ABSTRACT

Title of Thesis:

NOVEL APPLICATIONS OF WASTE-TREATMENT TECHNOLOGIES TO GENERATE ENERGY AND TREAT WATER

Kirkland Mahoney, M.S., 2023

Thesis Directed By: Assistant Research Professor Amro Hassanein, Environmental Science and Technology

This research sought to maximize energy generation and optimize water treatment from a combined waste stream of food waste and blackwater (FW-BW) and monitored a farm-scale anaerobic digestion (AD) incorporating co-digestion and composting. The FW-BW substrate was gravity separated to obtain a liquid fraction (~90% of the volume) and a solid fraction (~10%). Solids and liquids were pretreated with hydrodynamic cavitation (HDC) to investigate changes in water treatment and energy generation from this pretreatment strategy. Energy generation from the solid fraction was assessed using biochemical methane potential tests (BMP) that quantified methane (CH_4) production from AD and integrated a combined anaerobic digestion-microbial electrolysis cells (AD-MEC) design, which operated at various voltages (0.5, 0.9, and 1.2 V).

Electrocoagulation (EC) at various voltages (10-25 V) and timeframes (15-90 minutes) was assessed for contaminant removal from the liquid fraction.

AD at mesophilic conditions (35 °C) of solids after HDC (post-HDC solids) generated 81.2% of the cumulative CH_4 (348 mL CH_4/g volatile solids (VS) in 10 days. At longer digestion times (30 days), post-HDC solids generated significantly more CH_4 (63%, 429 mL CH_4/g VS) than solids before HDC (no-HDC solids) due to increased substrate availability and degradability. Energy generation from AD-MEC at 1.2 V was not significantly different from AD, with only 12.7% more CH_4 (292 mL CH_4/g VS) generated from post-HDC solids compared to no-HDC solids (259 mL CH_4/g VS) after 10 days.

Electrocoagulation conducted at 15 V for 90 minutes removed 96.2% of chemical oxygen demand (COD) and 100% of total suspended solids (TSS) from post-HDC liquids. Increasing EC conductivity with electrolytes decreased the timeframe (15 minutes) and voltage (10 V) needed for COD removal (66%) via increased contaminant flocculation.

The performance of a farm-scale AD system co-digesting FW and dairy manure (DM) was monitored and verified by analyzing the substrates for nutrient and solids content. A combined heat and power generator produced electricity from the biogas. The system parameters were monitored with an online data-logging system that collected data every 15 minutes for biogas, temperature, hydrogen sulfide (H_2S), energy generation, and CH_4 content. A life cycle assessment (LCA) was applied to explore the environmental impacts of the current condition compared to four alternative scenarios. The LCA suggested the co-digestion system as currently operated had the largest reductions in environmental impact in 8 out of the 10 impact categories compared to a baseline scenario (no digestion), generating substantial reductions in global warming (81%), eutrophication (442%), and acidification (321%). Using AD can successfully convert various wastes into energy, with HDC increasing overall energy production. Moreover, farm-scale AD showed substantial reduction in global warming.

NOVEL APPLICATIONS OF WASTE-TREATMENT TECHNOLOGIES TO GENERATE ENERGY AND TREAT WATER

By

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Dedication

To the Mahoney family. Thank you for the unconditional love, guidance, and support.

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List of Abbreviations

AD

Anaerobic digestion

OM

Organic material

WWTP

Wastewater treatment plant

SCFA

Short-chain fatty acids

VFA

Volatile fatty acids

HRT

Hydraulic retention time

VS

Volatile solids

DM

Dairy manure

FW

Food waste

BW

Blackwater

COD

Chemical oxygen demand

sCOD

Soluble chemical oxygen demand

BOD

Biological oxygen demand

TSS

Total suspended solids

MEC

Microbial electrolysis cell

OLR

Organic loading rate

BES

Bioelectrochemical system

EET

Extracellular electron transfer

TEA

Terminal electron acceptor

THP

Thermal hydrolysis pretreatment

HDC

Hydrodynamic cavitation

EC

Electrocoagulation

EDL

Electrical double layer

LCA

Life cycle assessment

CHP

Combined heat and power generator

Chapter 1: Introduction

1. Anaerobic Digestion

Anaerobic digestion (AD) is a four-stage microbial process that utilizes the oxidative potential of various microbes to break down organic matter (OM) (Figure 1.1). In the primary stage, hydrolytic microbes dissociate complex organic materials into proteins, carbohydrates, and lipids. Following the hydrolytic stage, acidogenic bacteria form short-chain fatty acids (SCFAs), known as volatile fatty acids (VFAs). VFAs are broken down by acetogenic bacteria into hydrogen gas (H_2), carbon dioxide (CO_2), and acetate (CH_3COO^-) intermediates. Methanogenic microorganisms transform the intermediates into biogas rich in methane (CH_4) (Hassanein, Lansing, et al., 2022). Anaerobic digestion converts abundant OM into usable energy and nutrient-rich fertilizers. AD is an adaptable technology, although the performance of AD can be adversely affected by several factors, including temperature, pH, hydraulic retention time (HRT), substrate availability, microbial composition, and substrate composition.

Temperature is a primary driver of the AD process. Shaping microbial communities and influencing performance stability, AD is operated at three temperature levels: psychrophilic (<20 °C), mesophilic (20–43 °C), and thermophilic (50–60 °C, 55 °C being optimal) (Nie et al., 2021). Mesophilic and thermophilic temperatures are the most ubiquitous levels in farm-scale AD, however, both conditions have advantages and disadvantages. Nie et al. (2021) found that temperature affects the performance of hydrolytic microorganisms. The function of extracellular hydrolytic enzymes, including urease, β-glucosidase, N-α-benzoyl-l-argininamide-hydrolyzing protease, and phosphatase increase with temperature, indicating thermophilic conditions are optimal (Lu et al., 2007; Nie et al., 2021). Furthermore, increased VFA production under thermophilic conditions boosts CH_4 production (J. K. Kim et al., 2006; Mengmeng et al., 2009; Nie et al., 2021). On the other hand, mesophilic temperatures increase microbial diversity compared to thermophilic digestion, where lower microbial diversity can precipitate higher microbial sensitivity and possible ammonia (NH_3) inhibition under thermophilic conditions (Ao et al., 2021; Wu et al., 2020). Therefore, digesting substrates with specific degradability and nutrient content is imperative to sensitive microbial interactions and subsequent digester performance.

pH is a vital parameter to the efficiency of AD due to its relationship with VFA generation. Previous literature reported that pH modifies the presence of certain VFAs, where alkaline influent pH (>7) produces primarily acetic acid, and acidic influent pH (<7) generates butyric acid (Lu et al., 2020). VFAs are the primary product of acidogenesis/acetogenesis, and a vital substrate for methanogenic pathways. VFAs are characterized as unsaturated or saturated carboxylic acids containing six or fewer atom chains of carbon (Feng et al., 2022). Prevalent VFAs in AD include acetic, propionic, butyric, valeric, and caproic acids (Wainaina et al., 2019). While VFA production is integral to the performance of AD, VFAs can also be detrimental. Excessive VFA production can lower the pH of the digester, eliminating sensitive methanogenic archaeans.

The proper HRT selection is crucial to the performance of AD. Low HRT ($\leq$ 5 days) can raise VFA production, harming microbes and lowering CH_4 yield and volatile solids (VS) removal (Bi et al., 2020). On the contrary, high HRT (HRT $\geq$ 15 days) can increase the maximum CH_4 yield at the expense of higher operational costs (Bi et al., 2020). Additionally, HRT can modify the microbial composition and CH_4 production pathways. Bi et al. (2020) found that an HRT of 5 days induced an H_2-mediated CO_2 reduction pathway favored by the hydrogenotrophic genus *Methanobrevibacter,* resulting in the domination of the microbial consortium by hydrogenotrophic archaeans. However, hydrogen influences digestion performance through pH variation.

1.2 Anaerobic Digestion Substrates

Substrates have one of the most profound effects on the performance of AD. Low C:N substrates lower biogas yield and process stability (Johannesson et al., 2020; Masih-Das & Tao, 2018). To mitigate low productivity, AD can be operated with a single feedstock, known as monodigestion, or with multiple substrates, known as co-digestion. When applied at the farm scale, anaerobic co-digestion can simultaneously transform abundant conventional wastes into renewable energy and manage nutrients. Successfully applied in food processing, livestock production, and sewage treatment, AD can treat a wide range of OM. AD efficiency and stability can be positively impacted by co-digesting high-nutrient substrates with low C/N substrates like dairy manure (DM).

1.2.1 Dairy Manure

Livestock substrates, such as DM, pose several benefits and drawbacks when monodigested. On one hand, the high lignocellulose content, low C:N ratio, and high moisture content of DM generates low biogas yield and necessitate higher HRTs (Li et al., 2021). On the other hand, DM benefits AD by supplying fermentative microbes that aid the start-up of the AD process, raises buffering capacity, and is commonly accessible (Acosta et al., 2021; Tallou et al., 2020). In the U.S. livestock industry, manure is typically stored in uncovered lagoons prior to crop application generating a substantial environmental burden and nuisance to the surrounding community. Moreover, stagnant manure storage in open lagoons creates anaerobic conditions. Unmitigated CH_4 production contributes to greenhouse gas (GHG) emissions with a global warming potential 27-30 greater than CO_2 (EPA, 2022). Consequently, livestock manure management contributes 10% of annual CH_4 emissions in the U.S., with anaerobic lagoons releasing the majority of CH_4 (368 ±193 kg CH_4 hd^{-1} yr^{-1}) (Owen & Silver, 2015).

In addition to GHG emissions, anaerobic conditions in open lagoons contribute to waterway pollution while producing negative odors from CH_4 emissions (Wilkie, 2005). AD mitigates the environmental burden from nutrient deposition and manure

GHGs by capturing CH_4 and utilizing it for energy production (Holm-Nielsen et al., 2009). With only 238 out of 20,000 livestock operations in the U.S. employing AD, nearly 1 billion pounds of produced animal manure is deposited into the environment on a yearly basis, representing an opportunity to generate energy and reduce GHG emissions between 978 to 1776 kg CO_2 eq year^{-1} per livestock (cow) unit (Cowley & Brorsen, 2018; Maranon et al., 2011; Zhang & Schroder, 2014). When utilized as a manure-treatment technology, AD has several benefits for farm-scale applications. The employment of AD to dairy farms can: 1) reduce the odors associated with DM storage, management, and disposal; 2) minimize the impacts of DM storage on the surrounding environment by COD reduction and total solids/volatile solids (TS/VS) removal (Lansing et al., 2010); 3) reduce the emission of GHG; and 4) generate CH_4-rich biogas for electricity production. While applying AD to DM is a promising approach for manure management, low productivity associated with DM mono-digestion inhibits further applications (Farghali et al., 2020). However, co-digesting DM with protein and carbohydrate-rich substrates, like FW, provides an opportunity to simultaneously improve biogas generation and lower environmental impacts associated with FW disposal.

1.2.2 Food Waste

Food waste constitutes a waste stream with substantially negative environmental impacts. In the U.S., 36.9 million metric tons of FW were disposed of in 2017, with 75.3% sent to landfills (Dalke et al., 2021). Food waste is an acidic to sub-acidic substrate comprised of proteins (15-25% TS), carbohydrates (50-60% TS), and lipids (13-30% TS) with variable moisture content (70-80%) (Negri et al., 2020). Food waste in the U.S. is composed of up to 57% carbohydrates and 23% proteins (Dalke et al., 2021). As proteins and carbohydrates are valuable compounds for AD due to their high biodegradability, digesting food waste presents an opportunity for improved CH_4 production and AD stabilization (Awosusi et al., 2021). However, the mono-digestion of FW faces challenges. Excessive loading of FW in AD decreases stability via VFA accumulation and NH_3 inhibition, resulting in a low process efficiency (Srisowmeya et al., 2020). Therefore, co-digesting FW with low-C, high-N substrates positively impacts stabilization and energy generation from AD. One unconventional FW substrate employed in co-digestion is dissolved air flotation waste (DAF). DAF is a flotation process where fats oils, and greases (FOG) are flocculated, floated, then collected, leaving a nutrient-rich, high VS liquid behind. Containing VS reported between 2.3-14.7%, DAF can boost CH_4 yield by 2,940% (Johannesson et al., 2020; Lisboa & Lansing, 2013), although loading rates must be considered as the acidic nature of DAF (pH ~5) can drop digester pH and inhibit methanogenesis (Logan et al., 2021). Co-digestion can mitigate stability issues arising from FW, with previous research reporting that co-digesting FW with substrates such as DM or blackwater (BW) improves CH_4 production by over 50%, enriches microbial communities, increases hydrolytic coefficients, and reduces burdens on landfills (Ebner et al., 2016; Wang et al., 2020; Zhang et al., 2013; Zhang et al., 2019).

1.2.3 Blackwater

Blackwater (BW) is a high-strength, human-derived waste stream procured from toilets. Representing roughly 30% of household water consumption, BW supplies 50-60% of OM and 75-95% of nutrients contained in domestic sewage (Wen et al., 2022; Zhang et al., 2019). Blackwater can be used as an AD substrate, although low C, sulfides, pharmaceuticals, and free NH_3 from high pH can negatively influence AD performance (Wang et al., 2020; Wen et al., 2022; Zhang et al., 2019; Zhang et al., 2020). Previous literature showed co-digesting FW and BW (2:1 FW-BW) enriched hydrogenotrophic methanogen and increased CH_4 yield from 51-85% compared to BW-only (Zhang et al., 2019). These data suggest that energy production from AD can be maximized by combining N-rich BW with C-rich FW.

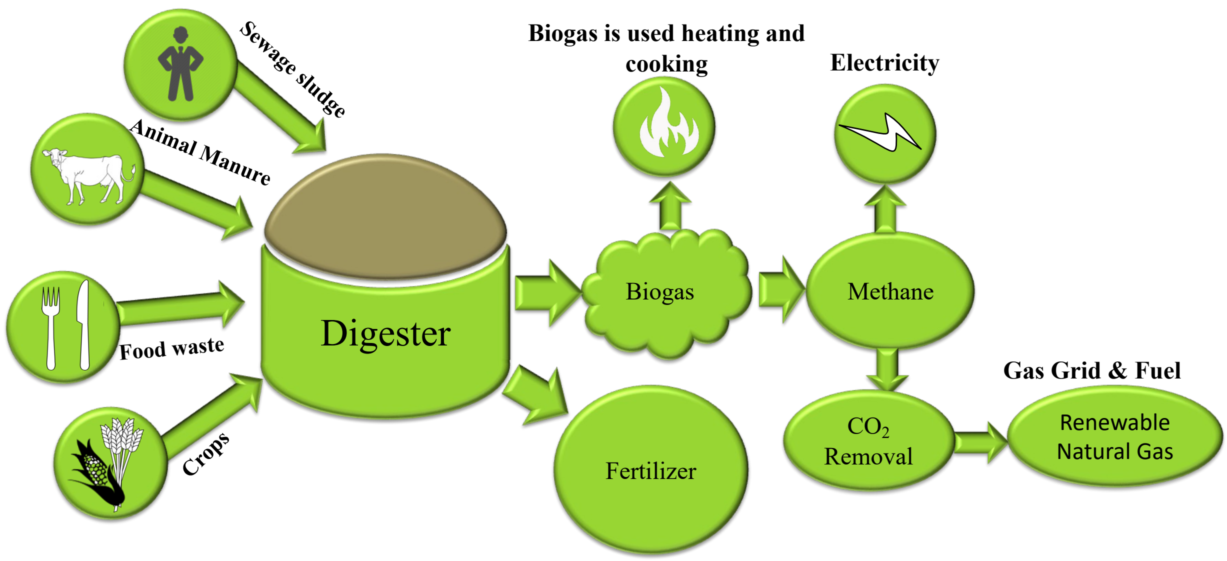

Figure 1.1: Anaerobic digestion basics (Hassanein et al., 2022)

1.3 Microbial Electrolysis Cell

Bioelectrochemical systems (BES) have been employed as a technology to increase energy generation from OM. Microbial electrolysis cells (MECs) are a BES technology that has risen to prominence as a simple, cost-effective renewable energy technology (Dange et al., 2021). By utilizing a small amount of electrical input to an anode and cathode, MECs influence the production of H_2 and CH_4 by harboring electrogenic organisms that assist in oxidizing OM in waste streams (Figure 1.2) (Zhang & Angelidaki, 2014).

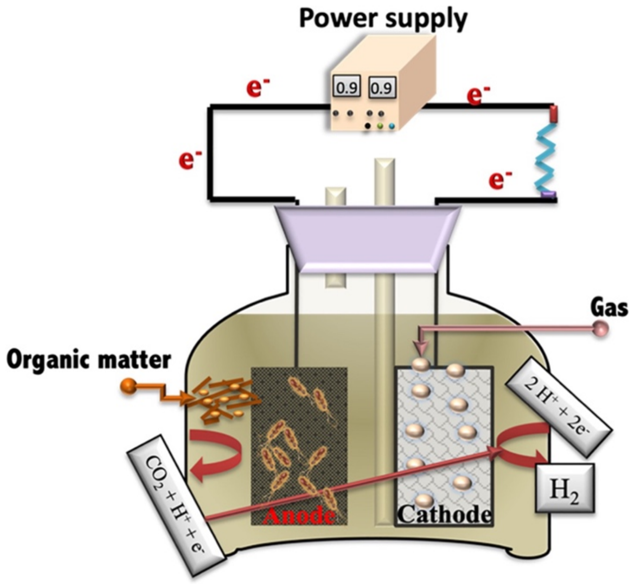

Figure 1.2: Diagram of a MEC (Hassanein & Lansing, 2022)

MEC operation is centered around extracellular electron transfer (EET) performed by electrogenic bacteria. Starting with the supplementation of a small voltage input (0.2-1.2 V), electroactive bacteria such as *Geobacter* and *Shewanella* oxidize OM at the bioanode via MtrAB porin cytochrome complex. *Shewanella oneidensis* performs EET using a metal respiratory system (Mtr) typically facilitated by various Fe and Mn species as terminal electron acceptors (TEA). On the other hand, *Geobacter sulfurreducens* employ C- and B-type cytochrome-containing complexes. After transferring electrons extracellularly to the anode, the electrons are transferred through an external circuit to the cathode. At the cathode, two pathways of CH_4 generation (electromethanogenesis) occur: direct and indirect. Direct electro methanogenesis consists of three routes for CH_4 production; 1) utilization of a series of redox-performing external membrane proteins (cytochromes) in direct contact with the cathode; 2) EET by various alternative membrane proteins, including hydrogenase, ferredoxin, or rubredoxin; 3) EET by nanowires (conductive pili). Indirect electro methanogenesis can be completed through multiple pathways: 1) hydrogen, formate, or acetate produced electrochemically/bioelectrochemically; 2) externally located electron mediators/shuttlers including quinones, phenazines, and riboflavins (Pawar et al., 2022).

1.3.1 Microbial Electrolysis Cell Design

Microbial electrolysis cells consist of either a membraned or membrane-less configuration with two bioelectrodes. Microbial electrolysis cell bioelectrodes are comprised of a wide range of materials including carbon, stainless steel, Ni-alloys, titanium, palladium, and polyaniline are all utilized in electrode configurations for MECs (Park et al., 2022). Generally, electrode material must be cost-effective, maintain a high surface area, have exceptional electroconductivity, good electrochemical stability, adequate electrocatalytic activity, and low hydrogen overpotential (Park et al., 2022).

1.3.2 Anaerobic Digestion Integration with Microbial Electrolysis Cell

MECs can be integrated with AD (AD-MEC) to increase the generation of biogas and biohydrogen via microbial transformations (Bo et al., 2014). CH_4 generation in AD-MEC is performed by many different archaens, with two primary metabolic pathways occurring: acetoclastic methanogenesis and hydrogenotrophic methanogenesis. The prevalent CH_4 production pathway (hydrogenotrophic) arises from the reduction of H_2 and CO_2. Performed by archaeans from *Methanobacterium, Methanospirillum, Methanobrevibacter, Methanosarcina, Methanoculleus,* and *Methanocorpusculum,* CO_2 is first reduced to a formyl group and combined with the carrier molecule methnofuran (MFR) to create formylmethanofuran (CHO-MFR). Next, tetrahydromethanopterin (H_4MPT) acts as a carrier for the formyl group (formyl-H_4MPT), followed by the generation of cyclic methenyl-H_4MPT resultant from dehydration. The methenyl group is reduced to a methylene group (methylene-H_4MPT), then a methyl group (CH_3- H_4MPT). Sulfhydryl-containing coenzyme M (HS-CoM), an intermediate of all methanogenic pathways, transfers the methyl group (CH_3-S-CoM) through the cell membrane. Finally, CH_4 is reduced from the methyl group via the oxidation of HS-CoM combined with sulfhydryl-containing coenzyme B (HS-CoB) (Costa & Leigh, 2014; Pawar et al., 2022). The acetoclastic methanogenic pathway, performed by archaea from *Methanotrix* and *Methanosarcina*, utilizes acetate for CH_4 production. Acetate is first activated through the transformation of ATP and coenzyme A (CoA) into acetyl-CoA. Acetyl-CoA is divided into a carbonyl and a methyl group. The methyl group is collected by H_4MPT and finishes the final two CO_2-reducing pathway steps found in hydrogenotrophic methanogensis. The remaining carbonyl group is transformed into

CO_2 by an oxidation reaction mediated by ferredoxin as the electron acceptor (Lyu et al., 2018). These pathways represent the primary routes of CH_4 production in AD-MEC.

1.4 Pretreatment of Substates

As high HRT and low biogas yield can restrict AD, pretreating waste increases carbon availability for energy production while lowering HRT (Ariunbaatar et al., 2014). Hydrolysis is generally the rate-limiting step in the AD of complex substrates, as hydrolytic enzymes are inadequate to thoroughly degrade intricate substrates. Hydrolysis of carbohydrates takes several hours, but hydrolysis of proteins, lipids, and cellulosic material can take several days (Atelge et al., 2020). As the destruction of complex OM maximizes substrate availability and digestibility, including pretreatment technologies before AD both lowers the HRT and enables a higher organic loading rate (OLR) (Atelge et al., 2020). Several pretreatment methods prevalent in waste treatment, including thermal and physicochemical, can increase AD process efficiency.

1.4.1 Thermal Hydrolysis Pretreatment

Pretreatment methods can increase economic and material inputs for waste treatment, generating feasibility concerns. Thermal pretreatment technologies, like thermal hydrolysis pretreatment (THP), utilize extreme temperatures to enhance energy production. By disintegrating sludge flocs and rupturing cell membranes with high temperatures (140 to 170 °C) and pressures (6 to 9 bar), THP increases VFAs via the reaction of free radicals (Ngo et al., 2021). Previous studies have reported a 55% increase in biogas production when AD is paired with THP (Abelleira-Pereira et al., 2015). However, Abelleira-Pereira et al. (2015) revealed that THP requires substantial energy input to maintain high temperatures and pressure. Alternatively, hydrodynamic cavitation (HDC), a physicochemical pretreatment, has been described as an eco-friendly, less energy-intensive process to enhance microbial biotransformation of OM in AD (Mancuso et al., 2022).

1.4.2 Hydrodynamic Cavitation

Hydrodynamic Cavitation is a physicochemical pretreatment technology that valorizes biomass by increasing the OM availability in liquid and solid fractions of waste. HDC triggers rapid pressure changes in a substrate by passing the material through a small constriction/orifice, or on the suction side of a rotating blade, simultaneously forming microscopic cavities in a phenomenon known as nucleation. Within the microscopic cavities, bubble growth occurs, followed by bubble implosion (Asaithambi et al., 2019). The collapse of the microscopic cavities generates enormous temperature and pressure changes (1,000-10,000 K and 100-5,000 bar, respectively) that lyse surrounding cells, effectively demolishing biomass while amplifying COD and biogas yield (Asaithambi et al., 2019; Nagarajan & Ranade, 2021). The breakdown of recalcitrant OM aids in the utilization of C as a substrate in CH_4 generation via AD. HDC has been shown to reduce the timeframe of AD, increase CH_4 production by over

20%, and lower the energy inputs (60-1200 kJ/kg TS) compared to thermal pretreatment (72,000 kJ/kg TS) technologies (Abrahamsson, 2016; Lee & Han, 2013).

1.5 Electrocoagulation

Water is one of the most valuable resources adversely affected by consumption in various industries. Further impacted by climate change, freshwater supplies have been drastically altered, leading to a need for conservation and improved management (Kundzewicz et al., 2008). Identifying alternative water treatment methods to reduce freshwater usage is crucial.

In distributed operations, water use is estimated to be between 93 to 225 liters of water per person each day. Between 131 to 138 liters per day of wastewater (blackwater) is generated. Furthering this dilemma, treating this nutrient-heavy wastewater can create a burden for WWTP, generating excess organic load on the environment in the form of COD and BOD. Water treatment technologies like chemical flocculation and electrocoagulation present a treatment approach for low-tier reuse.

Electrocoagulation (EC) is a flocculation technology that can treat water and remove contaminants (Figure 1.3). Treating water for low-tier reuse and recovery, EC utilizes an applied voltage at an anode and cathode to jumpstart a three-stage process where contaminants and COD are removed (Garcia-Segura et al., 2017). Contaminants and pollutants in wastewater are typically colloidal, stable, negatively charged, and highly resistant to destabilization. These properties allow contaminants and pollutants to resist filtration, flotation, or sedimentation. In the first step of EC, voltage (10-25 V) is applied to a sacrificial electrode causing the dissolution of metal ions into the solution. Typically, Al or Fe, due to cost and availability, ions generated at the anode destabilize surface charges held on contaminants and pollutants by overcoming Van der Waals forces. Once the surface charges of the contaminants are negated from the compression of the electrical double layer (EDL) via ionic species interaction and van-der-Waals forces are overcome, the oxidizing metal hydroxyl ions form flocculants with the contaminants. Larger flocs with adsorbed contaminants rise to the top of the solution with H_2 and O_2 bubbles generated at the cathode (Hakizimana et al., 2017; Tahreen et al., 2020). The water beneath the flocculated solids layer has reduced COD, color, and total suspended solids (TSS).

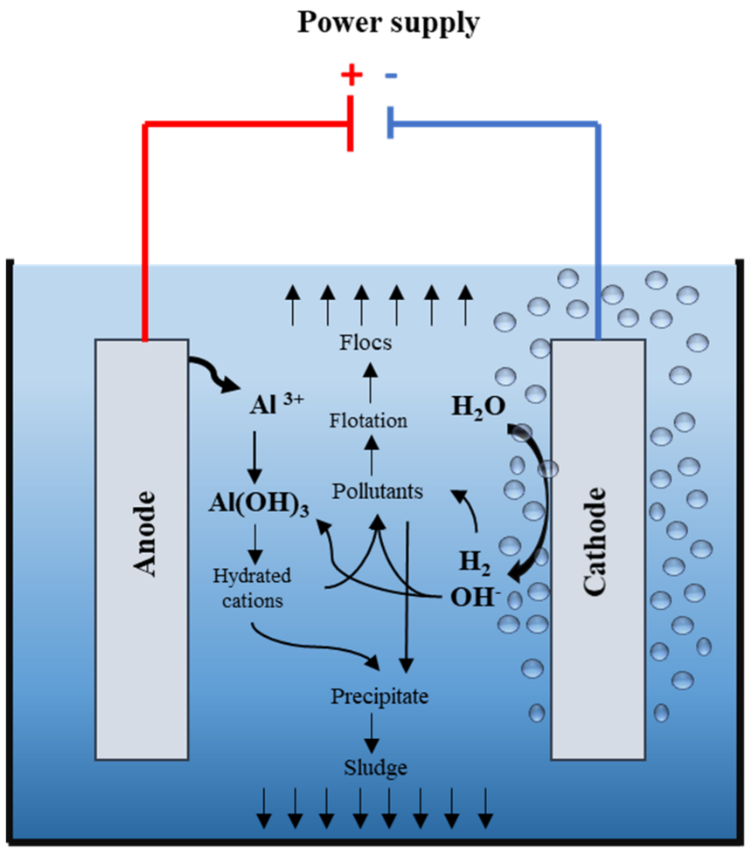

Figure 1.3: Electrocoagulation contaminant-removal process (Almukdad et al., 2021).

1.5.1 Electrocoagulation Substrates

Electrocoagulation applications include mineral processing water, municipal wastewater, and heavy-metal-laden wastewater, Applying EC to high-strength wastewater yields significant results in contaminant and pollution reduction. Jing et al. (2020) reported that EC treatment for 70 minutes at 19.23 mA/cm^2 using a Fe-anode yielded an 82.8% removal of COD (424.29 mg/L down to 72.9 mg/L) from a Pb/Zn sulfide waste stream. Intensified mineral recovery of galena and sphalerite occurred simultaneously. Similarly, EC treatment of domestic wastewater between 15-60 minutes, 9.23-45 A/m^2, and pH range of 3-9. removed 90% of COD (Bote, 2021). These studies suggest EC application can substantially remove contaminants from wastewater.

1.6 Life Cycle Assessment of Farm-Scale Anaerobic Digestion

Life cycle assessments (LCA) are a tool that evaluates the impacts of a product or process throughout its entire life. Encompassing a cradle-to-grave approach, LCAs have been used to compare the environmental impacts associated with AD of various substrates, configurations, and operational parameters (Duan et al., 2020; Fei et al., 2021; Wang et al., 2021). LCA can be broken down into four phases: 1) goal and scope: defining the boundaries, operational scenarios, functional unit, and intended goals and scope of the analysis; 2) life cycle inventory: identification of the raw material and energy inputs/outputs of the system; 3) life cycle impact assessment: evaluating and identifying environmental impacts of system components, inputs, and outputs; and 4) interpretation: analyzing system components that can be altered to minimize environmental impact and increase efficiency (Finnveden et al., 2009). Applying LCA to farm-scale AD can provide insight into pollution mitigation, broader environmental impacts, and human health impacts of the construction, operation, and output of the current system or alternative configurations.

In Maryland, agricultural operations cover 2,000,000 acres, with 40,000 dairy cattle producing over 20,000 pounds/head of milk (USDA, 2022). Previous literature reported that dairy operations with over 500 cows could produce 6.8 million MWh of electricity annually from farm-scale AD operations, although capital investments entail $1.5 million USD (Lansing & Klavon, 2014). LCA identifies processes with substantial environmental impact while providing an opportunity to optimize the efficiency of farm-scale AD systems.

1.7 Goals and Objectives

The primary goal of this research is to integrate HDC, AD, MEC and EC in a novel design to maximize energy production and increase the water recovery from DM, FW, and BW waste streams. A series of experiments explored the effect of HDC and gravity-separation on liquid and solid fractions of waste. Solids were utilized in energy production via AD and AD-MEC. The liquid fraction of the waste stream was treated with EC to recover water for low-tier reuse. The second goal of this research is to evaluate and perform LCA of an farm-scale AD operation of FW and DM to assess various impacts of the process input, operation, and outputs.

The objectives of this research were:

1. Test the ability of hydrodynamic cavitation (HDC) to increase organic material availability from food waste (FW) and blackwater (BW).

2. Determine the effect of combined AD-MEC to enhance CH_4 production from FW-BW after HDC.
3. Assess the ability of electrocoagulation to improve water quality.
4. Evaluate the energy, nutrient, and organic transformations for a novel AD farm design that digested DM and FW.
5. Determine the effects of improving the performance and operating conditions on the system's environmental impacts using LCA, including modification of substrates, solids composting, and biogas-fueled CHP efficiency.

Chapter 2: Energy Production and Waste Treatment from Food Waste and Black Water through Integrating Anaerobic Digestion, Hydrodynamic Cavitation, Microbial Electrolysis Cells, and Electrocoagulation

Abstract

This study examines a novel integration of microbial and physicochemical technologies to generate energy and treat water from combined food waste and blackwater (FW-BW) waste streams. The FW-BW substrate was gravity-separated, then pretreated with hydrodynamic cavitation (HDC) to collect a solid fraction (~10%) and a liquid fraction (~90%). Solids were used for energy generation via anaerobic digestion (AD), and AD integrated with microbial electrolysis cells (AD-MEC) operated at 0.5, 0.9, and 1.2 V. With AD of solids with HDC (HDC-AD), there was more methane (CH_4) production compared to solids without HDC applied (no-HDC AD), resulting in 51.6% more CH_4 after 5 days, 61.4% after 10 days, and 63% more CH_4 after 30 days. Additionally, AD of HDC-pretreated solids generated 81.2% of the cumulative CH_4 after 10 days of digestion. The AD-MEC at 1.2 V of post-HDC solids generated 12.7% more CH_4 compared to AD-only after 10 days. These data suggest HDC pretreatment maximizes energy generation and decreases the timeframe for CH_4

generation in AD-only. The liiquids treated with electrocoagulation (EC) at 15 V for 90 minutes removed 96.2% of the COD and 100% of the total suspended solids (TSS).

1. Introduction

Conventional anaerobic digestion (AD) microbially transforms organic material into methane (CH_4)-enrich biogas. During this biological degradation process, biogas containing CH_4 (50-70%), carbon dioxide (CO_2, 30-50%), and various other trace gasses (hydrogen sulfide (H_2S) and water vapor) is generated. Various substrates, including food waste (FW), blackwater (BW), livestock manure, and crop waste can be used for biogas production (Achi et al., 2020; Deena et al., 2022; Zamri et al., 2021). Although widely adaptable, various factors can reduce the energy production from AD, including low biogas production, destabilization, and recalcitrant compounds. The efficiency of the AD process is dependent up the temperature, substrate, hydraulic retention time (HRT), and pH within the digester. Co-digestion can increase biogas production by combining carbohydrate-rich substrates with low-C substrates (Karki et al., 2021). Previous literature reported that co-digestion of FW and sewage sludge improved CH_4 generation by over 85%, while lowering HRT by accelerating the degradation of complex lignin and carbohydrate-rich FW due to increased microbial activity (Karki et al., 2021; Lijó et al., 2014; Mehariya et al., 2018). Hydrolysis is generally regarded as the rate-limiting step (Atelge et al., 2020). VFA production is also dependent upon digester pH. High pH can inhibit enzymatic activity for VFA production, however, low pH alters VFA generation and inhibits acidogenic microorganism activity (Eryildiz & Taherzadeh, 2020; Lu et al., 2020). Due to the sensitivity of AD performance, additional treatment technologies can be incorporated to increase substrate degradability, process stability.

Bioelectrochemical systems (BES) encourage the colonization of electrochemically active microorganisms. BES can be integrated with AD to increase CH_4 yield, accelerate recalcitrant compound degradation, and promote process stability (Chandrasekhar et al., 2022; Wang et al., 2022; Yu et al., 2018). Microbial electrolysis cells (MECs) are a bioelectrochemical technology that can enhance substrate degradation by increasing syntrophic interactions of electrochemically active and inactive microorganisms in AD (Reddy et al., 2022). Reddy et al. (2022) determined that external voltage application to electrodes in MEC enriches microbial activity in the biofilm, accelerating the breakdown of OM to hydrogen gas (H_2) and CH_4. OM oxidation in MEC generates electrons at the anode that travel through the external circuit to the cathode. The electrons generate H_2 after combining with free protons, enriching archaea communities that generate CH_4 via hydrogenotrophic methanogenesis (Wang et al., 2022). Consequently, AD-MEC provides an opportunity for clean H_2 production and CH_4 generation from organic wastes via electrochemically induced activity. Previous research of AD-MEC vs AD-only determined that AD-MEC had 98% higher COD removal than AD-only, lowered CO_2 content in the biogas (2% vs 43.2%), and produced 3-6 times more CH_4 (Wang et al., 2021; Zou et al., 2021).

These data suggest that increased microbial diversity in AD-MEC increases H_2 and CH_4 production over AD-only via metabolic conversion from CO_2. Existing literature has indicated that digesting carbohydrate-rich substrates in AD-MEC can boost CH_4 production nearly 2 fold and accelerate reactor stabilization 4 times faster than AD (Park et al., 2018), although low COD can inhibit the performance of AD-MEC.

Physicochemical pretreatment technologies, like HDC, offer an approach to further valorize complex, recalcitrant substrates like lignocellulosic material, and increase the bioavailability of OM (Nagarajan & Ranade, 2021). Pretreatment technologies before AD increase the biodegradability of OM with the goal of valorizing biomass and reducing hydrolytic timeframe (Lanfranchi et al., 2022). Hydrodynamic cavitation (HDC) influences substrate parameters via a physicochemical approach, while increasing COD and substrate availability (Langone et al., 2015; Patil et al., 2016). By inducing rapid pressure and temperature changes across a rotating blade, nucleation and collapse of cavities in the substrate enhance free radical generation and subsequent chemical reactions (Nagarajan & Ranade, 2021). Additionally, HDC is advantageous over thermal hydrolysis pretreatment and biological pretreatment as it requires less energy input than, water consumption, decreased operation costs, and lower treatment time compared to other pretreatment technologies (Castro-Muñoz et al., 2023; Dular et al., 2016; Fedorov et al., 2022).

Electrocoagulation (EC) is an adaptable and cost-effective approach to treating wastewater (Tahreen et al., 2020). EC requires less economic input compared to chemical treatment and a lower footprint compared to biological treatment (Das et al., 2022). Previous studies have demonstrated the ability of EC as a water recovery and contaminant extraction technology by removing COD and total suspended solids (TSS) from wastewater (Bazrafshan et al., 2013; Mollah et al., 2001; Moreno-Casillas et al., 2007). Requiring less additives than conventional coagulation technologies, EC employs an electrical current to a metal electrode that induces sacrificial ion dissolution (Tahreen et al., 2020). Metal ions flocculate pollutants by electrostatic action before accumulating at the surface of the solution via flotation. Additionally, cathodic processes generate O_2 and H_2 as a useable byproduct for energy production (An et al., 2017). Previous EC research determined that addition of KCl enhances EC by nullifying the buildup of oxide films on the electrode surfaces (Shalaby et al., 2014).

Previous literature investigating the integration of AD with MEC has produced promising results by increasing substrate degradation rate 1.7-fold and lowering the timeframe for biogas production compared to AD while also obtaining an energy efficiency over 400% (Hassanein et al., 2017; Yu et al., 2018). However, limited studies have investigated the HDC of isolated solids before AD-MEC application. Moreover, research incorporating the benefit of EC treatment of isolated liquids combined with energy generation from AD-MEC is insufficient. The following research provides a novel approach to waste treatment by integrating HDC pretreatment, AD-MEC energy generation from FW-BW solids, and EC water treatment to maximize resource recovery from FW-BW liquids. The results will provide data that helps fill a knowledge gap regarding the combination of these four technologies. This research had the following objectives: 1) investigate the potential of HDC pretreatment to increase organic material availability of FW-BW waste stream; 2) explore how AD-MEC enhances CH_4 production from FW-BW solids over AD-only; and 3) assess the ability of EC to improve water quality from FW-BW liquid. This research will provide knowledge to fill the gaps on energy production and water treatment from a combined FW-BW waste stream, while assessing the novel integration of AD, MEC, HDC, and EC.

2. Materials and Methods

2.1 Substrate and Inoculum

The food waste (FW) substrate used in this study was prepared based on a recipe from previous literature where 567 g of white bread, 907 g of canned pork and beans, and 176 g of dried potato flakes were blended and homogenized (Yarberry et al., 2019). The resultant paste was mixed with 1 L of distilled water producing an average COD of 365 mg/L in the FW. The BW substrate was waste-activated sludge (WAS) supplied from a wastewater treatment plant in Maryland, USA. The WAS contained an average COD of 9,984 mg/L ± 1,448 mg/L. The combined FW-BW waste stream was supplemented with 1 L of DI water to reach a COD of 3,100 mg/L, where 50% of the

COD came from the BW portion and 50% from the FW portion of the waste stream to simulate blackwater (flush toilet) and food waste (rinse water) mixture collected from a household or outpost. The FW-BW waste stream was settled in 1 L Imhoff cones (Cole-Parmer, UX-25000-28, Chicago, IL, USA) for 30 minutes. Solids were collected from the bottom and averaged 0.160 L/L of FW-BW), with 84% of the volume being in the liquid portion (0.840 L/L of FW-BW mixture). Solids and liquid waste fractions were stored and processed separately. The AD inoculum was procured from a semi-continuous lab-scale digester (University of Maryland, College Park, MD, USA) in the experiment had a COD of 5,766 mg/L, a TS of 1.16%, and a VS of 0.84%. The liquid fraction of the gravity-separated FW-BW waste stream (84% of the waste stream) was treated using EC, while the solid fraction was used for energy generation from the AD-MEC system.

2.2 Experimental Overview

Food waste and blackwater were blended and combined as the initial substrate (raw FW-BW). The combined raw FW-BW was gravity settled for 30 minutes to separate the waste stream into solids (0.16 L) and liquids (0.84 L). After gravity separation, HDC was applied to the solids, liquids, and raw FW-BW to increase the COD in each waste stream (Figure 2.1). After HDC, the solids were used for incorporated into two separate energy generation experiment, while the liquid was treated for contaminant removal in two additional experiments. AD-only was utilized to test the effect of 5, 10, and 30-day HRT on energy generation from no-hydrodynamic cavitation (no-HDC) and post-hydrodynamic cavitation (post-HDC) solids. Post-HDC and no-HDC solids were also tested for energy generation via combined (AD-MEC) reactors.

Microbial electrolysis cells consisting of a non-membraned chamber with an anode and cathode were operated at 0.5, 0.9, and 1.2 V for 10 days to investigate the potential of AD-MEC to enhance energy generation from no-HDC and post-HDC solids compared to AD-Only. The liquid fraction (no-HDC and post-HDC) of the waste stream was treated with two iterations of EC. EC was initially conducted at 15, 20, and 25 V to analyze the impact of voltage on contaminant and COD removal in no-HDC and post-HDC liquids. EC was reapplied to raw FW-BW, no-HDC, and post-HDC liquids with KCl added to increase conductivity. Both liquids experiments assessed the ability of EC to reach parameter reductions for water reuse in a low-tier capacity.

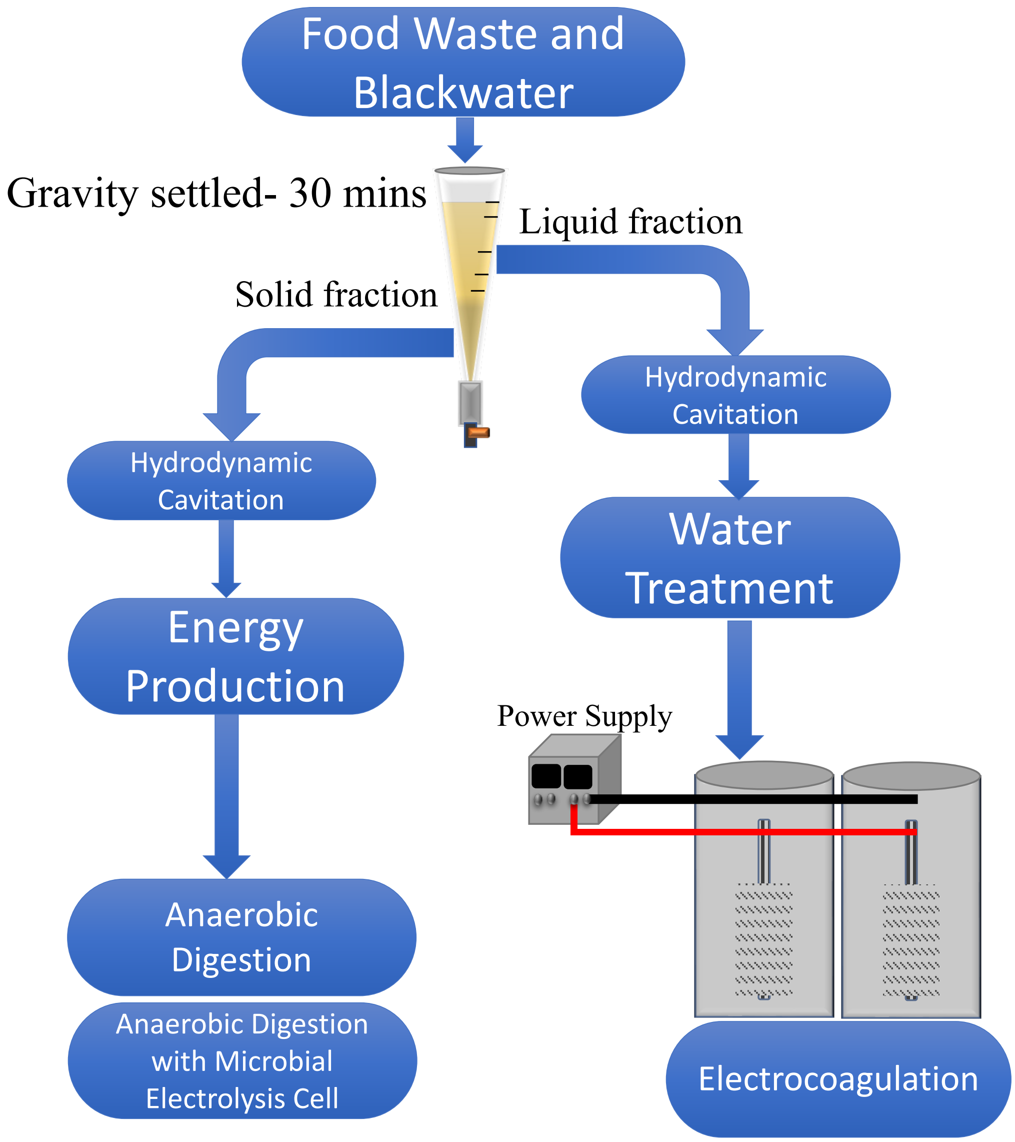

Figure 2.1: Treatment process of food waste and blackwater (FW-BW) via gravity separation, hydrodynamic cavitation pretreatment (HDC), anaerobic digestion (AD), anaerobic digestion with microbial electrolysis cell (AD-MEC), and electrocoagulation (EC).

Table 2.1: Experimental design for energy production and water quality improvement from food waste and blackwater (FW-BW).

Experiment	**Substrate Treated**	**Liquids, Solids, or No Separation**	**Technology Used**	**Experiment Overview**
AD-Only of Solids	NO-HDC Solids and POST-HDC solids	Solids-only digestion after settling	AD-Only	• Energy generation from solids • AD-Only • 5, 10, 30-Day HRT
AD-MEC of Solids	NO-HDC Solids and POST-HDC solids	Solids-only digestion after settling	AD-MEC	• Energy generation from solids • AD-MEC • 0.5, 0.9, 1.2 V
EC-Voltage	NO-HDC Liquids and POST-HDC Liquids	Liquid-only after settling	EC	• Water treatment • EC • 15, 20, and 25 V, for 30, 60, and 90 minutes
EC-Conductivity	1.) RAW, NO-HDC Liquids and POST-HDC Liquids	Liquid after settling	EC	• Water treatment • EC • KCl addition • 10 V for 15 minutes

2.3 Hydrodynamic Cavitation Reactor Design and Cavitation Procedure

The HDC reactor (6 L) consisted of a 15.2 cm x 30.4 cm baffled stainless-steel cylinder. HDC was generated using a proprietary rotary generator driven by a variable-speed electric motor. The rotary generator was specially designed to mix the solution, reduce the sizes of solid particles, and generate high-intensity cavitation in the solution. Two 600 W Incoloy heat elements were placed inside the reactor and controlled using an electronic PID temperature controller with thermocouple feedback. The reactor was pressure tight to allow subcritical heating of the FW-BW waste stream. Two HDC designs were tested at varying rotation speeds. A temperature of 110 °C and a pressure of 26 psi (5 psi above the vapor pressure of the water at 110 °C) were utilized to maintain subcritical conditions for the aqueous solution. Subcritical conditions were held for 30 minutes after the 25 minutes required for the reactor to reach >100 °C.

After gravity separation, no-HDC liquids had an influent COD of 930 mg/L, TS of 0.12%, and VS of 0.10%, while post-HDC liquids had an influent COD of 2,210 mg/L, TS of 0.16%, and VS of 0.14%. The no-HDC solids had an influent COD of 42,160 mg/L, a TS of 3.32%, and a VS of 3.10%. Post-HDC solids had an influent COD of 52,580 mg/L, 2.42% TS, and 2.22% VS. Inoculum had an influent COD of 5,766 mg/L, 1.16% TS, and 0.86% VS.

2.4 Experimental Design

The AD-MEC reactors consisted of a 250 mL serum bottle with a bioanode and biocathode inserted through a sealed rubber septum. The bioanode consisted of a 2-cm thick graphite rod (Graphtek, B000RC0SZ8, Northbrook, IL, USA) cut at 11 cm. A piece of 16-gauge titanium wire (TEMCo, RW0469, Fremont, CA, USA) was inserted through the graphite rod to suspend the bioanode. The biocathode consisted of fine steel mesh wrapped around a piece of 16-gauge titanium wire secured with rubber bands. Bioelectrodes were covered with shrink-tubing to eliminate contact between the bioanode and biocathode. The bioelectrodes were connected to a voltage regulator (Circuit Specialists Inc., CSI3645A, Tempe, AZ, USA) maintained between 0.5-1.2 V (Table **2.1**2.1).

2.4.1 Solids and Liquids Experiment Characterization

The first energy generation experiment tested AD-Only with no-HDC and post-HDC solids under mesophilic conditions (35 °C). Reactors consisted of a 250-mL serum bottle sealed with a septum. Each 250 mL reactor was loaded with 110 g of inoculum (COD 5,766 mg/L, 1.16% TS, and 0.86% VS). No-HDC AD-only reactors contained 70.2 g of no-HDC solids (3.32% TS and 3.10% VS) to reach a 2:1 inoculum-to-substrate ratio (ISR). Post-HDC AD-only reactors were loaded with 48.7 g of post-HDC solids (2.42% TS and 2.22% VS) to reach a 2:1 ISR based on VS. The experiment was run for 30 days, with solids sampling at 5, 10, and 30-days, and biogas sampling every day until gas production decreased. The septum was sealed with silicone after each sample extraction to minimize leakage.

Three voltages were tested (0.5, 0.9, and 1.2 V) for the AD-MEC energy generation experiment conducted at mesophilic conditions (35 °C). AD-MEC reactors (250-mL serum bottles). AD-MEC no-HDC reactors contained 28.9 g of no-HDC solids. AD-MEC post-HDC reactors contained 28.4 g of solids to reach a 2:1 ISR. All reactors in the energy generation experiments were continuously agitated and maintained at mesophilic temperatures in an incubator. AD-MEC reactors were connected to the power supply, with each circuit (anode and cathode) tested with a voltmeter to ensure conductivity.

The EC voltage experiment was run at 15, 20, and 25 V. The initial COD concentration of the post-HDC liquid was 5,330 mg/L, while no-HDC liquids possessed 2,960 mg/L COD. Each treatment group was run in triplicate. Samples were collected at 30, 60, and 90 min, with collection occurring 5 cm below the liquid surface (to avoid flocculant collection) using a Van Rogers and Waters 5-mL micro-pipette (989A0171, VWR, Radnor, PA, US). Collected samples were placed into a 15-mL centrifuge tube and analyzed for TSS and COD removal.

The second EC experiment analyzed no-HDC liquid, raw FW-BW, and post-HDC liquids using less voltage input and increased electroconductivity with added electrolytes. The same reactors used in the first experiment were employed, with the addition of KCl (1 g/L) to each reactor. Based on the preliminary experiment and current literature, KCl was added to boost the efficiency of EC via increased conductivity and subsequent current density through the liquid (Kadier et al., 2022). Prior to the addition of KCl, the electroconductivity of the no-HDC liquid was 48 μS/cm, post-HDC liquid was 513 μS/cm, and raw FW-BW was 282 μS/cm. After KCl, the conductivity of the no-HDC liquid was 2,323 μS/cm, with post-HDC liquid and raw FW-BW at 2,296 and 2,586 μS/cm, respectively. Voltage was maintained at 10 V with samples collected at 5, 10, and 15 minutes. Samples were collected with a 5-mL VWR micro-pipette before storage in 15-mL centrifuge vials for analysis.

2.4.2 Electrocoagulation Reactor Design

Electrocoagulation experiments utilized 600 mL aluminum (Al) reactors (450 mL working volume fabricated at the AGNR Shop, University of Maryland, College Park. The Al reactors were 18 cm tall, with a diameter of 7.5 cm. The circumference of the reactors was 24.5 cm. A 7 cm steel brush attached to a cylindrical section of Styrofoam (4 cm diameter by 2 cm thick) was placed inside each reactor and connected to the power supply to act as a cathode. Steel wire (24.5 cm) encircling the outside of the reactor supplied electricity to the anode (Figure 2.2). Voltage (10-25 V) was applied the anode and cathode between 5-90 minutes.

2.5 Analytical Methods

Biogas samples from the energy generation experiments were collected daily during the first week using a 50-mL gas-tight glass syringe. Biogas (0.5 mL) was injected and analyzed for CH_4 content via gas chromatography (GC) in an Agilent HP 7890 A gas chromatograph (Agilent Technologies, Santa Clara, CA, USA) equipped with a thermal conductivity detector (TCD) and a single HP porous layer open tubular (PLOT) Q column at injection temperature 250 °C, detector temperature 250 °C, and oven temperature 250 °C. Biogas samples from the AD-MEC experiment were collected with a 50-mL gas-tight, glass syringe daily through the first week, then every other day for the rest of the 10-day experiment. Collected biogas (0.5 mL) was analyzed for CH_4 quantity in the Agilent HP 7890 A GC. All reactors were analyzed for TS (TS, Method 2540B) and VS (VS, Method 2540E). COD concentration was analyzed with a Hach DR 5000 spectrophotometer (Hach Company, DR5000-03, Loveland, CO).

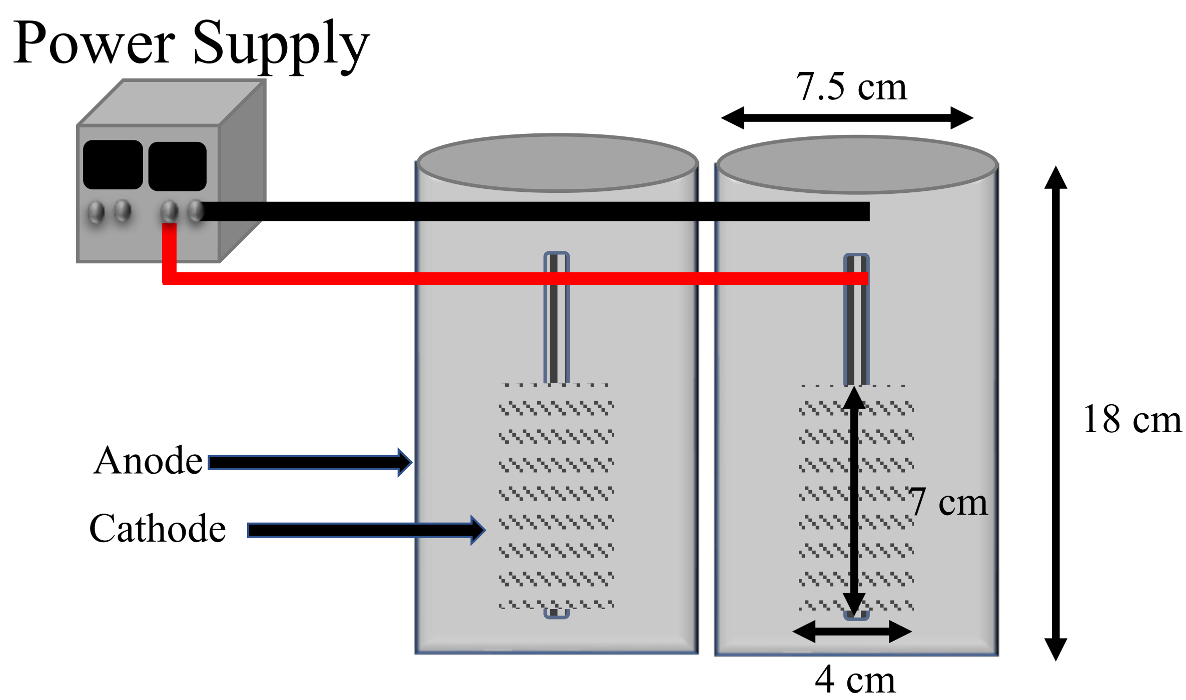

Figure 2.2: Aluminum electrocoagulation reactors (450 mL) design displayed with connection to the power supply, anode, and cathode.

2.6 Statistical Analysis

The data obtained were analyzed via analysis of variance (ANOVA). Data were compared for CH_4 production (cumulative CH_4 and mL CH_4 g/VS) from solids and percent COD removal from liquids for statistical significance (*p*-value <0.05). Tukey honestly significant variance (HSD) tests were applied to all analyses for multiple comparisons between different variables. Statistical analysis was performed with

StatPlus add-on for Microsoft Excel (Version v7, AnalystSoft Inc., Brandon, FL). Data are presented as averages with standard error (SE).

3. Results and Discussion

3.1 Effect of HDC on Organic Material Availability

Gravity separation was employed allowing for the isolated solid and liquid fractions of FW-BW to be analyzed individually for COD. The COD of the raw FW-BW substrate without HDC (no-HDC) was 2,960 mg/L. After HDC pretreatment, the COD of the FW-BW (post-HDC) substrate increased to 5,330 mg/L. Post-HDC solids contained 52,580 mg/L of COD, while no-HDC solids had 19.8% lower COD concentration (42,160 mg/L). Post-HDC liquids had 57.9% higher COD concentrations (2,210 mg/L) than no-HDC liquids (930 mg/L).

The results showed a substantial increase in COD from HDC of FW-BW. Previous literature has reported COD increases of 45% after HDC treatment of brewery-spent grain suspended in mature landfill leachate (Lebiocka et al., 2021). Lebiocka et al. (2021) found that the cavitation process increased OM concentration along with OM biodegradability. The substantial COD increase observed in this study was attributed to the nutrient rich FW. Similarly, HDC treatment of vegetable waste and sewage sludge for 50 minutes at 2.0 bar and 17 kW of applied power increased sCOD by 83% (Lanfranchi et al., 2022). HDC has also been shown to increase COD, where HDC employed as a water-treatment technology improved COD and bacterial reduction by 30-60% with decreased pressures (Doltade et al., 2019; Thanekar & Gogate, 2019). Compared with alternative literature, HDC has exhibited variable impacts on COD concentration, typically influenced by operational conditions (time, pressures, additives) (Agarkoti et al., 2021; Joshi & Gogate, 2019). HDC of industrial wastewater was shown to remove 63% of COD from a 70 L reactor over 180 minutes (Joshi & Gogate, 2019) However, Joshi and Gogate's (2019) study incorporated Fenton's reagent (H_2O_2 and $FeSO_4$), oxygen, and H_2O_2 loaded at 15 g/L. Their results identified increased oxidation was caused by the free radical generation from the additives. By promoting higher COD levels in the influent, this research determined that HDC has a considerable ability to increase substrate bioavailability for subsequent CH_4 generation.

3.2 Effect of HDC Process on Methane Generation in AD-Only

The AD-only experiment analyzed the effect of HDC pretreatment on energy production from solids with and without HDC pretreatment after 5, 10, and 30-day digestion. After 5 days, CH_4 production was 51.6% higher in post-HDC than no-HDC (*p*-value = 0.0001). After 10 days, post-HDC generated 61.4% more CH_4 compared to no-HDC (*p*-value=0.0001) (Figure 2.3B). At the end of the 30-day experiment, post-HDC (429 mL CH_4/g VS) produced 63% more cumulative CH_4 compared to no-HDC (263 mL CH_4/g VS, *p*-value=0.0001). In the solids with HDC pretreatment, 62% of the cumulative CH_4 production occurred in the first 5 days of the experiment, with 81.2%

of the cumulative CH_4 was generated after 10 days of digestion (Figure 2.3A). These data indicate that HDC pretreatment of FW-BW solids generated a significant difference in CH_4 generation while decreasing the HRT needed for maximum energy production.

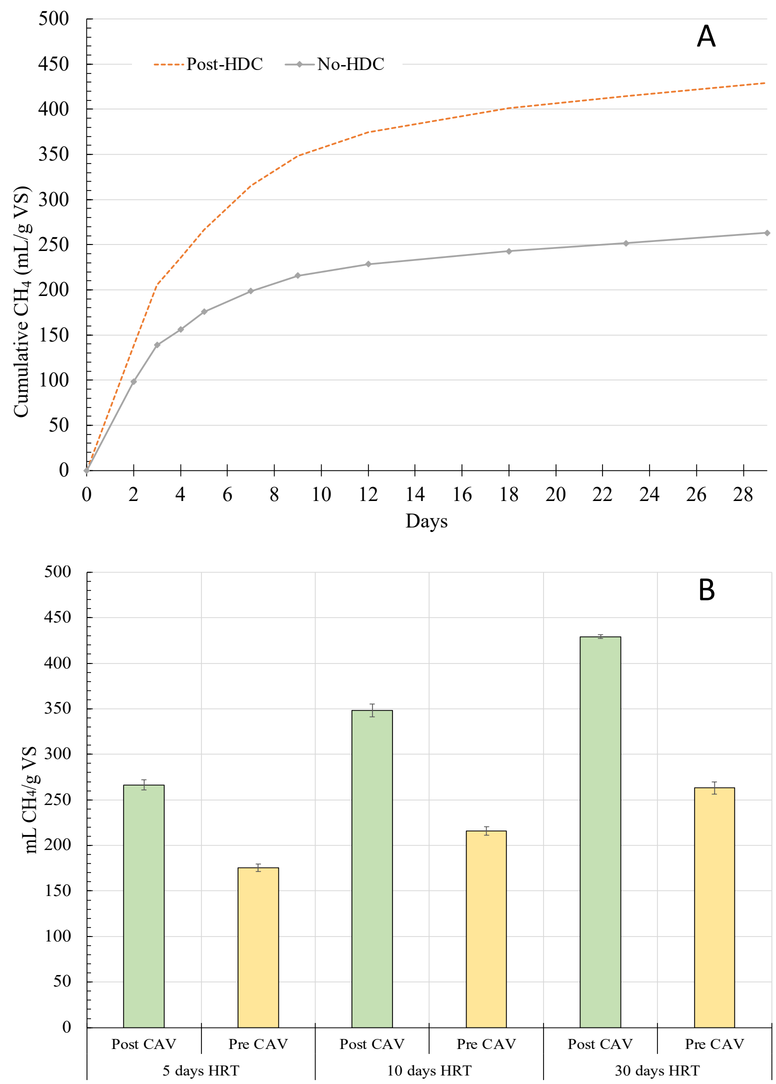

Figure 2.3: CH_4 production (mL/g VS) after 5, 10, 30-days (B), and cumulative CH_4 production (mL/g VS) over 30 days of anaerobic digestion (A) with standard error.

These results highlight how HDC has a positive effect on biogas production. The 886% COD increase in the solids generated by HDC amplified biogas production, where the 63% CH_4 increase in post-HDC solids (429 ±2.1 mL CH_4/g VS) compared to no-HDC solids (263 ±6.5 mL CH_4/g VS, *p*-value=0.0001) was attributed to the heightened microbial activity due to greater bioavailability of OM consumed during AD. Furthermore, with 81.2% of cumulative CH_4 (348 ±7.2 mL CH_4) generated in the first 10 days, HDC expressed a considerable reduction in the rate-limiting hydrolysis process. Previous literature supports these results, where HDC treatment before AD increased microbial productivity, increased substrate availability and biodegradability, and increased CH_4 generation (Blagojevič et al., 2023; Garlicka et al., 2020; Lanfranchi et al., 2022; Nagarajan & Ranade, 2021; Patil et al., 2016). In a similar study, HDC treatment resulted in a 144% increase in CH_4 generation compared to untreated wheat straw (Patil et al., 2016). Applying HDC to wheat straw resulted in a higher increase in CH_4, compared to the FW-BW substrate, due to the dissociation of lignin from radicals generated during the cavitation process (Zieliński et al., 2019). Additionally, HDC has been demonstrated to effectively valorize FW and WWTP sewage sludge, with HDC treatment increasing biogas production by over 32% (3.6 m^3/ kg substrate to 5.1 m^3/ kg substrate) for FW and sludge (Lanfranchi et al., 2022), and 152% for sludge (Garlicka et al., 2020). Although this study was similar to Lanfranchi et al. (2022) and Garlicka et al. (2020), temperatures were sustained at 37 °C in the Lanfranchi et al. (2022) experiment and 36 °C in the Garlicka et al. (2020) experiment, compared to 35 °C in the current study. Nonetheless, these results indicate that HDC positively impacts methanogenic efficiency in AD of carbohydrate-rich substrates.

3.3 Effect of HDC on TS/VS in AD-Only

The TS and VS concentrations of no-HDC and post-HDC solids before and after digestion are shown in

Figure **2.4**. After 30-days of AD, post-HDC solids displayed a VS removal of 28.1 ±0.01%, while no-HDC solids had a 12.7 ±0.02% VS removal. The VS reduction is contributed to the microbial conversion of solids to biogas. One study reported that HDC applied to brewery spent grain (BSG) decreased VS by 23% (5.5 g/kg down to 4.4 g/kg), although the authors noted that BSG maintained a naturally high VS content typically alleviated by mechanical dilution (Montusiewicz et al., 2017). The work by Montusiewicz et al. (2017) are consistent with this study, finding that HDC elevated VS removal by 55% in post-HDC solids compared to no-HDC solids after 30 days of digestion. However, one previous study examining HDC application to milk protein found no change in VS of the wastewater after treatment (Li et al., 2018).

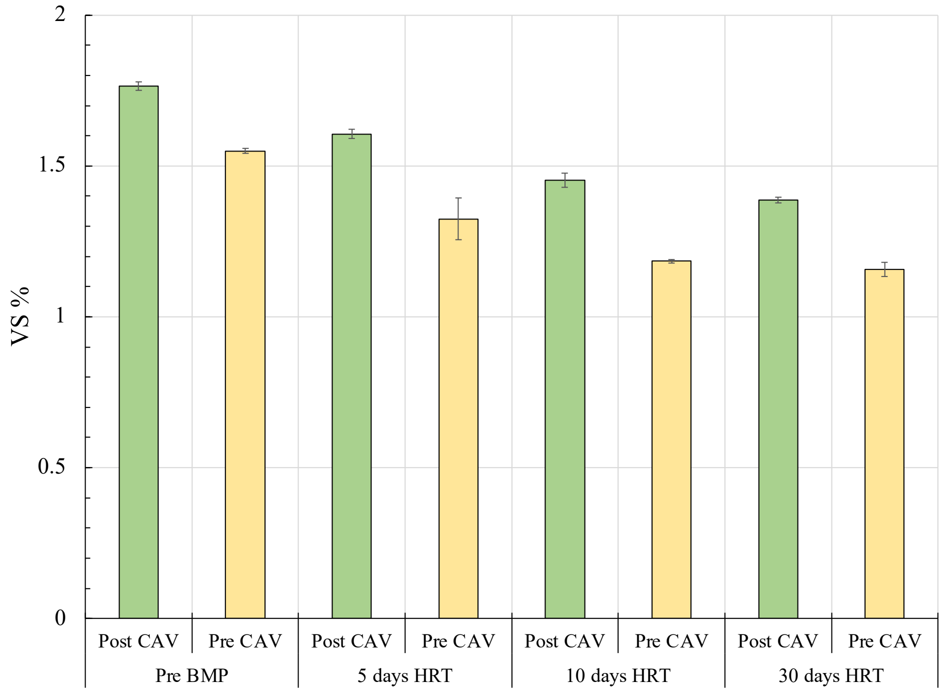

Figure 2.4: Volatile solids (%) of no-HDC and post-HDC solids after 5, 10, and 30-days of digestion with standard error.

3.4 Energy Generation from AD-MEC

The second energy generation experiment investigated the energy production of post-HDC and no-HDC solids in an AD-MEC system. The MECs were operated at 0.5, 0.9, and 1.2 V. After 10 days of digestion, AD-MEC of post-HDC at 0.9 and 1.2 V produced 9.8% and 12.7% more cumulative CH_4 (285 mL CH_4/g VS and 293 mL CH_4/g VS, respectively) compared to AD-only (259 mL CH_4/g VS). AD-MEC of no-HDC waste at 0.9 V produced 5.3% more cumulative CH_4 compared to AD-only (Figure 2.5). Furthermore, the results indicate that the inclusion of MEC with AD decreased the HRT for CH_4 production. After 5 days of digestion, AD-MEC at 0.9 V produced 82% of the cumulative CH_4 compared to 69% of cumulative CH_4 production from AD-only. On the contrary, there was no statistical difference between AD-MEC treatments and AD-Only treatments. While the 10-day timeframe was utilized based on optimal CH_4 production from the AD-Only experiment, it was hypothesized that the limited HRT (10 days) in this experiment did not provide adequate conditions for electrogenic microorganisms to colonize the anode. The data trend suggests that the CH_4 generation from AD-MEC began increasing over AD-only in the final two days of the experiment. From these findings, it can be implied that longer timeframes are pivotal for the formation of microbial communities enriched with electrochemically active microbes.

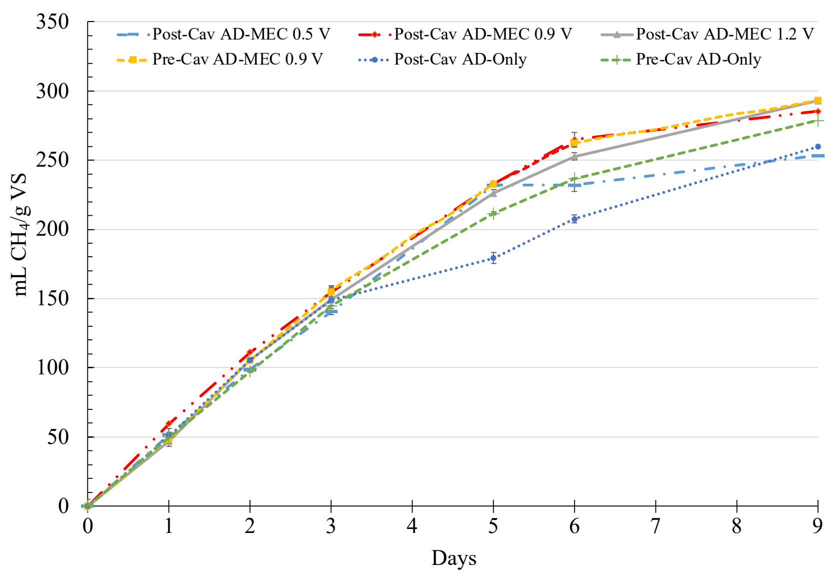

Figure 2.5: Cumulative CH_4 production after 10 days, normalized by grams of volatile solids (g/VS) with standard error shown by error bars.

Previous literature reported optimal CH_4 production occurs when MEC is integrated with AD. A similar analysis integrating AD-MEC reactors with swine manure were operated at 0.3, 0.6, 0.9, 1.2, and 1.5 V for 22 days under mesophilic conditions (Zou et al., 2021). AD-MEC 0.9 V generating the highest cumulative biogas and CH_4-content within the biogas at 36.25%, 53.17%, and 12.43% higher than the control, respectively (Zou et al., 2021). Zou et al (2021) found that AD-MEC at 0.9 V produced over 23% more cumulative CH_4 (347.7 mL CH_4/g TS) compared to AD-MEC at 1.5 V (275.7 mL CH_4 /g TS), while AD-MEC at 0.9 V produced 25% more cumulative CH_4 compared to AD-MEC at 0.3 V (269.8 mL CH_4/g TS). Additionally, Zou et al. (2021) stipulated that excessive voltage inhibits microorganism growth and metabolism, decreasing the biotransformation of organic material, with AD-MEC at any voltage accelerates lignocellulosic disassociation, although AD-MEC 0.9 V maximized complex feedstock destruction.

3.5 Effect of EC on Improving Water Quality

3.5.1 Electrocoagulation at 15, 20, and 25 V

The results of this experiment showed that the application of EC at all voltages to the liquid FW-BW waste stream had positive impacts on contaminant removal for both no-HDC and post-HDC (Figure 2.6). EC at 15 V and 30 minutes of post-HDC liquids yielded a 90.7% decrease in COD concentration (494 mg/L) from raw FW-BW post-HDC (5,330 mg/L, p=0.003). EC at 20 V and 30 minutes was less effective, with an 83.8% decrease in COD concentration (864 mg/L, p=0.006). EC of post-HDC liquids at 25 V and 30 minutes removed 83.9% of COD (858 mg/L), which was less efficient than 15 and 20 V (p=0.006). As the time applied increased, post-HDC liquid COD removal increased to 96.2% (200 mg/L) at 15 V and 90 minutes of treatment (p=0.0006). EC of post-HDC liquids at 25 V at 90 minutes saw the lowest COD decrease of all three treatments at 91.4% (460 mg/L), with significantly more COD removed compared to raw FW-BW (p=0.001). The raw FW-BW COD post-HDC before EC was 5,330 mg/L. Compared to raw FW-BW (5,330 mg/L) the post-HDC liquid EC treatment at 15 V-90 minutes (96.2% COD removal, 200 mg/L) exhibited significantly more COD removal than 20 V at 60 minutes (86.7% COD removal, 521 mg/L, p=0.04), 25 V and 30 minutes (83.9% COD removal, 858 mg/L, p=0.01), and 20 V 30 minutes (83.8% COD removal, 864 mg/L, p=0.01). However, all post-HDC liquid treatment groups saw significant COD removal compared to raw FW-BW. These data suggest that 15 V and 90 minutes of EC treatment were optimal for reducing water parameters, justifying a lower electrical input to remove contaminants.

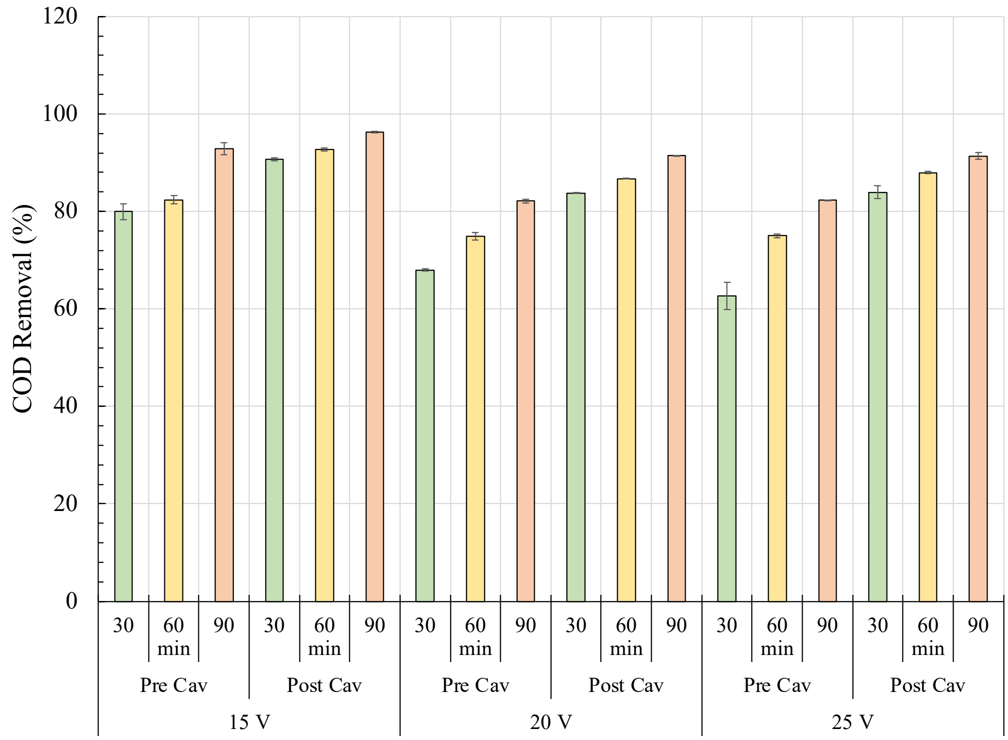

Figure 2.6: Percent chemical oxygen demand (COD) removal after 30, 60, and 90 minutes of electrocoagulation (EC) treatment on no-HDC and post-HDC liquids at 15, 20, and 25 V with standard error.

These results supported previous studies where substantial COD and TSS removal was observed from separated liquids with EC application. EC was applied from 10-60 V over 60 minutes to settled dairy wastewater resulting in 98.8% COD removal (6,114 to 70 mg/L) and 97.7% TSS removal from the influent (Bazrafshan et al., 2013). This research showed similar levels of COD removal with less electrical input, albeit a longer timeframe. On the contrary, Bazrafshan et al. (2013) used 2 L working volume reactors, while this study employed reactors with 450 mL working volume. Bazrafshan et al. (2013) also employed gravity settling to the dairy wastewater used in their experiment. However, the influent COD in their study (6,114 mg/L) was elevated compared to the COD of the raw FW-BW and post-HDC liquid influents used in this research (5,330 mg/L and 2,210 mg/L COD, respectively). Bazrafshan et al. (2013) determined that contaminant removal heightened with increasing voltage, however, 50% of wastewater parameters were eliminated within the first 15 minutes of treatment at 30 V. On the other hand, this research observed over 96% COD removal from post-HDC liquids after 30 minutes with half the voltage (15 V) applied.

Compared to raw FWBW without HDC (2,960 mg/L of COD), applying EC to no-HDC liquids at 15 V and 30 minutes removed 80% of COD (593 mg/L). EC at 20 V and 30 minutes demonstrated considerably less COD removal (67.9%, 948 mg/L), while 25 V of EC yielded the least COD removal after 30 minutes (62.7%, 1,105 mg/L). After 60 minutes of EC treatment to no-HDC liquid, 15 V and 20 V generated identical COD reductions (82.3%, 521 mg/L) with 25 V yielding the lowest COD removal (75%, 740 mg/L). EC of no-HDC liquid at 15 V removed 92.8% COD after 90 minutes (212 mg/L), where 20 V and 25 V demonstrated similar COD removals (82.1%, 529 mg/L and 82.2%, 525 mg/L, respectively). Differing from post-HDC liquid, none of the no-HDC liquid EC treatments exhibited significant differences between voltage, timeframe, or raw FW-BW. Moreover, there was no significant difference between post-HDC and no-HDC COD removal at voltage or timeframe. On the other hand, all treatment groups experienced 100% removal of TS, TSS, and significant color removal from the solution at the end of the experiment.

These results indicate EC can treat high COD and TSS/TS wastewater and enable the reutilization of wastewater. Nevertheless, the results of this research exhibited similar COD and TSS removal with lower voltage input (15 V). Furthermore, electrode efficiency impacts the availability and activity of sacrificial metal ions generated at the anode.

The data exhibited substantial COD removal from no-HDC (>62%) and post-HDC liquid (>83%) in aluminum (Al) reactors. Incidentally, electrode efficiency can impact EC performance, therefore, electrode materials have been analyzed for their contaminant removal efficiency (Chezeau et al., 2020; Devlin et al., 2019; Ingelsson et al., 2020). Al and Fe electrodes have been explored in their ability to remove contaminants from wastewater, where Al electrodes have been suggested to accelerate contaminant removal, however, Fe electrodes lower economic input (Chezeau et al., 2020). A similar study treating dairy wastewater influent (COD of 5,600 mg/L) applied 2.5 and 5 V at 120 minutes to combinations of Al and Fe electrodes (Akansha et al., 2020). Applying a Fe anode and Al cathode exhibited the largest COD removal at 74%. While COD removal was lower than in this study, the applied voltage was substantially lower (5 V) in Akansha et al. (2020) study. By comparison, this study's lowest COD removal from 15 V of post-HDC liquid was nearly 17% greater than the highest COD decrease observed in Akansha et al. (2020) without additives. Contrarily, Akansha et al. (2020) analyzed the addition of air into the EC process, resulting in a COD removal of 86.4% from an Al-Fe electrode combination. It was established that not only did air injection improve organic pollutant removal in EC for all electrode combinations, but an applied voltage of ≥ 5 V is necessitated to generate adequate metal hydroxides from the anode. On the other hand, this study found higher voltages (≥15 V) over a narrow timeframe (30 min) without incorporating additives such as electrolytes or air injection are sufficient for wastewater parameter reduction. Additionally, parameter reduction occurs cheaply and swiftly in Al EC reactors. Regardless, EC is advantageous and environmentally friendly compared to alternative flocculation technologies (chemical coagulation, biological treatment) as it necessitates minimal electrical, mechanical, and chemical input while expediting contaminant removal (Akansha et al., 2020; Ingelsson et al., 2020).

3.5.2 Effect of Increased Electric Conductivity on Electrocoagulation Performance

An additional EC experiment was performed to determine the ability of high electroconductivity (increased electrolyte content) and low voltage (10 V) on contaminant removal from raw FW-BW, no-HDC liquid, and post-HDC liquid (Figure 2.7). Potassium chloride (KCl 1 g/L) was added to each liquid to increase electroconductivity. Samples were collected at 5, 10, and 15 minutes with analysis conducted on COD removal. The addition of HDC to the liquid fraction of the FW-BW stream resulted in higher COD removal from the EC process compared to raw FW-BW due to the increase of organic material removed by flocculation.

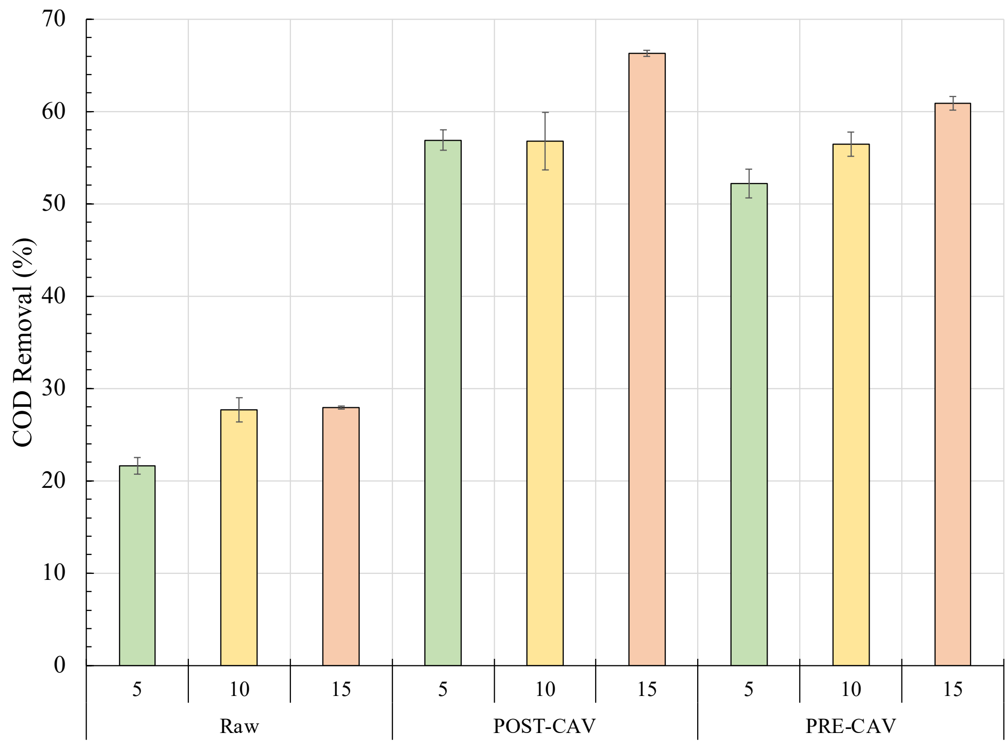

Figure 2.7: Percent chemical oxygen demand (COD) removal of raw food waste and blackwater (FW-BW), post-HDC liquid, and no-HDC liquid after 5, 10, and 15 minutes of electrocoagulation (EC) with KCl conducted at 10 V with standard error.

After 15 minutes of EC applied (with KCl) to post-HDC liquid, significantly more COD was removed compared to EC of raw FW-BW (66% vs 27%, respectively, p=0.0003). Similarly, no-HDC liquid saw significant COD removal compared to raw

FW-BW (60% vs 27%, respectively at p=0.0003). Moreover, no-HDC and post-HDC waste exhibited significant differences in COD removal at 5 minutes (no-HDC, 52%, p=0.0003, post-HDC 56%, p=0.0003) and 10 minutes (no-HDC, 56%, p=0.0004, post-HDC 56%, p=0.0005) compared to raw FW-BW (21% and 27%). On the contrary, although not significantly different, there was a slight variation in COD removal between no-HDC and post-HDC liquid at 5, 10, and 15 minutes.

The results of this experiment align with a similar EC experiment where greater than 60% of COD removal is observed (Tchamango et al., 2010). Compared to the first EC experiment, COD removal was slightly lower in this experiment (> 90% versus 66%, respectively), although the target for COD removal was met after 15 minutes. The lower timeframe (15 minutes vs. 90 minutes) for COD removal can be contributed to the addition of electrolytes (KCl) for increased conductivity.

Similar studies showed the addition of KCl at 1 g/L increased the contaminant removal (Aityoub et al., 2020; Hussein & Jasim, 2021). The results of this study align with previous studies, as the target COD removal was met in 15 minutes with 66% of COD removal from post-HDC liquids (1,058 mg/L to 356 mg/L, p=0.0003), while no-HDC liquids exhibited a significant increase of 61% COD removal (843 mg/L to 330 mg/L, p=0.0003) Previous research has indicated that KCl has a positive impact on pollutant removal as KCl not only increased the efficiency of the EC process via reduced electrode surface passivation, but engaged in a direct role of the EC process

(Aityoub et al., 2020). This phenomenon illustrated by Aityoub et al. (2020) explains the lower voltage and timeframe required for pollutant removal in this study. As KCl dissociates in aqueous solutions, Cl ions form complexes with flocculants while decreasing contaminant buildup (passivation) on the electrode surface.

The data obtained in this study imply that HDC does not significantly impact COD removal in EC. Consequently, HDC application to liquid before EC would most likely not be a fruitful allocation of resources. Instead, variations in voltage, electrolyte addition, and electrode configuration/material would plausibly yield superior reductions of wastewater parameters. Moreover, these results indicate moderate voltage (10 V) with added electrolytes yields adequate wastewater parameter reduction. However, higher voltages can be applied without the addition of supporting electrolytes, although the timeframe for EC application increases (15 minutes compared to 90 minutes). Moreover, employing solid-liquid separation proves beneficial to maximizing COD removal. It can be suggested that EC application to gravity-separated liquid is sufficient for recovering water for low-tier reuse.

4. Conclusions

This work investigated the novel integration of three waste treatment technologies and one water treatment technology with the goal of maximizing energy generation and treating water from a combined FW-BW waste stream. HDC of gravity-separated solids before AD proved advantageous, exhibiting significant increases in

COD, CH_4 quantity, and CH_4 generation rate. On the other hand, HDC application to gravity-separated liquids did not generate results that justified implementation. AD-MEC did not yield substantial CH_4 production over other treatment groups. However, the 10-day experiment did not provide adequate time for communities of electrogenic microorganisms to colonize the anode. Lengthening the experimental timeframe would yield more impactful results with the expansion and diversification of the microbial consortium. EC is a promising approach to wastewater treatment. Substantial reductions in COD were observed for all treatment groups, where lower voltages at longer timeframes removed the highest percentage of COD. Overall, applying HDC before AD of solids provides a promising approach to maximizing energy production from co-digestion, while gravity separation before EC of liquids reduces water parameters to a level sufficient for low-tier water reuse.

Chapter 3: Evaluation and Life Cycle Assessment of Farm-Scale Anaerobic Digestion of Dairy Manure and Food Waste with Composting of Manure Solids

Abstract:

This study evaluates a farm-scale anaerobic digester (AD) system that co-digests dairy manure and pre-consumer food waste (dissolved air flotation waste (DAF) and cranberry) for over 13 months. Cranberry had the highest total solids (TS, 20.1%), volatile solids (VS, 19.4%) and soluble chemical oxygen demand (sCOD, 91,513 mg/L), while DAF had the highest average volatile fatty acid (VFA) concentration (1,688 mg/L), total nitrogen (4,172 mg N/L), and total phosphorous (875 mg P/L). Data on biogas quantity, methane (CH_4) and hydrogen sulfide (H_2S) concentrations before and after the H_2S scrubbing system, temperature (°F), and electricity production (kWh) from the digester was collected. The average daily energy production was 5,378 kWh/day from the combined heat and power (CHP) generator, with 72.2% of the produced biogas sent to CHP (1,732,188 m^3 biogas) and the rest flared over the monitoring period (2,603,362 kWh total). The digester efficiency for turning the organic matter in the manure and food waste to energy was 631 L CH_4/kg VS. A life cycle assessment (LCA) was performed to analyze the farm-scale AD system under five operating scenarios: 1) baseline scenario without anaerobic digestion or DM solids

separation and composting: 2) the current condition of the farm-scale AD system; 3) DM and FW co-digestion with no DM solids separation and composting (Scenario A); 4) DM monodigestion with DM solids separation and composting (Scenario B); 5) DM monodigestion with no DM solids separation and composting (Scenario C). Compared to the baseline scenario, the current condition had the largest reductions in environmental impacts from acidification (320%), eutrophication (447%), carcinogens (42%), and ecotoxicity (347%). The current condition (-65 T N eq/yr) removed 442% of N. Furthermore, the current condition reduced the CO_2 emissions by over 81% (4,495 T CO_2 eq/yr) compared to the baseline scenario (23,751 T CO_2 eq/yr, 19 T N eq/yr).

1. Introduction

Anaerobic digestion (AD) entails a series of microbial transformations converting organic wastes into biogas containing 50-70% methane (CH_4), 30-50% carbon dioxide (CO_2), and trace hydrogen sulfide (H_2S) (Li et al., 2019; Zhang et al., 2019). AD applications encompass a wide range of commercial, farm, and small-scale settings, offering approach to dispose of greenhouse gas (GHG)-emitting organic material. Biogas can be produced from various organic materials, including food waste (FW), dairy manure (DM), municipal solid waste (MSW), and crop waste (Achi et al., 2020; Deena et al., 2022; Zamri et al., 2021). In U.S. cattle operations, DM management accounts for roughly 43% of agricultural CH_4 emissions (Owen & Silver, 2015). DM has a higher heating value (HHV) of approximately 13,560 MJ/kg, contains N (1.92%), C (34.42%), and P (6.00±3.33 g/kg) (Shen et al., 2015). With 1.2 billion pounds of cattle manure produced annually, there is considerable potential for energy generation from AD (Marin-Batista et al., 2020; Spencer & Guan, 2004). However, DM faces challenges in AD from low biogas production. To overcome these challenges, DM can be co-digested with various agricultural and organic substrates.

Typical DM consists of a slurry containing roughly 20-30% solids (Petersen, 2018; Rico et al., 2007). DM is usually flushed with water or scraped from dairy barns, often along with bedding substrate. Sand is a common, non-digestible bedding material in DM operations, but can hinder the AD process by decreasing digester volume as it settles (Wilkie, 2005). Additionally, DM typically has low nutrient and carbon availability, contributing to excess recalcitrant lignocellulosic compounds loaded into AD systems (Tallou et al., 2020; Zhu et al., 2021). Substrates with high lignin content do not readily degrade, decreasing hydrolysis efficiency and extending the hydraulic retention time (HRT) needed for biogas production (Bi et al., 2020; Shrestha et al., 2017). Moreover, the total solids (TS) and volatile solids (VS) content impact the rates of hydrolysis and methanogenesis from the AD of DM. While VS degradation to biogas is essential, TS is pivotal in determining digester configuration and biogas production (Wang et al., 2019). The solid fraction of DM (<30%) contains up to 75% of TS and 80% VS, where using solid separation can result in a liquid fraction with a TS of 4 - 7% (Rico et al., 2007). Rico et al. (2007) reported that peak energy generation (213 mL

CH_4/g VS) came from separated DM with 3.78% TS. Similar research discovered that CH_4 production efficiency decreased by 76% (422 mL CH_4/g VS down to 189 mL CH_4/g VS) when TS was elevated from 5 to 20% (Abid et al., 2021; Yan et al., 2022).

Incorporating high-carbohydrate FW in AD faces challenges, including volatile fatty acid (VFA) accumulation and process instability, resulting in less than 2% of FW being anaerobically digested in the U.S. (Xu et al., 2018). Upwards of 40 million metric tons of FW are disposed of in the U.S., representing a significant loss of nutrient-rich substrates (Dalke et al., 2021). Food processing wastes, such cranberry processing waste and dissolved air flotation (DAF) provide a nutrient-rich substrate that can increase CH_4 production from AD (Kondusamy & Kalamdhad, 2014; Lisboa & Lansing, 2013). DAF is a high-COD substrate composed of fat, oils, and grease (FOG). Derived from food processing industries, lipid-rich DAF can be used as an AD substrate. However, the presence of long-chain fatty acid (LCFA) and excess COD in DAF can harm methanogenic activity (Logan et al., 2021). Cranberry is a low-pH, high-VS substrate with limited research exploring inclusion in AD (Lisboa et al., 2014; Lisboa & Lansing, 2013). Employing AD to FW can result in poor digester performance, NH_3 inhibition, and digester failure (Xu et al., 2018). To overcome the challenges of anaerobically digesting FW, low-carbon substrates, such as DM, can be co-digested. Co-digesting DM and FW has been shown to enhance the biodegradation of substrates, improve microbial synergy, and increase CH_4 generation between 20 - 2,940% compared to mono-digestion (Lisboa & Lansing, 2013; Xu et al., 2018).

Life cycle assessment (LCA) quantifies impact of cradle to grave processing and offers insight into environmental consequences and mitigations by comparing two or more systems, products, or services. LCA is an effective tool to determine the environmental impact of employing AD (Czyrnek-Delêtre et al., 2017), including biogas combustion in CHP from a wide range of substrates from manure and crops to FW (Fei et al., 2022; Hassanein, Moss, et al., 2022; Righi et al., 2016). Previous studies exploring the application of CHP to farm-scale co-digestion have resulted in GHG emissions reductions compared to manure mono-digestion without CHP (Aui et al., 2019; Li et al., 2020). However, there is limited research applying LCA to farm-scale co-digestion with DM and FW, as well as incorporating on-site composting of manure solids, and no studies including energy generation from a CHP generator from DM/FW co-digestion.

This study provides new data by exploring a farm-scale co-digestion of liquid DM, DAF, and cranberry waste with on-site composting of the DM solids, which were sold as a value-added product. An LCA was applied to the system to analyze five scenarios that include or eliminate various aspects of the AD system. The baseline scenario investigated the system with no AD or solids composting, the current condition analyzed the system as it currently operates, Scenario A explored the co-digestion system with no solids composting, Scenario B examined monodigestion with solids composting, Scenario C reviewed monodigestion with no solids composting. With current knowledge, this research is the first to apply LCA to a farm-scale anaerobic co-digestion of liquid DM/FW, explore on-site DM solids composting, and analyze CHP energy generation in one study. The goals of this study were to: 1) evaluate the energy, nutrient, and organic transformations for a novel AD farm design that digested DM and FW; and 2) determine the effects of improving the performance and operating conditions on the system's environmental impacts using LCA, including modification of substrates, solids composting, and biogas-fueled CHP efficiency.

2. Materials and Methods

2.1 Farm-Scale Anaerobic Digestion

2.1.1 System Description

The farm-scale AD was located in the Mid-Atlantic of the USA. The system consisted of a sand separation lane, solids-liquid separation unit, covered lagoon digester, H_2S scrubber, CHP generator (240 kWh, Siemens, Germany), FW storage tanks, and an open effluent storage lagoon (Figure 3.1). The system operation was as follows: 1) the DM stream was washed from the barn onto an angled concrete slab for sand-separation; 2) the DM was processed to separate the solids and the liquids, where the solids were transferred to an on-site composting facility; 3) liquid DM traveled into the covered lagoon digester and mixed with FW; 4) liquid effluent was stored in an open lagoon before being applied to fields as a fertilizer; and 5) biogas was processed through the H_2S scrubbing system before being burned in the CHP to produce electricity or burned at a flare. Electricity was used to heat the digester, maintaining mesophilic temperatures (35 °C). FW was DAF waste from poultry processing, milk waste, and cranberry processing waste delivered to the farm.

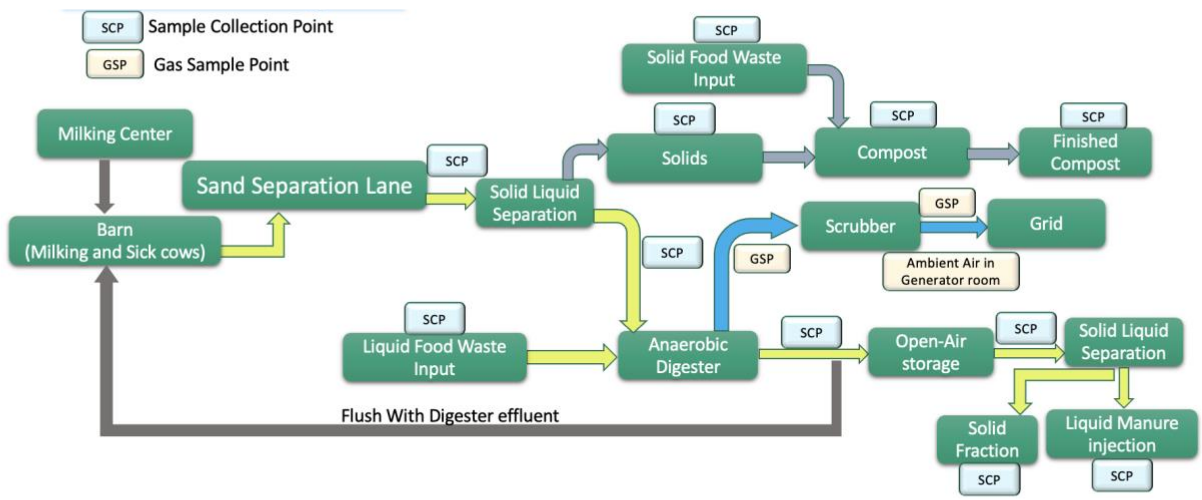

Figure 3.1: The solid, liquid, and gas sampling points for the farm-scale AD system and composting facilities.

2.1.2 Digester Operation

The covered lagoon digester has a HDPE cover that 96 m long by 24 m wide and has an area of 2,476 m^2. Heat from the CHP is recirculated to the AD system to heat the digester. The DAF and cranberry wastes are stored in 375,000 liter concrete pits, with agitation completed before injection into the system. DAF (16,071 liters/day 18.52% of volume), milk waste (3,214 liters/day, 3.71% of volume), and cranberry (3,214 liters/day, 3.71% of volume) are mixed with the liquid DM (64,285 liters/day, 74.07% of volume) and continuously fed into the digester via a 240-V pump (Gormann-Rupp, 13A20-B, Mansfield, Ohio) operating 24 hours per day. Effluent is stored in an open lagoon before field application, occurring in Spring and Fall of each year. Effluent is recirculated to flush the manure and bedding from the stables. The compost was turned twice per week, then allowed to set before being sold. The CHP was a 240 kW (CHP240 kW, Siemens, Germany) generator.

2.1.3 Sampling and Monitoring

The monthly sampling was performed over 13 months. Liquid from 6 sampling points and 3 solid sampling points were collected monthly from the farm-scale AD system. Liquid sample points included DM before solids separation (SS), DM after SS, DM digester influent, digester effluent, DAF, and cranberry. Solids sampling points included DM solids, initial DM solids for composting, and final compost. DAF and cranberry samples were initially agitated in the concrete containers for five minutes before collection. Each sample was collected while mixing was performed. The FW samples were mixed again using a drill before collecting a subsample to ensure a homogenized sample. All liquid samples were stored in 1 L bottles and brought to the lab for analysis. Solids were collected using a shovel, with samples taken at a depth of 12.7 cm. Solid samples were stored in 3.75-liter plastic bags until analysis. Samples were stored at ~0 ºC until analyses were completed.

Biogas and energy data from an on-site data-logging system was obtained for 13 months. This data included biogas quantity, CH_4 content, H_2S content before and after the scrubber, temperature, and electricity production every 15 minutes. All data points were combined into a Microsoft Excel spreadsheet, with the data point analyzed for daily and monthly production values for biogas, CH_4 efficiency, and electricity production. The biogas production: total biogas (% biogas in the CHP engine and the flare) was multiplied by the liters of FW, and DM added per day (87,598 L) and divided by the combined influent VS average (5%). This number was calculated daily and monthly. The digester efficiency (L CH_4 /kg VS) was calculated by multiplying the biogas production by the average percent CH_4 in the biogas and dividing by the weight of VS in the combined FW and DM substrate. Energy generation (kWh) calculations from the CHP generator were performed by averaging the energy production over 24 hours to derive the daily electricity production (kWh/day).

2.1.4 Biogas and Chemical Analysis

Liquid samples were analyzed for TS, VS, soluble chemical oxygen demand (sCOD), volatile fatty acid (VFA), ammonia (NH_3), total Kjehldahl nitrogen (TKN), and total phosphorous (TP). Each sample analysis was performed in triplicate. Standard Methods were used to analyze the TS (method 2540B) and VS (method 2540E) of all samples. The sCOD and COD of liquid samples were analyzed using a HACH digestion method, where sCOD was filtered through a 0.45 um membrane before analysis. Liquid NH_3 samples were acidified using 15 N sulfuric acid to reach pH 2 before filtration through a 0.45 um membrane. NH_3 analysis was performed with a Lachat instrument (QuikChem Method 10-107-06-2-O). TKN and TP for liquid samples were completed by acidification to pH 2 using 15 N sulfuric acid and $CuSO_4$-$5H_2O$ before analysis with the Lachat QuikChem Method 13-107-06-2-D for TKN and QuikChem Method 13-115-01-1-B for TP). VFAs of liquid samples were analyzed by acidification to pH 2, followed by centrifuging for 20 minutes (5000 RPM), and filtration of the supernatant with 0.22 um membranes. VFAs, including acetic acid, propionic acid, butyric acid, and valeric acid, were analyzed via a 7890A Agilent gas chromatograph (GC). From the collected online monitoring data, biogas quantities consumed in the CHP and flare were obtained, and CH_4 content was calculated based on the online data, with data collected on H_2S concentration before and after the scrubbing system.

2.2 Life Cycle Assessment

2.2.1 Goal and Scope, Inventory, and Impact of Life Cycle Assessment

This study performed an LCA using SimaPro software developed by PRé Consultants (Version 9.0, Amersfoort, Utrecht, Netherlands) to analyze the environmental impacts and mitigation of electricity production and nutrient flows through the farm-scale AD and composting system. The scope of the LCA was a cradle-to-grave analysis of the system, including DM storage, FW storage, FW transportation, solids composting, and electricity generation from biogas combustion. Twelve months of data sampling, material inputs, and processing conditions were incorporated into the LCA to investigate the inputs and outputs of the farm-scale anaerobic co-digestion and composting processes (**Error! Reference source not found.**). The impact of the construction of the AD, compost facility, food waste containers/transport, lagoon manure storage, and electricity input/generation were determined using Ecoinvent 3.0 database. The inputs and outputs were translated into 10 impact categories and analyzed to evaluate the environmental impacts, GHG emissions, and human health impacts of farm-scale AD and composting. The LCA tracked energy and carbon flows, reductions and increases in the cradle-to-grave environmental impacts through an on-farm mass balance of materials in the system (Figure 3.2).

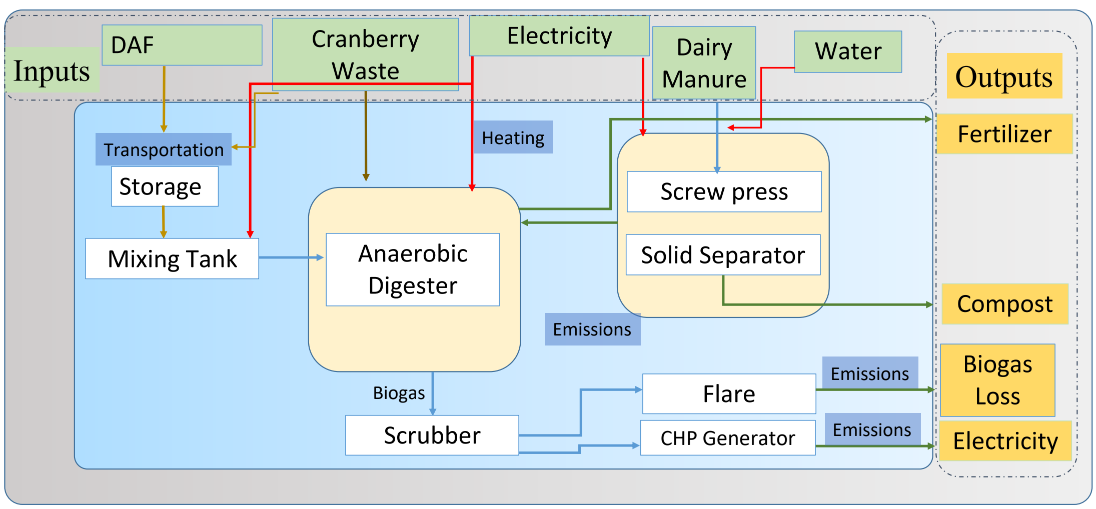

Figure 3.2: Boundaries for LCA of farm-scale AD of cranberry, dissolved air flotation (DAF) and dairy manure (DM).

Five scenarios were analyzed from the LCA of farm-scale anaerobic digestion (**Error! Reference source not found.**3.1). The baseline scenario analyzed no AD or solids composting, the current condition and Scenario A explored co-digestion with or without solids composting, Scenario B investigated monodigestion with solids composting, while Scenario C examined monodigestion and no solids composting. The LCA explored the impacts on the environment, GHG emissions, and human health impacts from each of the five scenarios. The scenarios allowed the implementation of waste ratios that maximize energy production and minimize environmental impacts.

Table 3.1: Table of LCA scenarios based on the functional unit of 1 year and 1 ton of organic waste.

Scenario	**Analysis Categories**	**Description of Scenario**
No AD, no DM solids composting (Baseline)	• Eutrophication • Global Warming (GHG) • Acidification • Ozone Depletion • Fossil Fuel depletion • Carcinogens • Non-carcinogens • Respiratory Effects • Ecotoxicity • Smog	No Anaerobic digestion system, manure storage in an open lagoon, and no DM solids separation or composting

AD of FW, DM, DM solids composting (Current Condition)	• Eutrophication • Global Warming (GHG) • Acidification • Ozone Depletion • Fossil Fuel depletion • Carcinogens • Non-carcinogens • Respiratory Effects • Ecotoxicity • Smog	Anaerobic digestion of liquid DM and FW, with DM solids composting, CHP energy production from biogas
AD of FW, DM, no DM solids composting (Scenario A)	• Eutrophication • Global Warming (GHG) • Acidification • Ozone Depletion • Fossil Fuel depletion • Carcinogens • Non-carcinogens • Respiratory Effects • Ecotoxicity • Smog	Anaerobic digestion of FW and DM, no DM solids separation and composting
AD of DM, DM solids composting (Scenario B)	• Eutrophication • Global Warming (GHG) • Acidification • Ozone Depletion • Fossil Fuel depletion • Carcinogens • Non-carcinogens • Respiratory Effects • Ecotoxicity • Smog	Anaerobic digestion of DM with no FW co-digestion, no DM solids separation and composting
AD of DM, no DM solids composting	• Eutrophication • Global Warming (GHG)	Anaerobic digestion of DM with no FW co-digestion, no DM

(Scenario C)	• Acidification • Ozone Depletion • Fossil Fuel depletion • Carcinogens • Non-carcinogens • Respiratory Effects • Ecotoxicity • Smog	solids separation and composting

Two LCA analysis were performed for all scenarios using two functional units. The first LCA functional unit was 1 year (365 days), with the project inputs, outputs, and environmental impacts based on material input and outputs over 1 year of operation. The second LCA functional unit was 1 ton of organic waste, with inputs, outputs, and environmental impacts based on material input. The baseline scenario investigated the system with no AD or composting where all manure was stored in an open lagoon, with analyses on the impacts of manure storage and infrastructure without mitigating technologies, such as biogas capture or compost products. The baseline scenario considered the GHG emissions that would potentially be emitted from anaerobic conditions in an uncovered lagoon storing all the DM manure produced on-farm. The current condition explored the system as it operates with AD of liquid DM/FW and solids composting. The results provided data regarding the current system's efficiency, including energy generation from anaerobic co-digestion and solids composting.

Scenario A examined the AD of DM/FW that excluded solids composting, with all manure produced on-farm digested. Scenario B investigated the AD of DM only with solids composting, excluding the transportation, infrastructure, and incorporation of FW into the system. Biogas reductions and energy generation decrease (82.5%) associated with mono-digestion of DM were included based on data from our previous biochemical methane potential (BMP) of the wastes obtained from the farm-scale AD system (data not presented here). Scenario C investigated the farm-scale AD of DM only, with no solids composting. All scenarios with AD included the electricity consumption for the AD system, such as feeding pumps, and mixing pumps. Based on data obtained from the farm-scale AD system, all scenarios incorporating compost generated a value-added product of 3,822 cubic meters per year, assuming all separated DM solids (~27%) were composted.

3. Results and Discussion

3.1 Energy and Biogas Production

The yearly biogas production from the AD was 1,831,450 m^3/yr. The majority of biogas was consumed in the CHP (72.2%, 1,322,306 m^3/yr) while 27.8% (509,134 m^3/yr) of the biogas was flared, as the CHP operated at capacity (224 kW/hour). Based on the yearly biogas production, roughly $4.6*10^{10}$ BTU was generated annually. The biogas used in the CHP (1,322,306 m^3/yr) for energy generation was approximately

$3.3*10^{10}$ BTU, while flared biogas (509,134 m^3/yr) equated to $1.2*10^{10}$ BTU not utilized.

The percent CH_4 data in the biogas from the on-site biogas analyzer from month 1 through month 16 is shown in Figure 3.3B. During normal operation, the average CH_4 concentration was 56.2%. From Month 1 to 6, the average percent CH_4 in the biogas decreased to 54.1%, which increased to 60.1% from Month 6 to 13, and 63.2% from Month 13 to 16, with a decrease during the final month to 56.1% CH_4.

The digestion efficiency of the farm-scale AD system over the monitoring period was 631 L CH_4/kg VS, with 74% DM and 26% FW volumetric inputs to the system (Figure 3.3A). Previous literature reported a CH_4 production of 388 L/kg VS co-digesting DM and FW (1:2) and 603 L CH_4/kg VS at 70% DM and 30% FW (Prabhu et al., 2021; Zhang et al., 2013). Contrarily, Zhang, et al. (2013) determined 65% CH_4 content (70% DM, 30% FW), compared to an average concentration of 56.2% CH_4 in this study.

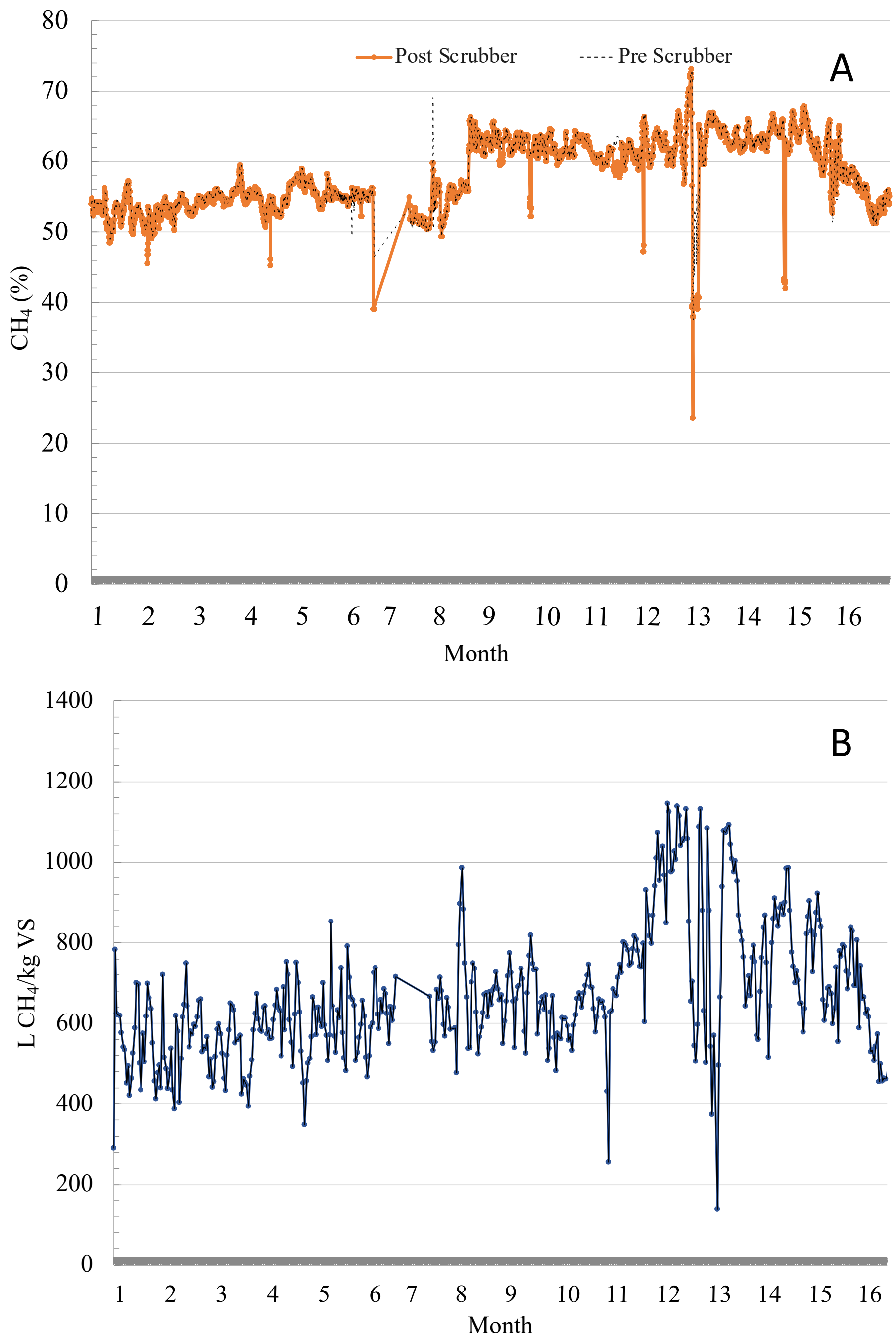

Figure 3.3: CH_4 production (L/kg VS) (A), and CH_4 content (%) (B) of farm-scale AD of FW and DM.

The average kWh production of the CHP was 5,378 kWh/day, with an average energy production of 224 kW per hour, which was the capacity of the generator (224 kW). The CHP output was 1,983,722 kWh/yr. A similar farm-scale AD operation found that AD of DM alone decreased the net energy of CHP production, where co-digestion of FW and DM increased the energy balance of the CHP system from the high degradability of FW (Usack et al., 2019). Usack et al. (2019) reported that DAF sludge contributed the greatest to CH_4 production (0.99 m^3 biogas /kg of DAF) and the lowest parasitic energy loss (14%), while DM alone led to 54% parasitic energy loss (Usack et al., 2019). Similarly, in the current study, the high VS from cranberry (19.4%) and DAF (10.9%) increased the substrate compared to DM influent (2.53%), enabling increased CH_4 production and subsequent energy generation from the CHP. Another study exploring a continuously stirred tank reactor (CSTR) found that the digestion of DM alone did not generate sufficient energy in CHP to maintain mesophilic conditions from September to June, generating only 3,298 kWh compared to 61,995 kWh from co-digesting DM and corn stover (Li et al., 2020).

3.1.1 Total Solids and Volatile Solids

The average TS% over 13 months for DAF and cranberry was 11.6 ±2.2% and 10.1 ±3.1%, respectively (Figure 3.4A). The average TS% for DM before SS, DM after SS, DM influent, and effluent was 2.75 ±0.2%, 2.59 ±0.2%, 2.53 ±0.2%, and 2.63 ±0.2%, respectively. While DAF TS% has been reported in previous literature (2.3-

14.7%) (Johannesson et al., 2020; Logan et al., 2021), limited studies analyze cranberry waste as a substrate for farm-scale AD. DM influent represents the liquid manure entering the digester after solids removal. DM solids were removed through a solids-separation (SS) unit. Expectedly, a decrease in TS% was observed between DM before SS and DM after SS due to the removal of the DM solids (~27%). Similar research found that DM treated with SS had 6.9% TS before SS and 5.4% TS after SS (Aguirre-Villegas et al., 2019). These studies suggest SS application to DM is beneficial, as lowering the TS of DM can improve AD performance (Wang et al., 2019).

The average VS% for DAF and cranberry over 13 months were 10.9 ±2.2% and 19.4 ±3.1%, respectively (Figure 3.4B). The average VS% over 13 months for DM before SS, DM after SS, DM influent, and effluent was 1.9 ±0.1%, 1.8 ±0.1%, 1.7 ±0.1%, and 1.8 ±0.1%, respectively. The combined influent (DM influent, DAF, cranberry) had a VS of 5.0 ±1.8% over the 13-month sampling period. The VS from FW is pivotal in biogas production, represented by higher influent VS availability and decreased effluent VS (Johannesson et al., 2020; Logan et al., 2021). DAF has a reported VS range of 2.3 - 14.7% (Johannesson et al., 2020; Logan et al., 2021). Originating from poultry processing waste, the DAF in this research contained a moderate VS (10.9%) compared to the previous study by Johannesson et al. (2020) (2.3-14.7%). VS conversion is critical for CH_4 generation, combining low-VS DM with high-VS cranberry and DAF positively impacted biogas generation. Previous supporting data reported that the addition of FW to DM for AD increases CH_4 yield

(55%) and increases VS conversion (39.5%) (Abbas et al., 2023; Kim et al., 2022; Orfanoudaki et al., 2020). While the previous studies found increased biogas and CH_4 yield from the co-digestion of FW and DM, our study included cranberry waste, a high-VS% substrate not typically used in farm-scale AD. A pervious study of waste from this sample covered lagoon AD system revealed that co-digesting DM with cranberry increased biogas production by over 1,200% compared to DM alone (Lisboa & Lansing, 2013). As the TS% and VS% of the DM alone are relatively low (1.9%), incorporating FW helped boost the influent VS to 5.0%, adding more substrate for microbial degradation.

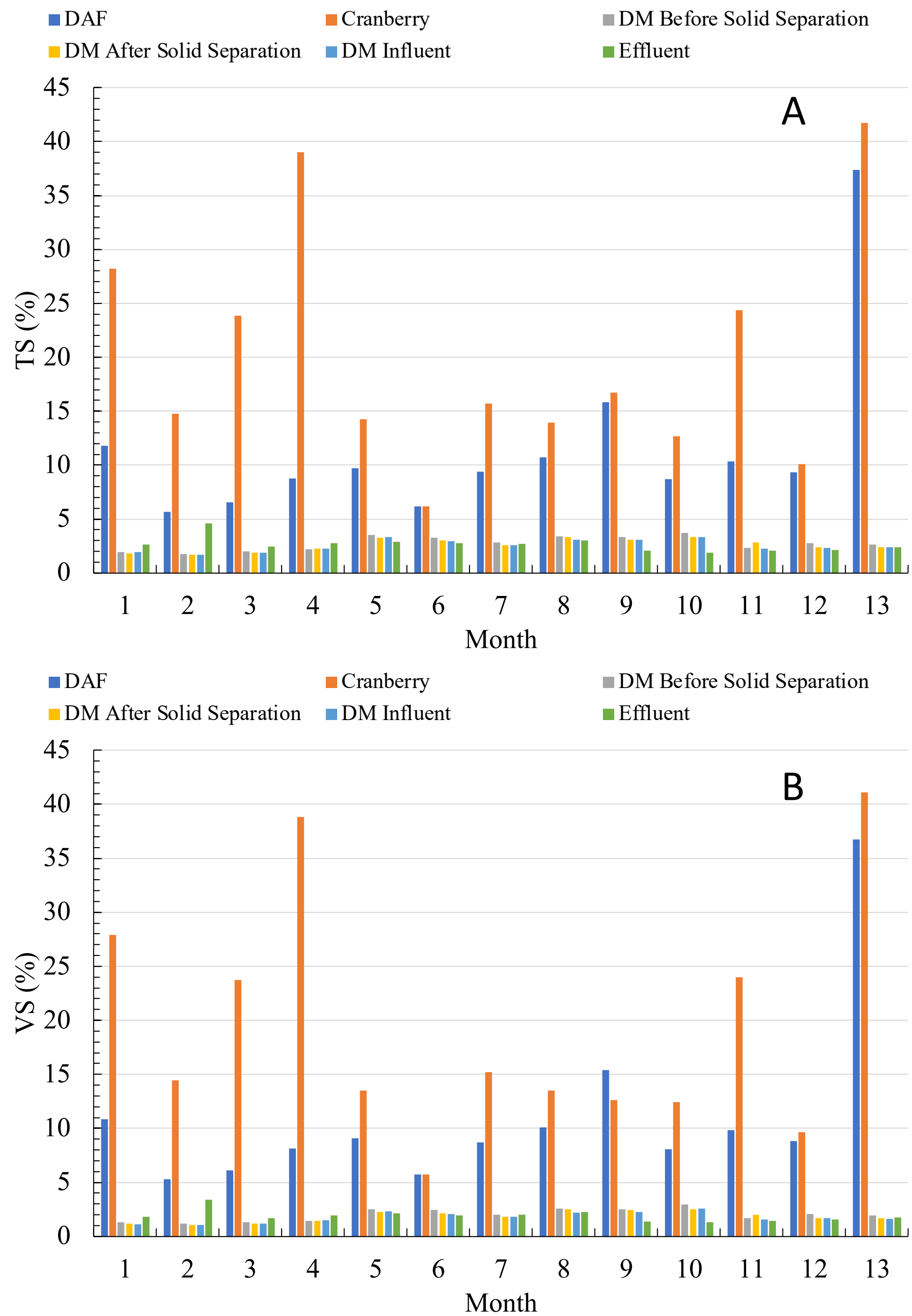

Figure 3.4: Total solids (TS%) over 13 months for liquid samples (A) and volatile solids (VS%) for liquid samples (B) over the sampling period.

3.1.2 Soluble Chemical Oxygen Demand

Soluble chemical oxygen demand (sCOD) represents the amount of organic matter in the specific substrates for biogas production as microbial activity will reduce sCOD due to the methanogenesis converting the organic material to biogas. The average sCOD over 13 months for DAF and Cranberry was 26.7 ±4.5 g/L and 91.5 ±13.4 g/L, respectively (Figure 3.5). The average sCOD for DM before SS, DM after SS, DM influent, and effluent over 13 months was 9.6 ±1.7 g/L, 6.5 ±1.1 g/L, 6.2 ±1.2 g/L, and 1.9 ±0.5 g/L respectively. The lowest substrate sCOD was DM influent.

Correlating with low biogas generation, low sCOD also signifies the efficiency of hydrolytic rates (Dasgupta & Chandel, 2020). Co-digestion increased CH_4 yield by 25% when high-sCOD DAF (38 g/L) was combined with DM (21 g/L) (Johannesson et al., 2020). Additionally this study found that cranberry had the highest average sCOD, indicating a high potential for microbial degradability and biogas generation when co-digested. A past study analyzing various substrates' biomethane potential (BMP) reported a COD of 436 g/L for cranberry (Lisboa & Lansing, 2013). The sCOD reduction between the influent and effluent represents a stable anaerobic digestion process.

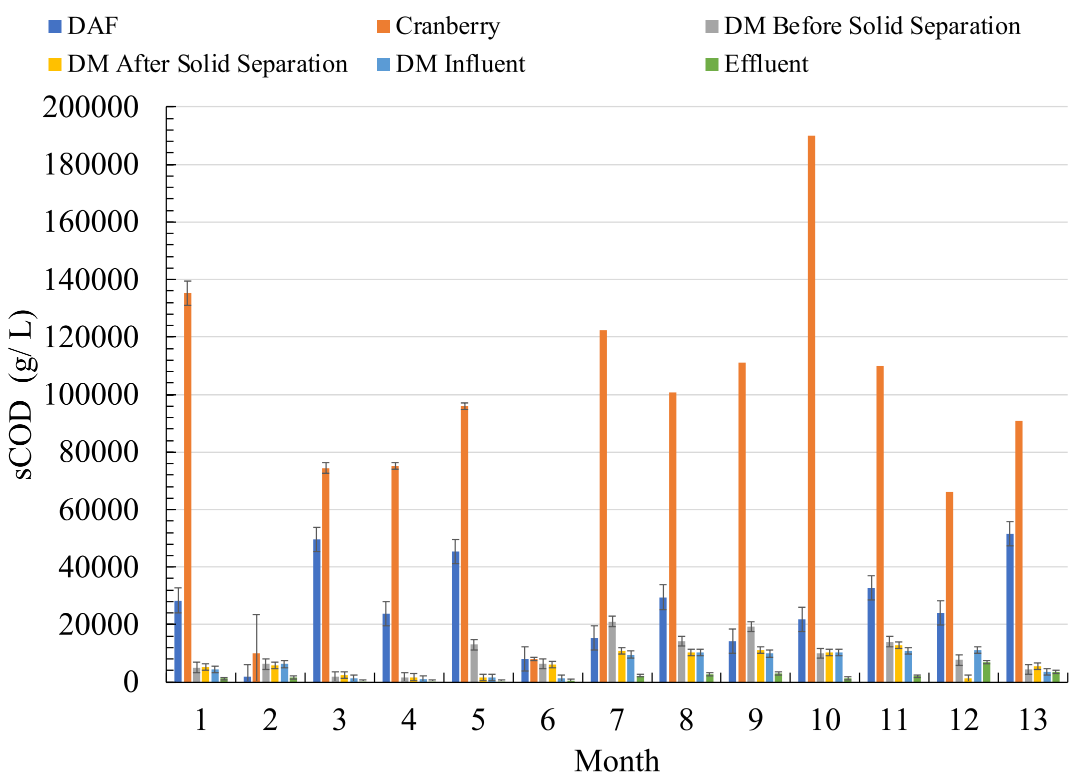

Figure 3.5: Soluble chemical oxygen demand (sCOD) for liquid samples over the sampling period.

3.1.3. Ammonia, Total Nitrogen, and Total Phosphorous

Tracking the ammonium (NH_3), total nitrogen (TKN), and total phosphorous (TP) allows the nutrient content traveling through the system to be analyzed. The average TKN concentration for DAF and cranberry over 13 months was 4,170 ±280 mg-N/L and 2,840 ±330 mg-N/L, respectively (Figure 3.6B). The average TKN for DM before SS, DM after SS, DM influent, and effluent over 13 months was 1,890 ±170 mg-N/L, 1,950 ±180 mg-N/L, 1,860 ±200 mg-N/L, and 4,030 ±260 mg-N/L, respectively.

The average TP concentration for DAF and cranberry over 13 months was 870 ±130 mg-P/L and 500 ±70 mg-P/L, respectively (Figure 3.6C). The average TP for DM before SS, DM after SS, DM influent, and effluent over 13 months was 300 ±40 mg-P/L, 300 ±40 mg-P/L, 450 ±90 mg-P/L, and 770 ±100 mg-P/L, respectively.

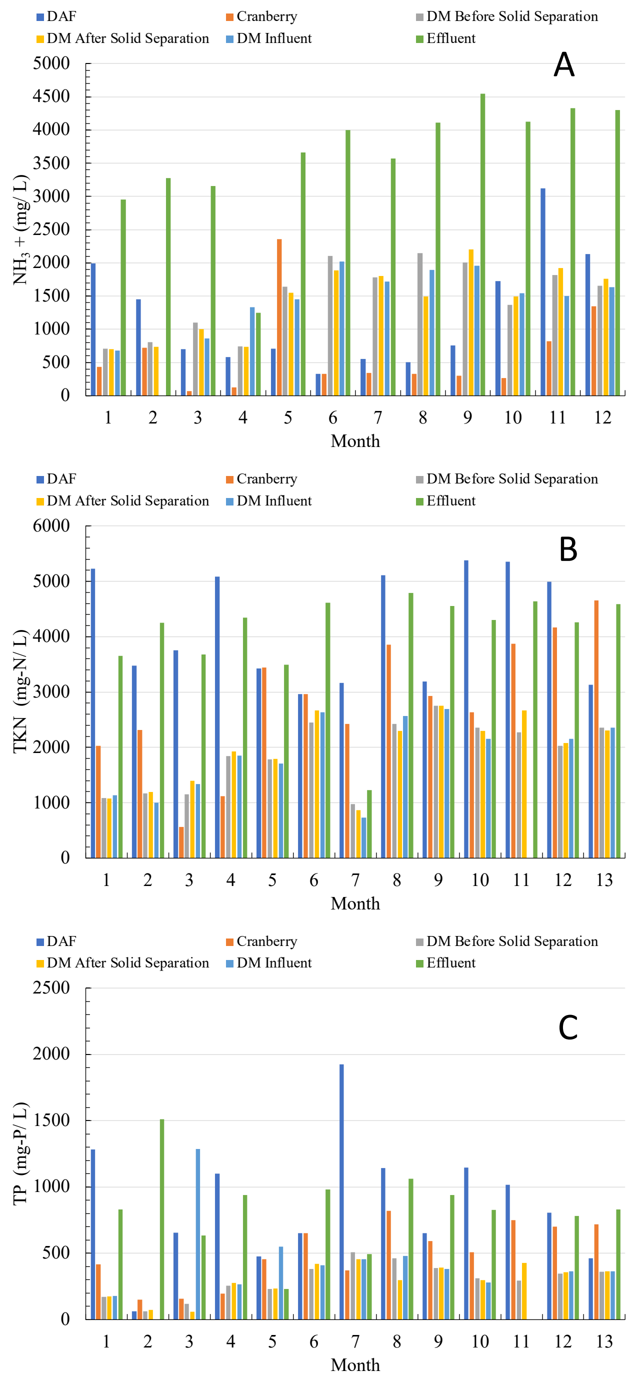

Figure 3.6: Average ammonia (NH_3) (A), total nitrogen (TKN) (B), and total phosphorous (TP) (C) concentrations for liquid samples over the sampling period.

3.1.4. Temperature and pH

The digester was maintained near 35 °C (mesophilic conditions) throughout the monitoring period (Figure 3.7). The average temperature throughout the monitoring period was 36.3 °C, with a high of 38.7 °C. This temperature provided adequate conditions for digestion and gas production while limiting the energy input for heating.

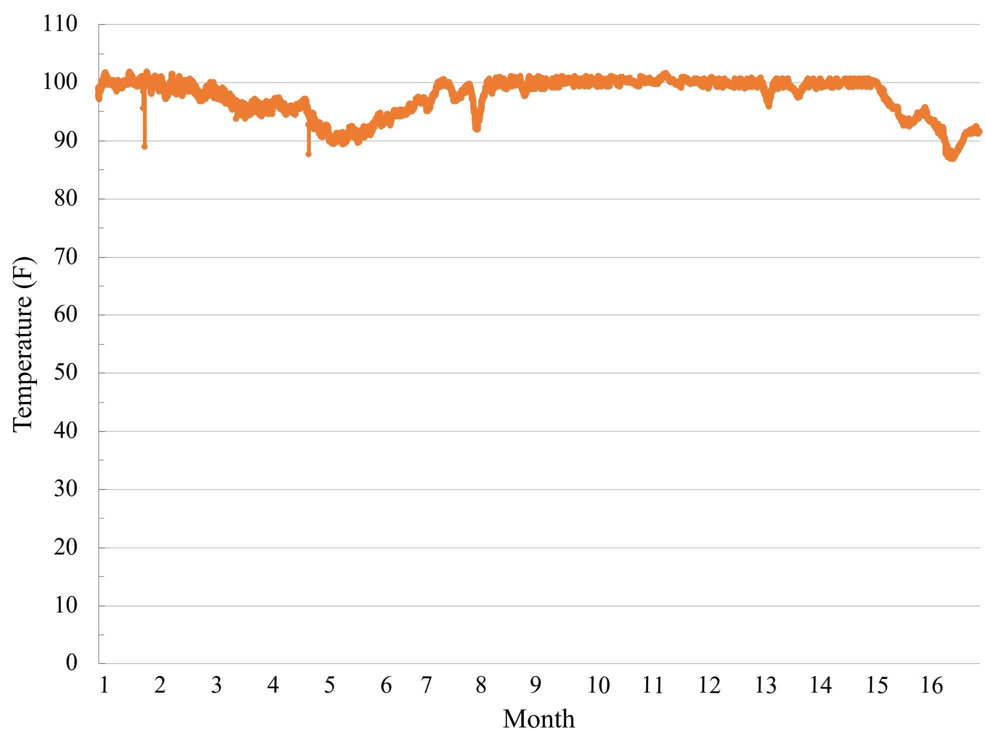

Figure 3.7: Temperature over the monitoring period for farm-scale AD system (°F).

The highest pH observed over the sampling period was observed for DM influent (8.2), while the lowest pH was observed in cranberry (3.01) (Figure 3.8).

Throughout the sampling period, cranberry had the lowest average pH (3.6 ±0.1), the AD effluent had the highest average pH (7.90 ±0.1), while the average feedstock influent (DAF, cranberry, DM influent) had a pH of 5.6 ±0.1. DAF had an average pH of 5.2 ±0.1), DM influent had an average pH of 7.8 ±0.03 over the 13-month sampling period.

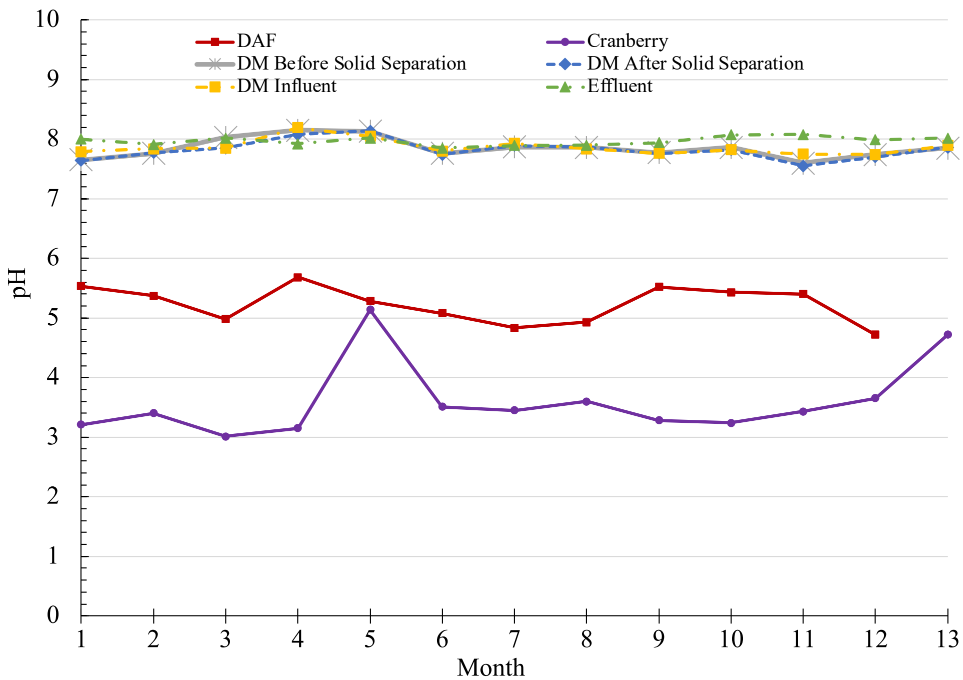

Figure 3.8: pH for liquid samples over the sampling period.

3.1.5. Volatile Fatty Acids

Volatile fatty acids (VFAs) are produced during the acidogenic and acetogenic stages, providing a substrate for methanogens to produce biogas. The four primary VFAs analyzed were acetic acid, propionic acid, butyric acid, and valeric acid (Figure

3.9A, B, C, D). Influent DM had the lowest average VFA concentration (693 ±115 mg/L). DAF had the highest average VFA concentration (1,688 ±191 mg/L), followed closely by cranberry (1,526 ±273 mg/L). The AD effluent had an average total VFA concentration of 301 ±49 mg/L. Similar research reported VFA concentrations of DM between 3,432 mg/L to 23,000 mg/L (Page et al., 2014; Zhang et al., 2011). Adding FW increases the amount of available substrate, in the form of VFAs, available for biogas production.

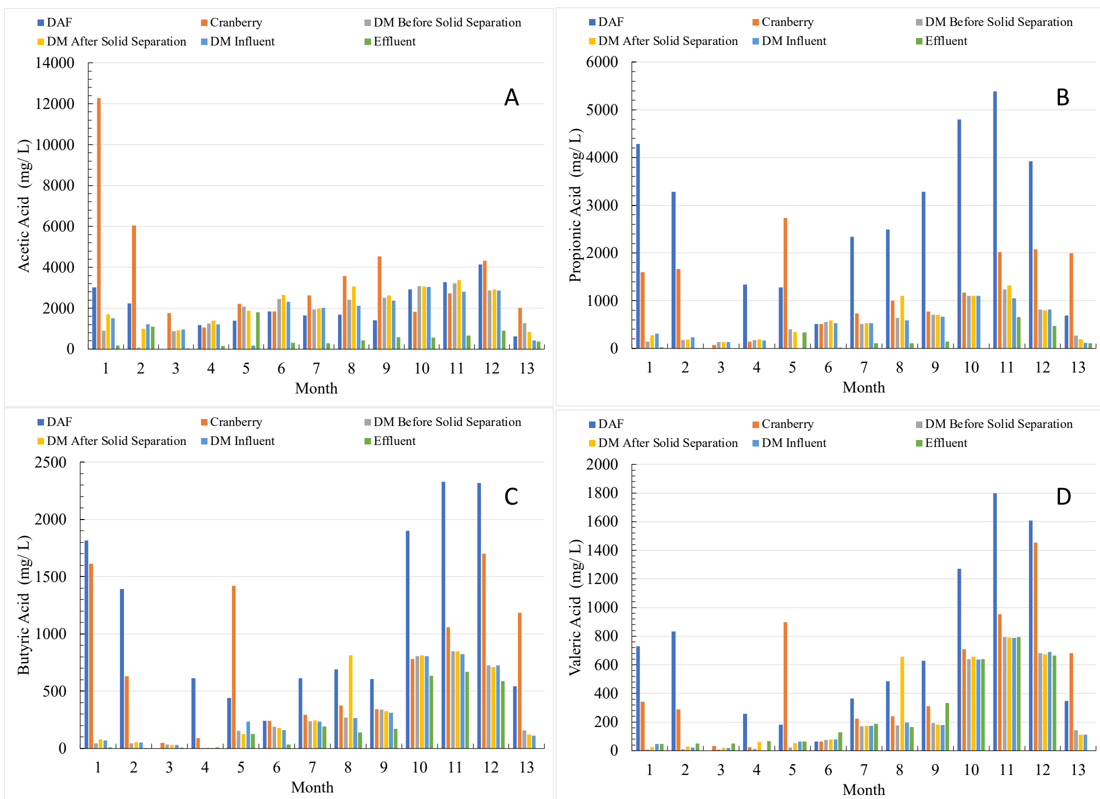

Figure 3.9: VFA concentrations (mg/L) of acetic acid (A), propionic acid (B), butyric acid (C), and valeric acid (D) over the sampling period.

3.2. Life Cycle Assessment

3.2.1. Energy Generation from Farm-Scale AD

Annually, the farm-scale AD system processed 17,362 m^3/yr of liquid DM (~73% of DM waste stream) and received 8,290 m^3/yr of FW. The solid separation unit separated 4,703 tons/yr (~27% of DM waste stream) of solid DM. The average BTU for biogas with 60-80% CH_4 is 600-800 BTU (EPA, 2004). Based on 600 BTU, the yearly biogas production (1,831,450 m^3/yr) equated to roughly $3.8*10^{10}$ BTU generated over the year. The biogas used in the CHP (1,322,306 $m^{3/}$yr) for energy generation was equal to $2.8*10^{10}$ BTU, while flared biogas (509,143 m^3/yr) equated to $1.07*10^{10}$ BTU lost. As 72.2% of the produced biogas was combusted for energy production, the net energy balance was positive, with the heat from the CHP used for heating.

3.2.2. Environmental Impacts from Farm-Scale AD

The baseline scenario (no AD, no DM solids compost) had the following impacts: global warming (GHG emissions, 23,751 T CO_2 eq/year), eutrophication (19 T N eq/yr) acidification (75 T SO_2 eq/yr), ecotoxicity (62 CTUe x1,000,000/yr), ozone depletion (0.54 kg CFC-11 eq/yr), smog (461 T O_3 eq/yr), carcinogens (1.5 CTUh/yr), respiratory effects (9 kg PM2.5 eq/yr), and non-carcinogens (2.5 CTUh/yr) (

Table **3.2**). The CO_2 emission of the baseline scenario totaled 23,751 T CO_2 eq/year, where constructing the DM storage lagoon (6,024 T CO_2 eq/year) and environmental emissions of DM from the lagoon (17,727 T CO_2 eq/year) were the largest contributors to total CO_2 emissions. The current condition displayed less emissions in global warming (4,495 T CO_2 eq/year), smog (380 T O_3 eq/yr), acidification (-166 T SO_2 eq/yr), eutrophication (-65 T N eq/yr), ecotoxicity (-154 CTUe x1,000,000), respiratory effects (0.74 kgPM 2.5 eq/yr), non-carcinogenics (0.38 CTUh/yr), and carcinogenics (0.88 CTUh/yr).

When comparing the current condition to the baseline scenario, there were large reductions in almost all assessment categories: acidification (320%), eutrophication (447%), ecotoxicity (348%), respiratory effects (92%), global warming (81%), non-carcinogenics (85%), carcinogenics (43%), and smog (18%) (Figure 3.10). Global warming reductions were nearly equivalent for all other scenarios at >81% reductions compared to the baseline scenario. However, the current condition increased fossil fuel depletion (3%) and ozone depletion (6%) more than the baseline scenario due to the transport of FW.

Table 3.2: Impact quantities for all scenarios.

Impact Categories	**Unit**	**Baseline Scenario**	**Current Condition**	**Scenario**		
				A	**B**	**C**
Global Warming	T $CO_{2\ eq}$/yr	23,751	4,495	4,506	4,284	4,296
Eutrophication	T N_{eq}/yr	19	-65	-64	-58	-57
Smog	T $O_{3\ eq}$/yr	480	380	364	324	307
Acidification	T SO_2 eq/yr	75	-166	-166	-151	-151
Carcinogens	CTUh/yr	1.55	0.89	0.9	0.91	0.92
Non-carcinogens	CTUh/yr	2.5	0.38	0.6	-0.71	-0.49

Respiratory Effects	kg $PM_{2.5}$ $_{eq}$/yr	9	0.74	0.71	0.82	0.79
Ecotoxicity	CTUe X 1,000,000	62	-154	-149	-137	-132
Fossil Fuel Depletion	MJ Surplus/yr	5,134,437	5,268,569	5,325,492	4,738,453	4,795,376
Ozone Depletion	kg CFC-11 eq/yr	0.54	0.57	0.56	0.51	0.5

Compared to the current condition, Scenario C (DM monodigestion, no DM solids separation and composting) had the largest reductions in ozone depletion (8%) and smog (33%), while Scenario B (DM monodigestion, DM solids composting) had the largest reduction in fossil fuel depletion (8%). These reductions were attributed to excluding FW transport and composting, as FW transport, the construction of the FW containers, diesel used for compost agitation, and concrete used for composting pads increased fossil fuel use, ozone depletion, and smog. To more efficiently investigate GHG emissions and nutrient flow, separate analyses were conducted for global warming and eutrophication process contributions to quantify emissions from each scenario.

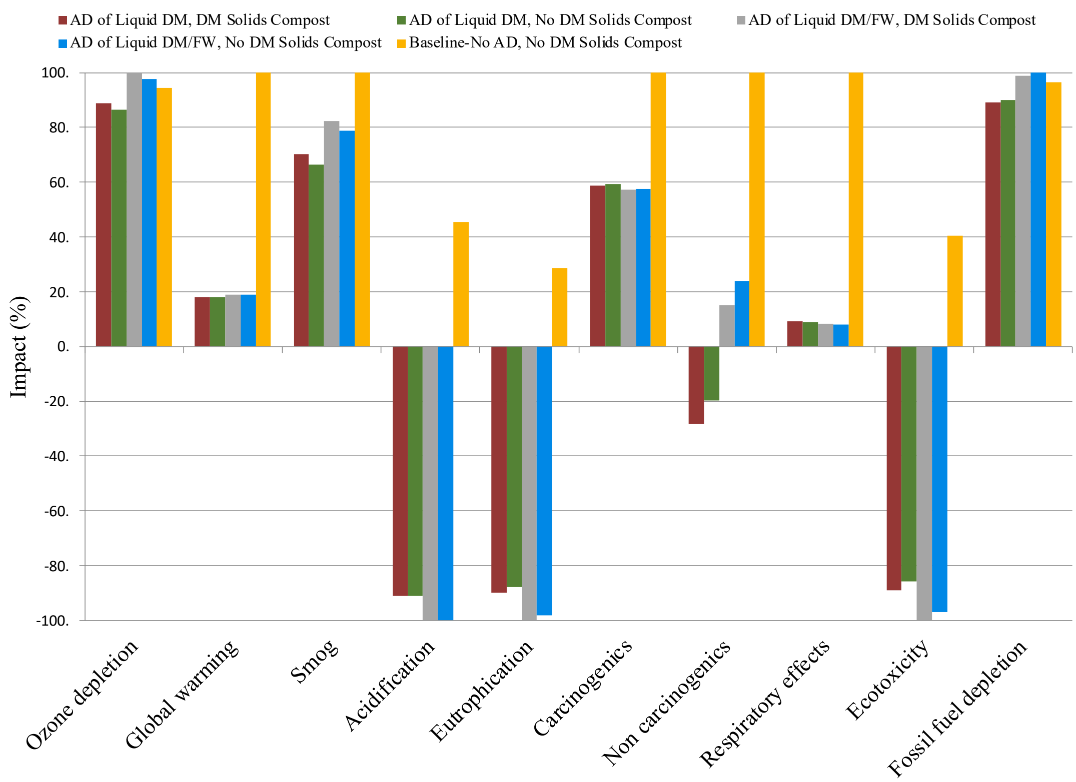

Figure 3.10: Environmental impacts for life cycle assessment of all scenarios for farm-scale AD with a functional unit of 1 year.

3.2.2.2. Eutrophication, Global Warming, and Carbon Dioxide Emissions

Global warming is one of the most important categories due to the GHG potential and climate impact of CO_2 emissions. LCA data indicated that incorporating AD, in any configuration, drastically reduced global warming (>81%) compared to the baseline scenario (Figure 3.11). Compared to the baseline scenario, large reductions

(>81%) in CO_2 emissions were observed in the current condition (4,495 T CO_2 eq/year), Scenario A (4,506 T CO_2 eq/year), Scenario B (4,284 T CO_2 eq/year), and Scenario C (4,296 T CO_2 eq/year). When the digester was added, the avoided CO_2 emissions from electricity production in AD of FW/DM (2,242 T CO_2 eq/year) substantially offset the emitted C_{O2} from the AD cover (6.7 T CO_2 eq/year), pump electricity consumption (5.8 T CO_2 eq/year), concrete (442 T CO_2 eq/year), and pump construction (175 T CO_2 eq/year). This resulted in a net 1,612 T CO_2 eq/year in avoided emissions.

Food waste heavily impacted global warming in the Scenarios A-C and the current condition. The transportation in ton-kilometers (tkm) and concrete for the FW storage pits contributed to the overall emissions by greater than 15% (682 T CO_2 eq/year). This impact was completely offset by the increased electricity production from DM/FW co-digestion (2,242 T CO_2 eq/year), yielding a net CO_2 decrease of 1,560 T CO_2 eq/year (30%). The AD of DM without FW in Scenarios B and C led to a decrease in biogas and electricity production (1,769 T CO_2 eq/year). This was displayed in the LCA as less reduction in CO_2 emissions and global warming impacts, as the energy generation was less effective at offsetting the impacts of the AD lagoon (6,024 T CO_2 eq/year). Nonetheless, the similarity in global warming of all scenarios compared to the baseline Scenario does not justify the exclusion of FW due to the substantial offset from increased electricity production.

In the current condition, the reduction in CO_2 emissions of compost production (611 T CO_2 eq/year) was nearly eliminated by the global warming impact of the material and energy inputs from compost operation and facility construction (concrete 571 T CO_2 eq/year, diesel 25 T CO_2 eq/year, and wood 2.5 T CO_2 eq/year). Consequently, compost contributed to a minor net reduction in global warming (11 T CO_2 eq/year). Despite the loss of energy generation from excluding FW, AD of DM in Scenarios B and C, Scenario B was most effective at decreasing CO_2 emissions due to the small benefit of compost (1% reduction in T N eq/yr).

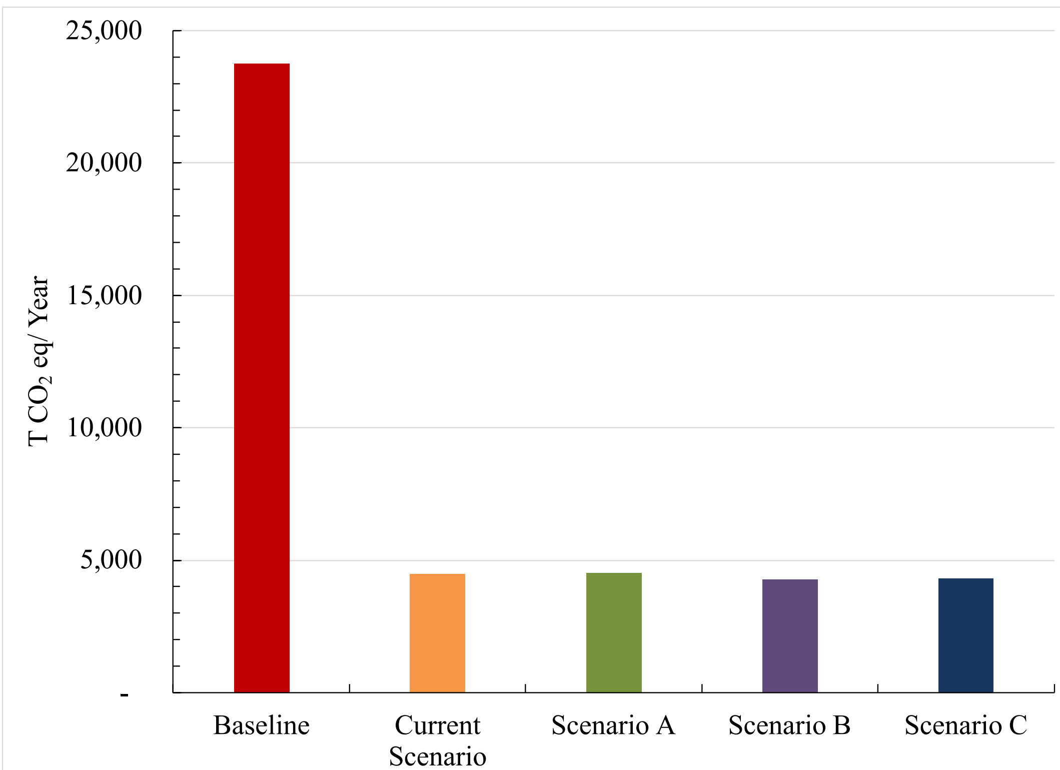

Figure 3.11: Global warming impacts from all LCA Scenarios (T CO_2 eq/year).

Compared to the baseline scenario, large N reductions were observed in the current condition (448%), Scenario A (436% reduction), Scenario B (405% reduction), and Scenario C (400% reduction) (Figure 3.12). Integrating FW in AD exhibited larger reductions in N emissions compared to AD of DM alone in Scenarios B and C. The electricity generation from co-digestion (current condition and Scenario A) decreased N emissions by 82.5% (10.2 T N eq/year) compared to Scenarios B and C (1.78 T N eq/year). Eliminating composting (1.19 T N eq/year) in Scenario A removed 1 ton less N compared to the current condition. On the other hand, reducing energy generation 82.5% and eliminating compost in Scenario C increased N emissions by nearly 10 T. These data indicate that the current condition is the most sustainable approach to eutrophication reduction.

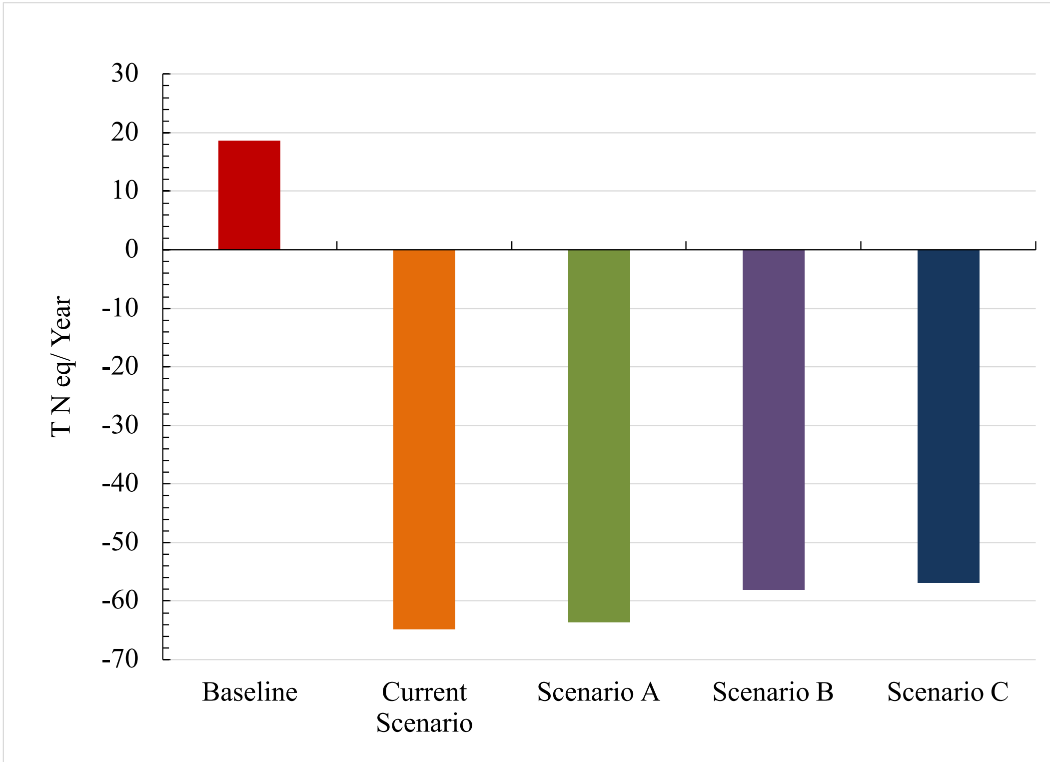

Figure 3.12: Eutrophication impacts from all LCA Scenarios (T N eq/year).

Overall, the LCA results showed that co-digesting FW and AD using the conditions from the current condition led to the greatest reductions in acidification, eutrophication, ecotoxicity, and carcinogens due to increased biomass valorization and subsequent gas capture. However, compared to the current condition, Scenarios B and C were advantageous to fossil fuel depletion, smog, global warming, non-carcinogens, and ozone attributed to excluding composting and FW. The LCA suggested that increased electricity from co-digestion resulted in a positive net energy gain while yielding the least environmental impacts. Moreover, compost had a lesser impact, though still generating a decrease in global warming and eutrophication. From this study, the LCA results indicate that CHP is a sustainable approach to energy generation from farm-scale AD due to efficient energy production, although reducing FW transport distances and higher compost generation would yield greater emission mitigation.

4. Conclusions

The results showed that AD can be used to sustainably convert DW and FW to renewable energy (1,831,450 L biogas/kg VS; 1,983,772 kWh/yr) in the farm AD system, with the creation of solids compost. The current condition revealed the lowest impacts in acidification, eutrophication, ecotoxicity, and carcinogens over three alternative Scenarios and a baseline Scenario. However, the lowest global warming (4,284 T CO_2 eq/yr) came from Scenario B (DM monodigestion, DM solids separation and composting). The DM storage lagoon had the largest global warming emissions from all scenarios, suggesting modifying the DM storage lagoon would generate the most significant reductions in global warming impact. On the other hand, the current condition portrayed the highest reductions in eutrophication (T N eq/yr) compared to all other scenarios. Nonetheless, all scenarios significantly reduced every impact category compared to the baseline scenario. From this study, it was revealed that the current condition is the optimal scenario.

Conclusions

Intellectual Merit

Chapter 2 of this thesis investigated integrating three energy generation technologies and one water treatment technology to maximize CH_4 production and remove contaminants from a combined food waste and blackwater waste stream. The results showed that pretreating gravity-separated liquid and solid fractions of the waste stream increased the organic material availability by 886% in solids and 57.9% in liquids due to cell lysis via high temperatures and pressures induced by cavitation and nucleation. After 30 days of digestion, solids with HDC pretreatment at 35 C generated 63% more CH_4 compared to non-pretreated solids, attributed to the increased substrate availability. Pretreated solids produced CH_4 at a faster rate (81.2% cumulative CH_4 after 5 days) compared to non-pretreated solids. This indicated that HDC pretreatment of solids maximized energy production while lowering the time frame for CH_4 production. Integrating MEC with AD did not yield significant energy increases with an increase of only 12.3% CH_4 generated from MEC inclusion. However, the duration of the experiment (10 days) was not sufficient for electrogenic microbes colonization at the anode. Liquids treated with EC at 15, 20, and 25 V reduced wastewater contaminants by up to 96.2% over 90 minutes. Adding KCl to EC removed up to 66% of contaminants over a reduced time frame (15 minutes) and applied voltage (10 V). These data suggested lower timeframes and voltage paired with increased electroconductivity (KCl) increased electrode efficiency and adequately met water quality parameters. HDC pretreatment of liquids did not generate significant differences in contaminant removal.

Chapter 3 investigated, monitored, and verified an farm-scale AD system co-digesting liquid DM and FW with composting of DM solids. The biogas from the system was burned in a CHP generator. An LCA was applied to the material inputs from the system to explore the environmental impacts from the current condition and 4 scenarios adapting various system configurations. Six liquid samples and three solid samples were collected over 13 months and analyzed for solids and nutrient content. Over the course of 16 months, data from an online monitoring system was provided every 15 minutes and analyzed for biogas, temperature, H_2S, CH_4 content, and energy production. This data was calculated on a yearly basis. Material inputs and emissions outputs were integrated into the LCA, with an emphasis on eutrophication and global warming impacts.

The sampling data suggested FW contributed the highest average VS (15.1%), TS (10.8%), sCOD (59,086 mg/L), nutrients (total nitrogen 3,507 mg-N/L, total phosphorous 686.5 mg-P/L and NH_3 972 mg NH_3/L), VFAs (1,607 mg/L), and CH_4 production (82.5%) in the AD system. The yearly biogas production was 1,831,450 m^3/yr with an average CH_4 content of 56.2%, where 72.2% of the biogas (1,322,307 m^3/yr) was burned in the CHP to generate 1,983,772 kWh/yr of energy. Compared to a baseline scenario with no AD or composting, the LCA data suggested the current condition generated the largest eutrophication reductions (442%), while Scenario B had the highest global warming reduction (81.9%). Co-digesting FW and DM had the highest positive impact due to the increased biogas production and subsequent energy production which offset environmental impacts in every category.

Appendix

Table A1: Material inputs and outputs for life cycle assessment (LCA).

Material/Item	Input/Output	Amount	System Component
Cows	Input	750	Cows at farm
DAF	Input	113,562 L/week	Loading rate
Cranberry	Input	22,712 L/week	Loading rate
DM Influent	Input	450,000 L/week	Loading rate
DM Solids	Input	4,651 m^3/year	DM separated solids
Concrete	Input	380 m^3	Solid separation structure
Concrete	Input	650 m^3	Sand lane
Concrete	Input	408 m^3	Concrete travel lanes, barn to parlor
Concrete	Input	1,244 m^3	Compost building
Concrete	Input	1,350 m^3	Compost pad-outside
Concrete	Input	538 m^3	Compost pad-inside
Concrete	Input	650 m^3	Food waste storage
Wood	Input	100 m^3	Compost building, barn
Electricity	Input	19,272 kWh	Pump consumption for AD per year (Gorman-Rupp 13A2-B)
DAF	Input	287,799 tkm	Transport (50 km)
Cranberry	Input	1,174,381 tkm	Transport (167 km)
HDPE	Input	3,494 kg	Digester cover
Lagoon	Input	36,416 m^3	DM/FW available storage from 3 lagoons
HDPE	Input	11,933 kg	Lagoon lining for 3 lagoons
CHP	Input	1	CHP generator for biogas combustion (230 kWh)
Diesel	Input	284,544 MJ	Energy from diesel consumption for compost agitation per year
Tractor	Input	1	Doosan DL-350
Pump Station	Input	1	Pump construction for AD
H_2S Scrubber	Input	25 m^3	Volume of scrubber for AD system

Iron	Input	184,595 kg	Iron for H_2S scrubber
Air Pump	Input	63,072 MJ	Pump consumption for H_2S scrubber (SST10-231)
Total Biogas	Input	2,403,465 m^3	Overall biogas production from AD
Annual Biogas	Input	1,831,450 m^3	Biogas per year from AD
Biogas CHP/year	Input	1,322,307 m^3	Biogas combusted per year (72.2%)
Biogas Flare/year	Input	649,570 m^3	Biogas flared off per year (28.8%)
Total Electricity	Output	2,603,362 kWh	Electricity from CHP
Annual Electricity	Output	1,983,772 kWh/year	Electricity generation from CHP per year
Compost	Output	5,000 ton/year	Compost for market

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