter Estimation

Non-linear regression of the first-order reaction equation and modified Gompertz equation for each GTW feed ratio are shown in Figure 3.3 and Figure 3.4. Markers represent experimental results, while model results are shown with solid lines in Figure 3.3 and 3.4. Estimated model parameters along with the coefficient of determination and experimental methane potentials are presented in Table 3.5.

Methane potentials followed approximately first order kinetics after lag phases of 1 to 3 days. The first-order reaction model over estimated the final cumulative methane potentials and was not able to simulate the lag phase at all. Therefore, it yielded a lower curve fitting performance compared to the modified Gompertz equation with R^2 values that varied between 0.8 and 0.94. The first order rate constants for methane potentials estimated in this study are in agreement with the literature values (Donoso-Bravo et al., 2010; Li et al., 2011). In general, the modified Gompertz model provided a better fit to the experimental data as compared to the first-order reaction model. The overall methane potential as a function of time, including lag phase, was successfully described for all combinations with R^2 values that varied between 0.94 and 0.99. The model estimates for the length of lag phase (λ) and maximum methane potential rate (R_m) were similar to the measured values for all feed combinations except for the 24% GTW digestion. In this experiment, there was an interruption in methane production between day 2 and day 6. Increase in methane potential started earlier for 24% GTW digestion compared to MWS alone digestion and 14% GTW digestion; however 24% GTW addition was not enough to maintain the increase in methane potential as observed in the 32%, 39% and 100% GTW digestions. Therefore, methane production stopped after the easily degradable GTW portion of the feed was digested. As expected, the estimated values of R_m were the lowest, while λ was the highest for the digestion of the inoculum alone. Estimated R_m and

λ values for MWS alone and 14% GTW digestions were similar because low volumes of un-dewatered GTW addition did not make a considerable change in the substrate mixture. The results support the findings that low GTW additions have an insignificant contribution to methane generation potential. The maximum methane potential rate was the highest for MWS alone digestion because of the drastic increase in methane production after the lag phase. Although GTW alone digestion provided the maximum methane potential, lag phase duration showed an increasing trend with increasing GTW addition in general which supports the possibility of process inhibition due to excessive GTW addition as mentioned earlier.

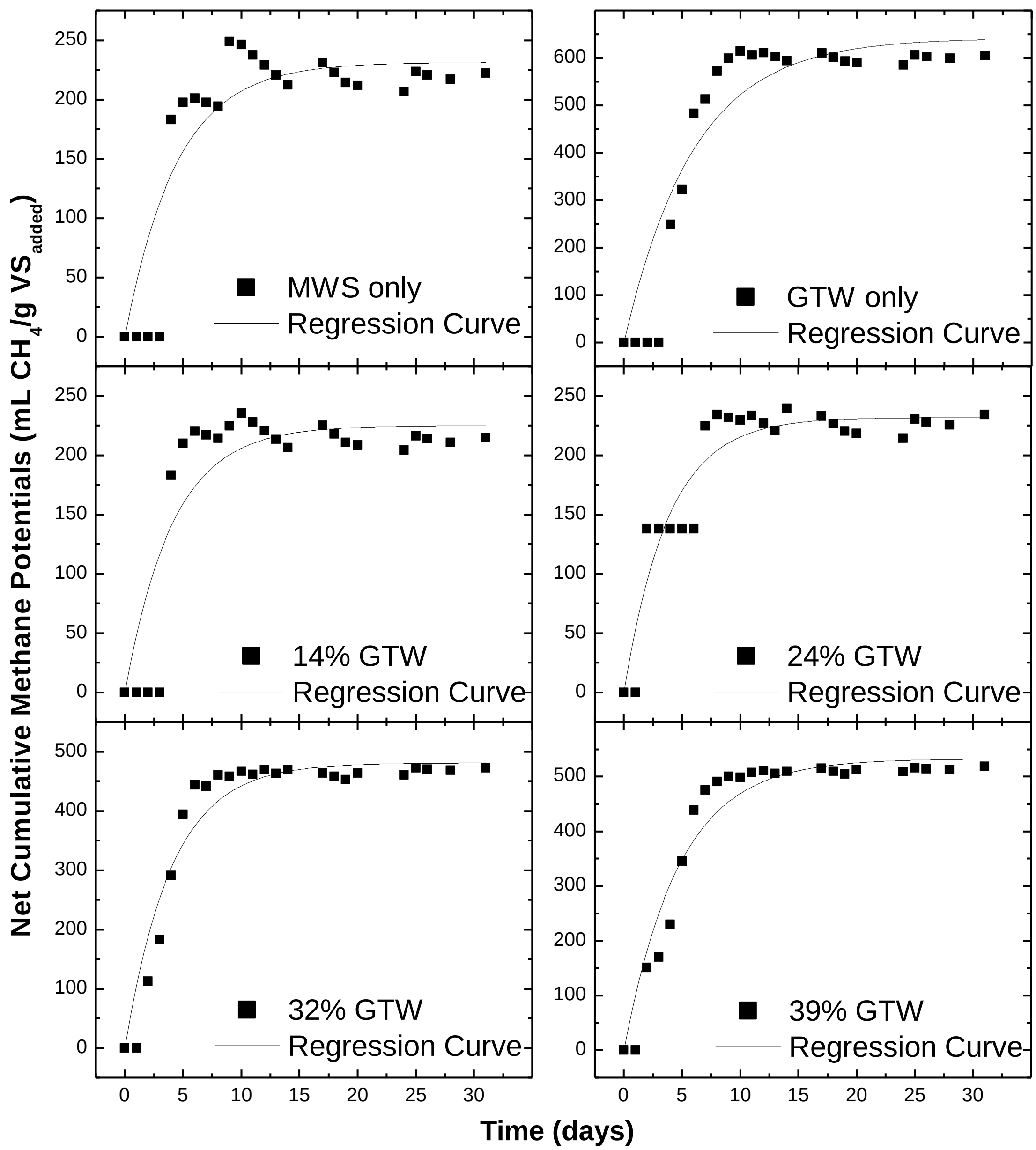

Figure 3.3: Non-linear regression of the first order reaction equation for cumulative methane potentials at each grease trap waste (GTW) feed ratio. MWS, municipal wastewater sludge.

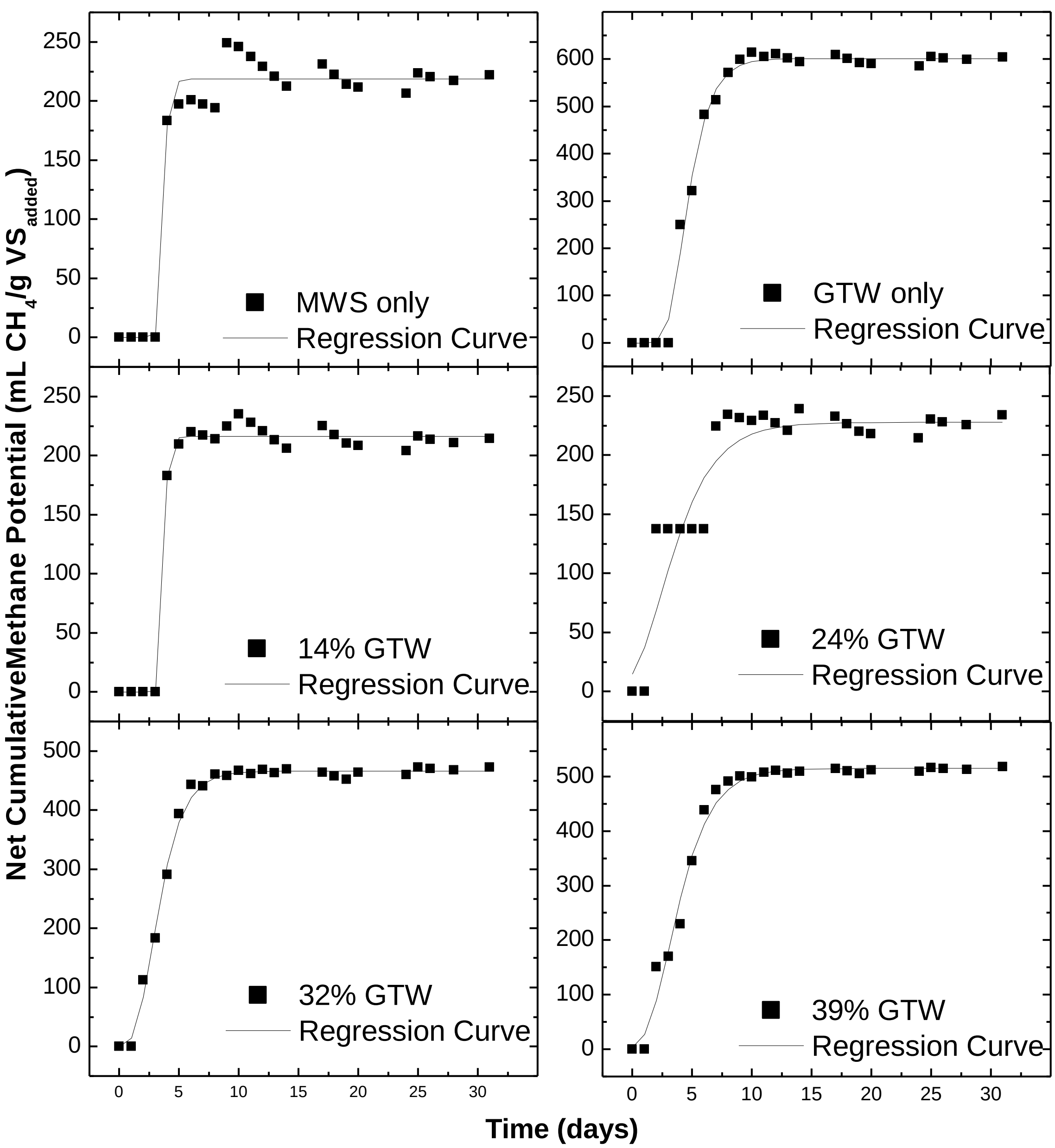

Figure 3.4: Non-linear regression of the modified Gompertz equation for cumulative methane potentials at each grease trap waste (GTW) feed ratio. MWS, municipal wastewater sludge.

Table 3.5: Comparison of experimental data and model parameters from non-linear regression.

	Experimental	Modified Gompertz Eq.				First Order Reaction Eq.		
	B_u (mL CH_4/g VS_{added})	**B_u (mL CH_4/g VS_{added})**	**λ (h)**	**R_m (mL CH_4/g VS_{added}.h)**	**R^2**	**B_u (mL CH_4/g VS_{added})**	**k (1/d)**	**R^2**
Inoculum	35	35	188.5	0.11	0.998			
MWS alone	223	219	75.5	10.40	0.986	231	0.226	0.795
GTW alone	606	602	69.1	7.14	0.996	642	0.168	0.863
14% GTW	214	216	76.7	11.44	0.996	225	0.246	0.775
24% GTW	233	228	0.0	1.43	0.941	232	0.265	0.894
32% GTW	470	466	32.4	5.07	0.997	481	0.252	0.929
39% GTW	517	515	27.6	4.10	0.993	533	0.213	0.939

Abbreviations: MWS, municipal wastewater sludge; GTW, grease trap waste; B_u, ultimate methane potential; λ, length of lag phase; R_m, maximum methane potential rate; R^2, coefficient of determination; k, first order rate constant.

Experimental results from the present study and two previous studies, on AcoD of MWS and concentrated/dewatered GTW, (Davidsson et al., 2008; Kabouris et al., 2008) were used to evaluate the how well the developed equation (Eq. 3) could be fit to the data. A least squares algorithm in Matlab® 7.14 was used to perform the nonlinear regression. The regression analysis was continued until the change in residuals was less than 1e-6. The equation results (solid lines) using Eq. 3 and experimental methane potentials (markers) are compared in Figure 3.5. The equation (Eq. 3) results fit very closely to the experimental results. Coefficient of determination values (R^2) were over 0.98 for all three studies. Smaller GTW feed ratios up to 1.7% on a volume basis did not make a considerable change in methane potential in the present study. The lower FOG content of un-dewatered GTW might have caused the initial methane potential lag phase (X_o) to be longer than that observed for concentrated/dewatered GTW digestions. Indeed, there is a clear lag phase observed in the present study. In the study by Kabouris et al. (2007), irregular methane potentials were observed between 14% and 20% GTW additions (Kabouris et al., 2007), because two different types of GTW, polymer and lime dewatered GTW, were used at different mixture ratios. The measured experimental and estimated model parameters acquired by fitting the data to Eq. 3 are shown in Table 3.6. In general, the model over estimated B_u and slightly under estimated B_i parameters. However, X_o and maximum methane potential rate (R_m) parameters closely matched the measured experimental values. The methane potential increased significantly after an initial methane potential lag phase for un-dewatered GTW digestion. As a result, the un-dewatered GTW digestion yielded the highest R_m value for GTW mixtures ranging from 24% to 39%. However, at a GTW feed ratio above 39%, the methane production rate plateaued. This could be due to the similar FOG content in 39% and 100% GTW digestion combinations. A continuous increase in methane potentials was observed with increasing GTW feed ratios for dewatered GTW digestions (Davidsson et al., 2008; Kabouris et al., 2007) In addition, the X_o and R_m values were similar for dewatered GTW digestions. Overall, the results indicate that a minimum FOG content might be needed before there is a measurable effect of GTW addition on methane potential. This may be the reason for the higher X_o and the drastic increase in methane potential (R_m) observed for un-dewatered GTW digestion as a function of GTW feed ratios; these lags and rapid increases in methane potential were not observed in the dewatered GTW digestion results.

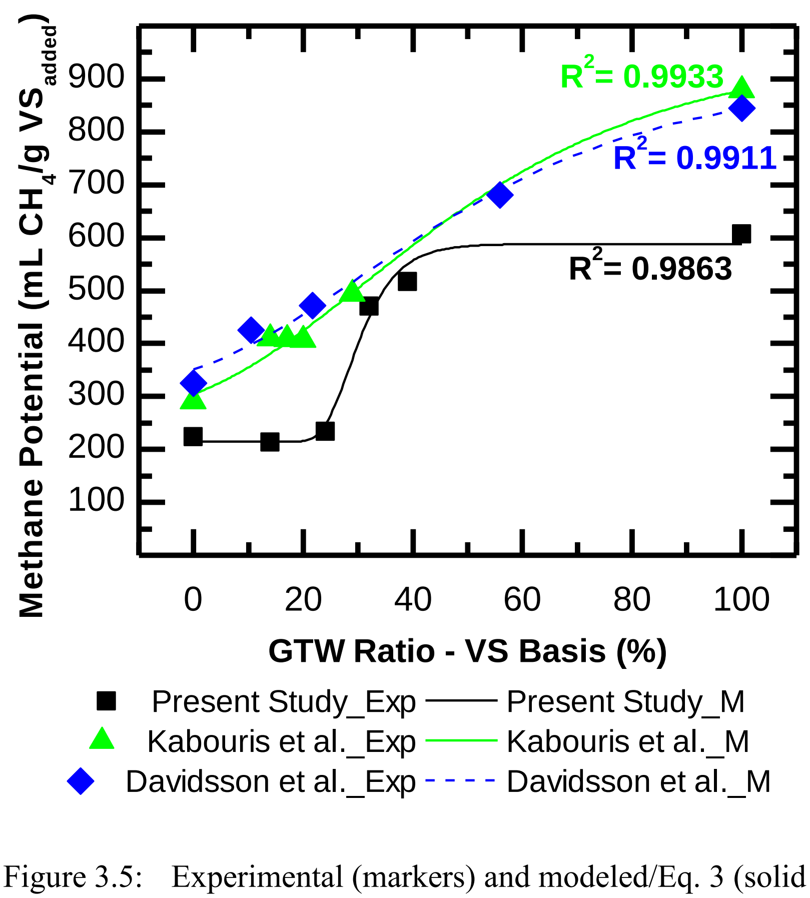

Figure 3.5: Experimental (markers) and modeled/Eq. 3 (solid lines) methane potentials vs. grease trap waste (GTW) feed ratios.

Table 3.6: Comparison of experimental and model (Eq. 3) parameters.

	Experiment				Model				
	B_i	B_u	X_o [a]	R_m [a]	B_i	B_u	X_o	R_m	R^2
	(mL CH_4/g VS_{added})		% GTW	(mL CH_4/g VS_{added}-%GTW)	(mL CH_4/g VS_{added})		% GTW	(mL CH_4/g VS_{added}-%GTW)	
Present study (2014)	215	605	24	29.8	215	588	23.5	28.7	0.9863
Davidsson et al. (2008)	325	845	0	9.5	313	902	0	7	0.9911
Kabouris et al. (2007)[b]	291	878	0	8.5	259	948	0	8.2	0.9933

[a] Estimated values based on experimental data.

[b] Bottle numbers 5, 7, 9, 10, 11 and 12 from the original paper were used in model evaluation.
Municipal wastewater sludge loadings were constant and dewatered sludge cake was not added in these bottles.
Polymer and lime dewatered GTW samples were used in this study.

3.4 CONCLUSIONS

The results of this phase of the study indicate that anaerobic co-digestion of MWS with un-dewatered GTW yields over two times more methane potential than anaerobic digestion of MWS alone Anaerobic digestion of un-dewatered GTW alone also resulted the much higher methane potential than anaerobic digestion of MWS alone. Inhibition or failure of the process was not observed even in the digestion of un-dewatered GTW alone. Anaerobic co-digestion of MWS and un-dewatered GTW may reduce the inhibition risk comparing to co-digestion of concentrated/dewatered GTW. Periodic/10-day testing provided a better understanding of the co-digestion process, however following a more frequent testing schedule in the first 10 days is suggested. The methane potential as a function of time and as a function of GTW feed ratio were measured. The experimental data was successfully fit with a modified Gompertz equation (Eq. 3) and yielded parameter estimates for maximum methane potential rate and the initial methane potential lag phase. The results indicate that monitoring FOG concentrations may provide insight into process performance and interpretation of the equation parameters. Fundamental information compiled from the batch experiments can be used to guide additional semi-continuous feed reactor experiments to evaluate the process performance and potential inhibition.

Chapter 4: Semi-Continuous Feed GTW Co-Digestion Experiments

ABSTRACT

Anaerobic co-digestion of municipal wastewater sludge (MWS) and un-dewatered grease trap waste (GTW) was investigated in terms of substrate degradation, methane potential and process inhibition in two laboratory scale, semi-continuous feed reactors under mesophilic conditions (35 ^{0}C). The performance and stability results observed in this study were compared with data from previous studies of co-digestion of dewatered GTW. The volatile solids content was less in the un-dewatered GTW than in the dewatered GTW. However, anaerobic digestion of dewatered and un-dewatered GTW resulted in similar methane generation potentials, methane content and volatile solids reduction. Successful co-digestion was possible for GTW feed ratios of up to 46% on a volatile solids basis. Co-digestion failed at the 70% GTW feed ratio. Comparison of stability data observed in this study with un-dewatered GTW and previous study results with dewatered GTW indicates that anaerobic co-digestion of MWS with un-dewatered GTW reduced the risk of inhibition compared to anaerobic co-digestion of MWS with dewatered GTW. Recovery of the digesters following upset conditions was achieved by feeding digester effluent from a full scale digester to the laboratory-scale co-digestion system for 10 days. Complete recovery of biogas production relative to that produced in the undisrupted digester was achieved in both reactors. The performance data collected in this study can be used as a guide for the development of full-scale applications in Austin and other municipalities.

4.1 INTRODUCTION

Discharge of fats, oils and grease (FOG) from food service establishments into the sewer systems is illegal in many states as a precaution to minimize the risk of line blockages (EPA, 2014). An increasing number of local POTW authorities are implementing management practices including the use of grease traps." (EPA, 2007). Grease traps physically separate FOG and solids such as food particles from kitchen wastewaters. Separated FOG and solids along with some kitchen wastewater are often called grease trap waste (GTW). GTW periodically is collected by private haulers and transferred to wastewater treatment plants (WWTPs) to feed into digesters. Since not all WWTPs accept GTW into their digesters, transferring GTW to further WWTPs is common practice. This practice requires a licensed transfer station, dewatering unit, chemical addition, discharge of high BOD filtrate into the municipal wastewater collection system, and creates extra carbon dioxide emission resulting from additional truck trafficking. Using raw/un-dewatered GTW at local WWTPs in close proximity to food service establishments may eliminate these additional steps in GTW management system. However, the effects of using un-dewatered GTW as the co-substrate instead of dewatered GTW on methane production and system inhibition have not been established.

Using dewatered/concentrated GTW as a co-substrate in anaerobic digesters treating municipal wastewater sludge (MWS) has been identified as an attractive option in previous studies. This process not only stabilizes the GTW, but also enables resource recovery due to the high methane potential of FOG in GTW (Davidsson et al., 2008; Kabouris et al., 2009b; Kabouris et al., 2008; Liu & Buchanan, 2011; Luostarinen et al., 2009; Razaviarani et al., 2013; Wang et al., 2013). However, inhibition and even failure of the digestion process have been reported when anaerobic digesters are fed with lipid rich GTW (Angelidaki & Ahring, 1992; Suto et al., 2006). Although accumulation of long chain fatty acids (LCFAs) resulting from feeding high concentrations/loading rates of GTW were identified as the source of inhibition in anaerobic co-digestion of MWS with GTW, the effects that co-digestion of MWS with un-dewatered GTW would have on methane production and system inhibition have not been established. Batch experiments presented in Chapter 3 reported the possibility of reducing the inhibition risk, while increasing methane potential and volatile solids degradation using un-dewatered GTW. Neither inhibition nor failure of the process was observed even at 100% un-dewatered GTW loading after 31 days of batch digestion.

The fundamental information on process behavior obtained during the batch experiments was used to design a semi-continuous feed digestion system that can more accurately simulate the digestion process typical of full-scale applications. Raw samples of MWS and un-dewatered GTW were anaerobically digested in semi-continuous feed reactors under mesophilic conditions (35 ^{0}C). The objective of the experiments was to determine the effects of un-dewatered GTW addition on co-digestion performance and stability under semi-continuous feed conditions. The results were compared with previously reported performance data for anaerobic co-digestion of MWS and concentrated/dewatered GTW. In addition, recovery of the digesters following system upset was evaluated.

4.2 Materials and Methods

4.2.1 Materials

Municipal wastewater sludge (MWS), digester influent, and untreated grease trap waste (un-dewatered GTW) were used as the substrates, while digester effluent before further thickening was collected as inoculum (INO) for the experiment. The anaerobic digestion inoculum (INO) and the MWS samples were obtained from the Austin Water Utility Hornsby Bend Biosolids Management Plant (HBBMP) in Austin, Texas; at this facility, solids consisting of primary sludge (PS) and waste activated sludge (WAS) are treated. The plant receives MWS from two WWTPs in Austin, TX. The MWS was screened and thickened before treatment in mesophilic (35 ^{0}C) anaerobic digestion (30-day). Grease trap waste was obtained from the transfer station of a GTW hauler in Austin, Texas. The hauler collects GTW from grease traps from restaurants, food courts, and food processing facilities. Grease trap waste samples for this study were collected before dewatering at the transfer station. The GTW is a new kind of waste stream; therefore, a comprehensive characterization was performed. Six separate samples of GTW that were collected during the experiment from different trucks/sources were tested to determine the characteristics of GTW. The samples were stored in a constant temperature room at 4 ^{0}C.

4.2.2 Experimental Setup

The anaerobic co-digestion experiments were conducted in two identical co-digestion systems in parallel. Each co-digestion system consisted of a digester (Reactor 1 (R1) and Reactor 2 (R2)), a gas collection unit and a liquid collection unit as shown in Figure 4.1. Detailed technical drawings and photos of the digestion system are presented in Appendix B. The digesters and gas collection units were constructed of acrylic, while the liquid collection units were made of high-density polyethylene. The volume of each digester was 6 L with a liquid volume of 5.25 L. Each gas collection unit had an 18 L volume and was fully filled with displacement liquid in order to reduce air entrance during gas release. The volume of each liquid collection unit was 19 L. The liquid collection units were maintained open to the atmosphere. A mechanical mixer with two wide blades was mounted on the top of each digester.

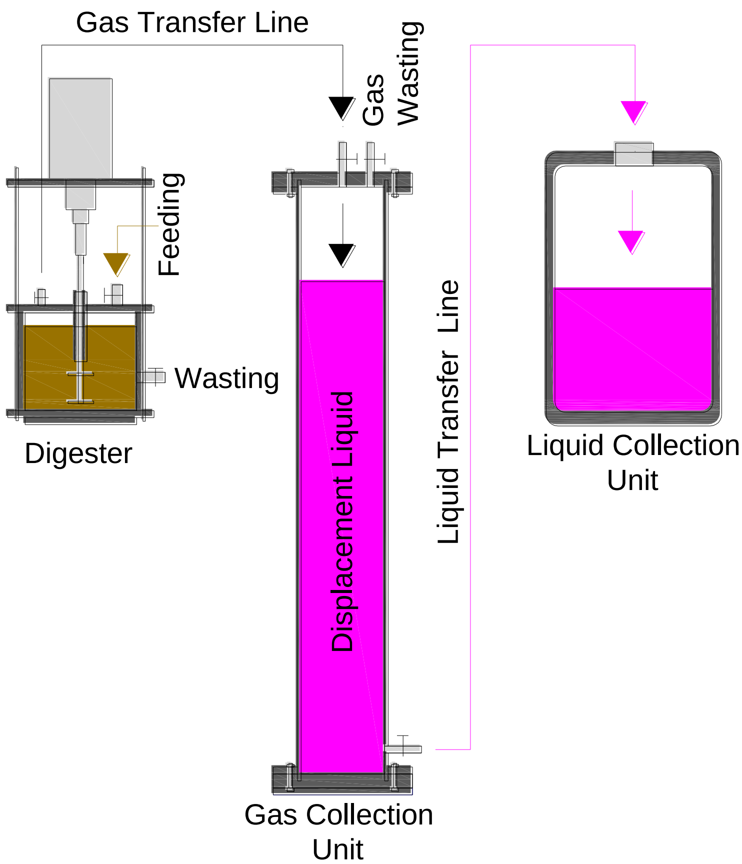

Figure 4.1: Schematic of experimental setup.

The digestion system was designed to operate in the simplest way possible. Feeding sludge and GWT was performed with a funnel through the feeding valve on top of the digester. The biogas generated in the digester moved from the digester to the gas collection unit through the vinyl gas transfer line. Biogas collected in the gas collection unit forces the displacement liquid into the liquid collection unit through the vinyl liquid transfer line until equilibrium with atmospheric pressure is reached. The volume of produced biogas was determined by measuring the volume of displaced liquid.

4.2.3 Experimental Procedure

The digestion systems were housed in a constant temperature room at 35 ^{0}C during the entire experiment in order to maintain mesophilic conditions. The volume of feed provided to the digester was the same as the volume of sludge wasted from the reactor daily. The hydraulic retention time (HRT) was maintained at 15 days. The organic loading rates (OLR) for each experiment are shown in Table 4.1. The daily feed mixture to the reactors was stored at the 35 ^{0}C for one hour to attain 35 ^{0}C before feeding the mixture into the digesters. Both digesters were operated at identical conditions; the conditions in terms of feed ratios, environmental, operational conditions. Operational specifications for each study phase including temperatures, OLR, HRT, durations and GTW feed ratios are presented Table 4.1.

The experiment was started up with 2 L INO in each digester and batch fed with MWS until the volume reached 5.25 L. Biogas production was not monitored during the start-up. The experiment was split into six (6) different phases based on GTW feed ratio, the ratio of VS from GTW to the sum of VS from MWS and GTW (GTW/ (MWS+GTW)). Only MWS was fed to the digesters during the first study phase which served as the reference condition. The GTW feed ratios were sequentially increased during the 2nd, 3rd and 4th phases. Recovery of the digesters following upset conditions was investigated during the 5th phase. Only MWS was fed to the digesters during the 6th phase; these results were compared with the 1st phase results in order to evaluate the performance of the system following recovery. In order to isolate the effect of changing the composition of the feed from a change in VS loading, the volatile solids concentrations of GTW and MWS in each study phase were maintained constant at 50 and 40 g/L, respectively. Samples were periodically taken from both feed and effluent from each reactor and all the tests (see Analytical Methods) were performed. During the first 3 phases, the tests were performed on samples collected on 16 separate days per each phase and less frequently during the last 3 phases. A more frequent testing schedule was applied during the 2nd HRTs as compared to the 1st HRTs in each phase because of the stability criteria. Criteria to determine when anaerobic co-digestion of MWS and GTW stabilizes have been proposed in previous studies by Creamer et al. (2010) and Liu and Buchanan (2011). More aggressive stability criteria were targeted in this study to simulate anaerobic co-digestion process closer to full-scale applications. The digesters were assumed to reach stability when the total solids (TS), volatile solids (VS), chemical oxygen demand (COD), pH, alkalinity, ammonia nitrogen concentrations and biogas production values lay within two standard deviations of the corresponding mean values for at least 5 consecutive days. After the system was determined to have stabilized the process parameters were used to calculate average results for each phase.

Table 4.1: Operational Specifications.

	Phase					
Parameters	**1**	**2**	**3**	**4**	**Recovery**	**6**
Temperature (^{0}C)	35	35	35	35	35	35
Active Volume (L)	5.25	5.25	5.25	5.25	5.25	5.25
Feed &Waste (mL/d)	350	350	350	350	350	350
OLR (g VS/L-d)	2.66 ± 0.13[a]	2.94 ± 0.09	2.93 ± 0.04	3.14 ± 0.02	1.73	2.53 ± 0.06
HRT (d)	15	15	15	15	15	15
Duration (d)	30	31	29	16	10	22
GTW-VS basis (%)	0	25	46	70	0	0
GTW-volume basis (%)	0	22	42	65	0	0

[a] Mean ± Standard Deviation.

4.2.4 Analytical Methods

Total solids (TS), volatile solids (VS), pH, alkalinity, chemical oxygen demand (COD) and sulfate (SO_4^{-2}) concentrations were analyzed in accordance with Standard Methods (APHA et al., 2012). TS and VS were measured using the 2540 B and E methods; pH was measured using the 4500 B Electrometric Method; alkalinity was measured using the 2320 B Titration Method, COD was determined using the 5220 D Closed Reflux Colorimetric Method and sulfate was measured using the 4500 E Turbidimetric Method. Samples were centrifuged at 3900 rpm for 15 minutes. Supernatants were used in determination of COD, ammonia nitrogen (NH_3-N), sulfate, alkalinity and volatile acids (VA) concentrations. Chemetrics® (HR+) mercury free COD vials were used to measure COD concentrations. The test kit is able to measure COD concentrations between 0 and 15 g/L. pH was monitored using a pHmeter and dual junction pH/temperature electrode. Ammonia nitrogen concentration was measured using HACH® nitrogen-ammonia reagent test kit in accordance with the Salicylate Method. The test kit is able to measure NH_3-N concentrations between 0.4 and 50 mg/L. Dilutions were applied in COD, NH_3-N and SO4-2 measurements. VA was determined using the Volatile Acid Alkalinity Method described in Operations Manual Anaerobic Sludge Digestion (EPA, 1976). The acidic salt solution used as the displacement liquid to minimize the dissolution of biogas in the liquid (especially carbon dioxide), was prepared in accordance with Standards Method 2720 B. 3. d. (APHA et al., 2012). Gas production was measured daily. Methane content was measured with a gas chromatograph equipped with a thermal conductivity detector and two stainless steel Alltech® Hayesep Q80/100 column packed columns. Helium was used as the carrier.

4.3 Results and Discussion

4.3.1 Characteristics of the Materials.

The characteristics of the untreated samples are presented in Table 4.2. INO had lower VS, COD and VA concentrations as compared to the MWS and GTW samples because the organic components in the INO had already been degraded during the digestion. The alkalinity concentration in the INO was the highest amongst all samples due to the mineralization of organic nitrogen into ammonium ion (NH_4^+). Consequently, INO had the highest pH among the samples. The VS and COD concentrations in GTW were slightly higher than that in MWS, 49.69 ± 15.21 and 13.4 ± 4.2 g/L, respectively. The GTW samples had the greatest degradable solids content (VS/TS) 92 ± 3%, compared to 75 ± 11% and 56 ± 17% in MWS and INO, respectively. pH and alkalinity concentrations in the GTW samples were the lowest as a result of the acidic nature of GTW sample. The VA/Alkalinity ratio was much higher in the GTW samples. Although the MWS samples had a pH lower than the optimum range, pH 6.8-7.2 (Gerardi, 2003) and higher VA concentration, the alkalinity in the MSW was sufficient to initiate the anaerobic digestion process. The SO_4^{-2} and NH_3-N concentrations in the GTW and MWS samples were lower than INO. The most significant characteristic of the GTW samples was the high variation observed in each measured parameter. These results indicate that the GTW characteristics can vary considerably depending on the source and it will be important to maintain a proper GTW feed ratio to the anaerobic co-digestion process to prevent the system from becoming acidic and failing.

Table 4.2: Characteristics of inoculum (INO), municipal wastewater sludge (MWS) and un-dewatered grease trap waste (GTW).

Parameters	GTW Mean ± Stdev	MWS Mean ± Stdev	INO Mean ± Stdev
TS (g/L)	53.78 ± 23.27[a]	60.16 ± 13.28	53.33 ± 22.71
VS (g/L)	49.69 ± 15.21	44.22 ± 5.03	27.07 ± 1.29
VS/TS (%)	92 ± 3	75 ± 11	56 ± 17
COD (g/L)	13.4 ± 4.2	12 ± 0.5	2.24 ± 0.92
pH	4.71 ± 0.52	5.43 ± 0.24	7.72 ± 0.1
Alkalinity (mg $CaCO_3$/L)	965 ± 816	3098 ± 419	6219 ± 753
VA (mg/L)	2834 ± 895	5151 ± 371	139 ± 2
VA/Alkalinity	4.19 ± 3.15	1.68 ± 0.23	0.04 ± 0.03
Ammonia-N (mg NH_3-N/L)	177 ± 157	485 ± 138	1464 ± 88
Sulfate (mg SO_4^{-2}/L)	21	21.07	93

[a] Mean ± Standard Deviation. N=6, 6, 3 for GTW, MWS and INO, respectively.

4.3.2 Assessment of the Process Performance

The change in daily biogas production and methane content with increasing GTW feed ratios are shown in Figure 4.2. Biogas production is shown with 90% confidence intervals to provide a better assessment of gas production stability. Similar biogas production and methane content trends were observed in both reactors indicating that the anaerobic digestion process was well-controlled. Biogas production increased with increasing GTW feed ratios up to 46% on a VS basis. Similarly, the methane content of the biogas also increased with increasing GTW feed ratio up to the GTW feed ratio of 46%. However, biogas production started to decrease immediately after feeding at the 70% GTW ratio and eventually ceased. The immediate decrease in biogas production after feeding at the 70% GTW indicates that the reactors were already at the maximum GTW feed limits at 46%.

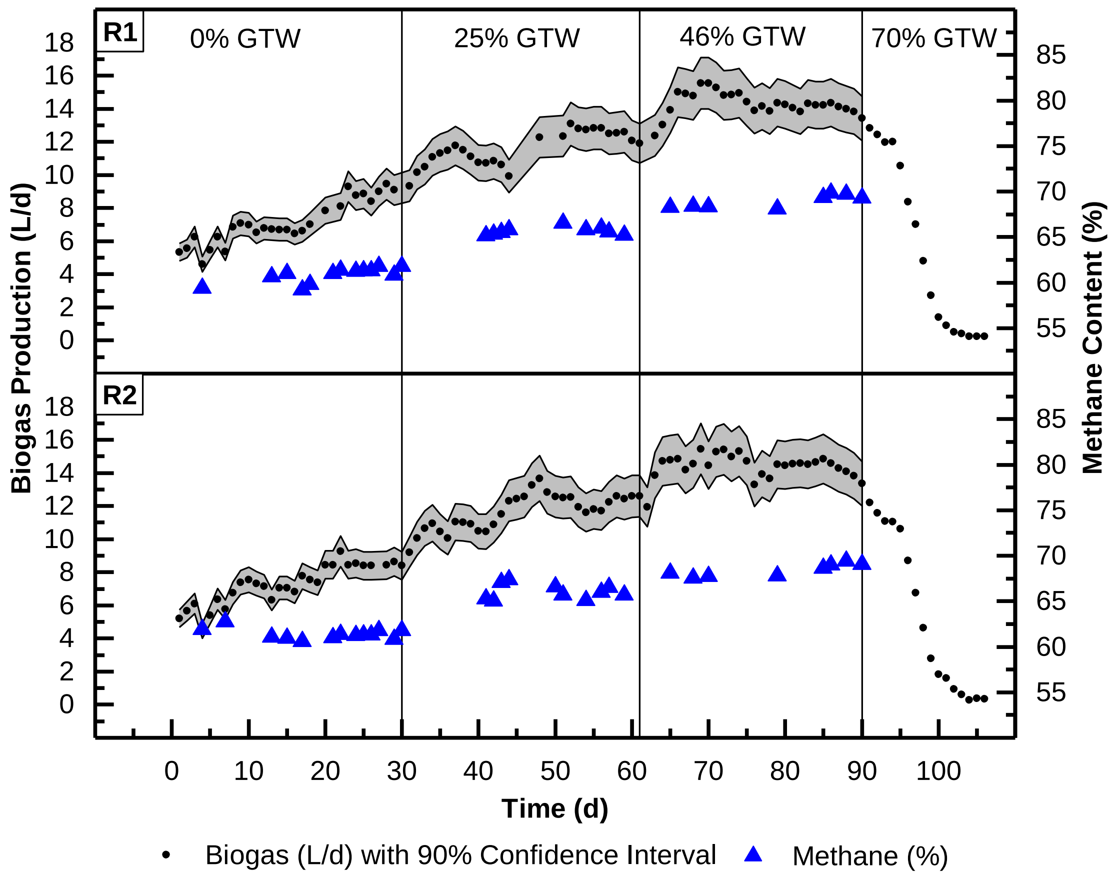

Figure 4.2: Change in daily biogas production and methane content in R1 and R2 over time.

The averaged performance of the two reactors after the system was determined to have stabilized is presented in Figure 4.3. Biogas production increased from 8.75 L/d to 12.5 L/d and 14.45 L/d at 0%, 25% and 46% GTW feed ratios, respectively. Similarly, the methane content increased from 61% to 66% and then to 69% at 0%, 25% and 46% GTW feed ratios, respectively. VS reduction increased from 47% to 50 and 59% at 0, 25 and 46% GTW feed ratios, respectively.

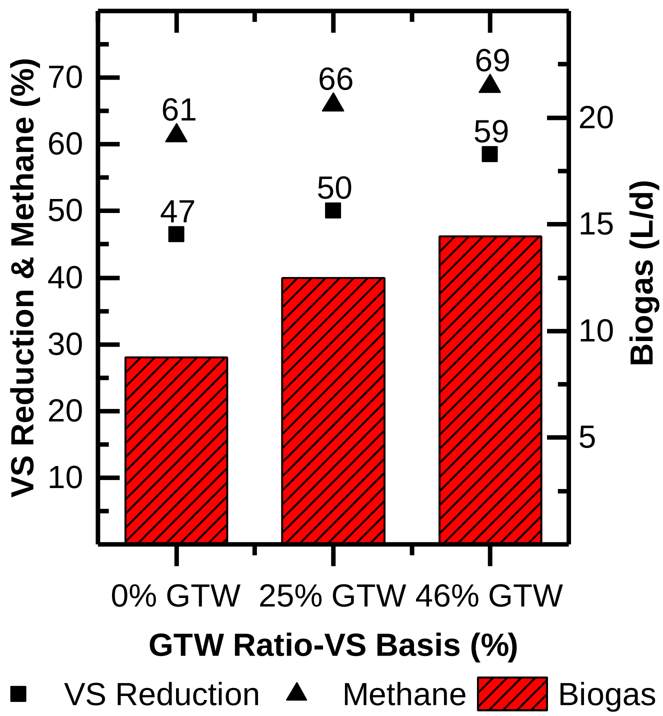

Figure 4.3: The average daily biogas production, methane content and VS reduction at 0%, 25% and 46% GTW feed ratios on VS basis.

Methane potential also increased as a result of the increase in both biogas production and methane content (Table 4.3). The methane yield increased from 384 mL to 536 mL to 641 mL CH_4/ g VS_{added} (average of the two reactors) at 0%, 25% and 46% GTW feed ratios, respectively. The results indicate that anaerobic co-digestion of MWS with un-dewatered GTW up to 46% GTW feed ratio not only increased the biogas production, but also increased the quality of gas and stabilized more organics. The increase in digestion performance up to a certain amount of GTW (46% on VS basis) can be attributed to the high methane potential and biodegradability of GTW.

Table 4.3: Stability and performance parameters in feed, R1 and R2 at 0, 25, 46 and 70% GTW feed ratios.

	Phase 1 (0% GTW)			**Phase 2 (25% GTW)**			**Phase 3 (46% GTW)**			**Phase 4 (70% GTW)**		
Parameters	**Feed**	**R1**	**R2**	**Feed**	**R1**	**R2**	**Feed**	**R1**	**R2**	**Feed**	**R1**	**R2**
TS (g/L)	50.4 ± 1.1[a]	31.9 ± 1.3	33.8 ± 2.4	51.7 ± 1.6	31.1 ± 1.2	28.8 ± 0.7	50.9 ± 0.6	25.6 ± 0.5	24.4 ± 0.9	52 ± 0	35.4 ± 0.1	37.4 ± 1.5
VS (g/L)	40 ± 2	21.3 ± 1.1	22.6 ± 1.4	44 ± 1.4	22.8 ± 1.1	21 ± 0.6	44.3 ± 0.6	18.7 ± 0.7	18.1 ± 0.5	47.2 ± 0.2	29.3 ± 0	31.1 ± 1.3
VS/TS (%)	79	67	67	85	73	73	87	73	74	91	83	83
COD (g/L)	13.3 ± 1.1	2.1 ± 0.3	2.4 ± 0.2	18.9 ± 3.6	3.2 ± 0.3	3 ± 0.3	14.7 ± 5.1	2.7 ± 0.3	2.6 ± 0.1	11.8 ± 1.5	12.4 ± 0.2	12.5 ± 1.2
pH	5.37 ± 0.03	7.63 ± 0.04	7.68 ± 0.04	5.34 ± 0.02	7.59 ± 0.04	7.63 ± 0.06	5.32 ± 0.02	7.54 ± 0.05	7.56 ± 0.04	5.21 ± 0.01	5.47 ± 0.13	5.53 ± 0.14
Alkalinity (mg $CaCO_3$/L)	2611 ± 63	5413 ± 128	5619 ± 120	2620 ± 30	5040 ± 47	5070 ± 65	2302 ± 64	4764 ± 40	4820 ± 48	2038 ± 53	3038 ± 124	3131 ± 115
VA (mg/L)	4572 ± 187	142 ± 21	389 ± 164	5017 ± 181	471 ± 117	678 ± 192	4511 ± 112	535 ± 103	607 ± 131	4369 ± 186	5466 ± 40	5569 ± 80
VA/Alkalinity	1.75 ± 0.05	0.03 ± 0.02	0.07 ± 0.03	1.91 ± 0.06	0.09 ± 0.02	0.13 ± 0.04	1.96 ± 0.06	0.11 ± 0.02	0.13 ± 0.03	2.14 ± 0.03	1.8 ± 0.09	1.78 ± 0.09
Ammonia-N (mg NH_3-N/L)	477 ± 28	1220 ± 48	1259 ± 63	495 ± 35	1179 ± 96	1263 ± 53	395 ± 21	1040 ± 46	1054 ± 91	355 ± 33	871 ± 27	993
Sulfate (mg SO_4^{-2}/L)	nm[b]	nm	nm	57 ± 23	56 ± 26	56 ± 22	30 ± 18	79 ± 17	56 ± 15	10 ± 3	51 ± 30	59
VS reduction (%)	N/A[c]	48 ± 3	45 ± 3	N/A	48 ± 2	52 ± 2	N/A	58 ± 2	59 ± 1	N/A	38 ± 0	34 ± 3
Biogas (L/d)	N/A	9 ± 0.4	8.5 ± 0.3	N/A	12.6 ± 0.3	12.4 ± 0.5	N/A	14.4 ± 0.6	14.5 ± 0.6	N/A	Failed	Failed
Methane Content (%)	N/A	61.3 ± 0.3	61.3 ± 0.3	N/A	65.6 ± 0.4	66.1 ± 0.7	N/A	68.9 ± 0.7	68.4 ± 0.7	N/A	Failed	Failed
Methane Yield (mL CH_4/ g VS_{added})	N/A	393 ± 15	374 ± 12	N/A	536 ± 15	535 ± 22	N/A	641 ± 25	640 ± 25	N/A	Failed	Failed

These are the values collected during the steady state periods of each phases and averaged.
[a] Mean ± Standard Deviation.
[b] Not measured.
[c] Not applicable.

4.3.3 Assessment of the Process Stability

Periodic monitoring of process stability provides insight into process performance. Observations on stability and performance parameters of the feed as well as the effluent from R1 and R2 at steady state are shown in Table 4.3. The NH_3-N concentration in the reactors decreased with increasing GTW addition because of the low NH_3-N concentrations in the GTW. The NH_3-N concentrations increased after digestion at all GTW feed ratios resulting from the breakdown of proteins and amino acids. However the increase in NH_3-N was not high enough to cause inhibition or failure of the process. Sulfate concentrations during the experiment also were not high enough to outpace the conversion efficiency of sulfate reducing bacteria which can compete with methanogens.

The change in Alkalinity and VA concentrations in feed, R1 and R2 with increasing GTW additions up to a 70% GTW feed ratio (on a VS basis) are shown in Figure 4.4. Although the VA concentrations in the feed were not significantly different at increasing GTW loading ratios, a decreasing trend in alkalinity concentration was observed because of the low alkalinity content of the GTW. The sudden decrease in alkalinity observed on day 17 was a result of a fresh feed provided that day. The decrease in alkalinity at increasing GTW loading ratios was much greater in the reactors than in the feed because of acidic fermentation in the digesters. Small increments in VA concentrations were observed in both reactors at increasing GTW feed ratios up to 46%. However, the VA concentration in the reactors peaked (5466 ± 40 and 5569 ± 80 mg/L in R1 and R2, respectively) at the 70% GTW feed ratio. Successful digestion was observed with feeds containing a VA/Alkalinity ratio of up to a 1.96 ± 0.06. However, a feed with VA/Alkalinity ratio of 2.14 ± 0.03 led to VA accumulation and consumption of the buffering capacity of the system (alkalinity). The VA/Alkalinity ratio in the reactors increased from 0.12 to 1.79 which is substantially greater than the suggested digester stability limits (0.3-0.4) (El-Fadel et al., 2013). Consequently the process pH decreased from 7.55 to 5.5 (average of two reactors) and the digesters failed (Table 4.3). These results indicate that feeds with VA/Alkalinity ratios greater than 2 are not suitable when co-digesting MWS with un-dewatered GTW.

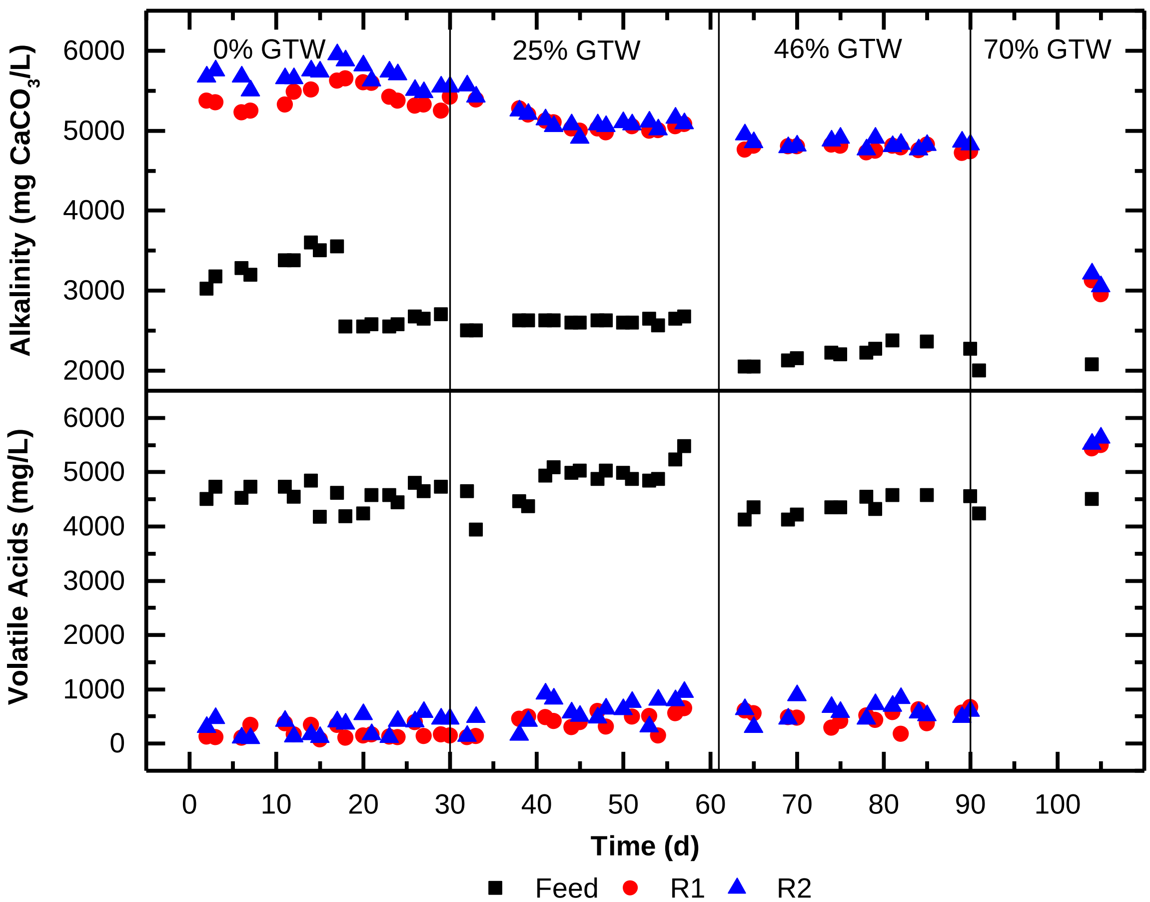

Figure 4.4: Change in alkalinity and volatile acid concentrations in feed, R1 and R2 over time.

4.3.4 Digester Recovery

In previous studies by Wan (2011) and Wang (2013), three methods were evaluated for their efficacy in recovering digester function following failure resulting from excessive GTW loadings: (1) addition of NaOH, (2) batch digestion and (3) switching the feed from co-digestion conditions to mono-digestion conditions. Addition of NaOH on a single day increased the pH from 5.6 to 7.72; however, biogas production did not recover (Wan et al., 2011). After 2 weeks of batch digestion, the pH increased from 5.6 to 6.3 with the second method. Batch digestion was continued for 40 days. Mono-digestion was performed by feeding only the major substrate after 40 days of batch digestion mode; however, the biogas production level only recovered to about half of the biogas production level of the undisrupted digestion (Wan et al., 2011). However, in a different study by Wang (2013), switching the feed from co-digestion conditions to mono-digestion conditions by feeding only the major substrate for 31 days was more successful. The pH increased from 6.6 to 6.9, the alkalinity increased from 3.64 to 4.29 g $CaCO_3$/L, and the biogas production and methane content reached levels similar to the undisrupted digestion values (Wang et al., 2013). When recovery was started (at pH 6.6) immediately after the first signs of digester inhibition as in the last study, full recovery was maintained. On the other hand, when the feed at excessive GTW loadings was continued (until pH 5.6) even after inhibition started as in the first two methods, biogas production did not recover. Therefore, some of the microbial activity was permanently damaged and a full recovery was not maintained.

In the current study, semi-continuous feed of digester effluent from a working digester into the failed digesters was evaluated as a recovery method. Once a day, 0.35 L of HBBMP digester effluent was fed into each reactor to maintain balanced conditions and 0.35 L of digested material was also discharged from each reactor during the 10-day recovery period. 10-day recovery period was followed by MWS alone feeding for 22 days as shown in Figure 4.5. The characteristics of feed, R1 and R2 at the end of the recovery phase and after 15 days (1 HRT) of MWS alone feeding (Phase 6) are shown in Table 4.4. The pH gradually increased from 5.5 to 7.24 and the VA/Alkalinity ratio declined from about 1.79 to 0.58 in the reactors by the end of the 10-day recovery period. Although the VA/Alkalinity ratios within the reactors remained greater than the digester stability limits (0.3-0.4) suggested by (El-Fadel et al., 2013) by the end of the 15 day of feeding MWS alone (Phase 6), a decrease in COD, VS and VA concentrations were achieved. An average of 8.75 L/d biogas production, 62% methane content and 413 mL CH_4/ g VS_{added} methane potential were observed towards the end of Phase 6 indicating that a healthy microbial activity was present. Although the biogas production had ceased as a result of the digester failure, 100% recovery of the undisrupted digester's methane production was observed in the present study as a result of the addition of the digester effluent feed. This recovery can be attributed to the addition of new microbial mass with the digester effluent feed. These results indicate that semi-continuous feed of digester effluent into upset digesters could provide a faster and more effective recovery as compared to the methods proposed previously in the literature.

During the 10-day recovery-period in this study, the effluent discharges from each reactor were stored in separate plastic containers under mesophilic conditions (35 ^{0}C). The characteristics of the stored effluents were determined and are presented in Table 4.4. Since the contents from the failed digester were diluted with HBBMP digester effluent during the 10-day recovery period, the effluents from the recovering digesters would be suitable for further digestion based on VA/Alkalinity ratio and pH levels. The main disadvantage of using a feeding digester effluent as a recovery method is the requirement to store the effluent from the unstable digester until the recovery is achieved.

At that point, the stored sludge can be fed back into the digesters for further digestion. This method requires that additional storage be available.

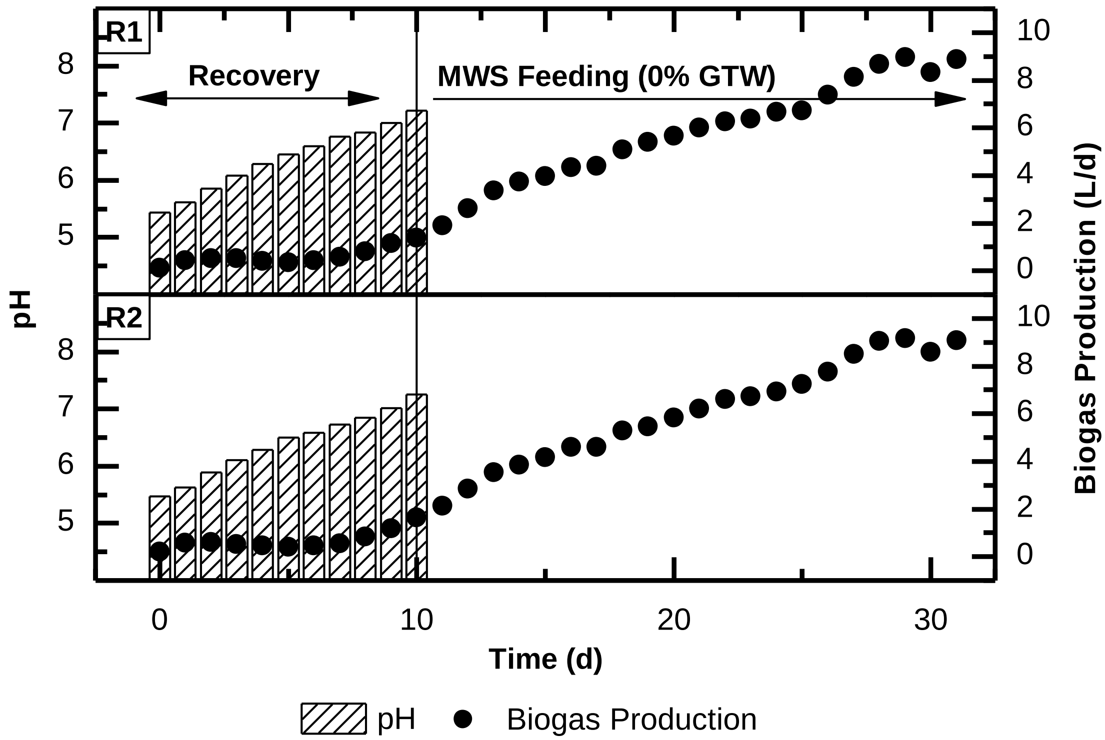

Figure 4.5: Daily biogas production and increase in pH in the digesters during the upset recovery and MSW-feed periods.

Table 4.4: Stability and performance parameters in feed, R1, and R2 during the recovery phase (digester effluent feeding), 6th phase (MWS alone feeding) and the characteristics of the stored effluents during the recovery.

	Phase 5 (Recovery)			**Phase 6 (0% GTW)**			**Recovery Effluent**	
Parameters	**Feed**	**R1**	**R2**	**Feed**	**R1**	**R2**	**R1**	**R2**
TS (g/L)	38.4	33.5	33.3	48.9 ± 1[a]	35.5 ± 1.6	35.2 ± 0.3	32.7	36.3
VS (g/L)	26.0	25.0	25.0	38 ± 0.9	25.5 ± 1.2	25.2 ± 0.1	25.3	29.0
VS/TS (%)	68	75	75	78	72	72	78%	80%
COD (g/L)	2.2	9.2	8.6	12.1 ± 0.6	6.8 ± 0.2	6.9 ± 0.1	10.20	10.48
pH	7.65	7.22	7.25	5.82 ± 0.04	7.48 ± 0.07	7.49 ± 0.03	7.07	7.07
Alkalinity (mg $CaCO_3$/L)	6100	4950	5050	2513 ± 796	3325 ± 18	3341 ± 58	4250	4300
VA (mg/L)	413	2813	2981	4772 ± 119	1744 ± 186	1838 ± 0	4069	4294
VA/Alkalinity	0.07	0.57	0.59	1.99 ± 0.58	0.52 ± 0.06	0.55 ± 0.01	0.96	1.00
Ammonia-N (mg NH_3-N/L)	1364	1396	951	536 ± 48	1169 ± 350	1345 ± 16	1013	1277
Sulfate (mg SO_4^{-2}/L)	93	67	88	27	43 ± 4	82 ± 15	78	34
VS reduction (%)					33 ± 1	34 ± 2		
Biogas (L/d)					8.6 ± 0.4	8.9 ± 0.3		
Methane Content (%)					62 ± 1.1	62 ± 1		
Methane Yield (mL CH_4/ g VS_{added})					405 ± 16	420 ± 14		

[a] Mean ± Standard Deviation.

4.3.6 Comparison with Previous Studies

Results from the present study as well as previous studies are shown in Table 4.5. The highest methane potentials without inhibition or failure of the process were obtained between 20% and 68% GTW additions when co-digesting with dewatered GTW (Davidsson et al., 2008; Kabouris et al., 2009b; Liu & Buchanan, 2011; Luostarinen et al., 2009; Wang et al., 2013), while the observed methane potentials ranged between 46% and 64% GTW additions when co-digesting with un-dewatered GTW (see Table 4.5). The GTW feed ratio that yielded successful digestion was increased by using un-dewatered GTW. The results of this study indicate that using un-dewatered GTW as the co-substrate yields more stable digestion at lower feed ratios (<46% on VS basis) as compared to dewatered GTW. This result supports the hypothesis that using un-dewatered GTW instead of dewatered GTW can reduce the inhibition risk of anaerobic co-digestion of MWS and GTW. Although the water content of un-dewatered GTW is higher than dewatered GTW, co-digestion of un-dewatered GTWs results in higher methane potentials than that reported in most previous studies of co-digested dewatered GTW (see Table 4.6). Significant increase in methane potential was reported in one study by Wang et al. (2013). The unusual increase reported by Wang et al. (2013) could be due to the pre-treatment applied to the GTW before it was fed into the digesters and the low methane potential of the major substrate (TWAS). In that study, the GTW was separated into components: FOG, food particles and associated wastewater. Food particles were mixed with a blender to homogenize. Feed GTW contained 10% FOG, 40% food particles and 50% wastewater by volume. The increase in methane content and VS reduction during co-digestion of both dewatered and un-dewatered GTW were also similar.

Table 4.6: Comparison of results from present and previous studies.

Reference	Substrates (MWS + GTW)	GTW loading		OLR	VS Reduction		Methane		Methane Potential		
		Optimum[a]	Failure[a]	Optimum	w/o GTW	Optimum	w/o GTW	Optimum	w/o GTW	Optimum	Increase
		(% VS basis)		(gVS/L-d)	(%)		(%)		(mL CH_4/g VS_{added})		%
Present Study (2014)	50% PS & 50% WAS + GTW (5% VS)[b]	46	70	2.93	47	59	61	69	384	641	67%
(Wang et al., 2013)	TWAS[b] + pre-treated GTW (≈19% VS)	20	40	2.16	30	55	60	70	180	752	318%
(Liu & Buchanan, 2011)	75% PS & 25% TWAS + GTW (≈64% VS)	68	na	4.24	na	73	na	66	na	620	N/A
(Wan et al., 2011)	TWAS + GTW (3% VS)	64	75	2.34	40	57	65	67	252	598	137%
(Kabouris et al., 2009)	PS & TWAS 1:5 w/w + polymer dewatered GTW (≈41% VS)	47.6	na	4.35	25	50	66	66	159	473	197%
(Luostarinen et al., 2009)	MWS and Thickened GTW from meat processing plant (25% VS)	46	55	3.46	52	67	63	62	278	463	67%
Davidsson et al. (2008)	50% PS + 50% WAS, and GTW (17% VS)	30	na	2.4	45	58	65	69	271	344	27%

Abbreviations: MWS, municipal wastewater sludge; GTW, grease trap waste; OLR, organic loading rate; VS, volatile solids; PS, primary sludge; WAS, waste activated sludge; TWAS, thickened waste activated sludge; na, not available.
[a] Optimum and failure GTW feed ratios.
[b] VS concentration of GTW used in the study was shown in parenthesis.

4.4 CONCLUSIONS

The results of the present study indicate that anaerobic co-digestion of MWS with grease trap waste without dewatering yields similar performance in terms of biogas production, methane content, and VS reduction as compared to previous studies of co-digestion of dewatered GTW. Successful co-digestion was observed up to 46% GTW feed ratio. AcoD of MWS with un-dewatered GTW yielded 67% increase in methane potential as compared to MWS alone digestion. The results also demonstrated that using un-dewatered GTW as the co-substrate can reduce the inhibition risk. The GTW feed ratio that yielded stable co-digestion was increased by using un-dewatered GTW. The maximum VA/Alkalinity ratio that did not result in digester failure was established as 2 for AcoD of MWS and un-dewatered GTW. The recovery method suggested in this study can provide a relatively rapid and effective recovery of digesters following upset conditions, if additional storage is available for the effluent discharged during the recovery. In conclusion, anaerobic co-digestion of MWS and GTW without dewatering at the nearest digesters, if possible, can improve digester performance and provide a more stable digestion. Also, additional processes associated with transferring and dewatering GTW can be eliminated. Further investigation is needed to support the feasibility of digesting GTW at the nearest WWTP instead of hauling the dewatered GTW long distances for treatment.

Chapter 5: Conclusions and Recommendations for Future Research

5.1 Conclusions

The primary objective of this research was to investigate the effect that blending un-dewatered grease trap waste with municipal wastewater sludge in anaerobic co-digestion reactors has on methane potential and system inhibition. To this end, the biochemical methane potential (BMP) of un-dewatered GTW was evaluated through a series of BMP tests, and the stability and performance of AcoD of MWS and un-dewatered GTW in semi-continuous feed reactors was investigated. The observed results were compared with previously published methane potentials of concentrated/dewatered GTW. The key observations and specific conclusions from this study are summarized as follows:

- The biochemical methane potential of un-dewatered GTW was established as 606 mL CH_4/g VS_{added}. Although this value was less than previously reported methane potentials of concentrated/dewatered GTW (845 – 1050 mL CH_4/g VS_{added}), the methane potential of un-dewatered GTW observed in the current study was more than twice the 223 mL CH_4/g VS_{added} determined for anaerobic digestion of MWS alone.
- Anaerobic co-digestion of MWS with un-dewatered GTW up to a 46% GTW feeding ratio not only increased biogas production, but also increased the quality of the gas generated and yielded greater organics stabilization relative to digestion of MWS alone. The biogas production increased from 8.75 L/d to 12.5 L/d to 14.45 L/d at 0%, 25% and 46% GTW feed ratios, respectively. Similarly, the methane content increased from 61% to 66% to 69% at 0%, 25% and 46% GTW feed ratios, respectively. VS reduction increased from 47% to 50% to 59% at 0%,

25% and 46% GTW feed ratios, respectively. Finally, the methane yield increased from 384 mL to 536 mL to 641 mL CH_4/ g VS_{added} at 0%, 25% and 46% GTW feed ratios, respectively.

- The highest methane potentials in previous studies without inhibition or failure of the process were obtained at GTW feed ratios between 20% and 68% (on VS basis) for anaerobic co-digestion of dewatered GTW under semi-continuous feed conditions. Inhibition of the digestion system was observed above these feed ratios. In the current study with un-dewatered GTW, the highest methane potential was observed at 46% GTW feed ratios on VS basis. The GTW feed ratio that yielded digestion inhibition was increased by using un-dewatered GTW. The results indicate that using un-dewatered GTW as the co-substrate yields a safer digestion at lower feed ratios (<46% GTW on VS basis) compared to dewatered GTW. These results supports the hypothesis that using un-dewatered GTW instead of dewatered GTW can reduce the inhibition risk of anaerobic co-digestion of MWS with GTW.
- Successful anaerobic digestion was observed at feeds up to 1.96 ± 0.06 VA/Alkalinity ratio. However, a feed of 2.14 ± 0.03 VA/Alkalinity ratio led to VA accumulation and consumption of the buffering capacity in the system. These results indicate that, feed ratios of VA/Alkalinity greater than 2 should not be applied when co-digesting MWS with un-dewatered GTW.
- A modified Gompertz equation (Eq. 3) describing the relationship between GTW feed ratio (on a VS basis) and methane potential was developed. The data collected in the current BMP tests and those reported in previous studies were successfully fit to the modified Gompertz equation (Eq. 3) and allowed estimation of key process parameters including the maximum methane potential rate (R_m)

and the initial methane potential lag phase (X_o). The X_o and Rm values for un-dewatered GTW digestion were higher than that observed for concentrated/dewatered GTW digestions values. The higher X_o and R_m values observed for un-dewatered GTW digestion indicated that a minimum FOG content might be needed before there is a measurable effect of GTW addition on methane potential.

- Modification of the standard BMP test (Periodic/10-day testing) provided a better understanding of the co-digestion process by allowing periodic monitoring of biochemical behavior, TS, VS, COD, N_{H3}-N, Alkalinity, VA, pH, for each feed combination/mixture; however, following a more frequent testing schedule in the first 10 days is suggested.
- Addition of digester effluent after biogas production ceased yielded 100% recovery of the methane production to the levels produced in the digester prior to disruption. Semi-continuous feeding of effluent from an operating anaerobic digester into an anaerobic digester following upset conditions provided a rapid and more effective recovery as compared to the previously reported methods. The main disadvantage of this method is the requirement to store the effluent from the unstable digester until the recovery is achieved. At that point, the stored sludge can be fed back into the digesters for further digestion. This method requires that additional storage be available.
- 172% and 67% increase in methane potentials relative to the methane potentials from MWS alone digestion were observed under batch and semi-continuous feed conditions, respectively. Dilution of inhibitory substances, high inoculum loadings and low organic loading rates can cause the higher methane potentials observed in the batch experiments. The observations from this study indicate that

semi-continuous feed conditions provide a better simulation of the AcoD process than batch feed conditions (BMP tests).

This study demonstrates that anaerobic co-digestion of MWS with un-dewatered GTW not only increases the methane production, but also reduces the inhibition risk of anaerobic co-digestion process. The performance data that are presented in the current study can be used as a guide for the development of real-scale applications in Austin, TX and other municipalities.

Many municipalities in Texas require FSEs to install grease traps and periodically empty the traps. However, most of the municipalities have no further GTW management strategies. Based on observations in Texas, operators are hesitant to feed GTW into anaerobic digestion facilities because of a lack of information on AcoD of GTW with MWS and the serious negative consequences of improper management. This lack of information leads to difficulties in GTW management in Texas. Not all municipalities are accepting GTW; therefore, additional treatment and long distance hauling are required. The collected GTW is first hauled to transfer stations. Dewatering, usually with chemical addition, is performed at the transfer stations to decrease the volume. The dewatered GTW is transferred to the WWTP. This practice requires a licensed transfer station, dewatering unit, chemical addition, discharge of high BOD wastewater into the municipal wastewater collection system, and creates extra carbon dioxide emissions resulting from additional truck trafficking. Anaerobic co-digestion of un-dewatered/raw GTW at the nearest WWTP after collection may eliminate these requirements and reduce the inhibitory effects of dewatered GTW on the anaerobic co digestion process.

5.2 Recommendations for Future Research

Ultimate disposal or resource recovery of GTW needs to be addressed in the management policies of municipalities and local solutions should be identified. The economic feasibility of AcoD of un-dewatered GTW with MWS at local anaerobic digestion facilities versus chemically dewatering at local transfer stations and hauling long distances for disposal or resource recovery must be evaluated.

In this study, the characterization of GTW did not include long chain fatty acids (LCFAs). High concentrations of LCFAs have been reported to cause inhibition of the digestion process. More comprehensive characterization of GTW including LCFAs may provide a better digestion strategy. Similarly, monitoring LCFAs and the change in microbial community at increasing GTW feed ratios may provide additional insight into the process.

Appendix A: Batch Experiment Digestion Bottles

Figure A.1: Batch digestion bottles in the incubator (35 ^{0}C).

Figure A.2: Biogas measurement from batch digestion bottles.

Appendix B: Semi-Continuous Feed Reactor Experiment Digestion System Drawings and Photos

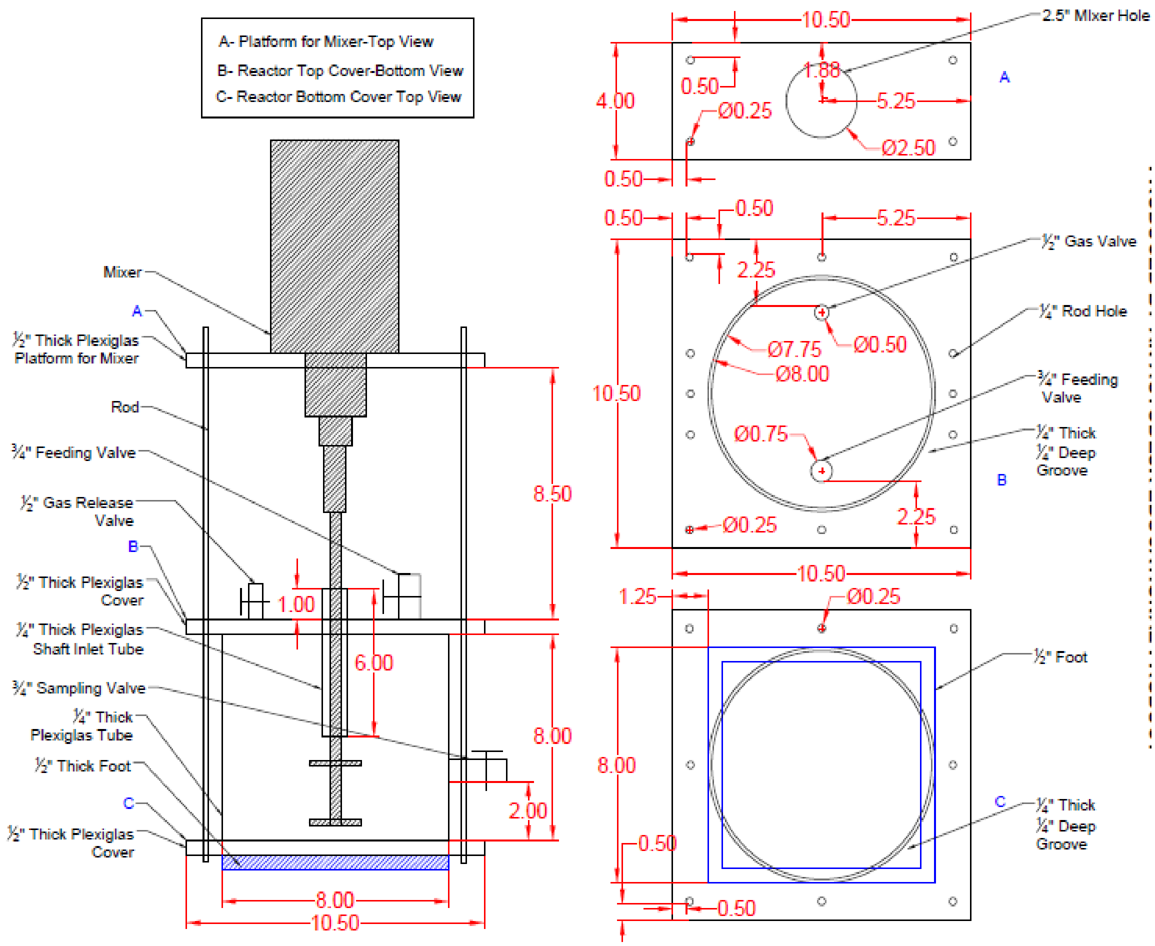

Figure B.1: Technical drawings of the semi-continuous feed digesters

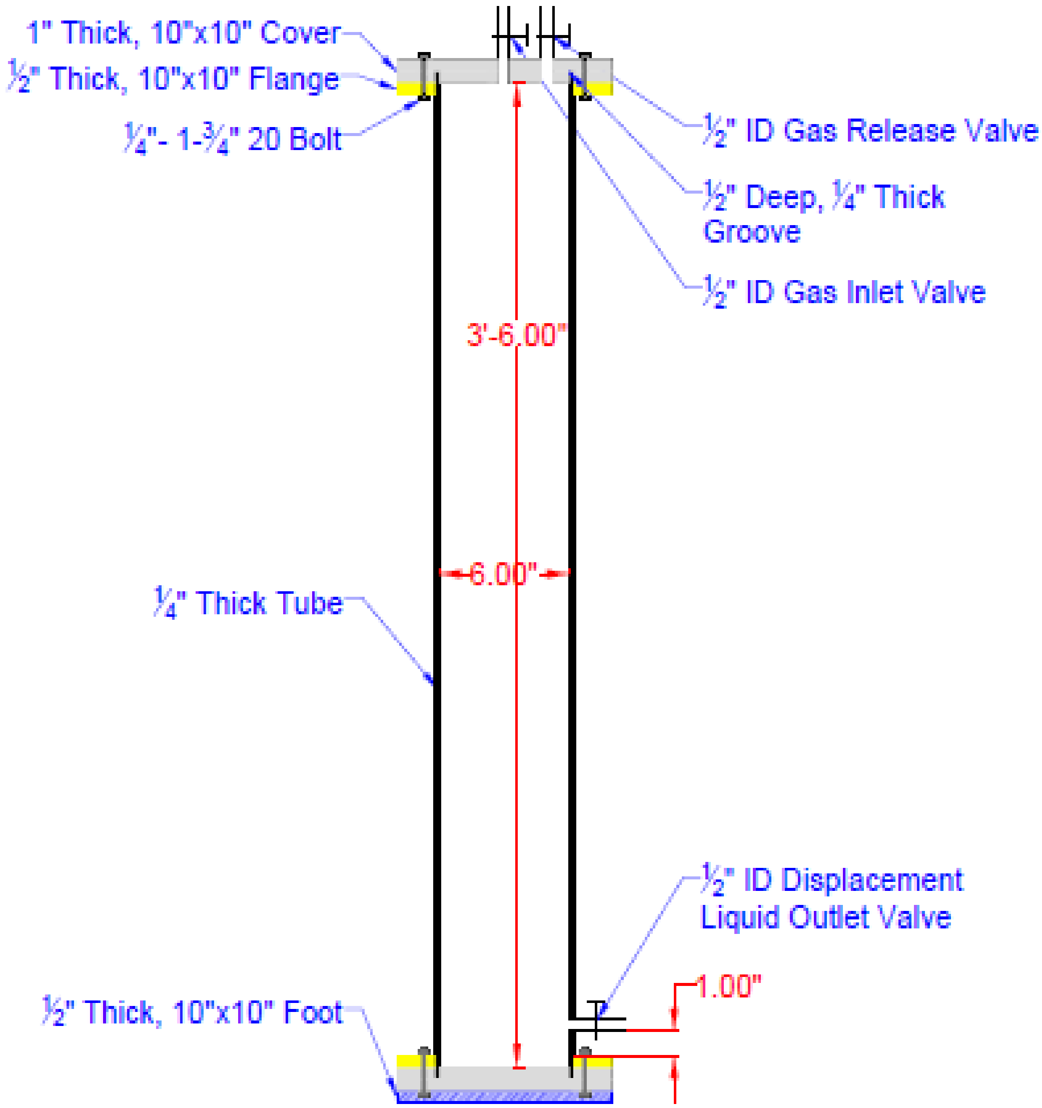

Figure B.2: Gas collection unit body drawing.

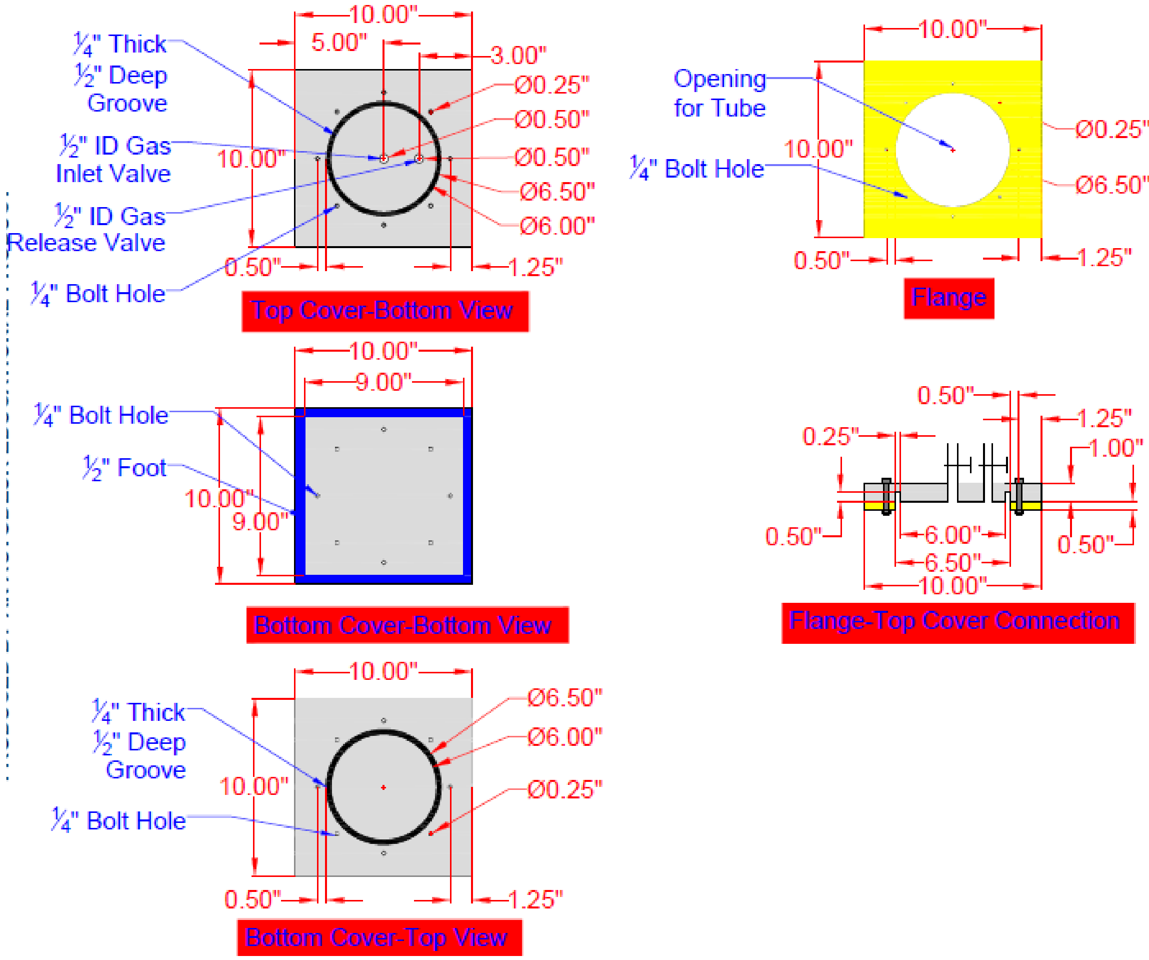

Figure B.3: Gas collection unit detail drawings.

Figure A.3: Anaerobic digestion system, Front-View.

Figure A.4: Anaerobic digestion system, Side-View.

Appendix C: List of Symbols and Nomenclature

GTW	Grease trap waste
MWS	Municipal wastewater sludge
FOG	Fats, oils and grease
FSE	Food service establishments
POTW	Publicly-owned treatment works
AcoD	Anaerobic co-digestion
BMP	Biochemical methane potential
AD	Anaerobic digestion
OFMSW	Organic fraction of municipal solid wastes
MWWTP	Municipal wastewater treatment plant
DLVO	Derjaguin and Landau, Verwey and Overbeek theory
HRT	Hydraulic residence time
PFGT	Passive flow grease trap
MFGT	Mechanical flow grease trap
WWTP	Wastewater treatment plant
GT	Grease trap
OHPA	Obligate hydrogen producing acetogens
HA	Homoacetogens
DAF	Dissolved air flotation
WAS	Waste activated sludge
TWAS	Thickened waste activated sludge
PS	Primary sludge
INO	Inoculum

HBBMP	Hornsby Bend Biosolids Management Plant
VA	Volatile acids
TS	Total solids
VS	Volatile solids
LCFA	Long chain fatty acids
FFA	Free fatty acids
BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
N:P	Nitrogen to phosphorus ratio
C:N	Carbon to nitrogen ratio
S:INO	Substrate to inoculum ratio
R^2	Coefficient of determination
B	Methane potential at "X" GTW feed ratio (mL CH_4/g VS_{added})
B_i	Minimum methane potential or methane potential at 0% GTW addition (mL CH_4/g VS_{added})
Bu	Ultimate methane potential (mL CH_4/g VS_{added})
R_m	Maximum methane potential rate (mL CH_4/g VS_{added} – % GTW)
E	Euler's number (2.7183)
X_0	Lag phase GTW ratio (%)
X	GTW ratio (%)

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Vita

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