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Modeling frameworks to evaluate energy autarky of wastewater treatment systems

By

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A Dissertation
Submitted to the Faculty of
Mississippi State University
in Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
in Civil Engineering
in the Department of Civil and Environmental Engineering

Mississippi State, Mississippi

May 2020

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Title of Study: Modeling frameworks to evaluate energy autarky of wastewater treatment systems

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This research demonstrates the use of two novel methodologies to evaluate energy autarky status of wastewater treatment plants (WWTPs) in two steps. Step I (analysis 1 and 2) focuses on overall energy performance evaluation of a conventional activated sludge process (CAS) using a quantitative mass balance model. Step II involves development of a dynamic model that simulates a future wastewater resource recovery facility (WRRF).

The step I (analysis 1) focused on small WWTPs with treatment capacities less than 5 MGD. The results revealed that a CAS process can achieve energy autarky or energy-positive status when old technology equipment is replaced with new, high efficiency equipment to save 10-12% energy; aeration energy is reduced by installing nitritation/anammox nitrogen removal process; and energy production is enhanced with the addition of FOG for co-digestion. Analysis 2 of step I focusing on large plant capacities (i.e., > 20 MGD) evaluated the effect of influent wastewater strength (IWWS), primary treatment COD removal efficiency (PT-COD), and proper design of combined heat and power (CHP) systems on the overall energy performance. The results showed that energy autarky is feasible when PT-COD is 60% for low IWWS, 40% or greater for medium IWWS, and 30% or greater for high IWWS.

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In step II analysis, a new and dynamic model was developed by integrating high rate algal pond (HRAP) and anaerobic digester (AD) systems. The model was calibrated using the experimental data from recent studies. The results showed that this system can achieve energy autarky when advanced solids separation and co-digestion systems are included. Solids separation efficiency was increased from 75 to 90% to reduce the winter effluent COD concentrations from HRAP (by 20%). Similarly, nitrogen effluent concentrations were reduced by increasing the solids retention time. Future studies should focus on techno-economic and environmental life cycle impact analysis of these novel process configurations.

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DEDICATION

I dedicate my entire doctoral program to God, my wife and kids.

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ACKNOWLEDGEMENTS

I would first of all like to give all glory, honor, and praise to the most-high God for being faithful to me throughout my entire life. I would like to show my appreciation to my research advisor Dr. Veera Gnaneswar Gude for his guidance, encouragement, and patience. I would also like to thank my committee members Dr. Benjamin S. Magbanua Jr., Dr. Dennis D. Truax, and Dr. Seamus Freyne for their willingness to serve on my committee. I am grateful for their contributions and making themselves available to provide feedback when necessary.

There would be no Ph.D. without my lovely and supportive wife Monica A Sarpong who has been a God sent angel in my life. I couldn't have done this without her selfless and relentless support. Words cannot express my gratitude to her. This is for you "babe", I love you. Also, I would like to express my gratitude to my four kids Amaris Sarpong, Brielle Sarpong, Annalisa Victoria Sarpong, and Gideon Sarpong Jr. for believing in me. Finally, I would like to thank my parents who currently reside in Ghana, for doing whatever they could to give me the life I have today.

This doctoral journey initially looked like a dark tunnel with no light at the end, now I have finally seen the light at the end of the tunnel and another journey begins.

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CHAPTER I

INTRODUCTION

Current environmental regulations are becoming stringent and at the same time there is a growing need for industries to reduce their carbon footprint. This presents a challenge to most wastewater treatment plants in the United States with mechanical infrastructure that has reached its design life. The solution to this dilemma is a paradigm shift focusing on planning, design, and management of infrastructure to produce systems that have greater capacity and longevity. As part of this approach, future wastewater treatment plant (WWTP) design should be based on the resource recovery potential, recognizing wastewater as a valuable source of energy and nutrients. This results in the design of Water Resource Recovery Facilities (WRRF). To achieve this goal, wastewater treatment facilities are carefully examining various pathways to exploit the energy and resource recovery possibilities of wastewater as it is being treated.

Different groups of researchers have proposed various methods through which a WWTP can become a net-positive energy producer. However, these recommendations have limited application and a holistic effect of implementing the recommendations in a prospective wastewater treatment plant design has yet to be reported. It would be beneficial to envision the compound effect of the best design practices in a prospective wastewater treatment plant design and operation to realize the maximum energy recovery potential. The goal of this research is to develop energy assessment tools that can be used to evaluate the energy performance and bridge this knowledge gap by incorporating the best design and management practices reported by actual plant

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performance reports and research studies into simple and dynamic quantitative models, so that a comprehensive solution to transform existing WWTPs into a WRRFs without major infrastructural changes can be developed. Further goal of this work is to propose a new WRRF configuration by integrating different energy-yielding biological operations for wastewater treatment.

1.1 Research Objective

The research objective is to develop a quantitative mathematical model to evaluate the energy related performance of wastewater treatment plants at small and large capacities The model will serve as an assessment tool for energy analysis of a given wastewater treatment plant (WWTP). The model outcome will then be used to propose a path to energy self-sufficiency in future WWTP designs. A dynamic mathematical model will then be developed to perform a comprehensive energy analysis of the proposed future WWTP systems.

1.1.1 Research Questions

Some questions have to be answered to achieve the research objectives which are listed below.

1. What can current conventional WWTPs (such as Conventional Activated Sludge) do right now to achieve energy self-sufficiency?
2. How would the integration of new technologies for nutrient removal affect the energy performance of current conventional WWTPs?
3. How does the sensitivity of operating parameters such as varying influent wastewater strength, varying primary settler efficiency etc. affect the overall energy performance?

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4. What proposed future configurations depict wastewater treatment process as energy source?
5. What are the operational parameters that impact the performance of this configuration?

1.1.2 Research Approach

In order to achieve the objectives of this research and answer the research questions, this study will be organized into two steps. Step I consists of three different analytical approach to answer research questions 1 to 3 in section 3.1. Whereas, step II will use a time dependent variable model to answer research question 4 in section 3.1. The presentation of the different approach for this research are briefly summarized in the subsequent sections:

1.1.2.1 Step I (Analysis 1)

Analysis 1 covered in "Chapter 3" presents hypothetical concepts for three process schemes which progressively build upon the concept of transformation of a conventional activated sludge wastewater treatment plant (CAS-WWTP) into a water resource recovery facility (WRRF). These schemes also include a theoretical (but practically feasible) WWTP configuration which represents an alternative energy self-sufficient wastewater process train for future designs.

1.1.2.2 Step I (Analysis 2)

Analysis 2 covered in "Chapter 4" uses a quantitative model to perform a detailed analysis of two (basic and moderate) energy-neutral or energy-positive wastewater treatment configurations. In addition, a novel and practically feasible energy-positive wastewater treatment scheme incorporating advanced solids separation is presented with energy analysis and a case

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study. This model can be useful to quickly assess the energy recovery potential of small scale wastewater treatment systems.

1.1.2.3 Step I (Analysis 3)

Analysis 3 covered in “Chapter 5” also uses a quantitative model to presents a systematic analysis of different wastewater treatment scenarios based on wastewater strength, plant capacity, primary treatment efficiency, and different supplemental feedstock to evaluate the potential for transitioning of WWTPs into WRRFs.

1.1.3 Step II

In step II, covered in “Chapter 6” present a novel coupled high rate algae pond model and anaerobic digestion model to simulate biological conversion of light energy into chemical energy (in the form of methane) for a future WRRF. A computer software (Matlab R2019a) was used to code series of ordinary differential equations using ODE45 solver.

1.2 Addressing Knowledge Gap

Currently used common methodologies to evaluate energy performance of a WWTP are carbon footprint analysis, data envelopment analysis, economic efficiency analysis, life cycle analysis, normalization, and plant-wide modeling. These methodologies are briefly discussed below:

Economic efficiency analysis (EEA) is exclusively based on the WWTP capital costs, operating costs and economic benefits. This is linked to the energy features of the process in terms of reducing operating costs by using advanced control systems and increasing economic benefit by increasing energy recovery (Piao et al 2016, Guerrine et al. 2017). Carbon footprint analysis (CFA) has been used to measure the total release of GHG emissions by WWTPs. The CFA

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methodology solely focuses on increasing aeration efficiency and reducing energy consumption by on-site energy recovery, which can help reduce the overall carbon footprint in a wastewater treatment process (Remy et al 2013, Daelman et al 2013, Haas et al 2014, Wang et al 2016). Life cycle analysis (LCA) is widely known to be a standardized procedure applied for analyzing environmental aspects of different processes (which in this case is a WWTP). Several studies have adopted LCA to analyze energy yielding AD process (Evangelisti et al 2014, Molinos et al 2014, Arashiro et al 2018, Polruang et al 2018). Data envelopment analysis (DEA) on the other hand is a technique that is widely applied for eco-efficiency assessment. This analysis is only useful when the data available is limited. The analysis links the economic cost, energy consumption, pollutant removal, and global warming effect during the wastewater treatment processes to interpret the eco-efficiency of WWTPs (Hernandez et al 2011, Garrido et al 2011, Lorenzo et al 2016, Guerrini et al 2017). Normalization has also been used for WWTP energy performance assessment. This approach consists of normalized energy performance indicators and ratios. In other words, it simply normalizes the energy use based on a given level of output or an activity (Hernandez et al 2011, Garrido et al 2011, Lorenzo et al 2016, Guerrini et al 2017). Finally, the plant-wide modeling provides a platform for multi-objective WWTP performance assessment (Flores et al 2014, Barbu et al 2017, Mannina et al 2016a, Zaborowska et 2017, Arnell et al 2017).

By examining the different energy performance analysis methods discussed above, two key limitations can be identified: a) none of these analyses includes the best design and management practices reported by actual plant performance reports and research studies into a simple quantitative model so that a comprehensive solution for transforming an existing WWTP into a WRRF can be evaluated and developed; and b) none of the methodologies have been used to

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evaluate a combined plant-wide energy performance analysis of a microalgae wastewater treatment system with bioenergy production.

1.3 Significance/Relevance of Research

Considering the knowledge gap presented in the previous section, it is clear that there is a need to develop a quantitative model that can predict the performance of the WWTPs with input from the field. The quantitative model developed in this study for this analysis is the first of its kind. In addition, a plant-wide dynamic model of microalgae and high performance sludge removal wastewater resource recovery facility will be the first of its kind as well.

It is important to note that, even though a plant-wide modeling of WWTP has been well established and the feasibility of bacteria-microalgae wastewater treatment process has already been demonstrated, further studies in the field of energy evaluation are needed to help overcome some of the technical difficulties in scaling up the technology for industrial application. It is also worth noting that existing microalgae models such as WASP, QUAL 2K, Lake 2K, CE-QUAL 2k, and River model 1 cannot be used for energy performance analyses such as this. Hence, this is a significant contribution to the field of energy positive wastewater treatment.

A few practical implications of this research are that: 1) the model can be a beneficial assessment tool for different wastewater treatment systems; 2) this study provides technical information to design engineers, stakeholders and decision makers considering expansion/ upgrading or building new WRRFs; and 3) this work presents several alternatives for existing and future plants to improve their energy performance.

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CHAPTER II

ENERGY CONSUMPTION AND GENERATION IN WASTEWATR TREATMENT

2.1 Introduction

Population growth, in general, increases the burden of managing higher volumes of waste, in the form of gas, liquid and solid. Domestic wastewater is the most common waste stream, which has important and adverse impact on the environment. About 78% of the United States' (U.S.) population receives collection and treatment services from over 15,000 municipal wastewater treatment plants (WWTPs). The energy consumption for the wastewater treatment accounts for nearly 4% of the entire U.S.'s electrical demand, treating an average wastewater flow of about 32,345 million gallons per day (MGD) (Mo and Zhang, 2013; Yanwen et al., 2015).

Wastewater treatment plants (WWTPs) contribute to anthropogenic greenhouse gas (GHG) emissions, which is a major cause of global warming although they are considered as natural cycle of emissions by the USEPA (USEPA 2006). Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are the main constituents of GHG that is emitted from wastewater treatment processes. CO_2 is formed under aerobic condition during microbial degradation and through combustion of organic matter. CH_4 is generated through the degradation of organics under anaerobic conditions, while N_2O is produced as the result of the biological removal of nitrogen (N) through enhanced nitrification and denitrification processes (Hiroko et al., 2014). The need to reduce these emissions and to identify the factors controlling the GHG emissions from wastewater treatment plants is on ascendency (Kampschreur et al., 2009). CH_4 emissions from WWTPs occurs mostly during

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anaerobic decomposition (and anaerobic digestion of sludge) whereby methanogens are activated. CH_4 can be collected and used as an energy source, indirectly reducing CO_2 emissions (Oshita et al., 2014). Besides the anaerobic decomposition or digestion process, the sludge thickener produces the highest amount of CH_4 of 2.1 gCH_4/kg BOD_5 whereas the aerobic reactor produces the highest N_2O of 1.26 gN_2O/ kg TN (Hwang et al., 2016). Therefore, WWTPs are recognized as one of the major sources of GHG emissions (Yan et al., 2014).

There is an increasing number of literature contributions focused on reducing energy requirements or even on energy positive wastewater treatment processes (WERF, 2009; Elías-Maxil et al., 2014; McCarty et al., 2011; Funamizu et al., 2001; Gude, 2015a; Chae and Kang 2013; Nowak et al., 2011; Frijns et al., 2013). The current technologies used in most of the WWTP were designed years ago when GHG emissions and energy consumption or production were not major concerns. According to the US EPA, there are over 14,700 municipal WWTPs and 48% of the plants use anaerobic digester (AD) for sludge stabilization and less than 10% actually uses the biogas produced from the AD for heat or electricity production. There are about five utilities in the US and four in Europe that have achieved 100% or more energy production. This chapter discusses energy consumption and recovery trends in wastewater treatment systems and presents three different classifications energy positive wastewater treatment configurations. Case studies including mass and energy balances are presented in detail.

2.2 Energy Consumption in Wastewater Treatment Systems

In the U.S., approximately 3% to 4% of national electricity consumption is used for transmitting and treatment of water and wastewater (Goldstein and Smith, 2002; Galbraith, 2011). Typically, about 30% of the operational cost is due to energy usage (Tchobanoglous et al., 2003). Energy costs represent a large portion of operating costs for utilities since it is normally required

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in all the stages of the treatment process, from influent pumping to discharge of treated effluent. Figure 2.1 (A-D) shows the variability of energy requirements within different wastewater treatment technologies (A); in different countries (B); at different capacities (C); and individual unit operations and processes (D).

Specific energy consumption for wastewater treatment depends on the process technology and configuration and treatment (Fig.1.1A and Table 1.1). According to Table 2.1, the specific electrical energy consumed for the different conventional treatment technologies range from 0.3 to 0.6 kWh/m^3. Adaptation of lagoon or pond type technology yield less energy consumption (0.07 – 0.3 kWh/m^3); whereas treatment technologies based on mechanical aeration such as oxidation ditch or high purity oxygen and activated sludge processes consume the highest energy (> 1 kWh/m^3).

Advanced technologies such as membrane reactors and extended aeration systems further increase the specific energy consumption. For instance, the addition of a reverse osmosis (RO) for water reuse will triple or quadruple the utilities energy consumption. Figure 2.1-C shows that energy consumption varies depending on the treatment technology and plant capacity. Specific energy consumption is inversely proportional to the plant capacity for plants with capacities under 10 MGD and it does not change significantly beyond that capacity.

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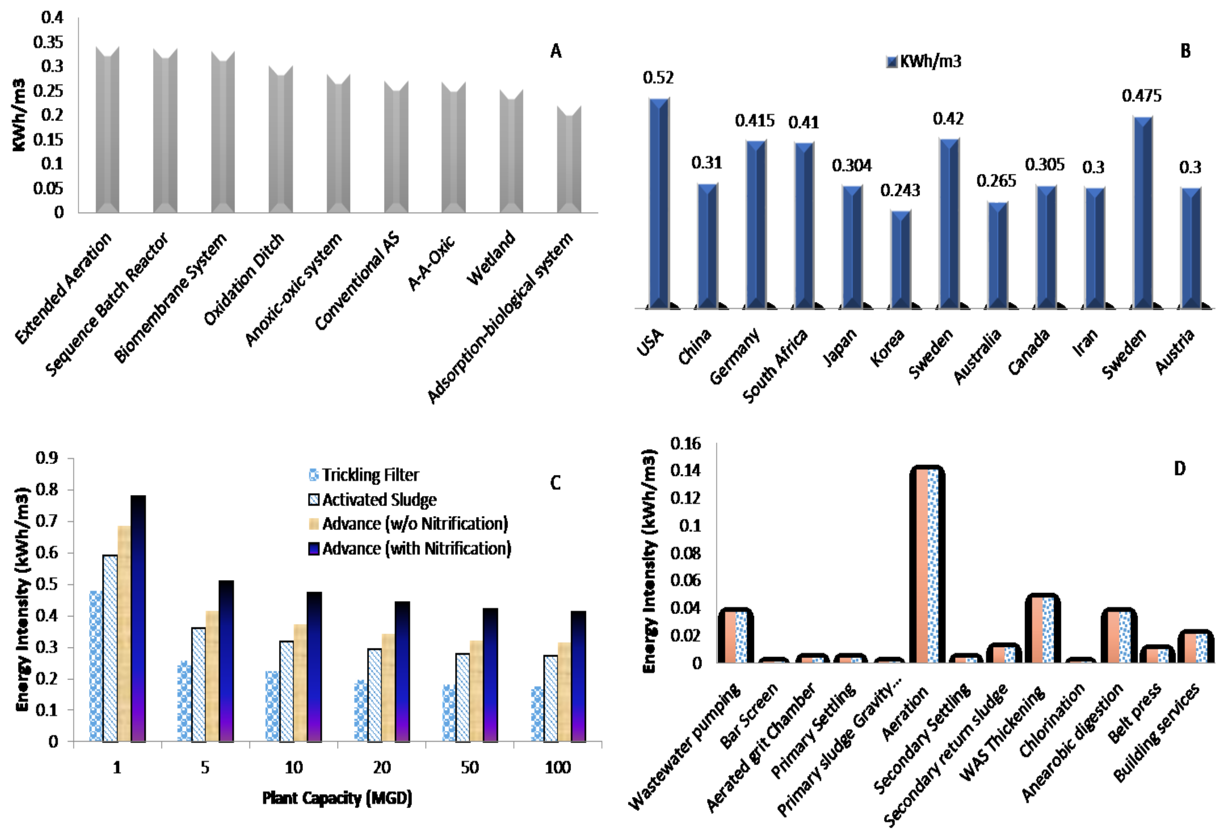

Figure 2.1 Specific energy consumption in wastewater treatment; (A) Energy Requirement for different biological treatment technologies; (B) Energy consumed by wastewater treatment processes across the world; (C) Energy consumption intensity for different treatment technologies at different capacities; and (D) Specific electricity consumption in individual unit processes (Renan et al., 2017; Goldstein and Smith 2002)

Among the few selected countries shown in Fig.2.1B, US plants consumed an estimate of 0.52 kWh of electrical energy for every cubic meter of wastewater treated; this is probably due to the aging infrastructure. The European countries and South Africa have the second highest (> 0.4 kWh/m^3) energy consumption; Australia, Iran and the Asian countries record the lowest energy requirement (< 0.31 kWh) for treating wastewater (Renan et al., 2017).

WWTP capacity has a significant impact on the specific energy consumption. Fig. 2.1C shows the effect of plant capacity on four different treatment technologies. The specific energy

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consumption for all the different technologies decreases as plant capacity or size increases. In recent years, design and operation of WWTP has increasingly focused on improving or minimizing energy consumption and reducing cost of operation, without compromising on the treated water quality. Fig.2.1D shows a typical distribution of energy use in a conventional activated sludge process with treatment capacity of 10 MGD (Goldstein and Smith 2002). About 44% (0.14 kWh/m^3) of the energy consumed by the wastewater operation is used for biological process such as the aeration tank, followed by waste activated sludge thickening process (~15%), anaerobic digestion and pumping both at 12%

The application of high efficiency equipment and improvement of design and operation can potentially lower energy consumption and maximize energy recovery. However, if additional energy present in wastewater were captured for use and even less were used for wastewater treatment, then wastewater treatment could become a net energy producer rather than a consumer (Logan, 2005).

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Table 2.1 Wastewater treatment technologies and their specific energy consumption

Technology Type	Plant Capacity (MGD)	Treatment Technology	Consumption (kWh/m^3)	Source
Conventional	4	Activated Sludge Process	0.353	Schwarzenbeck et al., 2008
	5	CAS with Nitrification	0.509	Goldstein and Smith 2002
	5	Activated Sludge Process	0.434	Wolfgangsee-Ischl WWTP
	5	Activated Sludge	0.362	Goldstein and Smith 2002
	15	Activated Sludge Process	0.308	Willis, 2012; Dorr, 2011; Theiszen, 2013
	12	Activated Sludge Process	0.341	USDOE, 2012; Proctor, 2011
	67	Activated Sludge Process	0.447	Joss et al., 2010; Cao, 2011
		Activated Sludge Process	0.33-0.60	Gude 2015a
Pond		Microalgae Stabilization Pond	0.079 - 0.28	Wang et al 2016
		Aeration Ditch	0.48 - 1.03	Wang et al 2016
		Lagoon	0.09-0.29	Gude 2015a
Filter	5	Trickling Filter	0.258	Goldstein and Smith, 2002
	20	Trickling Filter	0.198	Goldstein and Smith, 2002
		Thickening filter	0.19 - 0.41	Wang et al 2016
	10.1	Biotower/Activated Sludge	0.392	PG&E 2003
		Trickling Filter	0.18-0.42	Gude 2015a
Advance		Immersed biological membrane reactors	0.8	Ortiz et al 2007
	0.1	Primary filtration and Trickling Filter	0.087	Gikas 2016
	5.5	High Purity Oxygen Activated Sludge	1.06	PG&E 2003
	528.3	Anoxide-anaerobic-oxide	0.13	Kang and Chae 2013

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2.3 Energy Recovery Trends in Wastewater Treatment Systems

Wastewater contains approximately 60% (dry basis) of organic compounds; which is 50-55% carbon and mostly biodegradable (in the form of bCOD), 10 – 15% is nitrogen (as N) and 1-3% is phosphorus (as P) (Gude 2015b). The energy in the nutritional components of the wastewater such as N and P is approximately 0.7 kWh/m^3 (Chae and Kang 2013). The energy contained in wastewater solids is 3.2 kJ/g of total solids (Nowak et al., 2011). The sludge from the primary treatment is reported to contain 15 – 22.8 kJ/g; secondary is 12.4 – 16.1 kJ/g; digested sludge contains about 11 kJ/g on a dry mass basis (Figure 2) (Zanoni et al., 1982; Gude 2015b; Shizas and Bagley, 2004). Shizas and Bagley, 2004 reported that about 66% of energy content entering the WWTP is captured in the primary sludge, 42% of the remaining energy is retained in the secondary sludge, and the biogas contains 47% of the energy entering the digester. It is apparent that, there is enough energy in wastewater (in the form of biogas from the AD) which represents a renewable fuel source that could be converted into electricity and heat. The available thermal heat for heat-pump extraction is about 7 kWh/m^3. According Figure 3 it requires ~1.5 kWh/m^3 to treat 1 kg of COD which contains ~3.9 kWh/m^3 (Chae and Kang 2013). Similarly, the energy required (and contained) to remove nitrogen and phosphorus are ~13 kWh/m^3 (~19 kWh/m^3) and ~6.44 kWh/m^3 (~2 kWh/m^3) respectively (McCarty et al., 2011).

Energy contained in wastewater can be harvested using various physical chemical, and biological processes such as thermal treatment (gasification, incineration, liquefaction, and pyrolysis); composting to produce various valuable biofuels and nutrient-rich biosolids and finally anaerobic digestion (AD). Energy can be recovered from influent organic matter and nutrients, kinetic energy from wastewater flow, and residual heat in treated wastewater (Mo and Zhang 2013). The most common practice is that, resources recovered in the form of "energy" are used

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directly by the WWTPs and other facilities reducing potential environmental loads by WWTPs (Goldstein and Smith 2002; Wilkinson, 2000). Sometimes, onsite energy generation helps to not only reduce energy cost, but also remove the hazardous contaminants in the wastewater and improve treated water quality (Goldstein and Smith 2002). Some of the technologies used for wastewater resource recovery are; combined heat and power (CHP) (EPA, 2007; Stillwell et al., 2010), biosolids incineration, effluent hydropower, onsite wind and solar power, and bioelectrochemical systems.

According to the USEPA, the CHP units produce electricity at a cost below retail price, displace purchase fuel for thermal needs, qualify as a renewal source, reduce carbon footprint, and are reliable for onsite heat and power generation (EPA, 2007). However, CHP unit requires a high capital cost from \$2,000 – \$7,500/kW. EPA also reported that, reported that the CHPs are only cost-effective for the WWTPs with a flow rate above 5 million gallons per day (MGD). Stillwell et al., reported that, WWTPs could achieve a reduction of 26% in electricity consumption if CHP is adopted (Stillwell et al., 2010). Anaerobic Digestion (AD) is the key component for energy (biogas) production with a CHP. AD has been tested and demonstrated to be the best option for recovering the maximum energy from primary and secondary sludge of a municipal wastewater treatment plant through energy-rich biogas production. During the anaerobic digestion process the organic waste is decomposed to CH_4 (60% by volume) and CO_2 (30% by volume) (Tchobanoglous et al., 2003).

Hence, electricity is generated by using biogas as a fuel. Most municipal wastewater treatment utilities incinerate dewatered biosolids as a means of disposal, which requires dewatering prior to incineration. Other biosolids management methods include use as fertilizers or soil stabilizers or disposal in a landfill (Tchobanoglous et al., 2003). According to the US EPA and

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USDoE, the heating value of biogas is approximately 37.3 kJ/m^3 (550 BTU/ft^3), which is about 60% of the heat value of natural gas. An estimated 628–4,940 million kWh could be saved annually in the United States by AD if all WWTPs could use the biogas produced (Stillwell et al., 2010). The use of biogas by individual utilities can result in significant energy savings if done properly. Biogas can be used on-site in different ways, such as generating heat for the process; generating heat for space heating and cooling; powering engines used to drive equipment directly; powering engines used with generators to drive remote equipment; and powering engines used with generators to produce general purpose electrical power (EPA and USDE 1995).

Biosolids incineration is another technology that is widely employed in most utilities, however it comes with some major disadvantages which includes; the release of persistent environmental pollutants, quality inconsistency, and the relatively high capital investment ($66/dry Mg) and energy cost for dewatering the biosolids (EPA, 2007; Cartmell et al., 2006; Mahmood and Elliott, 2006; Wang et al., 2008).

One of the unique technologies mentioned above is onsite application of wind and solar power. This application produces electricity from wind and/or solar energy by taking advantage of the large available land of the WWTPs. Table 2.2 below shows a few state-of-the-art WWTPs with onsite wind and/or solar power generation. Location, climate condition and large capital investments are some of the drawbacks for solar and wind onsite electricity generation.

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Table 2.2 WWTPs with Solar and wind Electricity Generation

Technology Integration	Location	Utility Name	Energy Production Potential	Application	Reference
Solar	CA, USA	Oroville Wastewater Treatment Plant	520 kW	Provide 80% of facility needs	SPG Solar
Solar	CO, USA	Boulder Wastewater Treatment Plant	1000 kW	Provide 15% of facility needs	Boulder, 2012
Solar	NJ, USA	Atlantic County Utilities Authority	500 kW	Provide 660,000 kWh of energy to the facility per year	ACUA, 2011
Wind			7500 kW	Provide 70% of facility needs	ACUA, 2011
Wind	MT, USA	Browning Waste Water Treatment Plant	40 kW	Displace grid electricity used at facility	Browning, 2001

Electricity or energy production via heat pump has been reported to produce 597×10^3 MWh low-temperature heat energy using 199×10^3 MWh electrical energy for a treatment capacity of 119 MGD. Heat pumps are mainly useful when there is a need for onsite heating and cooling within a short range.

Microbial fuel cells (MFC), a type of bio-electrochemical systems, is another promising technology that has been widely studied over the last 15 years for resource recovery (Pant et al 2010). MFC directly converts microbial metabolic or enzyme catalytic energy into electricity by using conventional electrochemical technology. The technology has the potential of harvesting the energy contained in wastewater; however, it has only been applied on pilot scales for wastewater treatment so far (Allen and Bennetto 1993; Park and Zeikus 2000; Roller et al., 1984; Foley et al., 2010; Kim 2009). Beyond energy generation, another key advantage of the MFC is, it can also reduce the sludge by 20% when compared with the conventional treatment, thereby reducing the sludge disposal costs. However, there are some drawbacks prohibiting the large-scale use of MFC, which include energy loss during the electricity generation process, low organic utilization rates

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and high capital costs (around 800 times of an anaerobic system) (McCarty et al., 2011; Lui et al., 2004).

Phototrophic technology in terms of microalgae cultivation is another promising technology for onsite or offsite energy generation (Mo and Zhang 2013). Inorganic or organic carbon and nutrients from wastewater are utilized for microalgae cultivation and microalgae are reported utilize carbon dioxide much faster than conventional biofuel crops (ESMAP, 2008). Currently, integrating the phototrophic technology in WWTPs is still in research phase. The main challenges of this integration reported include: (a) algal cultivation cost reduction; (b) harvesting, dewatering and lipid extraction energy reduction; and (c) microalgae species selection for optimal performance (ESMAP, 2008).

2.4 Considerations for Energy-efficient Wastewater Treatment

Energy recovered in a utility can directly offset the energy costs of the WWTPs; however, there are several limitations and uncertainties, such as large capital costs, lack of reliability and specific requirements for climate and local conditions (Mo and Zhang 2013). In the case of biogas production for a CHP, the major challenge is economic and political factors, which often prevent the direct sale of digester gas. Given that over 90% of WWTPs in the U.S. are small plants, the major challenge is to improve/innovate technologies that have low capital costs, are simple and affordable to operate, and are easy to integrate into the existing small plants (Mo and Zhang 2013). Life cycle energy benefits associated with reducing and reusing organic and nutrient loadings from wastewater and waste volume for downstream handling are infrequently studied (Mo and Zhang 2013). Lack of life cycle analysis and lack of studies examining the integration and tradeoffs of for energy and resource recovery is another challenge. Studies are needed to evaluate the maximum

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amount of energy that can be generated onsite with consideration of such integrations and tradeoffs (Mo and Zhang 2013).

Reducing electricity consumption of WWTP can be approached through improvement of both the hardware (mechanical equipment) and soft technology (process and operation). Among the hardware, the biological process (i.e. aeration facility) is the main electricity consumer and minimizing energy consumption of the aeration process is the key. On the other hand, current sludge regulations on biosolids disposal have become the driving force for municipal wastewater plants to focus on energy recovery. The solids treatment process is another challenge; it significantly affects the cost of buildings and operating a WWTP, which accounts for about 50% of a wastewater plant's capital cost (Joss et al., 2010).

By employing the best available technologies, near and long term planning for energy self-sufficiency is achievable. The energy intensity of a conventional wastewater treatment plant with nutrient removal and tertiary treatment is assumed to be 0.47 kWh/m^3 for a 10 MGD plant capacity (Goldstein and Smith 2002). For example, up to 30% of energy achieved through improvement in the aeration system by selection of higher efficiency facilities and optimal process control. This will result in reduction of specific energy consumption from 0.473 to 0.331 kWh/m^3. Some of the recommended options to achieve energy-positive status are; enhancing primary settling tank performance by harvesting more bCOD to anaerobic digester; incorporating sludge pretreatment to increase VSS destruction; using high efficiency electrical generators; and co-digestion.

2.5 Enhancing Energy Recovery in Wastewater Treatment Systems

Figure 2.2 shows the possible ways of energy reduction and production in WWTPs classified as "basic", "moderate" and "advanced" configurations. The basic category is mainly focused on improving energy recovery with supplemental biogas production via co-digestion and minor

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upgrades to minimize energy consumption. The moderate configuration employs higher energy efficiency processes and process components to significantly reduce the energy consumption; and the advanced configurations consists of hypothetical designs that reduce energy consumption and enhance energy production. Some of the energy positive wastewater treatment systems are listed in Table 2.3.

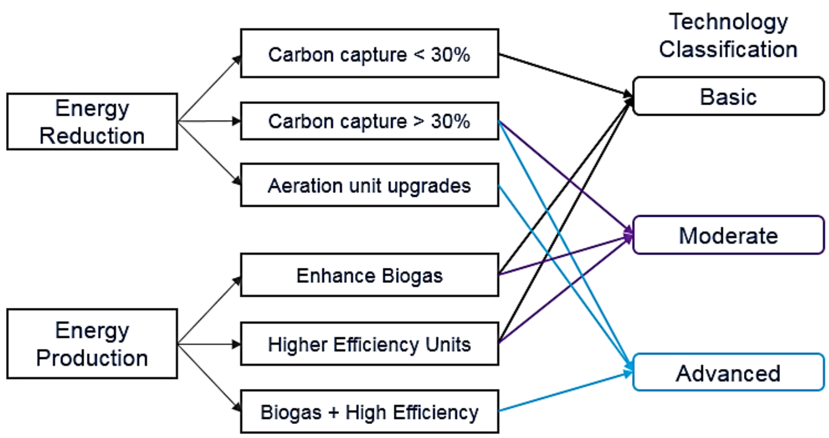

Figure 2.2 WRRF Classification – *basic* (possible with process upgrades) configuration consists of traditional wastewater treatment process with no upgrades; *moderate* (possible in near future with upgrades in equipment, process configuration, and treatment scheme) configuration is a modification of the basic process configuration to include the "Anammox" process which focuses of nutrients removal by using nitrite as electron acceptor and CO_2 as energy source; *advanced* (possible in future, more preliminary work is required) configuration incorporates major process modifications to replace the energy intensive biological process of the basic configuration with a less energy consuming treatment technology such as microalgae systems, trickling filter etc. This configuration also adopts an advanced primary treatment filtration, which focuses on higher biodegradable solids removal for enhanced energy production).

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Table 2.3 Net positive (100% plus) energy wastewater resource recovery facility

Location	Plant Name	Plant Capacity (MGD)	Energy Production (Biogas GWh)	Energy Produced - Biogas (kWh/y)	Energy Produced - Biogas (KWh/d)	Energy Produced - Biogas (kWh/m^3)	Anaerobic Digester Feedstock	Reference
NY, USA	Gloversville-Johnstown Joint WWTP	11	28	28×10^6	76.7×10^3	1.842	PS+WAS+HSW	Ostapczuk, 2011
WI, USA	Sheboygan Regional WWTP	11	32	32×10^6	87.7×10^3	2.105	PS+WAS+HSW+FOG	USDOE-Oregon, 2012; Doerr, 2011; Thieszen, 2013
OR, USA	Gresham WWTP	13	17.2	17.2×10^6	47.1×10^3	0.958	PS+WAS (~0.06 MGD)+FOG	Proctor, 2011
CA, USA	East Bay Municipal Utility District WWTP (EBMUD)	70	90	90×10^6	246.6×10^3	0.931	PS+WAS+HSW+FOG+FW	Williams, 2012; EBMUD
CA, USA	Point Loma WWTP	175	193	19.3×10^6	52.8×10^3	0.798	PS+WAS (~1 MGD)	Wiser et al., 2012; Boranyak, 2012;Greer, 2011; Mazanec, 2013
Germany	Grevesmuhlen WWTP	4	1.95	1.95×10^6	5.3×10^3	0.353	PS (10%)+WAS (60%)+GSS (30%)	Schwarzenbeck, 2008
Austria	Wolfgangsee-Ischl WWTP	5	3	3×10^6	8.2×10^3	0.434	PS+WAS	Nowak et al., 2011; Nowak, et al., 2015
Austria	Strass im Zillertal WWTP	6	10	10×10^6	27.4×10^3	1.206	PS+BNR/WAS+TG+FOG (0.009 MGD)	Crawford, 2010; Wett, 2007a
Switzerland	Zurich Werdholzli WWTP	67	41.6	41.6×10^6	113.9×10^3	0.449	PS+WAS+FOG	Cao, 2011; Williams, 2012

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It is evident that existing WWTPs in their current form cannot be energy self-sufficient unless process improvements such as; (1) reducing energy consumption by replacing energy-intensive mechanical equipment with more energy-efficient equipment; (2) minimizing aeration energy by implementing online monitoring and using micro-bubble diffusors; (3) implementing innovative energy-efficient nitrogen removal technologies; and (4) enhancing energy production by co-digesting supplementary feedstock such as FOG and sewage sludge are considered. The subsequent chapters will describe in detail the different approaches and methodologies considered in this research for energy assessment of wastewater treatment plants.

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CHAPTER III

NEAR FUTURE ENERGY SELF-SUFFICIENT WASTEWATER TREATMENT SCHEMES

3.1 Abstract

A new paradigm shift in municipal wastewater treatment plant (WWTP) operations is to achieve energy self-sufficiency, while simultaneously complying with permit requirements. Less than 10% of U.S. WWTPs produce energy for beneficial use and only a handful of these plants are energy self-sufficient. We propose three energy-positive WWTP operating schemes and use a quantitative mass-balance model to assess their potentials in carbon and nitrogen removal and energy generation from municipal wastewater. This research identifies potential challenges in the selection and implementation of energy recovery process configurations and, proposes practically feasible, energy-positive WWTP process configurations. Energy recovery through biogas production, and aeration energy optimization are the two main approaches to achieve energy self-sufficiency. Moving forward, the main alternative strategy to enhance energy recovery in the near future is (i) to enhance COD capture in primary sludge to boost energy production; (ii) replace activated sludge process with a less energy intensive biological treatment technology to conserve energy; and (iii) to increase energy production by adding fat, oil and grease containing supplementary feedstock in anaerobic codigestion. This chapter presents quantitative analysis of three process schemes which progressively build upon the concept of transformation of a conventional activated sludge wastewater treatment plant (CAS-WWTP) into a water resource recovery facility (WRRF). These schemes also include a hypothetical (but practically feasible)

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WWTP configuration which represents an alternative energy self-sufficient wastewater process train for future designs.

Keywords: codigestion; energy recovery; sludge; anammox; wastewater treatment; high rate algae pond; advance primary treatment filtration

3.2 Introduction

About 78% of the United States (U.S.) population receives collection and treatment services from over 15,000 municipal WWTPs contributing to more than ~4% of the entire U.S. electrical demand, treating an average wastewater flow of about 32,345 million gallons per day (MGD) (Mo and Zhag 2013; Shen et al. 2015). Typically, roughly 30% of wastewater treatment operational cost is assigned for energy use (Tchobanoglous et al. 2003). Conventional activated sludge process is the most commonly used method to convert waste organic matter is used water into biomass and carbon dioxide (CO_2).

In recent years, WWTP design and operations have increasingly focused on minimizing energy consumption and reducing the cost of operation, without compromising effluent quality. The specific energy consumption of wastewater treatment is about 0.5 kWh/m^3 as shown in **Figure 3.1** (Gude 2015a). In general, aeration energy is the largest energy consumer for CAS-WWTPs. It ranges between 49 and 60% of total energy consumption in a plant (Goldstein and Smith, 2002). Other studies show that reducing the activated sludge age by reducing mean cell residence time (MCRT) decreases the net energy use (Shi 2011, Joh et al., 2010). By reducing the sludge age, sludge production is increased, which may be desirable if the intent is to digest the sludge for methane production. The other higher energy consumers besides the aeration process are the waste activated sludge thickening process (11%), anaerobic digestion (9%) and pumping (8%) (Goldstein and Smith, 2002). Moreover, addition of advanced treatment technologies for tertiary

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treatment increases the specific energy consumption significantly. For instance, addition of a reverse osmosis (RO) unit for water reuse application will triple or quadruple the utility's energy consumption. However, application of high-efficiency equipment and improvement of design and operations could potentially lower energy consumption and maximize energy recovery. For instance, replacing coarse bubble aeration diffusers with fine pore diffusers will result in 45% energy reduction (Pakenas 1995). online sensors for a dynamic control of the process saves 30% energy (Monteith et al. 2007; Wett et al. 2007a, 2007b). About 10% of energy can be saved by including infield filters for aeration blowers (Jonasson 2007). Replacement of less-efficiency pumps with more-efficient pumps equipped with variable frequency drive (VFD) will save up to 80% of energy (EBMUD 2017, Pakenas 1995, Jonasson 2007).

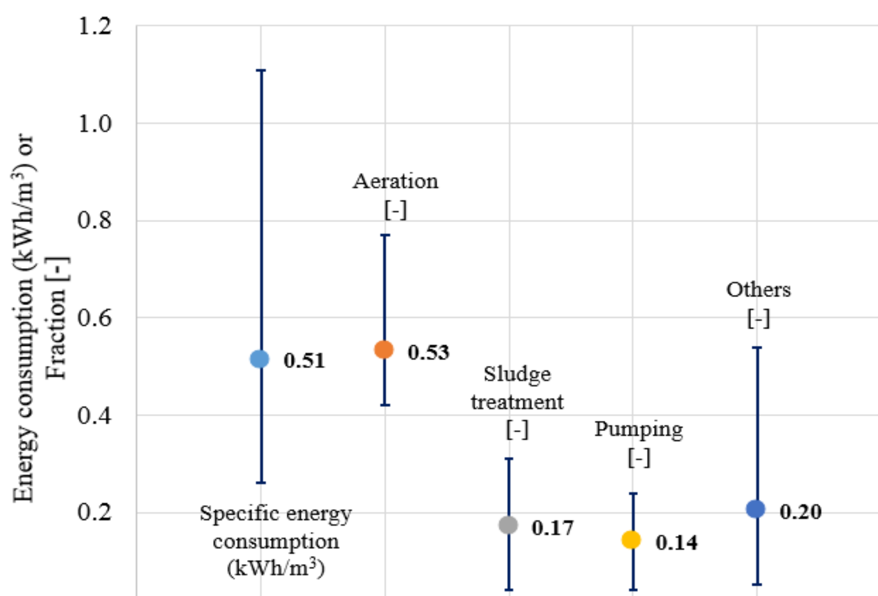

Figure 3.1 Specific energy consumption (kWh/m^3) of wastewater treatment in a conventional activated sludge process, aeration accounts for 53% with other significant components.

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Energy recovery assessment of a WWTP should account for both energy consumption and energy generation in different stages. Wastewater contains energy in different forms, organic carbon, nutrients and suspended solids as shown in **Figure 3.2** (Gude 2015a). The energy in the nutritional components of the wastewater such as N and P is approximately 0.79 kWh/m^3 (Chae and Kang 2013). The energy contained in wastewater solids, sludge, is about 2.72 kWh/m^3 of total solids (Gude 2015b). Sludge digestion produces biogas in anaerobic digesters, AD (Zanoni and Mueller 1982). About 48% of all the WWTPs in the U.S. use AD for sludge stabilization and less than 10% actually uses the biogas produced from the AD for heat or electricity. Despite the scientific evidence of wastewater treatment operation as energy producer, very few (~ 10) utilities in both the US and Europe that have utilized these concepts to achieve 100% or more energy efficiency (Shen et al. 2015).

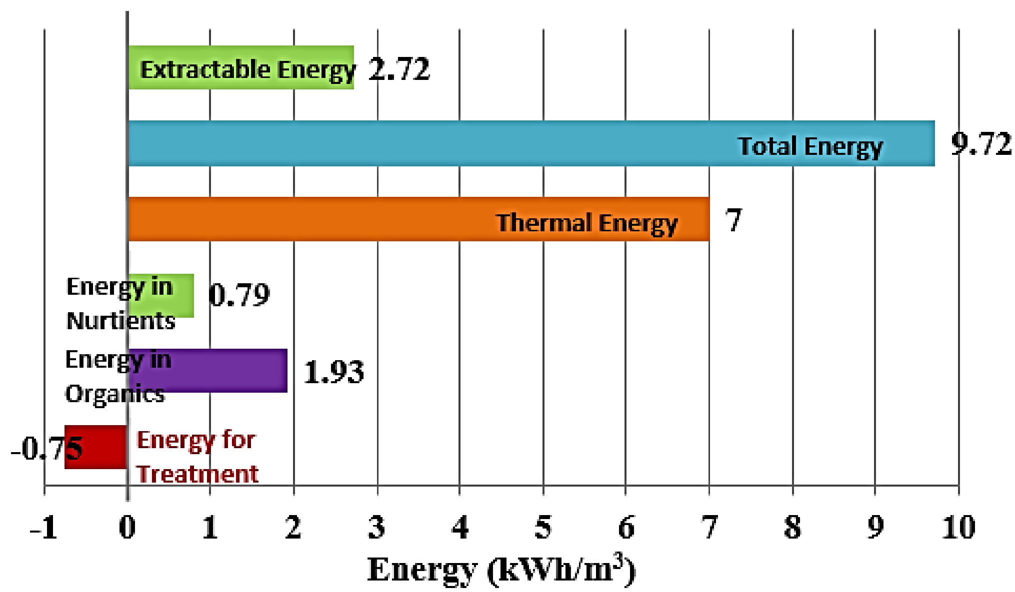

Figure 3.2 Energy content in municipal wastewater sources: energy extraction from organic compounds is a feasible method while thermal energy extraction requires more advances in research (Gude 2015a)

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Energy embedded in sludge varies; for instance, primary, secondary and digestate sludges contain about 15–22.8 kJ/g, 12.4–16.1 kJ/g, and 11 kJ/g, respectively (Zanoni and Mueller 1982; Shizas and Bagley 2004). About 66% of energy content entering the WWTP is captured in the primary sludge. The remaining energy from the primary effluent is retained in the secondary sludge, and biogas (Shizas and Bagley 2004). This information is critical for identifying ways to recover energy content in wastewater and to determine the efficiency of energy recovery schemes. Thus, enhanced energy recovery from wastewater can make the process a net energy producer (Logan 2005).

Approximately ~1.5 kWh/m^3 is required to treat 1 kg of COD, which contains ~3.9 kWh/m^3 (Chae and Kang 2013). Similarly, the energy required to remove nitrogen and phosphorus are ~13 kWh/m^3 (~19 kWh/m^3) and ~6.44 kWh/m^3 (~2 kWh/m^3), respectively (McCarty et al. 2011). Regarding sludge production in a CAS process, for every 1 kg of COD removed (assuming 0.5 g of dry biomass per gram of COD removed), 0.5 g dry biomass of sludge is produced. This large amount of sludge produced by a CAS process relates to higher energy consumption and significant CO_2 emissions.

Decision makers, stake holders and designing engineers often find it difficult to agree on a feasible path for transforming an existing WWTP into a WRRF based on the numerous possibilities reported in literature. The goal of this research paper is to bridge the knowledge gap by incorporating the best design and management practices reported by actual plant performance reports and research studies into a simple quantitative model, so that a comprehensive solution to transform WWTP at different process configurations to WRRF can be developed for decision makers or interest groups. This research explores energy positive operations at the wastewater plants, based on three treatment schemes. Scheme 1 focuses mainly on making non-infrastructural

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changes to the existing CAS-WWTPs to reduce energy consumption (through equipment upgrades) and enhancing energy production (through supplemental waste feedstock). Scheme 2 (Figure 3.4) builds upon Scheme 1 (Figure 3.3) and employs an innovative process to significantly reduce energy consumption for biological treatment, which requires a few infrastructure upgrades. While the advanced Scheme 3 (Figure 3.7) involves a treatment plan which can be considered for future or new designs. A quantitative mass-balance model for a CAS-WWTP was developed to account for mass and energy flows in different unit operations.

3.3 Materials and Methods

Figure 3.3 and 3.4 are shown below including mass and energy balances. Energy and mass balances were analyzed using population equivalent (p.e.) of 135,000; treating a wastewater flow of 39,217 m^3/d (~ 10 MGD) and a medium strength influent organic loading of 18,927 kg COD/d (500 mg/L) (Tchobanoglous et al., 2003). Population equivalent for this analysis was fixed at 0.14 kg COD/(p.e.d) and 0.01 kg N/(p.e.d). COD was used to measure the amount of organics in wastewater; hence, COD was used to evaluate the potential energy consumed and recovered in the different wastewater treatment configurations. COD removal efficiencies through specific unit processes were estimated based on reported representative fractions. The process train for this configuration includes at least primary sedimentation, secondary CAS process and anaerobic digester. Modification to the CAS process layout in includes three configurations that will be considered for COD removal and energy balance. The analysis assumed energy intensity values reported by Goldstein and Smith, 2002; influent equipment (pumps, bar screen etc ~ 0.041 kWh/m^3); primary and secondary sedimentation (~0.076 kWh/m^3); secondary CAS process (~ 0.232 kWh/m^3); and anaerobic digester (~ 0.1 kWh/m^3). As mentioned before, an overall energy

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savings for equipment upgrade was fixed at 11% of total plant energy consumed. Methane (CH_4) conversion to electricity efficiency for a combined heat and power (CHP) was assumed to be 35%.

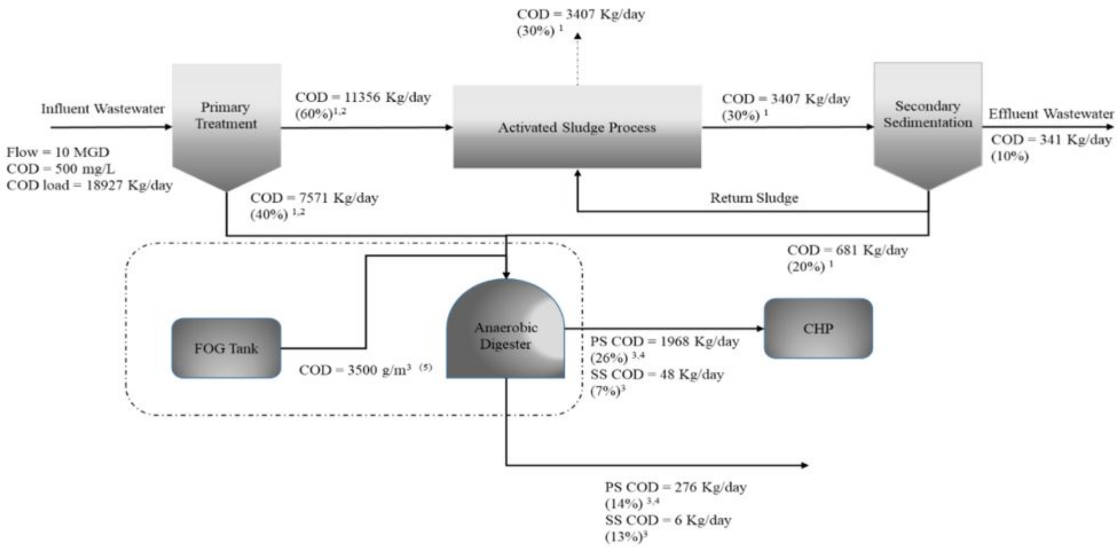

Figure 3.3 Mass and energy balance for equipment upgrade and addition of supplemental waste

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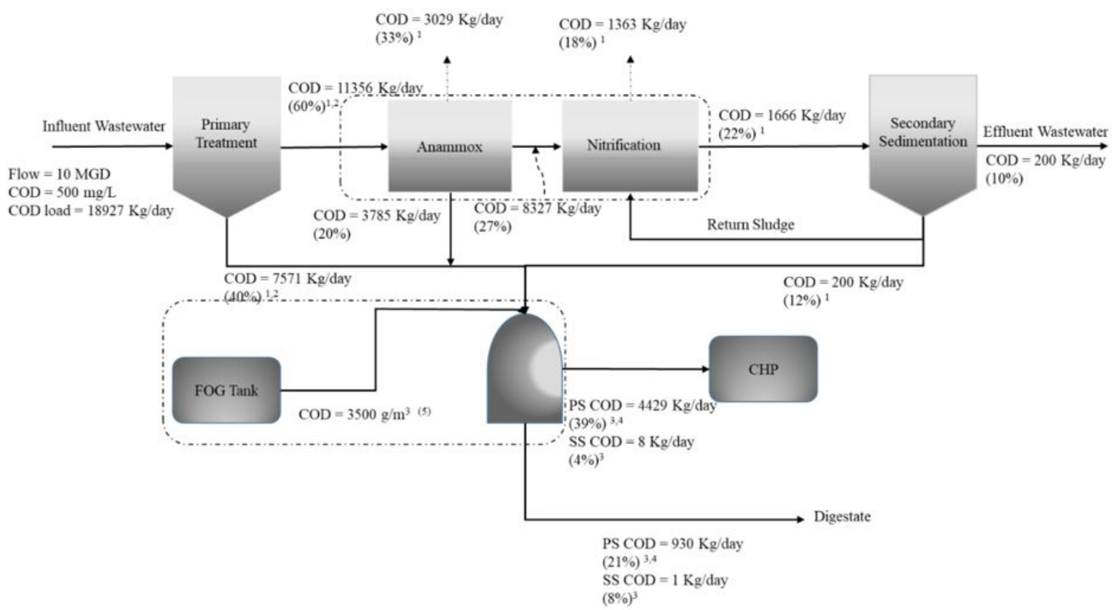

Figure 3.4 COD and energy mass balance - major modifications include equipment upgrades, use of FOG as supplemental waste and nitrogen removal with partial nitritation - anammox process (1- Tchobanoglous et al. 2003; 2- Rossle and Pretorius 2001; 3- Parkin and Owen 1986; 4- Miron et al. 2000; 5- John et al. 2009).

3.4 Results and Discussion

3.4.1 Scheme 1: COD removal – Equipment upgrades with supplemental biogas production

Scheme 1 represents changes or modifications that can be adopted by existing utilities to improve energy efficiency. Modifications applied in this scheme include equipment upgrades to minimize energy consumption, and codigestion of wastewater sludge with highly biodegradable sludge such as fat, oil and grease (FOG) to enhance energy production.

The influent characteristics were assumed as 10 MGD plant capacity, typical domestic wastewater COD concentration of 500 mg/L (~18,927 kg COD/day organics). As shown in Scheme 1, 40% (7,571 kg/day) of total COD entering the treatment plant was removed from the primary treatment tank (Tchobanoglous et al. 2003; Rossle and Pretorius 2001). Out of the 40%, 26% (1,968 kg/day)

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of primary sludge and 7% (48 kg/day) of secondary sludges were converted to biogas (Parkin and Owen, 1986). **Figure 3.5 (A)** shows energy consumption and generation profiles for Scheme 1. **Figure 3.5 (B)** shows a total of 0.378 kWh/m^3 of specific energy consumed by the plant, which includes 11% energy reduction via equipment upgrades. The published theoretical chemical energy obtained from converting 1 gram of COD to methane is 13.9 kJ (Heidrich et al. 2010).

Thus, the recoverable energy (to electricity) from both the primary and secondary sludge was estimated as the sum of (13.9 kJ/gCOD / 3,600 kJ/kWh) × 52 g/m^3 from primary treatment sludge and (13.9 kJ/gCOD/3,600 kJ/kWh) × 1.26 g/m^3 from secondary sedimentation sludge which is 0.21 kWh/m^3. This energy production represents almost 48% energy efficiency without equipment upgrade (total specific energy without equipment upgrade is 0.448 kWh/m^3) and approximately 56% with equipment upgrades. The only way for a utility of this kind to meet or even exceed the energy demand is to include supplemental waste for co-digestion. According to John et al., 2009, co-digestion of fat-oil-grease (FOG) with primary and secondary sludge will increase energy production by a factor 2.95 (this represent a soluble COD concentration of 3,500 mg/L). A combined energy production with co-digestion was estimated to be 0.42 kWh/m^3. This puts the plant above it energy by 111% (0.042 kWh/m^3 excess energy to the grid).

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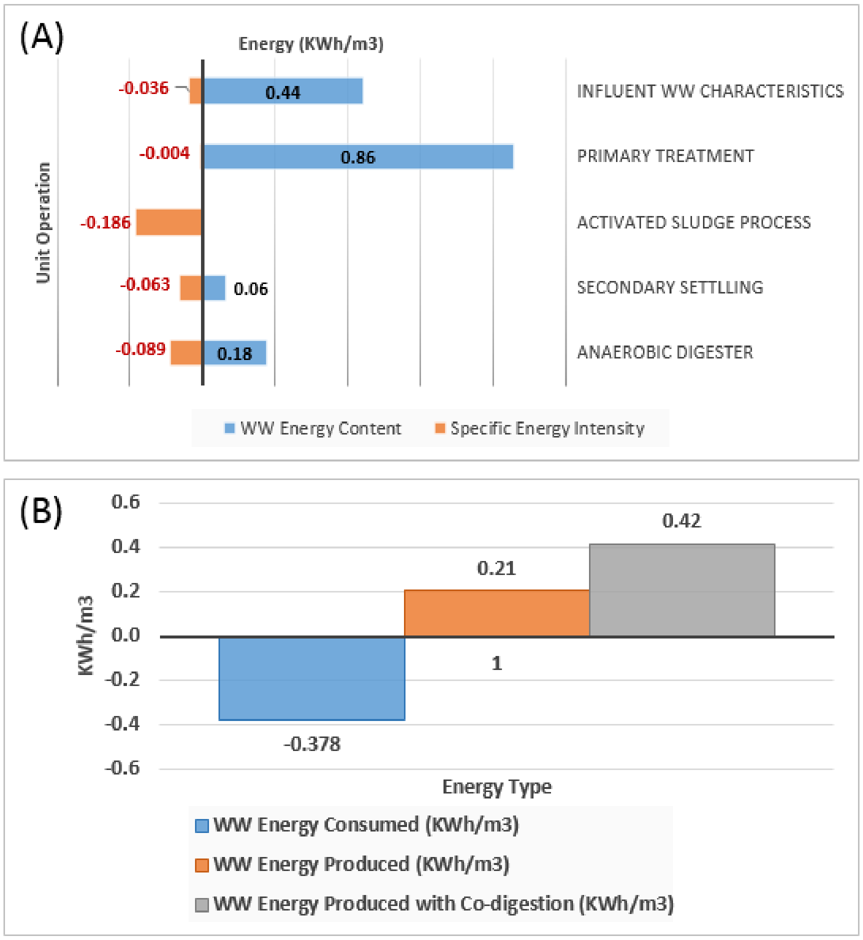

Figure 3.5 (A) Energy consumption and generation in different process units, and (B) energy balance of the process scheme 1

Combining high biodegradable waste such as FOG with wastewater sludge significantly improves energy production. Co-digestion is the best option for improving yields of the anaerobic digestion process, which improves biogas yields due to positive synergism established in the digestion medium and the supply of missing nutrients by the co-substrates (Mata-Alvarez et al. 2000; Edelmann et al. 2000; Mata-Alvarez et al 2011; Mata-Alvarez et al. 2014). Co-digestion of wastewater sludge with other organic wastes such as FOG and/or high-strength waste has received increasing attention over the years (Mata-Alvarez et al. 2000; Edelmann et al. 2000; Mata-Alvarez

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et al 2011; Mata-Alvarez et al. 2014). FOG under a mesophilic process has a high VSS destruction ratio (ranging from 70 to 80%) and potential biogas generation of up to 1.3 m^3/kg VSS destroyed, compared to a normal biosolids gas generation rate of 1 m^3/kg VSS destroyed (Johnson 2009). **Table 3.1** shows a comparison of the mass-balance analysis with actual plant data,

Table 3.1 Comparison of analysis output to actual plant data

Category	Unit	This Study	Gresham WWTP [1]	EBMUD WWTP [2]
Population	p.e.	135,000	125,000	685,000
Influent Flow	MGD	10	13	67
Influent COD	mg/L	500	518	~
FOG feed rate	MGD	~	0.10	0.60
Equipment upgrade	%	11	15	~
Energy consumed intensity	kWh/m^3	-0.378	-0.315	-0.408
Energy produced	kWh/m^3	0.42	0.385	0.55
Energy efficiency achieved	%	111	122	135

1 - Data obtained directly from Gresham Wastewater Treatment Plant
2. - Data directly obtained from East Bay Municipal Utility District

3.4.1.1 Case Study for Scheme 1 - Gresham WWTP

Gresham WWTP is located in Gresham, Oregon, serving a population of about 114,000 with an average daily wastewater flow of 13 MGD. The utility installed an AD in 1990 and started seeing problems with their 200 kW combustion engine after ten years due to untreated biogas. In 2005, Gresham addressed the problem by installing a 400 kW CHP CAT engine and biogas treatment system to remove siloxanes, hydrogen sulfide and moisture (Nora 2015). To improve on energy efficiency, equipment upgrades were implemented in 2010, including replacing the digester

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mixing equipment installed in 1990. Other equipment upgrades included installing a biogas mixing system with three 40-hp compressors and two linear motion mixers (one per digester) that require 5 hp per unit. Additionally, the city replaced two multi-stage blowers that supply air to the aeration basins with two turbo blowers. Fine bubble diffusers were also installed in the aeration basin.

These upgrades reduced the electricity consumption by 16 percent across the plant (Nora 2015; Gresham 2017). In 2012, the plant increased its biogas production by incorporating FOG as co-substrate for digestion. Prior to that, the city installed a 420 kW peak capacity ground-mounted solar energy system in 2009 that contributes approximately 5% of total energy produced. Biogas production increased from an average of 125 scfm before co-digestion to an average of 194 to 208 scfm — enough to operate two 400-kW CHP engines. Biogas contributes to about 95% of total energy production. Gresham has now achieved 122 percent energy efficiency (a net-positive 22 percent) (Nora 2015). It costs the district \$3.7 million to install the receiving and injection system for the supplemental waste unit. The utility receives a tipping fee of \$0.08/gal and the energy production saves the district about \$0.5 million per year (Nora 2015).

3.4.1.2 Challenges for implementing Scheme 1

Among all the benefits associated with integrating codigestion of mixed waste for energy production, there are a few challenges that have been reported which include upgrading existing facilities to incorporate the various waste sources, high variability of codigestion feedstock composition and volume, and digester overloading (Shen et al. 2015; Bond et al. 2012; Long et al. 2012; Zhang et al. 2014; USEPA 2006; Chapman and Krugel 2011; Ganidiet al., 2009; Kougias et al. 2014). The biggest challenge WRRFs encounter is the generation of excess sludge caused by digesting additional waste such as FOG with inconsistent characteristics. The additional sludge generated usually exceeds storage capacity which sometimes creates inventory issues.

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This means the thickened sludge has to be pumped back to the digesters to be managed especially during the winter months when it cannot be land spread. Other issues include high concentration of nitrogen in supplemental waste that presents challenges to meet permit limits, corrosion of pipping and tanks due to low pH, and finally, return waste stream from the AD with loaded FOG residue promotes growth of undesirable filamentous microorganism in the AS process, which causes effluent problems. Despite the risk of using FOG as a codigestate, its economic and energy benefits are attractive to WWTPs as discussed above.

3.4.2 Scheme 2: Nutrient removal in partial nitritation – anammox process and equipment upgrades

This scheme builds upon Scheme 1, with main focus on reducing aeration energy by increasing nutrient (such as N) removal (**Figure 3.6**). Anaerobic ammonium oxidation technology is added to the basic configuration layout to help minimize the energy required for aeration. Anaerobic ammonium oxidation process popularly known as Anammox has been proven to be the most effective way of reducing the oxygen demand of heterotrophic bacteria and significantly reduce energy requirement (Siegrist et al. 2008).

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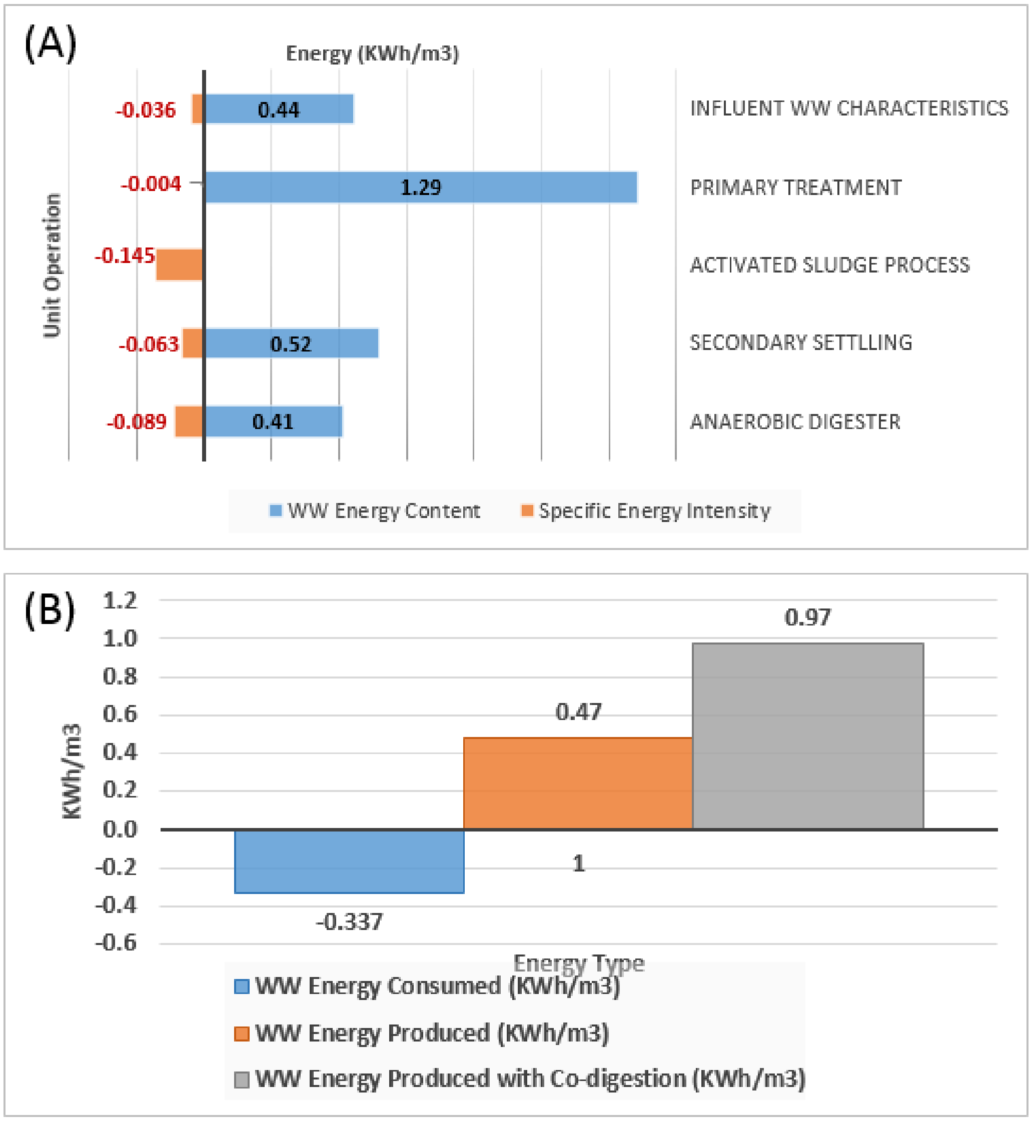

Figure 3.6 (A) Energy consumption and generation in different process units, and (B) energy balance of the process scheme 2.

The Anammox process uses nitrite as an electron acceptor and CO_2 as the energy source and was first reported previously (Mulder et al. 1995; WEF, Wett, 2007a; Wett 2007b; Siegrist et al., 2008). This technology was developed in Delft University of Technology and can reduce energy for biological treatment up to 63% (Siegrist, et al. 2008; Lackner et al. 2014). The Strass WWTP is widely known to be the first utility to implement the Anammox process on a full-scale.

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Other utilities reported to have adopted this technology besides Strass, includes Chesapeake Bay watershed, Alexandria Sanitation Authority WRF and Hampton Roads Sanitation District (HRSD)'s James River WWTP which recently upgraded to implement DEMON (Jin, et al. 2012). It is important to note that HRSD utility is the first full-scale anammox-based deammonification process in the US for side stream nitrogen removal (Nifong et al. 2013; Daigger 2011).

Energy consumption and generation profiles are shown in **Figure 3.6 (A)**. As shown in **Figure 3.6** (B), the total specific energy consumed was estimated to be 0.337 kWh/m^3; this represent a 10.8% (0.041 kWh/m^3) reduction of total energy intensity compared to the basic configuration. Also, aeration energy was reduced by 20% (0.145 kWh/m^3 compared to 0.182 kWh/m^3 for the basic configuration) by implementing the Anammox process. Hypothetically, to denitrify 1 gram of nitrite (NO^-_2 as N), it will require 1.7 g of COD. This means the remaining COD undergoes nitrification-denitrification.

Denitrification provides the opportunity to remove total nitrogen and by recovering energy. 1 kg of nitrate (NO_3^- - N) is denitrified when 2.86 kg of COD are oxidized; this reduces the amount of oxygen required for ammonia oxidation by almost 50%. According to Tchobanoglous et. al. (2003), ammonia oxidation requires a large amount of oxygen (~4.57 kg O_2/kg oxidized-N) (Tchobanoglous et al. 2003). In other words, 4.57-kWh electricity is lost with every kilogram of nitrate discharge or 2.86-kWh electricity is saved per kg (NO_3^- - N) (Tchobanoglous et al. 2003; Garrido et al. 2013). Scheme 2 shows 40% (7571 kg/day) removal of COD from the primary treatment tank; The Anammox technology reduces the COD entering the biological unit by 33% (equivalent to 3029 kg/day). A total of 43% (4437 kg/day) of the anaerobic digester sludge was converted to biogas. As mentioned above, the theoretical chemical energy obtained from

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converting 1 gram of COD to methane is 13.9 KJ. Hence, following similar steps discussed above; the estimated energy produced is 0.45 kWh/m^3. This alone makes the plant achieve 133% energy efficiency without co-digestion. The inclusion of the anammox technology makes it even more suitable for the implementation of co-digestion; because of the nutrient load cycle from the anaerobic digester. Thus, by adding FOG as a supplemental feedstock for co-digesting with primary and secondary sludge, a total energy production of 0.93 kWh/m^3 was estimated. This represent roughly 276% of energy efficiency. **Table 3.2** shows comparison of the output of this analysis to the Strass WWTP.

Table 3.2 Comparison of analysis with actual data from existing wastewater resource recovery plants

Category	Unit	This Studies	Strass WWTP
Population	p.e.	135,000	146000 [1]
Influent Flow	MGD	10	10 [2]
Influent COD	kg/d	500	463.3 [1,2]
FOG feed rate	MGD	~	~
Overall plant energy reduction	%	10.8	12 [3,4]
Energy consumed intensity [2]	kWh/m^3	-0.337	-2.076 [1]
Energy produced [2]	kWh/m^3	0.453[i]	2.243 [1]
Energy efficiency achieved	%	134	108

1 - Nowak et al. 2011; 2 - George 2010; 3 - Wett and Dengg 2010; 4 - Wett 2003
i- Energy produced excludes co-digestion; total energy produced with co-digestion is 0.93 KW/m3

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3.4.3 Scheme 3: Novel process with enhanced carbon capture and codigestion

The ultimate goal for a future WWTPs is to significantly reduce the net energy consumed and increase the net energy produced. This is due to the increasing concern over climate change and operating costs. **Figure 3.7** presents a recommended configuration for a future WRRF. For a WWTP to attain a net-positive energy status the following steps are recommended; (1) increase COD capture in primary treatment, (2) replace the aeration unit with a less energy demanding process, and (3) enhance energy production with codigestion. The future WRRF should focus on replacing the traditional primary settler and activated sludge process with an advanced primary treatment (APT) technology and a high rate microalgal pond (HRAP), respectively. These two unique unit operations will increase removal of biodegradable organics from the primary treatment to boost energy production and minimize or eliminate the need for aeration for biological oxidation and nitrification. In **Figure 3.7**, the APT technology follows a similar design as described by Gikas (2016). The APT uses a rotating fabric belt MicroScreen (pore size: 100-300 micro-meters), followed by a continuous backwash upflow media filter or cloth media filter (Aqua-Aerobic Systems Inc, 2014). This technology removes 60% of COD entering the wastewater treatment plant. The energy intensity for the APT technology is 0.034 kWh/day (**Table 3.3**) (Belinda and Stacey 2011).

The effluent from the APT goes to a HRAP, which substitutes the activated sludge process. In a wastewater microalgae cultivation, microalgae develop a synergistic effect with aerobic heterotrophs and autotrophs through exchange of organic substrates. During this interaction, microalgae generates O_2 that is required by heterotrophic bacteria to oxidize the substrates in the wastewater. While in the process of the substrate utilization or oxidation, the heterotrophic bacteria release CO_2, which is in turn used by microalgae growth (Alessandro and Joan 2019; Enrica et al.

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2018; Larissa et al. 2019). Replacing the activated sludge with HRAP provides several benefits such as low energy consumption and high energy efficiency. COD, nutrients and energy balances ae shown in **Figure 3.7** and **Table 3.3**.

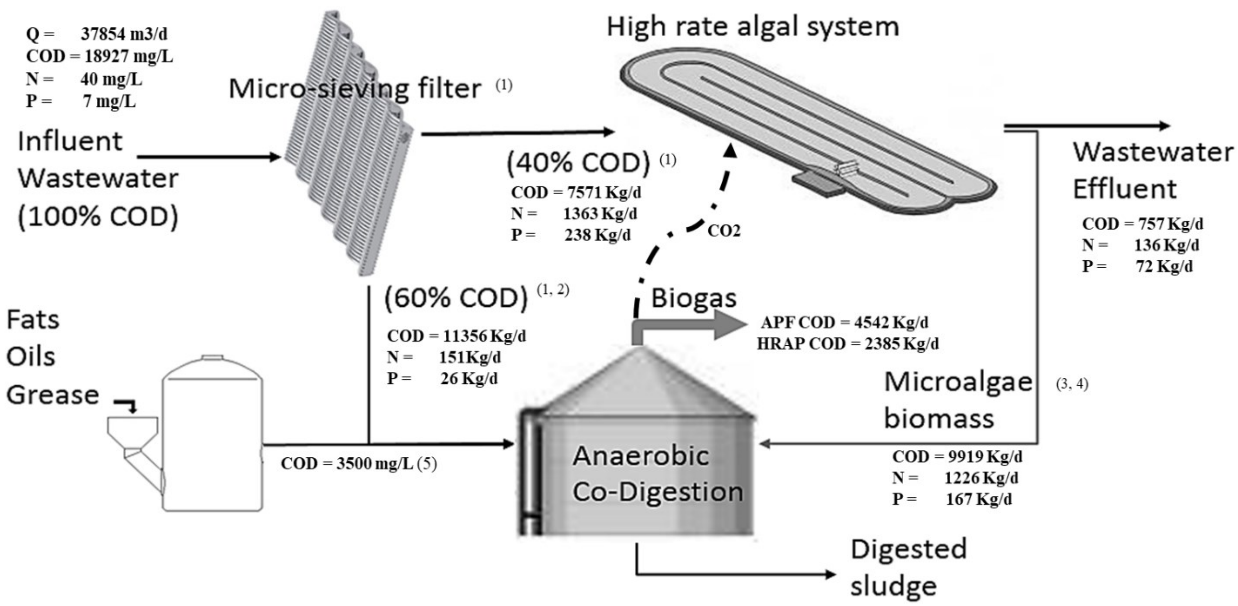

Figure 3.7 **Scheme 3** - COD, nutrient and energy mass balance; primary sedimentation replaced with advance primary filtration to increase COD removal and CAS replaced with HRAP (1- Gikas 2016; 2- Tchobanoglous et al., 2003; 3- Belinda and Stacey 2011; 4- Posadas et al., 2013; 5- 5- John et al., 2009)

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Table 3.3 Energy scenario analysis for the proposed future WRRF

Unit Process	Electrical Energy (kWh/m^3)
Influent Equipment	-0.036 [1]
Advance Primary Filtration	-0.01 [2]
HRAP	-0.007 [3]
Thickening/Dewatering	-0.011 [3]
Pumps	-0.036 [3]
Anaerobic Digester	-0.089 [1]
Total Energy Consumed	-0.188
Energy Produced (Co-digestion of Sewage-Algae)	1.02
Energy Produced (Co-digestion of FOG-Sewage-Algae)	1.87
Energy Balance	1.682

1 - Goldstein and Smith, 2002; 2 - Gikas 2016; 3 - Belinda and Stacey 2011

The HRAP system has attracted much attention recently because it is known to be low-energy, nutrient removal and energy production technology (Craggs et al., 2014). The total specific energy intensity for the configuration was estimated to be 0.188 kWh/m^3; almost 50% less compared to energy required for CAS. The hypothetical energy obtained from one gram of COD converted to methane is approximately 20 kJ/g (for co-digestion of sewage and microalgae) (Taira et al., 2017). According to **Table 3**, the maximum recoverable electrical energy for co-digestion of sewage and microalgae was estimated as (20 kJ/g /3,600 kJ/kWh) × 183 g COD/m^3 = 1.02 kWh/m^3. This represents roughly 5.4 times the energy required for treatment. By adding supplemental waste such as FOG for co-digestion, the estimated electrical energy recovery was 1.98 kWh/m^3. Hence, such significant increase in energy production presents opportunity to explore water reuse option whereby tertiary treatment such as membrane technology could be employed.

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3.5 Conclusions

The quantitative mass and energy balance analysis presented in this paper discussed three different schemes by which current WWTP facilities could become energy-neutral or energy-positive in their operations. The existing utilities can become energy self-sufficient by conserving energy and by producing additional biogas. The biological process (i.e., aeration unit) is the main energy consumer and minimizing energy consumption of the aeration unit is the key. Conventional method of removing nutrients from wastewater is an energy-intensive process. This can be better managed by adopting novel nitrogen removal techniques such as the one discussed in scheme 2. Finally, replacing the activated sludge process with a low energy demanding technology such as HRAP can transform a WWTP into an energy-yielding process.

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CHAPTER IV

ENERGY AUTARKY OF SMALL SCALE WASTEWATER TREATMENT PLANTS BY ENHANCED CARBON CAPTURE AND CODIGESTION - A QUANTITATIVE ANALYSIS

4.1 Abstract

Municipal wastewater treatment plants (WWTPs) can achieve energy self-sufficiency (autarky) while complying with increasing discharge standards. We evaluated three energy-positive wastewater treatment scenarios classified as “Basic”, “Moderate”, and “Advanced” configurations. A quantitative mass and energy balance model was developed to analyze the energy recovery potentials of these configurations. Enhanced COD capture, codigestion with locally available biodegradable wastes, and aeration energy optimization were considered as the main approaches to achieve an energy autarky or energy-positive status. Data from existing operating plants were used to validate the model performance. This chapter presents a detailed quantitative analysis of two (basic and moderate) energy-neutral or energy-positive wastewater treatment configurations. In addition, a novel and practically feasible energy-positive wastewater treatment scheme incorporating advanced solids separation is presented with energy analysis and a case study. This model can be useful to quickly assess the energy recovery potential of small scale wastewater treatment systems.

Keywords: codigestion; energy recovery; modeling; sludge; sustainability; wastewater treatment

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4.2 Introduction

Most of the wastewater treatment plants utilize aerobic biological processes to treat wastewater, which convert organic matter into biomass and carbon dioxide (CO_2). While this process is inherently energy-intensive due to many process and microbiological resource related limitations, increasing restrictive standards make the process even more energy demanding (Fraia et al 2018). According to an energy audit report, aeration energy is the largest energy consumer in conventional activated sludge (CAS) WWTPs amounting to 45-75% of the energy costs (Olsson and Carlsson 2013, EPRI 2002, Rosso and Stenstrom 2008). Wastewater treatment alone accounts for 1-4% of total electricity budget in many developed countries (Longo et al. 2016). Due to increasing concerns over global warming and climate change, associated with conventional energy sources, minimizing the energy needs for wastewater treatment has become an important priority at global levels (Mainnina et al 2016).

4.2.1 Energy conservation in wastewater treatment plants

Many process modifications have been implemented to minimize energy consumption in WWTPs as shown in **Table 4.1**. These include replacement of coarse bubble aerators with fine pore diffusers, installing dynamic flow control on-line sensors, variable speed flow devices, high-efficiency blowers, pumps and lighting in the WWTP facilities (Rosso et al. 2008, Rosso and Stenstrom 2012). Other process related improvements include adjusting the sludge retention time to lower aeration energy requirements while generating higher biomass for biogas production (Rosso and Stenstrom 2012, Shi 2011).

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Table 4.1 Suggested Measures to Minimize Energy Consumption in WWTPs

Suggested Upgrades for Basic Technology	% Suggested Reduction	Notes	Reference
Replacing aeration diffusors with fine pore diffusing system	45		(Rosso and Stenstrom 2012)
Applying dynamic control of on-line sensors	30		(Pakenas 1995)
Infield filter for aeration blowers	10	Reducing aeration energy consumption	(Montieth et al 2007)
Install high speed motor for blowers	5		(Wet B 2007a, 2007b (Jonasson 2007)
Replacement of low efficient WAS pumps with more efficient motors with VFD	74		
Replacement of low efficient RAS pumps with more efficient motors with VFD	80	Increasing energy efficiency	(EBMUD) (Pakenas 1995) (Jonasson 2007)
Replacing old effluent pumps with more efficient ones with VFD	50		
Lighting: replacing T-12 lamps with T-8 technology	30	Reducing energy consumption	(Pakenas 1995)

WAS – Waste Activated Sludge; VFD – Variable Frequency Drive; RAS – Return Activated Sludge

4.2.2 Energy losses in wastewater treatment plants

Although wastewater treatment is energy-intensive, from a thermodynamic standpoint, the organic content in wastewater is considered an energy-source, not an energy-sink. The organic compounds in the wastewater contain energy embedded within their chemical bonds (Frijns 2013). However, it is quite challenging to extract this embedded energy and convert it into a useful form of energy such as electricity or thermal energy. If not extracted, most of this energy will be lost through process inefficiencies. For instance, in the anaerobic treatment, around 8% of embedded energy is used for generating energy to break down the high energy organic compounds (larger molecules) into lower energy organic compounds, CH_4. Another 7% of this energy is used for cell synthesis in addition to wastewater treatment process inefficiencies (McCarty 2011). Around 35% of the methane (CH_4) from the anaerobic digester may be converted into power (electricity) and 65% as thermal energy (heat) through the use of cogeneration units, subject to further inefficiencies

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(McCarty 2011). For example, internal combustion units (for cogeneration) are widely used in the wastewater treatment industry for energy recovery. These engines convert about 30 – 45% of available energy into electricity as mentioned above (Garrido et al 2013). Other technologies such as microbial fuel cells and fuel cells can be used to recover energy from wastewater (Logan 2005). Fuel cells convert about 36 - 45% of available energy to electricity; however, their electrical efficiencies are lower and the unit itself is very expensive (Brown and Caldwel 2010). Microbial fuel cells, on the other hand, have some drawbacks such as low COD removal; inconsistent power density; and expensive electrode materials that make this technology infeasible for full-scale applications (McCarty 2011, Logan 2005, Gude 2016, Oon et al 2016). However, some important advances have been reported in this area in recent studies using microalgae and anammox bacteria in energy producing microbial desalination process configurations [Gude 2018, Ghimire and Gude 2019, Kokabian et al 2018a and 2019b).

4.2.3 Energy recovery possibilities in wastewater treatment plants

When evaluating the energy recovery possibility of a WWTP, both energy sinks and potential energy sources should be considered. Wastewater sludge (on a dry basis) contains approximately 60% organic compounds; that consist of 50-55% organic carbon and mostly biodegradable (in the form of bCOD), 10-15% is nitrogen (as N) and 1-3% is phosphorus (as P) (Zanoni and Mueller 1982, Gude 2015a, Le et al. 2019, Xinrui et al 2019). The energy in the nutritional components of the wastewater such as N and P is approximately 0.79 kWh/m^3 (Chae and Kang 2013). The energy contained in wastewater solids is 2.72 kWh/m^3 (Gude 2015b, Jos et al 2013). Detailed assessment of energy content for raw municipal wastewater has been reported elsewhere (Zanoni and Mueller 1982, Shizas and Bagley 2004, Heidrich and Curtis 2011). Energy content embedded in sludge varies; for instance, primary, secondary and digested sludge contain

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15–22.8 kJ/g, 12.4–16.1 kJ/g, and 11 kJ/g of energy, respectively (Zanoni and Mueller 1982, Shizas and Bagley 2004, Heidrich and Curtis 2011). Shizas and Bagley reported that about 66% of energy content entering the WWTP is captured in the primary sludge, 42% of the remaining energy from the primary effluent is retained in the secondary sludge, and biogas contains 47% of the energy entering the WWTP (Shizas and Bagley 2004). This information is critical for identifying ways to convert the embodied energy content in wastewater into a valuable resource, and to determine the efficiency of energy recovery from municipal wastewater streams. Thus, by recovering the additional energy contained in wastewater, the treatment process can be converted into a net-energy producer (Logan 2005).

Once developed for sludge management/stabilization as the main goal, anaerobic digesters are now increasingly being considered as a critical component of energy recovery schemes in WWTPs. Up to 35% of energy content in wastewater entering the WWTPs can be converted or recovered as biogas in anaerobic digesters. In optimized and high efficiency process schemes including a cogeneration unit, up to 80% of energy requirements for wastewater treatment can be supplied through biogas production (Silvestre et al 2015). To improve the energy sustainability of WWTP operations, renewable energy integration such as geothermal energy sources and solar energy sources has also been considered (Di et al 2019, Najafi et al2019). Biofuel production from microalgae grown in wastewater is also considered to achieve energy sustainability of wastewater treatment systems (Kadri et al 2018, Blair et al 2014, Otondo et al 2018).

4.2.4 Energy performance analysis in wastewater treatment plants

Energy performance evaluation depends on many factors and assumptions (Fraia et al 2018, Mojtaba et al 2018). Energy audits are usually conducted to identify inefficiencies in WWTPs. These audits are useful in assessing energy consumption trends and in identifying energy

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conservation measures (Mojtaba et al 2018). Over the years, a large number of benchmarking tools have been developed to evaluate the energy efficiency of WWTPs (Lindtner et al 2004, Maria et al 2018). Energy benchmarking is used to optimize WWTP operations, which is aimed to reduce operation costs and GHG emissions. Difficulties in comparing energy performance of different WWTPs is addressed by developing common Key Performance Indicators (KPIs) (Lorenzo et al 2015, May et al 2015, Benedetti et al 2008). Electrical energy consumption has been used in other studies as the Key Parameter Indicator (KPI) for determining the specific energy intensity (SEI) (kWh/m^3 treated wastewater) of a WWTP (Otondo et al 2018, Belloir et al 2008, Longo et al 2016). Other bench marking tools have been developed to help compare energy performance among WWTPs even for different wastewater configurations and operational modes (Lindtner et al 2004, Maria et al 2018). Common parameters such as influent and effluent wastewater characteristics (both quality and quantity), plant size and operations, and pollutant loading have been used to evaluate energy performance of WWTPs (Belloir et al 2008). Several methodologies have also been developed to assess the energy saving potentials in WWTPs. For instance, Longo et al. 2019, proposed a standard method for assessing and improving the energy efficiency of WWTPs. This method delivers an aggregated measure of WWTP's energy efficiency and uses a single universal energy label for WWTP's energy status (Longo et al 2019). Theoretical mass balance analysis comparing different scenarios of WWTP energy performance has been presented by different authors (Garrido et al 2013, Svardal and Kroiss 2011, Qunli et al 2017).

4.2.5 Proposed research methodology

The quantitative method presented in this study is the first of its kind developed for energy performance evaluation of a WWTP. The present study seeks to answer the following questions: 1) what can the existing WWTPs do right now to become a resource recovery facility without

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major infrastructural changes?; 2) How would the integration of new technologies for nutrient removal affect the energy balance of existing WWTPs?; and 3) which type of treatment scheme should be considered for future or new WWTPs?

Current biological processes for wastewater treatment are energy-intensive (Liu and Gu 2018). The pathway towards energy self-sufficient operation of biological processes is to maximize energy recovery, while minimizing energy consumption (Zhang et al 2019). With this ideology, many wastewater treatment plants are now being transformed into energy producing facilities by incorporating anaerobic digesters (Gude 2015a, Chen and Chen 2013, Mirmasoumi et al 2018). In this research, we have explored energy-positive treatment schemes for small scale wastewater treatment plants (flow capacity of 5 MGD or 18,925 m^3/d) and classified them: "basic- *Scenario 1*"; "moderate – *Scenario 2*" and "advanced – *Scenario 3*" configurations. The basic configuration involves enhanced energy recovery through supplemental biogas production via codigestion and equipment upgrades to minimize energy consumption. The term "equipment upgrade" refers to suggested measures listed in **Table 4.1**. The moderate configuration employs anammox denitrification process followed by CAS to reduce energy consumption and then codigestion, while the advanced configuration includes enhanced separation of solids in the primary settling unit followed by biological filtration for further treatment. A simple quantitative mass and energy balance concept was developed to analyze a 5-MGD capacity WTTP with varying influent COD concentrations. The effect of different COD to N ratios were also studied. The potential contributions of equipment upgrades, codigestion with highly biodegradable FOG feedstock and enhanced sludge recovery (i.e., carbon capture) in primary treatment were evaluated. Different alternatives for each configuration are presented.

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4.3 Methods

The treatment train considered for the analysis consists of a primary settling unit (PS), a biological treatment process (i.e., Conventional Activated Sludge process – CAS), a secondary sedimentation unit (SS); and an anaerobic sludge digester (AD) (**Fig. 4.1)**. Mass balances for COD and nutrients are analyzed quantitatively as shown in **Table 4.2**. Equations 1 through 14 show the different mass balance equations used in this study. All assumptions considered in this analysis are shown in **Table 4.3**. COD was used as a reference value to calculate the amount of organics in wastewater and the potential energy requirements and energy recovery in different wastewater treatment configurations. Influent concentrations such as COD and N were varied.

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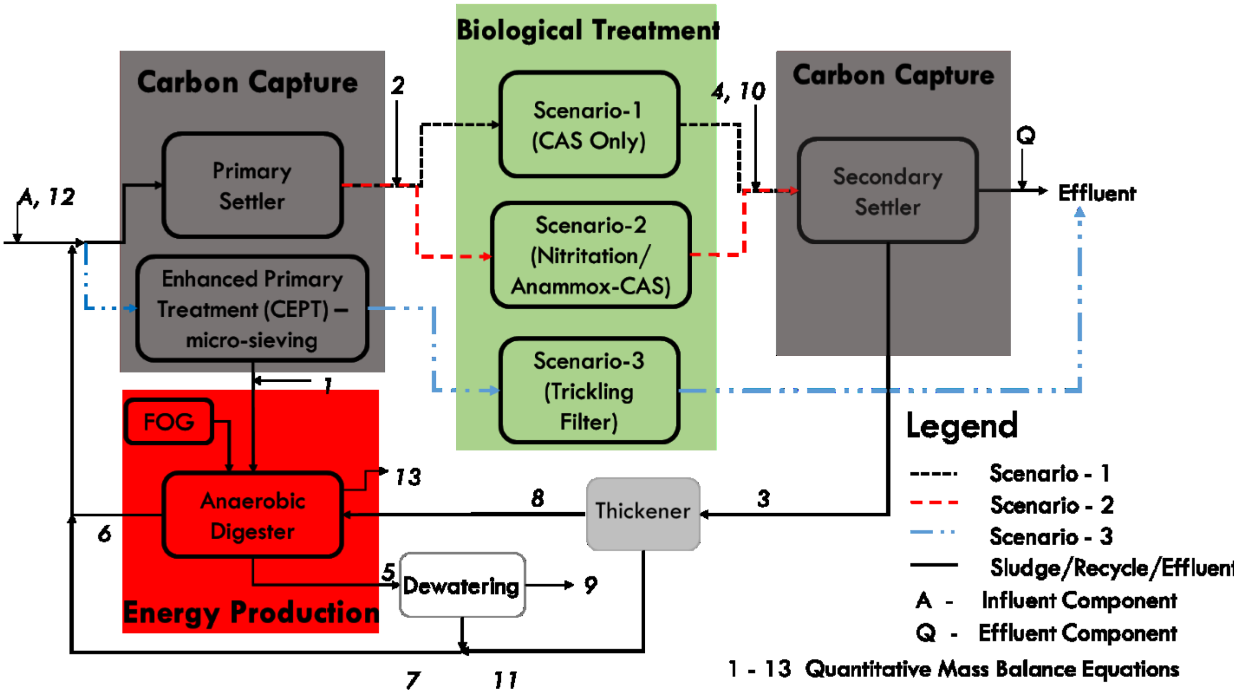

Figure 4.1 A generic scheme showing three different configurations of energy self-sufficient or energy-positive wastewater treatment plant, also known as Water and Resource Recovery facility: Scenario 1 – CAS with equipment upgrades and codigestion with FOG feedstock; Scenario 2 – CAS preceded by partial nitritation-anammox step and codigestion with FOG feedstock; Scenario 3 – enhanced separation of organic solids followed by carbon and nutrient removal in biological filters and codigestion with FOG feedstock. Enhanced primary treatment cab ne achieved by micro-sieving unit.

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Table 4.2 Summary of quantitative mass balance equations

Process	Symbol	Formulation	Equation#
Sludge Removed from Primary Sedimentation (PS) tank	E	$\dfrac{A-\alpha a-f_N a+\dfrac{f_R Q}{(1-f_R)}-\dfrac{f_G Q}{(1-f_R)}}{\left(\dfrac{1}{f_E}-\alpha\theta-f_N-f_G\beta\right)}$	(4.1)
where	α	$f_P(1-f_N-f_J)(1+f_S)$	
	β	$\dfrac{(1-f_E)(1-f_X)}{f_E}$	
	a	$\dfrac{Q(1-f_G)}{(1-f_R)}$	
	θ	$1+\beta(1-f_G)$	
PS tank effluent wastewater	B	$\dfrac{E(1-f_E)}{f_E}$	(4.2)
Sludge Removed from Secondary Sedimentation (SS) tank	F	$\dfrac{E(1-f_R)(1-f_E)}{f_E}-\dfrac{Q}{(1-f_R)}$	(4.3)
SS effluent wastewater	C	$\dfrac{Q}{(1-f_r)}$	(4.4)
Digestate from Anaerobic Digester (AD) to dewatering tank	K	$(E+H)(1-f_N-f_J)$	(4.5)
Recycle from AD to PS	N	$f_N(E+H)$	(4.6)
Recycle from Thickener back to PS	G	$f_G F$	(4.7)
Thickener underflow to AD	H	$F(1-f_G)$	(4.8)
Discharge from dewatering process to land application	L	$K(1+f_S)(1-f_p)$	(4.9)
Amount of COD synthesized by bacteria	X	$f_X B$	(4.10)
Dewatering recycle	P	$f_p(E+H)(1-f_N-f_J)(1+f_s)$	(4.11)
Total recycle back to PS	M	$\dfrac{E}{f_E}-(A+G)$	(4.12)

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Table 4.2 (continued)

Process	Symbol	Formulation	Equation#
Solids in AD converted gas	J	$f_J(E+H)$	(4.13)
Recycle from tertiary treatment	R	$\frac{f_R Q}{(1-f_R)}$	(4.14)

where f_R (tertiary recycle ratio) and f_S (conditioning chemicals ratio) are set to zero because the model assumed no tertiary treatment and no addition of chemicals to the anaerobic digester; f_E (Primary settler solids separation ratio) varies from 0.23 to 0.79; f_P (Dewatering recycle ratio) = 0.04; f_G (Thickener recycle ratio) = 0.03; f_J (Thickener solids removal ratio) = 0.53; and f_X (ratio of B destroyed) = 0.51.

4.3.1 Mathematical model

The mathematical procedure used to estimate loading rates of individual components in different wastewater unit processes was first developed by the US EPA. This procedure is based on material balance approach of a component. The removal efficiency of each unit operation is expressed as a fraction of substrate removed in each stage as explained at the end of **Table 4.2**, which represents the quantity of solids removed in each stage. Process equations listed in **Table 4.2** are derived by substituting the individual mass balance equations for unit processes. **Fig. 4.1** and **Table 4.2** show the complete methodology used for the quantitative mass and energy balance analysis. The mass balance procedure was then used to evaluate two different process configurations: *Scenario 1* and *Scenario – 2*. *Scenario – 1* follows the conventional wastewater CAS process configuration, where wastewater treatment train consists of primary treatment, activated sludge, secondary treatment and sludge management including anaerobic digestion. To enhance energy production, a fixed rate of supplemental feedstock such as Fat-Oil-Grease (FOG) was considered in a codigestion with sewage sludge scheme. The treatment train of *Scenario – 2* is similar to *Scenario -1*, however autotrophic nitrogen removal (a combined system of partial ammonia oxidation to nitrite and anammox denitrification) process was adopted to reduce energy

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consumption. Finally, a third scenario called advanced configuration includes enhanced solids separation by micro-sieving unit followed by biological filter system, discussed with a case study. A summary of assumptions made for this analysis is presented in **Table 4.3**.

Table 4.3 Assumptions used in the quantitative mass balance analysis of this study.

Process	Unit	Assumptions	Reference
Influent Wastewater			
Flow	MGD	5	This study
COD	mg COD/L	100 - 800	
Total Nitrogen (as N)	mg N/L	8 - 70.2	
CAS			
O_2 transfer	kg O_2/kWh	1	(Tchobanoglous et al 2013)
Biomass yield	g VSS/g $CODbs_{removed}$	0.5	(Tchobanoglous et al 2013)
Anaerobic Sludge Digester			
Heating Value	kJ/m^3	22,400	(Tchobanoglous et al 2013)
Biomass yield	g VSS/g $COD_{removed}$	0.08	(Tchobanoglous et al 2013)
CH_4 energy content	kJ/g CH_4	50.1	(Tchobanoglous et al 2013)
CHP			
Electricity recovery	kWh/SCFM	15	This study
Power to Heat Ratio	Ratio	0.6	
Codigestion			
Biogas yield of FOG	m^3/wet ton	970	(Lemoine et al 2006, EPA 1975)

Because the dynamics of COD fraction and its interactions in a CAS process are very critical for COD and energy balance analyses, influent raw municipal wastewater COD was divided into four fractions as suggested previously (Tchobanoglous et al 2013). COD fractionation

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profile incorporated in the mass balance analysis is shown in **Fig. 4.2**. The influent fractions assumed for this analysis were (% of total COD): soluble biodegradable (S_s = 0.15); particulate biodegradable (X_s = 0.7); inert soluble (S_i = 0.05); and inert particulate (X_i = 0.1). S_s fraction is the most easily available substrate for heterotrophic microorganisms. Hence, its quantity can be a determining factor for anaerobic and anoxic reactor volume design because phosphate release and denitrification processes are sensitive to easily accessible substrate fraction (Henze et al 2008). X_s fraction has high molecular weight, colloidal, and particulate organic substrate that must undergo external hydrolysis before it can be available for further degradation. In addition, X_s has the highest oxygen demand, hence it greatly influences the aeration requirements in the aeration tank and may be partly oxidized in the biological process or converted to methane in the anaerobic digester (Garrido et al 2013, Henze et al 2008). S_i, on the other hand, cannot be degraded in the CAS process, nor can it be separated by physical processes (Garrido et al 2013). X_i is partially separated in the primary sludge and finally removed in the secondary sludge (it can only be removed by clarification). This variable determines the sludge treatment capacity for the CAS process.

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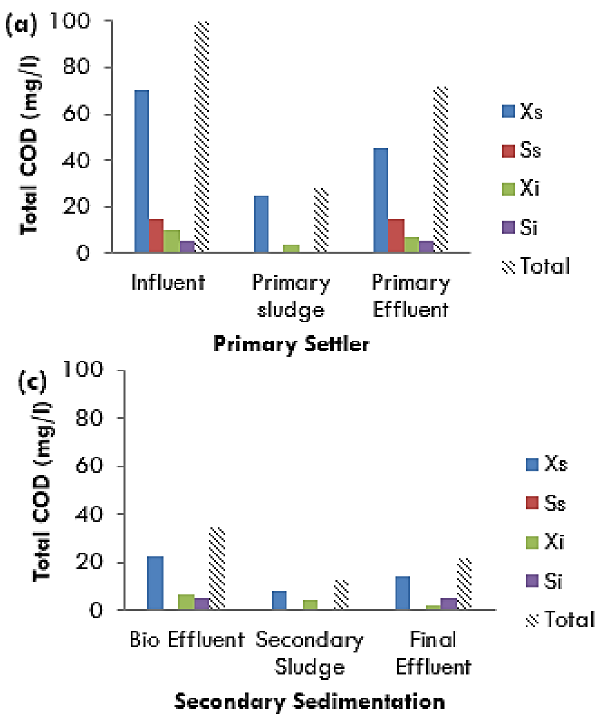

Figure 4.2 COD fractionation profile for CAS mass balance analysis.

Equations 4.15 - 4.19 were used to approximately estimate the energy intensity for the three different configurations with varying influent COD and N concentrations (and flowrate);

$$TEE_R = EE_P - \left(E_{O2-p} - E_{O2-C}\right) \tag{4.15}$$

$$EE_P = 0.6938 \times Q^{-0.132} \tag{4.16}$$

$$E_{O2-C} = \left(\frac{O_{2-R}}{Q \times O_{2-T}}\right)\left(\frac{MGD}{3785.4m^3d^{-1}}\right) \tag{4.17}$$

$$O_{2-R} = \left(Q(X) - 1.42P_{X.bio} + 4.33Q(NH_4\text{-}N)\right)(3785\frac{m^3d^{-1}}{MGD}) \tag{4.18}$$

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$$P_{X.bio} = Y_{obs}Q(X) \tag{4.19}$$

where TEE_R (kWh/m^3) is the total electrical energy required for a CAS WWTP; EE_P (kWh/m^3) represents expected electrical energy (practical) based on Equation 4.16 (EPRI 2002). $E_{O2\text{-}P}$ (kWh/m^3) is the aeration energy (practical) which is given as 44% of EE_P, adopted from EPRI report (EPRI 2002) while $E_{O2\text{-}C}$ (kWh/m^3) is the calculated aeration energy based on Equation 4.17. This equation accounts for the variations in the influent wastewater characteristics (such as the ratio of COD and N), to calculate actual energy consumption. Q is the wastewater flowrate (MGD). Data used to formulate Equation 16 was adopted from EPRI (EPRI 2002). Equation 18 and 19 were adopted from Metcalf and Eddy 2003 (Tchobanoglous et al 2013). $O_{2\text{-}R}$ (kg O_2/d) is the total oxygen required for oxidation and nitrification; $O_{2\text{-}T}$ (kg O_2/kWh) is oxygen transfer given in Table 3; X is the amount of biodegradable COD synthesized by biomass (g COD/m^3) as given in Table 2, Eq. 4.10. $P_{X.bio}$ is net daily waste activated sludge produced (kg/d); and Y_{obs} is the biomass yield given in **Table 4.3**. Thus, this was used to calculate the energy recovery status of all scenarios.

The cost of electrical energy production was presented as Levelized Electricity Cost per unit of Energy production (LEC, Eq. 20).

$$LEC = \frac{\sum_{t=1}^{25}\left(\frac{\left(ACC \times Q3650 \times (PA,\ i\%,\ t)\right) + (OMC \times Q3650)}{(1+i)^t}\right)}{\sum_{t=1}^{25}\left(\frac{(Electricity\ Produced)_t}{(1+i)^t}\right)} \tag{4.20}$$

where t (yr) is time, ACC (\$/yr) is the amortized capital cost for AD biogas production (assumed 1.1\$/yr), Q (MGD) is flow, and OMC (\$/yr) is the operation and maintenance costs for AD biogas production (assumed to be 0.38\$/yr) (1975). Equation 4.15 is a rough approximation

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normalized cost to evaluate the sensitivity of energy production form sludge digestion. According to Eq. 4.15, the discounted stream of annual cost for electrical energy over the life of the installed capital, which is assumed to be 25 years for this analysis, was divided by the discounted stream of electrical energy produced over the same time period. The analysis also assumed an inflation-adjusted discount rate of 6% (i = 0.06) for both cases.

4.4 Model validation

To check the validity and accuracy of the equations and assumptions, the model output results were compared with real case or actual plant operational data (**Table 4.4**). To perform this validity check analysis, the input data of two operating plants' data were fed to the model for easy comparison of output data and the results are shown in **Table 4.4**. The root mean absolute error between the observed and predicted values for the two scenarios were about 20%. This error is probably due to other "real-case" operational factors not considered in the model. However, it should be noted that combined assumptions and mathematical expressions developed for this work are reasonably acceptable and have some practical relevance.

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Table 4.4 Model Validation of Quantitative Mass Balance Analysis

Category	Unit	Scenario 1 - Basic **Gresham WWTP** [1]	**This Study**	Scenario 2 - Moderate **Strass WWTP**	**This Study**
Input					
Influent Flow	MGD	13	13	10 [a]	10
COD concentration	mg/L	518	518	463.3 [a]	500
FOG feed rate	tpd [2]	0.84	0.84	~	~
Equipment upgrade	%	15	15	12 [b, c]	12
Output					
Energy consumed intensity	kWh/m^3	-0.315	-0.36	-0.207 [b, c,d]	-0.25
Energy produced	kWh/m^3	0.385	0.3	0.224 [a]	0.273
Energy balance	kWh/m^3	0.07	-0.06	0.017	0.023
Root mean absolute error (RMAE) [3]					
Energy consumed intensity		0.14		0.17	
Energy produced		0.22		0.18	

a – Henze et al 2008, b – George and Crawford 2010, c – Wet B 2003, d - Wet B 2006
1- Data obtained directly from Gresham Wastewater Treatment Plant; 2 - Unit is tons per day;
3 - RMAE = |(Predicted - Observed)/(Observed)|

4.5 Results

4.5.1 Scenario 1a – Basic Configuration (COD removal – Supplemental biogas production) with varying N:COD ratio

The configuration for this scenario represents a traditional CAS WWTP with the process train shown in **Fig. 4.3**. CAS in this case represents COD removal but excludes nitrogen removal process. The main components of *Scenario 1* are codigestion of sewage sludge and high biodegradable supplemental feedstock such as FOG; and replacing old process equipment with high energy-efficient units. **Fig. 4.3** shows an example of a 5 MGD basic wastewater treatment configuration, with an influent COD and N concentrations of 400 mg/L and 35 mg/L respectively. This analysis assumes energy savings of 11% for equipment upgrades to improve energy efficiency

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and a fixed FOG feed rate of 1 ton per day for codigestion. Such a configuration will require (E_r) 8096 kWh/d (0.428 kWh/m^3) of electrical energy and achieves only 41% of energy self-efficiency or energy autarky without (WAS only) the use of supplemental feedstock. The energy efficiency increases to roughly 70% when codigestion of FOG and WAS is employed (WAS + FOG). Based on the calculated available energy (E_c) contained in the wastewater organic matter, the energy produced without codigestion is one fourth (25%) of the available energy. considering this limitation, we evaluated different options that would change the status of this 5-MGD WWTP to energy-neutral or energy-positive.

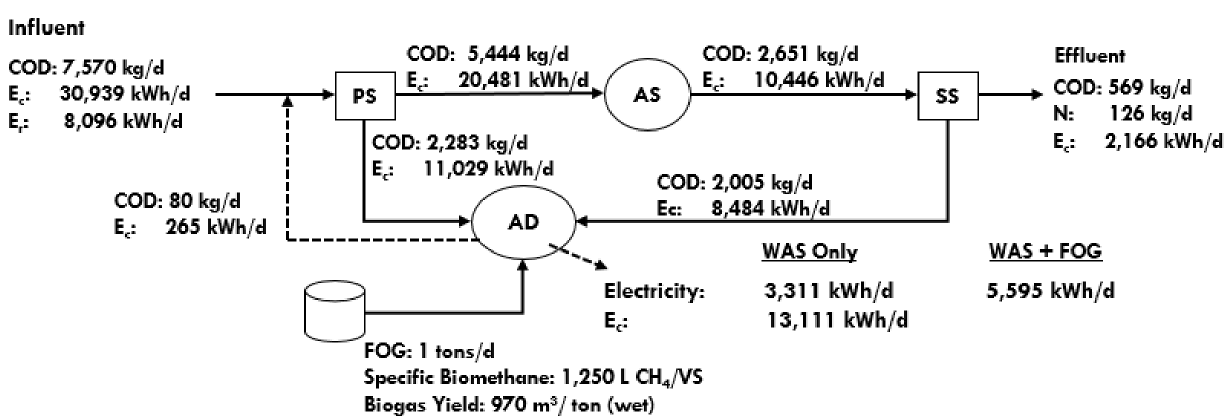

Figure 4.3 Mass and energy balance analysis for *Scenario 1* (basic configuration)

The largest energy consumer in this *Scenario* is the amount of oxygen required for carbon oxidation and nitrification, which represents 44% of the total energy requirements. **Fig. 4.4a** and **Fig. 4.4b** show the effect of varying N (from 8 to 70 mg/L) concentrations on energy consumption and production at a fixed COD concentration of 450 mg/L while **Fig. 4.4c** and **Fig. 4.4d** show the effect of COD (from 100 to 800 mg/L) concentration on energy consumption and production at a fixed N concentration. Nowak et al. 2011, reported that energy self-sufficiency is feasible for an

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influent N to COD ratio less than 0.1 (N:COD) with increased primary treatment efficiency, provided the aeration energy is significantly minimized (Nowak et al. 2011). **Fig. 4.4a** shows that energy efficiency increases as the influent N to COD ratio decreases. The basic configuration achieves a maximum energy efficiency of 43% at 0.02 (N:COD); this is because of the increase aeration energy demand for nitrification. Much higher energy efficiency (~67%) is obtained with a N:COD ratio of 0.05 when N is fixed and COD varies (**Fig. 4.4c**). Both scenarios for **Fig. 4.4a** and **Fig. 4.4c** correlate with Nowak's trend except that in this case primary sludge removal was fixed at 30% and there is no nitrogen removal. Increasing sludge removal efficiency in primary treatment unit as suggested by Nowak could move the basic configuration closer to energy autarky.

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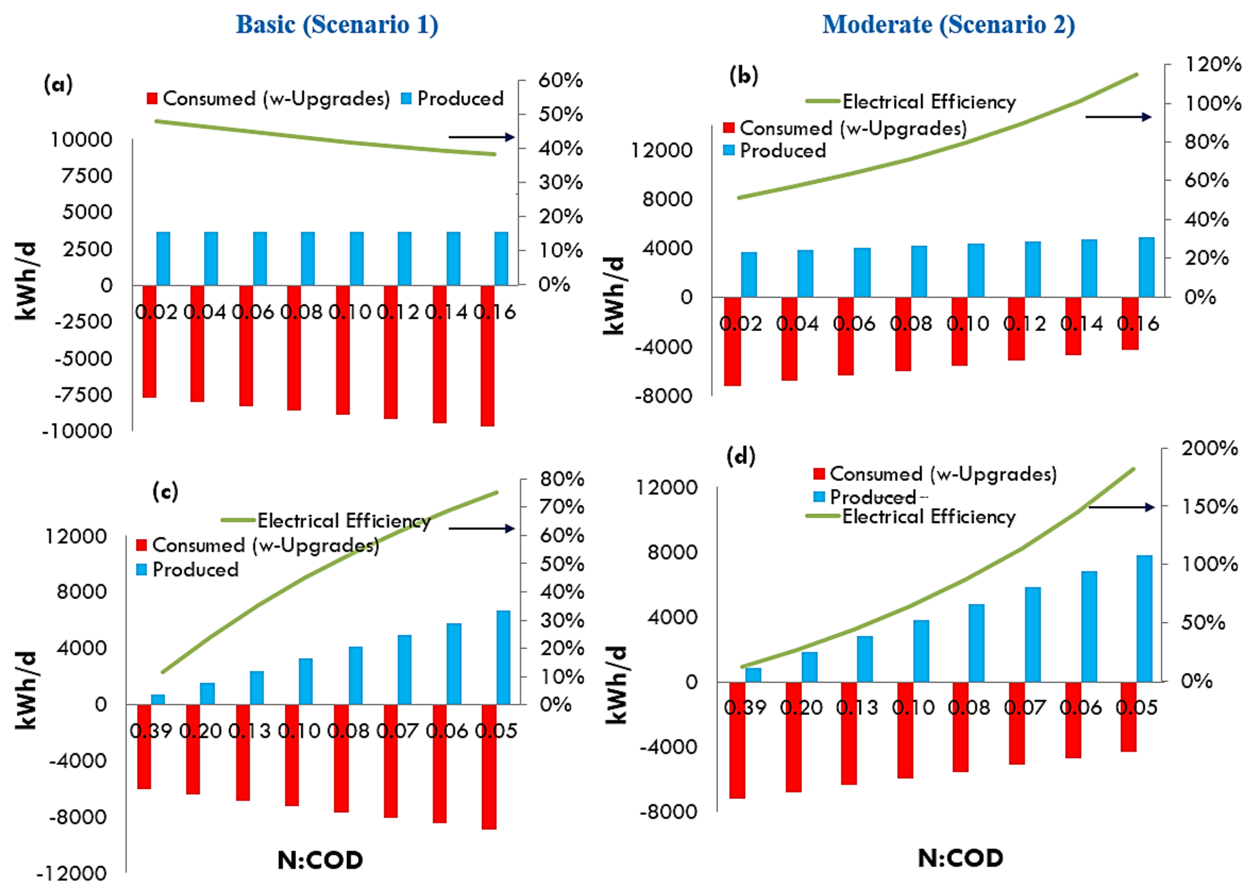

Figure 4.4 Effect of N:COD ratio on energy recovery: (a) basic and (b) moderate configurations varying N concentration from 8 – 70 mg/L with fixed COD concentration at 450 mg/L; (c) basic and (d) moderate configurations varying COD from 100 – 800 mg/L with fixed N at 39 mg/L)

4.5.2 Scenario 2a - Moderate Configuration (Nutrient removal - High-energy efficiency) with varying N:COD ratio

This configuration builds upon the basic configuration by adding a nitrogen removal process. Nitrogen removal with the anammox process provides the opportunity to remove total nitrogen and by recovering energy that would otherwise be used for aeration. 1 kg of nitrate (NO_3^- - N) is denitrified when 2.86 kg of COD are oxidized. This reduces the amount of oxygen required for ammonia oxidation by almost 50%. Ammonia oxidation requires a large amount of oxygen (~4.57 kg O_2/kg oxidized-N) (Tchobanoglous et al 2003). In other words, 4.57 kWh electricity is

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lost with every kilogram of nitrate discharge or 2.86-kWh electricity is saved per kg (NO_3^- - N) (Garrido 2013, Tchobanoglous et al 2003). The only difference between the basic and moderate process configurations is the addition of the anammox process. **Fig. 4.5** shows the process flow of the moderate configuration and a mass balance of COD and energy flow across the individual unit processes. By adding the nitrogen removal step for the 5 MGD WWTP, energy efficiency improves to 73% with WAS only; and 112% with FOG-WAS codigestion compared to the basic configuration.

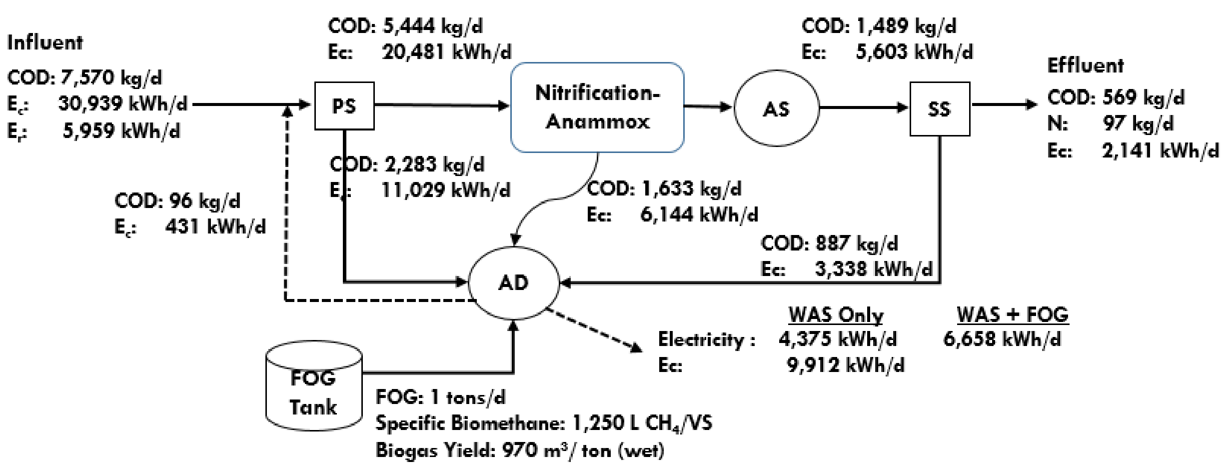

Figure 4.5 Mass and energy balance analysis for *Scenario 2* (moderate configuration)

Analysis of N:COD ratio in **Fig. 4.4b** for the moderate configuration shows a different trend compared to the basic **Fig. 4.4a**. **Fig. 4.4b** shows an opposite trend with energy efficiency increasing with increasing N:COD ratio. This implies that nitrogen removal technology helps to control any upset (that is higher energy demand) if and when influent N concentration is not constant but varies. Energy efficiency greater than 80% started at 0.1 N:COD, with an energy autarky of 102% at 0.16 N:COD ratio (Figure 4-b). However, if N is fixed and COD varies, the clock turns to favor lower N:COD ratio.

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4.5.3 Scenario 1b – Basic Configuration (COD removal – Supplemental biogas production) with equipment upgrades and codigestion

Combining high biodegradable waste such as FOG with wastewater sludge for codigestion significantly improves the overall energy balance. Codigestion is the best option for improving yields of the anaerobic digestion process, which improves biogas yields due to positive synergism established in the digestion medium and the supply of missing nutrients by the co-substrates (Mata-Alvarex et al. 2000, Mata-Alvarex et al. 2014, Edelmann et al 2000). Codigestion of wastewater sludge with other organic wastes such as FOG and/or high-strength waste has received increasing attention over the years (Mata-Alvarex et al. 2000, Mata-Alvarex et al. 2014, Edelmann et al 2000). FOG under a mesophilic process has a high VSS destruction ratio (ranging from 70 to 80%) and potential biogas generation of up to 1.3 m^3/kg VSS destroyed, compared to a normal biosolids gas generation rate of 1 m^3/kg VSS destroyed (Johnson et al 2009).

Fig. 4.6a shows the effect of combining FOG and sewage sludge for codigestion. FOG can be collected from other food and dairy industries in addition to the small amounts collected in preliminary treatment. It is evident that a WWTP under this category could easily become energy-neutral or even energy-positive when 1 ton per day (tpd) FOG is used. This is only true for wastewater with high COD strength (> 800 mg/L). Thus, a 5-MGD capacity WWTP with influent wastewater COD strength less than 800 mg/L will not achieve 100% energy efficiency unless significant efforts are made to improve process equipment energy efficiencies. The anammox process for the moderate configuration provides a superior advantage for codigestion; not just in terms of energy production but the ability to control “nutrient-shock” from AD recycle stream. The application of codigestion has a higher chance of having a nutrient impact on the wastewater treatment liquid stream; that eventually gets recycle/return back to the process stream after the biosolids have dewatered.

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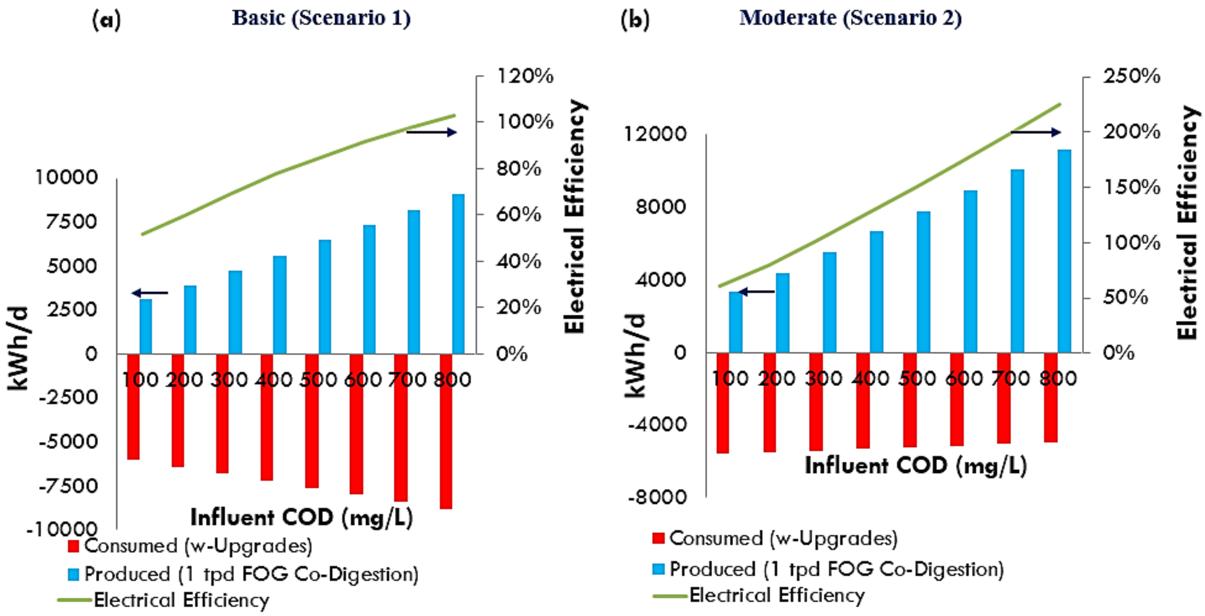

Figure 4.6 Effect of Codigestion and equipment upgrades on energy recovery (addition of 1 ton per day of FOG for codigestion)

4.5.4 Scenario 2b - Moderate Configuration (Nutrient removal with high energy efficiency) with equipment upgrades and codigestion

The addition of FOG for codigestion with sewage sludge to this configuration gives it a superior advantage (~35% more energy) compared to the basic configuration. This is because the addition of nitritation-anammox process removes nitrogen which significantly reduces the energy required for aeration. In other words, less organic matter and nutrients means less energy consumption. By incorporating FOG codigestion (**Fig. 4.6b**), influent concentration as low as 300 mgCOD/L could achieve approximately 102% energy efficiency.

Biogas production is a way of stabilizing waste sludge but flaring the gas without recovering usable energy into heat and power is much more expensive. There is a financial benefit associated with biogas utilization for either heating or on-site electricity generation. **Fig. 4.7**a and **Fig. 4.7c** indicate that the levelized cost per unit energy produced over a 25 year-life period is

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significantly less when an improvement is made to increase energy efficiency. The minimum levelized energy cost (LEC) achieved without codigestion was \$1.03/kWh at the highest influent COD concentration (800 mg/L) (**Fig. 4.7a**). By implementing codigestion to boost energy production, LEC drops to less than \$1/kWh at 500 mg COD/L (**Fig. 4.7c**). Codigestion generates extra revenue for WWTPs with a tipping fee varying from \$50 to \$170 per ton in the US (Parry 2013). The LEC for moderate configuration (*Scenario 2*) below \$1/kWh starts from an influent concentration of 600 mgCOD/L without codigestion and 400 mgCOD/L with codigestion (**Fig. 4.7b** and **Fig. 4.7d**).

4.6 Advanced Configuration

One key way to mitigate energy demands for wastewater treatment is by eliminating aeration needs. Demand for aeration accounts for 40 – 60% of the total energy consumption and it presents as a limiting factor for reducing energy consumption in WWTP operations (Shi 2011). The advanced configuration eliminates aeration needs by developing energy-yielding anaerobic treatment technologies (McCarty 2011). Many existing plants will consider facility upgrading instead of complete replacement, which makes the basic and moderate technologies of more favorable choices. Each step in the advanced configuration is highly focused on maximizing energy recovery from wastewater while minimizing energy consumption. The following section describes a recently proposed energy-positive wastewater treatment scheme (Gikas 2016).

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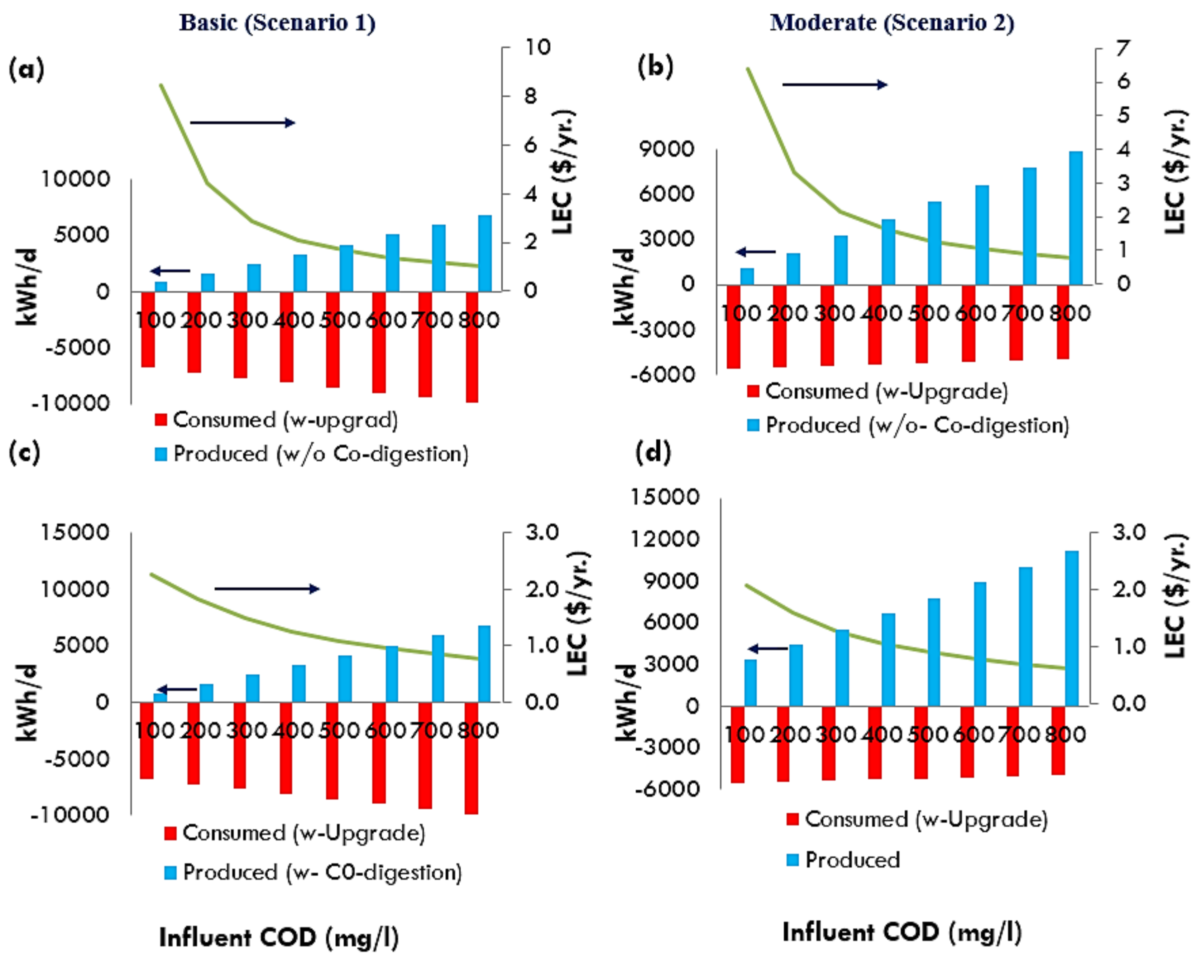

Figure 4.7 Energy production against LEC for different influent COD concentrations: (a) basic configuration without codigestion; (b) moderate configuration without codigestion; (c) basic configuration with codigestion and equipment upgrades; and (d) moderate configuration with codigestion and equipment upgrades

Wastewater entering the treatment plant contains nearly 10 times the energy required to treat the waste (WERF 2011, USDOE 2014). In a conventional activated sludge process, the biodegradable organic carbon contained in a primary sludge is higher than the biological sludge (secondary sludge), which is highly digested. Increasing the removal of biodegradable organic carbon in the primary treatment stage could potentially increase biogas production and at the same

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time, it could reduce energy consumption by lowering the amount of oxygen required by heterotrophic microorganisms for cell synthesis in the biological treatment process.

A combination of physicochemical processes (advanced micro-sieving and filtration processes) for upfront solids removal, along with downstream low-energy biological treatment process (low-height trickling filters and encapsulated denitrification) for complete wastewater treatment, appears to improve the net energy benefits. The premise for this approach is to capture the total suspended solids to the extent possible and digest them for energy production. The proposed approach uses a proprietary rotating fabric belt MicroScreen with pore size ranging from 100-300 mm, followed by a proprietary Continuous Backwash Upflow Media Filter or cloth media filter. The preliminary-primary treatment achieves about 80-90% reduction in TSS and 60-70% (a 30-45% dry solid cake) reduction in BOD_5 (Gikas 2016). Energy flow analysis for this process is shown in **Fig. 4.8a**. The estimated energy consumption for micro-sieving with auger press and primary filtration (cloth or sand media filters) are 0.005 kWh/m^3 and 0.010 kWh/m^3, respectively (**Fig. 4.8b**). This configuration was demonstrated in a recent pilot-scale study (Gikas 2016).

As shown in **Fig. 4.8c**, the energy required for wastewater treatment was estimated at 0.057 kWh/m^3, (or 0.087 kWh/m^3 if UV disinfection was used), which is nearly 85% less energy requirement when compared with a conventional activated sludge process. The biosolids produced during the process can generate about 0.172 kWh/m^3 of net electric energy making the process energy-positive.

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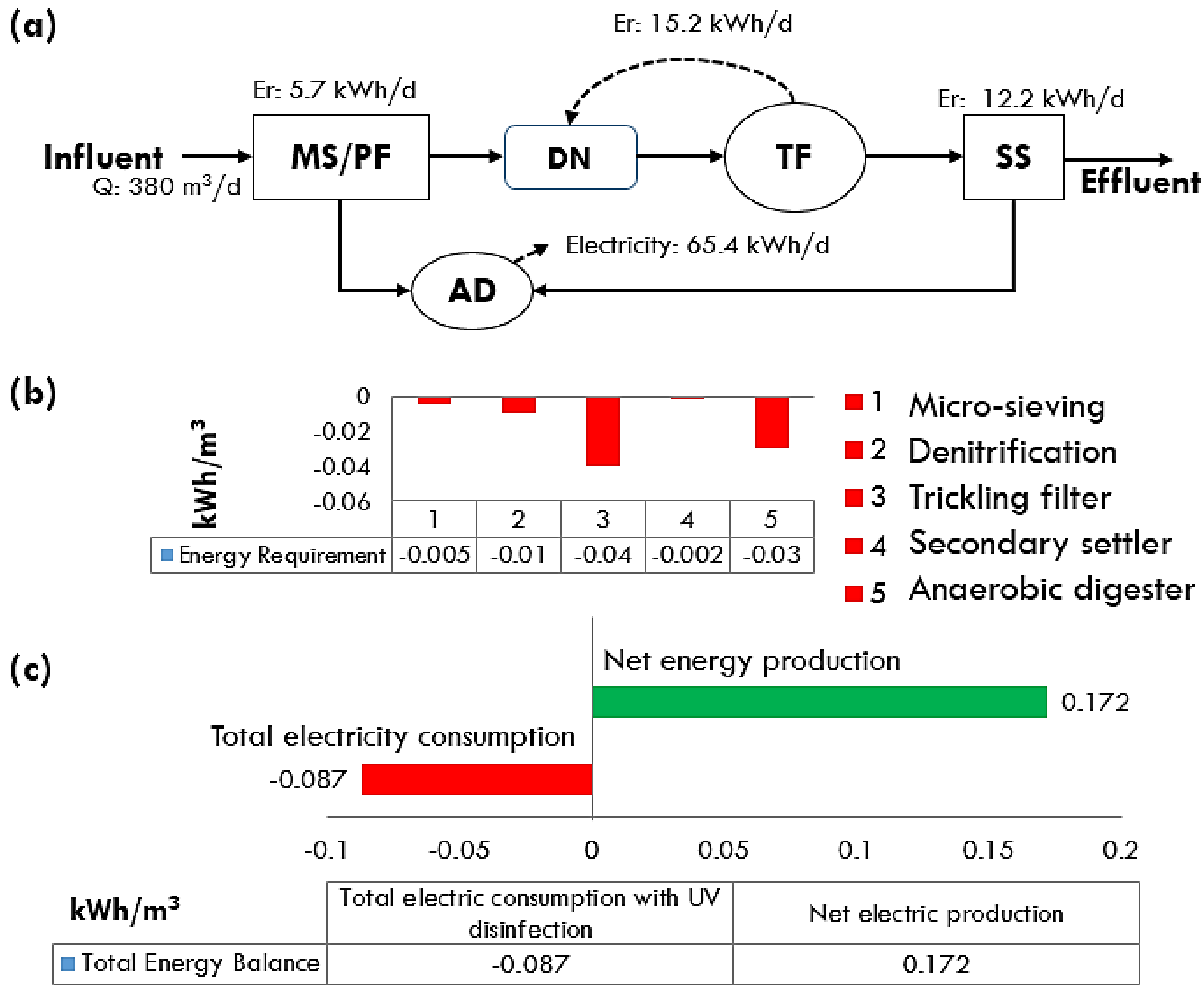

Figure 4.8 (**a**) Energy analysis of novel energy positive WWTP (MS- microsieving and PF – primary filtration; DN – denitrification; TF – trickling filter; SS – secondary settler); (**b**) energy consumption by individual process components; and (**c**) energy balance of the WWTP.

4.7 Discussion

Table 4.5 summarizes selected results obtained from the three different scenarios analyzed in this study. Although wastewater contains high energy content in the form of organic matter (i.e., COD), only a fraction of that can be recovered as electricity (Garrido 2013, Chae and Kang 2013, Shizas and Bagley 2004, Heidrich and Curtis 2011). Therefore, the first question this study seeks to answer is "what can be done to the existing CAS WWTPs to achieve an energy autarky (self-

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sufficiency) status?”. It should be noted that existing CAS WWTPs without any energy conservation measures or modifications are reported to recover less than 35% energy in the form of electricity (Bhatia et al. 2018) which is comparable to *Scenario 1a*. For instance, the Jurong Water Reclamation plant in Singapore and the Gaobeidian WWTP in China achieved 35 and 31% energy efficiencies, respectively (Zhou et al. 2013). Energy efficiency can be improved by considering process equipment upgrades as suggested in **Table 4.1** and energy production can be enhanced by co-digesting FOG and WAS (*Scenario 2a*). Utilities like Gresham WWTP in the U.S., Zurich Werdholzli WWTP in Switzerland and East Bay Utility District WWTP in the U.S., all achieved greater than 100% energy self-sufficiency (autarky) by implementing the above two approaches (Shen et al 2015). The next question is “How would the integration of nutrient removal affect the energy performance of the existing WWTPs (such as basic configuration)?”.

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Table 4.5 Comparison of selected results obtained from the three different scenarios.

Scenario	Description	Electricity Consumed (kWh/d)	Electricity Produced (kWh/d)	Energy Autarky (%)	Recovery status	Energy recovery factor*
1-a	Basic Configuration (N:COD; Fixed COD)	7707	3688	48	(-)ve	1.4
	Basic Configuration (N:COD; Fixed N)	8874	6678	75	(-)ve	2.1
1-b	Moderate Configuration (N:COD; Fixed COD)	4295	4941	115	(+)ve	3.3
	Moderate Configuration (N:COD; Fixed N)	4295	7796	182	(+)ve	5.2
2-a	Basic Configuration (Equipment Upgrades/Codigestion)	8798	9064	103	(+)ve	2.9
2-b	Moderate Configuration (Equipment Upgrades/Codigestion)	4964	11192	225	(+)ve	6.4
3	Advance Configuration	33	65	198	(+)ve	5.7

The results summarized in this table only shows the optimum/maximum output of each scenario; *Energy recovery factor is calculated by dividing the energy autarky percentage with an average energy recovery percentage of 35% through biogas production in conventional WWTPs.

Energy performance of a WWTP depends on the N:COD ratio when nitrogen removal is a design factor. *Scenario 1b* and *Scenario 2b* propose a configuration that includes a two-stage biological treatment process in which the first stage nitritation-anammox achieves both nitrogen/carbon removal while the second stage removes carbon providing a better opportunity to achieve energy autarky. The addition of the anammox process in a CAS process allows for an increase in primary sludge and biogas production for energy recovery (115%, for *Scenario 1b*) (Garrido et al 2011). Garrido et al. 2011 have shown on a theoretical basis that a municipal WWTP of this type (combined two-stage biological treatment) can produce 111% of the total electricity demand, an 11% higher electrical energy that that is required for plant's operations (Garrido et al 2011).

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By adding a codigestion unit to co-digest supplemental waste (such as FOG) greatly increases the energy efficiency (estimated 225%). Nowak et al. 2011 reported that the Strass WWTP (which is similar to *Scenario 2b*) produced up to 200% of total plant's electricity demand (Nowak et al 2011). Another study showed that a municipal WWTP of this type can produce about 1 kWh/(p.e.a) more electrical energy than is needed for the operation of the plant (Svardal and Kroiss 2011). The final question is "what should be considered for a new or future WWTP construction?". Scenario 3, as discussed above, is one of the many proposed new WWTP process configurations that could be considered for future WTTP designs. Other technologies such as combining a secondary high-rate CAS process with anammox ((Svardal and Kroiss 2011, Nowak et al 2011); and ZeroWasteWater WWTP technology proposed by Verstrate and Vlaeminck (Verstraete and Vlaeminck 2011) can be considered for future WTTP designs.

4.8 Conclusions

The quantitative mass and energy balance analysis presented in this chapter discussed the schemes by which current WWTP facilities could become energy-neutral or energy-positive. It is important to note that even though different quantities of FOG feedstock to the digester were not explored, the results discussed above show a linear correlation between the amounts of codigestion feedstock used to the amount of energy produced assuming that all conditions remain the same with no adverse effects. In addition, it should be noted that factors such as financial and operational challenges can affect the outcome of the result. It can be concluded that WWTPs with capacities less than 5 MGD could achieve energy neutrality if the wastewater N:COD ratio is less than 0.1 and a more energy-efficient ICE (greater than or equal to 40%), and codigestion are included for enhanced energy recovery.

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The roadmap for existing utilities to accomplish energy autarky begins by performing energy assessment of a WWTP to identify areas within the operations that need improvement. This leads to reducing electricity consumption, which can be achieved by improving both hardware (mechanical equipment) and soft technology (process control and operation). Among hardware, the biological process (i.e., aeration facility) is the main electricity consumer and minimizing energy consumption of the aeration unit is the key. It is recommended that more effort be put into nitrogen removal since higher nitrogen concentration increases the energy requirements of the WWTPs. The use of nitritation-anammox process reduces energy requirements. Improving primary treatment efficiency presents an opportunity to enhance overall energy production and to reduce energy consumption. The addition of FOG for codigestion has a positive effect on the digestion process with higher methane yields and stable operations. Biogas production due to FOG codigestion could also increase from 15 to 30%, which is a significant contribution to electricity and heat recovery. New WTTP designs should consider the advanced configuration after a detailed assessment and practical-scale demonstration. Overall, the model presented in this study can be a beneficial assessment tool for different wastewater treatment systems.

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CHAPTER V

ENERGY-EFFICIENT WASTEWATER TREATMENT SCHEMES FOR URBAN COMMUNITIES – A QUANTITATIVE MASS AND ENERGY BALANCE APPROACH

5.1 Abstract

Much of the wastewater infrastructure in the United States is reaching its design life at a time when environmental regulations are becoming stringent and there is a growing need for sustainable development in many urban communities. The solution to this dilemma is a paradigm shift, which focuses on planning, designing, and the management of infrastructure to produce systems that have greater capacity and longevity. As part of this approach, future wastewater treatment plant (WWTP) design should be based on the resource recovery potential, recognizing wastewater as a valuable source of energy and nutrients. This results in the design of Water Resource Recovery Facilities (WRRF). To achieve this goal, urban wastewater treatment facilities are carefully examining various pathways to exploit the energy and resource recovery possibilities of wastewater as it is being treated.

This chapter presents a systematic quantitative analysis of different wastewater treatment scenarios based on wastewater strength, plant capacity, primary treatment efficiency, and different supplemental feedstock to evaluate the potential for transitioning of WWTPs into WRRFs. Increasing the efficiency of primary treatment, process equipment upgrades, and use of supplemental biodegradable organic waste are identified as influential factors. Increasing the

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removal of chemical oxygen demand (COD) by 10% through enhanced primary settling resulted in an estimated reduction in total energy requirement by 8.5% and increased recoverable energy by 8.8%. This result illustrates that influent wastewater COD strength and the plant capacity can impact energy recovery potential. Energy production from a WRRF can be enhanced by co-digestion of sewage sludge with highly biodegradable organic waste. Other analyses show that specifying an appropriate Combined Heat and Power (CHP) engine is integral in minimizing energy losses.

Keywords: wastewater, water-energy nexus, process modeling, co-digestion, anaerobic digestion, primary settling.

Abbreviations

AEP	Actual Electricity Produced
AEEI	Average Electrical Energy Intensity
AHP	Actual Heat Produced
AD	Anaerobic Digester
APE	Actual Potential Energy
APT	Advanced Primary Treatment
APU	Aeration Power Usage
AS	Activated Sludge
ASRT	Aerobic Sludge Retention Time
BP	Biogas Produced
BW	Bakery Waste
CAS	Conventional Activated Sludge

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CHP	Combined Heat and Power
COD	Chemical Oxygen Demand
DM	Dairy Manure
EPT	Enhanced Preliminary Treatment
EPA	Environmental Protection Agency
FAS	Fast Activated Sludge
FOG	Fat-Oil-Grease
FW	Food Waste
HRT	Hydraulic Retention Time
HS	High Strength
LS	Low Strength
MCRT	Mean Cell Residence Time
MS	Medium Strength
NEI	Net Energy Intensity
ONPU	Oxidation/Nitrification Power Usage
PAE	Potential Available Electricity
PE	Population Equivalent
PHA	Potential Heat Available
SRT	Solid Retention Time
TPU	Total Power Usage
TSS	Total Suspended Solids
US	United States
VSS	Volatile Suspended Solids

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VFD Variable Frequency Drive

WWTP Wastewater Treatment Plant

WRRF Water Resource Recovery Facility

kWh kilowatts-hour

MGD Million Gallons per Day

MW Megawatts

mg/L milligrams per liter

kJ kilojoules

SCFM Standard Cubic Feet per Minute

kg kilogram

tpd tons per day

5.2 Introduction

Wastewaters generated by municipalities and industrial sectors contain harmful pollutants that can have adverse impacts on the environment. In developed countries, approximately 70 - 80% of the municipal and industrial water supplies are collected as wastewater requiring proper treatment and disposal (Davis and Cornwell 2000). The most commonly used wastewater treatment process is the activated sludge (AS) process with over one hundred years of history in process improvements and optimization (Lu et al 2018, Scholz 2016). The AS process uses an aerobic process for the removal of organic waste that makes it the highest energy consumer compared to all of the other unit operations and processes in the treatment scheme. In addition, nitrogen (N) removal processes traditionally used are aerobic resulting in an increase in net energy demand (McCarthy et al 2011, Tchobanoglous et al 2003, Jonasson and Jeppsson 2007).

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According to a report by EPRI (Electric Power Research Institute EPRI, 2002), nearly 4% of electrical energy used in the U.S. is for transport and treatment of wastewater, a level similar to other developed countries (Gude 2015a). Energy consumption for a WWTP varies with treatment configuration/technology, plant capacity, and concentration of influent pollutants (such as COD). The average electrical energy intensity is about 0.13–0.79 kWh per m^3 wastewater treated (Goldstein and Smith 2002). Aeration and pumping are the two major energy consumers within the AS process; combined, they account for 70-80% of the total energy required (Goldstein and Smith 2002). **Table 5.1** presents a summary of WWTP energy consumption values reported in previous studies.

It was reported that the primary sludge contains approximately 66% of the energy entering the treatment plant, with the rest entering secondary treatment (Shizas and Bagley 2004). Heidrich, et al., reported a much higher value for energy content with an approach that minimizes the loss of volatiles (Heidrich et al 2011). Subsequently, Shizas & Bagley estimate that the energy available in a typical municipal wastewater exceeds the power requirements of the processes used to treat it by a factor of ten (Shizas and Bagley 2004). It is admitted that not all of the available energy in wastewater can be harvested in a beneficial form, as no process is 100% efficient (Shizas and Bagley 2004, Heidrich et al 2011). However, an understanding of the available energy in the wastewater is a critical step towards developing energy and resource recovery schemes in WWTPs.

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Table 5.1 Energy consumption in WWTP operations

Remark	Total Electrical Energy Intensity (kWh/m^3)	Contribution of Unit Processes: Aeration Energy (%)	Energy for Sludge Treatment (%)	Energy for Pumping (%)	Other Unit Processes (%)	Year	Ref.
Average MBR treatment systems in Singapore	0.985	60	12	12	16	2017	Gu et al 2017
250,000 PE advanced WWTP in Poland	0.48	53		30	17	2017	Zaborowska et al 2017
Benchmarking study on 14 WWTP in Portugal	~	53	~	12	35	2017	Henriques and Catarino 2017
Average Energy Distribution in Germany	~	67	11	5	17	2016	Marner et al 2016
615,000 m^3/day advanced WWTP in Italy	0.3	51	29	~	20	2016	Panepinto et al 2016
18000 m^3/day WWTP in Spain	~	42	31	20	7	2015	Aymerich et al 2015
800000 m^3/day advanced WWTP in Singapore	0.89	13	9	24	54	2015	Yeshi 2015
81,000 m^3/day CAS WWTP in Japan	0.32	46	31	18	5	2010	Mizuta 2010
CAS WWTP in Singapore	~	50	30	15	5	2009	NEWRI 2009
500,000 PE CAS WWTP in Sweden	0.48	48	14	9	29	2007	Jonasson and Jeppsson 2007

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Table 5.1 (continued)

Remark	Total Electrical Energy Intensity (kWh/m^3)	Contribution of Unit Processes				Year	Ref.
		Aeration Energy (%)	Energy for Sludge Treatment (%)	Energy for Pumping (%)	Other Unit Processes (%)		
Benchmarking study on advanced WWTP in Austria	0.3	70	13	4	13	2007	Jonasson and Jeppsson 2007
250,000 PE advanced WWTPs in Austria	0.32	57	13	9	21	2007	Jonasson and Jeppsson 2007
2.4 million PE advance WWTP in China	0.26	57	5	~	38	2007	Gans et al 2007
WWTP in Iran	0.3	77	7	11	7	2006	Nouri et al 2006

5.3 Current state-of-the-art of energy scenario in WWTPs

Increasing operating costs due to environmental performance regulations and increasing power costs provides impetus for most utilities to lower their energy consumption and enhance energy recovery. Energy consumption in different stages of WWTP operations as presented in **Table 5.1** affirms that the aeration process consumes most of the energy, followed by sludge treatment and pumping. Hence, reducing the need for aeration through process modification or use of alternative treatment strategies which minimize the requirement for adding oxygen become a logical target in this endeavor.

Going further, many utilities are now seeking ways to make WWTPs breaks even on energy utilization, or even become net energy producers. The two main steps involved in transforming a WWTP to WRRF are: (i) reducing energy consumption and (ii) enhancing energy production

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(Tchobanoglous et al 2003, Joh and Olmstead 2010, Pakenas 1995, Huseyin et al 2019). Optimizing the AS aeration process will be the first measure for reducing the energy consumption (Garrido et al 2013). Reducing the mean cell residence time decreases the net energy use (Joh and Olmstead 2010, Shi 2011) as less energy is needed for aerobic solids stabilization in the reactor. Implementing dynamic controls by applying on-line sensors may effectively control aeration demand and can save up to 30% of total aeration energy (Pakenas 1995, Michela et al 2016). Other measures include improving process control (Tchobanoglous et al 2003, Pakenas 1995, Monteith et al 2007, Wet et al 2007, Wet 2007), including variable frequency drives (EBMUD 2012 2018), conducting energy audits (Tchobanoglous et al 2003, Garrido et al 2013, Shi 2011), improving process design, and integrating energy-efficient biological processes such as anammox side-line or up-flow anaerobic sludge blanket process (Goldstein and Smith 2002, Wet et al 2007, Wet 2007).

Higher sludge removal efficiencies in primary and secondary treatment units will increase biogas production potential allowing for higher energy recovery. For instance, Oregon County Sanitation District enhanced its primary treatment process, which increased its biogas production by 18% (EPA 1995 and 2014). Alternative feedstocks such as food waste (FW), fats, oils and grease (FOG) and other organic wastes have been adopted by many utilities to increase biogas production. The high resource potential contained in organic wastes such as FOG suggests their use as feedstock for biogas production. It is reported that FOG not only increases biogas production, but also stabilizes digester operation (Shi 2011, Columbus 2010).

There are many options to enhance biogas production. Process changes within WWTP operations that result in increased biogas production include pre-concentration of solids, anaerobic digestion (AD) performance improvement, cogeneration, waste sludge pretreatment, and co-

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digestion (Tchobanoglous et al 2003, Huseyin et al 2019, Johason 2009, EPA 1995 and 2014, Piate et al 2009, Columbus 2010, Guibelin 2004, Shen et al 2015, Caliskaner et al 2016 and 2017, EPA 2007, Francesco 2017, Johnston 2015). Other measures such as energy benchmarking programs and initiating incentive policies for energy recovery can help promote the concept of developing resource recovery facility (EPA 1995, Johason 2009, Piate et al 2009). It has been reported that some WWTPs have increased their revenues by as high as $500,000 per year after adopting co-digestion schemes (EBMUD 2012 and 2014, EPA 1995).

Use of supplemental waste in many cases has provided a better economic justification for implementing CHP (combined heat and power). In addition, the use of biogas in a CHP scheme has several benefits. The CHP scheme is widely used in the wastewater treatment plants and is considered the most cost-effective technology for harvesting energy from biogas (Columbus 2010). Use of biogas in CHP schemes has the potential to offset energy consumption by up to 40% (Columbus 2010). For example, at a 415-MGD WWTP plant with a primary sludge produced from 30% COD removal in the primary settling, and adding 200 tons per day of food waste as supplemental feedstock to the digester treating the primary sludge, can produce 2 MW of electrical energy (Shen et al 2015, Johnston 2015).

Other energy recovery processes such as thermochemical and microbial fuel cell (MFC) technologies are also being explored for recovering energy from wastewater (Manara and Zabaniotou 2012) and have been well studied as potential technologies for wastewater sludge energy recovery in recent years (Gude 2016). However, such technologies present some challenges that will make it difficult to immediately integrate into existing wastewater processes. For instance, the applicability of the thermochemical process may be limited due to its reliance on solvents that have to be recovered to make the process economical. MFCs present their own

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limitations such as low COD removal, inconsistent power density, cathode performance and many more (Gude 2015 and 2016). Hence, CHP currently appears to be the most established, viable, and ready to use technology for existing WWTP for enhancing energy recovery.

5.4 Research approach

Although the above mentioned practices are proven effective, there are many barriers for their implementation in wastewater treatment plants. These include aging infrastructure, lack of financial packages for capital investment, inadequate information on payback or return periods, and a lack of interest or opportunity on the part of utilities in utilizing biogas for energy recovery. Different groups of researchers have proposed various methods through which a WWTP can become a net-positive energy producer. However, these recommendations have limited application and a holistic effect of implementing the recommendations in a prospective wastewater treatment plant design has yet to be reported.

It would be beneficial to envision the compound effect of the best design practices in a prospective wastewater treatment plant design and operation to realize the maximum energy recovery potential. The goal of this research is to use a quantitative approach to evaluate energy performance that will help bridge this knowledge gap by incorporating the best design and management practices reported by actual plant performance reports and research studies into a simple quantitative model, so that a comprehensive solution to transform an existing WWTP into a WRRF without major infrastructural changes can be developed.

Different methods such as benchmarking (Vaccari et al 2018, Haslinger et al 2016, Belloir et al 2015, Lorenzo et al 2015), life cycle analysis (Rodriguez-Garcia et al 2011 and 2013, Xu 2013, Hospido et al 2004, Gallego et al 2008, Svardal and Kroiss 2011), and conceptual approaches (Garrido et al 2013, Svardal and Kroiss 2011) have been used to evaluate the energy performance

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of WWTPs. This study uses a quantitative assessment approach to evaluate the energy performance of a WWTP. This approach is unique when compared with other methodologies because the analysis includes of various practical considerations to enhance the energy performance of a WWTP.

First, a quantitative mass-balance model for a conventional activated sludge process was developed to account for mass and energy flows in wastewater treatment plant unit operations. The model was validated with actual plant data for its practical relevance. Second, key performance indicators that influence WWTP operations were analyzed for three different wastewater strengths (Low, medium and high). In addition, the following have been studied: i) the impact of varying primary treatment efficiencies (for COD removal) on energy balance and sustainability; ii) the effect of plant capacity on the energy balance of a conventional WWTP; iii) the impact of using different anaerobic digester supplemental feedstock for co-digestion on the overall energy balance and sustainability; and iv) optimization of combined heat and power scheme for enhancing energy recovery. Scheme 1 shows the logical sequence of how the overall energy performance analysis was executed.

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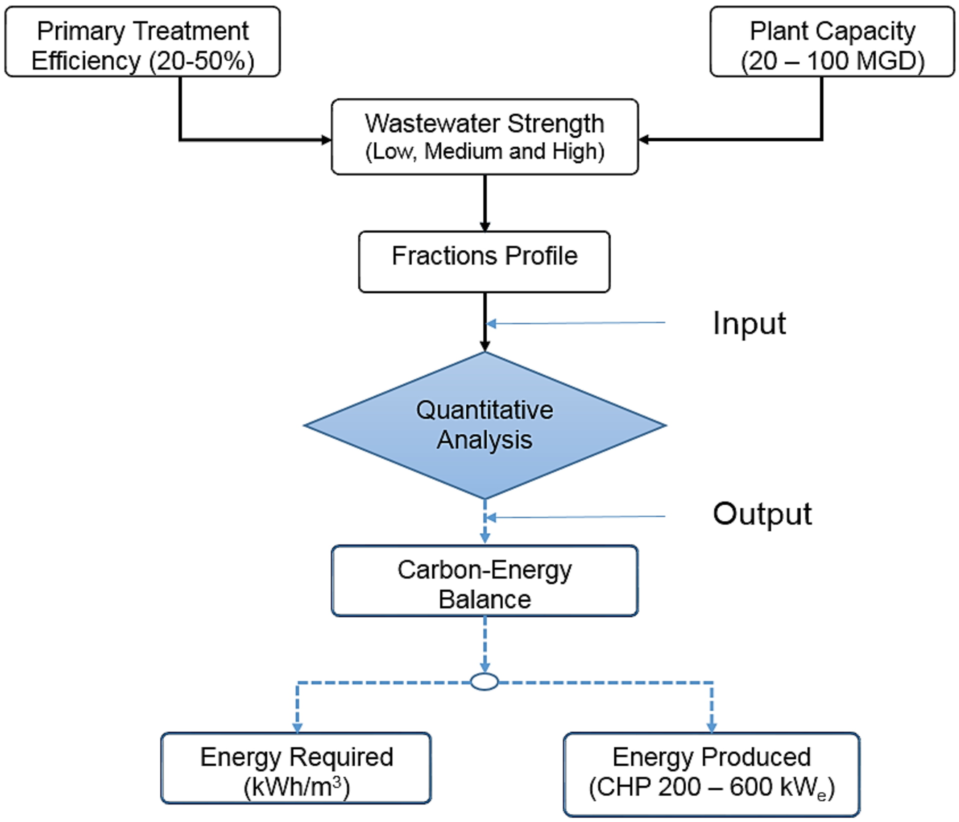

Figure 5.1 **Scheme 1**: Logical flow diagram of energy performance analysis

5.5 Materials and Methods

A quantitative mass and energy balance approach using a conventional activated sludge (CAS) process (see scheme 1) was developed to evaluate the effect of COD loading rates within different unit operations. This methodology was first developed by the USEPA to estimate the solids treatment and disposal rates in various wastewater treatment unit processes (USEPA 1979). The individual quantitative separation equations are given below. The two main components of the analysis are COD and N, which significantly affect the aeration requirements and these are actively monitored by the USEPA standards. The mass balance approach takes into account of different concentrations including recycle streams and process efficiencies. COD was used to account for the amount of organics in wastewater; thus, the potential energy consumed and

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recovered in different wastewater treatment configurations. Detailed formulation of quantitative mass-balance equations for Scheme 2 is described below.

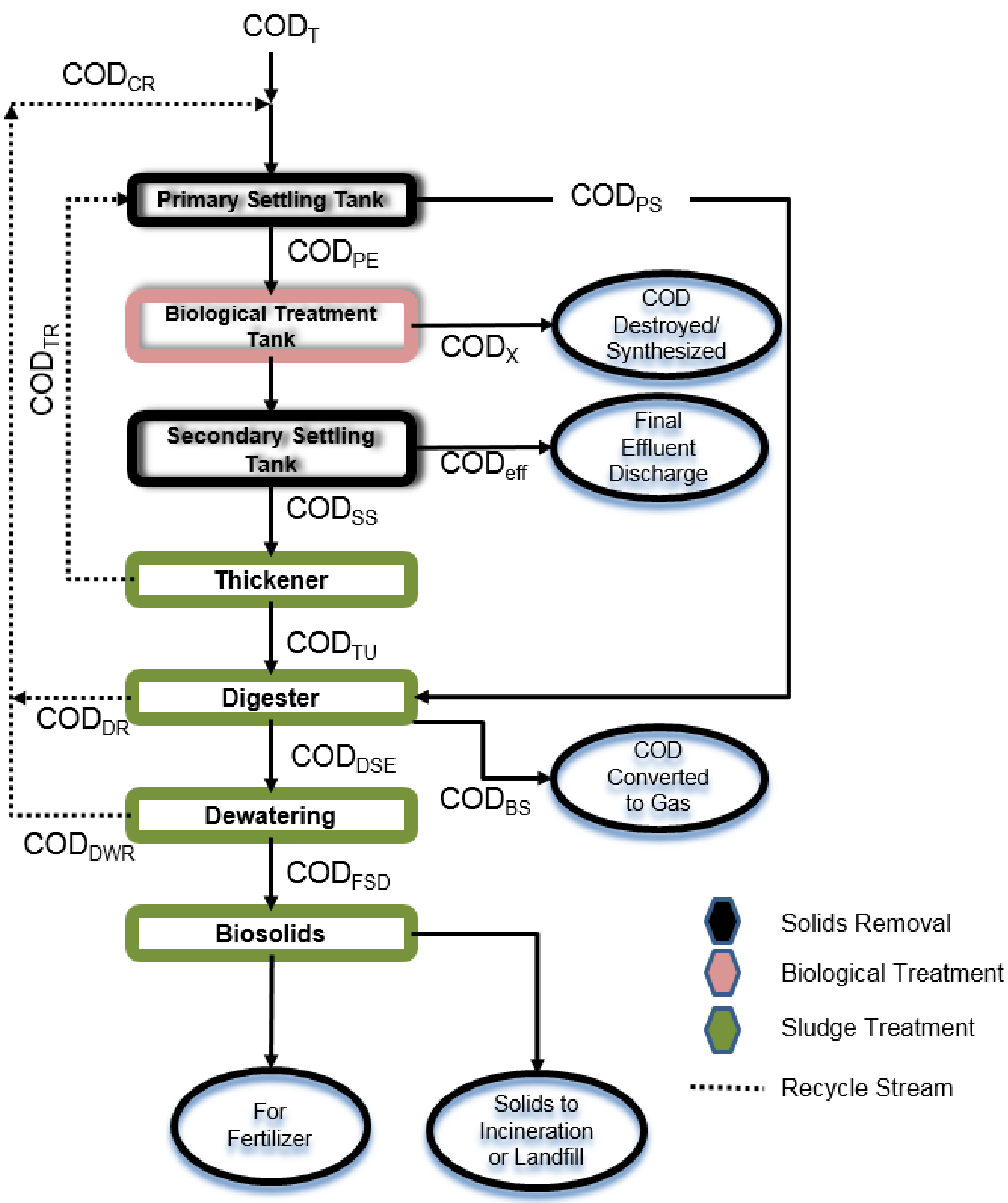

Figure 5.2 **Scheme 2**. Quantitative mass balance model for a conventional activated sludge process configuration; process configuration includes major unit operations such as: primary treatment; activated sludge process; secondary settling tank; dewatering; thickener and anaerobic digester.

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The quantitative mass balance assumes that the physical, chemical and biological processes of WWTP neither creates nor destroys matter; hence, this allows the general expression (Equation 5.1) to be adopted as:

$$\frac{dX}{dt} = X_{in} - X_{out} \tag{5.1}$$

where X_{in} and X_{out} represent the mass of dissolved components (such as COD and/or N), solids, or gas entering and exiting a unit process within the WWTP. By assuming a steady state condition without accumulation, Equation 5.1 then becomes:

$$X_{in} = X_{out} \tag{5.2}$$

Using this principle, several process interactions are examined together as shown in Scheme 1. The labels shown in the figure represent mass flow. Based on the concept explained above, several interrelated quantitative equations were developed (Equation 5.3 to 5.13). These equations establish the relationships between total influent and effluent mass and energy flows in interconnected unit processes (USEPA 1979).

Equations 3-13 represent the distribution of COD (in kg/day) in different process flows as shown in Scheme 2. The amount of COD removed as primary sludge can be expressed as:

$$COD_{PS} = \frac{COD_T - \alpha\, a - f_{DR}.a - f_{TR}COD_{eff}}{(\frac{1}{f_{PS}} - \alpha\ \theta\ - f_{DR} - f_{TR}\,\beta\,)} \tag{5.3}$$

where

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$$a = COD_{eff}\ (1 - f_{TR})$$

$$\propto = f_{DWR}\ (1 - f_{DR} - f_{BS})$$

$$\beta = ((1 - f_{PS})(1 - f\_x))/f_{PS}$$

$$\theta = 1 + \beta(1 - f_{TR})$$

COD_{PS} is the amount of COD (kg/day) removed in the form of primary sludge; COD_T is the total influent COD (kg/day); COD_{eff} is the effluent COD; f_{PS} represents the primary settler sludge removal ratio; f_{DR} and f_{TR} are recycle ratios for dewatering and thickening processes respectively; and f_X is the fraction of COD synthesized or converted to CO_2 during biological treatment. Primary treatment effluent COD (COD_{PE}), COD used for cell synthesis (COD_X) and removed as secondary sludge (COD_{SS}) are expressed in Equations 5.4 to 5.6;

$$COD_{PE} = \frac{COD_{PS}\ (1 - f_{PS})}{f_{PS}} \tag{5.4}$$

$$COD_X = f_x COD_{PE} \tag{5.5}$$

$$COD_{SS} \tag{5.6}$$

COD in thickener underflow (COD_{TU}), digester sludge effluent (COD_{DSE}), and dewatering final sludge discharge (COD_{FSD}) can be expressed as shown in Equations 5.7 to 5.9;

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$$COD_{TU} = COD_{SS}\,(1 - f_{TR}) \tag{5.7}$$

$$COD_{DSE} = (COD_{PS} + COD_{TU})(1 - f_{DR} - f_{BS}) \tag{5.8}$$

$$COD_{FSD} = COD_{DSE}\,(1 - f_{DWR}) \tag{5.9}$$

where f_{DWR} and f_{BS} are digester recycle ratio and digester solids to gas conversion ratio, respectively. The amount of COD in the digester converted to biogas is expressed as:

$$COD\_BS = f\_BS\,(COD\_PS + COD\_TU) \tag{5.10}$$

The model accounts for COD content in the liquid return (or recycle) from the thickener, digester and dewatering processes, which are expressed as;

$$COD_{TR} = f_{TR} COD_{SS} \tag{5.11}$$

$$COD_{DR} = f_{DR}(COD_{PS} + COD_{TU}) \tag{5.12}$$

$$COD_{DWR} = f_{DWR}(COD_{PS} + COD_{TU})(1 - f_{DR} - f_{BS}) \tag{5.13}$$

where COD_{TR} is the amount of COD in the thickener recycle; COD_{DR} represent COD content in digester recycle; and COD_{DWR} is the amount of COD in dewatering recycle.

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The analysis was performed by fixing the plant capacity and primary sedimentation sludge removal efficiency at 20 MGD and 30% respectively; and assuming an energy credit of 11% by improving energy performance such as replacing aeration diffusers; replacing low efficiency pumps with more efficient VFD pumps; and improving process performance (such as applying dynamic control of online sensors). This will be referred to as “equipment upgrades”. Next, the effects of plant capacity, primary treatment efficiency and co-digestion were studied. All scenarios were analyzed based on three different influent wastewater strengths (Tchobanoglous et al 2003). **Table 5.2** provides a summary of the basic assumptions considered in this work.

Table 5.2 Parametric assumptions for quantitative mass-balance model

Process	Unit	Assumptions	Reference
Influent Wastewater			
Low Strength (LS)	mg COD/L	390	This study
Medium strength (MS)	mg COD/L	720	This study
High strength (HS)	mg COD/L	1230	This study
CAS			
O_2 transfer	kg O_2/KWh	1	(Tchobanoglous et al 2003)
Biomass yield	g VSS/g $CODbs_{removed}$	0.5	
Anaerobic Sludge Digester			
Heating Value	kJ/m^3	22400	(Tchobanoglous et al 2003)
Biomass yield	g VSS/g $COD_{removed}$	0.06	
CH_4 energy content	kJ/g CH_4	50.1	
CHP			
Electricity recovery	kWh/SCFM	15	This study
Power to Heat Ratio	Ratio	0.6	
Co-Digestion			
Biogas yield of FOG	m^3/wet ton	970	FNR 2005, 2012, Moody et al 2011
Biogas yield of FW	m^3/wet ton	150	
Biogas yield of DW	m^3/wet ton	35	
Biogas yield of Bakery Waste	m^3/wet ton	700	

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The model breaks down the influent COD concentration into four different fractions (as suggested by Metcalf & Eddy and considering the typical characteristics of wastewater adopted in Activated Sludge Model 1) as: slowly biodegradable material (X_s), 70% of total COD; readily biodegradable material (S_s), 15% of total COD, inert soluble material (S_i), 5% of total COD; and inert particulate (X_i), 10% of total COD (Henze et al 1995). The model assumes that inert soluble materials (S_i) are generated through hydrolysis. S_i is non-biodegradable within a continuous AS process or cannot be separated either by physical or biological processes (Ekama et al 1986, Henze et al 1995). X_i is only removed by clarification and it generally determines the amount of sludge produced by both primary and secondary sludge due to its ability to form a floc with activated sludge. S_s is a readily available food component utilized by heterotrophic bacteria. X_s, on the other hand, influences the aeration requirement for biological treatment, as it is partly decomposed in the anaerobic digester [65, 66]. **Fig. 5.3** shows the graphical distribution of the COD fractions in the model.

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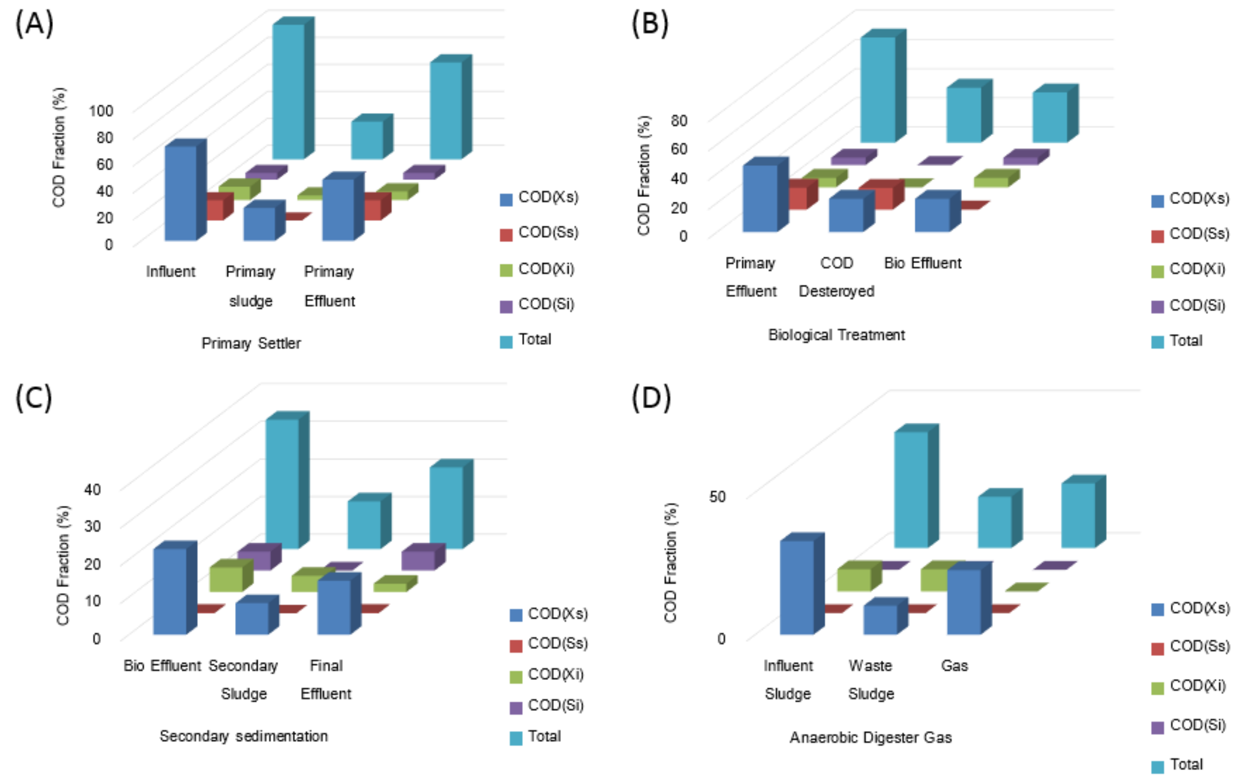

Figure 5.3 Graphical representation of COD fractions in the model; (A) influent COD fraction entering PS; removed as sludge and effluent leaving PS; (B) COD fractions entering the biological treatment process; utilized for cell synthesis and that leaving the reactor; (C) COD fraction in secondary settling; and (D) COD fraction available for gas generation in the anaerobic digester.

Energy intensity for different treatment conditions were estimated by using Eq. 5.14, which represents the total power used (TPU) in the form of electricity accounting for variation in treatment plant capacity and influent COD and N concentrations.

$$TPU = EEI_P - \left(APU_p - APU_C\right) \qquad (5.14)$$

where EEI_P (kWh/m^3) is the expected electrical energy intensity (practical) based on Eq. 5.15; APU_P (kWh/m^3) is the aeration power usage (practical) which is given as 44% of EEI_P,

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adopted from EPRI report (Goldstein and Smith 2002); APU_C (kWh/m^3) is the calculated aeration power usage based on Eq. 5. 16.

$$EEI_P = 0.6938 \times Q^{-0.132} \tag{5.15}$$

$$APU_C = \left(\frac{APU}{Q \times O_{2-T}}\right)\left(\frac{MGD}{3785.4m^3d^{-1}}\right) \tag{5.16}$$

Eqs. 5.16 and 5.17 account for the variations in wastewater treatment flow capacity (Q) and influent wastewater strength (such as the ratio of COD and N) respectively; Data used to formulate Equation 16 was adopted from EPRI (Goldstein and Smith 2002). APU is the aeration power usage or the total electrical energy required for carbon oxidation and nitrification, and $O_{2\text{-}T}$ (kg O_2/kWh) is oxygen transfer rate given in **Table 5.3**.

$$APU = \left(Q(COD_X) - 1.42P_{X.bio} + 4.33Q(NH_4\text{-}N)\right)(3785\frac{m^3d^{-1}}{MGD}) \tag{5.17}$$

$$P_{X.bio} = Y_{obs}Q(COD_X) \tag{5.18}$$

Eq. 17 and Eq. 18 were adopted from Metcalf and Eddy 2003 (Tchobanoglous et al 2003). COD_X is the amount of biodegradable COD synthesized by biomass (g COD/m^3); this is given as Equation 5. $P_{X.bio}$ is net daily waste activated sludge produced (kg/d); and Y_{obs} is the biomass yield given in **Table 5.3**.

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Table 5.3 Validation of model with actual utility data

Category	Unit	Gresham WWTP[1,a]	This Study[a]	EBMUD WWTP[2,b]	This Study[b]
Input					
Influent Flow	MGD	13	13	67	67
WW Strength	mg/L	518	MS	<600	MS
FOG feed rate	MGD	0.84	0.84	100 [3]	100
Equipment upgrade	%	15	15	5 - 8	11
Output					
Energy consumed intensity	kWh/m^3	-0.315	-0.35	-0.408	-0.323
Energy produced	kWh/m^3	0.385	0.365	0.55	0.469
Energy balance	kWh/m^3	0.07	0.015	0.142	0.146
Mean absolute error (MAE) [5]					
Energy consumed intensity		0.11		0.21	
Energy produced		0.05		0.15	

1 - Data obtained directly from Gresham Wastewater Treatment Plant
2 - Data directly obtained from East Bay Municipal Utility District
3 - USEPA 2015 (100 tons per day of Food waste feedstock) [23].
4 - Unit is tons per day
5 - MAE = |(Predicted - Observed)/(Observed)|
a - AD feedstock is Fat-Oil-Grease (FOG)
b - AD feedstock Food Waste (FW);

5.6 Results and Discussion

The energy demands for wastewater treatment can represent up to 40% of the energy budget for some small communities (Gude 2015b). A typical wastewater source with influent biodegradable COD of 250 mg/L requires 1,992 kWh to treat one million gallons (Gikas 2017), but it contains 4,800 kWh of chemical energy. This energy can be recovered in three major steps: (i) by enhancing primary treatment efficiency; (ii) by implementing equipment upgrades and

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process control; and (iii) by enhancing energy recovery through co-digestion and CHP schemes. The following sections will discuss the model outcomes considering these three major steps.

5.6.1 Mass and energy balance of carbon capture and energy production schemes

The process train for this configuration includes a primary sedimentation, a secondary CAS process and an anaerobic digester. Mass balances were performed using a medium strength influent COD (54,510 kg/d COD) mass loading and assuming a plant capacity of 20 MGD (representing a small-scale urban community) with a primary treatment efficiency of 30% (COD). The influent wastewater has an estimated energy content (E_c) of 81,664 kWh/d; which is two to three times higher than the energy required to treat it. Calculation of E_c (kWh/d) using Eq. 5.19 was adopted from (Shizas and Bagley 2004).

$$E_c = ((Q_i \times COD_i) \times \Delta U_{cs} \times f)(1000\ g/kg) \times (3785.41\ m^3/MGD) \qquad (5.19)$$

where E_c (kWh/d) is the chemical energy contained in wastewater; Q_i is the plant capacity (MGD); ΔU_{cs} is the influent energy content of wastewater, which is assumed to be 14.7 kJ/g COD; and f is a conversion factor; given as 3600 kWh/kJ.

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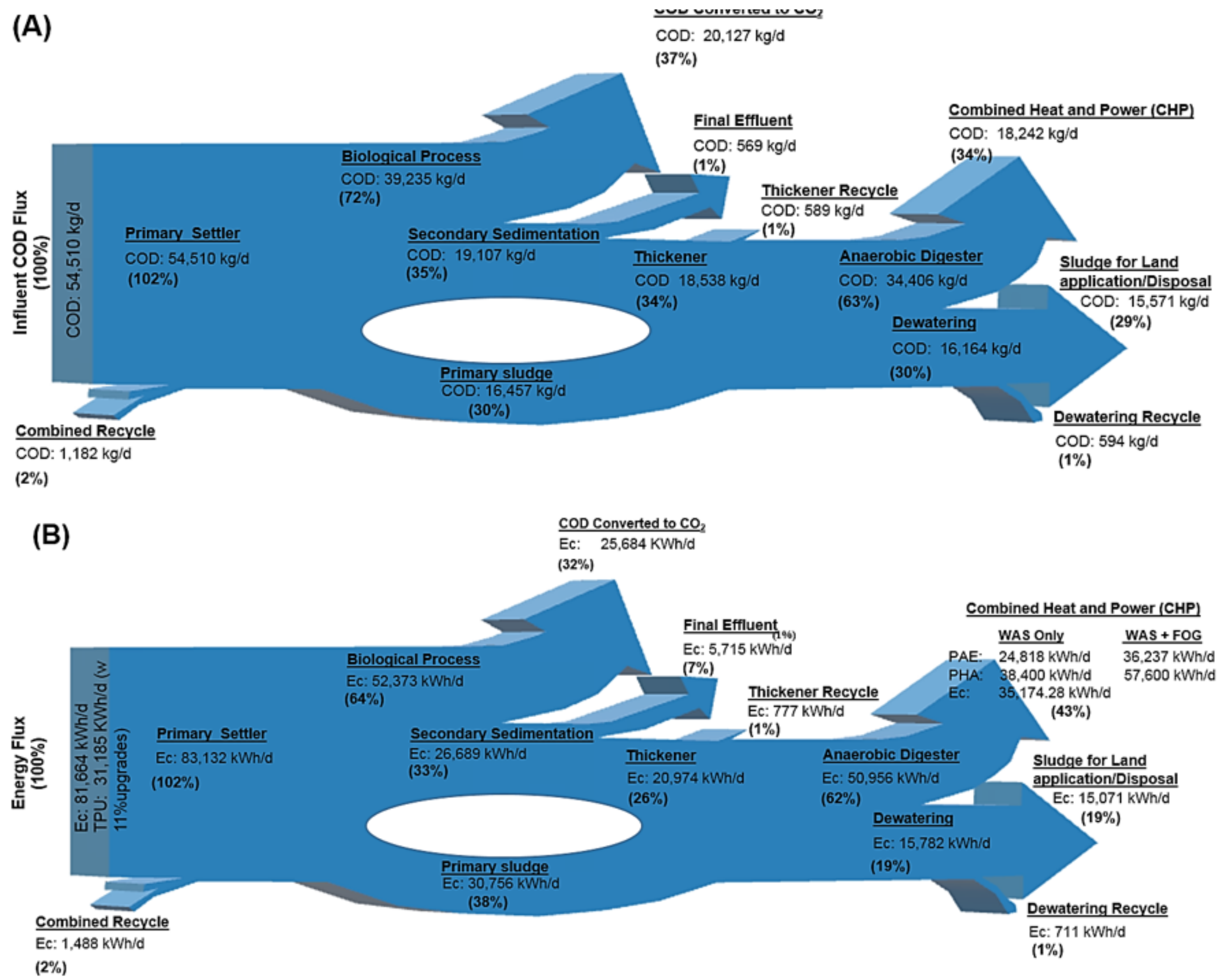

Figure 5.4 Quantitative mass balance analysis represented by a Sankey flow chart for a 20 MGD plant capacity and medium strength wastewater with a 30% COD removal in primary treatment: (A) carbon balance which accounts for recycle from both thickener and dewatering process (COD percentages are based on influent COD flux); and (B) energy balance showing the distribution of wastewater energy content within a CAS energy production (PAE for two scenarios - (a) WAS without the addition of FOG and (b) WAS with 5 wet tons per day FOG).

Energy balances were performed for **Scheme 2** using assumptions shown in **Table 4.2**. Assuming energy savings due to equipment upgrade of 11%, the resultant energy consumption was 26,965 kWh/d. The analysis shows that 20,128 kg COD/d representing 51% primary effluent COD was converted to CO_2 in the AS process. COD mass flow from both the primary and secondary sludge entering the AD accounted for 63% of the influent COD. The net CH_4

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production was 6,852 m^3/d (without co-digestion), equivalent to 18,242 kg COD/d, estimated according to the gas composition and stoichiometric coefficient of 0.4 m^3 CH_4/kg COD [5] and an SRT of 30 days at 35°C.

About 33.5% of the influent COD was converted to biogas in the digester. The biogas consists of 65% of CH_4 and 35% of CO_2 and other gases. The ratio of biogas produced to VSS destroyed was about 0.4 m^3 gas/kg VSS destruction. The estimated energy recovery potential (without co-digestion) represents approximately 80% of the TPU. Depending on the type and size of CHP engine used, the actual electricity produced could range from 40 to 70% of the TPU. The COD converted to biogas (18,242 kg COD/d) in the digester had an estimated energy content (E_c) of 35,215 kWh/d, which is 13% higher than the energy required to run the plant.

5.7 Model validation

To validate the assumptions made in the quantitative mass balance, the model was simulated to compare the outcomes with actual data from an operating WRRF. Inputs of the model were set to match (except wastewater strength was maintained at MS) the observed data for easy comparison of the output. The mean absolute error between the observed and predicted values for the two cases were low in both cases (**Table 4.3**). This means the assumptions made for this model are reasonably acceptable and have some practical relevance.

5.8 Sensitivity analysis

A series of sensitivity analyses were performed to characterize the reliability of this approach to producing energy from wastewater while illustrating operational alternatives that, in turn, might enhance approaches to limiting energy demand.

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5.8.1 Effect of wastewater strength on energy consumption and production

The wastewater strength was varied (± 20%) to predict model outcomes such as TPU, ONPU, APU, CH_4, PHA, and heat required and BP. Medium strength - MS (720 mg/L,) was set as the reference point. **Fig. 5.5** (**A)** shows a uniform balance across the predicted values; however, PHA in **Fig. 5.5** (**B)** shows slightly higher shift compared to the other parameters. This means model predictions for concentrations higher than MS are higher than its corresponding lower concentrations. As shown in **Fig. 5.5** (**A** and **B**), the predicted variables showed an output increase ranging from 15 to 25%. This also implies that the strength of the incoming wastewater has a significant impact on the overall energy balance of the wastewater process. Because it determines the energy demands and production in the process. Utilities with higher wastewater strength would benefit from higher energy production if the primary treatment efficiency can be improved.

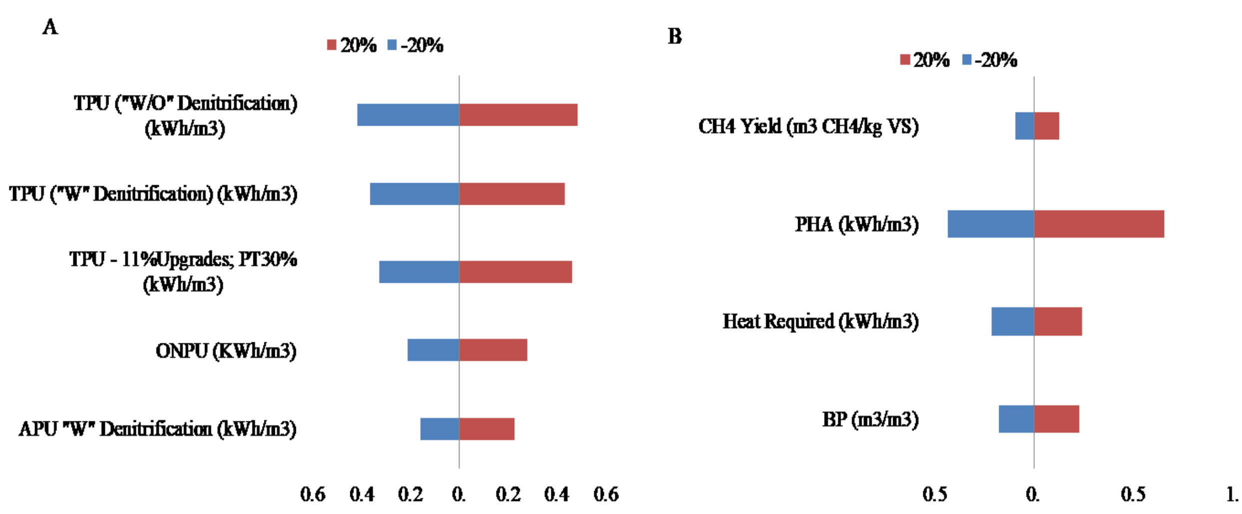

Figure 5.5 The effect of wastewater strength on biogas and energy production: Both A and B uses MS as reference point sensitivity analysis.

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5.8.2 Effect of primary treatment efficiency on energy balance

Primary treatment efficiency could have either a positive or negative impact on the overall energy balance. Less solids removal efficiency means more organic content is introduced to the secondary (biological) treatment and higher oxidation demand for cell synthesis which in turn means a higher energy demand and a lower energy production. On the other hand, higher primary treatment efficiency will have the opposite effect. **Fig. 5.6** compares the effect of varying primary sludge removal efficiencies on the overall energy balance.

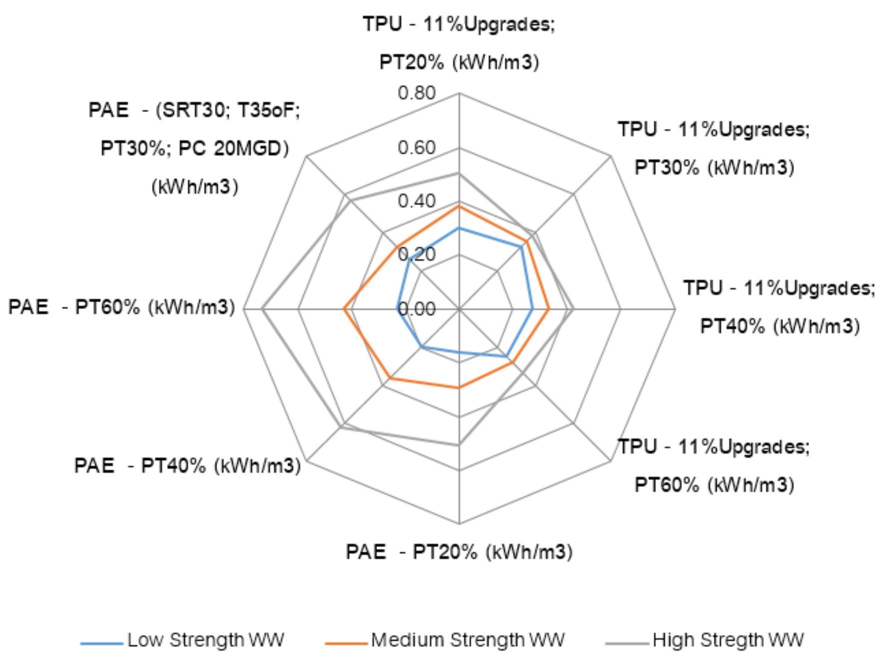

Figure 5.6 Effect of primary treatment efficiency on overall energy balance – potential available energy (PAE) at different primary treatment efficiencies

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Table 5.4 Energy balance output for varying COD removal in the primary clarifier

PT (%)	TPU (kWh/m^3)	PAE (kWh/m^3)	Energy Recovery Status	Energy Recovery Factor	EROI
Low					
20	0.35	0.23	ve(-)	0.7	1.8
30	0.33	0.26	ve(-)	0.8	2.0
40	0.31	0.29	ve(-)	0.9	2.2
60	0.27	0.34	ve(+)	1.3	2.7
Medium					
20	0.38	0.29	ve(-)	0.8	2.1
30	0.36	0.33	ve(-)	0.9	2.4
40	0.33	0.36	ve(+)	1.1	2.6
60	0.28	0.43	ve(+)	1.5	3.1
High					
20	0.42	0.35	ve(-)	0.8	2.4
30	0.39	0.39	Nuetral	1.0	2.7
40	0.36	0.43	ve(+)	1.2	3.0
60	0.3	0.52	ve(+)	1.7	3.6

The plant capacity for **Fig. 5.6** and **Table 5.4** analysis was fixed at 20 MGD and wastewater strength was varied as previously defined. It can be seen that if influent wastewater COD strength is low, energy recovery factor greater than one (>1) is only possible when COD removal in primary clarifier is 60% or greater. An energy recovery factor greater than one (~1.1) is obtained for medium strength when COD removal from primary clarifier is greater than 40 percent (>40%). With primary clarifier COD removal efficiency equal or greater than 30% high strength wastewater achieves at a minimum energy neutral; so any increase in primary clarifier COD removal efficiency increases the energy recovery factor. Nowak et al. 2011, reported that energy self-sufficiency or energy recovery factor >1 is feasible when a typical COD removal from primary clarifier is 33% (with a N:COD ratio of <0.1) (Nowak et al 2011). Also, more than 150% of the energy required is produced for both medium and high strength when primary clarifier COD removal efficiency is set 60%. This higher COD removal efficiency can only be achieved by adopting an advance primary treatment technology (APT). A similar application (that is APT) was adopted by Gikas (Henze et al 1995) at a pilot-scale level study; replacing the primary clarifier with a combination

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of micro-sieving and primary filtration (cloth or sand media filters) achieved 60-70% (a 30-45% dry solid cake) reduction in BOD_5 (Henze et al 1995). However, the Gikas application used trickling filter process for biological treatment (instead of activated sludge), the extra energy produced from ~60% primary treatment efficiency was nearly 3 times the energy required for the treatment (Henze et al 1995). Other researchers have reported primary treatment COD removal efficiencies between 45 and 65% using an APT technology (Caliskaner et al 2016 and 2017). The APT technology is reported to be in full-scale operation at the Linda County Water District WRRF (Olivehurst, California) since 2017. Hence, highly-efficient primary treatment systems can save between 15 and 30% of aeration energy in the activated sludge process while enhancing biogas production and also reducing the energy footprint for nutrient removal.

Table 5.4 also provides additional information on the energy return on energy investment (EROI). The EROI is "basic" or "simple" ration of energy produced relative to the amount of energy consumed in it production. This energy indicator can be valuable in energy performance evaluation. An EROI ratio of <1 shows that more energy is used up than generated, and a ratio of 3 or more has been proposed to be the minimum that can be considered as sustainable (Clarens et al 2011). It is reported that the EROI for microalgae used for wastewater treatment is greater than 3; this is because the use of nutrient from the wastewater by the algae species is a crucial component in producing a positive net energy balance (John et al 2015). In the same way, when higher amount of organic matter is removed from the primary treatment stage; then less amount of aeration energy will be required and greater chance of achieving energy sustainability. Based on the criterial set for energy sustainability, low strength will not achieve an energy sustainability. Medium strength will achieve a sustainability status only with a primary clarifier COD removal

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efficiency of 60% (equivalent to EROI of 3.1). High wastewater strength on the other hand achieves energy sustainability at a primary clarifier COD removal efficiency >40%.

5.8.3 Effect of varying plant capacity on energy consumption and production

This section presents an analysis of whether or not plant size affects the total energy balance. Different plant capacities were considered to evaluate their sensitivity to energy consumption and energy production. Adding supplemental waste in a co-digestion scheme enhances energy production and transforms a WWTP into a WRRF. It is quite evident that a plant with a smaller capacity and LS wastewater could easily achieve an energy-neutral or energy-positive status with supplemental feedstock in a co-digestion scheme. In **Fig. 5.7**, a 20-MGD plant with LS wastewater gains approximately 13% excess energy; whereas plants greater than 20-MGD stay below 100% efficiency. As the wastewater strength increases to MS, all the plant capacities exceed 100% energy efficiency. Also, **Fig. 5.7** shows higher energy production for smaller capacity plants; this is probably due to the fixed rate (5 tpd) of supplemental waste for co-digestion. In reality larger plants in big cities could easily increase the amount of supplemental waste to boost energy production due to the large amount of waste available. The specific energy intensity (kWh/m^3) decreases with increasing plant size. Moreover, there is no significant change in energy production and consumption for plants greater than 100-MGD. Larger plants (> 10-MGD) have the advantage of producing more energy due to the available organic solids and low specific energy consumption compared to smaller plants. Smaller plants (< 10-MGD) can overcome some of the barriers preventing them from becoming energy producers by using supplemental feedstock (as shown in **Fig. 5.7)**. Smaller plants can also consolidate sludge handling with other plants within the same district.

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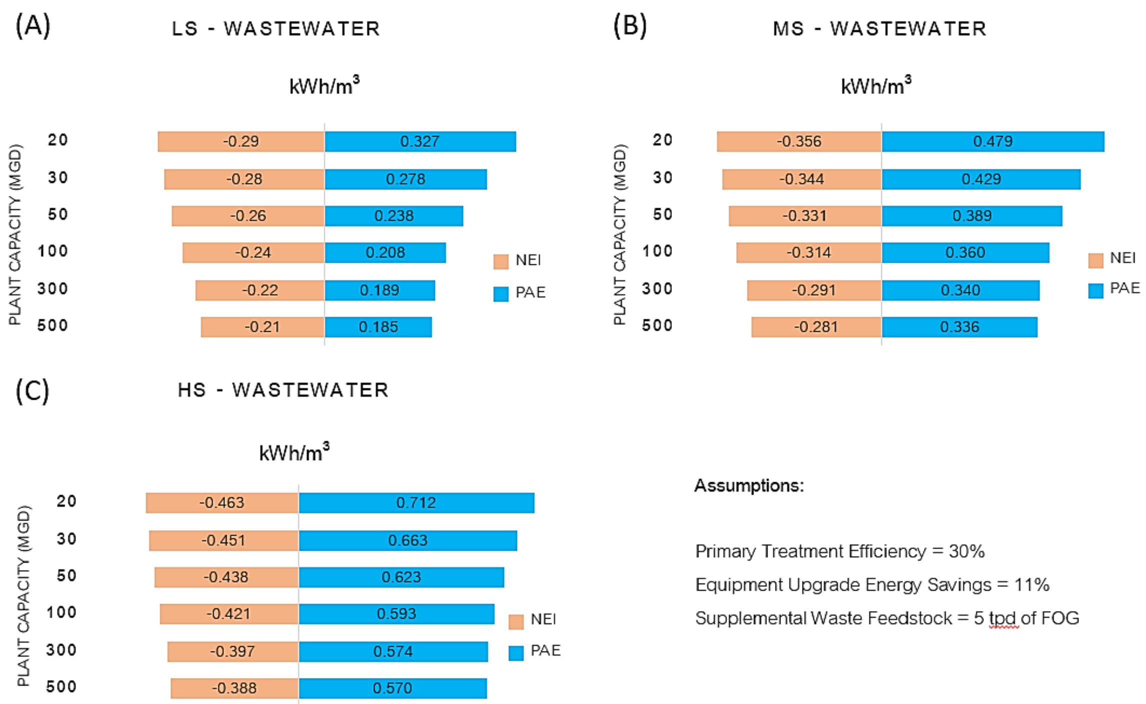

Figure 5.7 Effect of plant capacities on different influent wastewater strengths: (a) comparison of net energy intensity (NEI) and potential available energy (PAE) for a LS wastewater at different plant capacities; (b) NEI and PAE of a MS wastewater at different plant capacities; and (c) comparison of NEI and PAE of a HS wastewater at different plant capacities. All scenarios assumed a fixed 5 tons per day of FOG as feedstock to the digester.

5.9 Enhancing energy production by co-digestion

Energy production can be enhanced by adding supplemental feedstock containing high organic content. Addition of supplemental waste such as fat-oil-grease, manure, and diary waste can help increase biogas yield from the anaerobic digester. About 15% of the municipal solid waste generated in the U.S. is food waste, which contains approximately 140 trillion BTU energy. Co-digestion of food waste and municipal wastewater sludge have been practiced in recent years. The practice of co-digestion is simply the direct addition of supplemental organic waste to AD, usually by direct piping from the source or by hauling. The analysis assumed a 20-MGD plant capacity, 30% primary treatment efficiency, 30 days of SRT for AD, and AD temperature of 30°C.

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As shown in **Fig. 5.8**, using a 5-tpd FOG or bakery waste (BW), an additional 40% energy can be produced from a plant treating a HS wastewater.

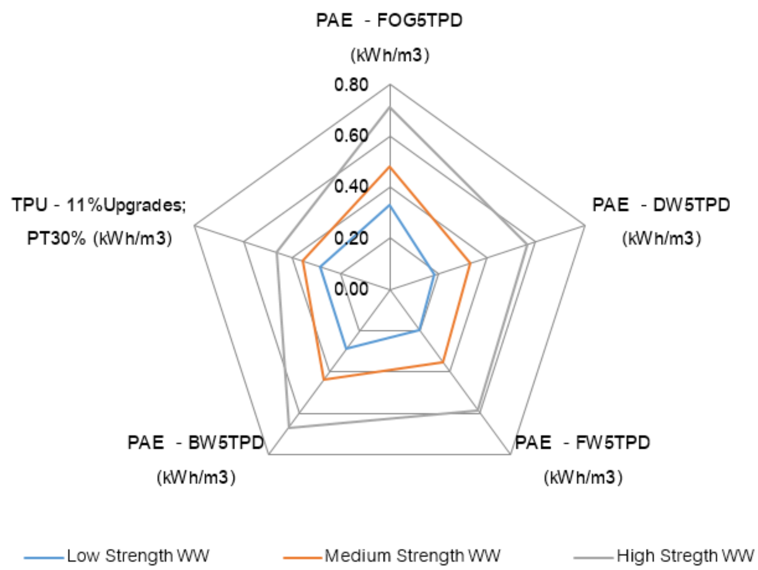

Figure 5.8 Effect of supplemental waste on energy production – Supplemental waste such as fat-oil-grease (FOG – 970 m^3 biogas per wet tons), dairy waste (DW – 35 m^3 biogas per wet ton), bakery waste (BW– 700 m^3 biogas per wet ton), and food waste (FW – 150 m^3 biogas per wet ton) evaluated for different wastewater strengths.

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Table 5.5 Energy balance output for different supplementary waste

Feedstock	TPU (kWh/m^3)	PAE (kWh/m^3)	Energy Recovery Status	Energy Recovery Factor	EROI
Low					
FOG	0.33	0.41	ve(+)	1.2	3.2
DW	0.33	0.27	ve(-)	0.8	2.1
FW	0.33	0.29	ve(-)	0.9	2.2
BW	0.33	0.37	ve(+)	1.1	2.9
Medium					
FOG	0.36	0.48	ve(+)	1.3	3.5
DW	0.36	0.33	ve(-)	0.9	2.4
FW	0.36	0.35	ve(-)	1.0	2.6
BW	0.36	0.44	ve(+)	1.2	3.2
High					
FOG	0.39	0.54	ve(+)	1.4	3.8
DW	0.39	0.4	ve(+)	1.0	2.8
FW	0.39	0.42	ve(+)	1.1	2.9
BW	0.39	0.5	ve(+)	1.3	3.5

Among all the feedstock considered, it can be seen that (**Fig. 5.8** and **Table 5.5**) DW obtains an energy recovery > 1 with only high strength wastewater; this is due to it low biogas yield. However using DW as a feedstock for co-digestion with sewage sludge has an EROI <3 for all the wastewater strength. In other words DW does not provide energy sustainability status for all conditions. The energy recovery factor for FW is >1 for wastewater strength medium and above. Similar to DW, the EROI for FW is <3 for all the three wastewater strength. The results also shows that, if the feedstock throughput for DW and FW is greater than 5 tpd, it is possible that the energy recovery and sustainability factors can increase. FOG shows a superior energy recovery potential (assuming operating conditions are stable) for all the wastewater strength. The energy recovery factor and EROI for FOG is greater than 1 and 3 respectively. Apart from FOG, for LS wastewater, using any other forms of supplemental waste will not yield an energy-neutral or energy-positive status. The scenario will be different for HS wastewater as more energy is produced from supplemental feedstock in the digester. Low strength plants can boost it energy production (by producing 0.08 kWh/m3) if FOG is used for co-digestion. Therefore, co-digestion can be a viable

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option to develop a WRRF (irrespective of the influent organic strength), although the impacts of nutrient loading, odor control and accessibility of the supplemental waste will have to be considered (EBMUD 2012, EPA 2014, Gude 2015b).

Among all the benefits associated with integrating co-digestion of mixed waste for energy production, there are a few challenges that are worth mentioning. One of the biggest challenges WRRFs encounter is the generation of sludge caused by digesting additional waste such as FOG with inconsistent characteristics. The additional sludge generated usually exceeds storage capacity which creates inventory issues; thus, thickened sludge has to be pumped back to the digesters to be managed especially during the winter months when it cannot be land spread. Other issues include high concentration of nitrogen in supplemental waste that presents challenges in meeting permit limits. In addition, tanks and piping used to handle the material continually fail due to corrosion issues, mainly due to low pH. Supernatant from the AD unit loaded with FOG residue promotes growth of undesirable filamentous microorganisms in the AS process which causes effluent issues. Despite the risk of using FOG as a co-digestate, it's economic and energy benefits are attractive to WWTPs as discussed above. Biogas production is doubled when FOG is used. Removing grease before secondary treatment helps reduce aeration energy in downstream process. Because, grease requires higher oxygen demand increasing the cost of oxygen supply in biological processes.

5.10 Combined heat and power (CHP) analysis

The amount of biogas produced by an AD is not proportional to the actual energy (electricity or heat) produced. Improper design of CHP could end up in higher energy losses. A proper evaluation of CHP should be based on different gas engine capacities. Biogas from AD can be used as a fuel source in a CHP generator set to produce electricity and heat simultaneously.

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The sizing of a CHP system can be either electric- or thermal-limited, depending on the size of the facility and energy needs. Defining the current usage of electric and thermal energies is the first step in determining the best suited CHP unit for a utility. Conventional electricity costs (demand and energy) and the cost per kWh of recovered energy are the two factors that must be considered to determine whether or not producing excess electricity is a viable alternative. Biogas-to-power efficiencies in CHPs vary, depending on the type of system. This analysis only considers different sizes of gas CHP engines. The required number of CHP units as well as optimum capacities should be determined to maximize the energy recovery from biogas.

The CHP analysis assumes a power-to-heat ratio of 0.6 and an electrical energy recovery factor of 15 scfm biogas/kWh. The engine availability was assumed to be 96 percent. CHP engine sizes are rated as kilowatt-power (kW_e). Four different CHP engine sizes as listed in **Table 5.3** were used to evaluate two operational scenarios (gas produced with and without co-digestion). **Table 5.6** summaries the maximum individual CHP capacity for heat and power. For instance, assume that a digester produces 10,000 kWh/d of energy, and a 100 kW_e CHP engine is selected. The number of CHP engines required to produce the maximum electricity will be 10,000/2,304 = 4.34. Thus, four 100-kW_e CHP engines produce 9216 kWh/d (i.e., 2304 × 4) of electricity 15,360 kWh_{th}/d (i.e., 9216 / 0.6) heat. Hence, the amount of gas to be wasted (gas to flare) will be 784 kWh/d (i.e 10,000 / (4.34 – 4)). In this case the available energy to be converted to electricity was fully utilized with four CHP engines. Increasing the engine count to five will be redundant just to capture 784 kWh/day. The goal is to maximize the energy utilization with minimum number of CHP engines and less gas wastage. The procedure described above was used to perform the CHP analysis described in the subsequent sections.

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Table 5.6 Maximum energy production of individual CHP engines considered in model analysis

Energy	100-kW$_e$	200-kW$_e$	400-kW$_e$	600-kW$_e$	800-kW$_e$
Electricity (kWh$_e$/d)	2,304	4,608	9,216	13,824	18,432
Heat (kWh$_{th}$/d)	3,840	7,680	15,360	23,040	30,720

5.11 CHP for electricity production

To select the right CHP engine size for gas production without co-digestion, both **Fig. 5.9A** and **Fig. 5.9C** must be reviewed together**.** As mentioned above, the optimum design will result in less number of engines and less gas wasting**.** HS will require about eighteen 100-kWe CHP engines with 0.02 kWh/m^3 of gas waste. Installation of 18 CHP engines is not practical considering operation and maintenance challenges. However, only three 600-kW$_e$ CHP engines will be required with 0.01 kWh/d gas to flare. Comparing all the different engines for HS, 600-kW$_e$ represents an optimum engine size for a plant treating HS wastewater. Because, less number of CHP engines are required and gas utilization is maximized. In the case of a plant treating MS wastewater, 200-kW$_e$ will be the optimum size requiring five engines and flaring 0.01 kWh/m^3. An 800-kW$_e$ engine capacity will be suitable for LS requiring only one engine with no gas wasting. **Fig. 5.9B** and **Fig. 5.9D** tell a different story when more gas is produced through co-digestion. An 800-kW$_e$ engine capacity will be the optimum engine size for both HS and MS, in both cases three engine sizes are required and no gas is flared or wasted. A 200-kW$_e$ will be selected for LS with five engines with 0.05 kWh/m^3 gas to flare.

Next, actual electricity and heat production (**Fig. 5.10**) can be estimated based on the outcome of **Fig. 5.9**. Three engine capacities (100-kW$_e$, 200-kW$_e$, and 600-kW$_e$ capacities) produce the same amount of electricity (0.55 kWh/m^3) for HS without co-digestion **Fig 5.10A**. Under the same condition, LS and MS will produce the same amount of power (at 0.18 kWh/m^3)

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using a 600-kW_e engine size. **Fig. 5.10A** and **Fig. 5.10B** show the electricity production potentials with and without co-digestion, respectively. It can be noted that the maximum power production at all wastewater strengths can be achieved by using a 200-kW_e CHP engine. Co-digestion configurations (**Fig. 5.10A**) also show that a 200-kW_e engine produces a reasonable amount of electricity at all wastewater strengths. 600-kW_e capacity is less favored for both MS and LS wastewaters, and 400-kW_e is less favored for HS wastewater. Heat production follows the same trend as electricity production. Both **Fig. 5.9** and **Fig.5.10** show the importance of CHP analysis in the overall energy balance for a WRRF.

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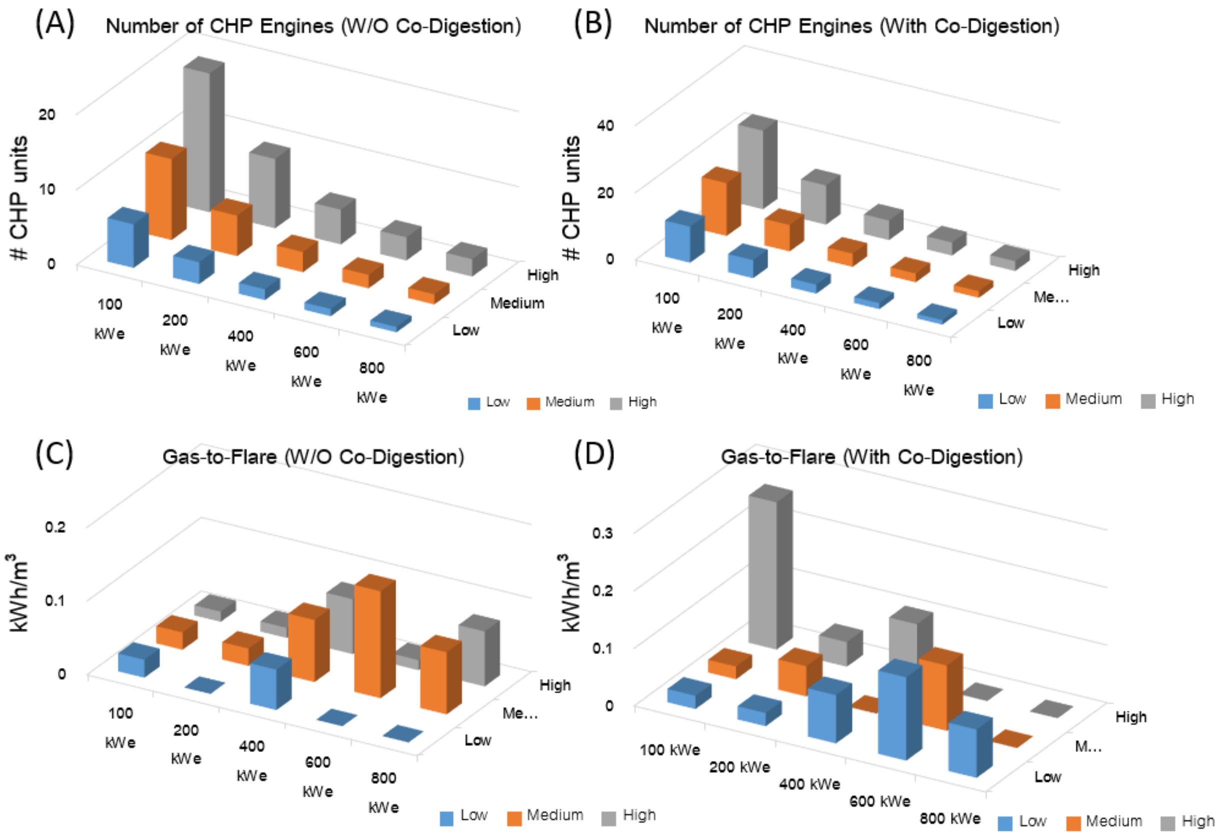

Figure 5.9 Biogas energy production analysis – Actual energy production depends on the size of a CHP engine: (A) number of CHP engines required to produce energy without co-digestion for all the three levels of wastewater strength; (B) similar situation as "A" but with co-digestion (FOG at 5 tons per day); (C) Gas-to-Flare Analysis or how much gas is lost without co-digestion; and (D) similar to "C" with co-digestion (FOG at 5 tons per day).

Table 5.7 present analysis of the actual electricity produced with respect to the type of CHP engine used. **Fig 5.9** and **Table 5.7** combined can be used as a great decision making tool for optimizing the design of CHP operation. To optimize the CHP process, the goal is to; maximize energy production, minimize gas wasting and number of CHP engines. For instance, considering biogas production without co-digestion (**Fig 5.9** a. and c; **Table 5.7**) and a medium strength wastewater. 200 kWh_e becomes the required engine because it has an energy recovery factor of 0.8 and gas wasting is less compared to the other engines. However, the number of engines

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required to achieve a minimum gas wasting is slightly higher. Compared to the other engines 200 kWh_e present the best option for biogas production without co-digestion. To analyze CHP optimization for biogas production with co-digestion (**Fig 5.10** b. and d; **Table 5.7**); it can be seen that CHP engine 800 kWh_e presents the best optimization option, with an energy recovery factor >1 (maximum of 1.3), minimum gas wasting, and less number of engines. Hence, analysis of this kind can be useful designing engineers and decision makers.

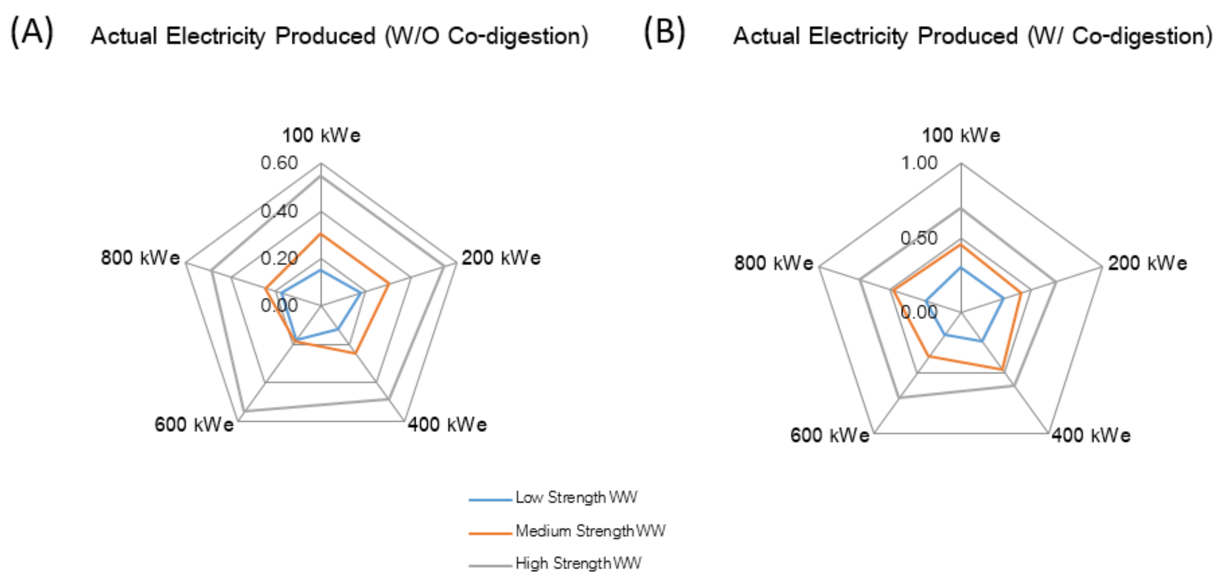

Figure 5.10 Biogas conversion to electricity and heat using for different CHP engine sizes - (A) actual electricity produced without co-digestion for different CHP engine sizes; (B) actual electricity produced with co-digestion (5-tpd FOG as supplemental feedstock).

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Table 5.7 Combined heat and power energy analysis

		W/O Co-Digestion			With Co-Digestion		
CHP Engine (kWe)	TPU (kWh/m^3)	PAE (kWh/m^3)	Energy Recovery Status	Energy Recovery Factor	PAE (kWh/m^3)	Energy Recovery Status	Energy Recovery Factor
Low							
100	0.33	0.15	ve(-)	0.5	0.3	ve(-)	0.9
200	0.33	0.18	ve(-)	0.5	0.3	ve(-)	0.9
400	0.33	0.12	ve(-)	0.4	0.24	ve(-)	0.7
600	0.33	0.18	ve(-)	0.5	0.18	ve(-)	0.5
800	0.33	0.18	ve(-)	0.5	0.24	ve(-)	0.7
Medium							
100	0.36	0.3	ve(-)	0.8	0.46	ve(+)	1.3
200	0.36	0.3	ve(-)	0.8	0.43	ve(+)	1.2
400	0.36	0.24	ve(-)	0.7	0.48	ve(+)	1.3
600	0.36	0.18	ve(-)	0.5	0.37	ve(+)	1.0
800	0.36	0.24	ve(-)	0.7	0.48	ve(+)	1.3
High							
100	0.39	0.55	ve(+)	1.4	0.7	ve(+)	1.8
200	0.39	0.55	ve(+)	1.4	0.67	ve(+)	1.7
400	0.39	0.49	ve(+)	1.3	0.61	ve(+)	1.6
600	0.39	0.55	ve(+)	1.4	0.71	ve(+)	1.8
800	0.39	0.49	ve(+)	1.3	0.71	ve(+)	1.8

5.12 Case Study – Gresham Wastewater Treatment Plant

Gresham WWTP is located in Gresham, Oregon, serving a population of about 114,000 with an average daily wastewater flow of 13 MGD. The utility installed an AD in 1990 and observed problems with their 200 kW combustion engine after ten years due to untreated biogas. In 2005, Gresham addressed the problem by installing a 400 kW CHP CAT engine and a biogas treatment system to remove siloxanes, hydrogen sulfide and moisture (Manara and Zabaniotou 2012). To improve the energy efficiency, equipment upgrades were implemented in 2010, including replacing the digester mixing equipment installed in 1990. Other equipment upgrades included a biogas mixing system with three 40-hp compressors and two linear motion mixers (one per digester) that require 5 hp per unit.

Additionally, the city replaced two multi-stage blowers that supply air to the aeration basins with two turbo blowers. Fine bubble diffusers were also installed in the aeration basin. These

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upgrades reduced the electricity consumption by 15 percent across the plant (Nora 2015). In 2012, the plant increased its biogas production by incorporating FOG as co-substrate for digestion. Prior to that, the city installed a 420 kW peak capacity ground-mounted solar energy system in 2009 that contributes approximately 5% of total energy produced. Biogas production increased from an average of 125 scfm before co-digestion to an average of 194 to 208 scfm — enough to operate two 400-kW CHP engines. Biogas contributes to about 95% of total energy production. Gresham has now achieved 122% energy efficiency (a net-positive 22%) (Nora 2015). It costs the district \$3.7 million to install the receiving and injection system for the supplemental waste unit. The utility receives a tipping fee of \$0.08/gal and the energy production saves the district about \$0.5 million per year (Nora 2015).

5.13 Conclusion

This study presented and analyzed various scenarios through which a wastewater treatment plant could possibly achieve an energy-neutral or energy-positive status. High impact best design practices such as increasing the primary treatment system efficiency, equipment upgrades, and co-digestion with supplemental waste were presented with detailed information. The analysis showed that replacing old equipment with highly-efficient ones is the first step for a WWTP to become a WRRF. In addition, improving primary treatment unit's efficiency will provide dual benefits of reducing downstream aeration energy consumption and increasing energy production. A WRRF can easily save over 20% of total energy demand when plant upgrades and primary treatment efficiency improvements are implemented. Also, increasing biogas production with alternative high-strength biodegradable waste through co-digestion is the most feasible method to achieve an energy-neutral or energy-positive status at the plant level. Co-digestion option also provides wastewater treatment plants with a new revenue stream in the form of tipping fees. Care must be

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taken when selecting a CHP engine to minimize energy losses. Replacing the aeration unit with a much less energy consuming technology such as a trickling filter or a high-rate microalgae pond seems to be a more promising alternative for future designs.

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CHAPTER VI

A COUPLED DYNAMIC MODEL FOR INTEGRATED ENERGY-POSITIVE MICROALGAE AND WASTEWATER TREATMENT SYSTEMS

6.1 Abstract

The energy and environmental performance of a full scale wastewater treatment system using a coupled dynamic simulation model of high-rate algal ponds and anaerobic digestion was evaluated through mass and energy balances. The process configuration involves wastewater primary treatment for sludge removal, microalgae-bacteria based secondary treatment, anaerobic co-digestion of microalgae-bacteria biomass, primary sludge, and biogas cogeneration. Furthermore, three scenarios were considered to enhance the biogas production based on different mixtures of FOG, primary sludge, microalgae and bacteria biomass: (i) no FOG addition to the mixed bacteria-microalgae sludge, (ii) 10% FOG addition to the mixed bacteria-microalgae sludge, and (iii) 20% FOG addition to the mixed bacteria-microalgae sludge. Carbon offset, an environmental impact factor, was analyzed for all three scenarios. Pumping wastewater to the primary settler, sludge pumping, pond paddle mixing and power for digester operations were the source of electricity consumption. In addition, digester heating was the only point for heat addition. Influencing operating parameters such as hydraulic retention time and seasonal temperatures were used to evaluate process performance. Results from the process simulations show that energy recovery is higher in summer than in winter. Improving primary treatment COD removal from 30 to 60% efficiency also improved energy recovery. The addition of FOG significantly enhanced energy production to improve electricity production. Winter effluent quality (for COD) was

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improved by increasing secondary solids separation efficiency from 75% to 90% and N effluent concentration was lowered by increasing solids retention time to 16 days.

6.2 Introduction

Adopting High Rate Algae Ponds (HRAP) for wastewater treatment and bioenergy production has attracted growing attention among researchers in recent years (Larissa et al. 2019, Rawat et al. 2011, Quinn et al. 2011, Park et al. 2011a, Craggs et al. 2013). The coupling of microalgae technology with bioenergy production presents remarkable benefits such as reducing energy consumption, reducing emissions and biomass production promote resource recovery such as nutrients and energy (Acien et. al. 2016, Park et al., 2011a). The implementation of HRAP is considered to be economically feasible as costs associated with the production of microalgae biomass and harvesting are part of the wastewater treatment costs, basically providing free feedstock for bioenergy (in the form of biogas) production (Delrue et al 2016, Benemann 2003, Rawat et al. 2011).

HRAPs are open raceway systems usually designed to be shallow (about 0.3 – 0.5 m deep) with a paddlewheel mixing unit where microalgae assimilate nutrients and generate oxygen, which is used by bacteria to oxidize organic matter (Craggs et al., 2014). Its operation performance depends on a synergistic relationship between bacteria and microalgae. HRAPs provide low-energy wastewater treatment, at the same time recover dissolved nutrients as harvested algal biomass that could be used as a biofuel feedstock (Craggs et al., 1999). However, the land area required for HRAP is large (1.7–2.7 ha/ML/day; Craggs et al., 2013) and the fact that it is a passive system presents some variability in treatment performance.

HRAP technologies are known to be a development of advanced integrated wastewater pond systems (AIWPS) which was first developed by Oswald and co-workers at the University of

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California at Berkeley in the late 1950s. (Craggs et al., 2014). Currently, most of these systems are operating in northern Californian cities, such as St. Helena (built-in 1967) and Hilmar (built in 2000). Temperature, light, and pH are the main parameters that affect the operation of HRAP systems. There are four main steps involved in HRAP process: a) solids removal; b) aerobic treatment by sunlight; c) biomass removal and conversion to bioenergy; and d) tertiary treatment of wastewater as required (Craggs et al., 2014).

Because of the increasing interest in the AIWPS as a future sustainable wastewater treatment technology, an integrated simulation of HRAP operation and energy performance analysis will be useful to help bridge the knowledge gap. One of the best ways to do that is through modeling. A coupled model of HRAP cultivation and energy recovery on a single platform for a plant-wide sustainability assessment will be the way forward for this development.

6.2.1 Review Microalgae Cultivation Models

Various models have been developed to simulate microalgae biomass growth and production. Microalgae growth models are categorized into three main groups; models based on a single nutrient substrate, light factor, and multiple limiting factors. The Monod and Droop models are examples of a single nutrient limiting model. Monod Model is mostly used when only nutrient limitation is considered. Because of its simplicity the Monod Model, has been used to describe the relationship between microalgae growth and a single nutrient concentration such as nitrogen, phosphorus, or carbon (Aslan et. al. 2006. Goldman et al. 1974). However, the main drawback of this model is its limited ability in describing microalgae growth inhibition due to high nutrient concentrations (Park et. al. 2010). Another type of model called the "Droop Model" which is solely based on the assumption that microalgae growth rate depends on the concentration of the internal nutrient in the algal cell, which is measured by the cell quota. This type of model is said to define

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the growth rate more accurately because it explains the growth in the absence of external nutrients due to accumulated nutrients in the cell (Groover et al. 1991, Eunyoung et al 2015). Tamiya model is an example of a "light factor" model which is a well-known theoretical model as well as the most widely applied model. It is comparable to a Monod-type model in describing the light effect on microalgae growth.

The multifactor models with co-limitation provide more accurate estimations and a deeper understanding of microalgae growth. The concept of co-limitation is widely applied in the development of kinetic models. Thus, the fundamental assumptions behind the co-limitation are that both multiple nutrient resources and availability of light, and their interactions control overall microalgae growth (Bello et al. 2017, Yang 2011, Terry 1980, Eunyoung et al 2015).

Researchers like Buhr and Miller (Buhr et al 1983) have described a kinetic growth modeling of biochemical interaction and synergetic relationship between photosynthetic microalgae and heterotrophic bacteria. Yang (Yang 2011) proceeded to expand on the mathematical model developed by Buhr and Miller to include the effect of pH, dissolved oxygen and substrate concentrations on carbon dioxide supply and utilization. Jupsin et al (2003) have presented a mathematical model of HRAP based on River Water Quality Model (RWQM) that was used to simulate HRAP's operating cycles considering sediment oxygen demand (SOD). Models such as WASP, QUAL 2K, Lake 2K, and CE-QUAL 2k have been used to simulate algae bloom in water bodies.

6.2.2 Energy Recovery – Anaerobic Digestion Modeling

Combining microalgae cultivation and anaerobic digestion (AD) systems for energy recovery is a promising technology to biologically convert light energy to chemical energy of methane. However, this approach faces many drawbacks due to its inherent complexity (Sialve et

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al. 2009). Hence, using a dynamic model to identify working strategies for microalgae digestion is critical. Anaerobic digestion is a complex biochemical process, where specific anaerobic bacteria degrade organic matter and produce biogas, which contains about 50 – 75% CH_4 and (20 – 45%) CO_2 (Harun et al. 2010). The AD process consists of multiple steps; hydrolysis, fermentation, acetogenesis, and methanogenesis (Diltz et al. 2013). It is worth noting that the AD of microalgae was first studied by Golueke and Oswald 1957.

Modeling of anaerobic digestion is a well-established field and has been extensively developed since the 1970s, from simple models (e.g. with one substrate limiting reaction) (Graef and Andrews, 1974) to more complex models (e.g. ADM1 with 12 reactions, (Batstone et al., 2002)). As mentioned above the model developed by Graef and Andrews is general and the only substrate considered is acetic acid. The biological step involves the conversion of volatile acids to CH_4 and CO_2 with five state variables. Hill and Barth (1977) modified Graef and Andrews's model by including a hydrolysis step with nine state variables, and is generally used for animal wast.

Husain 1998, modified Hill's model with more details concerning chemical reaction (Husain 1998). Several changes in model parameters and the death rate for both acedogens and methanogens as volatile fatty acid-based Monod functions were the key changes. Batstone et al., (2002) developed the popular ADM1 model. This model is general but complex enough to describe biochemical and physiochemical processes. The biochemical stage involves disintegration, hydrolysis, acidogenesis, and methanogenesis while the physiochemical expressions described the association and dissociation of ions, and gas-liquid transfer. The implementation of the ADM1 model is very stiff as pH and H^+ are relatively fast (Rosen 2006). Rosen et al. stated that the stiffness poses numerical challenges for implementation in e.g. MATLAB/SIMULINK, (Rosen et al. 2006). ADM1 has about 35 state variables and 12 reactions.

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Both HRAP and AD models are well studied and widely used as described earlier. However, an integrated model to simulate both HRAP cultivation and co-digestion (with a mixture of biomass including microalgae, primary sludge, bacteria sludge, and fat, oil and grease (FOG)) has not been developed. Integrated simulation of HRAP-AD can, therefore, be critical for understanding process and for identifying optimum working strategies. This study aims to develop a dynamic model that will simulate the biological conversion of photosynthetic energy to the chemical energy of methane using a coupled HRAP and AD models. The model will then be used to conduct an energy performance analysis by performing a sensitivity analysis of different parameters such as primary treatment (CPT and APT), seasonal variation (winter and summer), plant capacity (urban and rural population), SRT, and boosting biogas production with FOG for co-digestion. A detailed research matrix is present in the Appendix A. The schematics of an integrated HRAP-AD model is shown in Figure 6.1.

There are several benefits for this first coupled HRAP-AD dynamic model: (a) this will increase the application of the model for full-scale plant design, operation and optimization; (b) more developmental work on optimizing operation and control for full-scale plants; and (c) help transfer technology from research to field.

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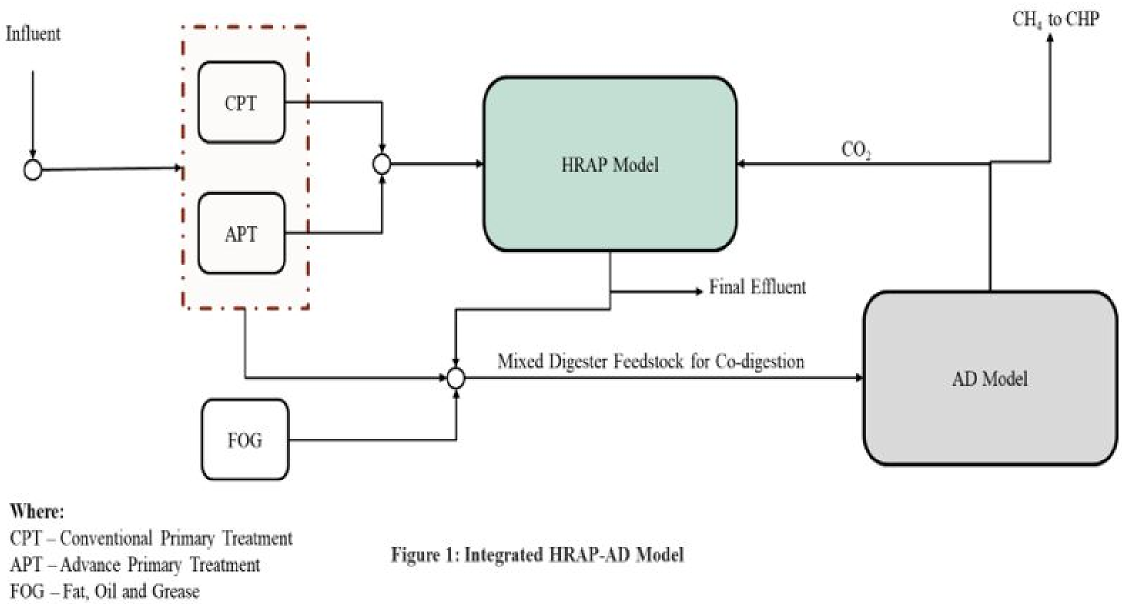

Figure 6.1 Integrated Model Schematic: This process schematic shows two different primary solids removal technologies (CPT and APT). CPT represent a conventional primary treatment and APT represent an advanced primary treatment. CPT and APT assumes 30% and 60% COD removal efficiency respectively. Primary solids, biological solids and a supplementary waste (such as FOG) are co-digested for energy recovery.

6.3 Methodology

6.3.1 Development of HRAP model

The HRAP system pond considered in this work is shown schematically in Figure 6.1. The influent characteristics of the wastewater can be described as a combination of biological oxygen demand (BOD), dissolved oxygen, dissolved inorganic nutrients and pH of wastewater; these are an important parameter that administers the biochemical transformation and substance balance in the reactor.

The effluent of the system includes water flow, gas flow, algal, and bacterial biomass. This analysis will focus on simulating variable behavior as a function of time. The HRAP model developed in this study follows the works of Bello et al. 2017, Yang 2011, and Buhr and Miller

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(1983) with a few modifications. The basic assumptions considered in developing the HRAP model are as follow.

(a) the HRAP is modeled as completely stirred tank reactors (CSTR) connected in series;

(b) the specific growth rate of microalgae is a function of light intensity (or solar radiation), temperature, dissolved CO_2, total inorganic nitrogen, and inorganic phosphorus;

(c) exchange of O_2 and CO_2 between the pond and the atmosphere; and

(d) the model does not include evaporative losses due to lower water loss.

The pond contains a microalgal-bacterial consortium. The exchanges between these two microorganisms considered in this work include the transfer of oxygen produced by the photosynthesizing microalgae to the heterotrophic bacteria and that of CO_2 generated in the oxidation process by the bacteria to microalgae. A schematic illustration of the integrated HRAP and AD model is shown in Figure 6.2. Detailed description of the model development is provided in subsequent sections.

As mentioned above the CSTRs connected in series has a recirculation loop to mimic a race-way type of hydrodynamics of the HRAP, which in most cases exhibits a certain degree of heterogeneity along with the flow in the race-way channel (Yang 2011). According to Buhr and Miller, a system configuration of about 10-25 CSTRs with a properly set recirculation flow rate can render a satisfactory approximation. The main difference between the HRAP model in this study and the works of Bello et al. 2017, Yang 2011, Buhr and Miller (1983) are that (a) this model includes phosphorus limitation for both microalgae and bacteria growth rate terms; (b) microalgae growth is affected by pH and water temperature multiplicative function; and finally (c) a nitrification of autotrophic Nitrosomonas bacteria is included.

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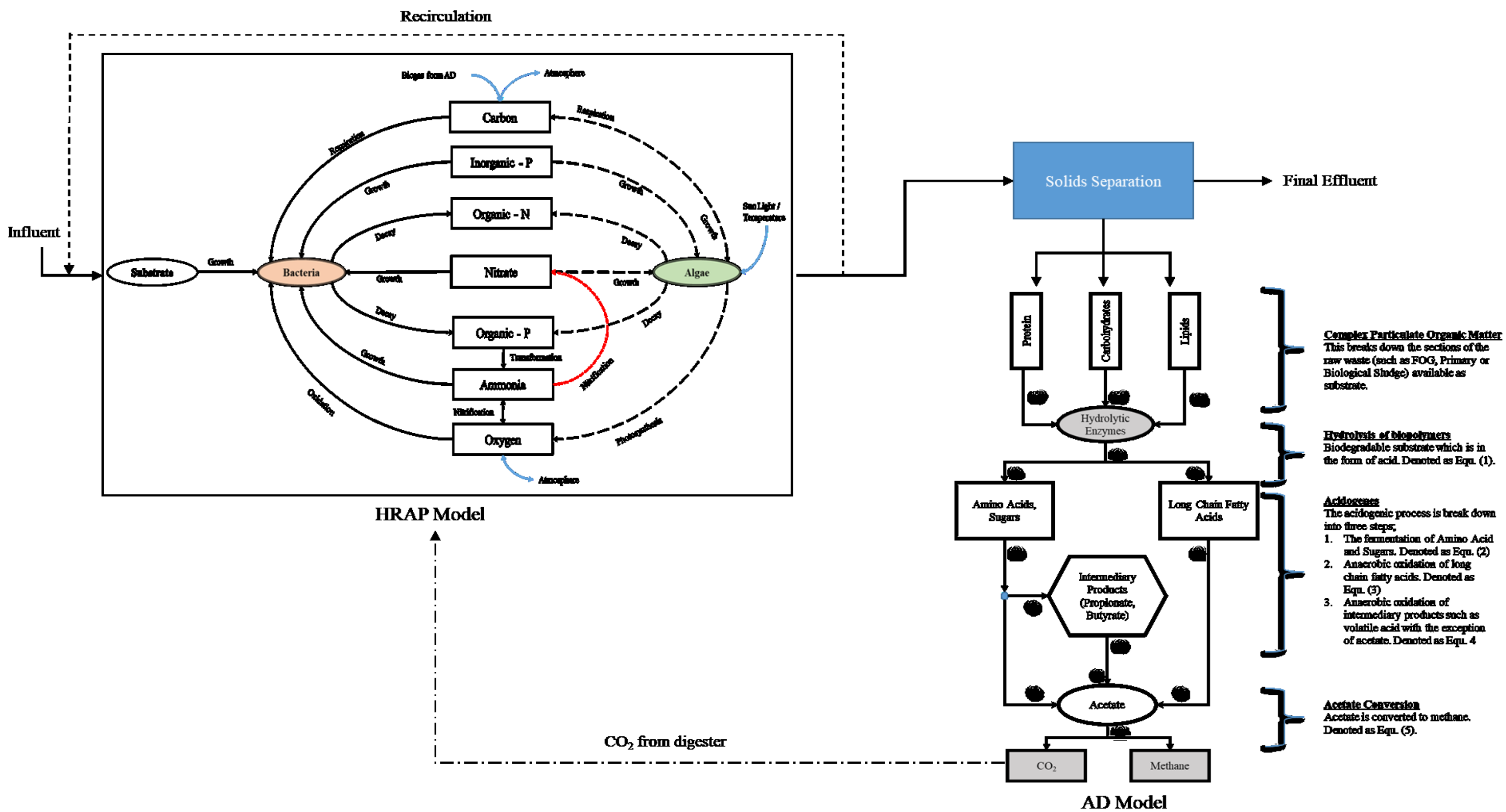

Figure 6.2 Schematic flow diagram of the HRAP-AD simulation model. The HRAP logic flow diagram represent the synergetic relationship between microalgae and bacteria. Arrows pointing a microorganism represent growth or respiration, arrows leaving the microorganism indicate death. The AD model follows a three stage process; hydrolysis, acidogensis and methanogenesis. Hydrolytic enzymes breaks down the complex organic matter to amino acid/simple sugars and long chain fatty acids.

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The equations for this model are presented in a Petersen's matrix format (Table 6.1).

The mass balance within any defined system boundary is given as;

$$[Accumulation] = [inflow] - [outflow] \pm \sum_j v_{ij}\rho_j \quad (6.1)$$

The transport terms (input and output) depends on the system of physical characteristics. The reaction term r_i is obtained by adding the product of the stoichiometric coefficient v_{ij} and the process rate expressions for the specific component ρ_j being considered in the mass balance. The HRAP-AD model was implemented on Matlab/Simulink R2019a platform. The ordinary differential equations were coded and implemented using the Matlab system function and integrated with the ODE45 solver.

The model that describes the growth of microalgae-bacteria consortium in HRAP is a set of nonlinear differential equations derived from mass balance equations for both liquid and gaseous species transformations. General model equations are presented in Table 6.1. The average solar radiation or light intensity in the pond is expressed in terms of concentration and pond depth (z) at a particular time using the Beer–Lambert's law (Bello 2017).

$$106\ CO_2 + 65\ H_2O + 16\ NH_3 + H_3PO_4 \leftrightarrow C_{106}H_{181}O_{45}N_{16}P + 118\ O_2 \quad (6.2)$$

Photosynthetic oxygen was modeled using (Eq. 6.2). This is the stoichiometric equation proposed by Stumm and Morgan (Stumm and Morgan 1970). Based on Eq. 6.2, there is 1.244 mg of O_2 produced for every milligram of microalgae synthesized.

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Equation 6.3 represents the stoichiometric relationship used to determine oxygen utilization by bacteria for respiration (Endogenous respiration). This relationship assumes a cellular composition of $C_5H_7NO_2$ for bacteria cells;

$$C_5H_7NO_2 + 5O_2 \rightarrow 5CO_2 + 2H_2O + NH_3 \tag{6.3}$$

Equations 6.2 and 6.3 form the basis for stoichiometric relationships applied throughout the model.

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Table 6.1 Petersen's matrix (Petersen 1965) for HRAP model - Process kinetics and stoichiometry for substrate oxidation, photosynthesis, and nitrification

	Component (i) →	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**Process Rate (ρi)**
j	Process (i) ↓	X_A	X_X	X_S	S_{O2}	S_{TIC}	S_{NH4}	S_{NO3}	S_{No}	S_{Pi}	S_{Po}	(g/m³.day)
1	**Growth of Phototrophs**	**1**			**1.244**	**-1.314**	**0.063**	**-0.063**		**-0.009**		$\mu_{A\,max}\cdot\left(\frac{S_{CO_{2D}}}{K_C+S_{CO_D}}\right)\cdot\left(\frac{S_{N_T}}{K_{NA}+S_{N_T}}\right)\cdot\left(\frac{S_{P_i}}{K_{PA}+S_{P_i}}\right)f(L).f(T).f(pH)$
2	Growth of Heterotrophs		1	$\frac{1}{Y}$	$0.42.Y_H$	$-0.388.Y_H$	$-0.124Y_H$	$-0.124Y_H$		-0.024		$\mu_{X\,max}\cdot\left(\frac{X_S}{K_S+X_S}\right)\cdot\left(\frac{S_{O_2}}{K_{O_2}+S_{O_2}}\right)\cdot\left(\frac{S_{N_T}}{K_{NA}+S_{N_T}}\right)\cdot\left(\frac{S_{P_i}}{K_{PA}+S_{P_i}}\right)$
3	**Decay of Phototrophs**	**-1**					**-0.063**	**-0.063**	**0.063**	**0.009**	**0.009**	$k_{dA}.X_A$
4	Decay of Heterotrophs		-1		-1.42	1.31			0.124		0.024	$k_{dX}.X_X$
5	**Nitrification**				$\frac{4.57}{Y_N}$		$-\frac{1}{Y_N}$	$\frac{1}{Y_N}$				$\mu_N\cdot\left[\frac{S_{NH_4}}{K_N+S_{NH_4}}\right]\cdot\left[\frac{S_{O_2}}{K_{NO_2}+S_{O_2}}\right].(fC_{pH}).(fC_T)$
6	Transformation of Organic N → NH_3						1		-1			$\propto_N.S_{N_o}$
8	**Transformation of Organic P → Pi**									**1**	**-1**	$\propto_P.S_{P_o}$
9	Interfacial O_2 transfer				1							$k_{la-O_2}(O_{2s}-O_2)$
10	**Interfacial CO_2 transfer**					$\frac{12}{44}$						$k_{la-CO_2}(CO_{2s}-CO_2)$
Observed conversion Rates (g/m³.day)						$r_i=\sum_j v_{ij}\rho_j$						
		Algae Biomass (g/m³.day)	Bacteria Biomass (g/m³.day)	Readlly biodegradable Substrate (g/m3.day)	Oxygen (g/m³.day)	Total Inorganic Carbon (g/m³.day)	Ammonia (g/m³.day)	Nitrate (g/m³.day)	Organic Nitrogen (g/m³.day)	**Inorganic Phosphorus (g/m³.day)**	**Organic Phosphorus (g/m³.day)**	

Stoichiometric Parameters:

Heterotrophic Yield: Y_H
Nitrosom Yield Coef.: Y_N

Kinetic Parameters:

Phototroph growth and decay:
μ_{A_max}, K_C, K_{NA}, K_{PA}, k_{dA}

Heterotrophic growth and decay:
μ_{X_max}, K_S, K_{O2}, K_{NA}, K_{PA}, k_{dA}

Nitrification: μ_N, K_{NO2}, K_N

Transformation Coefficients: α_N, α_P

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Table 6.2 Light, pH and Temperature functions for algae growth

Description	Equation	Ref.
Light Function		
Average Light Intensity with pond depth	$I_a = \frac{I_o}{d}\exp\left(\frac{1-e^{(K_{e1}+K_{e2}X_A)d}}{(K_{e1}+K_{e2}X_A)}\right)$	Belo et al. 2017
Diurnal Variation of Surface Light Intensity	$I_o(t) = \max\left(0, I_0\pi\left(Sin\left(\frac{(t-5)2\pi}{24}\right)\right)\right)$	Gomez et al 2016
Light Intensity Factor	$f(L) = \frac{I_a}{I_s}\exp\left(1-\frac{I_a}{I_s}\right)$	
pH Function	$f(pH) = \frac{[H^+]}{[H^+]+K_{OH}(T)+\frac{[H^+]^2}{K_H(T)}}$	James et al. 2013
Temperature Function	$f(T) = \begin{cases} \exp[-K_1^T(T_1-T)^2] & for\ T < T_1 \\ 1 & for\ T_1 \le T \le T_2 \\ \exp[-K_2^T(T-T_2)^2] & for\ T > T_2 \end{cases}$	Cossins and Bowler 1987

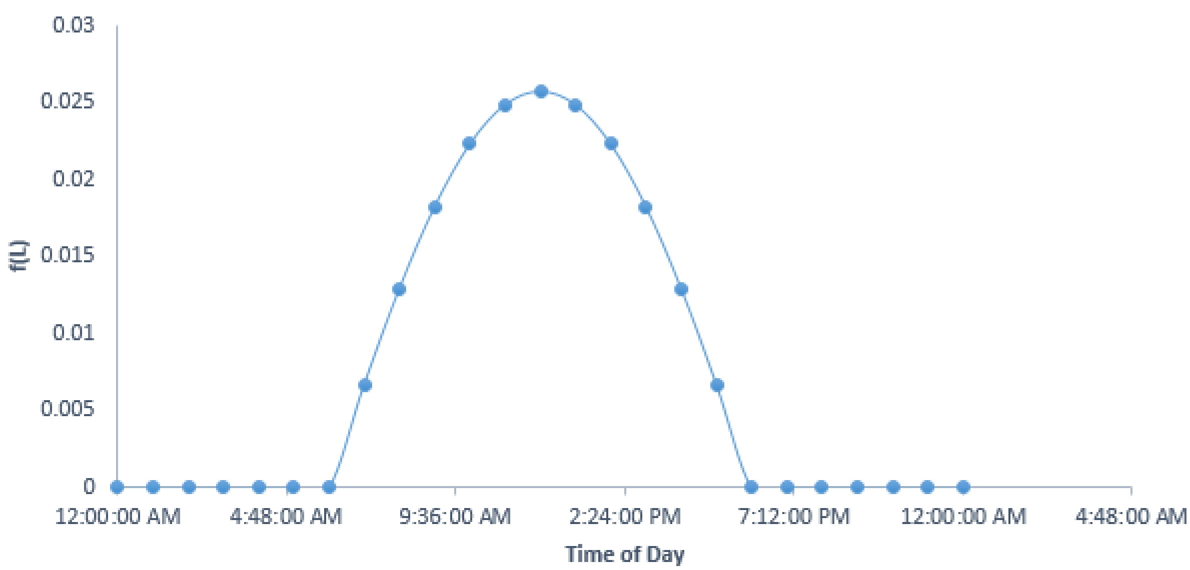

Figure 6.3 Growth limitation as a function of light intensity

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Microalgae biomass growth is directly proportional to incident light, which is a function of depth (d) of algae below the water surface, average daylight, light extinction due to algae biomass (K_{e1} and K_{e2}), light intensity at the water surface (I_a), and optimum light intensity (I_o) at a particular time using the Beer–Lambert's law as given in Table 6.2. As the light intensity increases microalgae grow to some saturation (optimum) intensity (I_s). The multiplicative light function *f(L)* given in Table 6.2 is illustrated in Figure 6.3. Microalgae growth start to decline beyond the optimum light intensity (James et al 2013, Belo et al 2017).

Total Inorganic Carbon (TIC) plays a very critical role in eutrophication processes and is present in numerous forms in water such as dissolved CO_2, carbonic acid (H_2CO_3), bicarbonates (HCO_3^-), and as carbonates (CO_3^{2-}) (James et al 2013). The relative amount of each carbon species present in the media is closely related to the pH of the media. When pH values are less than 6.5, the most dominant form of inorganic carbon species in the medium is free dissolved CO_2, whereas, at greater pH values above 10, inorganic carbon typically exists as carbonates. Between 6.5 and 10 pH values, bicarbonates are the predominant source of inorganic carbon. During photosynthesis, all microalgae species use free dissolved CO_2 although many other algal species prefer to use bicarbonates and some species can use carbonates and can grow in high-pH environments (e.g., *Scenedesmus quadricauda*) (James et al 2013, Belo et al 2017).

The simulation of pH limitation effect takes into account the estimate of maximum and minimum values of pH that support microalgae growth (James et al 2013). Thus, if the pH of the medium is increased, then the growth of algae may be inhibited due to the lack of dissolved CO_2. In most cases, microalgae species can maintain the growth of up to 8.6−8.85 pH values (James et al 2013, Belo et al 2017). However, with the availability of CO_2 from bicarbonates or carbonates, some algae species can grow up to pH values of 9.2−9.3. In other instances, most algae species do

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not grow well below pH values of 4.5−5.1, even though some species (e.g., *Euglena gracilis*) can grow in pH values as low as 3.9 (James et al. 2013). This HRAP model is designed to handle pH as input data to calculate *f*(pH) based on CO_2 concentrations. The variation of pH with temperature is ignored, since the change in effect is only by 0.1 unit per 20°C change in temperature (James et al 2013, Belo et al 2017).

During the interfacial exchange between air-water (for instance if CO_2 from AD biogas is bubbled through the growth medium), the exchange between H_2O and CO_2 results in the formation of carbonic acid (H_2CO_3) that dissociates into two protons (H^+) and carbonate (CO_3^{2-}) (James et al. 2013). Hence, with everything holding constant, higher CO_2 concentration in media should result in a decrease in the media pH (James et al 2013, Belo et al 2017).

Total inorganic carbon concentration (S_{TIC}) consists of dissolved carbon dioxide concentration ($C_{CO_{2aq}}$), carbonate concentration ($S_{CO_3^{-2}}$) and bicarbonate concentration ($C_{HCO_3^-}$) species which are generated in the system

$$S_{TIC} = S_{CO_{2aq}} + S_{CO_3^{-2}} + S_{HCO_3^-} \tag{6.4}$$

The principles of solution equilibrium and charge neutrality were applied to model the ionic equilibrium (pH estimation) (James et al. 2013);

When a gaseous CO_2 is introduced into H_2O, it becomes aqueous CO_2 ($CO_{2(aq)}$), which then reacts with H_2O to form H_2CO_3:

$$CO_{2(g)} + H_2O \leftrightarrow CO_{2(aq)} + H_2O \tag{6.5}$$

$$CO_{2(aq)} + H_2O \leftrightarrow H_2CO_3 \tag{6.6}$$

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The equilibrium reaction above shows that only a small fraction of $CO_{2(aq)}$ is converted into H_2CO_3, which can be presented by the hydration constant (at 25°C) as;

$$k_h = \frac{H_2CO_3}{CO_{2(aq)}} = 1.7 \times 10^{-3} \tag{6.7}$$

H_2CO_3 is a diprotic acid that can dissociate into two protons in a two-stage process:

$$H_2CO_3 \overset{k_1}{\leftrightarrow} H^+ + HCO_3^- \tag{6.8}$$

$$HCO_3^- \overset{k_2}{\leftrightarrow} H^+ + CO_3^{2-} \tag{6.9}$$

The acidity (or dissociation) constants for the two stages are given as:

$$k_1 = \frac{[H^+][HCO_3^-]}{[H_2CO_3]} \tag{6.10}$$

$$k_2 = \frac{[H^+][CO_3^{2-}]}{[HCO_3^-]} \tag{6.11}$$

Based on Eq. (6.4), (6.10), and (6.11), the concentration of carbonate ions as a function of pH are express as follows:

$$HCO_3^- = \frac{(S_{TIC})}{(1 + \left(\frac{H^+}{K_1}\right) + \left(\frac{K_2}{H^+}\right)} \tag{6.12}$$

$$CO_3^{2-} = \frac{(S_{TIC})}{(1 + \left(\frac{H^+}{K_2}\right) + \left(\frac{H^{+2}}{K_1K_2}\right)} \tag{6.13}$$

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According to James et al. 2013, it can be assumed that carbonic acid is a weak monoprotic acid, CO_3^{2-} formed during the second dissociation of HCO_3^- (Equ. 6.11) can be ignored. Therefore, the following expression can be derived (James et al. 2013):

$$k_1 = \frac{[H^+]([H^+] - [OH^-])}{k_h CO_{2\_aq} - [H^+] + [OH^-]} \tag{6.14}$$

Using the hydration constant for water, $k_w = [H^+][OH^-] = 1.008\times10^{-14}$ at a standard temperature of 25°C, and using $[OH^-] = k_w/[H^+]$, the simplified expression for k_1 in terms of $[H^+]$ is given as:

$$[H^+]^3 + k_1[H^+]^2 - (k_1 k_h[CO_2] + k_w)[H^+] - k_1 k_w = 0 \tag{6.15}$$

$k_1 k_w$ can be neglected due to its smaller value (~ $1.0(10^{-21})$); therefore Eq. 6.15 can be reduced to a quadratic equation

$$[H^+]^2 + k_1[H^+] - \left(k_1 k_h\left[CO_{2\_aq}\right] + k_1\right) = 0 \tag{6.16}$$

The quadratic equation above can then be solved numerically and approximated into a simple expression for H+ as a function of CO_{2aq} and is express as:

$$pH = -\frac{1}{2}\, log_{10}\left(k_w + k_1 k_h\left[CO_{2_{aq}}\right]\right) \tag{6.17}$$

An iteration method is used to calculate $CO_{2(aq)}$ which is then used to find pH using Eq. 6.17. With known pH, the multiplicative function f(pH) in Table 6.2 was modeled. The f(pH) multiplicative function is illustrated in Figure 6.4. It is worth noting that H+ was calculated from

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Eq. 6.17 and used in the f(pH) function calculation. The f(pH) functions show that microalgae growth is maximum in neutral to slightly alkaline water.

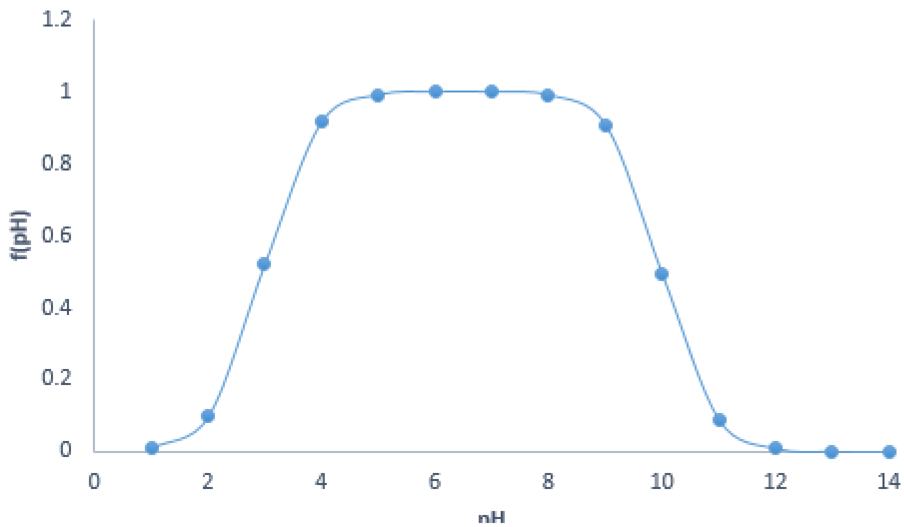

Figure 6.4 pH function growth limitation at 21ºC.

The HRAP system receives nitrogen through the influent wastewaters containing ammonia (NH_4), organic nitrogen (N_o), and nitrate (NO_3^-). Even though nitrogen fixation from the atmosphere can be realized by some algal species; yet, this process is outside the scope of this modeled. Organic nitrogen from both algae and bacteria biomass, in the form of proteins, is disintegrated by hydrolysis into amino acids that ends up in the form of ammonia through decomposition by bacteria. First of all, the soluble part of ammonia combines with hydrogen ion (H^+) to form ammonium ions as follows;

$$NH_3 + H^+ \leftrightarrow NH_4^+ \quad (6.18)$$

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When ammonia combines with hydrogen ions the pH increases. Oxidation occur through the activities of autotrophic Nitrosomonas and Nitrobacter bacteria to consecutively produce nitrite and nitrate.

$$NH_4^+ + O_2 \rightarrow H_2O + NO_3^- + H^+ \tag{6.19}$$

The two reactions (Eq. 18 and 19) above require 4.57 mg of oxygen for every milligram of ammonia nitrified as N. Nitrite concentration is always low because the formation of nitrate is faster than the formation of nitrite due to the fact that Nitrobacter needs roughly three times as much substrate as Nitrosomonas to obtain the same amount of energy and nitrification is usually rate-limited by the activity of Nitrosomonas. Hence, nitrite is ignored in this modeled (Fritz 1979). Nitrification in this model is described as a single-step process with multiplicative functional links to temperature, O_2, and pH. By using the Monod model, the autotrophic Nitrosomonas growth rate coefficient, is calculated as shown in Table I.

6.3.2 Development of AD Model

An AD process normally consists of a reactor with liquid volume and a gas headspace at atmospheric pressure with gas removed for downstream utilization (Batstone et al. 2002). The AD model is a CSTR with a single input and out-stream, and a constant volume. The AD model follows the structured ADM1 model proposed by Batstone et al. 2002 (ADM1). ADM1 was simplified (shown in Table 6.3.) in other to simulate four different feedstock for co-digestion with less difficulty. The four different feedstock considered for co-digestion is primary sludge, biomass (algae and bacteria), and fat-oil-grease (FOG). ADM1 by Batstone has 24 rate components with 12 soluble and 12 particulate parameters. Whereas, this model constitutes 10 components with

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three particulate and seven soluble parameters. The lower amount of rate components in this model is due to the fact that, this model was design to be simple in order to prevent any possible solver stiffness. The methanogens specific growth rate for this model uses a Haldane function in order to incorporate volatile fatty acid (VFA) inhibition associated to ammonia inhibition. Also, in this model a non-competitive inhibition function for long chain fatty acid (LCFA) was added to take into account for the inhibition of methanogenic steps by high total VFA concentration, especially when FOG is modeled. Finally, unlike ADM1, this model does not include process kinetics related to hydrogenotrophic methanogens.

The model presented is schematically illustrated in Figure 6.2. The model involves a single enzymatic process (that is the hydrolysis of undissolved organic matter) and four bacteria groups Figure 6.2. Overall five distinct processes are considered for the model: (i) hydrolysis of biopolymers (proteins, carbohydrates, and lipids); (ii) amino acids and sugars fermentation; (iii) anaerobic oxidation of LCFA; (iv) anaerobic oxidation of intermediary product (such as VFAs) with exception of acetate); and (v) conversion of acetate to methane. The model excludes the conversion of hydrogen to methane.

6.3.2.1 Biological reactions pathway

Enzymes discharged by acid-forming bacteria convert the complex particulate organic matter (proteins, lipids, and carbohydrates) into soluble organics (represented by glucose $C_6H_{12}O_6$) according to (Husain, 1998):

$$C_6H_{13}NO_5 + H_2O + H^+ \rightarrow C_6H_{12}O_6 + NH_4^+ \quad (6.20)$$

Anaerobic oxidation of acidogens can be express as;

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$$C_6H_{12}O_6 + 1.2NH_3 \rightarrow 1.2C_5H_7O_2N + 3.6H_2O \quad (6.21)$$

Once the organic matter is solubilized as $C_6H_{12}O_6$, acetogens degraders convert the waste into VFAs (namely; acetate, propionate, and butyrate) as given in the equation below;

$$\begin{aligned} C_6H_{12}O_6 \rightarrow\ & 0.1115C_6H_7NO_2 + 0.744CH_3COOH + 0.5CH_3CH_2CH_2OOH \\ & + 0.5CH_3CH_2COOH + 0.454CO_2 \end{aligned} \quad (6.22)$$

It is important to note that all the small fraction of the soluble organics are consumed to maintain the bacteria population represented by $C_6H_7NO_2$. Also, the nitrogen requirement for bacteria cell synthesis is obtained from the release of ammonium (NH_4^+) in Eq. 20.

Conversion of acetate to methane is achieved by;

$$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^- \quad (6.23)$$

Overall, the conversion of organic matter to CH4 involves a close relationship among four types of bacterial populations with the dynamic balance between production and utilization of the intermediate products being critical to the overall success of the fermentation (Batstone et al 2003). Disturbance of the dynamic balance would cause an accumulation of VFAs and eventually lead to digester failure.

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Table 6.3 Petersen's matrix (Petersen (1965) for AD model - Biochemical rate coefficients ($v_{I,j}$) and kinetic rate equations(ρ_j)

	Component (i) →	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**Process Rate (pi)**
j	Process (i) ↓	X_m	X_{LCFA}	X_{AA}	X_P	S_{hp}	S_{LCFA}	S_{AA}	$S_{Acetate}$	S_P	S_{TIC}	S_N	(g/L.day)
1	**Methanogenic Growth**	1							$-k_5$		k_9	$-k_9$	$\left(\frac{U_{m\_max} \times Acetate}{k_{vfa} + Acetate + \frac{Acetate^2}{Kim}}\right) \times I_{pH} \times I_N \times I_{NH_3} \times I_{VFA}$
2	Uptake of LCFA		1						$-k_5.q_5$		k_8	k_8	$\left(\frac{U_{LCFA\_max} \times LCFA}{k_P + LCFA}\right) \times I_{pH} \times I_N$
3	**Uptake of Amino Acid**			1				$-k_1$	$k_3.q_4$	$q_3.k_1$	$-k_7$	$-k_7$	$\left(\frac{U_{AA\_max} \times AA}{k_{s1} + AA}\right) \times I_{pH} \times I_N$
4	Uptake of Propionate				1		$-k_2$		$k_6.q_6$	$-k_6$			$\left(\frac{U_{P\_max} \times P}{k_{Prop} + P}\right) \times I_{pH} \times I_N$
5	**Hydrolysis**					-1	q_2	q_1					$k_{hp}.S_{hp}$
6	Decay of X_m	-1											$k_{d_m}.X_m$
7	**Decay of X_{LCFA}**		-1										$k_{d_{LCFA}}.X_{LCFA}$
8	Decay of X_{AA}			-1									$k_{d_{AA}}.X_{AA}$
9	**Decay of X_P**				-1								$k_{d_P}.X_P$
Observed conversion Rates (g/L.day)		$r_i = \sum_j v_{ij}\rho_j$											
		Acetoclastic Methanogens - (g/L)	**LCFA Degraders - (g/L)**	Amino Acid Degraders - (g/L)	Propionate Degraders - (g/L)	**Hydrolyzable Particulate - (g/L)**	Long Chain Fatty Acid - (g/L)	Amino Acid - (g/L)	Acetate - (g/L)	Propionate - (g/L)	Total Inorganic Carbon - (g/L)		

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Table 6.4 Inhibition Expressions

Description	Equation	Used for	Ref.
Non-competitive inhibition	$I_N = \frac{k_{in}}{(k_{in}+N)}$	Inorganic Nitrogen Limitation	Batstone et al. 2002
Substrate limitation	$I_{NH_3} = \frac{k_{i\_NH_3}}{k_{i\_NH_3}+NH_3}$	Ammonia Limitation	Batstone et al. 2002
Substrate limitation	$I_{VFA} = \frac{k_{im}}{k_{im}+LCFA}$	Volatile Fatty Acid Limitation	Batstone et al. 2002
Empirical	$I_{pH} = \frac{1+2\times 10^{\left(\frac{1}{2}\times(pH_{LL}-pH_{UL})\right)}}{1+10^{(pH-pH_{UL})}+10^{(pH_{LL}-pH)}}$	pH inhibition when only	Batstone et al. 2002

pH_{LL} and pH_{UL} for the pH (Empirical) function are the upper and lower limits where the groups of microorganisms are 50% inhibited.

6.3.2.2 Physio-chemical process

The pH value was computed by assuming all acid-base pairs are in equilibrium. It was assumed that carbonate concentration can be ignored when a pH range of operation is less than 8, that makes the C_{TIC} equal to the sum of $CO_{2\text{-aq}}$ and HCO_3^- (Mairet et al. 2011). HCO_3^- can be expressed in terms of the dissociation constant:

$$HCO_3^- = \frac{K_c}{H^+ + K_c} \times C_{TIC} \tag{6.24}$$

Dissociation of total inorganic nitrogen (N_T) follows the same suit as Equ. 22.

$$NH_4^+ = \frac{H^+}{H^+ + K_N} \times N_T \tag{6.25}$$

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Hence, hydrogen ion concentration $[H^+]$ can be calculated as:

$$[H^+] = \frac{K_{pH}[CO_{2-aq}]}{HCO_3^-} \tag{6.26}$$

$$pH = -log_{10}[H^+] \tag{6.27}$$

here K_C (M) is the dissociation constant for the coupled $HCO_3^- / CO_{2\text{-aq}}$ (Batstone et al. 2002), K_N (M) is the dissociation constant for the coupled NH_3/NH_4^+, and K_{pH} (M) is the first dissociation constant for carbonic acid system.

The effect of pH on bacteria growth was described by a Michaelis pH function (represented as an empirical formula in Table 6.4), the function is normalized to give a value of 1.0 as center value (Batstone et al. 2002). The form of the pH inhibition function given in Table 6.4 is shown in Figure 6.5.

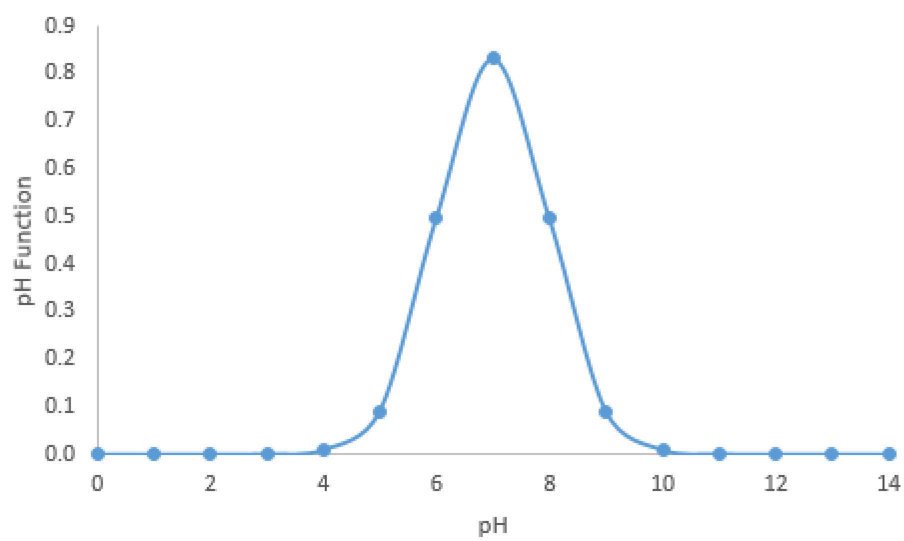

Figure 6.5 Normalized Michaelis pH function used in the AD model

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6.3.2.3 Liquid step equations

The mass balance equations used in this work to describe the dynamic behavior of soluble substrates and particulate substrates components in the liquid step are shown below:

$$\frac{dS_{liq-i}}{dt} = \frac{Q}{V_{liq}}.\left(S_{in-i} - S_{liq-i}\right) + \sum_{j=1-9} v_{ij}\rho_j \quad i = 5 - 11; \tag{6.28}$$

$$\frac{dX_{liq-i}}{dt} = \frac{Q}{V_{liq}}.\left(X_{in-i} - X_{liq-i}\right) + \sum_{j=1-9} v_{ij}\rho_j \quad i = 1 - 4; \tag{6.29}$$

where $S_{\text{liq-i}}$ represent each soluble state variable concentration, $X_{liq\text{-}i}$ is the concentration of each biomass state variable, V_{liq} is the volume of liquid in reactor, Q is the flow, $S_{\text{in-i}}$ is the input concentration of soluble components, $X_{\text{in-i}}$ is also the input concentration of biomass components and the term $\sum_j v_{ij}\rho_j$ is the sum of the specific kinetic reaction rates ρ_j for process j multiplied by the stoichiometric coefficients v_{ij} presented in Tables 6.3 and 6.4.

6.3.2.4 Gas-Step equations

Methane and carbon dioxide are the only two gases modeled. The rate transfer of carbon dioxide and methane into the gas step was determined from the general theory of two-film mass transfer (Whitman, 1923). All gases in the model were assumed to obey the ideal gas law and occur at a temperature comparable to the liquid step temperature in a constant volume (CSTR) and a constant pressure headspace (Batstone et al., 2002).

The liquid-gas mass transfer rate of CO_2 expressed in (mol/L.day) is given as:

$$\rho_{CO_2} = k_{la}\left(CO_2 - H_{CO_2}P_{CO_2}\right) \tag{6.30}$$

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where H_{CO_2} is the Henry's constant, P_{CO_2} is the partial pressure of CO_2 in the reactor headspace, k_{la} is the liquid-gas mass transfer coefficient. Because CH_4 has very low solubility, it is assumed that all the CH_4 produced is transferred to the reactor headspace; that is given as:

$$\rho_{CH_4} = k_{13} U_m X_m \tag{6.31}$$

Gas flow is then simulated assuming the headspace is over pressured:

$$q_{gas} = \max\left(0, k_p\left(P_{CH_4} + P_{CO_2} - P_{atm}\right)\right) \tag{6.32}$$

where kP is the pipe resistance coefficient (Batstone et al., 2002). The dynamics of the partial pressures are express as:

$$\frac{dP_{CO_2}}{dt} = -P_{CO_2}\frac{q_{gas}}{V_{gas}} + \rho_{CO_2}\frac{V_{liq}RT_{op}}{V_{gas}} \tag{6.33}$$

$$\frac{dP_{CO_2}}{dt} = -P_{CH_4}\frac{q_{gas}}{V_{gas}} + \rho_{CH_4}\frac{V_{liq}RT_{op}}{V_{gas}} \tag{6.34}$$

where TOP is the digester temperature and V_{liq} and V_{gas} are the volume of liquid and gas steps. Finally, the CH_4 content of the gas flow is given as:

$$\%CH_4 = \frac{P_{CH_4}}{P_{CH_4} + P_{CO_2}} \tag{6.35}$$

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6.3.3 Energy Assessment

The analysis includes: (a) electricity consumption for the HRAPs paddle-wheel; (b) electricity requirement for sludge pumping; and (c) electricity and heat for the anaerobic digester. The analysis assumes energy input for wastewater pretreatment, primary and secondary sedimentation to be negligible (Metcalf and Eddy, 2013). The theoretical energy balance analysis was derived using equations presented in Table 6.5 below:

Table 6.5 Equations used for energy calculations

Energy Source	Equations	Parameters	Ref.
HRAP mixing electricity requirement (kWh/day)	$G^2.\mu.V.24$	μ = 0.001 N.s/m^2; G = 50 s^{-1}	Metcalf and Eddy, 2013
Sludge pumping (kWh/day)	$Q_{Sludge} \times \varepsilon \times 0.0002778$	ε = 1800 kJ/m^3	Maria et al. 2018
Digester heat requirement (kWh/day)	$Q_{ld} \times UA\Delta T \times 2.778 \times 10^{-7}$	UA = 4200 J/kg.°C	Metcalf and Eddy, 2013
Digester electricity for mixing (kWh/day)	$Q_{Sludge} \times \emptyset \times 0.0002778$	$\emptyset$ = 300 kJ/m^3	Maria et al. 2018

Here μ is the assumed dynamic viscosity of the wastewater, G is the velocity gradient of the mixing paddle, ε is the electrical consumption for pumping, UA specific heat of sludge, Q_{ld} digester capacity (or sludge loading), ΔT temperature drop across reactor surface, Q_{Sludge} is the sludge flow rate and $\emptyset$ is the electrical consumption for sludge mixing. Some of the assumptions made for the energy analysis were; heat loss was assumed to be 49% of heat required, electricity production was based on CHP internal combustion engine with electrical conversion efficiency of 45%; and power to heat ratio of 0.5. Finally, net energy ratio was calculated as energy produced over energy consumed.

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6.3.4 Computation of influent and effluent COD concentration of HRAP system

The influent soluble COD fractions used in the mass balance were computed as follow (Fritz et al. 1979):

$$X_s = COD_{raw} \times [1 - Fr_{COD} + F_n] \tag{6.36}$$

$$F_n = \begin{cases} -e^{-0.16\times T} & if\ T < 15^oC \\ (T-15)^2/_{100} & if\ T > 15^oC \end{cases} \tag{6.37}$$

where Fr_{COD} is refractory (or inert) COD fraction; and *Fn* represent COD fraction in temporary storage (when T<15ºC) or release from temporary storage (when T>15ºC). (Fritz et al 1979). So, what the equation says is, the available soluble COD for oxidation depends on temperature that is Fn. In other words, when the temperature is less than 15ºC, *Fn* represent a portion or fraction of the raw COD that goes into temporary storage by sedimentation or goes through the pond without being oxidized. But, if the temperature is greater than 15ºC, *Fn* will be the portion of COD that is released from temporary storage. This increases the available COD to be oxidized. This means because COD solubility is temperature dependent, a fraction of the COD will settle and some of it will reach the effluent before it is oxidized. This equation was adopted from the work of Fritz et al. 1979.

On the other hand, total effluent COD was computed as follows (Fritz et al. 1979):

$$COD_{Total} = X_S + X_A + X_X + COD_{raw}[Fr_{COD} + F_t] \tag{6.38}$$

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$$F_t = \begin{cases} -e^{-0.16\times T} - e^{-0.37\times T} & if\ T \leq 15^oC \\ 0 & if\ T > 15^oC \end{cases} \tag{6.39}$$

where X_S, X_A, X_X are the effluent soluble COD, microalgae and bacteria respectively; and Ft is fraction of COD that flows through the pond without being oxidized (Fritz et al. 1979). Effluent total COD was computed from the mass-balance relationship presented above. Total effluent COD is the sum of soluble effluent, microalgae biomass, bacteria biomass, inert COD and any other COD that flows through the pond without being oxidized.

6.3.5 Model Validation

Both models were calibrated and validated against experimental data from literature and estimated values obtained from model calibration are provided in Table 6.7 and Table 6.8. Before the model was calibrated and validated a sensitivity analysis was performed. The sensitivity analysis determines uncertainty in the model; and helps understand how parameters and state variables influence the simulation against the measured data. A global sensitivity analysis using Monte Carlo technique was adopted. This approach uses a representative (global) set of samples (normal distribution) to explore the design space. After the sensitivity analysis, the parameters were ranked based on correlation, partial correlation and standardized regression. Parameters with higher correlation are selected for parameter estimation (new estimated parameters are provided in Table 6.7 and Table 6.8.). Hence, the new estimated parameters are then used to calibrate and validate the model. Results for sensitivity analysis are provided in Appendix B.

6.3.5.1 HRAP Model

Experimental and simulation data of Bai 2015 and Bello et al. 2017 were used for the HRAP model. Bai work focused on the impact of bacteria on microalgae cultivation in an open

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algal system by focusing on carbon limitation in open microalgae cultivation and the difficulties it presents for downstream processing. Bai also proposed a model to simulate the experimental work.

The HRAP model in this present study was compared to both Bai's experimental and simulated data (Bai 2015). Figure 6.7 shows a comparison between the current model and experimental data reported by Bai. It can be seen that microalgae biomass concentration of the HRAP model is comparable to both experimental (Adj. R^2 = 0.99; MAE= 8.5%) and simulation results (Adj. R^2 = 0.97; MAE = 7.9%). It is also worth noting that the simulated trend for Bai shows a "wavelike" profile as an indication to confirm that the HRAP model can reproduce microalgae growth and at the same time depict the inactivation cycles occurring during day and night times, respectively. In addition, the model was validated against the work of Bello et al 2017 (Figure 6.8). Bello developed a comprehensive mathematical model to simulate the production of microalgae in an HRAP. Similar to the HRAP model in this work, the Bello model established a synergetic relationship between the bacteria-algal system involving several interrelated biological and chemical systems. The HRAP model is comparable to that of Bello, but the correlation (Adj. R^2 = 0.95; Mean Absolute Error = 14%) is less than that of Bai. The statistical parameters show a strong correlation between the HRAP model and literature.

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Figure 6.6 Model validation plots, comparing the HRAP simulation with both experimental and simulation work of Bai (B) shows a comparison between the HRAP simulation and Bello et all 2017 simulation.

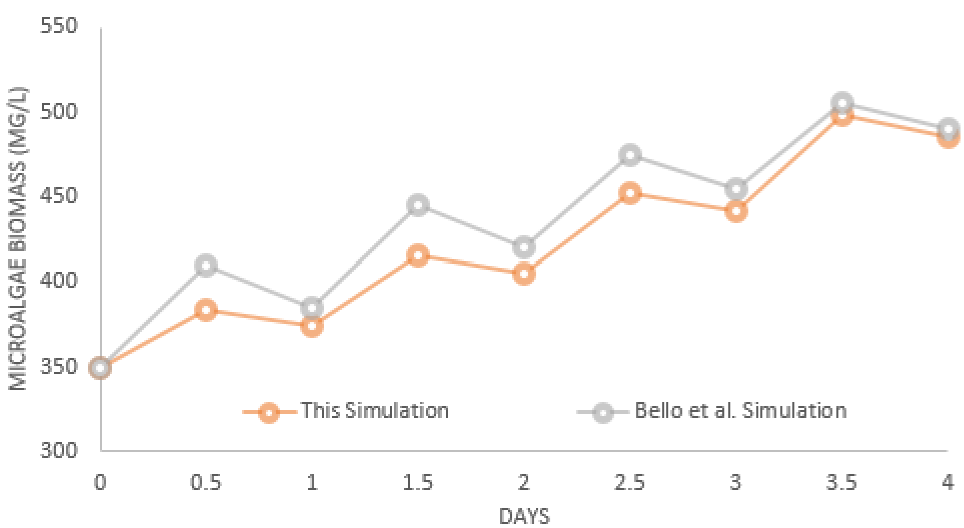

Figure 6.7 Model validation plots, comparing the HRAP simulation with simulation work of Bello et al. 2017.

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6.3.5.2 AD Model

The AD model consists of four feedstock (namely primary sludge, bacterial biomass, microalgae biomass, and FOG). Thus, obtaining literature data for works done using multiple (four) substrate for co-digestion has not been done (especial co-digestion of primary sludge, microalgae, and FOG). Because of the challenge in obtaining a similar work in literature for the model validation; two separate experimental studies from the literature were used to validate the AD model. Mahdy et al 2015 studied the comparison of anaerobic digestion of primary sludge, secondary sludge, and microalgae. Mahdy evaluated the effect of thermal pretreatment on methane yield and concluded that microalgae biomass is a potential co-substrate for biogas generation. The AD experimental test was conducted in batch mode with a reactor liquid volume of 0.07 L and maintained a sludge mixture COD/VS ratio of 0.5 g/g. AD temperature was kept at 35 ^{o}C. Mahdy also mentioned that raw algae biomass and bacterial biomass had a biodegradability of 33% and 23% respectively. Whereas, primary sludge has the highest biodegradability of 97%, making it suitable for higher biogas production when co-digested with other substrates such as algae and bacteria biomass. To validate the AD model, the simulation output was compared to the experimental data of Mahdy et al. 2015. Figure 6.8 and Table 6.7 shows the simulation profile correlation with experimental data. All simulation shows a strong agreement with Mahdy's experimental data with <1.5% MAE. Figure 6.9 shows validation graph of multiple (two substrate) substrate.

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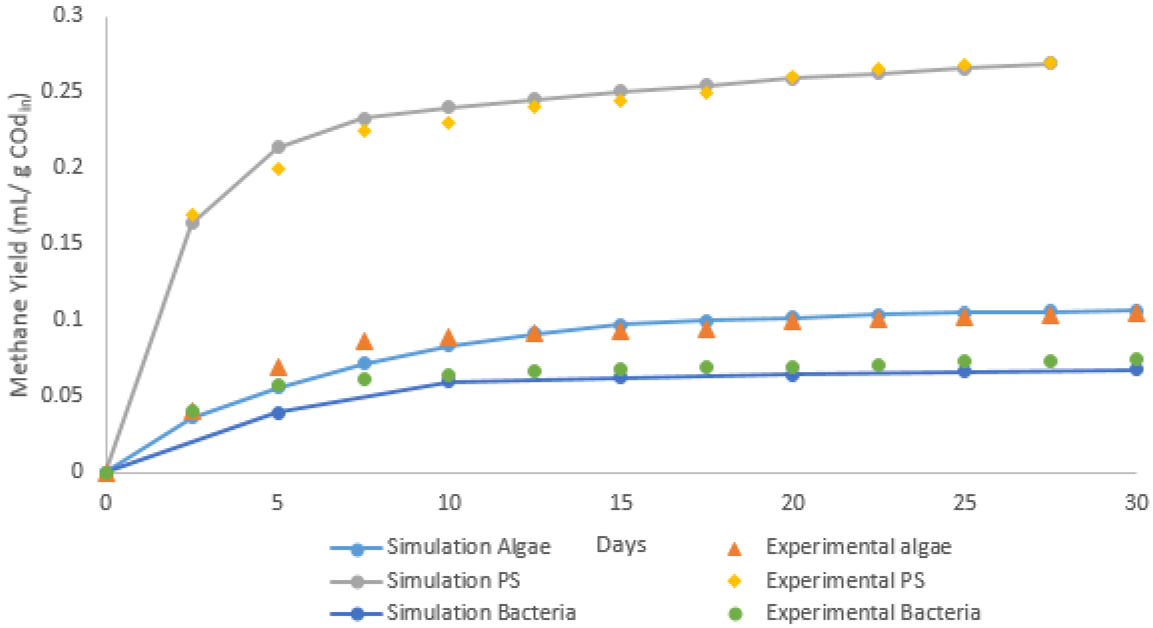

Figure 6.8 AD model validation profiles –mono or single substrate anaerobic digestion simulation compared to experimental data by Mahdy et al. 2015;

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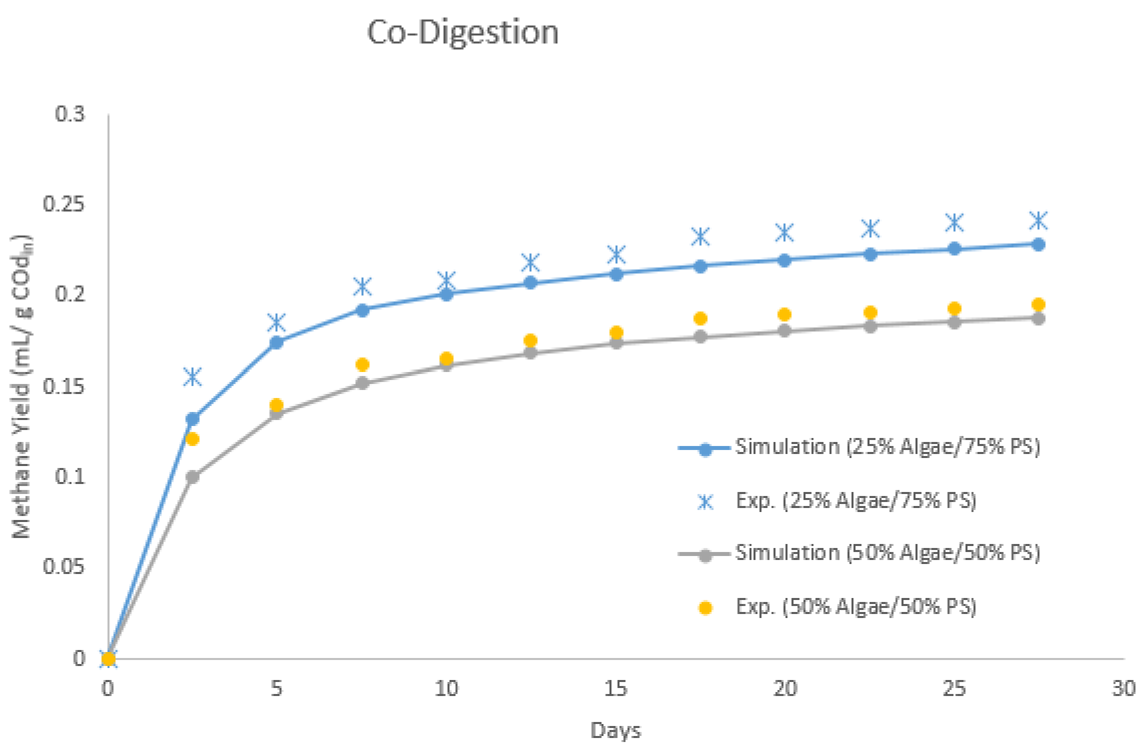

Figure 6.9 AD model validation profiles - represent multiple substrate (or substrate mixture) co-digestion simulation compared to experimental data by Mahdy et al. 2015.

Table 6.6 Statistical comparison of AD simulation against experimental data from the literature

Test Condition (Sim vs Exp)	Adjusted R^2	Mean Absolute Error (MAE)	Experimental Work [Ref.]
Mono Substrate Digestion			
Primary Sludge (PS) Only	0.99	0.005	Mahdy et al 2015
Microalgae Biomass Only	0.96	0.004	Mahdy et al 2015
Bacteria Biomass Only	0.99	0.0005	Mahdy et al 2015
FOG only	0.97	0.028	Davidsson et al. 2008
Co-Digestion			
25% Algae + 75% PS	0.99	0.012	Mahdy et al 2015
50% Algae + 50% PS	0.99	0.008	Mahdy et al 2015

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The experimental work of Davidsson et al. 2008 was used to validate the FOG AD simulation. Davidsson used sludge from trap grease and sewage sludge for co-digestion and experimented with batch tests and continuous pilot-scale digestion tests. The pilot-scale digesters were kept at a mesophilic temperature (35ºC) with an HRT 10-13 days. As shown in Figure 6.10 below and Table 6 above, FOG simulation agrees well with experimental data from Davidsson et al. 2008 with an MAE of <3%.

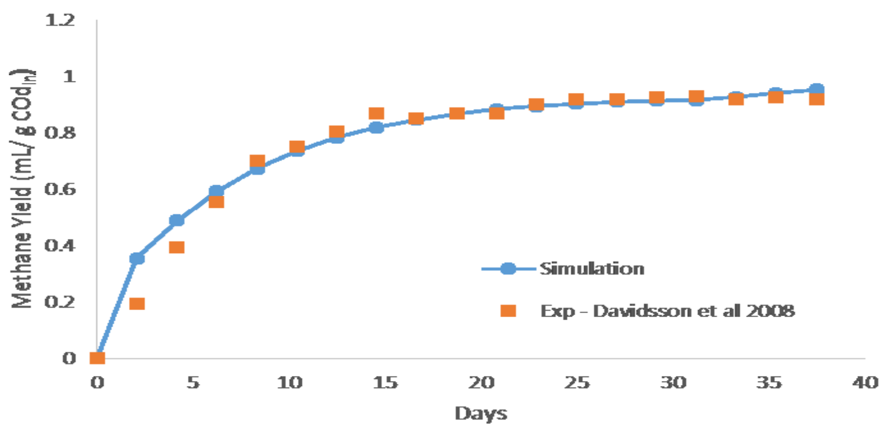

Figure 6.10 AD validation profile for FOG (methane yield vs. digestion time)

The simulation results show that each substrate has its own methane potential. As shown in Figure 6.10, the primary sludge obtained a steady-state methane potential of 0.25 L/g COD added which shows that primary sludge is approximately three times more biodegradable than microalgae and bacteria biomass. This is because, primary sludge are just colloidal organics readily available to be transformed into methane by anaerobes and also because the organic matter has no cell wall, hence no penetration is required during digestion. Methane yield for microalgae and

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bacteria biomass are less due to the thickness of it cell wall which may obstruct anaerobic bacteria attack (Mahdy et al 2015. Digestion of FOG presents an added advantage for co-digestion, as shown in Figure 6.10. The biomethane potential for FOG averages 0.8-0.9 L/ g COD added. The addition of FOG for co-digestion presents a promising scenario for energy self-sufficient wastewater treatment.

Table 6.7 Design parameters adopted in the HRAP model

Item	Parameter Description	Symbols	Numerical Values	Unit
HRAP				
	Hydraulic Retention Time	HRT	7	day
	Pond depth	d	0.4	m
	Temperature	T	Summer/Winter	ºC
	Number of CSTR	n	20	
	Photo Period		5 am – 6 pm	
Influent Wastewater				
	Flow	Q	60 - 120	gal/capita.d
	Substrate Concentration	COD	117	g/capita.d
	Total Inorganic Carbon	C_T	41	g/capita.d
	Organic Nitrogen	N_O	3.4	g/capita.d
	Ammonia	N_{NH3}	5.6	g/capita.d

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Table 6.8 Values of simulation parameters adopted in the HRAP model

Parameter Description	Symbol	Value	Unit	References
Maximum specific bacteria growth rate	$U_{X\ max}$	5	day^{-1}	Yang 2011
Maximum specific algae growth rate	$U_{A\ max}$	0.44	day^{-1}	Yang 2011
Maximum Substrate Utilization Rate @ 20°C	SUKC20	20	day^{-1}	Fritz et al 1979
Half Saturation constant for carbon	K_C	1	$mg.L^{-1}$	Fritz et al 1979
Half Saturation constant for oxygen	K_{O2}	1	$mg.L^{-1}$	Fritz et al 1979
Half Saturation constant for substrate	K_S	50	$mg.L^{-1}$	Fritz et al 1979
Bacteria Half Saturation constant for nitrogen	K_{XN}	0.01	$mg.L^{-1}$	Assumed
Bacteria Half Saturation constant for Phosphorus	K_{XP}	0.01	$mg.L^{-1}$	Assumed
Algae Half Saturation constant for Phosphorus	K_{AP}	0.02	$mg.L^{-1}$	Fritz et al 1979
Algae Half Saturation constant for nitrogen	K_{AN}	0.0995	$mg.L^{-1}$	Estimated
Arrhenius Constant	AC	1.07		Fritz et al 1979
Bacteria Decay coefficient at 20°C	BDC20	0.007	day^{-1}	Fritz et al 1979
Yield coefficient	Y_H	0.5	$mg.mg^{-1}$	Bello et al. 2017
Mass Transfer Coefficient of Oxygen	$K_{la\text{-}O2}$	24.95	day^{-1}	Bello et al. 2017
Mass Transfer Coefficient of Carbon dioxide	$K_{la\text{-}CO2}$	6.05	day^{-1}	Bello et al. 2017
Henry's Constant for oxygen	H_{O2}	0.044	$mg(L.atm)^{-1}$	Yang 2011
Henry's Constant for carbon dioxide	H_{CO2}	0.903	$mg(L.atm)^{-1}$	Yang 2011
Partial Pressure for oxygen	P_{O2}	0.21	atm	Yang 2011
Partial Pressure for carbon dioxide	P_{CO2}	0.00032	atm	Yang 2011
Oxygen-Nitrosom half saturation constant	K_{NO2}	1.3	$mg.L^{-1}$	Fritz et al 1979
Nitrosom growth rate	U_N	0.008	day^{-1}	Fritz et al 1979
Nitrosom yield coefficient	Y_N	0.15	$mg.mg^{-1}$	Fritz et al 1979
Algae decay constant	k_{dA}	0.05	day^{-1}	Estimated
Extinction Coefficient	K_{e1}	0.32	m^{-1}	James et al. 2013
Extinction Coefficient	K_{e2}	0.03	$m^{-1}.(mg/L)^{-1}$	James et al. 2013
Saturation Light Intensity	Is	14.342	$MJ.(m^2.day)^{-1}$	Yang 2011
Maximum Light Intensity	Io	77.225	$MJ.(m^2.day)^{-1}$	Yang 2011
Lower optimal growth temperature	T_1	17	°C	James et al. 2013
Upper optimal growth temperature	T_2	32	°C	James et al. 2013
Temperature effect coefficient	K_1^T	0.69	$°C^{-2}$	James et al. 2013
Temperature effect coefficient	K_2^T	0.007	$°C^{-2}$	James et al. 2013

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Table 6.9 Assumed model parameters for AD model

Parameter Description	Symbol	Value	Unit	Ref
Yield for sugar-lipid degradation	K_1	12.841	$gCOD.(gCOD)^{-1}$	Estimated
Yield for sugar-lipid degradation (FOG)	K_1	12.5	$gCOD.(gCOD)^{-1}$	Batstone et al 2002
Yield for protein degradation	K_2	12.5	$gCOD.(gCOD)^{-1}$	Batstone et al 2002
Yield of VFA production: acidogenesis of sugar-lipid degraders (algae)	K_3	13.24	$gCOD.(gCOD)^{-1}$	Estimated
degraders	K_3	1.366	$gCOD.(gCOD)^{-1}$	Estimated
Yield of VFA production: acidogenesis of sugar-lipid degraders (PS)	K_3	9.903	$gCOD.(gCOD)^{-1}$	Estimated
Yield of VFA production: acidogenesis of sugar-lipid degraders (FOG)	K_3	11.67	$gCOD.(gCOD)^{-1}$	Estimated
Yield of VFA production: acidogenesis of protein	K_4	11.5	$gCOD.(gCOD)^{-1}$	Batstone et al 2002
Yield of VFA consumption (algae)	K_5	13.1	$gCOD.(gCOD)^{-1}$	Batstone et al 2002
Yield of VFA consumption (bacteria)	K_5	8.75 x 10-5	$gCOD.(gCOD)^{-1}$	Estimated
Yield of VFA consumption (PS)	K_5	10.996	$gCOD.(gCOD)^{-1}$	Estimated
Yield of VFA consumption (FOG)	K_5	10.191	$gCOD.(gCOD)^{-1}$	Estimated
Yield of bacteria growth on propionic	K_6	71.43	g/g	Bryers 1984
Yield for ammonium consumption	K_7	0.006	$mol. (gCOD)^{-1}$	Batstone et al 2002
Yield for ammonium production	K_8	0.083	$mol. (gCOD)^{-1}$	Batstone et al 2002
Yield for ammonium consumption methanogenesis	K_9	0.006	$mol. (gCOD)^{-1}$	Batstone et al 2002
Yield of CO2 production: acidogenesis of sugar-lipid	K_{10}	0.04	$mol. (gCOD)^{-1}$	Batstone et al 2002
Yield of CO2 production: acidogenesis of proteins	K_{11}	0.04	$mol. (gCOD)^{-1}$	Batstone et al 2002
Yield of CO2 production: methanogenesis	K_{12}	0.12	$mol. (gCOD)^{-1}$	Batstone et al 2002
Maximum specific growth rate for Amino Acid degraders (algae)	$U_{AA\ max}$	0.293	day^{-1}	Estimated
Maximum specific growth rate for Amino Acid degraders (bacteria)	$U_{AA\ max}$	0.788	day^{-1}	Estimated
Maximum specific growth rate for Amino Acid degraders (PS)	$U_{AA\ max}$	1.503	day^{-1}	Estimated
Maximum specific growth rate for Amino Acid degraders (FOG)	$U_{AA\ max}$	0.203	day^{-1}	Estimated
Half-saturation constant of sugar-lipid acidogenic bacteria	k_{sl}	0.29	$g.L^{-1}$	Batstone et al 2002
Maximum specific growth rate for LCFA	$U_{LCFA\ max}$	0.053	day^{-1}	Estimated
Maximum specific growth rate for LCFA (bacteria)	$U_{LCFA\ max}$	0.038	day^{-1}	Estimated
Maximum specific growth rate for LCFA (PS)	$U_{LCFA\ max}$	0.0001	day^{-1}	Estimated
Half-saturation constant of protein acidogenic bacteria	k_p	0.046	$g.L^{-1}$	Batstone et al 2002
Maximum specific growth rate for Methanogenic degraders (algae)	$U_{M\ max}$	0.355	day^{-1}	Estimated
Maximum specific growth rate for Methanogenic degraders (bacteria)	$U_{M\ max}$	0.45	day^{-1}	Estimated
Maximum specific growth rate for Methanogenic degraders (PS)	$U_{M\ max}$	0.27	day^{-1}	Estimated
Maximum specific growth rate for Methanogenic degraders (FOG)	$U_{M\ max}$	0.38	day^{-1}	Estimated
Half saturation constant of methanogenesis	kvfa	0.003	$g.L^{-1}$	Batstone et al 2002
Saturation inhibition constant of methanogenic bacteria	Kim	16.4	$g.L^{-1}$	Batstone et al 2002
Death rate for methanogenic bacteria	$K_{d\ m}$	0.005	day^{-1}	Batstone et al 2002
Death rate for amino acid degraders	$K_{d\ AA}$	0.0025	day^{-1}	Batstone et al 2002
Death rate for long chain fatty acid degraders	$K_{d\ LCFA}$	0.001	day^{-1}	Batstone et al 2002
Maximum specific growth rate for propionic degraders	$U_{P\ max}$	0.08	day^{-1}	Batstone et al 2002

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Table 6.9 (Continued)

Parameter Description	Symbol	Value	Unit	Ref
Ammonia inhibition constant (algae)	$K_{i\,NH3}$	0.015	M	Estimated
Ammonia inhibition constant (algae)	$K_{i\,NH3}$	0.015	M	Estimated
Ammonia inhibition constant (bacteria)	$K_{i\,NH3}$	0.0015	M	Estimated
Ammonia inhibition constant (PS)	$K_{i\,NH3}$	0.003	M	Estimated
Ammonia inhibition constant (FOG)	$K_{i\,NH3}$	0.018	M	Estimated
Inhibition for inorganic nitrogen (algae)	$K_{i\,N}$	0.18	M	Estimated
Inhibition for inorganic nitrogen (bacteria)	$K_{i\,N}$	0.0014	M	Estimated
Inhibition for inorganic nitrogen (PS)	$K_{i\,N}$	0.78	M	Estimated
Inhibition for inorganic nitrogen (FOG)	$K_{i\,N}$	18	M	Estimated
Hydrolysis rate constant	$K_{h\,p}$	0.05	day^{-1}	Estimated
inhibition constant for VFA (algae)	$K_{im\,LCFA}$	210.96	$g.L^{-1}$	Estimated
inhibition constant for VFA (bacteria)	$K_{im\,LCFA}$	183.6	$g.L^{-1}$	Estimated
inhibition constant for VFA (PS)	$K_{im\,LCFA}$	165	$g.L^{-1}$	Estimated
inhibition constant for VFA (FOG)	$K_{im\,LCFA}$	249.7	$g.L^{-1}$	Estimated
Dissociation constant for coupled HCO_3^-/CO_2	K_C	4.9×10^{-7}	M	Batstone et al 2002
Dissociation constant for coupled NH_3/NH_4^+	K_N	1.58×10^{-9}	M	Estimated
Mass transfer coefficient	K_{la}	5	day^{-1}	
Henry's constant for carbon dioxide	H_{CO2}	0.027	$M.bar^{-1}$	
Gas law constant	R	0.0831	$Bar.M^{-1}.K^{-1}$	

Note: Estimated values are based on numerical values obtained after model was calibrated and validated with experimental data from literature.

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Table 6.10 AD model input characteristics

Input	Value	Unit
Algae		
Lipid Content	0.29	%
Carbohydrate Content	0.16	%
Protein Content	0.55	%
Bacteria		
Lipid Content	0.04	%
Carbohydrate Content	0.38	%
Protein Content	0.58	%
Primary Sludge (PS)		
Lipid Content	0.36	%
Carbohydrate Content	0.41	%
Protein Content	0.23	%
Fat-Grease-Oil (FOG)		
Lipid Content	0.94	%
Carbohydrate Content	0.01	%
Protein Content	0.05	%
Operating temperature	35	°C

6.4 Results and Discussion

With a satisfactory validation of the model against the experimental data from literature, the model was simulated for different scenarios or conditions. Initial influent COD concentration for the wastewater was kept at 515 mg/L (117 g/capita.day). Plant capacity was kept constant at 15 MGD (for a population of 250,000) unless specified otherwise. Most of the influent and operational parameters for both the HRAP and AD have been provided in Table 6.7 through Table 6.10. Solids concentration for primary sludge, biomass, and FOG were assumed to be 2.5%, 4%, and 10%, respectively. Different energy assessment scenarios (1 – 3) were considered for two different process configurations; CPT-HRAP-AD, and APT-HRAP-AD. The scenarios are based on the percent FOG added for biogas production. FOG volatile solids (VS) added was assumed to be 50 g VS/L. The combined heat and power electrical energy efficiency was assumed to be 45% and a power to heat conversion ratio of 0.6. The theoretical AD reactor volume for a 15 MGD

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plant capacity was 2337 m^3 with only 2050 m^3 as usable volume and 287 m^3 represent gas volume. All parameters used for energy assessment are presented in Table 6.5 of section 6.3.4.1.

Figures 6.12 to 6.17 provide general dynamic profiles of both the HRAP and AD system. As shown in Figure 9 below, the open pond reached a steady-state condition after day 12. Microalgae growth stabilized at a rate of 0.54 g/L (Figure 6.11). Pond pH ranged from 8.5 to 10, while maintaining a carbon concentration of 0.027 g inorganic carbon/L (Figure 6.13). The effluent COD, N, and P concentrations also reached a steady state of 0.05 gCOD/L, 0.002 g N/L (Figure 6.12), and 0.001 g P/L (Figure 14), respectively. The AD effluent for pH and soluble components such as amino acid, LCFA, and acetate are shown in Figure 6.15.

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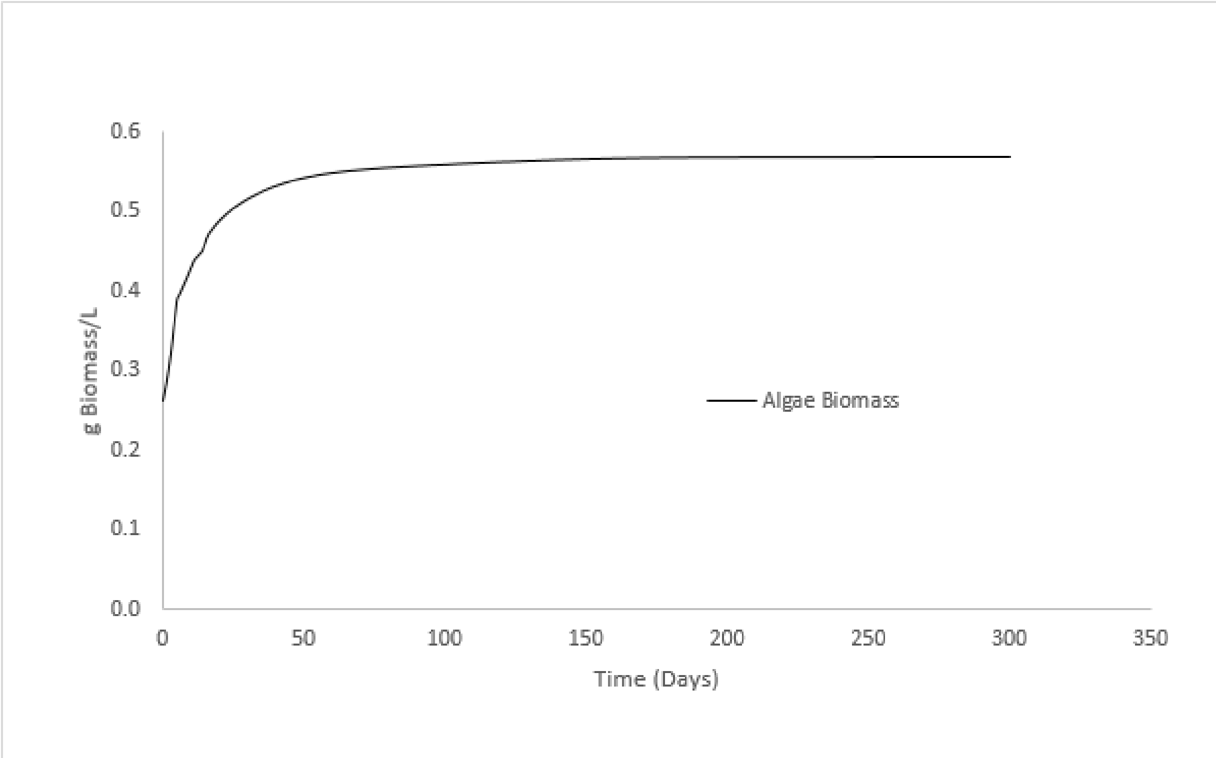

Figure 6.11 Simulation output result showing algae biomass growth for of a 15 MGD CPT-HRAP system.

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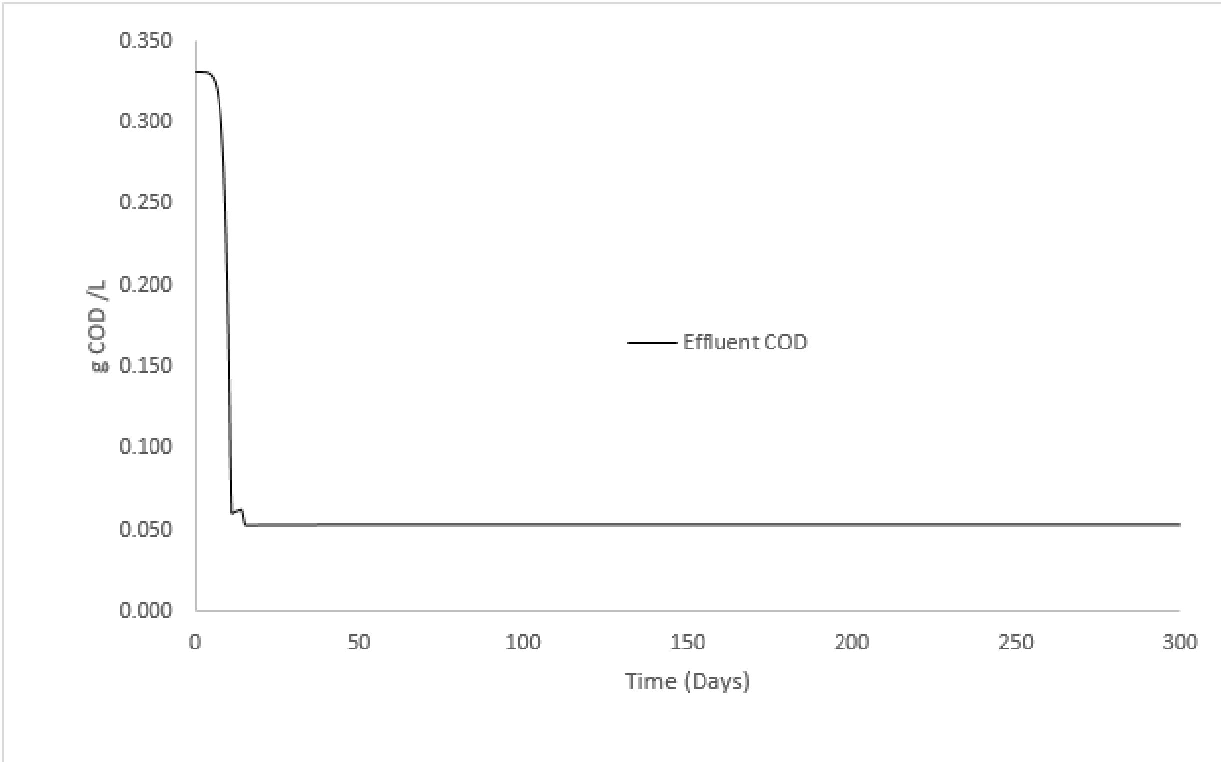

Figure 6.12 Simulation output result showing effluent concentration of COD

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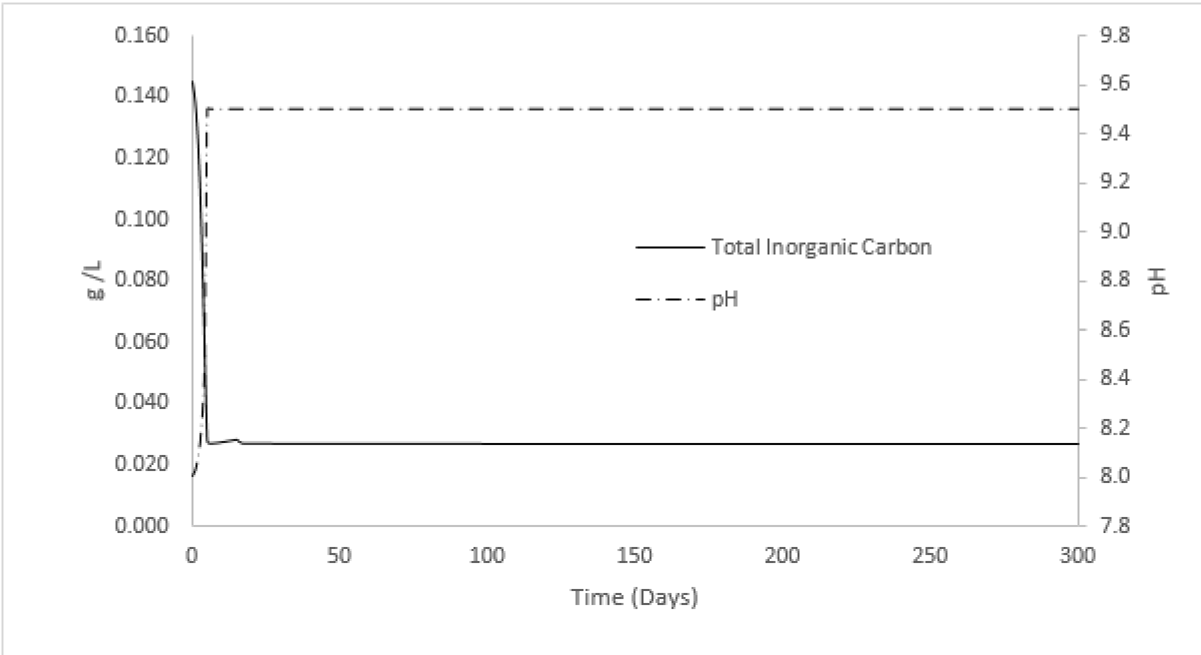

Figure 6.13 Simulation output result showing effluent total inorganic concentration and pond pH

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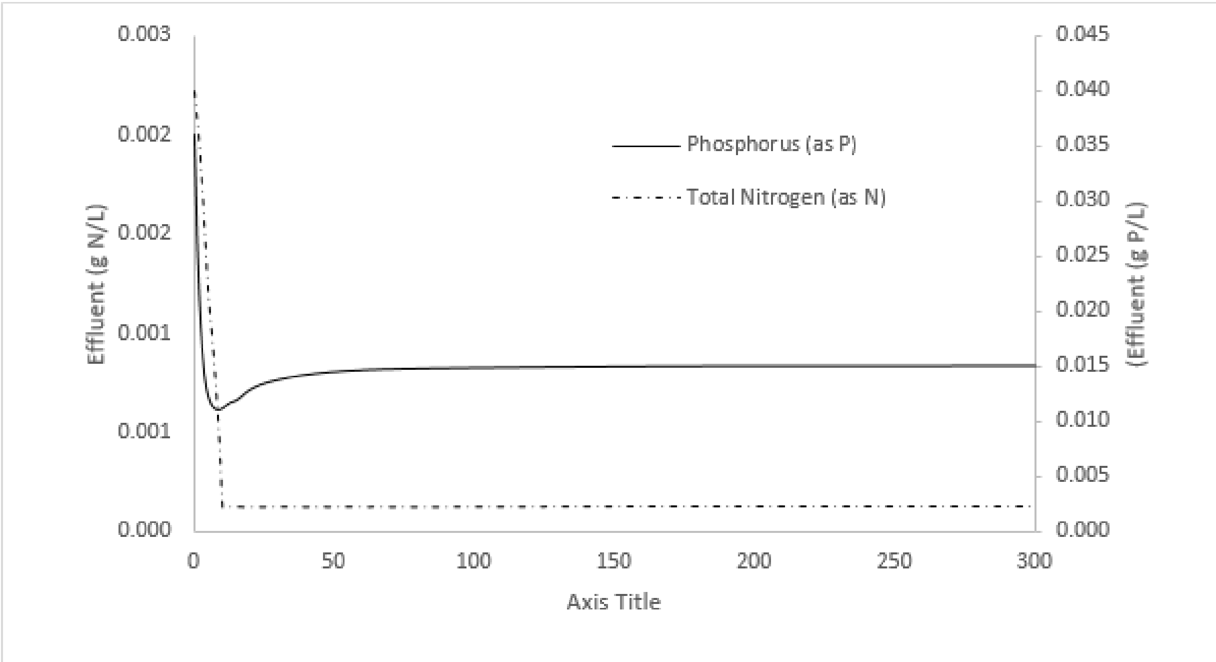

Figure 6.14 Simulation output result showing effluent concentration of phosphorus (as P) and total nitrogen (as N)

Both Figures 6.16 and 6.17 are the output of co-digestion of primary sludge, biological biomass (microalgae-bacteria) and FOG. The AD operation was assumed to maintain a mesophilic temperature of 35°C; a HRT of 30 days and a volumetric loading rate of 1 kg/m^3.day. The amount of volatile solids for FOG addition was fixed at 50 g VS/L. Methane potential as shown in Figure 6.17 maintained a steady-state yield between 0.9-1 L CH_4/g VS.

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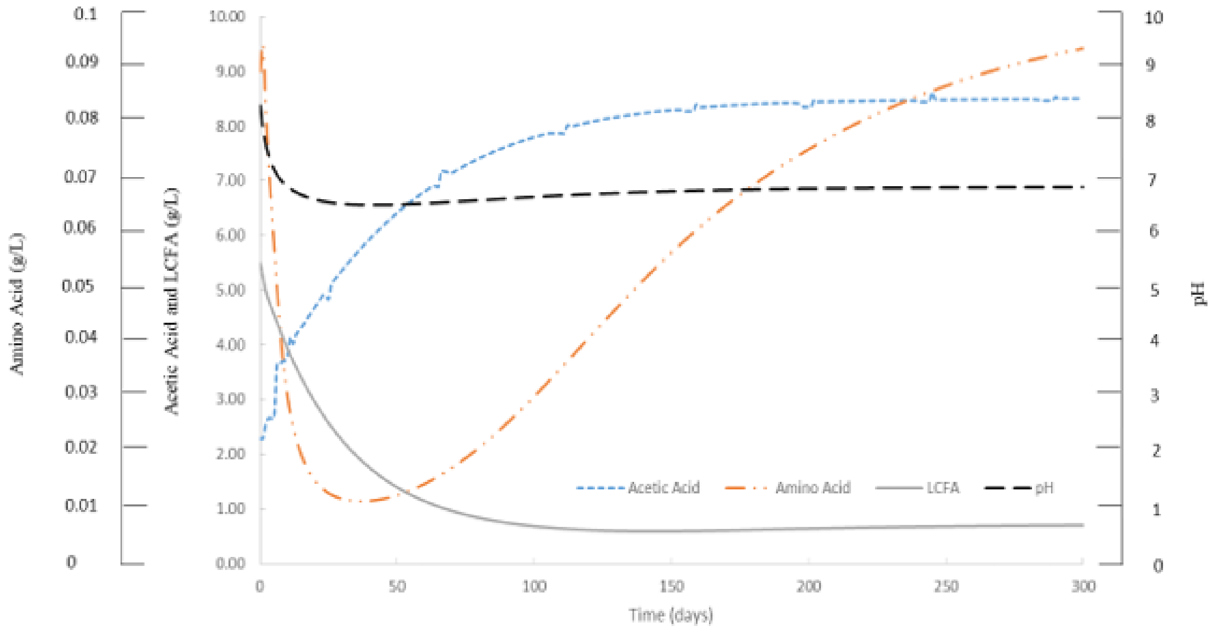

Figure 6.15 A mesophilic Co-digestion of primary sludge (30% COD removal), mixed algae-bacteria biomass; effluent concentration of simulation for pH, amino acid, LCFA, and acetic acid are presented;

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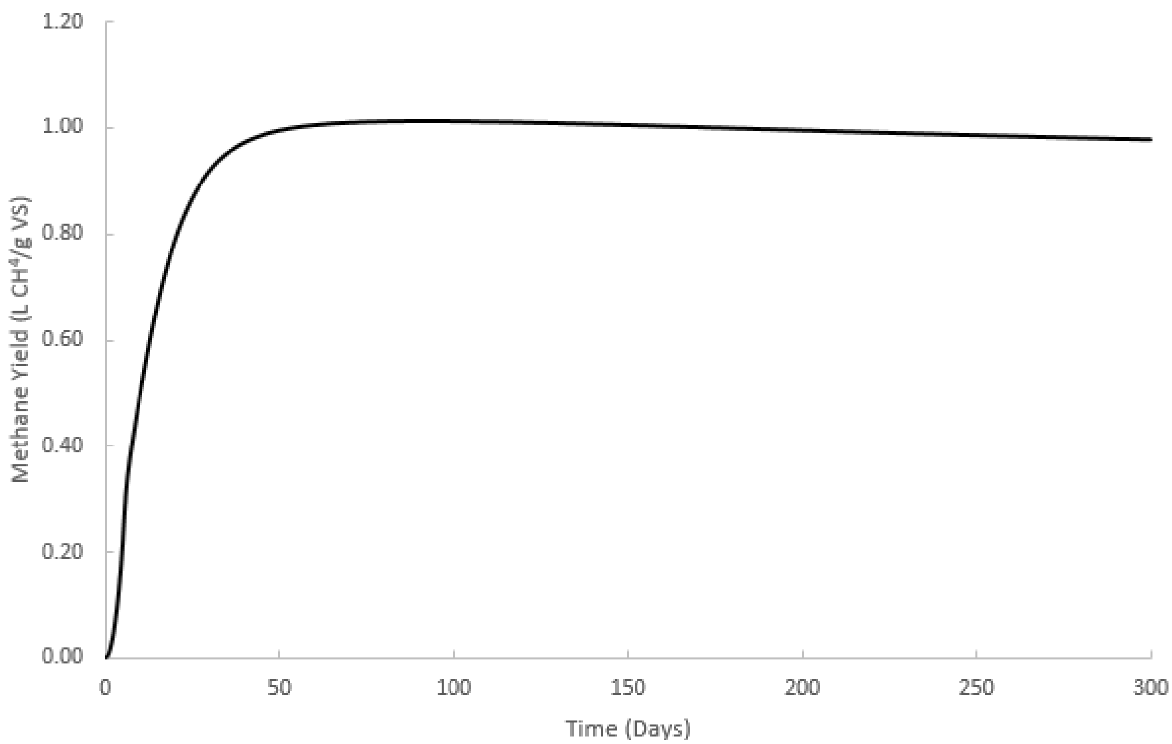

Figure 6.16 A mesophilic Co-digestion of primary sludge (30% COD removal), and mixed algae-bacteria biomass with no FOG feed - methane yield for the combined digestion of four substrates at a HRT of 30 days and volumetric loading of 1 kg/m^3.day.

6.4.1 Scenario 1: Energy assessment for wastewater process configuration without FOG co-digestion

This scenario assumes a 15 MGD plant capacity of a HRAP system with no zero FOG for co-digestion. Performance was evaluated over summer (scenario 1a) and winter (Scenario 1b) conditions. Other operational conditions considered for this analysis are provided in Table 6.11. Scenario 1a specifically focuses on the concept of adopting a conventional primary treatment (CPT) unit (such as sedimentation with at 30% COD removal efficiency), whereas, scenario 1b involves upgrading from conventional pretreatment to Advanced Primary Treatment (APT) technology (with a 60% COD removal efficiency) to improve both treatment efficiency and energy production.

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As provided in Table 6.11 below, in summer, the net energy ratio (NER) for electricity for both CPT and APT are nearly 59% and 79% respectively. The NER for scenario 1a for summer is net negative for both CPT and APT, which is below a sustainable energy recovery factor (SERF = 1). However, NER for heat showed net positive (>1) for both CPT and APT.

In the summer season, scenario the electrical NER for both CPT and APT were slightly higher than the winter season because the electrical energy required is less compared to that of summer. However, because of the lower temperature the heat required for AD process is higher. Hence, the heat NER for winter is less than that for summer. This is because the heating requirements for AD process in summer are lower due to higher influent temperature.

To know the impact a resource recovery facility has on the environment in terms of carbon-offset, the equivalent of carbon emissions reduction by energy recovery process was evaluated. despite the fact that the electricity production is not at the self-sufficient status, this process still has positive impact on the environment due to lower fossil fuel energy consumption.

This analysis also evaluated the possibility of supplying electricity to residential homes using the produced electrical energy. The basic assumptions for the individual indicators are provided at the end of Table 6.11. The effect of the treatment configuration on the environment in terms of carbon offset for both summer and winter season is virtually the same. The only difference is the type of primary treatment technology adopted (either CPT or APT). APT shows superior benefits in terms of energy production and higher carbon offsets on all categories as specified in Table 6.11.

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Table 6.11 HRAP-AD model output for scenario 1: 15 MGD with no FOG mixture

	Summer		Winter	
	Scenario 1a		Scenario 1b	
	CPT	APT	CPT	APT
Operating Condition				
Temperature (^oC)				
HRAP	21		10.77	
AD	35		35	
HRT (days)				
HRAP	7			
AD	30			
Biogas Characteristics				
Methane Production (m^3 CH_4/day)	304.5	409.9	295	400
Energy Assessment				
Consumed ($kWh.day^{-1}$)				
Electricity	1470	1504	1376	1412
Heat	1096	1298	1351	1723
Produced ($kWh.day^{-1}$)				
Electricity	871.2	1185	843.75	1158
Heat	1742	2370	1688	2316
Net Energy Ratio (NER)				
Electricity	0.59	0.79	0.61	0.82
Heat	1.59	1.83	1.25	1.34
Environmental Impact (Carbon Offset)[1]				
Gasoline offset (Mgal)	0.03	0.04	0.03	0.04
Coal offset (tons)	117	159	108	149
Forest saved (acres)	28	38	26	36
Waste offset (tons)	94	128	86	120
Single family home served (#)	71	96	65	90

6.4.2 Scenario 2: Energy assessment for wastewater configuration with 10% FOG mixture

The only difference between scenario 1 and 2 is the addition of 10% FOG mixture for co-digestion. Only APT shows a net positive ratio of electrical energy (+ve) for both summer and winter. The addition of 10% FOG feedstock improved CPT electrical NER from 59% (scenario

[1] The basic assumptions for the carbon offset analysis are; (a) the model assumes 0.086 gal of gasoline is offset for every kWh of electricity produced (b) 0.81 lbs of coal is offset for every kWh of electricity produced; (c) 8.78E-05 acres of forest is offset (or saved) for every kWh of electricity produced; (d) 0.65 lbs of waste is offset for every kWh of electricity produced; and (e) 4.51 MWh/yr of electricity used for a single family home.

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1a) to 93% for scenario 2a; and from 61% to 86% for scenario 2b with CPT. Net energy positive status for both electricity and heat was achieved for scenario 2a with APT.

According to Davidsson et al 2007, the addition of FOG for co-digestion with sewage sludge showed an increase in the methane yield by a quantitative sum of 9–27% when 10–30% of sludge from grease traps (on VS-basis) was added. By adding 10% FOG, biogas production increase by a factor of approximately 1.46 (20 - 36%) as shown in Table 6.12. This shows a similar trend with literature (Davidsson et al 2007). Overall, the heat NER decreased because of the high amount of heat required to treat increased sludge quantity. By adding supplemental feedstock to enhance energy production, the potential carbon offset increases as well.

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Table 6.12 Steady-state model output for scenario 2: 15 MGD with 10% FOG mixture

		Summer		Winter	
		Scenario 2a		Scenario 2b	
		CPT	APT	CPT	APT
Operating Condition					
Temperature (^oC)					
	HRAP	21		10.77	
	AD	35		35	
HRT (days)					
	HRAP	7			
	AD	30			
Biogas Characteristics					
	Methane Production (m^3 CH_4/day)	511.2	642.5	431.2	563.3
Energy Assessment					
Consumed ($kWh.day^{-1}$)					
	Electricity	1587	1636	1453	1504
	Heat	3138	3591	3782	4627
Produced ($kWh.day^{-1}$)					
	Electricity	1483	1874	1248.75	1642.5
	Heat	2966.00	3748.00	2498	3285
Net Energy Ratio (NER)					
	Electricity	0.93	1.15	0.86	1.09
	Heat	0.95	1.04	0.66	0.71
Environmental Impact (Carbon Offset)					
	Gasoline offset (Mgal)	0.05	0.06	0.04	0.05
	Coal offset (tons)	199	251	125	177
	Forest saved (acres)	48	60	30	42
	Waste offset (tons)	160	202	101	142
	Single family home served (#)	120	152	76	107

6.4.3 Scenario 3: Energy assessment of wastewater process configuration with 20% FOG mixture

Table 6.13 shows a positive electrical NER for both CPT and APT in summer and winter. Heat NER for scenario 3a and 3b (for both CPT and APT) decreased. Less heat is produced than required due to increase in sludge quantity. Table 6.13 also shows that with an increase in FOG mixture, the energy intensity between CPT and APT is approximately 63 $kWh.day^{-1}$. That means there is no significant difference in the NER for CPT and APT when a FOG mixture of 20% is co-digested. Overall, the three scenarios show that energy recovery is higher in summer than in winter.

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Improving primary treatment COD removal from 30 to 60% efficiency also help improve energy recovery. The addition of FOG significantly enhanced energy production through electricity production but the increase in sludge reduces the heat recovery. The environmental impact analysis also shows that improving energy production with 20% FOG mix compared to no FOG mixture; reflects a significant (about 200%) positive impact on the environment.

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Table 6.13 Steady-state model output for scenario 3: 15 MGD with 20% FOG mixture

	Summer		**Winter**	
	Scenario 3a		Scenario 3b	
	CPT	APT	CPT	APT
Operating Condition				
Temperature (^{o}C)				
HRAP	21		10.77	
AD	35		35	
HRT (days)				
HRAP	7			
AD	30			
Biogas Characteristics				
Methane Production (m^3 CH_4/day)	718	875	657.3	726.5
Energy Assessment				
Consumed ($kWh.day^{-1}$)				
Electricity	1704	1767	1531	1596
Heat	5180	5884	6212	7530
Produced ($kWh.day^{-1}$)				
Electricity	2095	2562	1653.3	2126.7
Heat	4190.00	5124.00	3306.60	4253.40
Net Energy Ratio (NER)				
Electricity	1.23	1.45	1.08	1.33
Heat	0.81	0.87	0.53	0.56
Environmental Impact (Carbon Offset)				
Gasoline offset (Mgal)	0.07	0.08	0.05	0.07
Coal offset (tons)	281	344	143	205
Forest saved (acres)	67	82	34	49
Waste offset (tons)	225	276	115	165
Single family home served (#)	170	207	86	124

Considering scenarios 1 to 3, adding a FOG mixture of 10% and improving primary treatment efficiency from 30% to 60% by adopting an APT technology will result in an energy positive status. Larissa et al. 2019, studied the effect of primary treatment of influent wastewater before the operation of a HRAP system, and its impact on bioenergy recovery. The authors concluded that HRAP with primary treatment improved the methane yield or biogas production. Comparing all the three scenarios, it becomes evident that a HRAP-AD configuration with APT presents a promising option with lower environmental impact potential irrespective of the season.

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6.4.4 Effect of Process Parameter - Hydraulic Retention Time

The effect of HRT on AD methane yield (Figure 6.17) and HRAP algae biomass production (Figure 6.17) was studied. Figure 6.17 shows the methane yield for three different HRTs (20, 30, and 40), as it can be seen the relationship between HRT and biomethane potential is directly proportional. Algae biomass growth on the other hand did not show any changes with respect to changes in HRT (Figure 6.18).

On a full-scale system design a low HRT (or SRT) may be preferred in order to decrease the reactor volume. But, as for other particulate organic substrates (e.g., waste activated sludge and lignocellulosic biomass), much longer SRTs may be preferred in order to attain a higher methane yields (Mahdy et al 2015, Passos and Ivert 2014). This is mainly ascribed to refractory substances such as the nature of microalgae cell wall. This is illustrated in Figure 6.8 section 6.3.5.2. In fact, methane yield tests with different microalgae species have proven that AD is strain-specific and it specifically depends on the composition and biodegradability of the microorganism cell wall, which is mainly composed by cellulose, hemicellulose and pectin (Mahdy et al 2015, Passos and Ivert 2014). For example, the cellulosic content of the microorganism cell wall may obstruct anaerobic bacterial attack, since it requires different enzymes for solubilization and it depends strongly on many factors such as the inoculum source, biomass concentration and cellulose bioavailability in the cell structure (Passos and Ivert 2014).

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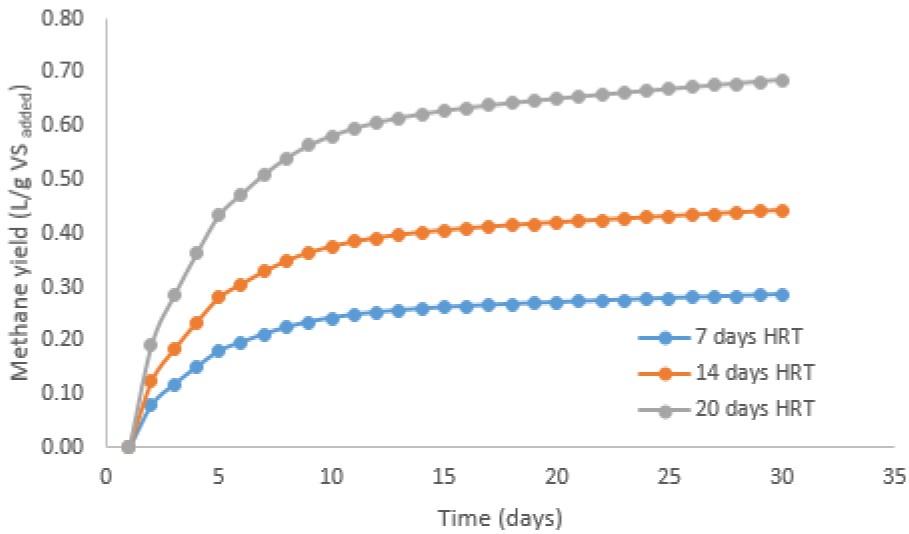

Figure 6.17 Represent the effect of HRT for a 15 MGD; a summer water temperature of 21ºC (30ºC air temperature) - shows HRT effect on AD methane yield

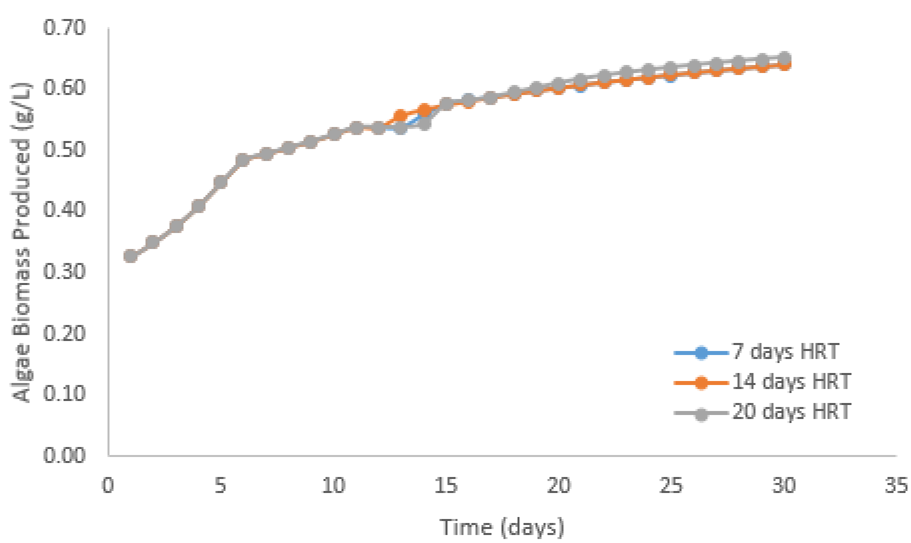

Figure 6.18 Represent the effect of HRT for a 15 MGD; a summer water temperature of 21ºC (30ºC air temperature) - shows HRT effect on HRAP biomass growth

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6.4.5 Effect of Process Parameter - Temperature

Figure 6.20 shows the effectiveness of the HRAP operation in both summer and winter. The summer season shows an effluent COD quality averaging 54 mg COD/L after day 12; for the winter season effluent COD stabilizes at 75 mg COD/L after 16 days. The HRAP treatment efficiency was evaluated to provide a complete picture of the feasibility of a HRAP-AD process as a resource recovery facility.

HRAP treatment effluent quality is affected by temperature and light intensity. Low temperatures affect microalgae growth and increase the amount of un-oxidized COD. For this reason, the effluent quality concentration for winter tends to be higher than summer. Additionally, in winter the time required for effluent quality to maintain a steady state is 4 days more than summer. Hence, the wastewater treatment process is less effective during the winter season. This is true because according to Rittmann and McCarty, with a drop in temperature two issues can be found in ponds of this type, first microbial activities slow down, and secondly BOD and ammonia nitrogen oxidation are slowed, which may jeopardize effluent quality.

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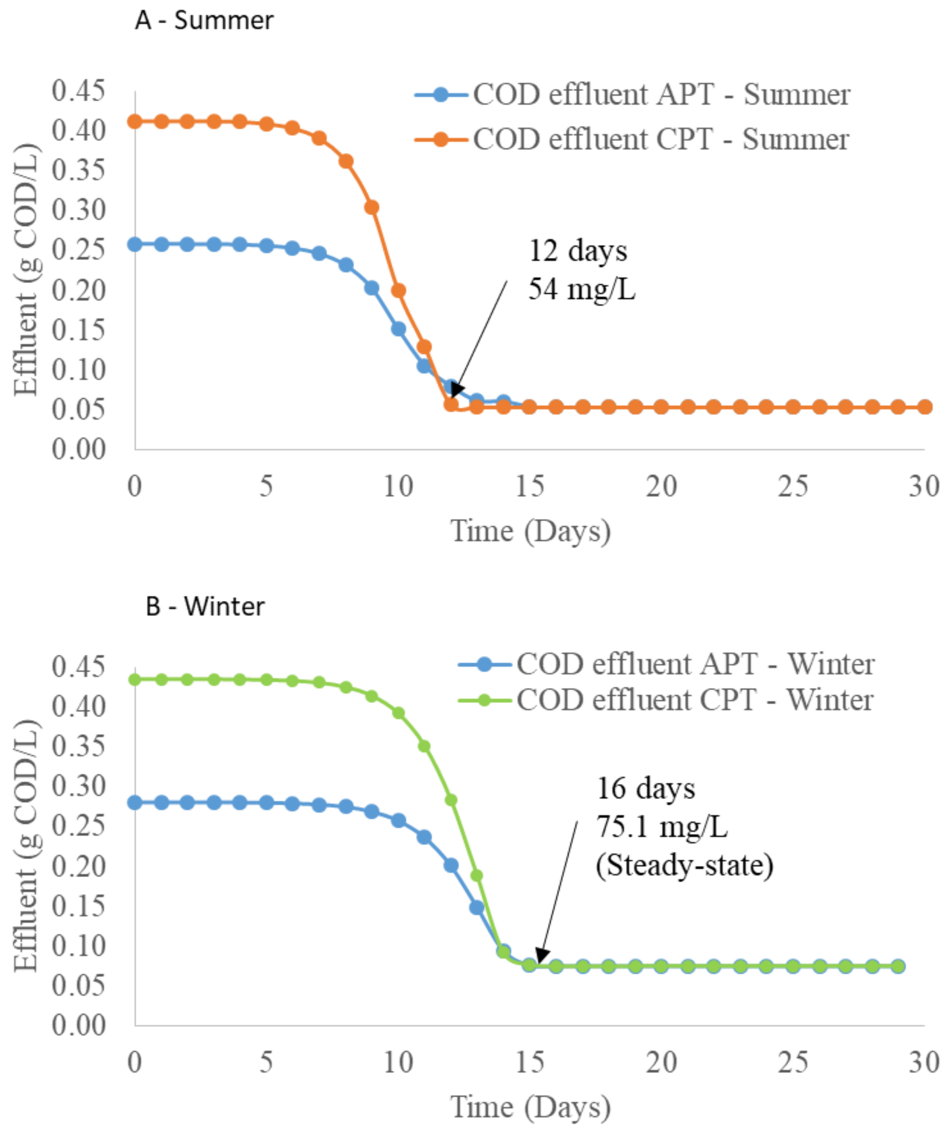

Figure 6.19 Represent the effect of HRT and seasonal change on the model output for a 15 MGD; a summer (A) water temperature of 21°C (30°C air temperature) and winter (B) temperature of 10.7 °C (air temperature of 8 °C) - Seasonal effect on COD effluent with summer water temperature at 21 °C and 10.7 °C for winter.

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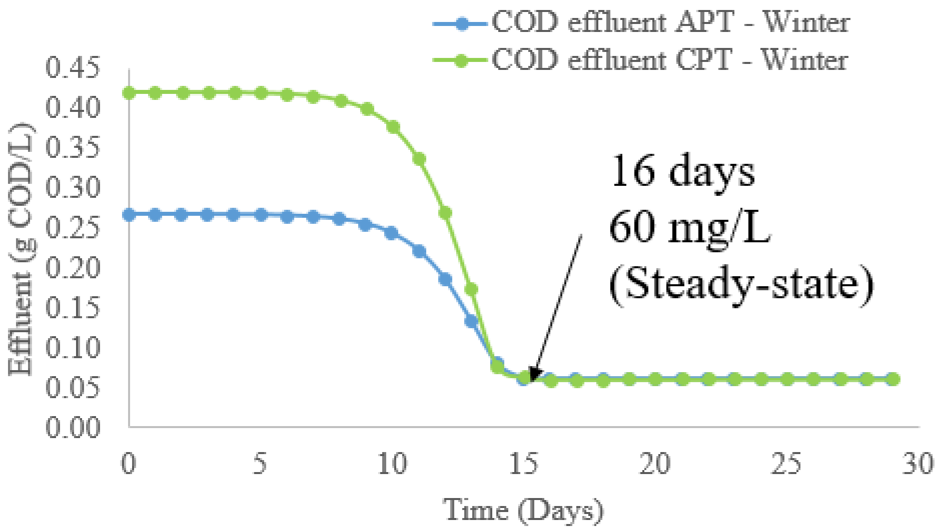

Figure 6.20 Improved effluent COD quality for winter by increasing the secondary solids separation efficiency from 75% to 90%.

It is obvious that the higher COD concentration for winter is due to the increased suspended COD particulates or un-oxidized COD. By improving solids separation efficiency from 75% to 90%, the effluent quality can be improved by 20% as shown in Figure 4.20.

On the other hand, Figure 6.21 shows effluent total nitrogen (as N) stabilizing at ~2 mg N/L after 8 days in summer and about 13 mg N/L in winter. Both summer and winter effluent concentrations show the impact of temperature on HRAP process. It is recommended that a treatment configuration of this type will require some form of tertiary treatment to polish the HRAP treated water before discharge. Similarly, a pilot HRAP system operated in California showed higher effluent total nitrogen (as N) above 10 mg N/L during winter and a tertiary treatment was recommended to polish effluent quality during winter (WEF 2016). The additional process additions recommended are continued aeration (day and night times) in HRAP and denitrification basins.

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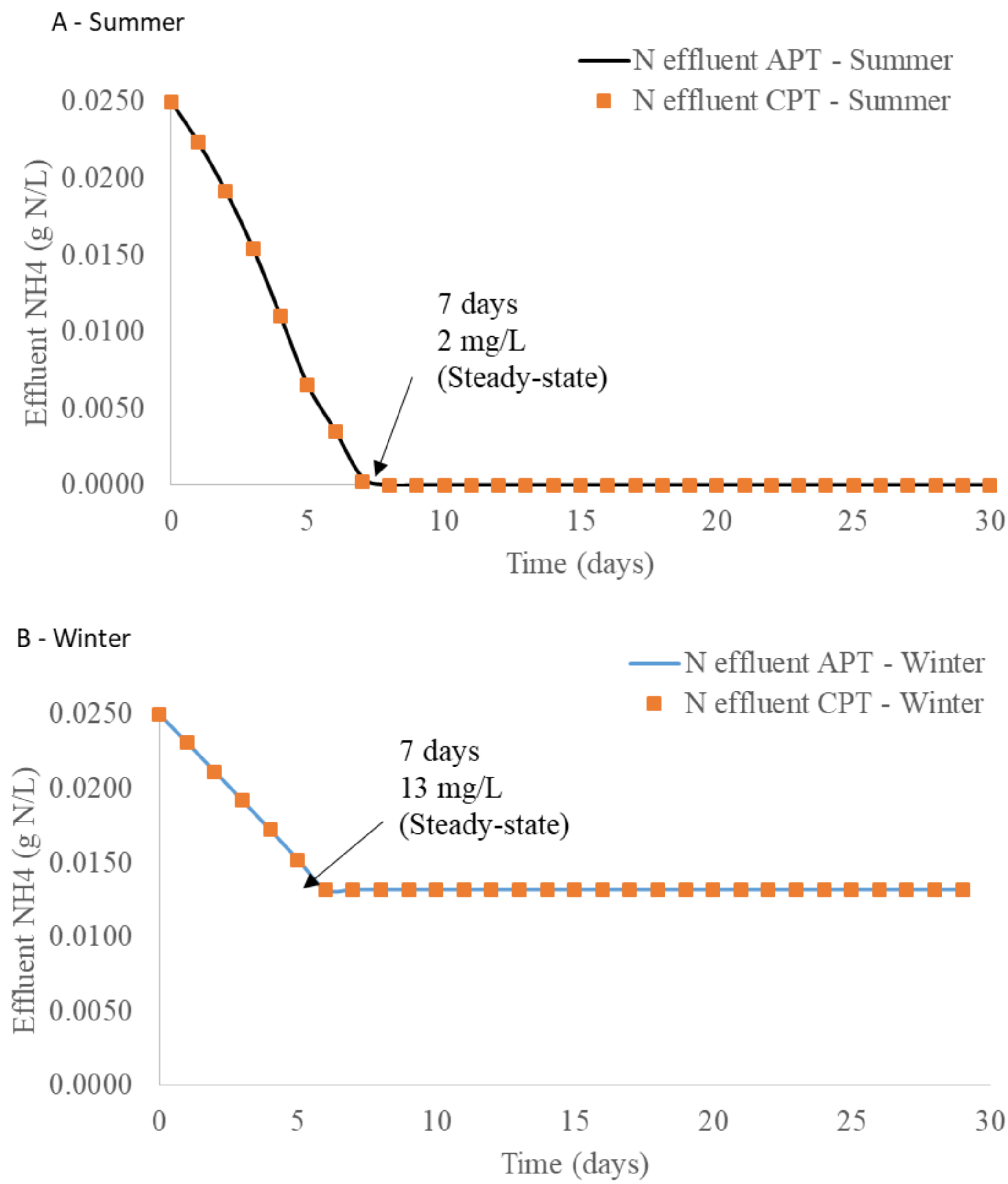

Figure 6.21 Represent the effect of HRT and seasonal change on the model output for a 15 MGD; a summer water temperature of 21ºC (30ºC air temperature) and winter temperature of 10.7 ºC (air temperature of 8 ºC) - Seasonal effect on COD effluent with summer water temperature at 21 ºC and 10.7 ºC for winter.

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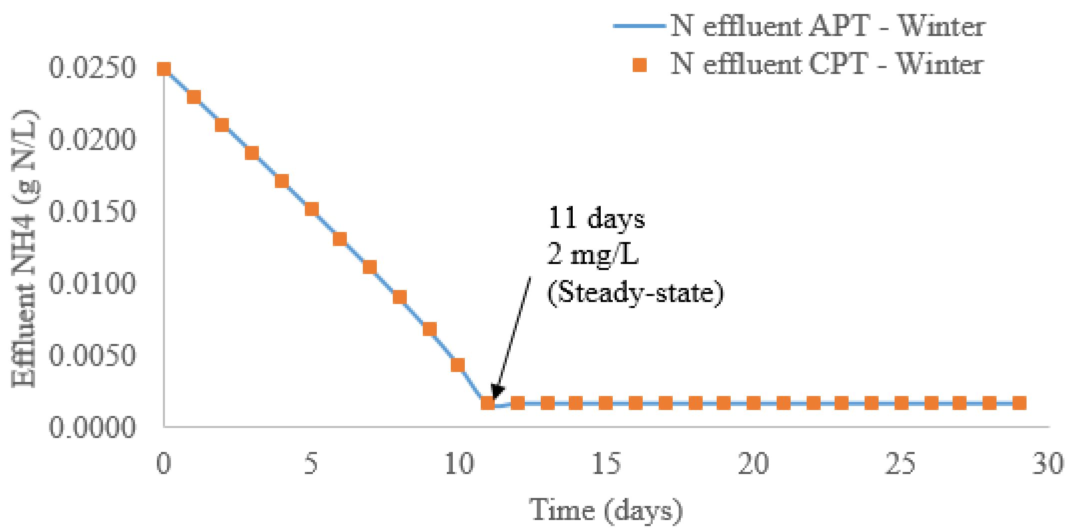

Figure 6.22 Improved winter N effluent quality

To improve the nitrogen effluent quality, the first step is to find ways of increasing oxygen concentration in the pond for nitrification. That can be done in two ways, through mechanical aeration or increase algae oxygen production. It is very obvious that improving in-situ oxygen production will be economical and energy efficient. So, algae biomass in pond was increased by increasing solids retention time that will increase the algae activities in the pond. Nitrogen effluent quality was significantly improved. This agrees with Rittman and McCarty, who suggested that employing a much longer detention time may help minimize the impact of poor effluent quality (Figure 6.22) (Rittman and McCarty 2001).

6.4.6 Effect of Process Parameters – Varying Plant Capacity

This analysis assumed a 20% FOG mixture, 30 day HRT for AD and 7 HRT for HRAP. CPT-HRAP-AD was assumed as the process configuration with water temperature maintained at

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21°C. It can be seen in Figure 6.21 that electricity production is net positive (+ve) as the plant capacity increases.

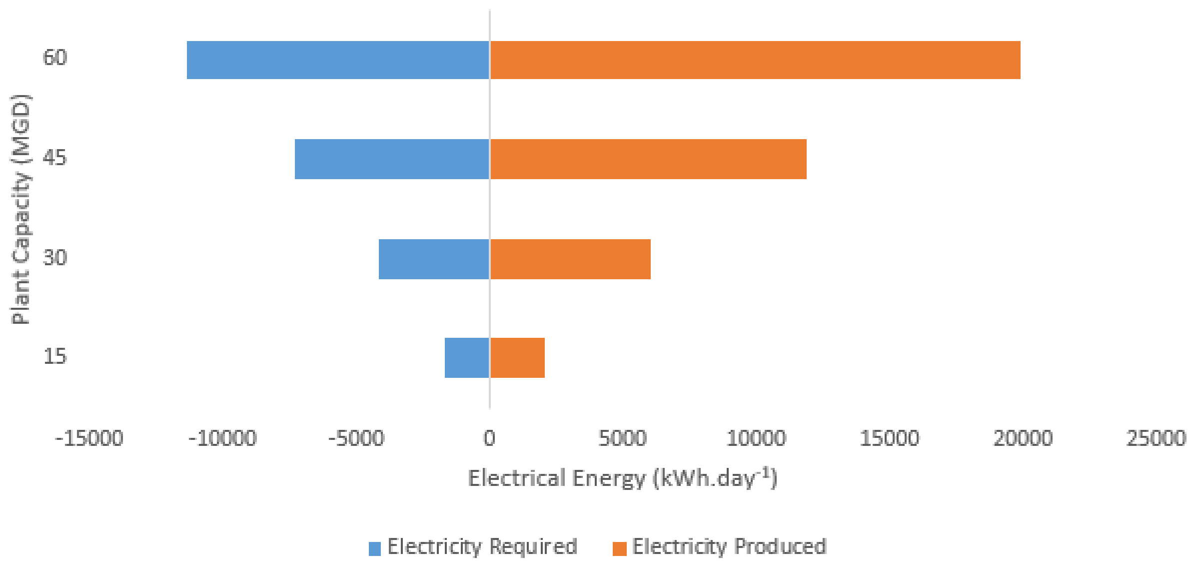

Figure 6.23 Effect of plant capacity on electricity energy

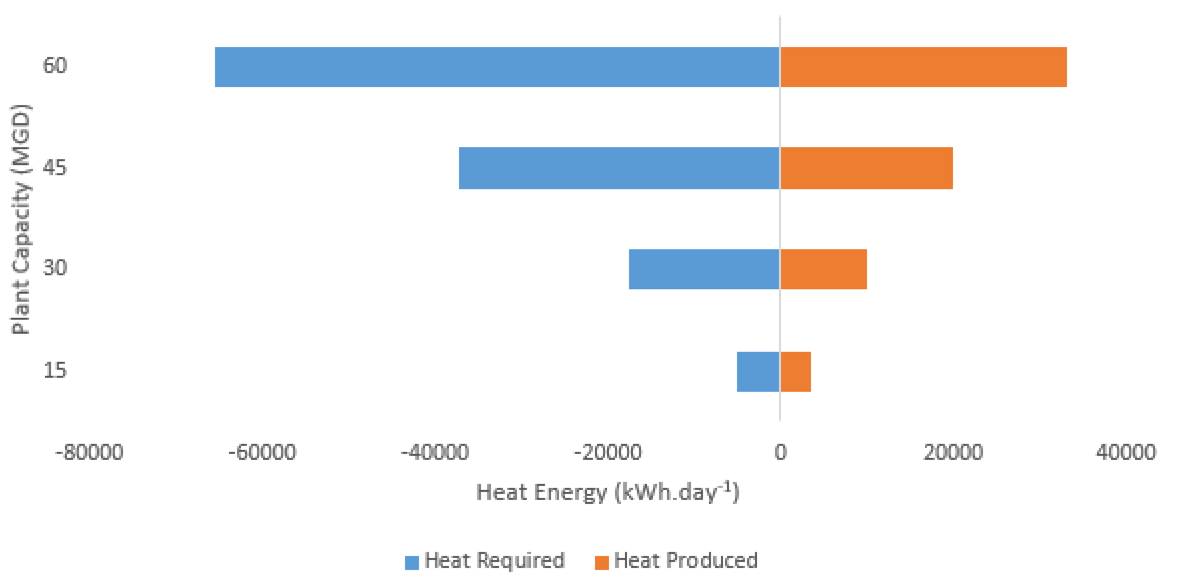

Figure 6.24 Effect of plant capacity on thermal energy

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Figure 6.24 shows a net (-ve) heat energy production for all plant sizes. This is due to higher amount of sludge generated than heat generated. Even though, the simulation output shows increasing net positive electrical energy production for all plant sizes; it may not be practical feasibility due to other unknown site specific conditions which were not considered in the simulation.

6.5 Conclusion

Simulation of combined HRAP wastewater treatment with anaerobic digestion of multiple substrates was studied. The models were calibrated against experimental results from literature to validate the simulation data. The coupled model was used to simulate different scenarios for energy assessment. Results of the simulation showed that a microalgae-bacteria wastewater treatment system alone cannot achieve energy autarky. Effluent concentration for both COD and N are lower in summer season compared to winter. Similarly, a pilot HRAP system operated in California showed higher effluent for total nitrogen (as N) during winter and a tertiary treatment was recommended to polish effluent quality during winter. To address this issue, continuous aeration of HRAP and addition of denitrification basins were recommended as options. The COD concentration in winter effluent can be improved by increasing solids separation efficiency and similarly N concentration can be improved by increasing solids retention time.

In order to improve the energy balance of the process, different compositions of FOG, primary sludge, microalgae, and bacterial biomass were evaluated. The favorable FOG mixture for a net positive heat and electricity was 10% FOG feedstock mixture with co-digestion; and this is only feasible during summer. Although, the theoretical analysis assumes a large-scale (>15 MGD) treatment plant, it is worth noting that the application of this size plant may be practically

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challenging due to many unknowns. This situation confirms the need for a coupled simulation of this process to identify the design challenges and evaluate possible alternatives. This model can be instrumental to study various other scenarios that may provide better treatment, energy recovery and environmental impact performance indicators.

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CHAPTER VII

SUMMARY AND FUTURE RECOMMENDATIONS

7.1 Summary

The goal of this research is to develop quantitative and dynamic process models to evaluate the energy performance of a wastewater treatment plant (WWTP). This quantitative model serves as an assessment tool for energy analysis of a WWTP. The model outcomes can then be used to propose feasible schemes to achieve energy self-sufficiency in future WWTP designs. An integrated dynamic model was also developed to perform a comprehensive energy and environmental performance analysis of the proposed future WWTP design.

First, a hypothetical concept of three process schemes which progressively build upon the concept of transforming a conventional activated sludge wastewater treatment plant (CAS-WWTP) into a water resource recovery facility (WRRF) was evaluated. It was concluded that, existing utilities can become energy self-sufficient by conserving energy and by producing additional bioenergy through biogas. The biological process (i.e., aeration facility) is the main energy consumer and minimizing energy consumption of the aeration unit is the key. Conventional method of removing nutrients from wastewater is an energy-intensive process. This can be better managed by adopting novel nitrogen removal techniques such as the one discussed in scheme 2 of chapter 3. Finally, replacing the activated sludge process with a low energy demanding technology such as HRAP can transform a WWTP into an energy-yielding process.

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Next, a quantitative model was developed to perform a detailed analysis of two (basic and moderate) energy-neutral or energy-positive wastewater treatment configurations. In addition, a novel and practically feasible energy-positive wastewater treatment scheme incorporating advanced solids separation was presented with energy analysis and a case study. This model can be useful to quickly assess the energy recovery potential of small scale wastewater treatment systems. It was concluded that, WWTPs with capacities less than 5 MGD could achieve energy neutrality if the wastewater N:COD ratio is less than 0.1 and a more energy-efficient ICE (greater than or equal to 40%), and codigestion are included for enhanced energy recovery.

It was recommended that more effort be put into nitrogen removal since higher nitrogen concentration increases the energy requirements of the WWTPs. Also, improving primary treatment efficiency presents an opportunity to enhance overall energy production and to reduce energy consumption. The addition of FOG for codigestion has a positive effect on the digestion process with higher methane yields and stable operations. Biogas production due to FOG codigestion could also increase from 15 to 30%, which is a significant contribution to electricity and heat recovery. New WTTP designs should consider the advanced configuration after a detailed assessment and practical-scale demonstration. Overall, the model presented in this study can be a beneficial assessment tool for different wastewater treatment systems.

In addition, the same quantitative methodology adopted in the previous analyses was used to develop a systematic analysis of different wastewater treatment scenarios based on wastewater strength, plant capacity, primary treatment efficiency, and different supplemental feedstock to evaluate the potential for transitioning WWTPs into WRRFs. In this analysis, it was concluded that, replacing old equipment with highly-efficient ones is the first step for a WWTP to become a WRRF. In addition, improving primary treatment unit's efficiency will provide dual benefits of

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reducing downstream aeration energy consumption and increasing energy production. A WRRF can easily save over 20% of total energy demand when plant upgrades and primary treatment efficiency improvements are implemented. Additionally, increasing biogas production with alternative high-strength biodegradable waste through co-digestion is the most feasible method to achieve an energy-neutral or energy-positive status at the plant level. Co-digestion option also provides wastewater treatment plants with a new revenue stream in the form of tipping fees. Care must be taken when selecting a CHP engine to minimize energy losses. Replacing the aeration unit with a much less energy consuming technology such as a trickling filter or a high-rate microalgae pond seems to be a more promising alternative for future designs.

Finally, an advanced treatment technology in the form of HRAP for wastewater treatment was studied. A novel treatment scheme including an advanced primary treatment system coupled with high rate algae pond model and anaerobic digestion model to simulate biological conversion of light energy into chemical energy (in the form of methane) for a future WRRF was studied. A computer software (Matlab R2019a) was used to code series of ordinary differential equations using ODE45 solver for the coupled model. The model was calibrated and validated against experimental data from literature. The adoption of HRAP technology minimizes greenhouse gas emissions such as CO_2. Winter effluent quality can be improved by increasing secondary solids separation efficiency and increasing solids retention time. Modeling of HRAP for this size (>15 MGD) plant capacity is an unchartered territory and this also presents an opportunity for future studies.

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7.2 Recommendations

Recommendations proposed for future studies may include;

(1) Calibrating the dynamic model with actual pilot plant data to incorporate site specific operational parameters will help improve the model usefulness.

(2) Expanding the model to explore the integration of other treatment technologies such as trickling filter for process optimization especially in winter.

(3) The coupled dynamic model could provide extensive platform for different studies in this field. Some of the studies may include process optimization, effect of sludge pretreatment for enhancing energy production, nitrogen fixation, carbon cycling and greenhouse gas emissions evaluation.

(4) Performing a detailed economic analysis could help relate the practical feasibility of the models developed in this study.

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APPENDIX A

RESEARCH ANALYSIS MATRIX FOR COUPLED DYNAMIC MODEL

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Table A.1 Research analysis matrix

	Summer						Winter					
	CPT – HRAP - AD (Scenario -1)			APT – HRAP - AD (Scenario – 2)			CPT – HRAP - AD (Scenario – 3)			APT – HRAP (Scenario – 2)		
	Final HRAP Effluent (COD and N)	Energy (Produced and Consumed)	Net Energy Ratio	Final HRAP Effluent (COD and N)	Energy (Produced and Consumed)	Net Energy Ratio	Final HRAP Effluent (COD and N)	Energy (Produced and Consumed)	Net Energy Ratio	Final HRAP Effluent (COD and N)	Energy (Produced and Consumed)	Net Energy Ratio
	1-a.	1-b.	1-c.	2-a.	2-b.	2-c.	3-a	3-b.	3-c	4-a	4-b	4-c
Plant Capacity (MGD)												
15	x	x	x	x	x	x	x	x	x	x	x	x
30	x	x	x	x	x	x	x	x	x	x	x	x
45	x	x	x	x	x	x	x	x	x	x	x	x
SRT												
HRAP (7, 14, 21)	x	x	x	x	x	x	x	x	x	x	x	x
AD (20, 30, 40)	x	x	x	x	x	x	x	x	x	x	x	x
Digester FOG mixture												
No FOG												
10% FOG		x	x		x	x		x	x		x	x
20% FOG		x	x		x	x		x	x		x	x

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APPENDIX B

SENSITIVITY ANALYSIS FOR COUPLED MODEL

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics		
Multiple R	0.988	
R Square	0.976	Goodness of Fit >= 0.80
Adjusted R Square	0.974	
Standard Error	0.044	
Observations	19	

ANOVA

	df	SS	MS	F	P-value
Regression	1	1.292635479	1.292635479	678.92297	0.000
Residual	17	0.032367152	0.00190395		
Total	18	1.325002632			

						Confidence Level			
						0.95			0.99
	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 99%	Upper 99%	
Intercept	-0.074986196	0.033148872	-2.262104027	0.037	-0.144924202	-0.00504819	-0.17106	0.021087	
FOG_Sim	1.085047401	0.04164266	26.05615033	0.000	0.997189069	1.172905733	0.964357	1.205737	

y = -0.075 +1.085*FOG_Sim

RESIDUAL OUTPUT

PROBABILITY OUTPUT

FOG_Sim	FOG_Exp	Observations	PredictedFOG	Residuals	Standard Resi	Sorted Res	Percentile	FOG_Exp	LCI	UCI	LPI	UPI
0	0	1	-0.07499	0.07499	1.76834	-0.11571	2.63158	0	-0.14437	-0.0056	-0.18968	0.039707
0.355461	0.195	2	0.31071	-0.11571	-2.72860	-0.06165	7.89474	0.195	0.269776	0.351636	0.210626	0.410786
0.489964	0.395	3	0.45665	-0.06165	-1.45379	-0.04016	13.15789	0.395	0.42521	0.488085	0.360061	0.553235
0.593485	0.555	4	0.56897	-0.01397	-0.32951	-0.02224	18.42105	0.555	0.543541	0.594404	0.47417	0.663775
0.67407	0.7	5	0.65641	0.04359	1.02790	-0.01412	23.68421	0.7	0.634195	0.678629	0.562421	0.750403
0.73629	0.75	6	0.72392	0.02608	0.61494	-0.01397	28.94737	0.75	0.702879	0.744968	0.630203	0.817644
0.783922	0.805	7	0.77561	0.02939	0.69315	-0.01129	34.21053	0.805	0.754541	0.796672	0.681881	0.869332
0.820157	0.87	8	0.81492	0.05508	1.29884	0.00281	39.47368	0.85	0.7933	0.836545	0.72107	0.908775
0.847604	0.85	9	0.84470	0.00530	0.12489	0.00320	44.73684	0.87	0.82237	0.867038	0.750685	0.938723
0.868331	0.87	10	0.86719	0.00281	0.06618	0.00530	50.00000	0.87	0.844172	0.890216	0.773009	0.961378
0.883934	0.87	11	0.88412	-0.01412	-0.33307	0.00712	55.26316	0.87	0.860506	0.907742	0.789792	0.978456
0.895612	0.9	12	0.89680	0.00320	0.07557	0.00770	60.52632	0.9	0.872691	0.9209	0.80234	0.99125
0.904245	0.92	13	0.90616	0.01384	0.32631	0.01006	65.78947	0.92	0.881677	0.930648	0.81161	1.000716
0.910439	0.92	14	0.91288	0.00712	0.16782	0.01384	71.05263	0.92	0.888115	0.937653	0.818257	1.007511
0.914505	0.925	15	0.91730	0.00770	0.18169	0.02608	76.31579	0.92	0.892335	0.942255	0.822618	1.011972
0.916946	0.93	16	0.91994	0.01006	0.23714	0.02939	81.57895	0.92	0.894868	0.94502	0.825236	1.014652
0.927406	0.92	17	0.93129	-0.01129	-0.26632	0.04359	86.84211	0.925	0.905705	0.956882	0.836449	1.026138
0.942099	0.925	18	0.94724	-0.02224	-0.52438	0.05508	92.10526	0.925	0.920892	0.973581	0.852185	1.042287
0.954013	0.92	19	0.96016	-0.04016	-0.94712	0.07499	97.36842	0.93	0.933176	0.987149	0.864931	1.055394
0.758868	0.748421 Xave											

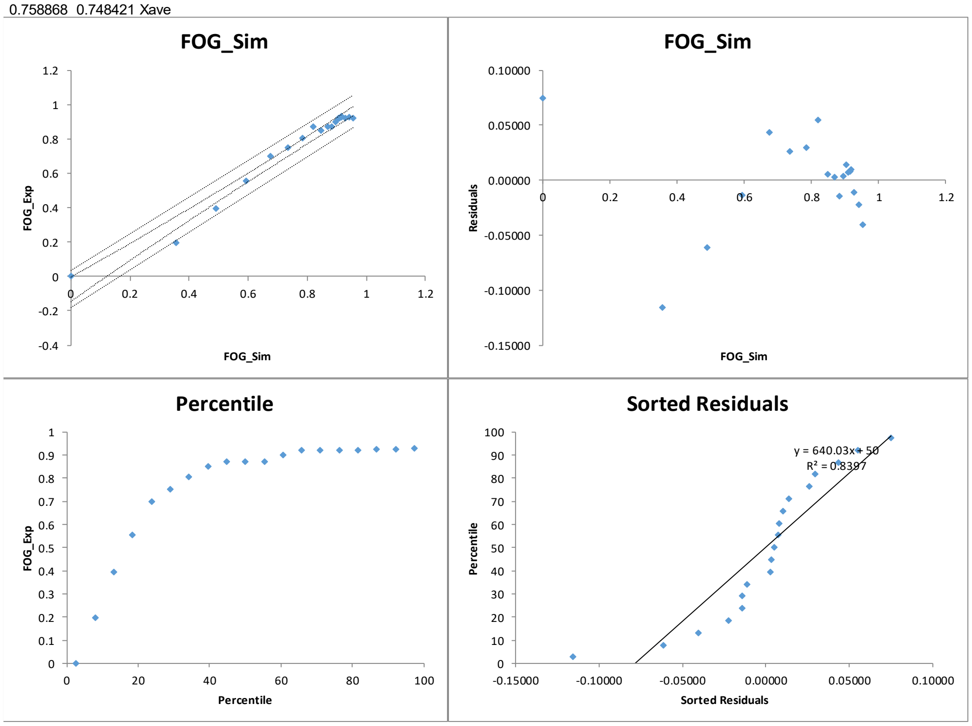

Figure B.1 FOG statistical result of AD model validation

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics		
Multiple R	0.982	
R Square	0.964	Goodness of Fit >= 0.80
Adjusted R Square	0.961	
Standard Error	0.006	
Observations	13	

ANOVA

	df	*SS*	*MS*	*F*	*P-value*
Regression	1	0.010793152	0.010793152	296.8675986	0.000
Residual	11	0.000399925	3.63568E-05		
Total	12	0.011193077			

Confidence Level

						0.95		0.99
	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 99%*	*Upper 99%*
Intercept	0.008040548	0.004654655	1.727420945	0.112	-0.002204278	0.018285374	-0.00642	0.022497
AlgaeSim	0.916078146	0.053168098	17.22984616	0.000	0.799055952	1.03310034	0.750948	1.081208

y = 0.008 +0.916*AlgaeSim

AlgaeSim	AlgaeExp	
0	0	
0.036577	0.04	
0.056297	0.07	
0.071742	0.086	
0.083463	0.089	
0.091859	0.092	
0.097279	0.093	
0.100162	0.095	
0.102204	0.1	
0.104002	0.101	
0.105327	0.103	
0.106275	0.104	
0.106919	0.1045	
0.081701	0.08288	Xave

RESIDUAL OUTPUT

PROBABILITY OUTPUT

Observations	*PredictedAlga*	*Residuals*	*Standard Resi*	*Sorted Residu*	*Percentile*	*AlgaeExp*	*LCI*	*UCI*	*LPI*	*UPI*
1	0.00804	-0.00804	-1.39279	-0.00804	3.84615	0	-0.00202	0.018096	-0.00842	0.024497
2	0.04155	-0.00155	-0.26818	-0.00480	11.53846	0.04	0.03523	0.047866	0.027071	0.056026
3	0.05961	0.01039	1.79918	-0.00416	19.23077	0.07	0.054969	0.064257	0.045784	0.073443
4	0.07376	0.01224	2.11999	-0.00231	26.92308	0.086	0.069972	0.077551	0.060195	0.087328
5	0.08450	0.00450	0.77967	-0.00167	34.61538	0.089	0.080881	0.088118	0.070979	0.098019
6	0.09219	-0.00019	-0.03306	-0.00155	42.30769	0.092	0.088394	0.095987	0.078623	0.105759
7	0.09716	-0.00416	-0.71993	-0.00153	50.00000	0.093	0.093124	0.101188	0.08352	0.110792
8	0.09980	-0.00480	-0.83093	-0.00149	57.69231	0.095	0.095608	0.103986	0.086114	0.11348
9	0.10167	-0.00167	-0.28879	-0.00140	65.38462	0.1	0.097355	0.10598	0.087946	0.115389
10	0.10331	-0.00231	-0.40095	-0.00019	73.07692	0.101	0.098886	0.107744	0.089556	0.117073
11	0.10453	-0.00153	-0.26476	0.00450	80.76923	0.103	0.10001	0.109047	0.090741	0.118316
12	0.10540	-0.00140	-0.24195	0.01039	88.46154	0.104	0.100812	0.109982	0.091587	0.119206
13	0.10599	-0.00149	-0.25750	0.01224	96.15385	0.1045	0.101356	0.110617	0.092162	0.119811

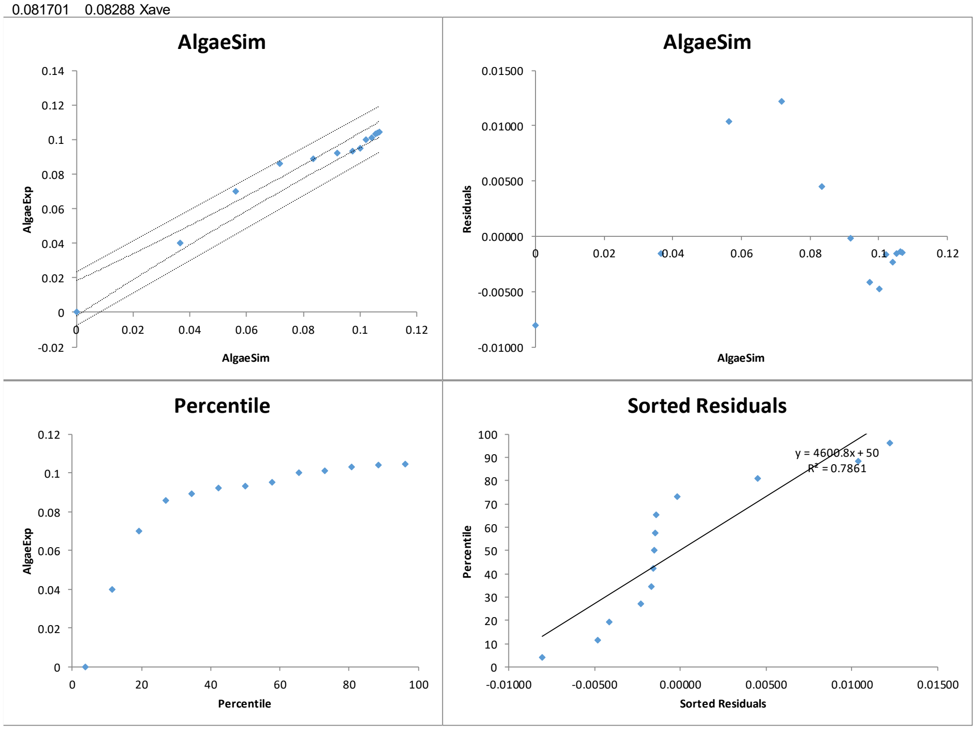

Figure B.2 Microalgae statistical results for AD model validation

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics		
Multiple R	0.997	
R Square	0.994	Goodness of Fit >= 0.80
Adjusted R Square	0.993	
Standard Error	0.006	
Observations	12	

ANOVA

	df	SS	MS	F	P-value
Regression	1	0.061357178	0.061357178	1574.314853	0.000
Residual	10	0.000389739	3.89739E-05		
Total	11	0.061746917			

Confidence Level

					0.95		0.99	
	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 99%	Upper 99%
Intercept	-0.000372538	0.00580519	-0.064173241	0.950	-0.013307307	0.012562231	-0.01877	0.018026
PS_Sim	0.988483169	0.024912854	39.67763669	0.000	0.932973872	1.043992467	0.909528	1.067439

y = 0 +0.988*PS_Sim

RESIDUAL OUTPUT

PROBABILITY OUTPUT

PS_Sim	PS_Exp	Observations	PredictedPS_E	Residuals	Standard Resid	Sorted Residu	Percentile	PS_Exp	LCI	UCI	LPI	UPI
0	0	1	-0.00037	0.00037	0.06259	-0.01122	4.16667	0	-0.01302	0.012276	-0.01895	0.018202
0.163702	0.17	2	0.16144	0.00856	1.43739	-0.00704	12.50000	0.17	0.156418	0.16647	0.146943	0.175945
0.214057	0.2	3	0.21122	-0.01122	-1.88486	-0.00459	20.83333	0.2	0.207272	0.215167	0.197056	0.225383
0.232645	0.225	4	0.22959	-0.00459	-0.77161	-0.00241	29.16667	0.225	0.22562	0.233566	0.215422	0.243763
0.240177	0.23	5	0.23704	-0.00704	-1.18239	-0.00218	37.50000	0.23	0.232983	0.241093	0.222844	0.251232
0.245613	0.24	6	0.24241	-0.00241	-0.40524	-0.00156	45.83333	0.24	0.238273	0.246551	0.228194	0.25663
0.250442	0.245	7	0.24718	-0.00218	-0.36704	0.00037	54.16667	0.245	0.242956	0.251414	0.23294	0.261429
0.254864	0.25	8	0.25156	-0.00156	-0.26137	0.00343	62.50000	0.25	0.247232	0.25588	0.237283	0.265829
0.25892	0.26	9	0.25557	0.00443	0.74506	0.00443	70.83333	0.26	0.251144	0.259986	0.241263	0.269868
0.262624	0.266	10	0.25923	0.00677	1.13782	0.00544	79.16667	0.266	0.254711	0.263744	0.244895	0.27356
0.265995	0.268	11	0.26256	0.00544	0.91416	0.00677	87.50000	0.268	0.257949	0.267168	0.248197	0.276921
0.269045	0.269	12	0.26557	0.00343	0.57550	0.00856	95.83333	0.269	0.260876	0.270273	0.251184	0.279965
0.221507	0.2186 Xave											

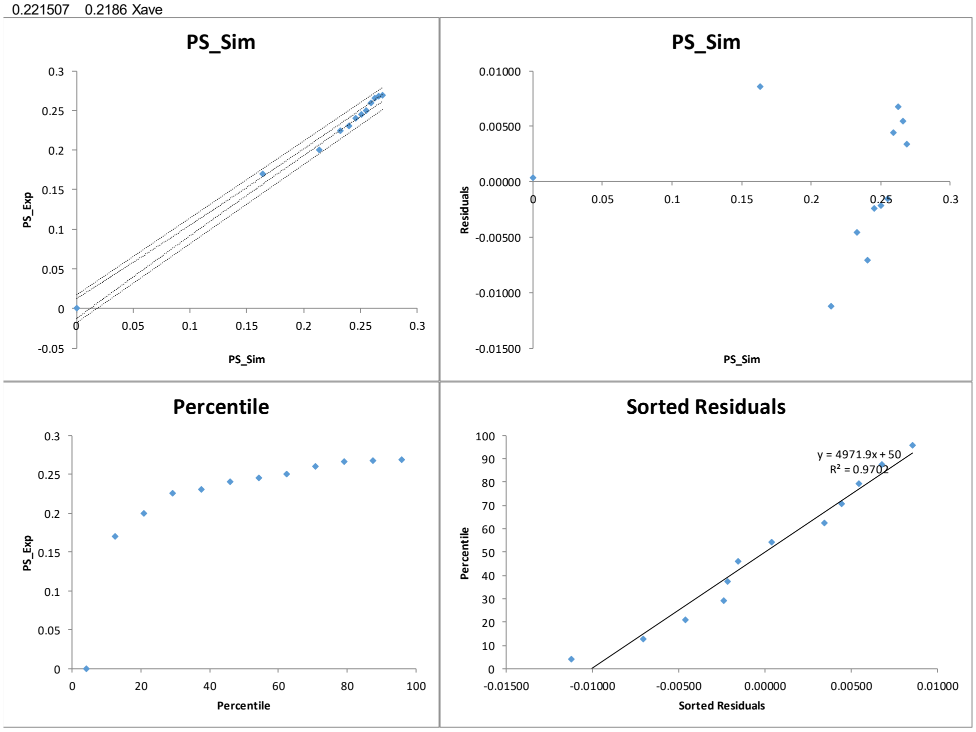

Figure B.3 Primary Sludge statistical results for AD model validation

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SUMMARY OUTPUT — Force Constant to Zero: FALSE

Regression Statistics		
Multiple R	0.999	
R Square	0.999	Goodness of Fit >= 0.80
Adjusted R Square	0.999	
Standard Error	0.001	
Observations	13	

ANOVA

	df	SS	MS	F	P-value
Regression	1	0.004998893	0.004998893	7996.227392	0.000
Residual	11	6.87672E-06	6.25156E-07		
Total	12	0.005005769			

Confidence Level: 0.95, 0.99

	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 99%	Upper 99%
Intercept	-8.70996E-06	0.000714588	-0.012188779	0.990	-0.001581508	0.001564088	-0.00223	0.002211
AS_Sim	0.994689404	0.011123589	89.42162709	0.000	0.97020655	1.019172257	0.960142	1.029237

y = 0 +0.995*AS_Sim

RESIDUAL OUTPUT — PROBABILITY OUTPUT

AS_Sim	AS_Exp		Observations	PredictedAS_E	Residuals	Standard Resi	Sorted Residu	Percentile	AS_Exp	LCI	UCI	LPI	UPI
0	0		1	-0.00001	0.00001	0.01151	-0.00196	3.84615	0	-0.00155	0.001535	-0.00231	0.002294
0.039533	0.04		2	0.03931	0.00069	0.90543	-0.00058	11.53846	0.04	0.038612	0.040017	0.037467	0.041162
0.060292	0.058		3	0.05996	-0.00196	-2.59339	-0.00038	19.23077	0.058	0.059489	0.060437	0.05819	0.061736
0.062927	0.062		4	0.06258	-0.00058	-0.77173	-0.00030	26.92308	0.062	0.062109	0.06306	0.060811	0.064357
0.064657	0.064		5	0.06430	-0.00030	-0.40220	-0.00015	34.61538	0.064	0.063823	0.064786	0.06253	0.066079
0.066276	0.067		6	0.06592	0.00108	1.43228	-0.00005	42.30769	0.067	0.065426	0.066405	0.064139	0.067693
0.067793	0.068		7	0.06742	0.00058	0.75999	0.00001	50.00000	0.068	0.066925	0.067925	0.065645	0.069204
0.069211	0.069		8	0.06884	0.00016	0.21775	0.00016	57.69231	0.069	0.068323	0.069347	0.067052	0.070618
0.070534	0.07		9	0.07015	-0.00015	-0.19969	0.00043	65.38462	0.07	0.069626	0.070676	0.068364	0.071938
0.071766	0.071		10	0.07138	-0.00038	-0.49750	0.00048	73.07692	0.071	0.070838	0.071915	0.069586	0.073168
0.072912	0.073		11	0.07252	0.00048	0.63990	0.00058	80.76923	0.073	0.071964	0.073067	0.070721	0.074311
0.073974	0.074		12	0.07357	0.00043	0.56502	0.00069	88.46154	0.074	0.073007	0.074138	0.071773	0.075372
0.074958	0.0745		13	0.07455	-0.00005	-0.06736	0.00108	96.15385	0.0745	0.073972	0.07513	0.072748	0.076354
0.061141	0.0608	Xave											

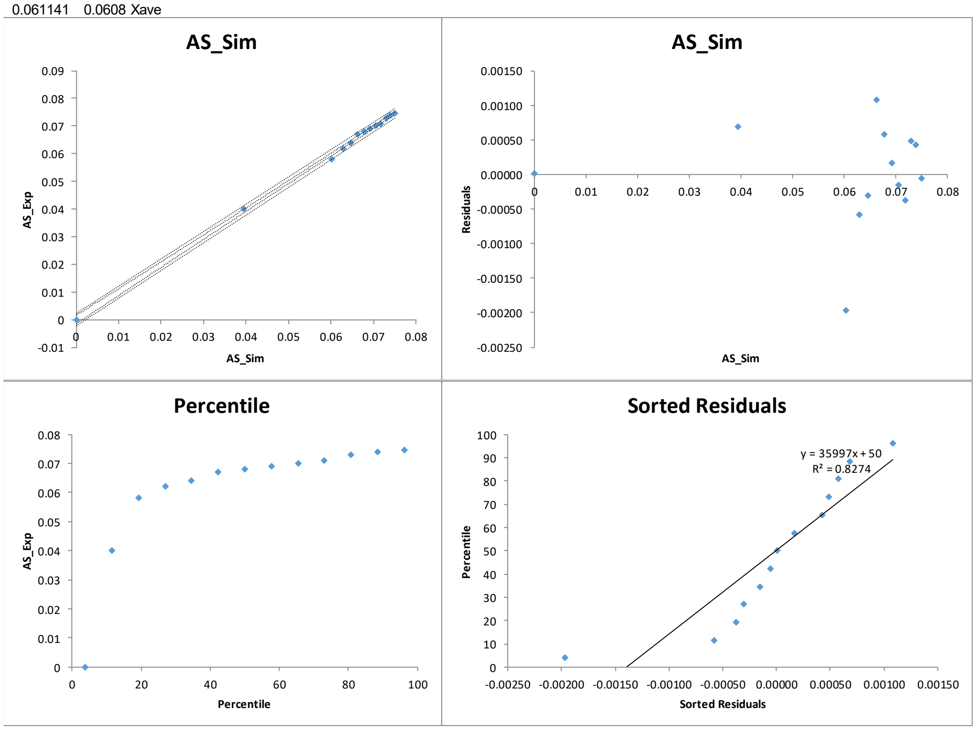

Figure B.4 Bacteria biomass statistical results for AD model validation

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics	
Multiple R	0.997
R Square	0.995 Goodness of Fit >= 0.80
Adjusted R Square	0.994
Standard Error	0.005
Observations	12

ANOVA

	df	*SS*	*MS*	*F*	*P-value*
Regression	1	0.049867925	0.049867925	1957.586072	0.000
Residual	10	0.000254742	2.54742E-05		
Total	11	0.050122667			

Confidence Level

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 99%*	*Upper 99%*
						0.95		0.99
Intercept	0.004312831	0.004620891	0.933333257	0.373	-0.005983155	0.014608818	-0.01033	0.018958
Sim	1.042953053	0.023572428	44.2446163	0.000	0.990430409	1.095475696	0.968246	1.117661

y = 0.004 +1.043*Sim

RESIDUAL OUTPUT

PROBABILITY OUTPUT

Sim	25 Algae/75 PS	*Observations*	*Predicted25 A*	*Residuals*	*Standard Resi*	*Sorted Residu*	*Percentile*	*25 Algae/75 P*	*LCI*	*UCI*	*LPI*	*UPI*
0	0	1	0.00431	-0.00431	-0.89621	-0.00594	4.16667	0	-0.00576	0.014381	-0.0106	0.019222
0.131921	0.155	2	0.14190	0.01310	2.72217	-0.00431	12.50000	0.155	0.137681	0.146119	0.130122	0.153679
0.174617	0.185	3	0.18643	-0.00143	-0.29724	-0.00258	20.83333	0.185	0.183202	0.189659	0.17497	0.197891
0.192419	0.205	4	0.20500	0.00000	0.00067	-0.00239	29.16667	0.205	0.201805	0.208188	0.193546	0.216447
0.200998	0.208	5	0.21394	-0.00594	-1.23527	-0.00147	37.50000	0.208	0.210678	0.217211	0.202473	0.225416
0.207175	0.218	6	0.22039	-0.00239	-0.49591	-0.00143	45.83333	0.218	0.217031	0.223742	0.208889	0.231884
0.212151	0.223	7	0.22558	-0.00258	-0.53538	0.00000	54.16667	0.223	0.22213	0.229023	0.214052	0.237101
0.216188	0.233	8	0.22979	0.00321	0.66767	0.00014	62.50000	0.233	0.226255	0.233319	0.218237	0.241337
0.219741	0.235	9	0.23349	0.00151	0.31336	0.00016	70.83333	0.235	0.229876	0.237108	0.221916	0.245068
0.222969	0.237	10	0.23686	0.00014	0.02933	0.00151	79.16667	0.237	0.233161	0.240557	0.225257	0.248461
0.225828	0.24	11	0.23984	0.00016	0.03314	0.00321	87.50000	0.24	0.236065	0.243616	0.228214	0.251468
0.228353	0.241	12	0.24247	-0.00147	-0.30633	0.01310	95.83333	0.241	0.238627	0.246322	0.230824	0.254125
0.18603	0.198333 Xave											

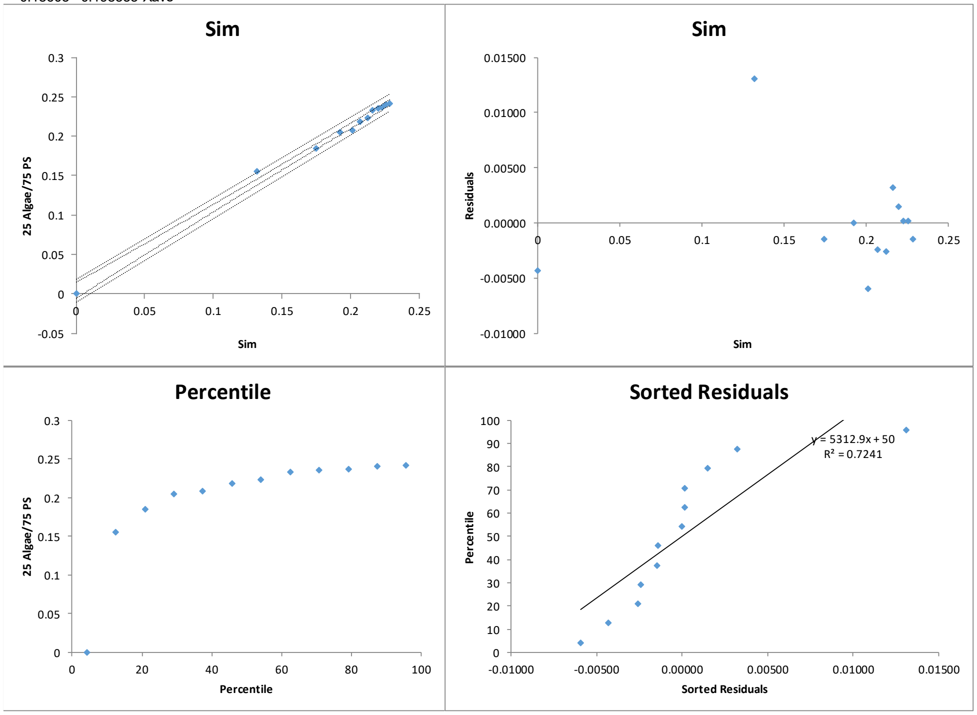

Figure B.5 Co-digestion 75% PS and 25% Algae statistical results for AD model validation

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics		
Multiple R	0.996	
R Square	0.992	Goodness of Fit >= 0.80
Adjusted R Square	0.991	
Standard Error	0.005	
Observations	12	

ANOVA

	df	SS	MS	F	P-value
Regression	1	0.03282844	0.03282844	1205.925211	0.000
Residual	10	0.000272226	2.72226E-05		
Total	11	0.033100667			

Confidence Level

	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 99%	Upper 99%
						0.95		0.99
Intercept	0.005263455	0.004658105	1.129956446	0.285	-0.005115449	0.015642359	-0.0095	0.020026
Sim	1.016717777	0.029277921	34.7264339	0.000	0.951482504	1.081953049	0.923928	1.109507

y = 0.005 +1.017*Sim

RESIDUAL OUTPUT

PROBABILITY OUTPUT

Sim	50 Algae/50 PS	Observations	Predicted50 A	Residuals	Standard Resi	Sorted Residu	Percentile	50 Algae/50 P	LCI	UCI	LPI	UPI
0	0	1	0.00526	-0.00526	-1.05804	-0.00526	4.16667	0	-0.00489	0.015413	-0.00998	0.020503
0.10014	0.121	2	0.10708	0.01392	2.79871	-0.00479	12.50000	0.121	0.102482	0.111672	0.094816	0.119339
0.135177	0.14	3	0.14270	-0.00270	-0.54287	-0.00270	20.83333	0.14	0.139276	0.146126	0.130828	0.154573
0.152193	0.162	4	0.16000	0.00200	0.40184	-0.00203	29.16667	0.162	0.156718	0.163284	0.148168	0.171834
0.16182	0.165	5	0.16979	-0.00479	-0.96256	-0.00182	37.50000	0.165	0.166429	0.173148	0.157934	0.181642
0.168736	0.175	6	0.17682	-0.00182	-0.36599	-0.00106	45.83333	0.175	0.17334	0.180301	0.164932	0.18871
0.173861	0.18	7	0.18203	-0.00203	-0.40817	-0.00103	54.16667	0.18	0.178428	0.185633	0.170105	0.193956
0.177513	0.188	8	0.18574	0.00226	0.45351	-0.00064	62.50000	0.188	0.182039	0.189449	0.173787	0.1977
0.180562	0.19	9	0.18884	0.00116	0.23243	0.00116	70.83333	0.19	0.185045	0.192643	0.176858	0.20083
0.183313	0.191	10	0.19164	-0.00064	-0.12892	0.00200	79.16667	0.191	0.187751	0.195532	0.179626	0.203657
0.185661	0.193	11	0.19403	-0.00103	-0.20667	0.00226	87.50000	0.193	0.190055	0.198001	0.181986	0.20607
0.18766	0.195	12	0.19606	-0.00106	-0.21327	0.01392	95.83333	0.195	0.192015	0.200107	0.183994	0.208128
0.150553	0.158333 Xave											

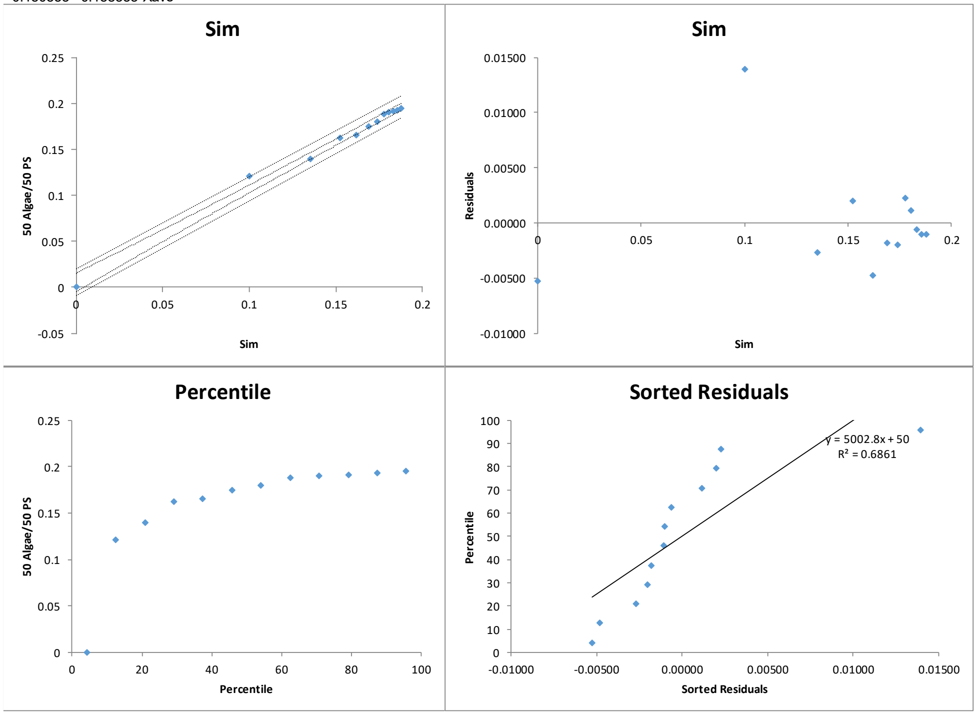

Figure B.6 Co-digestion 50% PS and 50% Algae statistical results for AD model validation

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics	
Multiple R	0.996
R Square	0.993
Adjusted R Square	0.991
Standard Error	5.229
Observations	5

Goodness of Fit >= 0.80

ANOVA

	df	*SS*	*MS*	*F*	*P-value*
Regression	1	11512.5117	11512.5117	421.0795941	0.000
Residual	3	82.02139355	27.34046452		
Total	4	11594.5331			

Confidence Level

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 99%*	*Upper 99%*
						0.95		0.99
Intercept	61.54913129	17.21101023	3.576148666	0.037	6.776015354	116.3222472	-38.9788	162.0771
Exp	0.837078641	0.040792861	20.52022403	0.000	0.707257552	0.96689973	0.598811	1.075346

y = 61.549 +0.837*Exp

RESIDUAL OUTPUT

PROBABILITY OUTPUT

Exp	Sim	*Observations*	*PredictedSim*	*Residuals*	*Standard Resi*	*Sorted Residu*	*Percentile*	*Sim*	*LCI*	*UCI*	*LPI*	*UPI*
350	350	1	354.52666	-4.52666	-0.99964	-5.11991	10.00000	350	345.2005	363.8528	338.1669	370.8864
370	374.8826	2	371.26823	3.61438	0.79818	-4.52666	30.00000	374.8826117	363.4281	379.1083	355.7077	386.8288
405	405.0068	3	400.56598	4.44086	0.98069	1.59132	50.00000	405.0068394	394.4023	406.7297	385.779	415.3529
460	441.4854	4	446.60531	-5.11991	-1.13065	3.61438	70.00000	441.4853961	439.1535	454.0571	431.2368	461.9739
505	485.8652	5	484.27385	1.59132	0.35142	4.44086	90.00000	485.8651691	473.3486	495.1991	466.9527	501.595
418	411.448 Xave											

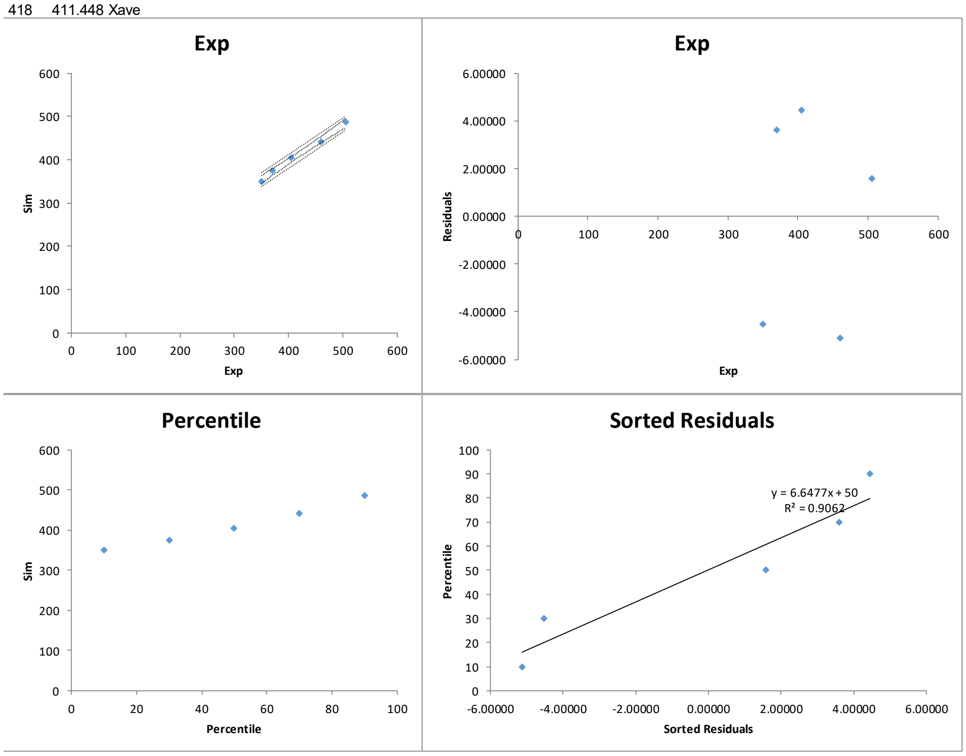

Figure B.7 HRAP simulation-Bai experimental statistical validation result

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SUMMARY OUTPUT

Force Constant to Zero: FALSE

Regression Statistics		
Multiple R	0.991	
R Square	0.982	Goodness of Fit >= 0.80
Adjusted R Square	0.979	
Standard Error	7.271	
Observations	9	

ANOVA

	df	SS	MS	F	P-value
Regression	1	19973.23277	19973.23277	377.8324986	0.000
Residual	7	370.0386545	52.86266493		
Total	8	20343.27143			

Confidence Level: 0.95, 0.99

	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 99%	Upper 99%
Intercept	17.21538716	21.0201119	0.818995981	0.440	-32.48927921	66.92005353	-56.3441	90.77492
Bai Sim	0.941675828	0.048445313	19.43791395	0.000	0.827120865	1.05623079	0.772142	1.111209

y = 17.215 +0.942*Bai Sim

Bai Sim	Sim	
350	350	
387	384.3728	
375	374.8826	
435	415.2596	
415	405.0068	
470	452.6617	
455	441.4854	
502	498.1649	
490	485.8652	
431	423.0777	Xave

RESIDUAL OUTPUT / PROBABILITY OUTPUT

Observations	PredictedSim	Residuals	Standard Resi	Sorted Residu	Percentile	Sim	LCI	UCI	LPI	UPI
1	346.80193	3.19807	0.47023	-11.58474	5.55556	350	336.3685	357.2353	327.3244	366.2794
2	381.64393	2.72888	0.40124	-7.14137	16.66667	374.8826117	374.3426	388.9452	363.6488	399.6391
3	370.34382	4.53879	0.66736	-4.19249	27.77778	384.3728102	362.1145	378.5731	351.9526	388.7351
4	426.84437	-11.58474	-1.70336	-3.00402	38.88889	405.0068394	421.3444	432.3443	409.5017	444.187
5	408.01086	-3.00402	-0.44170	2.72888	50.00000	415.2596364	402.2548	413.7669	390.5853	425.4364
6	459.80303	-7.14137	-1.05003	3.19807	61.11111	441.4853961	452.8514	466.7546	441.9469	477.6592
7	445.67789	-4.19249	-0.61644	4.53879	72.22222	452.6616521	439.5972	451.7586	428.1424	463.2133
8	489.93665	8.22825	1.20984	7.22863	83.33333	485.8651691	480.4182	499.4551	470.9336	508.9398
9	478.63654	7.22863	1.06286	8.22825	94.44444	498.1649044	470.1592	487.1139	460.133	497.1401

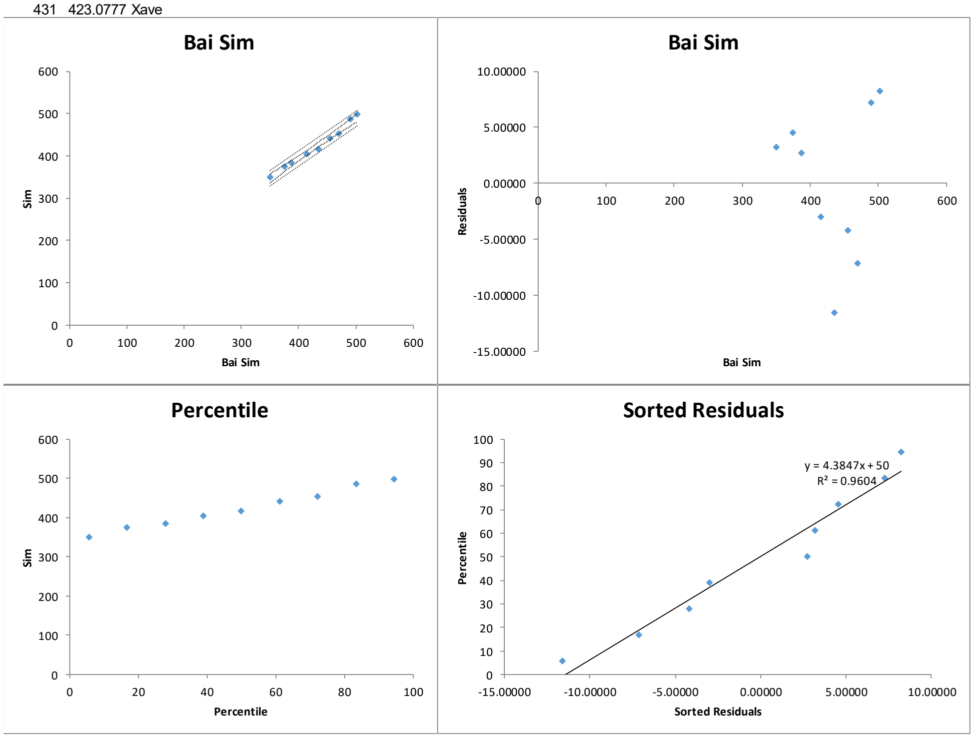

Figure B.8 HRAP simulation-Bai simulation statistical validation result

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SUMMARY OUTPUT

Force Constant to Zero
FALSE

Regression Statistics		
Multiple R	0.991	
R Square	0.982	Goodness of Fit >= 0.80
Adjusted R Square	0.979	
Standard Error	7.271	
Observations	9	

ANOVA

	df	*SS*	*MS*	*F*	*P-value*
Regression	1	19973.23277	19973.23277	377.8324986	0.000
Residual	7	370.0386545	52.86266493		
Total	8	20343.27143			

							Confidence Level	
						0.95		0.99
	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 99%*	*Upper 99%*
Intercept	17.21538716	21.0201119	0.818995981	0.440	-32.48927921	66.92005353	-56.3441	90.77492
Bello Sim	0.941675828	0.048445313	19.43791395	0.000	0.827120865	1.05623079	0.772142	1.111209

y = 17.215 +0.942*Bello Sim

RESIDUAL OUTPUT

PROBABILITY OUTPUT

Bello Sim	Sim	*Observations*	*PredictedSim*	*Residuals*	*Standard Resi*	*Sorted Residu*	*Percentile*	*Sim*	*LCI*	*UCI*	*LPI*	*UPI*
350	350	1	346.80193	3.19807	0.47023	-11.58474	5.55556	350	336.3685	357.2353	327.3244	366.2794
387	384.3728	2	381.64393	2.72888	0.40124	-7.14137	16.66667	374.8826117	374.3426	388.9452	363.6488	399.6391
375	374.8826	3	370.34382	4.53879	0.66736	-4.19249	27.77778	384.3728102	362.1145	378.5731	351.9526	388.7351
435	415.2596	4	426.84437	-11.58474	-1.70336	-3.00402	38.88889	405.0068394	421.3444	432.3443	409.5017	444.187
415	405.0068	5	408.01086	-3.00402	-0.44170	2.72888	50.00000	415.2596364	402.2548	413.7669	390.5853	425.4364
470	452.6617	6	459.80303	-7.14137	-1.05003	3.19807	61.11111	441.4853961	452.8514	466.7546	441.9469	477.6592
455	441.4854	7	445.67789	-4.19249	-0.61644	4.53879	72.22222	452.6616521	439.5972	451.7586	428.1424	463.2133
502	498.1649	8	489.93665	8.22825	1.20984	7.22863	83.33333	485.8651691	480.4182	499.4551	470.9336	508.9398
490	485.8652	9	478.63654	7.22863	1.06286	8.22825	94.44444	498.1649044	470.1592	487.1139	460.133	497.1401
431	423.0777 Xave											

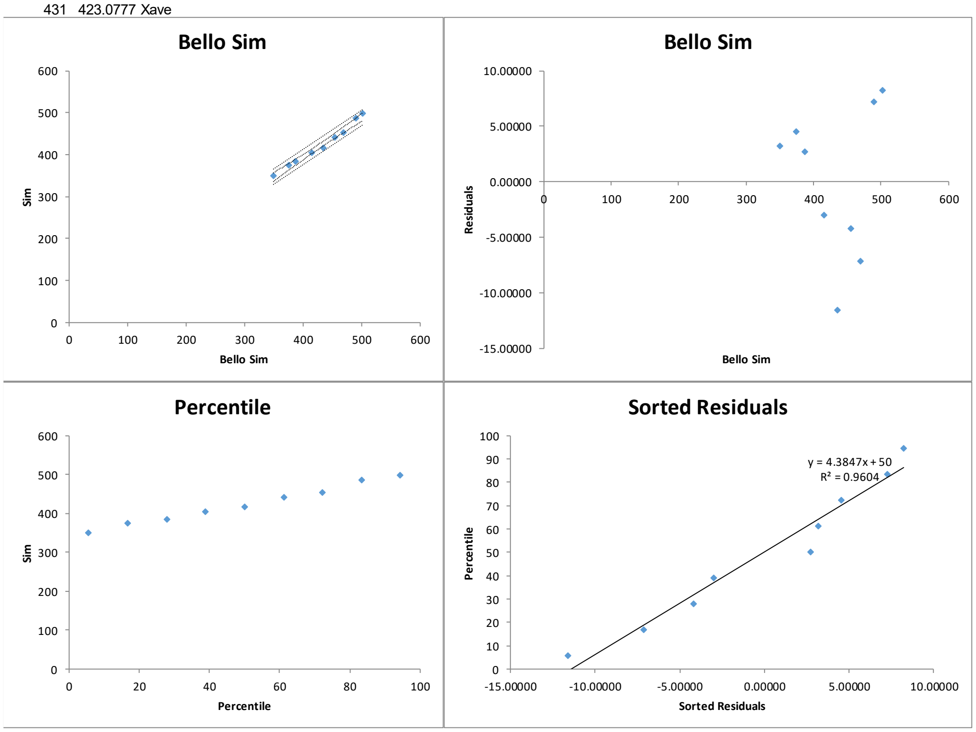

Figure B.9 HRAP simulation-Bello simulation statistical validation result

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