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Life Cycle Environmental Impacts for Anaerobic Treatment of Domestic Wastewater

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Life Cycle Environmental Impacts for Anaerobic Treatment of Domestic Wastewater

by

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ABSTRACT

The electricity requirement for aeration in conventional activated sludge treatment of domestic wastewater contributes to high costs and greenhouse gas emissions. Anaerobic treatment, on the other hand, requires no aeration and produces methane that can be converted to electricity and heat. However, the effluent from anaerobic treatment contains dissolved methane—a potent greenhouse gas—that would ultimately be released to the atmosphere. Therefore, there is a need to assess the life cycle environmental and economic impacts of anaerobic and aerobic treatment technologies to understand the long-term sustainability of these wastewater treatment options.

The first objective of this research was to measure the performance of a laboratory-scale Anaerobic Baffled Reactor (ABR) treating municipal wastewater at 15°C and 20°C. At 15–20°C treatment, the ABR produced an average of 0.17 m^3 of biogas per kg COD removed (approximately half of the theoretical methane yield) and the average concentration of BOD_5 in the effluent was 72 mg/L which is nearly 2.5 times greater than the EPA discharge limits of 30 mg/L. The effluent also contained approximately 28 mg/L of dissolved methane on average which represented nearly 35 percent of the total methane produced by the ABR and, therefore, represented a significant loss of energy (e.g., electricity and heat) and a liability with respect to climate change.

The second objective of this research was to use life cycle assessment (LCA) and techno-economic assessment (TEA) to assess the environmental and economic performance of an ABR, aerobic trickling filter, ABR with trickling filter post-treatment (ABR + Trickling Filter), and ABR with constructed wetland post-treatment (ABR + Constructed Wetland) for treatment of 2 MGD. Full-scale ABR performance was modeled using measured data from the laboratory-scale

ABR and sensitivity and uncertainty analyses were included to account for variability in LCA parameters.

LCA results showed that the ABR + Trickling Filter assembly was the most environmentally preferable technology with net beneficial impacts on human health, ecosystem quality, resource depletion, and climate change for the base case. Uncertainty analysis, on the other hand, showed that the ranges of impacts overlap for all damage categories except climate change (i.e. ecosystem quality, resource depletion, and human health). This suggests that the variability in overall model impacts may be too great to determine a single treatment system which is most preferential for these impacts. The ABR + Trickling Filter (base case), however, resulted in significant benefits to climate change, as this assembly sequestered approximately 650 kg CO_2-eq per day due to electricity produced via cogeneration, which provides a substantial advantage to the 4500 kg CO_2-eq emitted by the ABR and 4200 kg CO_2-eq emitted by the ABR + Constructed Wetland each day. The poor performances of the ABR and ABR + Constructed Wetland can be attributed to dissolved methane in ABR effluent which contributed nearly 95 percent of the total impact on climate change and was nearly five times greater than the beneficial impact on climate from bio-electricity generation. However, if dissolved methane was captured as biogas instead of being released with ABR effluent, the climate change impact would be eliminated, and the energy production of the ABR would increase to 9.7 MJ/kg COD which would likely make the ABR or ABR + Constructed Wetlands more ideal than the Trickling Filter or ABR + Trickling Filter from environmental and economic perspectives.

TEA results indicated that each assembly had a payback period less than the plant life and, therefore, were economically viable technologies. The ABR was the most ideal assembly in terms of capital cost, operating costs, and present worth due to its low energy requirements, low

solids production, and revenue generated from cogeneration. Specifically, the present worth of the ABR was nearly 36 percent lower than the Trickling Filter, 32 percent lower than the ABR + Trickling Filter, and 14 percent lower than the ABR + Constructed Wetland. The higher costs of the Trickling filter assembly is largely due to anaerobic digestion (used to digest solids from primary treatment, as well as solids produced in the trickling filter), which contributed approximately 35 percent of total capital costs, and disposal of solids which cost more than $36,000 annually compared to the ABR. Cogeneration was relatively inexpensive (less than5 percent of total capital costs) for all assemblies, and provided the additional benefit of annual revenue ranging from $28,000-$46,600 which suggests that it may be an economically viable option for 2 MGD treatment. Finally, the economic analysis showed that implementation of constructed wetlands as a post-treatment instead of trickling filters as a post treatment would avoid nearly $2,051,000 in capital costs and $101,700 per year in operating costs. Furthermore, if no liner was required for the constructed wetland, its capital cost would be reduced by an additional $494,000, making constructed wetlands even more economically favorable.

No single technology was optimal from both an environmental and economic perspective since the ABR and ABR + Constructed Wetland had the lowest present worth and the ABR + Trickling Filter and Trickling Filter had the lowest environmental impact. However, if methods are implemented to extract dissolved methane prior to effluent release, the ABR would likely be the most ideal technology from both an environmental and economic perspective.

NOMENCLATURE

ABR	Anaerobic Baffled Reactor
AD	Anaerobic Digestion
AeMBR	Aerobic Membrane Bioreactor
AnMBR	Anaerobic Membrane Bioreactor
BFD	Block Flow Diagram
BOD	Biochemical Oxygen Demand
CAS	Conventional Activated Sludge
CI	Confidence Interval
COD	Chemical Oxygen Demand
CW	Constructed Wetland
EPA	Environmental Protection Agency
FID	Flame Ionization Detector
FWS	Free Water Surface
GC	Gas Chromatography
GHG	Greenhouse Gas
GWP	Global Warming Potential
HRAS	High Rate Activated Sludge
HRT	Hydraulic Retention Time
IMPACT	Impact Assessment of Chemical Toxicants
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDPE	Low-density Polyethylene
MDL	Minimum Detection Limit
MGD	Million Gallons per Day
MACRS	Modified Accelerated Cost Recovery System
MRSA	Milton Regional Sewer Authority
PA	Partial Alkalinity
PVDC	Polyvinylidene Chloride
SCOD	Soluble Chemical Oxygen Demand
SF	Safety Factor
SRT	Solids Retention Time
TCOD	Total Chemical Oxygen Demand
TCD	Thermal Conductivity Detector
TEA	Techno-economic Analysis
TRACI	Tool for the Reduction and Assessment of Chemical and other environmental Impacts
TS	Total Solids
TSS	Total Suspended Solids
VA	Volatile Alkalinity
VS	Volatile Solids
VSB	Vegetated Submerged Bed
VSS	Volatile Suspended Solids

CHAPTER 1

INTRODUCTION

1.1 Conventional wastewater treatment deficiencies

Approximately 80 percent of the wastewater produced worldwide remains untreated before being discharged to waterways which poses an immediate and significant threat to human health and the environment (Tauseef et al. 2013). A majority of populations suffering from a lack of proper sanitation reside in the developing world where implementation of conventional wastewater treatment methods has been hindered by process dependence on reliable electricity that is costly, unreliable, or unavailable in the developing world. The few developing countries that utilize conventional wastewater treatment may allocate over 50 percent of their total municipal budget to the energy intensive process of water collection, aerobic treatment, and distribution (ASE 2002).

Conventional treatment of municipal wastewater utilizes aerobic microorganisms to biodegrade organic matter in the wastewater to meet the 30 day, 30 mg/L BOD and TSS secondary treatment effluent standards required by the EPA. Aerobic treatment is effective from an effluent quality perspective, which has resulted in its widespread implementation in the developed world. However, aerobic microbes require a steady supply of oxygen which corresponds to high electricity demand and substantial greenhouse gas (GHG) emissions. In the United States alone, wastewater treatment generates 45 million tons of GHGs, which represents approximately 5 percent of the total U.S. emissions (USEPA 2013).

1.2 Anaerobic wastewater treatment as an alternative

The negative impacts of aerobic treatment may be overcome via implementation of an alternative technology: anaerobic treatment. Anaerobic treatment utilizes anaerobic microorganisms which do not require oxygen to biodegrade organics. Therefore, wastewater treatment using anaerobic organisms does not require electricity for aeration and, consequently, reduces anthropogenic carbon dioxide emissions. Furthermore, a high-energy biogas containing 50 to 70 percent methane is produced as a byproduct of anaerobic digestion. Biogas is a renewable energy resource that can be used to generate electricity and heat that may result in a net positive wastewater treatment process.

Although anaerobic treatment has potential as an alternative to aerobic treatment, it may not be as environmentally beneficial as previously assumed due to high levels of dissolved methane present in anaerobic effluent (Liu et al. 2014). Dissolved methane represents lost energy production in anaerobic systems because only gaseous methane can be harnessed for electricity production. In addition, methane trapped in the aqueous phase is eventually discharged to waterways. Discharged methane— a potent GHG with a global warming potential (GWP) approximately 28 times greater than carbon dioxide (Myhre et al. 2013)—is eventually released to the atmosphere which may offset reductions in GWP compared to aerobic treatment (Cakir and Stenstrom, 2005).

There are numerous pros and cons, with respect to environmental impacts and economics, associated with aerobic and anaerobic wastewater treatment technologies. Consequently, robust analyses that quantify the environmental sustainability and economic viability of wastewater treatment technologies are necessary to properly evaluate the performance of treatment systems and make appropriate recommendations for implementation.

1.3 Evaluation and comparison of wastewater treatment technologies

Life cycle assessment (LCA) is one commonly used tool for quantification and comparison of the environmental impacts of wastewater treatment technologies. LCA is a methodology used to evaluate environmental impacts of a product or process over its entire lifetime—from the impacts generated by the extraction of raw materials to the final disposal of generated waste. LCA models are used to compare multiple technologies and identify the contribution of specific process components to the overall environmental footprint of a given process. Although LCA has historically been used independently, coupling LCA with economic evaluations, such as techno-economic analysis (TEA)—resulting in an "environomic" model—prevents modelers from recommending process alternatives that are environmentally favorable but not economically viable (Gerber et al. 2011). Furthermore, conducting a comprehensive analysis of uncertainty in conjunction with "environomic" modeling produces a range of model results instead of single point values. Providing stakeholders and decision makers with ranges of results instead of single values allows them to draw more meaningful conclusions from LCA and economic models (Gerber et al. 2011; Sills et al. 2013).

1.4 Goals and objectives

The goal of this research was to quantify the environmental impacts and economic viability of full-scale domestic wastewater treatment using an aerobic trickling filter, anaerobic baffled reactor (ABR), ABR with trickling filter post-treatment (referred to as ABR + Trickling Filter), and ABR with constructed wetlands post-treatment (referred to as ABR + Constructed Wetland). The assessed impacts reflect 15-20°C treatment and include impacts generated from dissolved methane present in anaerobic effluent. The first task implemented to achieve the goal

was to operate a laboratory scale ABR at15°C and 20°C to obtain steady-state treatment performance and dissolved methane concentration values to be used in environmental and economic models. The second task was to construct LCA models, coupled with uncertainty analysis, for full-scale trickling filter, ABR, ABR + Trickling Filter, and ABR + Constructed Wetland treatment to compare environmental impacts of the various methods in terms of climate change, ecosystem quality, human health, and nonrenewable resource consumption. The third and final task was to conduct a TEA of each treatment system to assess and compare total lifetime costs.

The results obtained via the objectives were combined into an overall "environomic" analysis which compared the trickling filter, ABR, ABR + Trickling Filter, and ABR + Constructed Wetland and identified environmental and economic tradeoffs associated with each treatment technology. A comprehensive uncertainty analysis was coupled with LCA to ensure robust analysis and interpretation of results.

1.5 Organization

In Chapter 2, we present a literature review of the ABR, dissolved methane in anaerobic effluent, trickling filter modeling, post-treatment with constructed wetlands, LCA, TEA, and the combined LCA and TEA approach used for evaluation of wastewater treatment technologies. In Chapter 3, an experimental study, which used a bench-scale ABR to treat municipal wastewater at low temperatures, is described. In Chapter 4, a study which used LCA to assess the environmental impacts of an ABR, trickling filter, ABR with trickling filter post-treatment, and ABR with constructed wetlands post-treatment is described. Chapter 4 also details the methods used to for system design and development of input-output models for use in LCA and TEA. In Chapter 5, a

study which used TEA to assess the economic impacts of an ABR, trickling filter, ABR with trickling filter post-treatment, and ABR with constructed wetlands post-treatment is described. Chapter 6 presents a summary of the conclusions reached in this study and their broader implications, and suggests topics that should be addressed in future work.

CHAPTER 2

LITERATURE REVIEW

2.1 Anaerobic Baffled Reactor

The ABR converts organic waste to methane and carbon dioxide. An ABR consists of up to eight compartments separated by vertical baffles that direct incoming wastewater through a series of upflow anaerobic sludge blankets (Figure 2.1) (Krishna et al. 2008). The resulting flow pattern is an intermediate between plug flow and perfectly mixed flows (Stuckey 2010). Krishna et al. (2009) showed that three compartments were adequate for treating dilute wastes (e.g., municipal wastewater), whereas four to eight compartments were required for treating wastes with high concentrations of organic matter. ABRs have a simplistic design with no moving parts or mechanical mixing, which translates to low capital and operating costs (Barber and Stuckey 1999).

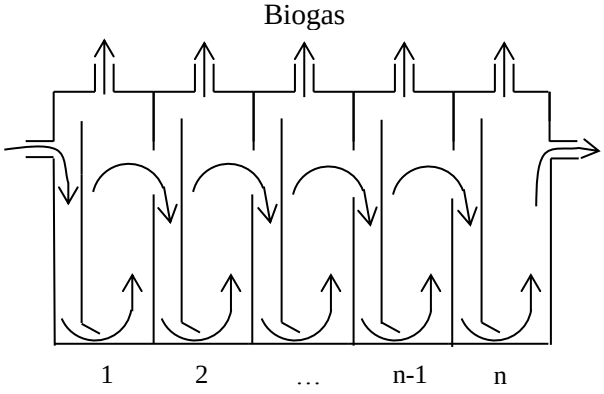


Figure 2.1. n-chambered ABR design (adapted from Barber and Stuckey, 1999)

There are multiple advantages to using the ABR. For instance, the compartmentalized design of ABRs allows the reactor to behave as a two-phase system that separates acidogenesis and methanogenesis longitudinally down the reactor which provides favorable conditions for the microorganisms in each phase to develop (Cohen et al. 1980). Two-phase systems have the potential to quadruple the activity of methane-producing methanogens in one-phase systems, which results in a greater potential for renewable energy production (Cohen et al. 1980). Consequently, the ABR is able to achieve significant gas production with a small reactor volume (Bachmann et al. 1985). Specifically, the ABR reportedly achieves 47.5 percent energy recovery which is greater than the energy recoveries for other anaerobic treatment technologies including the upflow anaerobic sludge blanket (24 percent), anaerobic membrane bioreactor (35.4 percent), anaerobic fluidized bed reactor (33.8 percent), anaerobic sequencing batch reactor (17.7 percent), microbial electrolysis cell (14.3 percent), and microbial fuel cell (1.6 percent) (Shoener et al. 2014). Consequently, the ABR is the most preferable of the anaerobic technologies from an energy production perspective.

A further advantage of the ABR is its design which prevents sludge bed expansion – a common problem for the Upflow Anaerobic Sludge Blanket Reactor (UASB) and anaerobic filter (Manariotis and Grigoropoulos, 2002). In addition, the ABR decouples SRT and HRT which simultaneously retains slow-growing anaerobes in the reactor and treats large wastewater volumes while minimizing reactor volume (Nachaiyasit and Stuckey, 1997) and its compartmentalized design which minimizes the effects of hydraulic and organic shocks (Manariotis and Grigoropoulos, 2002).

The disadvantages of the ABR (and anaerobic treatment in general) are that reactor performance deteriorates as operating temperature or influent substrate concentration decreases

(Langenhoff and Stuckey, 2000) and low pathogen removal (Dama, 2002; Foxon et al. 2004). Dissolved methane remaining in the reactor effluent represents an additional drawback, as it contributes to greenhouse gas emissions if the effluent is not treated before being released to waterways (Cakir and Stenstrom, 2005).

Several laboratory- (Krishna et al. 2008; Nasr et al. 2009; Wang et al. 2004; Manariotis and Grigoropoulos, 2002; Yu and Anderson, 1996; She et al. 2013; Bodkhe 2009; Feng et al. 2008; Jamshidi et al. 2014) and pilot- scale studies (Foxon et al. 2004; Singh et al. 2009; Dama 2002; Motteran et al. 2013) have been conducted to assess the performance of the ABR under various operating conditions. A pilot-scale ABR treating a mixture of 50 percent domestic wastewater and 50 percent textile effluent was able to achieve COD removal of 70-80 percent and a pH between 5.5 and 7.5 which is sufficient for discharge in South Africa (Dama, 2002). However, a pilot-scale ABR treating domestic wastewater achieved 50-75 percent COD removal, and produced effluent which required post-treatment to remove pathogens before the water could be discharged or reused for irrigation (Foxon et al. 2004). Constructed wetlands proved to be an effective post-treatment of effluent from a pilot-scale ABR treating domestic wastewater with the system achieving removal of 98 percent of fecal coliforms, 90 percent of COD, and 96 percent of total suspended solids (Singh et al. 2009). Performance data for ABRs that treated synthetic and municipal wastewaters are described in Table 2.1 and Table 2.2, respectively.

Table 2.1. Review of ABR performance for treatment of synthetic wastewater. Values are represented as "mean ± standard deviation" and "minimum – maximum".

Volume (L)	Feed (mg/L COD)	Avg. Temp. (°C)	HRT (hrs)	OLR (kg COD/m^3 d)	COD Removal (%)	CH_4 Prod (m^3/kg COD_r)	References
10	504 ± 7.6	26.9	20	0.61 ± 0.01	92.5 ± 0.8	0.18 ± 0.01	Krishna et al. 2008
		24	15	0.81 ± 0.01	90.5 ± 0.9	0.21 ± 0.01	
		27	10	1.20 ± 0.02	91.0 ± 0.8	0.21 ± 0.01	
		31	8	1.51 ± 0.02	89.0 ± 0.9	0.23 ± 0.01	
		29	6	2.00 ± 0.03	88.7 ± 1.2	0.19 ± 0.01	
10	500	35	10	-	95	-	Langenhoff and Stuckey, 2000
		20		-	70	-	
		10		-	60	-	
6060	14381	20	18	17.85 ± 5.1	80	-	Motteran et al. 2013
13	8000	-	50	2.5	93	0.35	Bachmann et al. 1985
		-	48	4.2	88	0.45	
		-	22	8.9	81	0.37	
		-	18	11.4	91	0.36	
		-	12	15.3	77	0.31	
		-	10	20	75	0.28	
		-	6.7	27.3	68	0.27	
		-	5.6	31.8	55	0.30	
		-	4.8	36.2	60	0.26	
14.7	400 ± 104	26.6	24	0.4	87.2	0.184	Manariotis and Grigoropoulos, 2002
	318 ± 32	24.5	12	0.636	91	0.102	
	282 ± 38	25.7	300-48	0.024-0.130	85.3	-	
	303 ± 37	25.9	24	0.303	91.9	0.077	
	330 ± 18	26.1	12	0.662	82.6	0.198	
	318 ± 54	25.9	24	0.318	91.4	0.211	
	330 ± 25	15.6	24	0.33	91.4	0.16	
	331 ± 18	15.4	12	0.662	83.7	0.136	

Volume (L)	Feed (mg/L COD)	Avg. Temp. (°C)	HRT (hrs)	OLR (kg COD/m^3 d)	COD Removal (%)	CH_4 Prod (m^3/kg COD_r)	References
92.4	481 ± 21	-	48	-	93.5	-	Sarathai et al. 2010
		-		-	91.6	-	
		-		-	92.6	-	
		-		-	87.7	-	
	877 ± 91	-	48	-	94.5	-	
		-		-	92.0	-	
		-		-	91.5	-	
		-		-	92.6	-	
	1404 ± 219	-	48	-	95.2	-	
		-		-	95.6	-	
		-		-	95.5	-	
		-		-	96.5	-	
17.8	300	20.3 ± 0.7	0.77 ± 0.08	0.40 ± 0.06	85.3 ± 4.5	0.11 ± 0.02	Wacker, 2014
			0.52 ± 0.02	0.59 ± 0.04	79.1 ± 3.0	0.11 ± 0.01	
			0.52 ± 0.02	0.58 ± 0.03	74.9 ± 2.9	0.11 ± 0.02	
100	1462-1720	17-25	24	1.46-1.72	>90	-	She et al. 2013
	936-1110		24	0.94-1.11		-	
	441-602		24	0.44-0.60		-	
	432-535		12	0.86-1.07	83-89	-	
	453-529		8	1.36-1.59		-	
10	4000	35	20	-	96[1]	-	Nachaiyasit and Stuckey, 1997
		25		-	93[1]	-	
		15		-	83[1]	-	
10	4000	-	24	4	74	-	Ruchiraset and Chinwetkitvanich, 2009
		-		4	78	-	
		-		4	83	-	

[1] SCOD removal

Table 2.2. Review of ABR performance for treatment of municipal wastewater. Values are represented as "mean ± standard deviation" and "minimum – maximum".

Volume (L)	Feed (mg/L COD)	Avg. Temp. (°C)	HRT (hrs)	OLR (kg COD/m^3 d)	COD Removal (%)	CH_4 Prod (m^3/kg COD_r)	References
15	682 ± 154.1	25.1	24	0.669	82.0 ± 4.8	-	Nasr et al. 2009
		23.3	18	0.958	79.7 ± 5.2	-	
		23	12	1.3	75.6 ± 4.6	-	
		24	8	2.1	67.5 ± 7.0	-	
3000	716 ± 54.4	-	22	-	72 ± 3	-	Foxon et al. 2004
42000	2914 ± 1406	-	28.8	-	47.2 ± 26.1	-	Singh et al. 2009
10.8	386	18-28	10	0.92	83	0.09	Yu and Anderson, 1996
	394		9	1.05	81	0.10	
	398		8	1.19	79	0.10	
	413		7	1.42	75	0.12	
	447		6	1.79	73	0.11	
	482		5	2.31	70	0.12	
	405		4	2.43	68	0.10	
16	564	21.1	24	0.58 ± 0.20	86 ± 4.4	0.13 ± 0.06	DiStefano, 2010
3000	360-930	-	42	-	83	-	Hudson, 2011
32	400	35	6	1.6	84	0.34	Bodkhe, 2009
17	305 ± 36	28	48	-	79	-	Feng et al. 2008
			18	-	69	-	
60	736 ± 51.3	25-29	15	0.622	43[1]	-	Jamshidi et al. 2014
	713 ± 21.4		11	0.843	59[1]	-	
3200	350-1200	-	60	-	<60	-	Dama, 2002
		-	32	-	80	-	
		-	20	-	70-90	-	

[1]calculated based on ABR influent and effluent COD data

Tables 2.1 and 2.2 show that a majority of ABR studies were conducted at temperature of 20°C and above. However, the average temperature of domestic wastewater in the United States is 16°C (Tchobanoglous et al. 2003), and performance of the ABR has been shown to start decreasing at 20°C (Langenhoff and Stuckey, 2000). Although psychrophilic methanogens can oxidize organic matter at 1°C, (Stuckey, 2010), reducing temperature of an ABR from 35ºC to

10ºC reduced COD removal from 95 percent to 60 percent for the treatment of synthetic wastewater with 500 mg/L COD (Langenhoff and Stuckey, 2000). On the other hand, when the temperature was decreased from 25°C to 15°C during treatment of a synthetic wastewater with 300 mg/L COD no negative effects on organics removal were observed, although total suspended solids removal decreased with temperature (Manariotis and Grigoropoulos, 2002). Nachaiyasit and Stuckey showed that the COD removal for an ABR that treated wastewater with 4000 mg/L COD with a 20 hour HRT decreased from 93 percent to 83 percent when temperature was decreased from 25°C to 15°C (Nachaiyasit and Stuckey, 1997).

One study which assessed low temperature treatment of municipal wastewater (386-482 mg/L as COD) between 18°C and 28°C using a modified ABR showed that HRT can also have an impact on reactor performance (Yu and Anderson, 1996). Specifically, 83 percent COD removal could be achieved at a 10 hour HRT, but removal efficiency decreased when HRT was decreased from 4 hours to 2 hours (Yu and Anderson, 1996). Furthermore, decreasing HRT from 24 hours to 8 hours resulted in decreased COD removal from 82.0 percent to 67.5 percent (Nasr et al. 2009).

To enhance understanding of the relationships between ABR operating parameters and performance, we conducted a correlation analysis for the parameters listed in Table 2.1 and 2.2. Correlations that visually appeared to be linear were analyzed with the Spearman method (Gardener, 2012), whereas correlations that visually appeared non-linear were analyzed with the Pearson method (Gardener, 2012). Parameters with significant correlation, i.e. $p \leq 0.05$, are summarized in Table 2.3, and plots of the correlation analysis are presented in Figures A1-A4 in the appendix.

Table 2.3. ABR parameter pairings resulting in a statistically significant correlation ($p \leq 0.05$)

Feed Type	Operational Parameter	Performance Parameter	R^2	Method
Municipal	Feed Concentration	Spec. CH_4 Prod. (gas)	0.68	Spearman
Municipal	Feed Concentration	COD Removal	-0.55	Pearson
Municipal	Temperature	Spec. CH_4 Prod. (gas)	0.96	Pearson
Synthetic	Temperature	Spec. CH_4 Prod. (gas)	0.58	Spearman
Synthetic	OLR	Spec. CH_4 Prod. (gas)	0.78	Spearman
Synthetic	Temperature	COD Removal	0.68	Pearson
Synthetic	HRT	COD Removal	0.57	Pearson
Synthetic	OLR	COD Removal	-0.82	Pearson
Synthetic	Feed Concentration	Spec. CH_4 Prod. (gas)	0.85	Pearson

The correlation analysis showed that feed concentration and temperature are significantly correlated with gaseous specific methane production regardless of feed type (municipal or synthetic). In addition, COD removal is significantly correlated to temperature, HRT, and OLR for systems treating synthetic wastewater. Finally, there is a significant correlation between OLR and gaseous specific methane production in ABRs that treat synthetic wastewater.

In summary, numerous studies have assessed the performance of the ABR. However, none have assessed the lifetime environmental and economic impacts of wastewater treatment with an ABR. Since over half of the total methane produced by anaerobic systems may be lost in anaerobic effluent (Noyola et al. 2006), there is a need to quantify the concentrations, impacts and recovery of dissolved methane in ABR effluent to properly evaluate the technology (Liu et al. 2014). Only one study, however, considered dissolved methane in its performance analysis of an ABR, but the environmental impacts associated with that methane release to the atmosphere were not discussed (Krishna et al. 2008). There is a need to quantify the concentrations, impacts, and recovery of dissolved methane in ABR effluent to properly evaluate the technology (Liu et al. 2014).

2.2 Dissolved Methane in Anaerobic Effluent

Anaerobic digestion produces methane that, at equilibrium, partitions between the gas and liquid phases according to Henry's law. Methane recovered from the gas phase can be used to produce renewable energy. However, an anaerobic system, treating domestic wastewater at 30°C, may contain nearly 45 percent of produced methane in dissolved form, with that percentage increasing as operating temperature decreases (Liu et al. 2014). When the effluent is discharged into waterways, dissolved methane is released into the atmosphere resulting a GWP approximately 25 times greater than carbon dioxide (Myhre et al. 2013).

The concentration of dissolved methane in anaerobic reactors varies depending on operating conditions, including SRT, temperature, and pressure in anaerobic digesters with floating caps that compress the headspace. Specifically, increasing SRT from 20 days to 40 days decreased the methane concentration of an anaerobic membrane bioreactor from 9.9 mg/L to 4.3 mg/L (Yeo and Lee, 2013). In addition, a pilot-scale anaerobic fluidized bed reactor (AFBR) produced effluent with a dissolved methane concentration of 76 mg/L at low temperatures (9-11°C), but the concentration decreased to 33 mg/L when temperature was increased to 15-20°C (Shin et al. 2014). Another study found that the percent of influent COD converted to dissolved methane increased from 31 percent to 52 percent when temperature was decreased from 15°C to 3°C (Smith et al. 2015). These results show the significant influence of SRT and seasonal temperature fluctuations on dissolved methane levels in anaerobic reactor effluent and highlight that anaerobic systems implemented in cool climates may incur larger impacts on climate change because of increased concentrations of dissolved methane in the effluent.

Dissolved methane can be removed from effluent via biological or physical mechanisms. Multiple methods may be used to extract dissolved methane from anaerobic effluent including

degassing membranes, biological oxidation, micro-aeration, and utilization of hollow-fiber membranes. A degassing membrane system was able to reduce the concentration of dissolved methane in UASB effluent from over 0.94 mM to 0.13 mM – nearly 86 percent (Luo et al. 2014). Biological oxidation involves the addition of an aerobic post-treatment step to anaerobic treatment to oxidize the dissolved methane, but the energy required to supply oxygen may make this technology inefficient (Cakir and Stenstrom, 2005). An encapsulated down-flow hanging sponge reactor, used as a post treatment to biologically oxidize dissolved methane, removed 95 percent of the methane in the influent (Hatamoto et al. 2010). An alternative method, micro-aeration, which removes dissolved methane from effluent via gas mixing of the reactor, decreased methane losses in anaerobic effluent to less than 11 percent of the total methane produced, such that more than 89 percent of produced methane was stripped into the gas phase (Hartley and Lant, 2006). An anaerobic reactor followed by a hollow-fiber degassing membrane as a degasification technology at 35°C increased total methane recovery from 89 to 97 percent (Bandara et al. 2011). All five methods resulted in a significant decrease in the methane lost from the systems via effluent, with the hollow-glass fiber degassing membrane reporting the most significant decrease in dissolved methane present in reactor effluent. However, it was determined that the removal of methane via degassing membrane is an energy intensive process that requires 300 times the energy equivalent of daily methane collected (Bandara et al. 2011).

2.3 Trickling Filter Models

Trickling filters, an aerobic fixed-film process, are a common low-energy alternative to conventional wastewater treatment (Tchobanoglous et al. 2003). Trickling filter treatment involves uniformly distributing wastewater over the top of a porous bed media to which

microorganisms have attached. Rock, vertical-flow plastic, and horizontal-flow plastic are commonly used packing media—although plastic media is generally considered preferable due to its high specific surface area, high void space, and low unit weight (Daigger and Boltz, 2011). The bed contains void space which allows for the vertical passage of air through the system by either natural draft or mechanical means, allowing the wastewater to be aerobically treated as it trickles through the bed (Grady et al. 2011). Usually, recirculation of the trickling filter effluent is employed to improve flushing of the filter medium and increase the removal rate (EPA, 2000). Recirculation is generally employed to achieve a 0.5 L/m^2-s hydraulic application rate which has been reported to achieve maximum efficiency (Tchobanoglous et al. 2003).

Much effort has gone into developing empirical formulas and deriving fundamental equations which describe trickling filter treatment based on organic loading, hydraulics, and mass-transfer (WEF, 2006). No equation is universally correct, so it is important to find the equation best-suited for a site-specific application. For instance, Schulze developed Equation 2.1 to describe trickling filter treatment for a non-specific media (WEF, 2006)

$$\frac{L_e}{L_o} = e^{-\frac{kD}{Q^n}} \tag{2.1}$$

where L_e is BOD of the settled effluent [mg/L], L_o is BOD of filter influent [mg/L], k is an experimentally determined rate constant [d^{-1}], D is filter depth [ft], Q is hydraulic loading rate [gpm/ft^2], and n is the packing-specific constant [unitless]. Germain later modified the Schulze equation to specifically describe treatment using trickling filters with plastic packing using Equation 2.2 (Tchobanoglous et al. 2003)

$$\frac{S_e}{S_o} = e^{-\frac{kD}{q^n}} \tag{2.2}$$

where S_e is BOD of the settled effluent [mg/L], S_o is BOD of filter influent [mg/L], k is the wastewater treatability and packing coefficient for n=0.5 [$(L/s)^{0.5}/m^2$], D is filter depth [ft], q is the hydraulic application rate of primary effluent (excluding recirculation) [L/m^2-s], and n is the packing constant for plastic packing- typically 0.50. The Germain equation is representative of treatment with plastic packing depths of 6.1-6.7 m and specific surface areas of approximately 90 m^2/m^3 (Tchobanoglous et al. 2003).

Eckenfelder developed an alternate equation, known as the modified Velz equation (Equation 2.3), to describe the treatment of a trickling filter which considers the surface area of the packing media and recirculation (WEF, 2006). The modified Velz equation is

$$S_e = \frac{S_o}{(R+1)\exp\left(\frac{k_{20}A_S D\theta^{T-20}}{[q(R+1)]^n}\right)-R} \quad (2.3)$$

where S_e is effluent BOD_5 [mg/L]. S_o is influent BOD_5 [mg/L], R is recycle ratio (R/Q), k_{20} is the rate constant at 20°C, A_s is specific surface area [m^2/m^3], D is media depth [m], θ is the temperature correction coefficient, T is temperature [°C], q is influent hydraulic rate [L/m^2-s], and n is hydraulic coefficient (WEF 2006). Equations 2.1-2.3 are just a few examples of the numerous approaches used for trickling filter design. This research utilized the Germain equation (Equation 2.2) for trickling filter design since the following typical values were assumed: plastic packing media with a 90 m^2/m^3 surface area and a tower height of 6.1 meters (Tchobanoglous et al. 2003).

An additional consideration in trickling filter design is whether the system will utilize natural draft or mechanical draft for aeration. Generally, low pressure fans are recommended to ensure adequate airflow, although natural draft may be sufficient for aeration if the pressure head due to temperature difference exceeds the pressure drop through the filter packing

(Tchobanoglous et al. 2003). The natural draft and pressure drop through packing of a trickling filter can be calculated using Equations 2.4-2.6 from (Tchobanoglous et al. 2003)

$$D_{air} = 353\left(\frac{1}{T_c} - \frac{1}{T_h}\right)Z \tag{2.4}$$

$$N_p = 10.33(D)e^{\left(1.36*10^{-5}\right)\left(\frac{L}{A}\right)} \tag{2.5}$$

$$\Delta P = N_P\left(\frac{v^2}{2g}\right) \tag{2.6}$$

where D_{air} is the natural draft of air [mm water], T_c is cold temperature [K], T_h is hot temperature [K], Z is filter height [m], N_p is packing head loss in terms of velocity heads, D is packing depth [m], L is liquid loading rate [kg/h], A is area of tower cross-section [m^2], ΔP is total head loss [kPa], g is 9.81 m/s^2, and v is superficial velocity [m/s].

Typical values and ranges for trickling filter treatment (BOD removal only) with plastic media can be found in Table 2.4. These values may be used as checks for calculated trickling filter design values.

Table 2.4. Typical and full-scale values for trickling filter treatment with plastic media

	Typical Value/Range	Source
Packing depth	5-7 m (16.4-23.0 ft)	(Grady et al. 2011)
	6 m	(Tchobanoglous et al. 2003)
	5-8 m	(Vesilind 2003)
	4.6 m (15 ft)[1]	(Jerry Smith, personal communication, 2014)
OLR (kg BOD/m^3-d)	0.3-1.0	(Tchobanoglous et al. 2003)
	0.4-0.8	(Henze 2008)
Recirculation ratio	0.5-4.0	(Vesilind 2003)
	1-2	(Tchobanoglous et al. 2003)
BOD removal (%)	60-90%	(Tchobanoglous et al. 2003)
Effluent BOD_5 (mg/L)	20-30	(Daigger and Boltz 2011)
	15-30	(Tchobanoglous et al. 2003)
Influent BOD_5 (mg/L)	200[1]	(Jerry Smith, personal comunication, 2014)
Number of towers (#)	2[1]	(Jerry Smith, personal communication, 2014)
Tower diameter (ft)	60[1]	(Jerry Smith, personal communication, 2014)

[1]represents a known design value from a full-scale trickling filter treatment plant

2.4 Constructed Wetlands Post-Treatment

Constructed wetlands are simple, low-energy systems which can be used as a post-treatment process to remove organics and nutrients from wastewater (Maheesan et al. 2011). There are two main types of constructed wetlands: vegetated submerged bed (VSB) and free water surface (FWS). Advantages of VSB wetlands are a greater tolerance to cold temperatures and reduced odors which make it ideal for implementation in cooler climates near residential communities (WEF, 1990). An advantage of FWS wetlands is a lower capital cost (WEF, 1990). Both VSB and FSW constructed wetlands have low operational and maintenance costs, requiring only occasional operator labor to clean inlet/outlet structures, maintain the berms, and monitor

performance (WEF, 1990) and provide the benefit of carbon sequestration via the planted vegetation totaling approximately 3.3 kg CO_2/m^2-year (Kalbar et al. 2013).

Constructed wetlands are a useful polishing step for effluent from secondary treatment that does not meet discharge limits. A study assessing ABR and constructed wetlands post-treatment of high-strength wastewater showed that an average of 90 percent BOD_5, 96 percent TSS, and 98 percent fecal coliforms could be removed by the system (Singh et al. 2009). A second study which used a laboratory-scale trickling filter and constructed wetlands post-treatment to treat domestic wastewater was able to achieve average concentrations of 22 mg/L of BOD and 28 mg/L of TSS in discharged effluent (Maheesan et al. 2011).

2.5 Life Cycle Assessment

LCA is a quantitative methodology used to assess the environmental impacts of products or processes. LCA originated in the early 1960s as a tool for industries to understand manufacturing systems, supply chains, and market behavior in order to make educated decisions when assessing competing designs, processes, and products (Theis and Tomkin, 2013). At that time, LCA focused primarily on the "life cycle analysis of cost" which involved developing a total cost of a product that included purchase, use, development, and end-of-life operations costs (Huppes and Curran, 2012).

The first documented attempt to shift LCA from a cost analysis tool to an environmental impact assessor occurred in 1969 when Coca Cola studied the resource use and waste management associated with their product (Huppes and Curran, 2012). The scope of the study was not broad; however, its exposure of the potential of LCA as an environmental tool combined

with the realization of the negative global impacts of industry, such as GHG emissions, instigated the development of environmental LCA over the next few decades.

In the 1990s, LCA served primarily as a tool for product development and comparison from an environmental management perspective, but was criticized for its lacked of a well-defined methodology (Gerber et al. 2011). That criticism led to the establishment of a series of standards and technical reports for LCA by the ISO commonly referred to as the ISO 14000 series for life cycle assessment (Heijungs and Guinee, 2012). Table 2.5 contains descriptions of the ISO 14000 series documents as of 2012.

Table 2.5. ISO documents on LCA (Heijungs and Guinee, 2012)

Number	Type	Title	Year
14040	International Standard	Principles and framework	1996, 2006
14041	International Standard	Goal and scope definition and inventory analysis	1998[a]
14042	International Standard	Life cycle impact assessment	2000[a]
14043	International Standard	Life cycle interpretations	2000[a]
14044	International Standard	Requirements and guidelines	2006[b]
14047	Technical Report	Examples of application of ISO 14042	2003
14048	Technical Report	Data documentation format	2001
14049	Technical Report	Examples of application of ISO 14041	2000

[a] Updated in 2006 and merged into 14044.
[b] Replaced 14041, 14042, and 14043

The ISO 1400 series ensures uniform conduction of LCA by providing a standard methodological framework. The framework decomposes LCA into four different stages: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation. LCA is an iterative, as opposed to linear, process. Figure 2.2 depicts the iterative nature of the ISO's suggested LCA framework by highlighting the two-way relationship between the various stages.

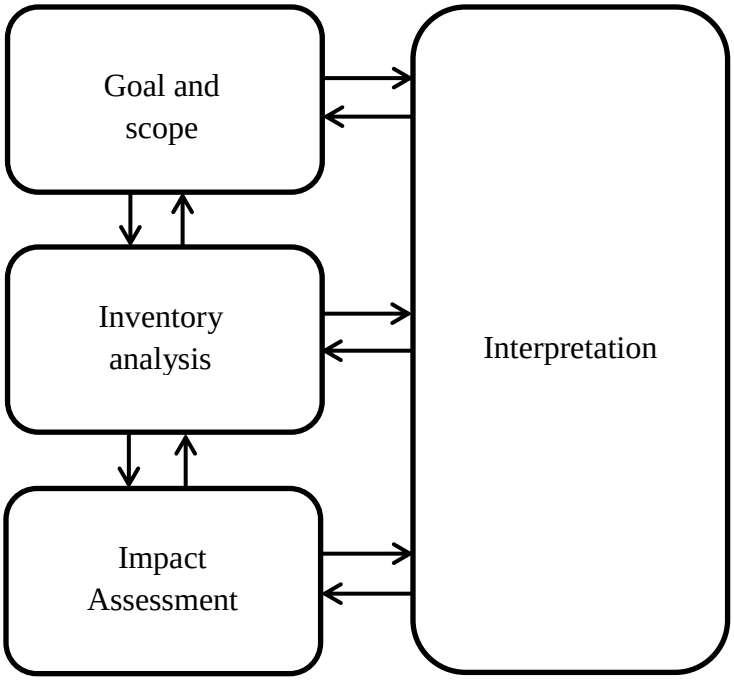


Figure 2.2. General methodological framework for LCA (adapted from "ISO 14040", 2006)

2.5.1 Goal and Scope Definition

Although LCA is not a linear process, the first step is often to clearly define the goal and scope of the LCA to be conducted. However, due to the iterative nature of LCA, it may be necessary to modify the scope as the study progresses and additional information regarding limitations and constraints is acquired.

The key components of goal and scope definition are to state the study objectives, define the functional unit, set the system boundaries, describe initial data quality, state assumptions, and explain the type of critical review to be performed. The standards for goal and scope definition are explicitly stated in ISO 14044.

2.5.1.1 Goal Definition

According to ISO, the goal definition must "unambiguously state the intended application, including the reasons for carrying out the study and the intended audience, i.e. to whom the results of the study are intended to be communicated" (Jensen et al. 1997). Furthermore, the goal definition must explicitly state whether or not the LCA results will be for private use or for comparative assertions that will be disclosed to the public ("CAN/CSA-ISO 14044", 2006).

2.5.1.2 Scope Definition

Scope definition involves the clear statement and description of the product system, performance characteristics of the system to be studied, functional unit, system boundaries, allocation procedures, types of impact and the methodology of impact assessment and subsequent interpretation to be used, data requirements, assumptions, limitations, initial data quality requirements, and type of critical review, if any (Jensen et al. 1997). A clear scope definition is vital to ensure proper comparability of LCA results.

Two systems must have the same function and functional unit to have comparable LCA results. A functional unit is a reference quantity to which the input and output data for the system are normalized (Jensen et al. 1997). The required quantity of process outputs for a given product system to fulfill the function expressed by the functional unit is called a reference flow ("CAN/CSA-ISO 14044", 2006). Two products with the same function can be compared by quantifying the same functional unit in the form of their reference flows ("CAN/CSA-ISO 14044", 2006).

2.5.2 Inventory Analysis

The primary task associated with LCI is the compilation and quantification of all relevant inputs and outputs of the system through data collection from literature and data calculation via system modeling (Heijungs and Guinee, 2012). In this stage, the various emissions and extractions associated with the individual inputs of the considered system are quantified and summed in a vector of substances, considering the different elements of the LCI (Gerber et al. 2011). Emissions are expressed in terms of the designated functional unit using Equation 2.7

$$Em_{j,i}^{FU} = \frac{Em_{j,i}}{FU_{tot}} \tag{2.7}$$

where $Em_{j,i}$ is the emission of the elementary flow *i* for the LCI element *j* and FU_{tot} is the total functional unit quantity (Gerber et al. 2011). Total emissions are quantified using Equation 2.8:

$$\forall i = 1 \dots n_i \colon Em_i^{FU} = \sum_{j=1}^{n_j} Em_{j,i}^{FU} \tag{2.8}$$

where n_i is the total number of elementary flows and n_j is the total number of LCI elements (Gerber et al. 2011).

Inventory analysis contributes uncertainty to the LCA as a result of the compounded effects of input uncertainties and data variability ("CAN/CSA-ISO 14044", 2006). Consequently, it is recommended by ISO to characterize the uncertainty introduced during LCI through uncertainty ranges and/or probability distributions.

2.5.3 Life Cycle Impact Assessment

The objective of life cycle impact assessment (LCIA) is to evaluate the environmental impacts of the material and energy flows determined through inventory analysis. ISO standard 14042 decomposes impact assessment into three processes: category definition and classification, characterization of inventory data, and optional normalization and weighting of all environmental impacts (Jensen et al. 1997).

Numerous environmental impact categories – or "midpoints" - are utilized in LCIA including acidification potential, eutrophication potential, GWP, and ozone depletion potential (Heijungs and Guinee, 2012). The input and output data collected during the LCI phase is categorized into relevant environmental impact categories. Then, the total impact for each category is calculated using a characterization model which expresses the LCI component impacts in terms of category indicators, such as CO_2 equivalents for GWP. This process is completed using a matrix, which contains the characterization factors for each of the elementary flows (emissions or depletions) considered in the LCI:

$$\begin{bmatrix} F_{1,1} & \cdots & F_{1,n_i} \\ \cdots & \cdots & \cdots \\ \cdots & F_{l,i} & \cdots \\ \cdots & \cdots & \cdots \\ F_{n_l,1} & \cdots & F_{n_l,n_i} \end{bmatrix} \cdot \begin{bmatrix} Em_1^{FU} \\ \cdots \\ Em_i^{FU} \\ \cdots \\ Em_{n_i}^{FU} \end{bmatrix} = \begin{bmatrix} I_1^{FU} \\ \cdots \\ I_l^{FU} \\ \cdots \\ I_{n_l}^{FU} \end{bmatrix} \tag{2.9}$$

where $F_{l,i}$ is the characterization factor to convert the emission or depletion for LCI elementary flow *i* into the midpoint impact category *l*, Em_i^{FU} is the emission or depletion for elementary flow *i* calculated using Equation 2.2, and I_l^{FU} is the impact category l of the impact assessment method (Gerber et al. 2011).

Optional components of LCIA are normalization and weighting of results. Normalization allows "midpoints" to be grouped into "endpoints" –or damage categories- which include climate change, ecosystem quality, human health, and nonrenewable resource consumption. Endpoint category impacts are determined using a variation of Equation 2.9, in which the characterization matrix is replaced with a similar normalization matrix. Normalized endpoint impacts are reported in units of "points" which correspond to the total impacts for a given area on a per capita basis, which allows for comparison of different impacts ("CAN/CSA-ISO 14044", 2006).

The second optional LCIA technique - weighting of results - involves modifying the indicator results of the environmental impact categories using numerical factors based on value-choices ("CAN/CSA-ISO 14044", 2006). Weighting is a tool used to synthesize impact scores by considering the relative importance of each impact category to the overall environmental performance of the product or process being considered (Lippiatt, 2007). A single impact score can be determined using Equation 2.10:

$$I_{tot}^{FU} = \sum_{l=i}^{n_l} I_l^{FU} \cdot w_1 \tag{2.10}$$

where w_1 is the weighting factor.

Some of the most common methods used to assess environmental impacts include CML 2001, Eco-indicator 99, EPS 2000, IMPACT 2002+, IPCC, and TRACI (Rosen et al. 2012; Margni and Curran, 2012). The choice of LCIA methodologies is largely based on the geographic location of the study. EPS 2000, CML 2001, and IMPACT 2002+ are the most commonly used methods for Swedish, Spanish, and European studies respectively (Yoshida et al. 2013). It is important to use more than one impact assessment method to check consistency and confirm legitimacy of LCIA results (Gerber et al. 2011).

2.5.3.1 IPCC 2013

The IPCC developed their most current impact assessment method – IPCC 2013 – to assess the impacts of air emissions on GWP. Three versions of the IPCC method exist to assess climate change factors over 20, 100, and 500 year timeframes (Goedkoop et al. 2008). GWP is the only impact category assessed via IPCC 2013. Consequently, weighting and normalization are not applicable to results.

Several considerations underlie the characterization factors used by IPCC 2013. It should be noted that biogenic carbon dioxide is considered an impact credit that reduces the calculated net impact for the product or process (Goedkoop et al. 2008). However, biogenic methane is not considered a credit, and, therefore, produces an impact on climate change.

2.5.3.2 TRACI 2.1

TRACI was developed by the U.S. EPA to assess characterization factors that correspond to various U.S. locations. TRACI characterizes non-weighted impact categories at the midpoint level which eliminates the uncertainty that arises when defining the relationships between midpoint and damage categories (Bare et al. 2003). The most recently updated version of this method is TRACI 2.1.

2.5.3.3 IMPACT 2002+

The IMPACT 2002+ life cycle impact assessment methodology was created at the Swiss Federal Institute of Technology. IMPACT 2002+ utilizes a combined midpoint/damage approach that links all flows defined through LCI via 14 midpoint categories to the following four damage categories: human health, ecosystem quality, climate change, and resources (Humbert et al. 2005).

Figure 2.3 shows the distribution of midpoint categories between damage categories for the IMPACT 2002+ method. Note that the solid arrows represent pathways that are known and quantitatively modeled, whereas the dashed arrows represent assumed correlations between midpoint and damage categories that are not included in the quantitative model (Humbert et al. 2005).

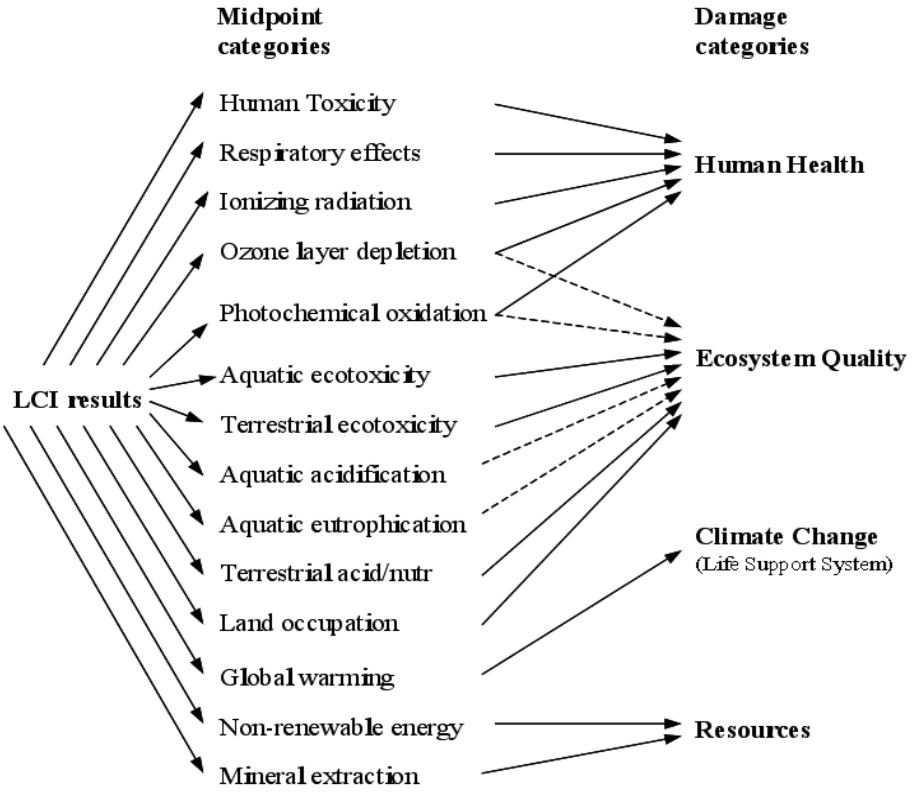


Figure 2.3 IMPACT 2002+ framework (Humbert et al. 2005)

Damage (or endpoint) categories may be normalized by dividing the impact per unit of emission by the total impact of a specific category, per person per year (Goedkoop et al. 2008). Data used for normalization reflect the impacts associated with an average European lifestyle.

However, research has been conducted to create normalization factors associated for the U.S. and Canada (Lautier et al. 2010). Weighting is optional, but not recommended, for IMPACT 2002+.

2.5.4 Interpretation

According to ISO 14044, life cycle interpretation is the phase of LCA in which the results of impact assessment and inventory analysis are evaluated in relation to the defined goal and scope to draw conclusions and make recommendations. In addition, interpretation involves the evaluation of the completeness, sensitivity, consistency, and overall quality of the LCA results.

2.5.4.1 Uncertainty and Sensitivity Analysis

Uncertainty analysis quantifies uncertainties and considers their effect on decision-making (Smith, 2002). A useful technique to incorporate in environmental assessments is Monte Carlo analysis – which is a computer-based method of analysis that utilizes statistical sampling techniques to obtain a probabilistic approximation to the solution of a mathematical model (Huijbregts et al. 2003). Application of Monte Carlo technique to LCA involves the conversion of the deterministic LCA model to a probabilistic model that is used to forecast the entire range of likely observations in a given situation (Shih-Chi et al. 2005). Often the minimum, maximum, and mode of parameter values are the basis for the triangular probabilistic distribution, used for input parameters with scarce data (Smith, 2002). It is important to consider what parameters should be modeled, since a large number of parameters requires a large number of Monte Carlo simulations to produce defensible results (Smith, 2002). One way to choose parameters to include in uncertainty analysis is to conduct sensitivity analysis. A sensitivity analysis is conducted by

changing the values of single parameters and determining the impact of these single changes on model results.

2.5.5 LCA of Wastewater Treatment

LCA was first applied to wastewater treatment in the 1990s to evaluate the CO_2 emissions of various technologies (Corominas et al. 2013). Since then, the usage of LCA in wastewater treatment has broadened to include numerous environmental impacts (e.g. eutrophication, ozone depletion, resource depletion) and has been used to identify and explore problem areas (in terms of environmental impacts) for several wastewater treatment processes. For instance, a number of studies have focused on environmental impacts associated with treatment and disposal of sludge (Hospido et al. 2005; Hong et al. 2009; Houillon and Jolliet, 2005; Suh and Rousseaux, 2002; Yoshida et al. 2013). Results showed that overall environmental impacts were lower for sludge treatment that included anaerobic digestion to reduce the volume of sludge for disposal (Hong et al. 2009; Suh and Rousseaux, 2002). Additional researchers concluded that, in terms of global warming potential, incineration of sludge in cement kilns is a more ideal end-of-life use for sludge than agricultural spreading, fluidized bed incineration, wet oxidation, pyrolysis, and landfilling (Houillon and Jolliet, 2005). However, the results of LCA of sludge disposal vary significantly between studies due to differences in defined goal, scope, and local conditions (Yoshida et al. 2013).

LCA is also used to compare conventional activated sludge with less frequently implemented, yet promising, alternatives for secondary treatment. For instance, LCA was used to compare environmental impacts associated with activated sludge to those of a sequencing batch reactor, up-flow anaerobic sludge blanket reactor followed by facultative aerobic lagoon, and

constructed wetland treatment of wastewater (Kalbar et al. 2013). This study concluded that constructed wetlands produced the lowest overall environmental impact and a negative impact on GWP due to carbon sequestration; however, sequencing batch reactors resulted in the lowest eutrophication potential impact due to their ability to produce high quality effluent in terms of organic and nutrient removal (Kalbar et al. 2013). A study that used LCA to compare activated sludge (conventional and high-rate) with anaerobic membrane and aerobic membrane treatment concluded that high-rate activated sludge combined with anaerobic digestion of solids was the most advantageous technology in terms of global warming impact and net energy, but that future advancements in anaerobic membrane treatment may make the technology competitive, in terms of environmental performance, with high-rate activated sludge (Smith et al. 2014). LCA has also been used to determine that low-energy alternatives to activated sludge, such as constructed wetlands and slow-rate infiltration, would reduce impacts on abiotic (non-living) resource depletion by approximately 85 percent, acidification by nearly 94 percent, and could result in a net beneficial impact on global warming (Nogueira et al. 2009).

Although LCA of wastewater treatment is common, a review of 45 LCA studies on wastewater treatment reported that 62 percent did not correctly follow ISO standards for LCI and 67 percent did not include a thorough interpretation of results that included a sensitivity analysis and/or limitations of the methodology (Corominas et al. 2013). Furthermore, uncertainty is inherent in LCA studies of wastewater treatment due to the variability in wastewater quality, operating temperature, operating methods, and treatment configurations yet only one study – out of 35 published studies on LCA of sewage sludge – incorporated a thorough uncertainty analysis (Yoshida et al. 2013). In addition, only three of 45 studies on wastewater treatment considered the effects of the chosen impact assessment method on LCA results (Corominas et al. 2013). The lack of consistency in methods makes it difficult to compare results across studies and suggests that

there is a need to develop a standardized framework for conducting LCA for wastewater treatment.

In addition to inconsistent methodology, the presence of dissolved methane in anaerobic effluent was not considered in numerous existing LCA studies of anaerobic wastewater treatment including Hospido et al., Foley et al. and Kalbar et al. (Hospido et al. 2008; Foley et al. 2010; Kalbar et al. 2013), yet has been shown to have a significant environmental impact (Smith et al. 2014). An LCA study which did not consider dissolved methane found that an upflow anaerobic sludge blanket with a facultative lagoon for post-treatment produced a global warming impact of 7.67 kg CO_2-eq (for 1 p.e./year) which was 58 percent less than the global warming impacts produced from aerobic activated sludge treatment (Kalbar et al. 2013). Similarly, anaerobic treatment resulted in approximately 58 percent less GHG emissions compared to aerobic treatment when dissolved methane was not included in the analysis (Keller and Hartley, 2003). However, research that considered the influence of dissolved methane reported that the GWP for anaerobic treatment is 36 percent higher than for aerobic treatment (Liu et al. 2014). These results indicate that dissolved methane can have a significant influence on the overall environmental impact of anaerobic treatment and should be considered in LCA studies.

2.6 Techno-economic Analysis

TEA is method used to determine the economic feasibility of a product or process by coupling its technical and economic aspects. The first step of TEA is to clearly define the goal and scope of the analysis and define the product(s) and corresponding function(s). The next stage is to create a block flow diagram of the process which consists of blocks representing unit operations or equipment linked by input and output flows (Turton et al. 2012). The block flow

diagram used for TEA may be the same model used to develop LCIs, although it is important to note that not all inventory flows will contribute to the economic analysis – e.g., CO_2 emissions currently incur no cost. Once the goal, scope, and process have been defined, capital costs, operating costs, and profitability of the process can be analyzed.

2.6.1 MACRS Depreciation of Capital Costs

Total capital cost estimation for equipment and raw materials requires consideration of all costs associated with purchase and installation. Capital depreciation is the difference between the costs associated with the purchase and installation of equipment (negative cash flow) and the salvage cost (positive cash flow) of the equipment after the plant is closed (Turton et al. 2012). Over time, the value or worth of equipment and buildings decreases; however, the capital cost associated with land and working capital does not depreciate (Turton et al. 2012). The MACRS may be used for depreciation costing. MACRS uses the time period over which capital cost is depreciated, or recovery period, and the class life to determine the allowable cost-recovery period for a property (Collier and Glagola, 1998). MACRS dictates the maximum depreciation allowance as a percentage of capital investment for a given year which results in the quickest depreciation of an investment – the ideal approach from an economic perspective (Turton et al. 2012).

2.6.2 Economics of Wastewater Treatment

In the United States, it is estimated that an average of 35 percent of a municipality's total budget is spent on wastewater treatment (US EPA, 2013). One of the most costly operations at a conventional aerobic treatment plant is electricity for aeration which may contribute 45 to 75

percent of total energy costs (Rosso et al. 2008). In Pennsylvania, annual electricity requirements for an average aerobic municipal wastewater treatment plant may cost approximately $140,000 (PADEP, 2011). Consequently, exploration and implementation of various low-energy aerobic and anaerobic technologies is of interest.

Disposal of sludge is an additional costly operation and, in Pennsylvania alone, nearly 300,000 tons of dry solids are produced from treatment of wastewater each year (Elliot et al. 2007). These costs can be substantial since disposal of sludge via incineration can cost $87[a]-143[a] per wet ton on average ($349 to $572 per dry ton if 25 percent solids content is assumed) (NEBRA, 2001). One method to reduce the cost of sludge disposal is to treat wastewater anaerobically since it generates approximately 12 percent of the solids produced via aerobic treatment (Ashrafi et al. 2014). An additional option is to reduce solids via anaerobic digestion prior to disposal. Alternatively, careful choice among sludge disposal options can reduce costs. For instance, when transportation and processing is considered, landfilling of solids in Virginia costs an average of $431[a] per dry ton but land application only costs approximately $238[a] (NBP, 2005). Land application of solids may be less expensive than landfilling or incineration since it uses the nutrient-rich solids in lieu of fertilizer which can potentially generate a revenue of $28[a] to $187[a] per dry ton (processing and transportation not considered) for high quality biosolids pellets (Girovich, 1996). However, the cost of processing Class B biosolids to meet Class A standards is approximately $193[a] per dry ton (McMillon et al. 2000), which offsets the revenue from the sale of solids.

Although there are high costs associated with wastewater treatment, there is potential for revenue if combined heat and power (CHP) systems are implemented to convert the methane-rich

[a] Converted to 2015 dollars using ENR cost indices

biogas produced via anaerobic digestion into electricity. CHP technology could potentially generate 26 kW of electricity from each MGD of wastewater generated in the United States (US EPA, 2013). It was previously thought that cogeneration was not economically viable for treatment less than 5 MGD, however microturbines have been successfully implemented at a 2 MGD facility which generates nearly $44,000 worth of electricity each year (Eaton and Jutras, 2005). Microturbines are generally considered the most preferential CHP technology for influent flows less than 10 MGD, and have been successfully implemented at facilities with flows less than 5 MGD (ERG and RDC, 2011).

The economics of wastewater treatment are often evaluated in terms of capital cost, operating cost, and present worth. Common methods for deriving capital and operating costs for planning cost estimates include cost curves generated using data from existing facilities, unit costs provided by industry or the RS Means construction cost data (Waier and Charest, 2012), or economic modeling software packages such as CapdetWorks. Regardless of the method of estimation used, it is important to adjust costs for time using the Engineering News Record cost index, and location using local cost estimates or a location factor obtained from RS Means.

Cost curves are developed by fitting equations to known cost data from existing facilities and are advantageous when developing preliminary cost estimates since they require minimal information (e.g. influent flow, solids production) and produce estimates for entire treatment plants or treatment plant processes (e.g. anaerobic digestion, primary clarification) quickly. Cost curves have been used to analyze and compare reported capital and operating costs for anaerobic digestion of a wide range of solids and to determine that an economy of scale applies to anaerobic digesters (CAEPA, 2008).

If the specifics for treatment (e.g. location, site layout drawings, major required equipment) are known, unit costing can be used to provide more detailed cost estimates than cost curves. For instance, unit costing was used to determine budgeting level cost estimates with a -20% to +30% accuracy for algae ponds implemented for wastewater treatment and biofuel production in the Imperial Valley in California (Lundquist et al. 2010). The study utilized industry data specific to the geographic location and RS Means construction cost data (Waier and Charest, 2012) to generate the cost estimates (Lundquist et al. 2010).

Economic modeling of wastewater treatment using computer modeling programs, such as CapdetWorks (Hydromantis ESS, Inc. 2014), has become increasingly popular for providing preliminary cost estimates. CapdetWorks uses a combination of parametric cost curves generated by the Army Corp. of Engineers (US EPA, 1982) and unit costing to determine the capital costs, operating costs, and present worth of treatment plants. CapdetWorks has been used to determine preliminary estimates of the capital cost and annual maintenance, electrical, and chemical costs for new plants and plant expansions in Detroit, MI (Tetra Tech MPS, 2003) and has been used in research to evaluate the present worth of anaerobic membrane treatment, aerobic membrane treatment, and activated sludge (conventional and high-rate) for treatment of wastewater and to evaluate the economics of incineration, land application, and landfilling of solids (Smith et al. 2014).

2.7 Combined LCA and TEA of Wastewater Treatment

TEA may be combined with LCA to avoid the recommendation of a product that is not economically viable. However, many LCA studies on wastewater treatment have not been coupled with economic evaluation which raises doubt as to whether the technologies assessed in

those studies are economically viable treatment methods. A study which compared microbial fuel cell and microbial electrolysis cell systems showed that microbial electrolysis cell systems provide a significant environmental benefit relative to conventional anaerobic digestion, but that microbial fuel cells do not (Pant et al. 2011). However, the economic analysis determined that bioelectrical systems are not an economically viable alternative to anaerobic digestion due to the high cost of electrodes and membranes (Pant et al. 2011). The study highlights the importance of considering economic impacts when recommending a wastewater treatment technology.

A combined economic and environmental approach was used in a study that aimed to evaluate the sustainability of 24 Spanish wastewater treatment plants using two different functional units: volume of water treated and eutrophication reduction (Rodriguez-Garcia et al. 2011). The plants were classified into six "typologies" according to the quality requirements that their effluents have to meet such as removal of organic matter for discharge to non-sensitive areas (Type 1), removal of organic matter and nutrients to non-sensitive areas (Type 2), and plants that reuse the treated wastewater for aquifer recharge (Type 6). Eutrophication potential, GWP, and operational costs were considered, and it was concluded that a trade-off exits between environmental and economic impacts when volume is used as the functional unit (Rodriguez-Garcia et al. 2011). The two plant typologies (Type 1 and Type 2) that produced the most significant eutrophication impacts had costs that were approximately 40 percent and 64 percent less than the plant typology that produced the lowest eutrophication impact (Type 6) - approximately 45 percent and 40 percent that of the eutrophication impacts of Type 1 and Type 2, respectively (Rodriguez-Garcia et al. 2011). As a final conclusion, the study recommended the incorporation of socio-cultural impacts in addition to environmental and economic impacts to obtain a more complete set of sustainability indicators for the wastewater treatment plants (Rodriguez-Garcia et al. 2011).

One study that coupled LCA, life cycle costing (LCC), and a comprehensive analysis of uncertainty, compared aerobic and anaerobic membrane bioreactors (AeMBR and AnMBR) to activated sludge at 15°C and accounted for dissolved methane in anaerobic effluents (Smith et al. 2014). AnMBR capital costs were higher than HRAS+AD, but the cost of disposal of sludge for HRAS+AD could offset those capital costs depending on whether sludge was landfilled, incinerated, or land applied (Smith et al. 2014). Regardless of the practice used for disposal of sludge, AeMBR+AD had the highest life cycle costs of the technologies assessed in the study due to the electricity costs for aeration and high capital costs associated with the membrane system (Smith et al. 2014). In addition, the AnMBR generated the highest global warming impact. Seventy-five percent of the global warming impact resulted from dissolved methane in the effluent (Smith et al. 2014). The study concluded that with current technology, AnMBR treatment is neither economically viable nor environmentally favorable due to the high cost of membranes, energy required for membrane sparging, and impact of dissolved methane in anaerobic effluent on global warming.

These examples demonstrate that the most advantageous technology from an environmental perspective is often not the most economically favorable option. Therefore, an economic analysis should be included in LCA studies for a more comprehensive analysis and comparison of technologies.

CHAPTER 3

PERFORMANCE OF A LABORATORY-SCALE ANAEROBIC BAFFLED REACTOR FOR TREATMENT OF MUNICIPAL WASTEWATER

3.1 Introduction

Anaerobic treatment of domestic wastewater is an attractive alternative to conventional aerobic treatment (e.g., activated sludge) due to its production of methane-rich biogas and elimination of electricity requirements for aeration. Furthermore, the ABR is amongst the most desirable anaerobic reactors for treatment of domestic wastewater due to its high energy recovery and compartmentalized design which allows for high biomass retention, separation of acidogenesis and methanogenesis, and reduced expansion of the sludge (Shoener et al. 2014; Barber and Stuckey, 1999; Foxon et al. 2004).

Despite the advantages of the ABR, two main drawbacks may limit full-scale implementation: (1) decreased performance at low temperatures, and (2) the presence of dissolved methane in the effluent. Studies that examined ABR performance at psychrophilic temperatures (12-20°C) report a wide range of COD removals, spanning from 60 to 91 percent, depending on reactor operating conditions. In addition, many researchers only assessed low-temperature treatment of synthetic wastewater (Langenhoff and Stuckey, 2000; Motteran et al. 2013; Manariotis and Grigoropoulos, 2002; She et al. 2013; Nachaiyasit and Stuckey, 1997). One study, however, concluded that an ABR could attain nearly 86 percent COD removal when treating municipal wastewater at 21.1°C with a 24 hour HRT (DiStefano, 2010), which indicates that high removals are possible at a low operating temperature.

Although previous studies showed that the ABR may potentially achieve moderate to high organic removals at low temperatures (DiStefano, 2010; Manariotis and Grigoropoulos, 2002), none of those studies considered the presence of dissolved methane in the effluent. Dissolved methane represents a lack of energy efficiency, is a potent GHG, and its release in effluent may contribute significantly to climate change. Therefore, dissolved methane may negate ABR benefits such as electricity production from biogas. Although there are no reports in the literature on the presence of dissolved methane in effluent from an ABR, Liu et al. (2014) estimate that at 30°C nearly 45 percent of total methane produced during anaerobic treatment of domestic wastewater may remain in the dissolved phase (Liu et al. 2014). Furthermore, Shin and colleagues reported that a pilot-scale anaerobic fluidized bed reactor produced effluent with an average dissolved methane concentration of 76 mg/L COD (19 mg CH_4/L) at 9-11 °C and 33 mg/L COD (8.3 mg CH_4/L) at 15-20 °C (Shin et al. 2014). Therefore, the presence and impacts of dissolved methane should be accounted for to obtain a holistic understanding of anaerobic wastewater treatment impacts.

In this study, we quantified the performance of an ABR treating domestic wastewater at 15°C and 20°C. Removal of organics and solids, production of biogas, and concentrations of dissolved methane were used to model environmental and economic performance for a full-scale ABR in subsequent research, presented in Chapters 4 and 5.

3.2 Methods

Sections 3.2.1, 3.2.2, 3.2.3, 3.2.5, and 3.2.7 are modified versions of materials and methods described by Wacker in his Master's Thesis (Wacker, 2014).

3.2.1 Anaerobic baffled reactor description

A laboratory-scale ABR consisting of three chambers with a total empty bed volume of 17.8 L was used for data collection (Figure 3.1).

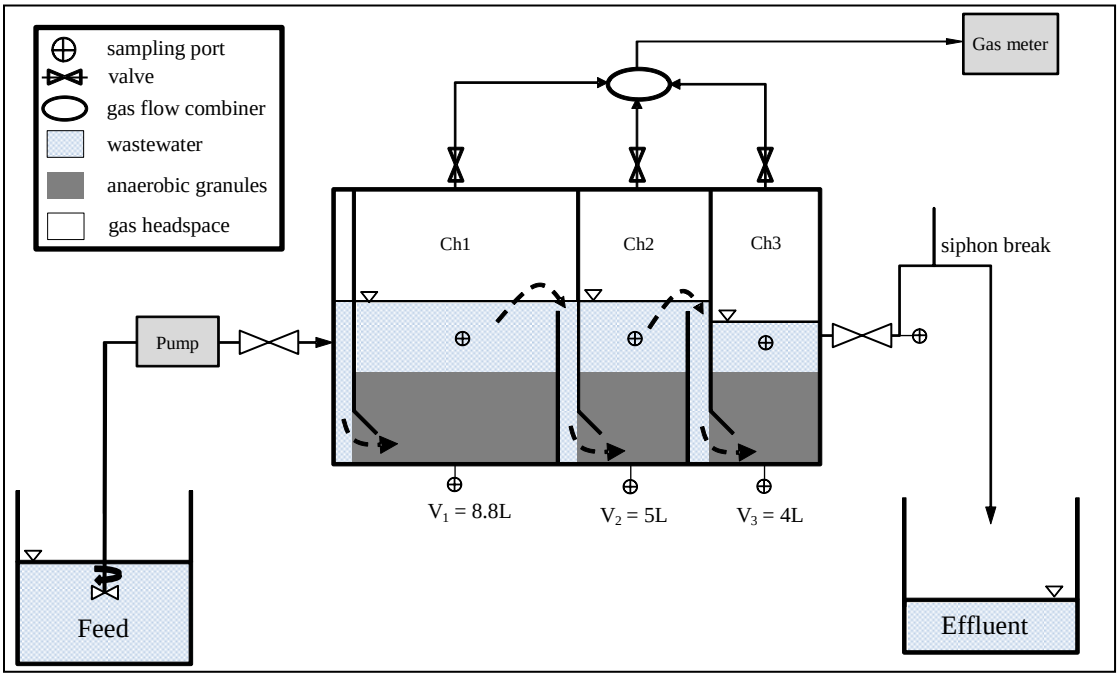


Figure 3.1. Schematic of ABR used for data collection (adapted from Wacker, 2014).

The empty bed volumes for Chambers 1, 2, and 3 were 8.8 L, 5.0 L, and 4.0 L, respectively. Liquid flowed into each chamber via a vertical down-flow baffle with a 45° bend along the bottom edge and trickled out of the first and second chambers over a vertically oriented up-flow baffle. Effluent exited the reactor through PVC piping attached to an opening on the side of the third chamber.

The ABR was equipped with biosolids sampling ports on the bottom panel of each chamber and liquid sampling ports on the front and back panels of each chamber. A sampling port was also attached to the effluent tubing between the reactor and a siphon break to obtain

liquid samples that were representative of ABR effluent prior to atmospheric exposure. Tubing with gas sampling ports, equipped with natural rubber septa (Sigma-Aldrich, Bellefonte, PA), were attached to the center of the top panel of each chamber. The three tubes connected at a joint which directed the combined gas flow to a tipping meter (Rebel Point Wet Tip Gas Meter Company, Nashville, TN) to measure gas production.

ABR temperature was controlled by a MM7 water chiller (VWR, Radnor, PA) which directed a flow of cooled water into six tubes oriented horizontally from entrance to exit across the ABR. A seven day test was conducted using LabQuest software (Venier, Portland, Oregon) in which a thermocouple continuously collected temperature data from Chamber 2 of the ABR to determine the accuracy of the temperature settings on the MM7 water chiller.

3.2.2 Reactor operation

Steady-state reactor performance was assessed at 20°C for 43 days and at 15°C for 56 days. The performance and maintenance testing schedule can be found in Table B.1 of the appendix.

Raw wastewater collected daily from Milton Regional Sewer Authority was used as feed. Sodium bicarbonate (EMD Chemicals Inc., Darmstadt, Germany) was added to the feed every other day to achieve alkalinity of 1100 mg/L as $CaCO_3$ at 20°C and 800 mg/L as $CaCO_3$ at 15°C to ensure that the pH within the reactor remained in the range of 6.5-7.5.The feed was continuously mixed with a paddle mixer (REX Engineering Co., Titusville, FL) and fed into a side port of the ABR using a peristaltic Masterflex® L/S® pump (Model #7518-00, Cole-Parmer Instrument Company, Vernon Hills, IL). A unit (XT Timer ChronTrol Corporation, San Diego, CA) automatically controlled operating cycles for the pump and achieve a 0.55 day HRT. The

OLR varied as described in Table 3.1 during 20°C and 15°C operation due to daily differences in feed concentration.

Table 3.1. Values of OLR and HRT at 15°C and 20°C (average [number of samples] (95% CIs)).

Parameter	**Unit**	**15°C**		**20°C**	
		Average [n]	**95% CI**	**Average [n]**	**95% CI**
OLR	g BOD_5 fed/L-reactor-d	0.48 [51]	(0.43, 0.53)	0.48 [38]	(0.42, 0.54)
	g COD fed/L-reactor-d	1.06 [51]	(0.95, 1.16)	1.06 [38]	(0.92, 1.20)
HRT	d	0.57 [54]	(0.56, 0.58)	0.53 [39]	(0.52, 0.55)

Solids were wasted from Chamber 1 two times per week to reduce the build-up of non-biodegradable solids in Chamber 1, which obstructed the liquid sampling port. However, solids still accumulated within the reactor at a rate of approximately 3.5 g per day. Wasting of solids from Chambers 2 and 3 was not required. Butyl rubber septum on sampling ports was replaced twice per week and influent tubing was replaced on an as needed basis due to varying rates of solids buildup within the tubing.

3.2.3 Characterization of organics and solids

TCOD and SCOD of the influent and effluent were measured according to the standard closed reflux calorimetric method (APHA, AWWA, and WEF 2012). COD samples were collected after feed preparation each day and were added to pre-made COD vials (VWR, Radnor, PA, cat. #: 80094-566; 80094-558). TCOD samples were transferred directly into COD vials. SCOD samples were filtered through a 1.2 µm glass-fiber filter (EMD Millipore, Billerica, MA) using a vacuum filter, and the filtrate was transferred directly into COD vials.

Influent COD was measured using medium range (0-1500 mg/L as COD) vials. During 20°C operation, effluent was measured using low range (0-150 mg/L as COD) vials. During 15°C operation, effluent concentrations increased as a result of the temperature change and, therefore, required preparation of both a medium and low range sample to ensure that the COD reading fell within the appropriate concentration range. The measurement that fell within the lowest acceptable measurement range was used for data analysis.

Testing was conducted to determine a BOD_5/TCOD ratio for ABR influent and effluent to convert measured TCOD concentrations to BOD_5 concentrations. The 5-Day BOD test was conducted on influent and effluent samples according to standard procedure 5210 (APHA, AWWA, and WEF 2012). A Hach HQ40d dual input portable multi-parameter meter (Hach Company, Loveland, CO) was used to measure dissolved oxygen concentrations. Feed samples from three separate days were run in triplicate to account for variability in the influent.

TSS and VSS of the influent and effluent were measured three times per week using standard methods (APHA et al. 2012). TS/VS testing was conducted on solids wasted from Chamber 1 to determine the VS content of the wasted sludge. In addition, an aliquot of the wasted sludge was diluted and blended (with a kitchen blender) for COD analysis. COD content of wasted solids was measured on three separate days, and was used to convert wasted volatile solids into an equivalent mass of COD.

3.2.4 Feed characteristics

The strength of the feed varied, having an average ± standard deviation COD concentrations of 531 ± 217 mg/L and 576 ± 190 mg/L during the 20°C (Sept.-Oct. 2014) and 15°C (Nov. 2014-Jan. 2015) operating periods, respectively (Table B.3, Figure 3.2). But average

values shown in Table B.3 indicate that the feed can be characterized as a low strength municipal wastewater (Henze 2008).

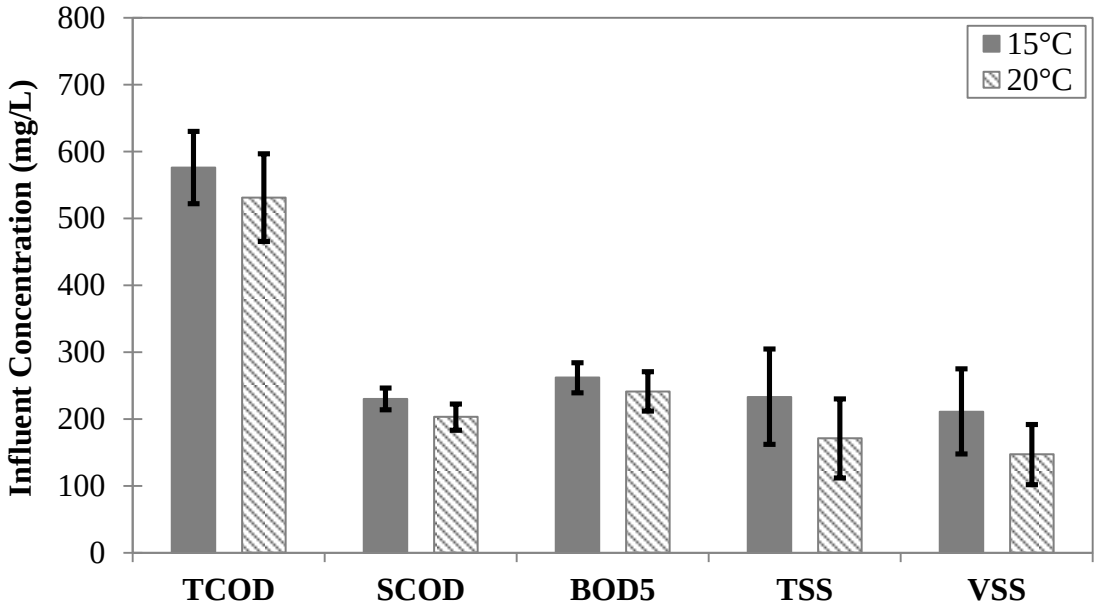


Figure 3.2. Feed characteristics for 15°C and 20°C. Error bars represent 95% CIs.

Figure 3.2 shows that the average values for influent TCOD, SCOD, TSS, VSS, and BOD_5 concentrations were higher at 15°C than 20°C and that feed characteristics varied. Results from a statistical analysis that compares feed characteristics are described in section 3.3.1.

3.2.5 Methane production

A calibrated precision wet tip gas meter was used to quantify gas production by the ABR. Biogas composition was analyzed using a HP6890 GC coupled with a 15 ft x 1/8 in x 2.1 mm 60/80 Carboxen 1000 packed column (Supelco, Bellefonte, PA) and a TCD. Nitrogen gas flowing

at a rate of 27.1 mL/min through the packed column served as the carrier gas. The oven temperature was maintained at 150°C throughout the seven minute run.

A 1 mL Hamilton® SampleLock syringe (Hamilton Company USA, Reno, NV) was used to take a 0.3 mL headspace sample from each chamber of the ABR. A standard curve generated using 99.99 percent methane gas (Associated Gas Products, Everett, MA) and a 60 percent methane and 40 percent carbon dioxide gas mixture (Praxair, Danbury, CT) was used to convert peak area to percent methane. The standard curve is presented as Figure B.1 in the appendix. Daily methane production was quantified by multiplying the total biogas production by the average percent methane in the headspace of the three ABR chambers.

3.2.6 Dissolved methane

Dissolved methane in ABR effluent was measured daily. The dissolved methane test was conducted by injecting 3 mL of ABR effluent into an 11.5 mL serum bottle sealed with a butyl rubber stopper and crimped aluminum cap (Armenio, 2013). Samples were drawn directly from a sampling port attached to an extension on the third chamber which minimized loss of methane to the atmosphere. The serum bottle was shaken for 90 seconds to allow the system to equilibrate. Then, a 100 µL headspace sample was taken with a 100 µL Hamilton® SampleLock syringe (Hamilton Company USA, Reno, NV) and injected into a Hewlett Packard (HP) 5890A GC System coupled with a 2 m x 2 mm Rt-sulfur micropacked Silcosteel® column (Restek Corporation, Bellefonte, PA) and FID. Nitrogen carrier gas was used and the oven temperature for a given sample increased from 140°C to 230°C at a rate of 15°C per minute, and then remained at 230°C for 2.33 minutes.

A standard curve (shown in Figure B.2 of the appendix) created using 99.99 percent methane gas (Associated Gas Products, Everett, MA) was used to convert peak area to moles of methane in the injected headspace sample. The partial pressure of methane in the headspace was used in conjunction with the ideal gas law, Henry's law, and a mass balance on the contents of the serum bottle to determine the total moles of methane in the serum bottle. Dimensionless Henry's constants (C_{aq}/C_g, where C is in unit of mol/L) of 0.0356 and 0.0384 were calculated for 20°C and 15°C, respectively, using Equation 3.1 (Tchobanoglous et al. 2003)

$$\log H_T = -\frac{675.74}{T} + 6.88 \tag{3.1}$$

where T is temperature (K) and H_T is Henry's constant at temperature T [atm]. Henry's constants were converted from atm units to a dimensionless value using Equation 3.2

$$K_{H,T} = \frac{55.6\frac{mole\ H_2O}{L\ H_2O} * R * T}{H_T} \tag{3.2}$$

where $K_{H,T}$ is the dimensionless Henry's constant for temperature T [C_{aq}/C_g], R is the gas constant [L-atm/mole-K], and T is temperature [K].

The concentration of dissolved methane in the effluent was then calculated by assuming that the sum of methane present in the dissolved and gaseous phases in the serum bottle came from the 3 mL of ABR effluent – as demonstrated in Equation 3.3

$$M_{CH4,sb} = C_{g,s} * V_g + C_{aq,s} * V_l \tag{3.3}$$

where $M_{CH4,sb}$ is total moles of methane in serum bottle, $C_{g,s}$ is molar concentration of methane in the gaseous portion of the serum bottle [mole/L], V_g is volume of the gaseous portion of serum bottle [L], $C_{aq,s}$ is molar concentration of methane in the aqueous portion of the serum bottle [mole/L], and V_l is volume of the liquid portion of serum bottle [L]. A blank was run to confirm that the methane concentration of air was zero. Sample calculations used to determine dissolved methane concentration can be found in Appendix B.

3.2.7 Volatile Fatty Acids

VFAs in Chambers 1, 2, and 3 were analyzed three times per week to serve as an early indicator of reactor malfunction. Liquid samples were drawn from the front sampling ports of Chambers 1 and 2. Liquid samples for Chamber 3 were extracted from the sampling port fixed to the outlet of Chamber 3. The liquid samples were filtered through a 25 mm syringe filter with 0.45 µm cellulose acetate membrane (VWR International, Radnor, PA) into a 1.5 mL vial containing 10 µL of ≥ 95 percent formic acid (Sigma-Aldrich, Bellefonte, PA) diluted to 25 percent.

A VFA standard curve was generated by using a serial dilution to prepare ten samples of volatile acid standard mix (Supelco, Bellefonte, PA) containing 10 mM concentrations of the following VFAs: acetic acid, propionic acid, iso-butyric acid, butyric acid, iso-valeric acid, valeric acid, iso-hexanoic acid, hexanoic acid, heptanoic acid. (Supelco, Bellefonte, PA). Standards were run at the beginning and end of each set of samples to ensure that an appropriate standard curve was being used for analysis. A blank was run between ABR samples to prevent cross-contamination.

Prepared samples and standards were analyzed using a Hewlett Packard 6890 series GC coupled with 30 m x 0.53 mm x 1.0 µm fused silica capillary column equipped with a FID and nitrogen carrier gas (Wacker, 2014). Minimum detection limits were determined as described in Appendix B. The GC inlet septum and inner liner were replaced once per week.

3.2.8 Alkalinity and pH

The pH and ratio of volatile alkalinity to partial alkalinity (VA/PA) was measured for each ABR chamber three times per week as an additional indicator of reactor health. An AR50 digital pH meter (Fisher Scientific, Waltham, MA) was used for pH measurement. The meter was calibrated prior to each use and pH buffer solutions (Hach, Loveland, CO) were replaced every other week.

The initial pH of a 30 mL sample (undergoing continuous mixing with a magnetic stir bar) was measured, the sample was titrated to a pH of 5.75 with 0.1 N HCl (diluted from 1.0 N) (BDH Acids, Poole, England) to measure PA, and the sample was further titrated to a pH of 4.00 with 0.1 N HCl to measure VA. The VA/PA of the sample was then calculated using sample calculations described in Appendix B.

3.2.9 Statistical analysis of data

The Students t-test for samples with unequal variances and sample sizes was used to test whether differences between data for 15°C and 20°C were significant using the following null hypothesis: the mean value of a parameter at 15°C and 20°C are equal at the 95 percent confidence level. A description of this statistical analysis is found in Appendix B.

3.3 Results & Discussion

3.3.1 Organics and solids removal

The average concentrations of TCOD, SCOD, TSS, VSS, and BOD_5 in the effluent at 15°C and 20°C are shown in Figure 3.3 and Table B.4. Average ± standard deviation TCOD concentrations of 136 ± 26 mg/L and 100 ± 24 mg/L (from about 550 mg/L in the influent) at were measured at 15°C and 20°C, respectively. Average ± standard deviation TSS concentrations of 11 ± 3 mg/L and 8 ± 4 mg/L were measured at 15°C and 20°C, respectively. Concentrations of TCOD, SCOD, and TSS in the effluent agreed with a previous study on the same wastewater which reported effluent concentrations of 99, 71, and 12 mg/L, respectively, with ABR treatment at 63°F (17.2°C) (DiStefano, 2010).

On average, the ABR produced effluent with 81 ± 15 mg/L of BOD_5 during 15°C operation and 60 ± 14 mg/L of BOD_5 during 20°C operation, although the calculated BOD_5 to TCOD ratio of 0.6 ± 0.05 (8 percent error) is likely overestimating BOD_5 in the effluent (Abdalla and Hammam, 2014). Therefore, an ABR that is operated as described here may not be adequate as a sole treatment option at either operating temperature since it produces effluent, which exceeds the 30 mg/L as BOD_5 concentration required by the EPA. ABR treatment should be optimized for enhanced BOD removal—perhaps by increasing HRT. Otherwise, a post-treatment process would be required to reduce organic matter in ABR effluent to safe discharge levels in a full-scale application.

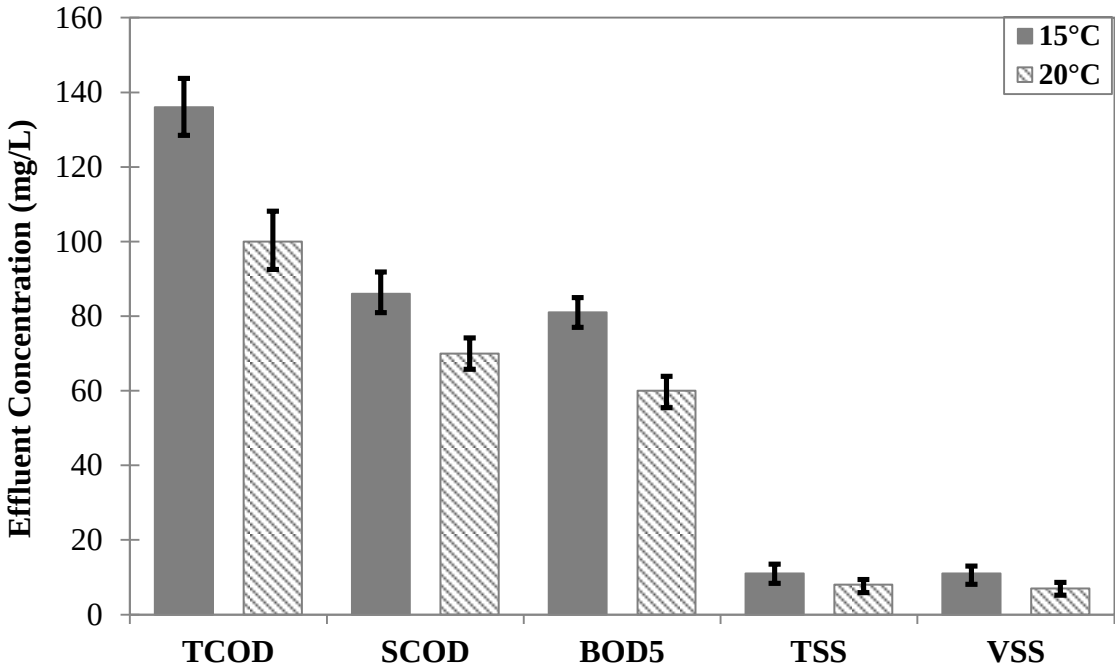


Figure 3.3. Effluent concentrations of TCOD, SCOD, TSS, VSS, and BOD_5 at 15°C and 20°C. Error bars represent 95% CIs.

Concentrations of TCOD, SCOD, TSS, VSS, and BOD_5 in the effluent were lower at 20°C than at 15°C. However, there was substantial variability in these values. A statistical analysis was conducted which determined that the difference between 15°C and 20°C effluent TCOD, SCOD, TSS, VSS, and BOD_5 concentrations were statistically significant at the 95 percent confidence level. Table 3.2 contains a summary of the results of the statistical analysis. Although all effluent concentrations are significantly different at the 95 percent confidence level, it is not clear that decrease of 5°C in operating temperature had a practical impact on effluent quality.

Table 3.2. Statistical analysis comparing influent and effluent concentrations for 15°C and 20°C at a 95% confidence level.

Parameter	Units	$t_{calculated}$	t_{table}	Statistically Different?
Effluent TCOD	mg/L	7.15	1.99	Y
Effluent SCOD	mg/L	4.87	1.99	Y
Effluent TSS	mg/L	2.73	2.03	Y
Effluent VSS	mg/L	2.67	2.10	Y
Effluent BOD_5	mg/L	7.15	1.99	Y
Influent TCOD	mg/L	1.04	1.99	N
Influent SCOD	mg/L	2.07	1.99	Y
Influent TSS	mg/L	0.63	2.04	N
Influent VSS	mg/L	2.67	2.10	Y
Influent BOD_5	mg/L	1.08	1.99	N

The average percent removals achieved by the ABR at 20°C and 15°C are listed in Table 3.3. On average, the ABR achieved a 5 percent higher TCOD removal at 20°C than at 15°C. The difference in TCOD removal values is statistically significant and is comparable to that obtained by Langenhoff and Stuckey in a study which assessed treatment of a 500 mg/L COD wastewater at 35°C, 20°C, and 10°C at a 10 day HRT using an ABR. The authors reported a 25 percent decrease in COD removal when temperature decreased from 35°C to 20°C and a further 10 percent decrease in COD removal when the temperature was lowered to 10°C (Langenhoff and Stuckey, 2000).

Differences in SCOD and VSS removal for 15ºC and 20ºC, however, were not statistically significant. Statistically significant differences between the influent SCOD and influent VSS concentrations during 15°C and 20°C operation may explain why effluent SCOD and VSS concentrations were statistically different, whereas percent removals of SCOD and VSS at 15°C and 20°C were not.

Table 3.3. Steady-state ABR performance at 15°C and 20°C. (average [number of samples] (95% CIs)).

Parameter	Unit	15°C	20°C	Statistically Different?
TCOD removal	%	75 [47] (72,78)	80 [33] (75,85)	Y
SCOD removal	%	61 [46] (57,65)	64 [33] (58,69)	N
TSS removal	%	94 [18] (84,103)	93 [17] (84, 103)	N
VSS removal	%	95 [8] (74,115)	93 [17] (83,103)	N

Figure 3.4 shows the consistent concentrations of TCOD in the effluent achieved at steady-state, even with a highly variable feed. This highlights the ability of the ABR to withstand organic shock loads without significant deterioration in effluent TCOD concentration – a well-documented advantage of the ABR design (Barber and Stuckey, 1999; Foxon et al. 2004; Manariotis and Grigoropoulos, 2002).

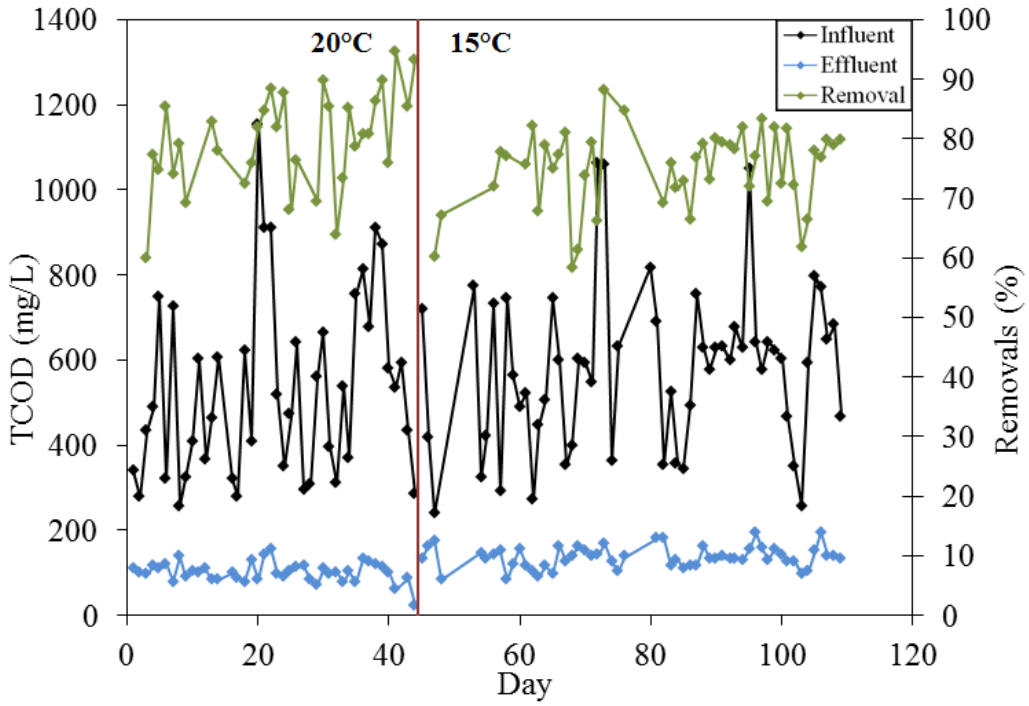


Figure 3.4. Influent TCOD concentration, effluent TCOD concentration, and percent removal of TCOD in a bench-scale anaerobic baffled reactor (ABR) operated at 15ºC and 20ºC.

Data collected for pH, VFAs and VA/PA was used to assess the "health" of the reactor. A "healthy" reactor is indicated by a pH of 6.5 to 8.2 and total VFA concentration lower than influent COD concentration (Speece, 1983). Furthermore, inhibition of methane production may occur if the VA/PA ratio exceeds 0.8 (WEF, 2007). These criteria were met throughout ABR operation. Raw data for pH, VFA, and VA/PA testing in presented in Figures B4-B9 in the appendix.

3.3.2 Methane production

Theoretically, the ABR has the potential to produce a total of 0.376 m^3 CH_4/kg COD removed at 20°C and 0.369 m^3 CH_4/kg COD removed at 15°C at 1 atm pressure. For both temperatures, the ABR produced an average of 2.5 L/d of gaseous methane which translates to an average gaseous specific methane production of 0.17 ± 0.05 m^3/kg COD removed. The gaseous specific methane value is within the range of reported values for laboratory-scale ABRs spanning from 0.09 m^3 CH_4/kg COD removed reported by Yu and Anderson when treating municipal wastewater with an average COD concentration of 386 mg/L COD to 0.18-0.23 m^3 CH_4/kg COD achieved by Krishna and colleagues when treating synthetic wastewater with a COD concentration of 500 mg/L (Yu and Anderson, 1996; Krishna et al. 2008). Furthermore, DiStefano (2010) reported 0.13 m^3/kg COD removed when treating municipal wastewater with the same ABR used in this study at 21°C, but that value decreased to 0.02 m^3/kg COD with a 4°C decrease in temperature – although the decrease in specific methane production was attributed to heavyprecipitation which diluted the feed. Note that these specific methane values represent only the gaseous portion of the methane produced via organics removal. About one-third of the total methane produced by the ABR is trapped in the aqueous phase. When the dissolved methane is

included in the calculation for specific methane yield, the average value for the ABR was 0.28 m^3/kg COD removed. Table 3.4 summarizes values for aqueous and gaseous specific methane production for the ABR.

Table 3.4. Specific methane production at 15°C and 20°C (average [number of samples] (95% CIs)).

	Specific CH_4 production (m^3/kg COD_r)	
	15°C	**20°C**
Gaseous	0.17 [45] (0.15, 0.18)	0.17 [30] (0.15, 0.18)
Aqueous	0.11 [46] (0.10, 0.12)	0.10 [30] (0.09,0.12)

The distribution of produced methane among the gaseous and aqueous phases (Figure 3.5) shows that 35 and 36.5 percent of the total methane produced by the ABR was dissolved at 15°C and 20°C, respectively, which is undesirable from both a GHG emission and energy production standpoint. Furthermore methane solubility increases as temperature decreases which results in higher dissolved methane concentrations in anaerobic reactor effluent. The average percentage of the total produced methane trapped in the aqueous phase increased by approximately 2 percent as a result of the 5°C temperature decrease which, again, indicates a need to consider temperature effects on overall ABR performance.

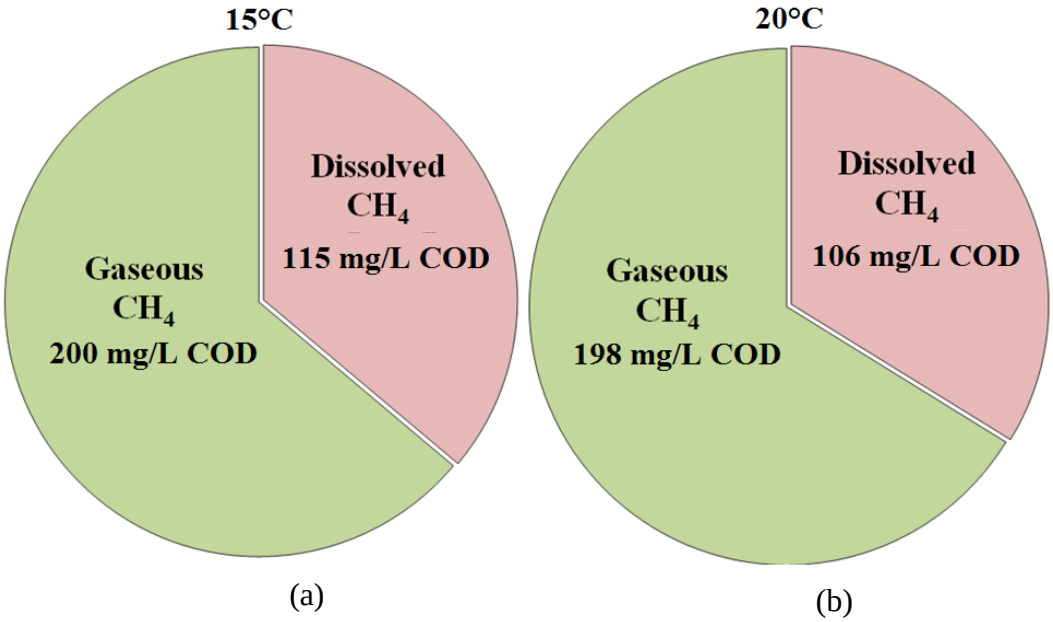


Figure 3.5. Methane (in units of mg COD per L wastewater) distribution in an anaerobic baffled reactor (ABR) among the gaseous and liquid phases at (a) 20°C and (b) 15°C.

High concentrations of dissolved methane in anaerobic treatment systems are not uncommon – particularly for low temperature and dilute feeds. We measured average ± standard deviation dissolved methane concentrations of 29.0 ± 3.3 mg/L and 26.9 ± 2.6 mg/L for 15°C and 20°C operation, respectively. Concentrations of dissolved methane measured in this study are similar to the 20-25 mg/L effluent methane concentration range obtained in a study which used a pilot-scale UASB to treat municipal wastewater at 21-26°C (Urban et al. 2007). In addition, Shin and colleagues measured dissolved methane concentrations of 33 mg/L COD (8.3 mg CH_4/L) and 76 mg/L COD (19 mg CH_4/L) when treating domestic wastewater with an anaerobic membrane bioreactor in fall (20-15°C) and winter (15-8°C), respectively (Shin et al. 2014). Shin et al. also observed a decrease from 61 percent of total methane produced trapped in the dissolved phase to 28 percent as a result of the temperature increase, for winter compared to fall temperatures (Shin et al. 2014).

An average dissolved methane concentration of 39.9 ± 1.1 mg/L was measured for treatment of 300 mg/L COD synthetic wastewater at 20°C by the same ABR used in this research (Wacker, 2014). An experiment was conducted to determine if the 12 mg/L difference was due to differences in the methods used to measure dissolved methane (Appendix B). We concluded that methods were not the source of the discrepancy since the method used in the present study resulted in concentrations of dissolved methane that were 5 mg/L higher than those that resulted from the method used by Wacker. Differences in feed characteristics (e.g., a synthetic feed was used by Wacker, whereas domestic sewage was used for the present study) may have lead to differences in concentrations of dissolved methane.

Several factors may affect dissolved methane concentration in anaerobic effluent such as HRT, influent COD concentration, and SRT. One study estimated that, at 25°C, decreasing the HRT from 28 h to 9 h resulted in an increase in dissolved methane from 1.8 ± 0.2 NL/d[b] to 6.1 ± 0.3 NL/d (Agrawal et al. 1997). The same study determined that decreasing the COD of a synthetic influent from 500 to 300 mg/L resulted in a 20 percent increase in the portion of aqueous phase methane (Agrawal et al. 1997). Similarly, Sanz estimated that methane loss increased from 33 percent to 66 percent as a result of a 400 mg/L to 200 mg/L decrease in influent COD concentration at 20°C which further indicates that influent COD concentration has a noteworthy influence on dissolved methane concentration in anaerobic effluent (Sanz and Fdz-Polanco, 1989). A third study concluded that an increase in SRT from 20 days to 40 days resulted in a decrease in average dissolved methane concentration from 9.9 mg/L to 4.3 mg/L for anaerobic membrane treatment at 23°C (Yeo and Lee, 2013). These studies demonstrate that HRT, strength of influent, and SRT have significant effects on effluent methane concentration.

[b] One normal liter (NL) equals the mass of 1 liter of gas at 1 atm and a standard temperature (0°C or 20°C).

Although the experimentally obtained values for concentration of dissolved methane are consistent with literature, the values for both temperatures were nearly 50 percent greater than theoretical dissolved methane concentrations determined using Henry's Law constants – as seen in Table 3.5. Note that Henry's constants for methane at 15°C and 20°C were calculated using Equation 3.1.

Table 3.5. Dissolved methane concentration, expected dissolved methane concentration based on Henry's Law, and corresponding supersaturation values for 15°C and 20°C. (average [number of samples] (95% CIs)).

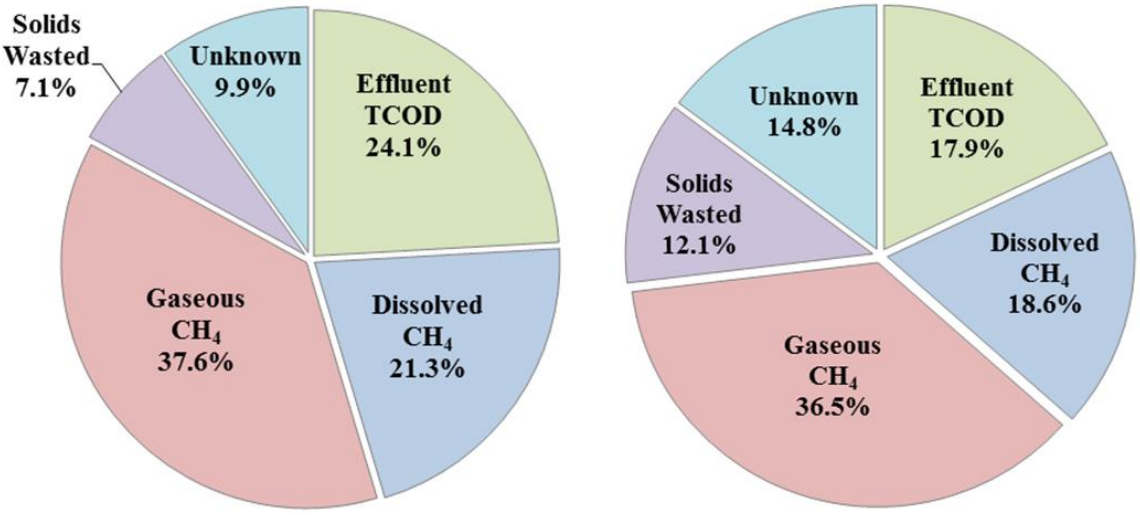
Operating Temperature	Experimental Dissolved CH_4 in Effluent (mg/L)	Expected Dissolved CH_4 in Effluent (mg/L)	Factor Supersaturated
15°C	29.0 [59] (28.1, 29.8)	19.2 (18.9, 19.5)	1.51 (1.47,1.55)
20°C	26.9 [41] (25.5, 28.2)	17.2 (16.4, 17.9)	1.57 (1.48, 1.65)

The ratio of experimental to theoretical dissolved methane (based on Henry's Law) defines the factor by which the system is supersaturated. Souza et al. reported that a supersaturation range of 1.4-1.7 was representative of 24.1-25.1°C treatment with lab- , pilot- and full-scale UASB reactors treating 442 and 520 mg/L COD domestic sewage at 5 and 7 hour HRTs, respectively (Souza et al. 2011). Similarly, a supersaturation value of approximately 1.94 was measured for an anaerobic membrane reactor treating 513 mg/L COD synthetic wastewater (Kim et al. 2011) at 35ºC. However, Wacker observed a supersaturation level of approximately 2.4 when treating 300 mg/L COD synthetic wastewater at 20°C (Wacker, 2014). The value of supersaturation measured by Wacker is comparable to the upper limit of the 0.8-2.2 range reported in a study that assessed an anaerobic migrating bed reactor that treated sewage with an average COD concentration of 550 mg/L at 16-28°C (Hartley and Lant, 2006). Overall, supersaturation is widely reported for anaerobic treatment of low-strength feeds (e.g., domestic

wastewater), demonstrating the need to use measured treatment data for modeling studies that assess the impact of dissolved methane on environmental performance.

3.3.3 COD mass balance

A COD mass balance analysis was conducted to understand the distribution of influent TCOD among effluent TCOD, solids wasted, dissolved methane, and gaseous methane (expressed as COD). The total mass of influent COD over the 15°C and 20°C operating periods was determined and the percentage of the total influent COD that was present as effluent TCOD, solids wasted, dissolved methane, and gaseous methane was evaluated. The calculated distribution of percentages is presented in the COD mass balances (Figure 3.6). The average ± standard deviation of 0.91 ± 0.13 g COD/g VS was used to convert wasted volatile solids into an equivalent mass of COD and all other values except the "Unknown" were measured. The "Unknown" COD was calculated by subtracting all known average COD parameters from the influent TCOD.



(a) (b)

Figure 3.6. COD mass balance for (a) 15°C and (b) 20°C. One hundred percent represents the influent TCOD.

Approximately 10 and 15 percent of influent COD was unaccounted for in the mass balances for 15°C and 20°C, respectively. On average, low to medium strength wastewater typically contains 20-30 mg/L of sulfate (Tchobanoglous et al. 2003). Theoretical calculations using Equations 3.4-3.6 estimated that approximately 0.5-0.7 g COD/d may have been depleted via sulfate reduction assuming a 20-30 mg/L influent sulfate concentration range (Tchobanoglous et al. 2003). Therefore, at 15ºC sulfate reduction could account for 3–5 percent of the influent of COD, reducing the percent of unaccounted-for COD from 10 percent to 5–7 percent. Similarly, at 20ºC, sulfate reduction could account for 3–4 percent of the influent COD, reducing the percent of unaccounted for COD from 15 percent to 11-12 percent.

$$\frac{1}{8}SO_4 + \frac{19}{16}H^+ + e^- \rightarrow \frac{1}{16}H_2S + \frac{1}{16}HS^- + \frac{1}{2}H_2O \quad (3.4)$$

$$\frac{1}{2}H_2O \rightarrow \frac{1}{4}O_2 + H^+ + e^- \quad (3.5)$$

$$\frac{1}{8}SO_4^{2-} + \frac{3}{16}H^+ \rightarrow \frac{1}{16}H_2S + \frac{1}{16}HS^- + \frac{1}{4}O_2 \quad (3.6)$$

Similar results were obtained in a study conducted by Shin and colleagues that considered the fate of COD in a pilot-scale anaerobic fluidized membrane reactor. In that study, approximately 9-18 percent of influent COD was unaccounted for when bulk wasting, sulfate reduction, gaseous methane, dissolved methane, and permeate were considered (Shin et al. 2014). COD mass balances with 98.9-92.7 percent closure conducted for UASB treatment of 300 mg/L synthetic wastewater at 25°C determined that 18-35 percent of influent COD was present as dissolved methane depending on HRT and organic loading (Agrawal et al. 1997). Wacker conducted COD mass balances on the same ABR used in this study using 19 data points and achieved a 90.8 percent recovery at 20°C (Wacker, 2014).

3.4 Conclusion

Temperature had a statistically significant effect on the average TCOD removal of the ABR – resulting in 70 percent removal at 15°C and 80 percent removal at 20°C. However, temperature did not significantly affect the SCOD, TSS, or VSS removal of the system. The ABR effluent contained dissolved methane concentrations of 26.9 mg/L at 20°C and 29.0 mg/L at 15°C which represents nearly one-fifth of the influent COD. Furthermore, nearly one-third of the total methane produced was trapped in the dissolved phase. Consequently, methods to extract dissolved methane prior to discharging effluent would improve the treatment process by increasing the energy efficiency and reducing the greenhouse gas emissions of the ABR.

CHAPTER 4

ENERGY RECOVERY FROM WASTEWATER: LIFE CYCLE ENVIRONMENTAL IMPACTS FOR ANAEROBIC BAFFLED REACTOR AND TRICKLING FILTER WITH ANAEROBIC DIGESTION

4.1 Introduction

Conventional wastewater treatment consumes significant amounts of energy and incurs substantial economic costs from sludge disposal and electricity for aeration. As a result, research into low-energy aerobic and anaerobic treatment alternatives has commenced. One promising aerobic technology, the trickling filter, utilizes natural convection and turbulence within the reactor for aeration and meets effluent quality standards when implemented at full-scale for treatment of domestic wastewater (Jerry Smith, personal communication, 2014). Although the trickling filter reduces electricity requirements for aeration (compared to activated sludge), high solids production and consequent disposal remains a concern. Anaerobic technologies produce less sludge than aerobic systems, eliminate aeration requirements, and produce a methane-rich biogas, which can be harnessed for electricity production, making them an attractive alternative to aerobic treatment. The efficacy of anaerobic treatment, however, decreases as temperature decreases, making it less appealing and potentially unacceptable from an effluent quality perspective at low temperatures. In addition, the presence of dissolved methane in anaerobic effluent may have a detrimental impact on climate change (Liu et al. 2014).

Numerous anaerobic treatment technologies exist, but the ABR achieves a 47.5 percent energy recovery which is higher than that of an upflow anaerobic sludge blanket, anaerobic membrane bioreactor, anaerobic fluidized bed reactor, anaerobic sequencing batch reactor, microbial electrolysis cell, and microbial fuel cell (Shoener et al. 2014). In addition, the ABR decouples solids retention time and hydraulic retention time, allowing for high retention of biomass and, therefore, low solids production (Foxon et al. 2004). However, depending on operating conditions (e.g. temperature, hydraulic retention time, etc.) the ABR may require post-treatment to meet effluent quality standards. An aerobic post-treatment, such as a trickling filter or constructed wetland, may be used to remove residual BOD_5. Constructed wetlands have been proven to effectively treat ABR effluent and provide the additional advantages of low operation and maintenance requirements (Maheesan et al. 2011) and carbon sequestration via foliage (Kalbar et al. 2013).

LCA is a method used to assess and compare the environmental impacts of processes and products over their lifetimes. Several LCA studies have been conducted to compare wastewater treatment technologies (Smith et al. 2014; Cornejo et al. 2013; Pant et al. 2011; Rodriguez-Garcia et al. 2011), but many studies have neglected to include a thorough analysis of sensitivity and uncertainty (Corominas et al. 2013). Furthermore, only one study included impacts from dissolved methane in anaerobic effluent (Smith et al. 2014).

In this study, we assess the life cycle environmental impacts of four process trains that treat 2 MGD of domestic wastewater at 15-20°C: (1) ABR (2) Trickling Filter (3) ABR + Trickling Filter and (4) ABR + Constructed Wetland. We modeled the removal of organics and solids, but assumed no nitrification. The ABR model was constructed using experimental data, described in Chapter 3 and, therefore, does not achieve effluent with a BOD_5 concentration below

30 mg/L which is a limitation of this study. All other models produce effluent with a BOD_5 concentration that meets effluent quality standards. This study includes a sensitivity analysis and a comprehensive analysis of uncertainty conducted with Monte Carlo simulations.

4.2 Methods

4.2.1 Goal & Scope

The goal of this LCA was to use ISO 14040 framework ("CAN/CSA-ISO 14040" 2006) in conjunction with the SimaPro 8.0.4 software (PRé Consultants, Amersfoort, NL) to compare the environmental impacts associated with an ABR, Trickling Filter, ABR with trickling filter post-treatment (ABR + Trickling Filter), and ABR with constructed wetland post-treatment (ABR + Constructed Wetland) for treatment of domestic wastewater. In all cases, the functional unit was treatment of 2 MGD of domestic wastewater for one day in Pennsylvania. The intended use of this LCA is to assess the environmental impacts, particularly those associated with dissolved methane in anaerobic effluent, of four low-energy wastewater treatment systems. The results of this study are intended for use in the scientific community to help guide and influence future research on wastewater treatment.

The boundaries for all scenarios include organics removal, biosolids treatment and disposal, effluent BOD_5 and methane releases, and biogas conversion to electricity. Specifically, the ABR model includes impacts from ABR treatment, cogeneration, dewatering, and disposal of solids (Figure 4.1). The Trickling Filter model includes impacts from primary clarification (required to reduce influent solids to the trickling filter), trickling filter treatment, secondary clarification, thickening, anaerobic digestion, dewatering, and disposal of solids (Figure 4.2). The ABR + Trickling Filter model includes ABR treatment, trickling filter post-treatment, secondary

clarification, thickening, anaerobic digestion, dewatering, and disposal of solids (Figure 4.3). The ABR + Constructed Wetland model includes impacts from ABR treatment, cogeneration, constructed wetlands post-treatment, dewatering, and disposal of solids (Figure 4.4).

Environmental impacts associated with wastewater collection were not considered because those impacts were common to all models (Hospido et al. 2005). Environmental impacts corresponding to construction of clarifiers and reactors were not considered, because construction of wastewater treatment plants accounts for only about 1 percent of total life cycle environmental impacts of wastewater treatment (Kalbar et al. 2013). However, construction impacts of the AD and CHP plants were considered to account for the variation in impacts that may occur due to differences in biosolids and biogas production rate.

Biogenic carbon dioxide emissions were not accounted for, because they are part of the natural short-term carbon cycle which does not contribute to climate change (Kalbar et al. 2013). Carbon dioxide emissions resulting from electricity consumption were modeled in this analysis because those processes are introducing more carbon into the short-term carbon cycle. The GWP of biogenic methane was accounted for, because it contributes significantly to climate change (Solomon et al. 2007). Biogenic methane accounts for approximately 70 percent of total global methane emissions and contributed to the long-term cycle of greenhouse gases (Denman et al. 2007).

4.2.2 System Design & Life Cycle Inventory

Process flow diagrams that model material and energy inputs and outputs were created for the four process trains described in section 4.2.1. Each treatment process was designed for 15°C, but the systems were assumed to operate at 15-20°C.

4.2.2.1 ABR

The base-case model for the ABR was modeled using average values measured for 15-20°C treatment (Table 4.1), and variation in model parameters was accounted for as part of the sensitivity and uncertainty analysis described in Section 4.2.3. The ABR was assumed to be a gravity-flow system with a total volume of approximately 4200 m^3 and a 0.55 day HRT. An input–output block flow diagram that depicts the ABR model is presented in Figure 4.1.

Table 4.1. Experimental performance values used to model 15-20°C with a full-scale ABR.

Parameter	Units	Value	Source
HRT	d	0.55	ABR data
Specific CH_4 production (gas)	L/g COD_r	0.17	ABR data
Dissolved CH_4	mg/L	28	ABR data
Effluent BOD_5	mg/L	72	ABR data

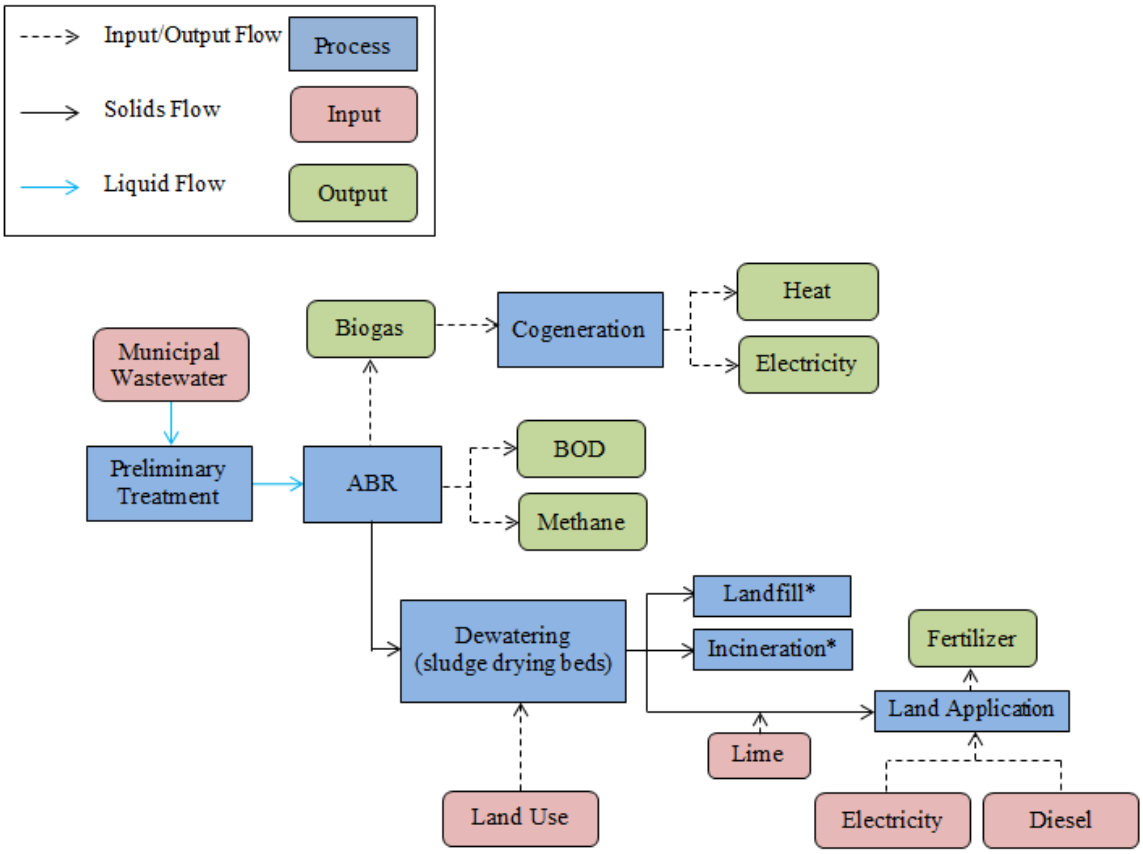


Figure 4.1. Block flow diagram of ABR for primary and secondary treatment of domestic wastewater, with inputs, process names, and outputs. * indicates a process from the ecoinvent database was used.

Methane gas produced via ABR treatment was assumed to be collected and combusted in a 56 kW microturbine CHP unit (based on the kW of electricity produced from the generated biogas) to produce heat and electricity. For the base case, overall CHP efficiency was assumed to be 62 percent and CHP electrical efficiency was assumed to be 25 percent based on performance parameters for microturbine CHP units (US EPA, 2007). The range of reported microturbine efficiencies (24-31 percent) (US EPA, 2007) was accounted for in the uncertainty analysis.

Fugitive methane losses were assumed to be 3.1 percent (for the base case) and 1.7-5.2 percent (Flesch et al. 2011) in the uncertainty analysis. We also assumed that gas clean-up to remove H_2S and other contaminants (e.g. siloxane) consumed approximately 0.25 kWh/kg VS processed (0.41 kWh/m^3 CH_4 produced) during anaerobic digestion (Sills et al. 2013).

Sludge drying beds were modeled for dewatering solids produced by the ABR. Land requirements for the sludge drying bed were calculated using a 120 kg dry solids/m^2-yr sludge loading rate (Tchobanoglous et al. 2003) and the impacts of land use for dewatering were considered in the model. Fine sand and gravel layers were assumed to be 13 inches and 0.6 inches respectively which reflect the averages of reported typical ranges for the layers (Crites and Tchobanoglous ,1998). A 1.56 g/cm^3 bulk density was assumed to calculate the mass of sand required (Ratnayaka et al. 2009). The bulk density of gravel was assumed to be 100 lb/ft^3 (Beverly, 2011).

Dewatered solids were stabilized via addition of 200 kg quicklime per dry ton TS (Suh and Rousseaux, 2002) to achieve standards for class B biosolids required for application to agricultural land (Tchobanoglous et al. 2003). Lime addition is not required for incineration or landfilling (Andreoli et al. 2007). Therefore, lime addition was only included for land application.

The method of sludge disposal can have a significant impact on LCA results (Barberio et al. 2013) and economics (Smith et al. 2014). In this study, an aggregate method of sludge disposal consisting of 66 percent land application, 20 percent incineration, and 14 percent landfilling was used to reflect typical sludge disposal practices in the United States (EPA, 1999). Sludge was transported by a lorry freight truck, and sludge transport distances for incineration, land application, and landfilling were 50 km, 50 km, and 100 km, respectively (Smith et al. 2014).

Energy consumed and fertilizer displaced, for land application, were dependent on the calculated biosolids production for each model (Table 4.2).

Table 4.2. Values used to generate land application process model

Parameter	Unit	Value	Source
Electricity consumption	kWh/ton TS[1]	58.5	Hospido et al. 2005
Diesel consumption	kg/ton TS[1]	0.73	Hospido et al. 2005
Nitrogen fertilizer avoided	kg fertilizer/ton TS[1]	17.87	Hospido et al. 2005
Phosphorus fertilizer avoided	kg fertilizer/ton TS[1]	14.32	Hospido et al. 2005

[1]TS represents the total solids produced by the system (pre-AD where applicable)

Incineration, modeled using a process from the ecoinvent 3.1 database, assumed an autothermal process, and that no net energy was generated from the incinerated solids (assuming an electrical efficiency of 13 percent and a heat efficiency of 25.6 percent) since the solids content of the dewatered sludge was less than 30 percent (Jungbluth et al. 2007).

Landfill impacts associated with disposal of biosolids were approximated using a weighted average of two processes from the ecoinvent database: (1) for landfilling of pulp and paper sludge (for VS), and (2) inert waste (for FS). The process for pulp and paper sludge assumed 60 percent degradability and that all methane and carbon dioxide produced is biogenic (Doka, 2009). The process for landfilling of inert material assumes no emissions from the landfill (Doka, 2009). The calculated biodegradability of disposed solids was 20 percent for the ABR (20 percent VS and 80 percent FS), 40 percent for the ABR + Trickling Filter (40 percent VS and 60 percent FS), and 50 percent for the Trickling Filter (50 percent VS and 50 percent FS).

4.2.2.2 Trickling Filter

Two trickling filter towers (operated in parallel) were designed to treat 2 MGD of domestic wastewater at 15°C. An input/output block flow diagram that depicts the trickling filter model is depicted in Figure 4.2.

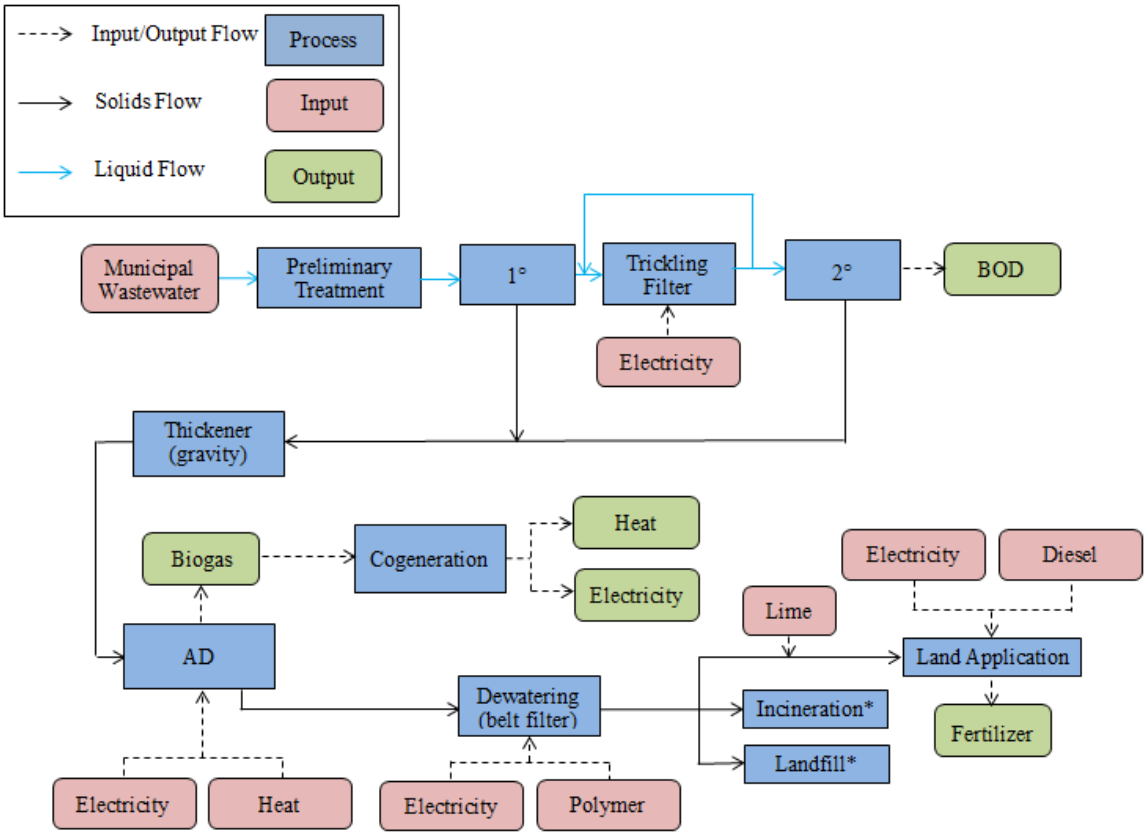


Figure 4.2. Block flow diagram of trickling filter used to for primary and secondary treatment of domestic wastewater, with inputs, process names, and outputs. * indicates a process from the ecoinvent database was used.

Trickling filters require primary clarification to remove solids from the influent and, therefore, reduce the organic loading to the filters to prevent clogging. Primary treatment was modeled for the following conditions: an HRT of 2.5 days and an overflow rate of 1000 gpd/ft^2 were assumed to achieve a clarifier depth of 14 ft and a weir overflow rate of 12700 gpd/ft. The

HRT, overflow rate, clarifier depth, and weir overflow rate are within reported typical ranges (Tchobanoglous et al. 2003). Concentrations of influent TCOD and TSS were based on measured values (Chapter 3), and concentrations of primary effluent TCOD, BOD_5, and TSS were based on values measured by the Milton Regional Sewer Authority.

Primary clarifier effluent was fed into the trickling filter. The trickling filter was designed using the Germain equation (adapted from the Schulze equation) for trickling filters with plastic packing to meet the 30 mg/L BOD_5 EPA limit. Design parameters for 15°C operation were modeled using the following equations adapted from (Tchobanoglous et al. 2003) where

k_2: normalized, site-specific rate constant [$(L/s)^{0.5}/m^2$]
k_1: known rate constant at 6.1 m depth and150 mg/L BOD influent [$(L/s)^{0.5}/m^2$]
h_1: known packing depth corresponding to k_1 [m]
h_t: tower height [m]
S_1: known influent BOD_5 concentration corresponding to k_1 [mg/L]
S_o: site-specific influent BOD_5 concentration [mg/L]
k_T: rate constant at temperature T [$(L/s)^{0.5}/m^2$]
T: air temperature (°C)
HLR: hydraulic loading rate, without recycle [L/m^2-s]
S_e: desired effluent BOD concentration [mg/L]
n: packing coefficient [unitless]
A_t: total tower area [m^2]
Q: influent flow rate [L/d]
V_t: total tower volume [m^3]
D_t: diameter for a single tower
x: number of towers
q_r: recirculation rate [L/m^2-s]
q_{min}: minimum required wetting rate [L/m^2-s]
TOL: total organic loading [kg/m^3-d]
R: recirculation ratio [unitless]

$$k_2 = k_1 \left(\frac{h_1}{h_t}\right)^{0.5} \left(\frac{S_1}{S_o}\right)^{0.5} \quad (4.1)$$

$$k_T = k_2(1.035)^{T-20} \quad (4.2)$$

$$HLR = \left[\frac{k_T h_t}{\ln\left(\frac{S_o}{S_e}\right)}\right]^{\frac{1}{n}} \quad (4.3)$$

$$A_t = \frac{Q\left(\frac{1d}{86400s}\right)}{HLR} \quad (4.4)$$

$$V_t = A_t * h_t \quad (4.5)$$

$$D_t = \sqrt{\frac{4A_t}{x*\pi}} \quad (4.6)$$

$$q_r = q_{min} - HLR \quad (4.7)$$

$$TOL = \frac{QS_o\left(\frac{1\ kg}{10^6\ mg}\right)}{V_t} \quad (4.8)$$

$$R = \frac{q_r}{HLR} \quad (4.9)$$

The constants used in Equations 4.1-4.9 are shown in Table 4.3.

Table 4.3. Constants and assumed values used in trickling filter design calculations (Tchobanoglous et al. 2003)

Constant	Units	Value
k_1	$(L/s)^{0.5}/m^2$	0.21
h_1	m	6.1
S_1	mg BOD_5/L	150
n	unitless	0.5
q_{min}	L/m^2-s	0.5

The design parameters for trickling filter treatment at 15-20°C are listed in Table 4.4. The effluent BOD_5 concentration for the designed system at 20°C was calculated to be 20 mg/L using Equation 4.3.

Table 4.4. Trickling filter design and operating parameters based on a 15-20°C operating temperature.

Parameter	Units	15°C Value	Source
TF influent BOD_5	mg/L	200	MRSA primary clarifier effluent data
TF effluent BOD_5	mg/L	20[a]-30[b]	Calculated[a] (Tchobanoglous et al. 2003), EPA limit[b]
Total tower volume	m^3	2450	Calculated, (Tchobanoglous et al. 2003)
Tower depth	m	6.1	Assumed
Tower diameter	m	16	Calculated
Number of towers	piece	2	Assumed
Total organic loading	kg BOD/m^3-d	0.62	Calculated, (Tchobanoglous et la. 2003)
HLR, including recycle	L/m^2-s	0.5	Minimum required value (Tchobanoglous et al. 2003)
Recirculation ratio	unitless	1.2	Calculated, (Tchobanoglous et al. 2003)

[a]value for treatment at 20°C
[b]value for treatment at 15°C

Although calculations (Equations 2.4-2.6) determined that natural draft would provide sufficient aeration to the system at 15°C and 20°C, one 0.5 kW fan was included per tower as a safety precaution to ensure sufficient aeration (WEF, 2006).

The electricity required to pump influent was also considered in the model. Typically, influent must be pumped the height of the trickling filter tower plus 2.5-3.5 ft (WEF, 2006). The resulting height was used as the total dynamic head to determine pumping requirements. The power consumption for the influent pumps were calculated using Equations 4.10 and 4.11 (Shoener et al. 2014)

$$BHP = \frac{Q*TDH}{3960*\varepsilon_{pump}} \quad (4.10)$$

$$E = \frac{0.746*BHP}{\varepsilon_{motor}} \quad (4.11)$$

where E is energy consumption per pump [kW], BHP is brake horsepower per pump [hp], Q in influent flow rate [gpm], ε_{motor} is motor efficiency [%/100], and ε_{pump} is pump efficiency [%/100]. The pump and motor efficiencies were assumed to be 67.5 percent and 87.5 percent (Spellman 2013). The calculated total power consumption by pumps for 2 MGD treatment was 12.7 kW.

A clarifier depth of 14 ft and an overflow rate of 825 gpd/ft^2 were assumed based on typical clarifier design criteria for trickling filter effluent (Tchobanoglous et al. 2003). An HRT of 3 hours was also assumed to achieve a tank diameter of 56 ft and weir loading rate of approximately 11300 gpd/ft which are within typical secondary clarifier design ranges (Tchobanoglous et al. 2003). Settled biosolids that were produced in the trickling filter exited the secondary clarifiers and combined with solids from the primary clarifier in a single stream before entering a gravity thickener.

A gravity thickener was modeled to increase the sludge concentration of the combined solids stream from primary clarification (4 percent solids) and secondary clarification (1 percent solids) to 5 percent solids, which reflects a typical underflow concentration value for both trickling filter and mixed-primary sludge and trickling filter sludge (WEF et al. 2010). A 3.5 m depth was assumed for the thickener, which is the average of the reported 3-4 m typical range (WEF et al. 2010). Modeling in CapdetWorks determined that a 90 ft diameter was both sufficient and appropriate for the thickener (Hydromantis ESS, Inc. 2014). The gravity thickened solids were then anaerobically digested.

A process for single-stage high-rate anaerobic digestion was included in the model to reduce solids for disposal and produce biogas. The digester was assumed to be a cylindrical tank with a 7 m diameter and 1V:6H bottom slope operating at 35°C with a 15 day SRT

(Tchobanoglous et al. 2003). It was assumed, as recommended by Tchobanoglous, that the digester walls were constructed from plain concrete surrounded by dry earth, and the floors were constructed from plain concrete in contact with moist earth (Tchobanoglous et al. 2003). The cover was assumed to be floating and insulated which is typical for single-stage high-rate digestion (Tchobanoglous et al. 2003; Smith et al. 2014). The dissolved methane present in the liquid digestate from anaerobic digestion was not accounted for.

Anaerobic digestion design assumptions (Table 4.5) were used to calculate the heat and electricity requirements, heat losses, and methane production for anaerobic digestion (Table 4.9) in each model using Equations 4.12-4.16 (adapted from Tchobanoglous et al. 2003)

Table 4.5. Values used in anaerobic digestion calculations. All values were obtained from Tchobanoglous et al. 2003 unless noted otherwise.

	Symbol	Unit	Value
VS destroyed in digester	VS_{des}	%	56
CH_4 content of biogas	P_{CH4}	%	68[1]
Specific heat coefficient, walls	U_{walls}	W/m^2-°C	0.63[1]
Specific heat coefficient, floor	U_{floor}	W/m^2-°C	2.85
Specific heat coefficient, roof	U_{roof}	W/m^2-°C	0.95[1]
Solids content of sludge	S	g VSS/100 g slurry	5[2]
Specific gravity of sludge	γ	-	1.02
Sludge density	ρ	kg/m^3	1020
Specific heat of sludge	c	MJ/kg-°C	0.0042
Biogas yield	Y_{biogas}	m^3 biogas/kg VS destroyed	0.94[1]
CH_4 lower heating value, 20°C	LHV_{CH4}	MJ/m^3	35.8
Power required, mixing	P_{mix}	kW/m^3 digester volume	0.0065[1]

[1]indicates the average value of a reported typical range (Tchobanoglous et al. 2003)
[2]typical value for gravity thickened trickling filter sludge (WEF et al. 2010)

$$V = \frac{VS*SRT}{S*\rho} \quad (4.12)$$

$$q_{sludge} = \frac{V*\rho*\Delta T*c}{SRT} \quad (4.13)$$

$$q_{tank} = \frac{\left(U_{walls} * SA_{walls} + U_{floor} * SA_{floor} + U_{roof} * SA_{roof}\right) * \Delta T * 86{,}400 \frac{s}{day}}{10^6 \frac{J}{MJ}} \quad (4.14)$$

$$E_{mix} = P_{mix} * V * 24 \frac{h}{day} \quad (4.15)$$

$$Y_{CH4} = VS * VS_{des} * Y_{biogas} * P_{CH4} \quad (4.16)$$

where V is digester tank volume [m^3], VS is volatile solids entering the digester [kg/d] (calculation detailed in Appendix D), q_{sludge} is the heat requirement for sludge entering the digester [MJ/d], ΔT is the difference between the digester temperature (35°C) and ambient temperature (15°C or 20°C) [°C], q_{tank} is the heat lost from the tank due to conduction [MJ/d], $E_{mix}$is the electricity required for mechanical mixing of the digester [kWh/d], Y_{CH4} is the total methane yield of the anaerobic digester per day [m^3/d], SA is surface area [m^2], and all other variables are as defined in Table 4.5.

The biogas produced via anaerobic digestion was collected and then combusted in a microturbine to produce heat and electricity. A 43 kW turbine was used in this model and was designed as described in Section 4.2.2.1.

Solids exiting the digester were dewatered using a belt filter press with a belt width of 1 meter. Electricity consumption [kWh] and acrylonitrile (polymer) consumption [kg] required for dewatering were calculated using the total solids production (Table 4.6). The dewatered solids cake was assumed to have a 22 percent solids content, which is a typical value reported for dewatered digestate (Tchobanoglous et al. 2003).

Table 4.6. Values used to generate dewatering process model

Parameter	Unit	Value	Source
Electricity consumption	kWh/ton TS	49.09	Hospido et al. 2005
Polymer consumption	kg/ton TS	5.50	Hospido et al. 2005

The dewatered sludge exiting the digester was disposed of using the methods described in Section 4.2.2.1.

4.2.2.3 ABR + Trickling Filter

The ABR + Trickling Filter scenario consists of an ABR followed by a trickling filter, which was designed to oxidize dissolved methane present in the ABR effluent and reduce effluent BOD_5 concentration to 30 mg/L. The solids produced by the ABR and trickling filter were combined, thickened, and anaerobically digested—as presented in Figure 4.3. It was assumed that all dissolved methane produced by the ABR was removed via the trickling filter post-treatment. Therefore, impacts associated with methane emissions are assumed to be negligible in this model. Although this assumption is most likely not correct for a trickling filter, it allowed us to compare impacts for a post-treatment process that oxidizes all dissolved methane (e.g., biological filter, or perhaps a rotating biological contactor) to a post treatment process in which all dissolved methane is lost to the atmosphere (e.g., constructed wetlands).

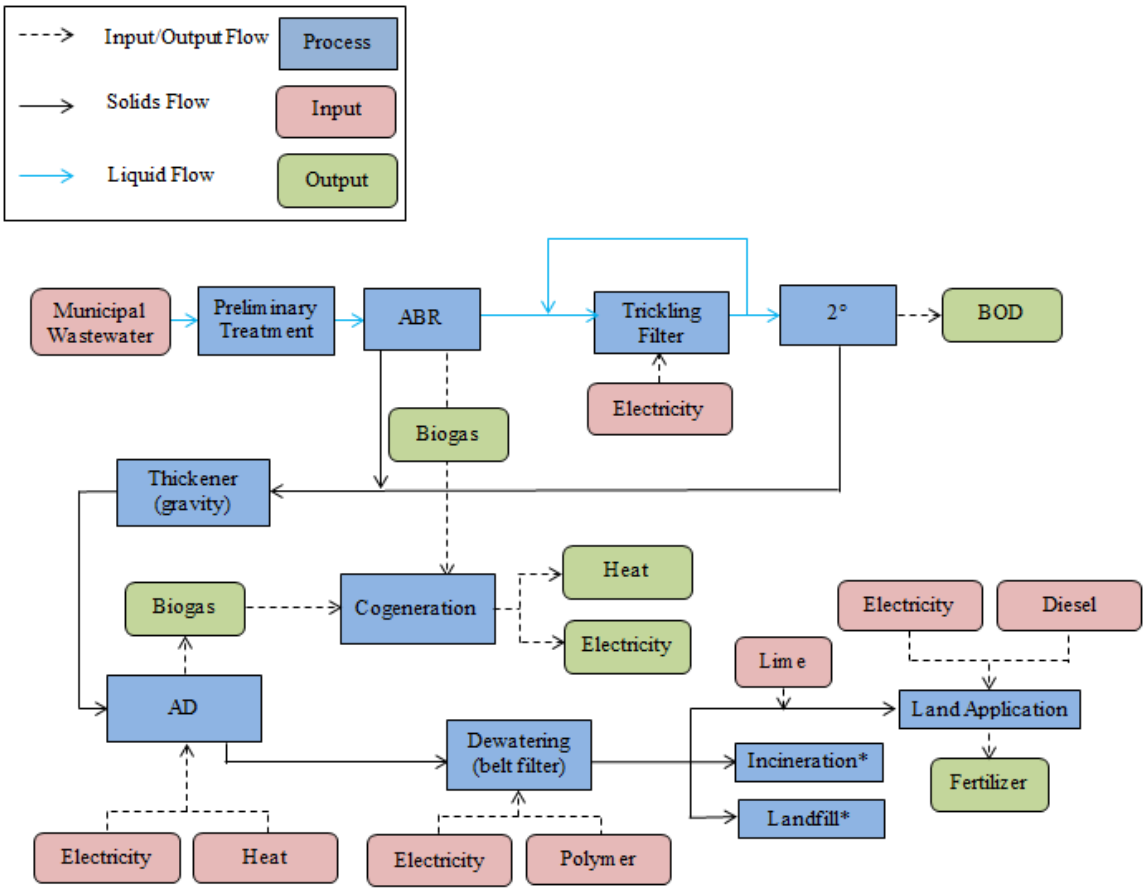


Figure 4.3. Block flow diagram of ABR + Trickling Filter for primary and secondary treatment of domestic wastewater, with inputs, process names, and outputs. *indicates a process from the ecoinvent database was used.

The trickling filter was designed to treat ABR effluent at 15°C, and was assumed to be operated at 15–20ºC. The design and operating parameters are listed in Table 4.7. Note that the sum of BOD_5 and dissolved methane concentration was used as the influent BOD_5 concentration to the trickling filter. Otherwise, design calculations were conducted as described in section 4.2.2.2. Since the trickling filter was designed for 15°C, the BOD_5 concentration in the effluent of the trickling filter was reduced to 18 mg/L for 20°C. In addition, the organic loading at 20°C

decreased to 0.54 kg BOD/m^3-d as a result of the decreased concentrations of BOD_5 and dissolved methane in the ABR effluent.

Table 4.7. Design and operating parameters for a trickling filter used as post-treatment at 15-20°C.

Parameter	Units	Value	Source
ABR effluent BOD_5	mg/L	60[a]-81[b]	ABR effluent data
ABR effluent, dissolved CH_4	mg/L COD	108[a]-116[b]	ABR effluent data
TF influent BOD_5	mg/L	168[a]-197[b]	ABR effluent data
TF effluent BOD_5	mg/L	18[a]-30[b]	Calculated[a] Tchobanoglous et al. 2003, EPA limit[b]
Total tower volume	m^3	2450	Calculated, (Tchobanoglous et al. 2003)
Tower depth	m	6.1	Assumed
Tower diameter	m	16	Calculated
Number of towers	piece	2	Assumed
Total organic loading	kg BOD/m^3-d	0.52[a]-0.61[b]	Calculated, ABR effluent data
HLR, including recycle	d	0.5	Minimum required value, (Tchobanoglous et al. 2003)
Recirculation ratio	unitless	1.2	Calculated, (Tchobanoglous et al. 2003)

[a]value for treatment at 20°C
[b]value for treatment at 15°C

Biogas produced by the ABR and anaerobic digester were combined before entering a 71 kW microturbine for cogeneration. Otherwise, cogeneration and disposal of solids were modeled as described previously in section 4.2.2.1. Gravity thickening and belt-filter press dewatering were modeled as described in section 4.2.2.2. Anaerobic digestion was modeled as described in section 4.2.2.2 except that the diameter of the digester was adjusted to 4 m to reflect the reduced solids production of the ABR + Trickling Filter assembly.

4.2.2.4 ABR + Constructed Wetland

In this scenario, the ABR effluent is directed into a sub-surface horizontal-flow constructed wetland to reduce organics in ABR effluent to a safe discharge level. All dissolved methane present in ABR effluent was assumed to be released to the atmosphere through the free water surface of the constructed wetlands. The block flow diagram for the ABR + Constructed Wetland is the combination of the ABR block flow diagram (Figure 4.1) and the CW post-treatment process shown in Figure 4.4. Since the constructed wetlands are an addition to the ABR model, the same methods used to model cogeneration, dewatering, and disposal of solids used in the ABR model were used in this model (Figure 4.1).

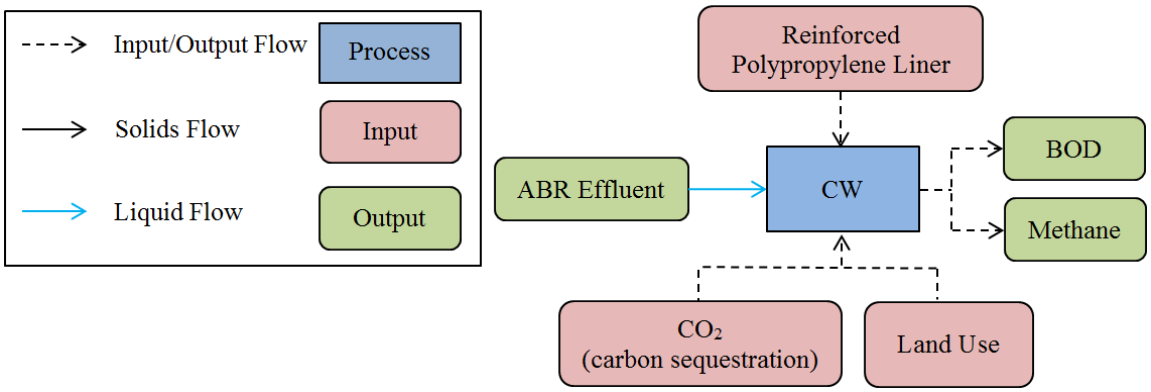


Figure 4.4. Block flow diagram of constructed wetlands used as a post-treatment portion of the ABR + Constructed Wetland model.

Constructed wetlands have negligible nonrenewable energy consumption which make them a relatively inexpensive post-treatment option (Mannino et al. 2008; Kalbar et al. 2013). In addition, constructed wetland post-treatment does not require biosolids disposal (Cheema et al. 2014), although the solids from ABR pre-treatment must still be disposed. A final advantage of constructed wetland treatment is the reduction in carbon dioxide impacts via carbon sequestration

at a rate of approximately 3.3 kg/m^2-yr which result in a negative GWP for the constructed wetland (Kalbar et al. 2013). To quantify the amount of carbon sequestered and determine the total land required for the CW, the required surface area was calculated using Equation 4.17 (adapted from Vesilind, 2003)

$$A_{cw} = \frac{Q*HRT*3.07}{\eta*d} \quad (4.17)$$

where A_{cw} is the wetland surface area [ha], Q is the influent wastewater flow rate [m^3/d], HRT is the hydraulic retention time [days], η is the void ratio of the wetland [unitless], and d is depth [m]. A void ratio of 0.7 was assumed since the typical range is 0.65-0.75 (Crites and Tchobanoglous, 1998). The HRT [days] was calculated using Equation 4.18 (adapted from Crites and Tchobanoglous, 1998)

$$HRT = -\frac{ln\left(\frac{C_e}{C_o}\right)}{k} \quad (4.18)$$

where C_o is the influent BOD_5 [mg/L] concentration to the CW, C_e is the effluent BOD_5 [mg/L] concentration of the CW system, and k [d^{-1}] is the temperature-corrected apparent BOD_5 removal-rate constant (Crites and Tchobanoglous, 1998). The known k value at 20°C (0.678 d^{-1}) was adjusted for 15°C treatment using Equation 4.19 (adapted from Crites and Tchobanoglous. 1998).

$$k_{15} = k_{20}(1.02)^{15-20} \quad (4.19)$$

Table 4.8 contains a summary of the values pertaining to the CW model. Since the CW was designed for 15°C, it produced higher quality effluent (20 mg/L as BOD_5) at 20°C operation.

Table 4.8. Design and operating parameters for a constructed wetland used as post-treatment at 15–20°C.

Parameter	Units	15°C Value	Source
A_{cw}	ha	3.8	Calculated, (Vesilind, 2003)
Average depth	ft	1.5	Assumed, (Crites and Tchobanoglous, 1998)
k	d^{-1}	0.61	Calculated, (Crites and Tchobanoglous, 1998)
HRT	d	1.6	Calculated, (Crites and Tchobanoglous, 1998)
Void ratio	unitless	0.7	Assumed, (Crites and Tchobanoglous, 1998)
CW influent BOD_5	mg/L	60[a]-81[b]	ABR effluent data
CW effluent BOD_5	mg/L	20[a]-30[b]	Calculated[a], EPA limit[b]

[a]value for treatment at 20°C
[b]value for treatment at 15°C

A 1 mm thick reinforced polypropylene liner totaling 35950 kg was included in the 3.8 ha wetland design to reduce seeping of contaminated water (Sills et al. 2013). However, the addition of a liner is conservative since it may not be necessary if the soil along the bottom of the wetland is clay-rich (Crites and Tchobanoglous, 1998).

4.2.2.5 Life Cycle Inventories

The system design and input-output models were used to construct life cycle inventories (LCIs) for treatment at 15-20°C (Table 4.9). Separate inventories for treatment at 15°C and 20°C (Table E.1 and Table E.2 in the appendix) were used to assess eutrophication, which varied more significantly with temperature. All electricity impacts presented in Table 4.9 reflect the medium voltage mix for the Northeast United States (Reliability First Corporation (RFC) region). Resources used to produce electricity for the RFC grid consist of 60 percent coal, 29 percent nuclear, 8 percent natural gas, and the remainder from wind and hydropower. In addition, all background life cycle inventories were taken from the Ecoinvent 3.1 database except the

following: acrylonitrile (USLCI database), quicklime (USLCI database), and diesel (ELCD database).

Table 4.9. Life cycle inventory for 15-20°C treatment of 2 MGD domestic wastewater.

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
ABR	Effluent BOD_5	mg/L	72	-	-	-
	Effluent dissolved CH_4	mg/L	28	-	-	28
	Gaseous CH_4 production	m^3/d	2 E+05	-	2 E+05	2 E+05
Trickling filter	Pump, electricity	kWh/d	-	307	307	-
	Fan, electricity	kWh/d	-	24	24	-
	Effluent BOD_5	mg/L	-	30	30	-
Constructed wetland	Sequestered carbon dioxide	kg	-	-	-	343
	Land occupation	ha	-	-	-	3.8
	Polypropylene liner	kg	-	-	-	3.60 E+04
	Effluent BOD_5	mg/L	-	-	-	30
Anaerobic digestion	Mixing, electricity	kWh/d	-	58	20	-
	Heat losses	MJ/d	-	4.22 E+02	1.89 E+02	-
	Heat consumed	MJ/d	-	2.11 E+03	6.55 E+02	-
	Gaseous CH_4 production	m^3/d	-	1.52 E+05	4.72 E+04	-
	Construction, AD plant for sewage sludge	piece	-	5.61 E-02	1.74 E-02	-
Cogeneration	Electricity generated	kWh/d	1.35 E+03	1.03 E+03	1.67 E+03	1.35 E+03
	Heat generated	MJ/d	7.26 E+03	5.51 E+03	8.99 E+03	7.26 E+03
	Construction, heat and power cogeneration unit, 160 kW_e	piece	0.354	0.268	0.453	0.354
Dewatering, sludge drying bed	Land occupation	m^2	741	-	-	741
	Sand	kg	1.82 E+04	-	-	1.82 E+04
	Gravel	kg	3.81 E+05	-	-	3.81 E+05
Dewatering, belt filter	Electricity	kWh/d	-	89	34	-
	Acrylonitrile (polymer)	kg/d	-	10	4	-
Sludge disposal (incineration only)	Sludge treated, municipal incineration	kg/d	248	1100	472	248
	Sludge transport, freight, lorry	kg*km/d	1.24 E+04	5.51 E+04	2.36 E+04	1.24 E+04

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
Sludge disposal (landfill only)	Solid waste treated, sanitary landfill	kg/d	248	1100	472	248
	Sludge transport, freight, lorry	kg*km/d	2.48 E+04	1.11 E+05	4.71 E+04	2.48 E+04
Sludge disposal (land application only)	Nitrogen fertilizer avoided	kg/d	4	20	8	4
	Phosphorus fertilizer avoided	kg/d	4	16	7	4
	Quicklime	kg/d	50	220	95	49
	Electricity, consumed	kWh/d	15	106	41	15
	Diesel, from crude oil, consumption mix	kg/d	0.2	1.3	0.5	0.2
	Solids for land application	kg/d	248	1100	472	248
	Sludge transport, freight, lorry	kg*km/d	1.24 E+04	5.51 E+04	2.36 E+04	1.24 E+04

Although heat consumption, generation, and losses are listed in the LCI, excess heat (beyond what is needed to heat the anaerobic digester) would likely be wasted in practice. Therefore, we considered impacts for heat, only if net heat consumption occurred. The heat components and net heat associated with 15-20°C treatment are shown in Table 4.10.

Table 4.10. Daily heat consumption, generation, and losses for treatment at 15-20°C.

		ABR	TF	ABR+TF	ABR+CW
Heat lost, AD	MJ/d	-	4.22E+02	1.89E+02	-
Heat required, sludge	MJ/d	-	2.11E+03	6.55E+02	-
Heat generated	MJ/d	-7.26E+03	-5.51E+03	-8.99E+03	-7.26E+03
Net heat consumption	**MJ/d**	**-7.26E+03**	**-2.98E+03**	**-8.15E+03**	**-7.26E+03**

As seen in Table 4.10, the model generated more heat than it consumed. Therefore, the impacts of heat generation and consumption from cogeneration were not considered in the analysis. This approach produces more conservative impacts for each model, although it

underestimates potential benefits if the facilities were to utilize excess heat for purposes such as improving reactor performance, heating the treatment facility, or drying solids.

4.2.3 Life Cycle Impact Assessment

We used IPCC 2013, IMPACT2002+ and TRACI 2.1 for impact assessment. IPCC 2013 converts GHG emissions to impacts on climate change and IMPACT2002+ converts elementary flows to impacts on four damage categories: climate change (kg CO_2 equivalents), human health (disability adjusted life years (DALYs)), resource depletion (MJ primary energy), and ecosystem quality (potentially disappeared fraction of species (PDF)-m^2-yr). IMPACT2002+ converts the four damage categories to units of LCA "points" using the European factors shown in Table 4.11 (factors for the United States are also shown) which allows for direct comparison of impacts across the categories. One point represents one year of impacts for one person. TRACI 2.1 assesses impacts for 14 midpoint categories (section 2.5.3.2) including eutrophication. An analysis (described in Appendix F) was conducted to compare mid-point impacts from IMPACT 2002+ and TRACI 2.1.

Table 4.11. Factors used for normalization of damage category impacts (adapted from (Humbert et al. 2012).

	Europe	United States	Units
Resource Depletion	152,000	203,000	MJ Primary/point
Human Health	0.0071	0.0388	DALY/point
Ecosystem Quality	13,700	4380	PDF-m^2-yr/point
Climate Change	9,950	20,600	kg CO_2-eq/point

Since IMPACT2002+, IPCC 2013, and TRACI 2.1 quantify impacts on climate change, an analysis was conducted to determine which method should be used in this study. A major impact of interest in this study is biogenic methane released in the effluent after anaerobic

treatment. The global warming potential values assigned to biogenic methane using four impact assessment methods were evaluated (Table 4.12).

Table 4.12. Global warming potential of 1 kg biogenic methane using various impact assessment methods.

Method	**GWP, 1 kg biogenic CH_4 (kg CO_2 eq.)**
IMPACT 2002+	4.9
IPCC 2013, 20 yr	82.3
IPCC 2013, 100 yr	25.3
TRACI 2.1	22.3

The results shown in Table 4.12 indicate that using IPCC 2013, 100-year as an impact assessment method instead of IMPACT 2002+ would increase its impact by a factor of approximately 5. Table 4.12 also shows the effect of time horizon on climate change impacts. A 20-year time horizon for assessment with IPCC 2013 would result in a climate change impact nearly three times greater than the impact resulting from a 100-year time horizon. Although both values are correct, the 20-year time horizon is generally used in cases where the rate of climate change is of interest and the 100-year time horizon is generally used where the eventual magnitude of climate change impact is the main focus (IPCC 1994). Since this study is interested in the eventual total impact on climate change, and since the 100-year time horizon is most often used in LCA, IPCC 2013 with a 100-year time horizon was used in this study.

Although the Climate Change impact was determined with the IPCC 2013, 100 yr method, we converted GWP values from kg CO2-eq to LCA points using the factor shown in Table 4.11, so we could compare climate change impacts to the other three damage categories (resource depletion, ecosystem quality, and human health) assessed with IMPACT 2002+.

Effluent quality was inconsistent between the models due to differences in treatment efficiency at 15°C and 20°C. TRACI 2.1 was used to account for that difference since it expresses the impact of BOD in terms of its eventual resulting impact on eutrophication. IMPACT 2002+ does not associate an impact with BOD and, therefore, does not account for the differences in effluent quality between treatment systems.

4.2.3.1 Sensitivity Analysis

Several model parameters are variable as a result of day-to-day fluctuation in the unit processes (e.g. dissolved methane concentration, solids production) and/or uncertainty. Much of the uncertainty in the present study results from variability in the experimental data used to construct the ABR model, and simple untested designs of unit processes, such as the trickling filter and anaerobic digester. A sensitivity analysis was conducted to assess the effect of parameter variations on LCA impacts.

Ranges for "Dissolved CH_4" and "gaseous CH_4 yield" (Table 4.13) reflect the average, minimum, and maximum values obtained during 15-20°C ABR operation. Ranges for the "disposed solids" (Table 4.13) were calculated to reflect the average, highest, and lowest potential solids production of each model. All other sensitivity analysis values and ranges were obtained from the literature.

Table 4.13. Values considered in sensitivity analysis of ABR and Trickling Filter models.

Process	Parameter	Unit	Baseline Value	Low Case	High Case
TF Treatment	Motor & pump efficiency	%	59[a]	40[a]	81[a]
ABR Treatment	Dissolved CH_4	mg/L	28	22	35
	Gaseous CH_4 yield	m^3/kg BOD_r	0.17	0.07	0.29
Anaerobic Digestion	Biogas yield	m^3 biogas/kg VS_r	0.94[b]	0.75[b]	1.12[b]
	VS removed	%	56[b]	50[c]	62[c]
	CH_4 content, biogas	%	65[b]	60[b]	70[b]
	Power, mixing	kW/m^3 digester	0.0065[b]	0.005[b]	0.008[b]
Cogeneration	Fugitive CH_4 emissions	%	3.1[d]	1.7[d]	5.2[d]
	Electrical efficiency	%	25[e]	24[e]	31[e]
Sludge Disposal	Transport distance, incineration	km	50[f]	10[f]	160[f]
	Transport distance, landfill	km	100[f]	10[f]	160[f]
	Transport distance, land application	km	50[f]	10[f]	160[f]
	Disposed solids, ABR	kg TS/d	272	26	377
	Disposed solids, TF	kg TS/d	1109	1012	1263

[a]Spellman 2013
[b]Tchobanoglous et al. 2003
[c]estimated using ± 10 percent of typical value from Tchobanoglous et al. 2003
[d]Flesch et al. 2011
[e]US EPA 2007
[f]Smith et al. 2014

To conduct a sensitivity analysis, we varied each parameter individually to its minimum and maximum values (Table 4.13) while keeping all other parameters constant. The maximum value corresponds to a "high case" and the minimum value corresponds to a "low case" relative to the base case generated using the average parameter value (Figures 4.11-4.18). The resulting changes in overall impacts were assessed for each model using IPCC 2013 (GWP) and IMPACT2002+ (resource depletion, ecosystem quality, and human health).

4.2.3.2 Uncertainty Analysis

We conducted an uncertainty analysis by running 1,000 Monte Carlo simulations within SimaPro for each model to assess the effect of simultaneous variations in multiple parameters on LCA impact categories. The uncertainty analysis used the following probability distribution functions (PDFs) to represent the variability of model parameters: normal, log-normal, triangular, or uniform. Parameter values that were calculated or obtained from literature were assigned a triangular distribution if derived from limited data sets with a defined mid-point, and uniform distributions if derived from limited data sets without a well-defined mid-point (Smith et al. 2014). Where sufficient data sets were available, normal and log-normal distributions were fit to the data using the Distribution Fitting Toolbox in Matlab (2013, Mathworks, Natick, MA).

The distribution, mean, and standard deviation of ABR data (Table 4.14) for gaseous methane production, dissolved methane concentration, COD concentration, and effluent BOD concentration were determined using the Distribution Fitting Toolbox in MATLAB. We chose between normal and log normal distributions by visual inspection of the fits. MATLAB was used to determine the standard deviation (σ) which was transformed to input values for normal and lognormal distributions in SimaPro using Equations 4.20 and 4.21, respectively

$$2\sigma = 2 * \sigma \tag{4.20}$$

$$GSD^2 = (e^{\sigma})^2 \tag{4.21}$$

where GSD is the geometric standard deviation. Note that the left and right sides of Equation 4.20 and Equation 4.21 represent values that where entered into SimaPro and obtained from Matlab, respectively.

Table 4.14. Distributions, modes, calculated geometric standard deviation (GSD), and standard deviation (σ) values used for uncertainty analysis.

Data Set	Distribution Type	MATLAB Values		SimaPro Values	
		σ	**Mean**	**GSD^2**	**2σ**
Specific CH_4 production, gas	Lognormal	0.34	0.17	2	-
Dissolved CH_4	Normal	3	28	-	6
Influent COD	Normal	189	576	-	378
Effluent COD	Normal	25	136	-	50
Effluent BOD_5	Normal	18	72	-	36

All other value ranges were calculated or defined based on average, minimum, and maximum values obtained from literature (Table 4.15).

Table 4.15. Baseline, minimum, and maximum values and distribution types used for uncertainty analysis.

Process	Parameter	Unit	Baseline Value	Distribution	Min.	Max.
TF Treatment	Motor efficiency	%	87.5[a]	Uniform	80[a]	95[a]
	Pump efficiency	%	67.5[a]	Triangle	50[a]	85[a]
Anaerobic Digestion	Biogas yield	$m^3/kg\ VS_r$	0.94[b]	Triangle	0.75[b]	1.12[b]
	VS removed	%	56[b]	Triangle	50[c]	62[c]
	CH_4 content, biogas	%	65[b]	Triangle	60[b]	70[b]
	Power, mixing	kW/m^3 digester	0.0065[b]	Triangle	0.005[b]	0.008[b]
	Influent VS, TF	kg VS/d	1257	Triangle	1160	1423
	Influent VS, ABR+TF	kg VS/d	390	Triangle	327	512
Cogeneration	Fugitive CH_4 losses	%	3.1[d]	Triangle	1.7[d]	5.2[d]
	Electrical efficiency	%	25[e]	Triangle	24[e]	31[e]
Sludge Disposal	Transport distance, incineration	km	50[f]	Uniform	10[f]	160[f]
	Transport distance, landfill	km	100[f]	Uniform	10[f]	160[f]
	Transport distance, land application	km	50[f]	Uniform	10[f]	160[f]
	Disposed solids, TF	kg TS/d	1100	Triangle	1002	1257
	Disposed solids, ABR & ABR+CW	kg TS/d	248	Triangle	58	653
	Disposed solids, ABR+TF	kg TS/d	472	Triangle	283	1063

[a]Spellman 2013
[b]Tchobanoglous et al. 2003
[c]estimated using ± 10 percent of typical value from Tchobanoglous et al. 2003
[d]Flesch et al. 2011
[e]US EPA 2007
[f]Smith et al. 2014

4.3 Results & Discussion

Assessment of climate change impacts with IPCC 2013 (Figure 4.5) showed that the Trickling Filter and ABR + Trickling Filter treatment scenarios resulted in a global warming potential (GWP) that is about five times lower than for the ABR and ABR + Constructed Wetland scenarios. The large gap between climate change impacts is due to dissolved methane which contributes approximately 95 percent of the GWP for the ABR model. Smith et al. concluded that

dissolved methane contributed to 75 percent of the GWP from treatment via anaerobic membrane reactor while 19 percent of the remaining impact resulted from electricity requirements for the reactor (Smith et al. 2014). Results also show that the ABR produced 0.6 kg CO_2-eq per m^3 wastewater treated, which is about two-thirds of the 0.9 kg CO_2-eq per m^3 value obtained for an anaerobic membrane reactor treatment of wastewater at 15°C (Smith et al. 2014). These results indicate that dissolved methane has a noteworthy impact in both low- and high-energy anaerobic systems.

Results also confirm allegations that dissolved methane may negate the beneficial impact obtained from electricity production in anaerobic systems (Liu et al. 2014). Dissolved methane produced an impact approximately five times greater than the benefits resulting from electricity production from ABR biogas (which was assumed to replace grid electricity). Even post-treatment with constructed wetlands, which sequesters carbon dioxide at a rate significant enough to produce a negative overall impact on GWP (Kalbar et al. 2013), was not able to counteract the detrimental impact of dissolved methane (Figure 4.5). This indicates that ABR treatment at 15-20°C is not beneficial in terms of climate change if no post-treatment or modification is made to mitigate dissolved methane in the effluent.

Knowing that dissolved methane can have a significant impact at 20°C indicates that the dissolved methane present in the effluent of the anaerobic digester may also have a substantial impact since the operating temperature was assumed to be 35°C. For instance, a digester at 35°C with 10 percent TS, a supersaturation of 1.5 for dissolved methane, and 68 percent methane content in its biogas would result in a climate change impact of approximately 8 kg CO_2-eq per day for 2 MGD treatment. That value is negligible compared to the 5342 kg CO_2-eq produced from dissolved methane present in effluent of the ABR, indicating that the Trickling Filter and ABR + Trickling Filter would still be preferable to the ABR in terms of GWP if the climate

change impact of dissolved methane from the liquid digestate from the digester were considered.

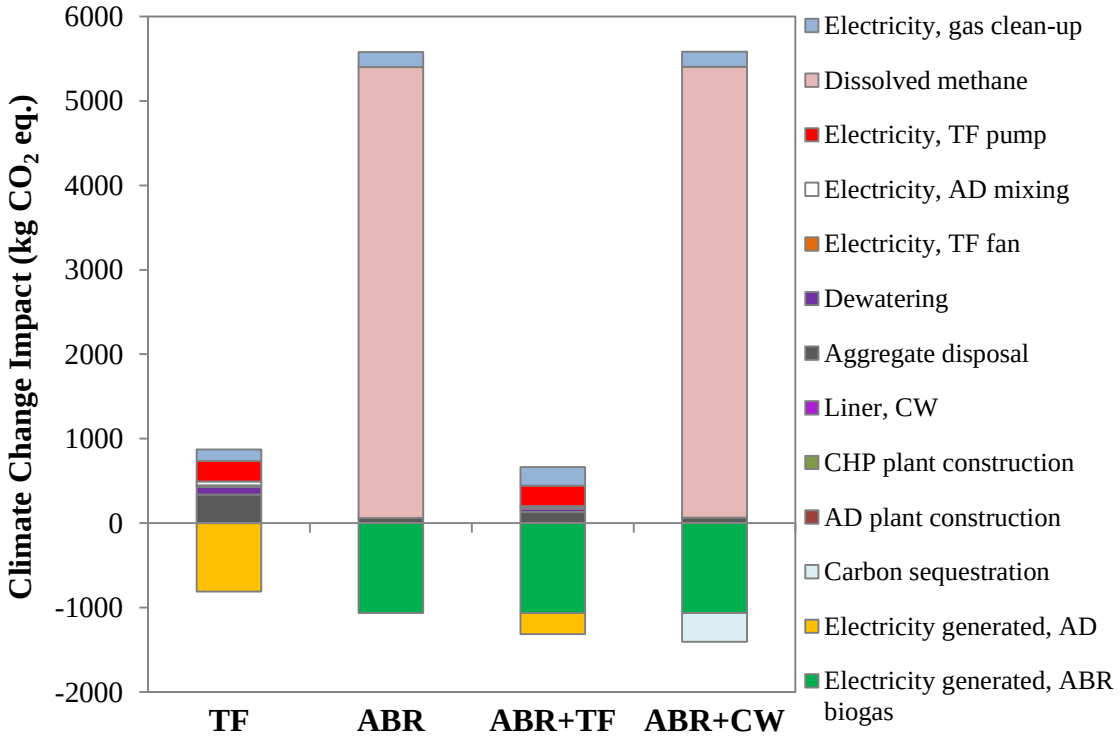


Figure 4.5. Daily impacts on climate change for 15-20°C treatment assuming an aggregate method for disposal of solids.

The impacts for three categories, Resources, Ecosystem Quality, and Human Health were similar, in that beneficial impacts resulted from bio-electricity production, and harmful impacts resulted primarily from consumption of grid electricity (Figure 4.6, Figure 4.7, and Figure 4.8). This is a direct reflection of the electricity grid used in this analysis in which 60 percent of electricity production is generated using coal. Consideration of the type of electricity used in wastewater treatment, therefore, is critical for proper understanding of life cycle environmental impacts. For instance, if the electricity grid consisted primarily of electricity generated from renewables, the

magnitude of impacts resulting from electricity generation and consumption would be reduced. Results presented in Figures 4.6-4.8 also indicate that impacts associated with construction of anaerobic digesters and combined heat and power (CHP) plants are negligible which agrees with the findings of previously conducted LCA studies which concluded that construction impacts are negligible (Renou et al. 2008; Kalbar et al. 2012).

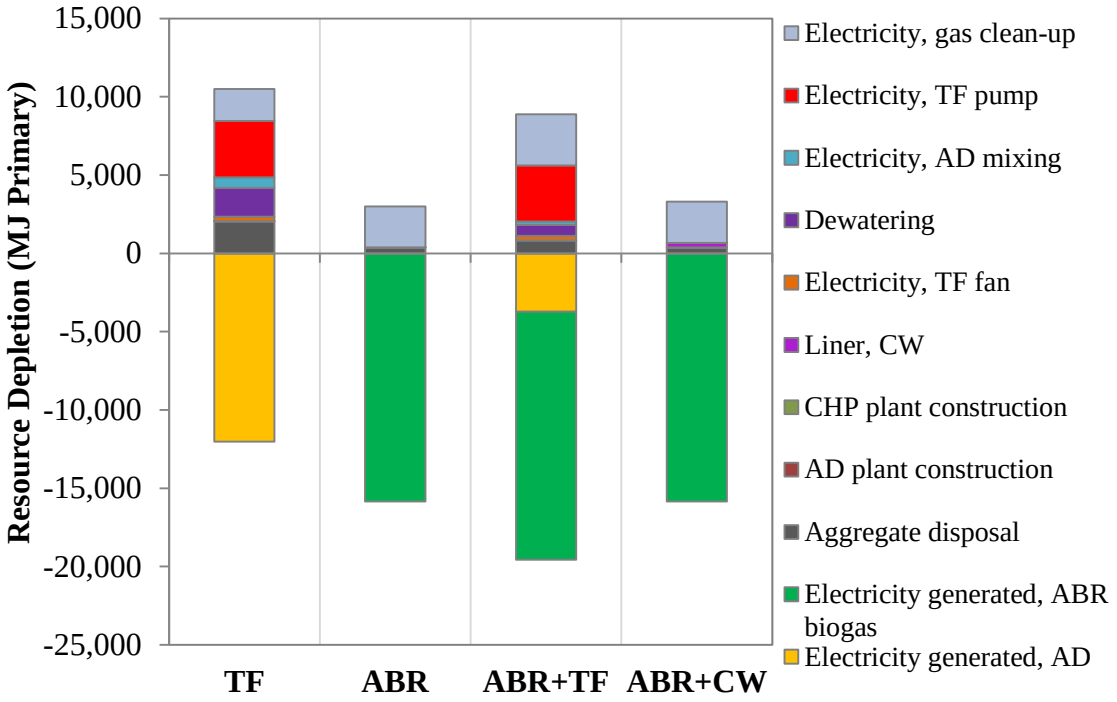


Figure 4.6. Daily impacts on resource depletion for 15-20°C treatment assuming an aggregate method for disposal of solids.

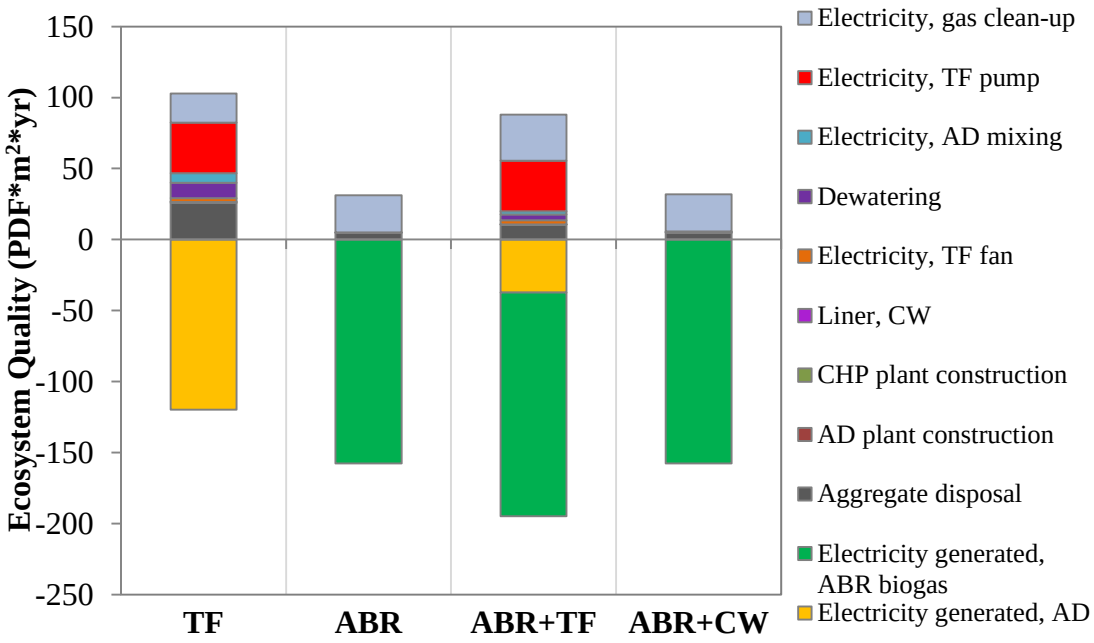


Figure 4.7. Daily impacts on ecosystem quality for 15-20°C treatment assuming an aggregate method for disposal of solids.

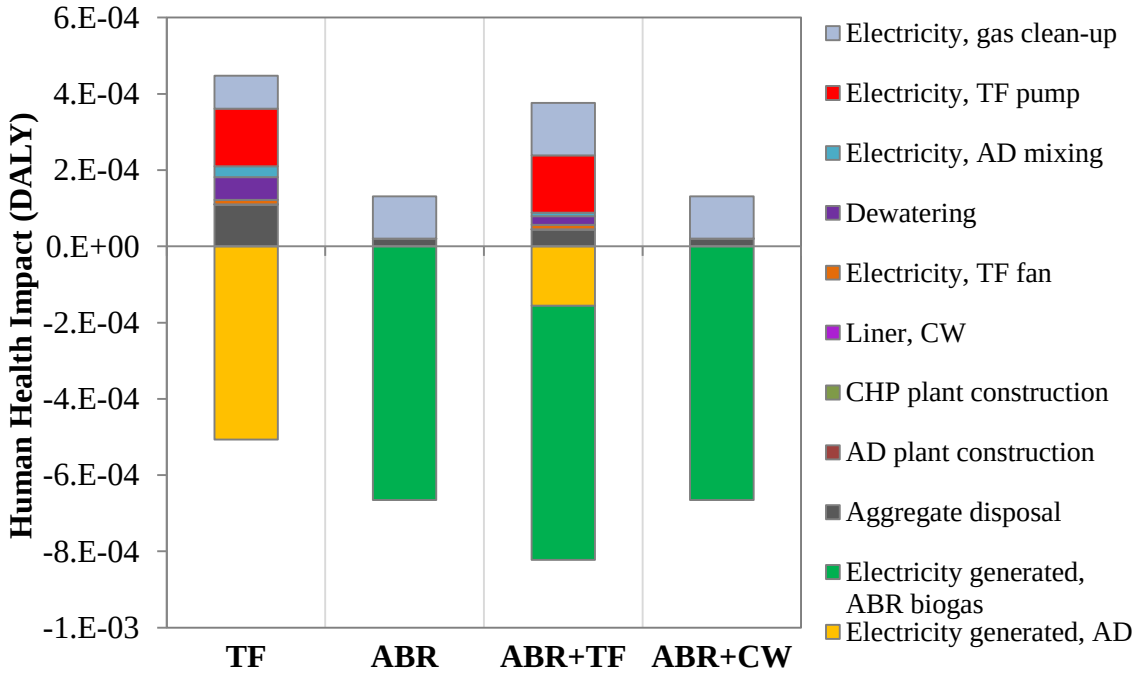


Figure 4.8. Daily impacts on human health for 15-20°C treatment assuming an aggregate method for disposal of solids.

Further assessment of Resource impacts showed that the ABR and ABR + Constructed Wetland models recovered more energy than the other models due to high-energy production from biogas and low energy requirements for operation. The energy production of each of each treatment model was determined by dividing the total energy impacts (MJ primary) determined for each system using IMPACT2002+ by total COD removed and the total wastewater treated to obtain the unit energy values shown in Table 4.16.

Table 4.16. Unit energy impacts for overall treatment systems. Negative values indicate net production of energy.

	Total Impact (MJ Primary/d)	**Total impact (MJ Primary /m^3 WW)**	**Total Impact (MJ Primary/kg COD_r)**
TF	-1520	-0.2	-0.4
ABR	-12829	-1.7	-3.9
ABR + TF	-10681	-1.4	-2.7
ABR + CW	-12530	-1.7	-3.1

On average, wastewater treatment in the United States consumes approximately 1.1 MJ/m^3 wastewater treated (EPA, 2009). Each of the technologies assessed in this study demonstrated the potential to be a net energy producer, with the ABR treatment plant model producing an average of 1.7 MJ primary energy per m^3 of water treated, assuming that bio-electricity produced replaced grid electricity. The energy produced solely via gaseous methane from the ABR was 5.9 MJ/kg COD removed which is approximately 81 percent of the average energy recovery of 7.3 MJ/kg COD removed reported for the ABR, but within the reported range for various anaerobic technologies of 0.5-7.3 MJ/kg COD removed (Shoener et al. 2014). If dissolved methane were captured as biogas instead of released with ABR effluent, the energy production of the ABR would increase to approximately 9.7 MJ/kg COD (4.9 MJ/m^3) removed – a value nearly 4.5 times greater than the magnitude of average energy consumption (1.1 MJ/m^3) of wastewater treatment in the United States (EPA, 2009).

Despite the high energy recovery of the ABR, a comparison of overall normalized impacts, presented as LCA points, (Figure 4.9) on Human Health, Ecosystem Quality, Resources, and Climate Change showed that the harmful impact of climate change dwarfs the beneficial impacts of electricity from biogas production in the ABR and ABR + Constructed Wetland models. The ABR + Trickling Filter model produced a beneficial environmental impact for the four endpoint categories, which indicates that it is the most desirable treatment technology from an environmental perspective, assuming all methane in the ABR effluent was removed in the trickling filter. Furthermore, all assemblies result in a beneficial impacts for Human Health, Ecosystem Quality, and Resource impacts (Figure 4.9) as a result of electricity produced from biogas. However, the impacts of pathogens in ABR effluent are not accounted for in this analysis and may have a harmful impact on Human Health.

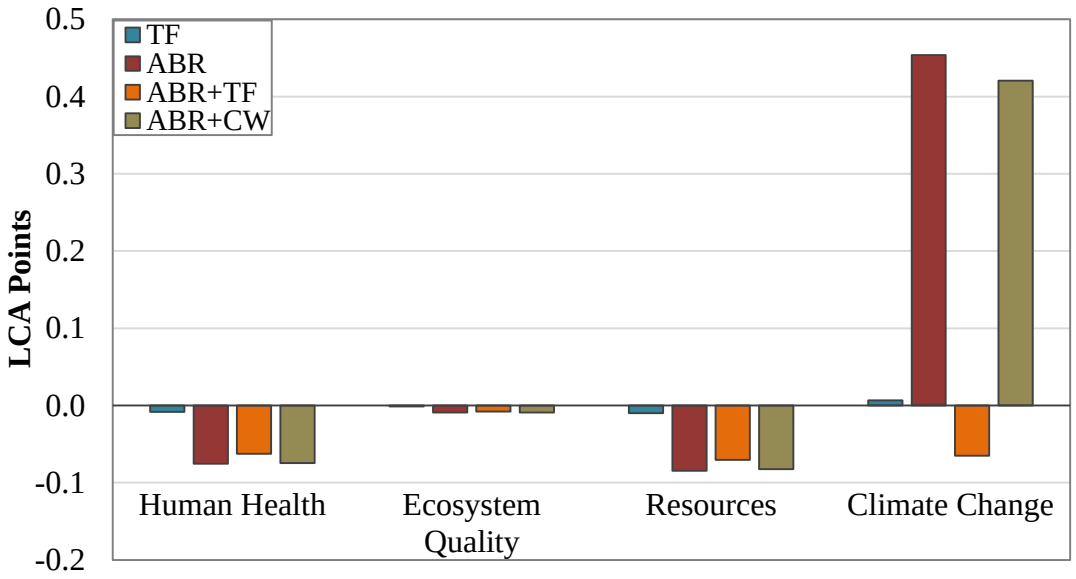


Figure 4.9. Normalized daily impacts for 15-20°C treatment determined using IMPACT 2002+ and IPCC 2013 (Climate Change) assuming an aggregate method for disposal of solids.

In addition to climate change, the poor effluent quality of the ABR negatively impacts eutrophication if distributed to waterways since it produces effluent with concentrations of BOD_5 that exceed the 30 mg/L limit by a factor of approximately 2 to 3 depending on temperature. The eutrophication impacts of the ABR and Trickling Filter are presented for comparison (Figure 4.10). The poor effluent quality of the ABR makes it the least ideal treatment method from a eutrophication perspective with nearly double the impact of the Trickling Filter at 15°C and 20°C (Figure 4.10). However, BOD_5 impacts were reduced by 26 percent when operating temperature increased to 20°C and a previous study concluded that the ABR could achieve nearly 95 percent COD removal at 35°C (Langenhoff and Stuckey, 2000) which would bring the concentration of BOD_5 in the effluent below 30 mg/L. Furthermore, the ABR has been shown to achieve higher organic removal with an increase in HRT (Nasr et al. 2009;Yu and Anderson, 1996) which suggests that the effluent quality of the ABR may not be an issue if operating conditions are adjusted.

Results also indicate that total ABR impacts on eutrophication for 15°C treatment were reduced by approximately 70 percent with addition of a post-treatment step (Figure 4.10). Comparatively, an increase in temperature from 15°C to 20°C reduced the impact of BOD_5 for the Trickling Filter, ABR + Trickling Filter, and ABR + Constructed Wetland by 33, 43, and 33 percent respectively. Thus, increasing in operating temperature and addition of post-treatment technology to remove BOD_5 are effective ways to reduce impacts on eutrophication.

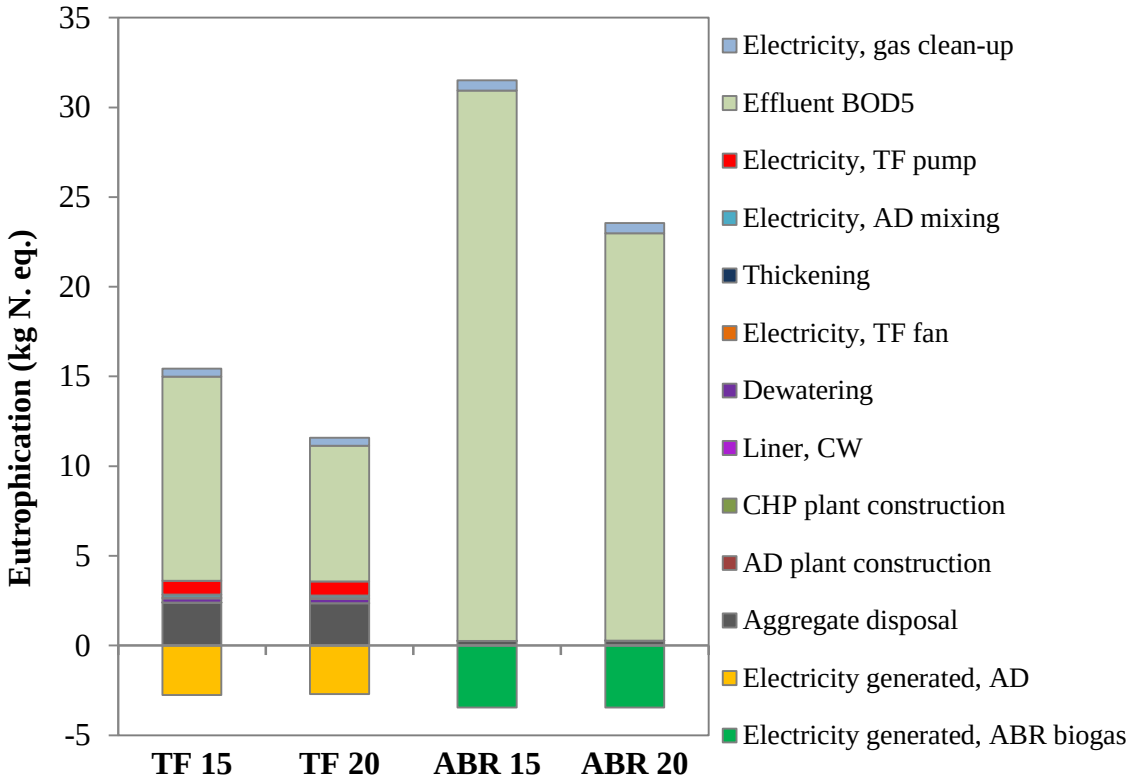


Figure 4.10. Daily impacts on eutrophication for 15 and 20°C treatment assuming an aggregate method for disposal of solids.

4.3.1 Sensitivity Analysis

For the ABR assembly, climate change is most sensitive to dissolved methane concentration, gaseous methane yield, and electrical efficiency of the microturbine (Figure 4.11). Dissolved methane alone has a sensitivity range of approximately ± 0.15 points which, considering the base case value, may change ABR impacts by ± 33 percent. The high sensitivity of dissolved methane observed in this study is in agreement with a previously conducted LCA study which determined that the GWP impact of an anaerobic membrane reactor was sensitive to methane recovery when treating medium and high strength domestic wastewater (Smith, 2014). Unlike the ABR, which has only three parameters that contribute significantly to sensitivity, the

Trickling Filter assembly is highly sensitive to several parameters (e.g. biogas yield, electrical efficiency, methane content of biogas, etc.) with biogas yield, the most sensitive parameter, potentially causing an approximate change of ± 114 percent on climate change impact (Figure 4.12). In addition, the Trickling Filter assembly is sensitive to solids disposal due to the larger mass of solids produced during aerobic treatment compared to anaerobic treatment.

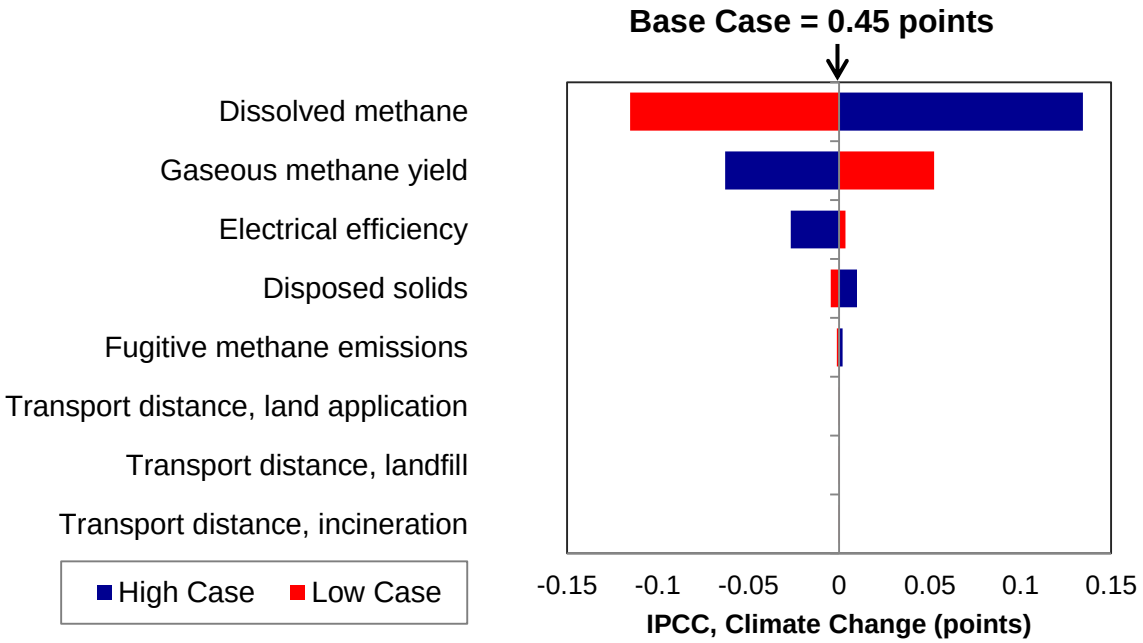


Figure 4.11. Sensitivity analysis for Climate Change impacts of the ABR using IPCC 2013. The x-axis corresponds to a ± change in LCA points relative to the base case (0.45 points).

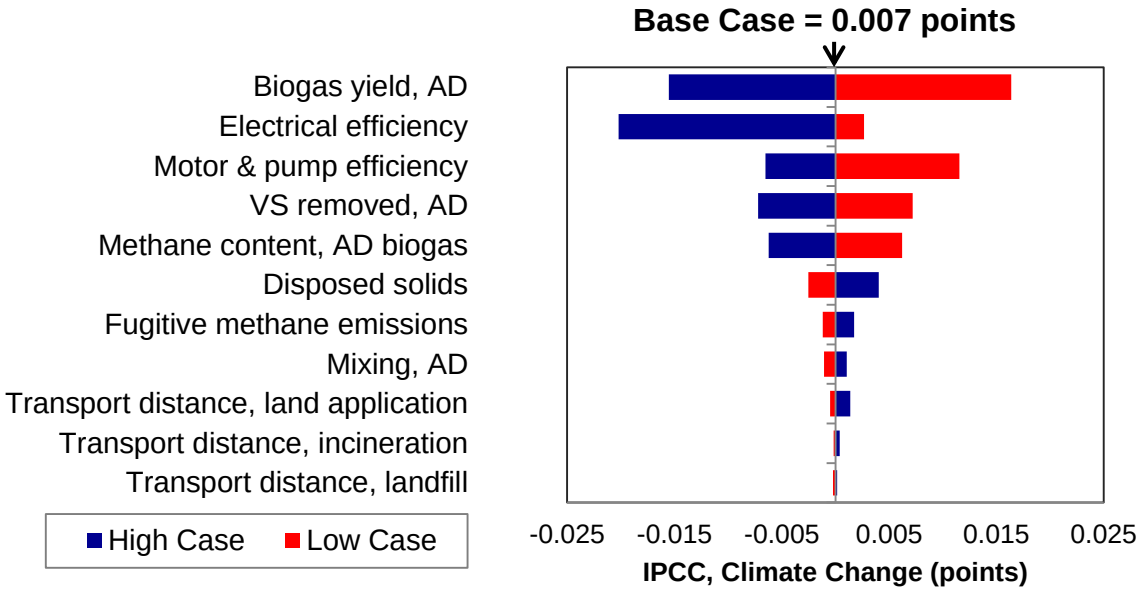


Figure 4.12. Sensitivity analysis for Climate Change impacts of the Trickling Filter using IPCC 2013. The x-axis corresponds to a ± change in LCA points relative to the base case (0.007 points).

For the Resources, Human Health, and Ecosystem Quality impacts using IMPACT 2002+ (Figures 4.13-4.15) gaseous methane yield is the most sensitive parameter in the ABR model, followed by electrical efficiency of the microturbine. This demonstrates that the ABR model is highly sensitive to parameters which influence the quantity of electricity production from biogas. Since the ABR has minimal solids production, variation in transport distances for solids disposal does not greatly effect impacts on Resources, Human Health, or Ecosystem Quality (Figures 4.13–4.15), which agrees with the negligible sensitivity of an LCA model of an anaerobic membrane reactor to variation in transport distances for sludge disposal (Smith, 2014). The sensitivity analysis also shows that the change in impacts for each parameter relative to the base case is similar for Resource Depletion, Human Health, and Ecosystem Quality. Note that dissolved methane has no effect on Resources, Human Health, or Ecosystem Quality impacts.

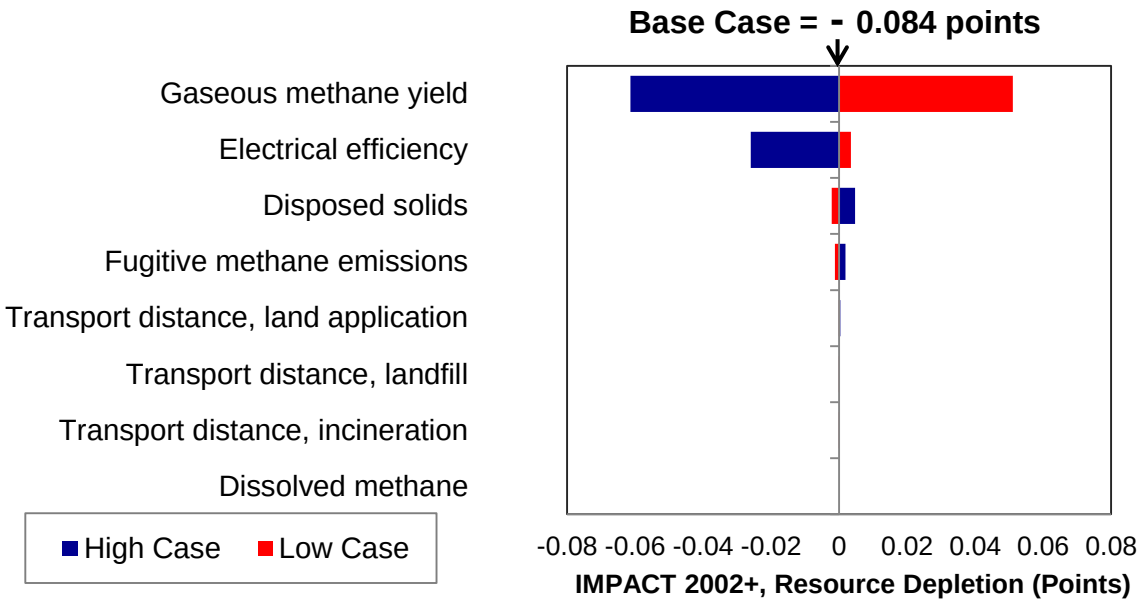


Figure 4.13. Sensitivity analysis for Resource Depletion impacts of the ABR using IMPACT 2002+. The x-axis corresponds to a ± change in LCA points relative to the base case (-0.084 points).

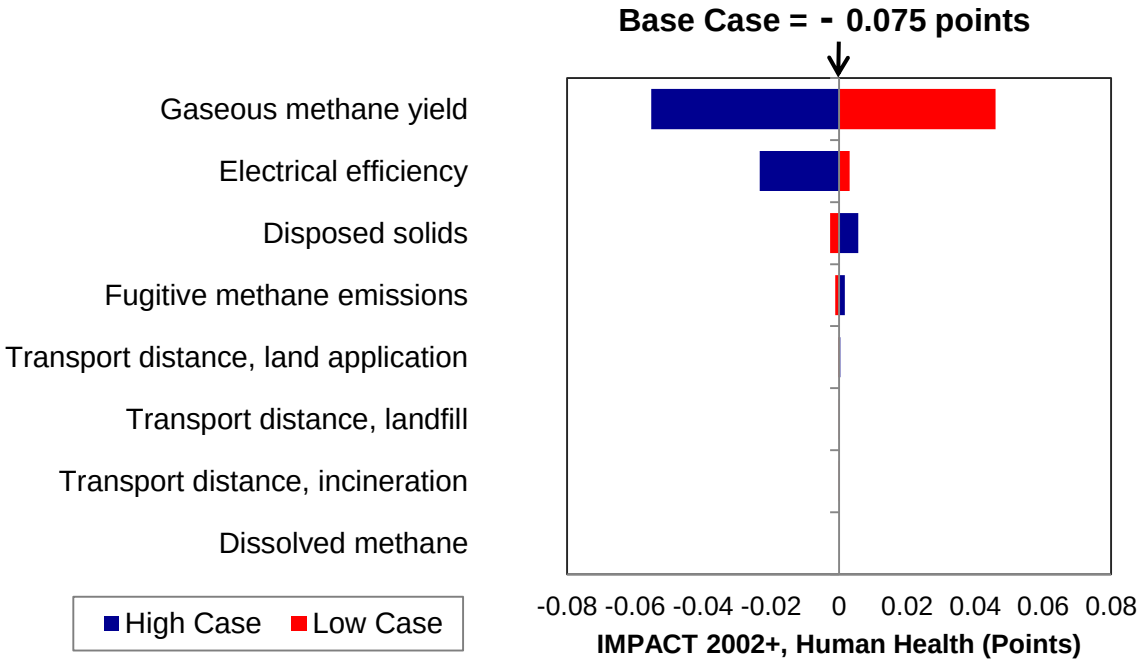


Figure 4.14. Sensitivity analysis for Human Health impacts of the ABR using IMPACT 2002+. The x-axis corresponds to a ± change in LCA points relative to the base case (-0.075 points).

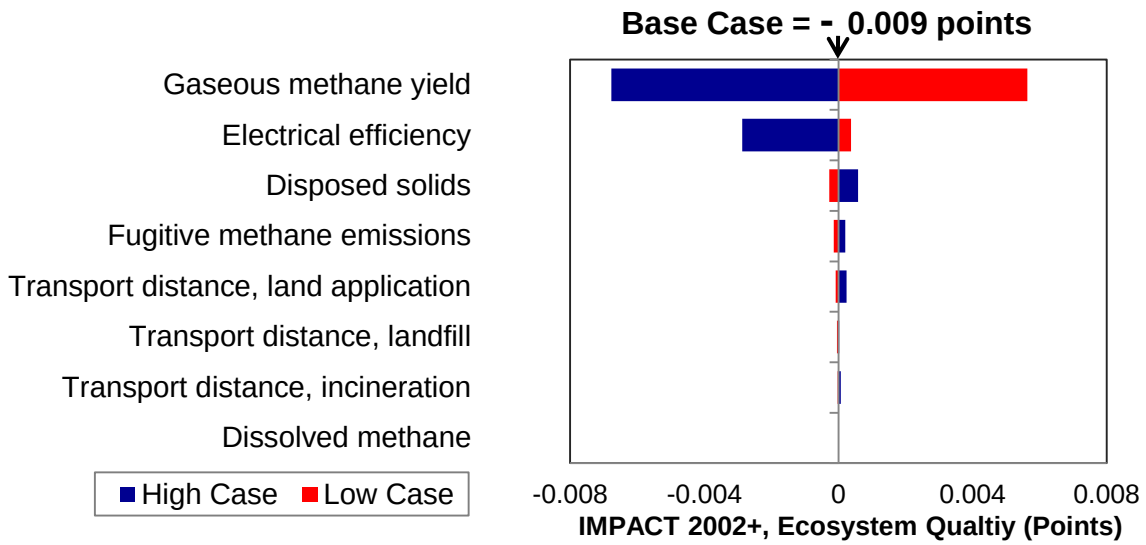


Figure 4.15. Sensitivity analysis for Ecosystem Quality impacts of the ABR using IMPACT 2002+.The x-axis corresponds to a ± change in LCA points relative to the base case (-0.009 points).

As with the ABR, the Trickling Filter showed a similar change in impacts relative to the base case for Resource Depletion, Human Health, and Ecosystem Quality (Figure 4.16, Figure 4.17, and Figure 4.18). Unlike the ABR, the assumed transport distances for solids disposal could potentially alter the base case impact of the Trickling Filter on Ecosystem Quality by a total of +0.0015 points (+150 percent) or -0.0007 points (-68 percent). These results are consistent with a sensitivity analysis of three aerobic systems (conventional activated sludge, high rate activated sludge, and aerobic membrane reactor) which showed that ecotoxicity impacts were highly sensitive to solids transport distances, but that all other TRACI midpoint categories were not when treating medium strength wastewater (Smith, 2014).

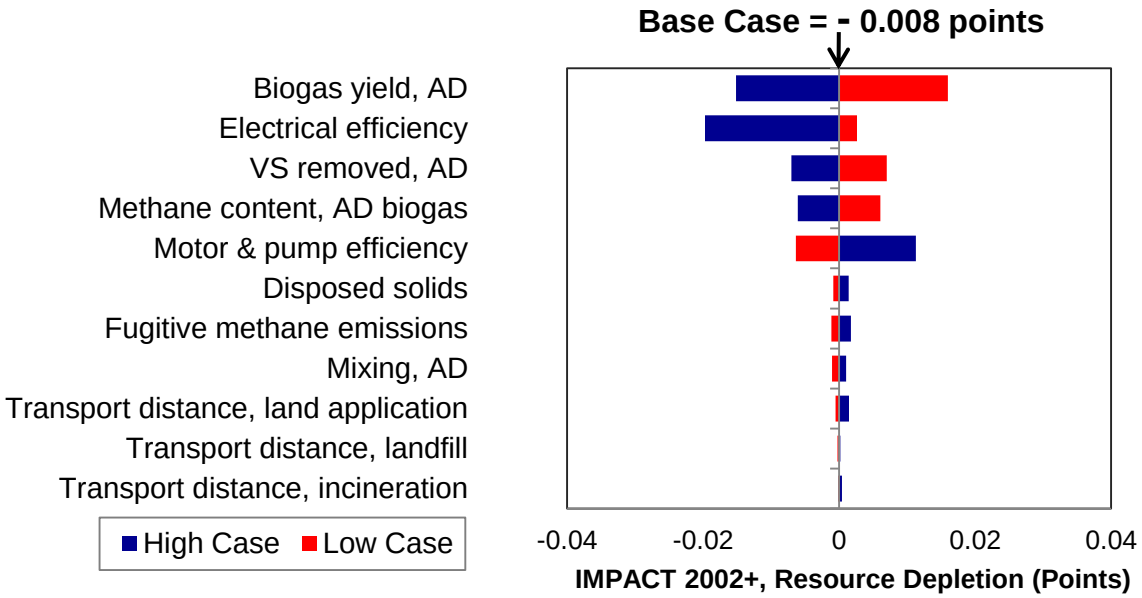


Figure 4.16. Sensitivity analysis for Resource Depletion impacts of the Trickling Filter using IMPACT 2002+.The x-axis corresponds to a ± change in LCA points relative to the base case (-0.008 points).

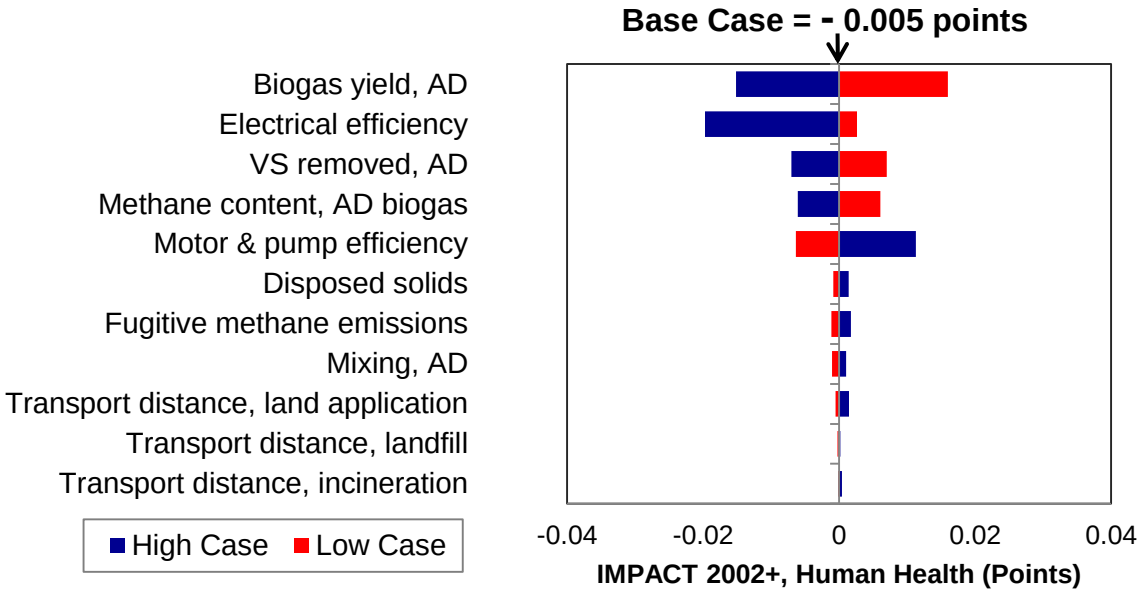


Figure 4.17. Sensitivity analysis for Human Health impacts of the Trickling Filter using IMPACT 2002+. The x-axis corresponds to a ± change in LCA points relative to the base case (-0.005 points).

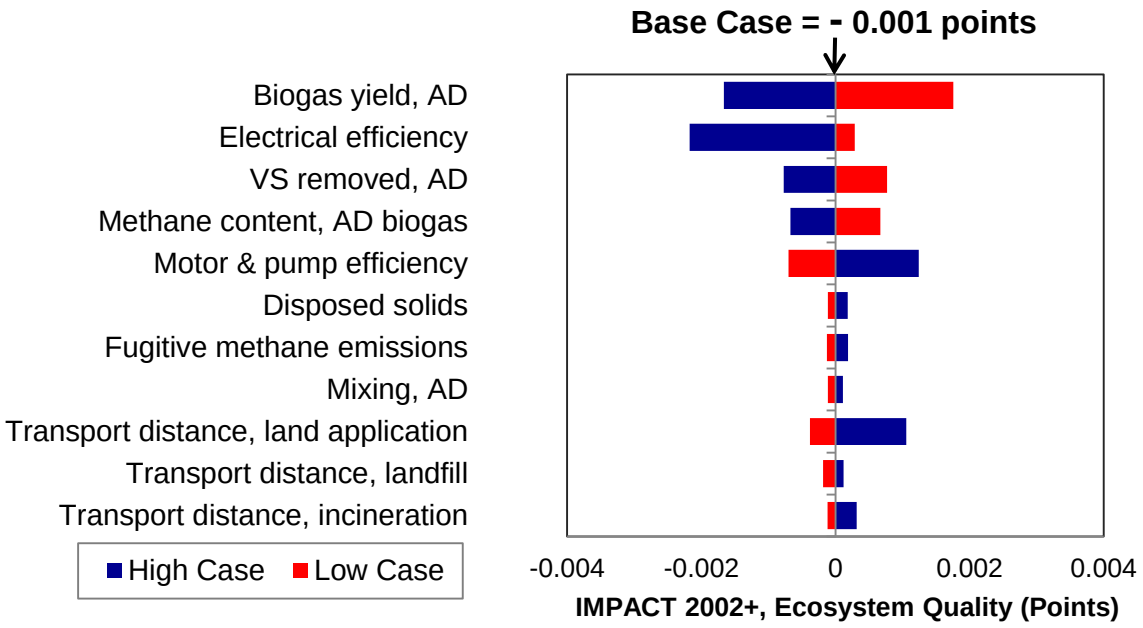


Figure 4.18. Sensitivity analysis for Ecosystem Quality impacts of the Trickling Filter using IMPACT 2002+. The x-axis corresponds to a ± change in LCA points relative to the base case (-0.001 points).

The sensitivity analysis provided insight on the high sensitivity of climate change impacts to dissolved methane concentrations in the ABR model and biogas yield and electrical efficiency in the Trickling Filter model. In addition, transport distance for disposal of solids has a significant effect on impacts only for Ecosystem Quality in the Trickling Filter due to its high production of solids. Finally, the change in impacts relative to the base case for Resources, Ecosystem Quality, and Human Health are similar for both the ABR and Trickling Filter.

4.3.2 Uncertainty Analysis

The ABR and ABR + Constructed Wetland assemblies have larger climate change impacts than the Trickling Filter and ABR + Trickling Filter, even when uncertainty ranges are considered (Figure 4.19). However, the uncertainty ranges for the ABR, ABR + Trickling Filter, and ABR + Constructed Wetland scenarios overlap for Ecosystem Quality, Human Health, and

Resources, suggesting that the variability in overall model impacts may be too great to determine a single treatment system which is most preferential. It is clear, however, that the Trickling Filter is likely to be the least beneficial treatment train in terms of impacts of Resources, Ecosystem Quality, and Human Health.

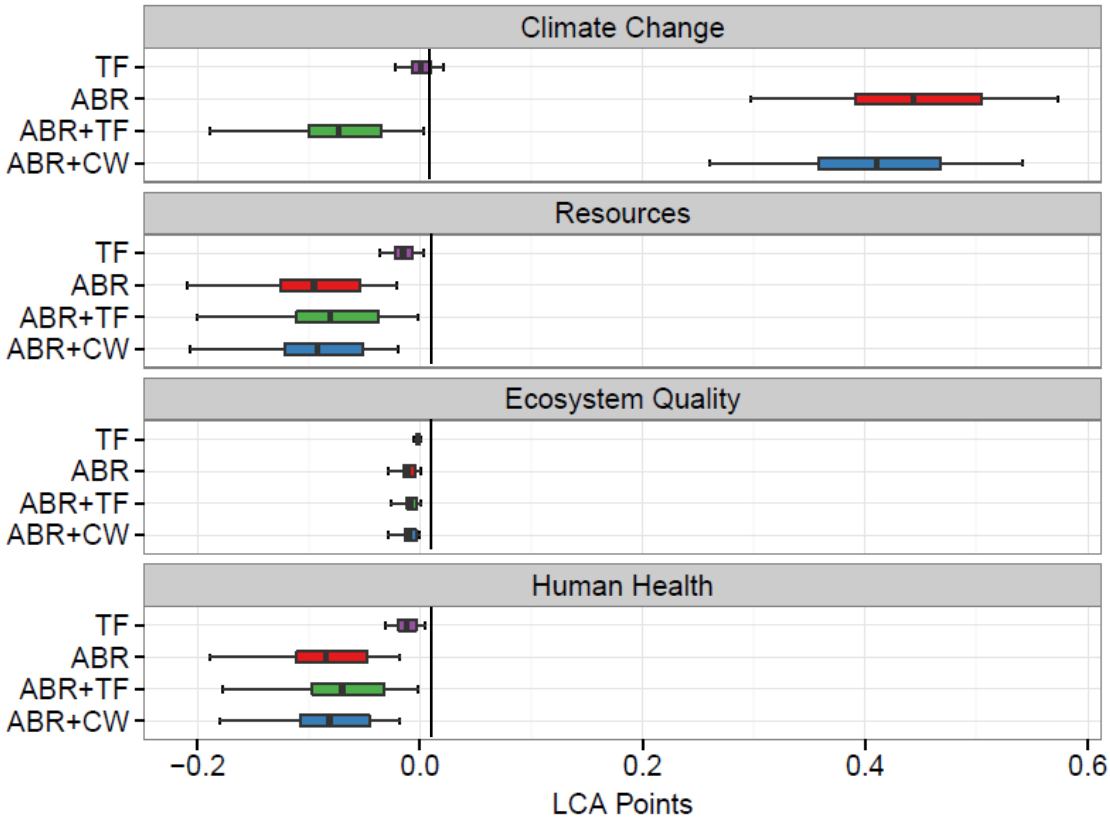


Figure 4.19. Depiction of uncertainty distributions for Climate Change, Resources, Ecosystem Quality, and Human Health determined from 1000 Monte Carlo simulations. Center lines represent median values, edges of colored boxes represent the 25th and 75th percentiles, and limiting bars represent the 5th and 95th percentiles of each distribution, respectively.

The uncertainty for the ABR, ABR + Trickling Filter, and ABR + Constructed Wetland are larger than the uncertainty for the Trickling Filter (Figure 4.20) because the ABR was modeled using variable experimental data, whereas the trickling filter was modeled using design equations coupled with point values for a number of model parameters. Figure 4.20 also demonstrates that the range of uncertainties for Resources and Human Health are nearly four times larger than those for Ecosystem Quality which is a direct reflection of the smaller impact associated with the base case for Ecosystem Quality compared to the base cases for Resources and Human Health. The relative changes for each impact (relative to the median impact), however, are consistent with the sensitivity analysis, in that the percent change from the base case for each model is similar for Resources, Ecosystem Quality, and Human Health.

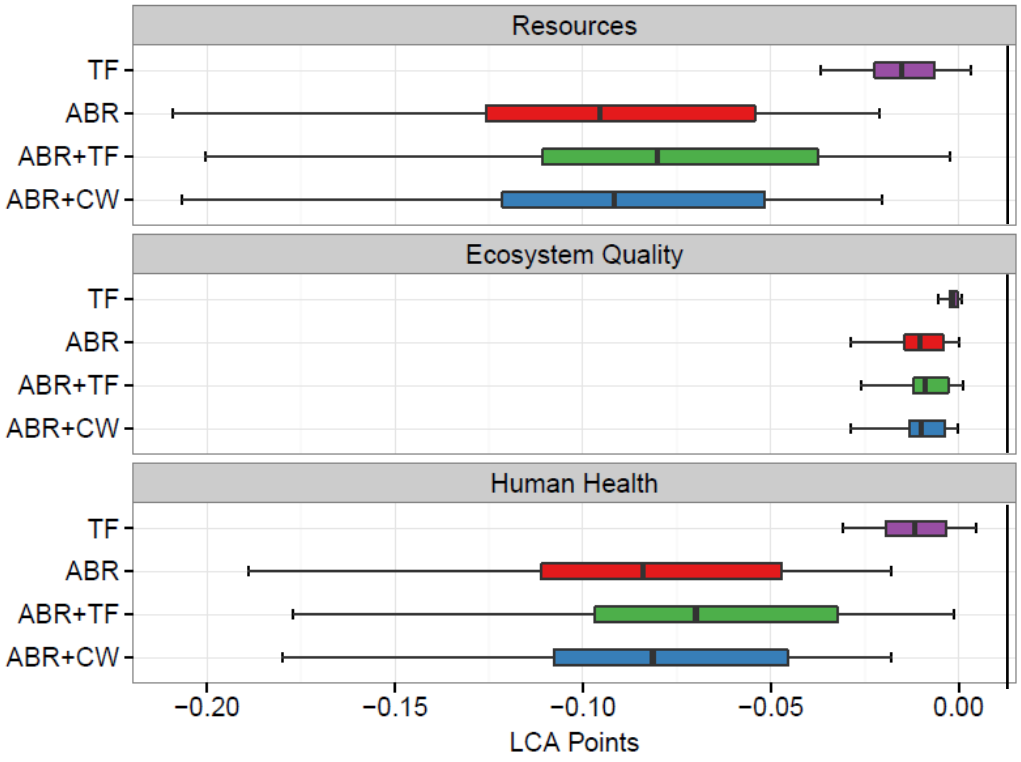


Figure 4.20. Depiction of uncertainty distributions for Resources, Ecosystem Quality, and Human Health determined from 1000 Monte Carlo simulations. Center lines represent median values, edges of colored boxes represent the 25th and 75th percentiles, and limiting bars represent the 5th and 95th percentiles of each distribution, respectively.

4.4 Conclusion

Accounting for dissolved methane in the effluent of anaerobic treatment is imperative for LCA modeling due to both its magnitude and the sensitivity of global warming potential to this parameter. For the base case ABR scenario, dissolved methane contributed 95 percent of the climate change impact. In addition, model results changed by ± 33 percent when the magnitude of dissolved methane was varied to its minimum and maximum values. For Climate Change, the harmful impact of dissolved methane in the effluent was five times greater than the beneficial impact from bio-electricity that replaced grid electricity. At an operating temperature of 15°C and an HRT of 0.55 days, the ABR posed the additional issue of high concentrations of BOD_5 in the effluent which proved to have an impact on eutrophication that was nearly twice as large as the Trickling Filter, although the eutrophication impact decreased by approximately 70 percent with addition of a post-treatment step. However, under different operating conditions, the ABR has been shown to achieve higher BOD removal (Nasr et al. 2009; Yu and Anderson, 1996; Langenhoff and Stuckey, 2000) and may achieve concentrations of BOD_5 that are low enough to eliminate the need for post-treatment.

The ABR + Trickling Filter assembly proved to be the most environmentally preferable technology with net beneficial impacts on Human Health, Ecosystem Quality, Resource Depletion, and Climate Change for the base case. However, results showed that if all the dissolved methane from the ABR were captured in the form of biogas, it could potentially produce 9.7 MJ/kg COD of energy while simultaneously avoiding the release of methane to the atmosphere which would likely make the ABR or ABR + Constructed Wetlands the most optimal treatment systems due to their low energy requirements for operation and minimal production of solids. Even without considering the capture of dissolved methane, the energy production of the

ABR is nearly 2.7 times greater than the average energy consumed by conventional wastewater treatment (EPA, 2009).

A limitation of this study is that it was assumed that all dissolved methane produced in the ABR was destroyed in the trickling filter via biological oxidation, and that all dissolved methane produced in the ABR was released to the atmosphere in the constructed wetland. Those assumptions serve as an approximation of the behavior of post-treatment methods, but may not accurately reflect the dissolved methane released during full-scale operation. We also assumed no nitrification and did not account for human health impacts due to pathogens in the effluents. In addition, ABR treatment did not meet effluent quality standards, and the full-scale model was constructed using laboratory-scale data. The trickling filter, anaerobic digestion, and constructed wetlands processes, on the other hand, were designed using design equations based on first principles and experience with large-scale systems. Additional experimental work using bench-, pilot-, and full-scale reactors should be done to improve the accuracy of the models and to test for variation in biogas production rates and effluent quality. Future research should refine the LCA models to address these limitations.

Future work should focus on the development of methane recovery strategies to lessen the environmental impacts and increase the energy production of anaerobic technologies to expose their potential as net energy producers that are competitive with conventional means of wastewater treatment. Furthermore, work which improves the modeling framework presented in this study and address its limitations should be conducted. Specifically, the model should be refined and expanded to address assumption made regarding dissolved methane release from post-treatment systems and to account for removal of nutrients. In addition, the effect of varied

HRT, organic loading rate, and temperature on the concentration of BOD_5 in ABR effluent should be incorporated into the models.

CHAPTER 5

ENERGY RECOVERY FROM WASTEWATER: LIFE CYCLE ECONOMIC IMPACTS FOR ANAEROBIC BAFFLED REACTOR AND TRICKLING FILTER WITH ANAEROBIC DIGESTION

5.1 Introduction

The United States emits 45 million tons of greenhouse gas emissions each year via wastewater treatment (USEPA, 2013) and electricity requirements alone can cost a 3 MGD wastewater treatment plant an average of $140,000 per year (PADEP, 2011). Note that the average electricity cost was calculated by summing the annual electricity costs for 133 wastewater treatment plants, and dividing that total ($19,000,000) by 133. Consequently, there is a need to research alternative methods of wastewater treatment and their corresponding environmental and economic impacts.

Extensive research has been conducted to explore the potential environmental advantages of implementing alternative wastewater treatment technologies using life cycle assessment (LCA); however, few LCA studies on wastewater treatment have also included an analysis which evaluates their economic feasibility. A study which included an economic analysis of the wastewater treatment technologies considered in the LCA concluded that the most environmentally preferable alternative is not always the most economically preferable (Pant et al. 2011). In addition to ensuring the economic feasibility of recommended technologies, economic

analysis has also played a pivotal role in guiding the future work on anaerobic membrane reactors by pinpointing processes which contribute the greatest economic burden (Smith et al. 2014).

One approach used for cost estimation of wastewater treatment facilities included unit costing which uses a calculated number of "units" (e.g. volume of slab concrete required, kWh of electricity) and unit costs (e.g. slab concrete cost [$/m^3], electricity [$/kwh]) typically obtained from RS Means (Waier & Charest, 2012) or a local engineering consulting firm to determine the total cost of a treatment facility. Unit costing with RS Means supplemented with consultant estimates was previously used to obtain planning level estimates to assess and compare the economics of wastewater conveyance and treatment facilities (Young and Tull, 2003; Lundquist et al. 2010).

A second approach uses parametric cost curves generated from cost data for constructed wastewater treatment plants to calculate costs from a given plant design value (e.g. influent flow rate) (EPA, 1980). Cost curves have also been used to determine the capital and operating costs for single processes in a wastewater treatment plant, such as anaerobic digestion, to estimate costs for a range of treatment capacities and to assess the economy of scale for a given technology (CAEPA, 2008). The CapdetWorks software combines unit costing with parametric cost estimation to generate planning level cost estimates of wastewater treatment plants (US EPA, 1982). CapdetWorks has been successfully implemented to assess and compare the economics of numerous wastewater treatment plant configurations (Tetra Tech MPS, 2003; Zahid, 2007; Koopman et al. 2006) and has been used in a previous study which coupled LCA with an economic assessment (Smith et al. 2014).

In this study, we compare life cycle economic impacts of four assemblies for treatment of 2 MGD of domestic wastewater at 15°C: (1) ABR (2) Trickling Filter (3) ABR + Trickling Filter

and (4) ABR + Constructed Wetland. We calculated capital costs, annual operating costs, present worth, and payback period for each assembly.

5.2 Methods

CapdetWorks 2.5 (Hydromantis ESS, Inc., 2014) was used to calculate capital and operating costs for all unit processes except cogeneration and constructed wetlands. Capital and operating costs were used to calculate the present worth of each assembly. Finally, the payback period for each assembly was calculated to assess economic viability.

The assemblies used for the economic analysis— ABR, Trickling Filter, ABR + Trickling Filter, and ABR + Constructed Wetland— are based on the life cycle inventories described in Chapter 4. All assemblies include preliminary treatment and use an aggregate method for disposal of solids which assumes 66 percent land application, 20 percent incineration, and 14 percent landfilling (EPA, 1999). Note that using a combination of technologies for disposal of sludge incurs extra costs for a single plant, and would not likely occur in practice. The ABR assembly (Figure 4.1) includes ABR treatment, cogeneration, and dewatering (sludge drying bed). The Trickling Filter assembly (Figure 4.2) includes primary clarification, trickling filter treatment, secondary clarification, gravity thickening, anaerobic digestions, cogeneration, and dewatering (belt-filter). The ABR + Trickling Filter assembly (Figure 4.3) includes ABR pre-treatment, trickling filter post-treatment, secondary clarification, gravity thickening, anaerobic digestion, cogeneration, and dewatering (belt-filter). The ABR + Constructed Wetlands assembly is the ABR assembly (Figure 4.1) with constructed wetlands post-treatment (Figure 4.4).

5.2.1 Capital and Operating Costs

CapdetWorks provides planning-level cost estimates using unit costs (Table 5.1) to estimate the costs of process components (e.g. slab concrete, excavation, electricity) and statistically generated cost curves to determine site-specific costs (e.g. site electrical, yard piping, raw waste pumping) (Hydromantis, Inc., 2003). Detailed descriptions of the specific procedures used in the CapdetWorks modeling software can be found in the accompanying EPA manual (US EPA, 1982).

Default unit costs from CapdetWorks were used (Table 5.1) except for electricity cost (EIA, 2015) and land cost (USDA, 2014) which were adjusted from default values to reflect average values for Pennsylvania since those unit costs vary greatly with location.

Several design criteria (Table G.1 in the appendix) were adjusted from default values to tailor the CapdetWorks models to reflect the life cycle inventories described in Chapter 4 (e.g. ABR volume, TF diameter, operating temperature). Design criteria for the influent (e.g. suspended solids, volatile solids, COD, pH) reflect measured values obtained via 15°C ABR operation (see Chapter 3). Because an ABR unit process was not available in CapdetWorks, we modified the UASB unit process to mimic a full-scale ABR (Table G.1 in the appendix). Specifically, the reactor volume, COD removal, and operating temperature were adjusted.

Land application of solids, for which information was also not available in CapdetWorks, was modeled by modifying the landfill unit process (available in CapdetWorks) to include revenue for biosolids as fertilizer. Annual revenue from biosolids used as fertilizer in land application was calculated assuming a purchase price of $35.80 per ton biosolids (Sullivan, Cogger, & Bary, 2007). However, prices for biosolids fertilizer range from $15-$100 per ton

(Girovich, 1996). Costs associated with lime addition were also added to the land application unit process. The default CapdetWorks unit cost of $0.18/lb hydrated lime was used.

Table 5.1. Unit costs used for CapdetWorks modeling

Unit Costs	Value	Unit
Building cost	146	$/ft^2
Excavation	8	$/yd^3
Wall concrete	500	$/yd^3
Slab concrete	350	$/yd^3
Crane rental	200	$/hr
Canopy roof	16	$/ft^2
Electricity	0.11	$/kWh
Handrail	75	$/ft
Land costs	5600	$/acre

Table 5.2 contains the labor rates used for modeling, which were adjusted from default values to reflect the most current national average wage values reported by the Bureau of Labor Statistics (BLS, 2014).

Table 5.2. Labor rates used for CapdetWorks modeling

Labor Rates[a]	Value	Unit
Construction labor rate	17	$/hr
Operator labor rate	23	$/hr
Administration labor rate	26	$/hr
Laboratory labor rate	21	$/hr

[a]Note that reported labor rates are straight-time wages which exclude overtime pay, severance pay, nonproduction bonuses, benefits, and tuition reimbursements (BLS, 2014)

The default costs of chemicals were used with the exception of "polymer" which was adjusted to $1.36 per pound to reflect average market prices of acrylonitrile – the specific polymer chosen for addition during belt filter press dewatering (ICIS, 2008).

The Marshall and Swift Equipment Cost Index and the Engineering News Record (ENR) Construction Cost Index, and Pipe, Valve, and Fitting Cost Indices were used to bring equipment and construction costs specified by CapdetWorks for 2007 to current market prices using the following expression (Turton et al. 2012)

$$C_2 = C_1 \left(\frac{I_2}{I_1}\right) \tag{5.1}$$

where C is purchase cost, I is cost index, subscript 1 refers to base time when cost is known, and subscript 2 refers to time when cost is desired. The current index values used are shown in Table 5.3.

Table 5.3. Cost indices used for CapdetWorks modeling

Cost Indices	Value
Marshall and Swift	1585.7
Engineering News Record	9972
Pipe Cost Index	868.9

Finally, several cost assumptions were specified to account for the indirect costs shown in Table 5.4. The percentage of total capital costs applied for engineering design fees was determined by summing the average values for electrical, site development, mechanical, and structural engineering fees defined in Division 1 of RS Means (Waier & Charest, 2012). The contingency value reflects the percentage associated with project in the "Conceptual stage" and the "Profit and Overhead" value is an average percentage (Waier & Charest, 2012). Default CapdetWorks values were used for all remaining items in Table 5.4.

Table 5.4. Additional cost assumptions used for CapdetWorks modeling

Additional Cost Assumptions	Value (%)
Engineering design fee	20
Miscellaneous	5
Administration /legal	2
Inspection	2
Contingency[a]	20
Technical	2
Profit and overhead	10

[a]The contingency value reflects the reported percent contingency typically assumed for a project estimate in the conceptual stage (Waier & Charest, 2012)

A location factor of 97.2 percent was used to adjust all costs to reflect prices in Harrisburg, PA as of 2013 (Waier & Charest, 2012). Specifically, all operating costs and capital costs determined by CapdetWorks were multiplied by a factor of 0.972. The design lives of structural components, pipes, and pumps were assumed to be 40, 20, and 25 years, respectively (Hydromantis ESS, Inc., 2014).

5.2.1.1 Costs for Cogeneration and Constructed Wetlands

Because CapdetWorks does not include process models for cogeneration and constructed wetlands treatment, their capital and operating costs were calculated individually then added to costs calculated with CapdetWorks.

Subtotal capital costs for cogeneration were determined by assuming a value of $2,689/kW which is considered typical for microturbine capital cost and was used for this analysis (EPA, 2015). The ABR, Trickling Filter, ABR + Trickling Filter and ABR + Constructed Wetland produced 56 kW, 43 kW, 71 kW, and 56 kW, respectively, which are comparable to the reported estimate that each MGD of wastewater in the United States can generate 26 kW of electricity (US EPA, 2013). The subtotal capital cost was adjusted to account for the additional

fees listed in Table 5.4 to calculate a total capital cost for the microturbine. Annual maintenance costs for cogeneration were calculated assuming a typical value of $0.01/kWh of electricity generated by the system (EPA, 2015). Annual electricity generation for each system in kWh/yr was used to calculate the potential electricity savings, assuming an electricity cost of $0.11 per kWh (EIA, 2015). Annual electricity costs for gas clean-up to remove H_2S from biogas were also accounted for by assuming 0.25 kWh/kg VS processed (0.41 kWh/m^3 CH_4 produced) for anaerobic digestion (Sills et al. 2013).

Capital costs for constructed wetlands consist of land purchase, liner, excavation, gravel, planting, and plumbing costs. It was assumed that the 3.8 ha wetland was purchased at $5,600 per acre (USDA, 2014) which totals $52,583. The constructed wetland was lined with a 1 mm thick reinforced polypropylene liner (Sills et al. 2013) which at a cost of $13 per square meter (Beal et al. 2015) totaled $494,000. Note that a plastic liner was included to provide a conservative cost estimate, but may not be necessary if the soil along the bottom of the wetland is clay-rich (Crites & Tchobanoglous, 1998). Capital costs for excavation, gravel, planting, and plumbing were determined using reported unit costs from 1997 (Table 5.5) which were updated to 2015 dollars using Engineering News Record cost indices. The sum of the capital costs is considered a subtotal cost which included the additional fees listed in Table 5.4 to calculate a total capital cost for the constructed wetland. The total cost of the wetland translates to about $65/$m^2$ which is comparable to the $60/$m^2$ capital cost reported by Singh and colleagues for constructed wetlands (Singh et al. 2009). A summary of determined capital costs are shown in Table 5.5.

Table 5.5. Capital costs for constructed wetlands scenario. Unit prices for 1997 were obtained from (EPA, 2000) and were adjusted using Engineering News Record cost indices.

Item	Unit	Unit Price ($, 1997)	Unit Price ($, 2015)	Item Price
Excavation/Compaction	m^3	$2.30	$3.92	$68,069
Gravel (60 cm)	m^3	$20.95	$36	$813,670
Planting Cost	ha	$12,355	$21,046	$79,975
Plumbing	lump sum	$7,500	$12,776	$12,776
Control Structures	lump sum	$7,000	$11,924	$11,924
RPP Liner	m^2	-	$13	$494,000
Land Cost	acre	-	$5,600	$52,584
			Capital cost (subtotal)	$1,532,998
			Capital cost (total)	$2,468,127

Operation and maintenance costs for constructed wetlands have been reported as negligible (Cheema et al. 2014); however, an estimated 10 to 15 hours of operator maintenance are generally required per month (WEF, 1990). The annual operation and maintenance costs for post-treatment with constructed wetlands were calculated by assuming a 15 hour per month operator labor requirement and the assumed operator labor rate of $23/hour (see Table 5.2) which resulted in annual costs of $4,140.

5.2.2 Present Worth Analysis

The present worth of each system was calculated to compare the overall project costs of each system. Present worth was calculated by adding the capital costs determined via CapdetWorks to the series present worth of the uniform annual operating costs, P, determined using Equation 5.2 (Newnan et al. 2014)

$$P = A\left[\frac{(1+i)^n-1}{i(1+i)^n}\right] \quad (5.2)$$

where A is net annual payment ($/yr), i is interest rate (%), and n is the payment period (yrs). A 30-year payment period was used based on values assumed in previously conducted economic analyses of wastewater treatment plants (Lundquist et al. 2010; Zahid, 2007). An interest rate of 3.375 percent was used for analysis since it reflects the interest rate reported by the Water Resources Council for 2015 (NRCS, 2015).

5.2.3 Profitability Analysis

A profitability analysis was conducted to determine the number of years required to achieve a positive cash flow– referred to as the payback period. The profitability analysis utilized operating costs and capital costs determined via CapdetWorks. As with the present worth analysis, a 30-year term was assumed (Lundquist et al. 2010; Zahid, 2007).

The after-tax cash flow (CF) for each year *k* of operation was calculated using Equation 5.3 adapted from (Turton et al. 2012)

$$CF_k = (R - OC - d_k) \cdot (1 - t) + d_k \quad (5.3)$$

where R is annual revenue, OC is annual operating cost, d_k is depreciated capital, and *t* is tax rate. Three sources of "revenue" were considered in the payback period analysis: electricity produced via cogeneration, sale of biosolids for land application, and revenue from sale of wastewater treatment services. However, revenue from biosolids for land application may not be applicable depending on the quality of the biosolids. Electricity generation was assumed to generate $0.11 per kWh (EIA, 2015), land applied solids were assumed to sell for $44.60 per ton ($35.80 per ton

updated to 2015 dollars using ENR cost indices) (Sullivan et al. 2007) although this is an optimistic value since Class B biosolids may not be marketable. In addition, revenue from water treatment services was assumed to be $7 per 1000 gallons ($5 per 1000 gallons updated to 2015 dollars using ENR cost indices) (Sharpe & Swistock, 2004). For the profitability analysis only, a conservative tax rate of 35 percent was used under the assumption that the plant is operated by a private contractor (Beal et al. 2015), but an additional analysis which used a tax rate of zero percent was conducted to provide a best-case estimate for the payback period.

The MACRS method was used to assess the depreciation of treatment plant costs over time. The salvage values of the plants were assumed to be zero, as is customary with the MACRS depreciation method (Newnan et al. 2014). Municipal wastewater treatment plants have an Asset Depreciation Range (ADR) class life of 24 years and are, therefore, a part of the 15-year property class which required use of a 15-year depreciation schedule (Newnan et al. 2014).

The cash flows determined using Equation 5.3 were used in Equation 5.4, which considers the time-value of money, to calculate the discounted cash flow (DCF) for each year *k* of operation

$$DCF_k = \frac{CF_k}{(1+i)^k} \tag{5.4}$$

where *i* is the effective interest, or discount, rate which includes the effect of inflation (Turton et al. 2012). As with the present worth analysis, an interest rate of 3.375 percent was assumed (NRCS, 2015). Land cost (LC) and capital cost (CC) values were obtained from CapdetWorks and the cumulative discounted cash flow (DCF_C) of the project for a given year was determined using Equation 5.5

$$DCF_C = LC + DCC + \sum_{k=1}^{n_k} DCF_k \quad (5.5)$$

where LC is land cost, DCC is discounted capital costs, and n_k is the operating life of the plant in years. The year at which the DCF_C first became positive is the payback period.

5.3 Results & Discussion

5.3.1 Capital and Operating Costs

Capital costs presented in Figure 5.1 show that the highest capital costs are associated with anaerobic digestion which results in 35 percent and 23 percent of total capital costs for the Trickling Filter and ABR + Trickling Filter assemblies, respectively. The capital cost of anaerobic digestion in the Trickling Filter assembly ($2,570,000) is nearly 64 percent of the total capital costs for the ABR assembly, highlighting the economic advantage of mainstream anaerobic treatment with low solids production. The capital cost for anaerobic digestion in the Trickling Filter assembly ($2,570,000) is significantly larger than the expected capital cost ($1,300,000[c]) determined using cost curves (CAEPA, 2008) generated from anaerobic digester data for 16 facilities (Tsilemou & Panagiotakopoulos, 2006). A possible explanation for the discrepancy is economy of scale since the digester data used to generate the cost curve represents treatment of a larger quantity of solids (2,500-100,000 tons per year) (Tsilemou & Panagiotakopoulos, 2006) than the digester in the Trickling Filter assembly (402 tons per year).

The capital cost of microturbines for cogeneration, however, are minor ranging from approximately $186,000 (Trickling Filter) to $307,000 (ABR + Trickling Filter) which

[c] Converted to 2015 dollars using ENR cost indices

correspond to only 2.5 percent and 4.0 percent of total plant capital costs, respectively. The microturbine capital costs determined in this analysis are comparable to the $275,000-$303,000 range reported for existing 60 kW microturbines which utilize anaerobic digester biogas (ERG & RDC, 2011) and the $303,000 capital cost reported for construction of two 30 kW microturbines implemented at a 2 MGD wastewater treatment facility (Eaton & Jutras, 2005).

Post-treatment of ABR effluent with constructed wetlands has lower capital costs than post-treatment with a trickling filter due to the costs associated with processes required to treat the solids produced by the trickling filter (i.e. anaerobic digestion, gravity thickening, belt-filter press). Specifically, capital costs for trickling filter post-treatment and the solids disposal processes it requires are nearly $2,051,000 greater than those of the constructed wetland. If no liner was required for the wetland, the capital costs for the constructed wetland would be reduced by $494,000, which would provide even greater savings compared to trickling filter post-treatment.

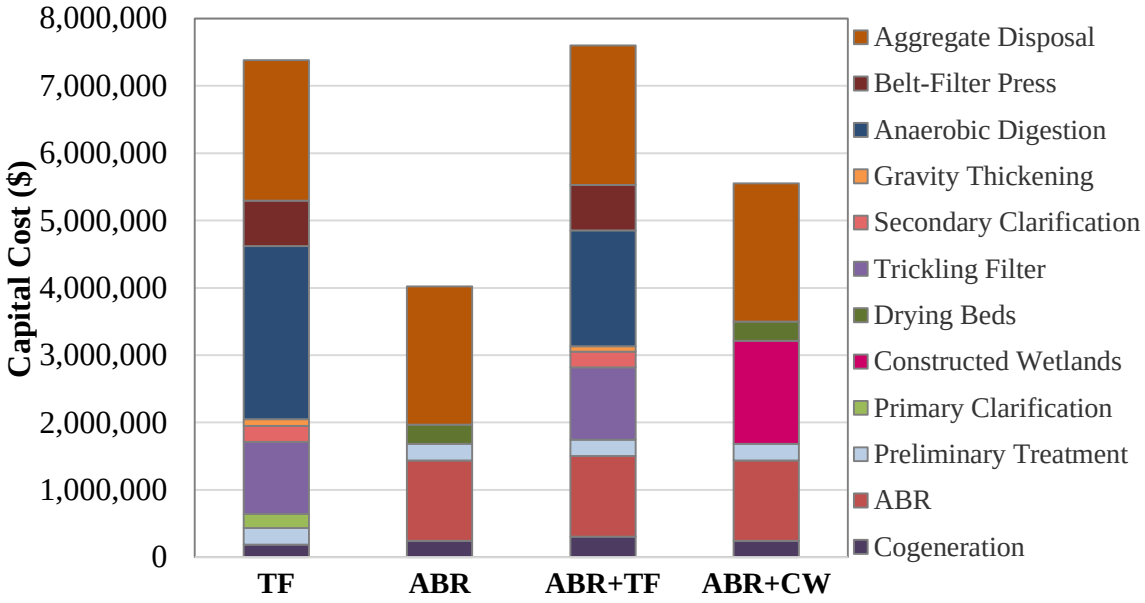


Figure 5.1. Capital costs for 2 MGD treatment with and an aggregate method for disposal of solids (66% land application, 14% landfill, 20% incineration).

The ABR and ABR + Constructed Wetland assemblies incur net operating costs that are approximately 57 percent of the operating cost for the Trickling Filter assembly (Figure 5.2), which is largely due to the minimal sludge production of anaerobic treatment. The lower solids production of the ABR compared to the Trickling Filter results in an annual savings of over $36,000 on solids disposal. The annual cost of sludge disposal using an aggregate disposal method for the Trickling Filter totals $117,000 ($289/dry ton) which is comparable to the annual costs of $256/dry ton[d], $335/dry ton[b], and $367/dry ton[b] reported for land application, landfilling, and incineration of solids produced from 1-5 MGD treatment plants in Pennsylvania (Elliot et al. 2007). The cost of solids disposal for the Trickling Filter is also consistent with the average costs of $431[a] per dry ton for landfilling and $238[a] per dry ton for land application reported for solids disposal in Virginia (NBP, 2005).

[d] Converted to 2015 dollars using ENR cost indices

Cogeneration incurs annual savings from electricity production ranging from approximately $28,000 (Trickling Filter) to $46,600 (ABR + Trickling Filter). The annual savings are comparable to the $43,000 in annual savings reported for a 2 MGD facility using two 30 kW microturbines for electricity generation (Eaton & Jutras, 2005). The ABR generates approximately $9,000 worth of electricity more than the Trickling Filter via cogeneration due to its higher rate of biogas production. Although anaerobic digestion coupled with cogeneration reportedly lacks economic viability for treatment less than 5 MGD (Smith et al. 2014; Naik-Dhungel, 2010), it has been shown that microturbines have been implemented successfully for flows less than 5 MGD (Eaton & Jutras, 2005; ERG and RDC, 2011). The low capital costs and significant annual savings from cogeneration determined in this research further indicate that microturbines may be economically viable for 2 MGD treatment – particularly for anaerobic treatment facilities which avoid the high capital cost of digesters.

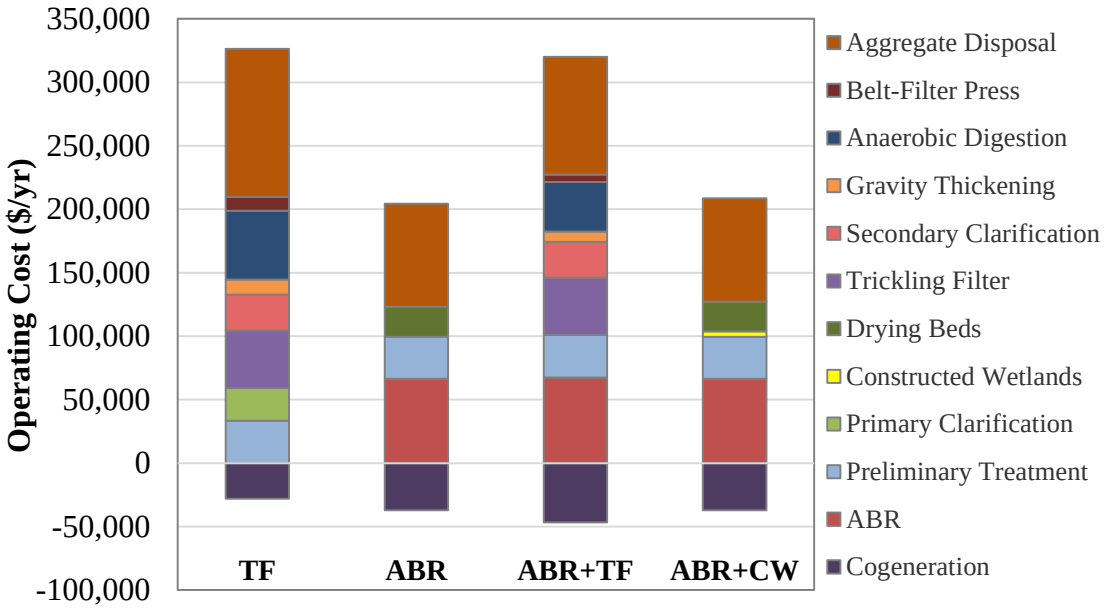


Figure 5.2. Annual operating costs for 2 MGD treatment with and an aggregate method for disposal of solids (66% land application, 14% landfill, 20% incineration).

5.3.2 Present Worth Analysis

The operating and construction costs were used to determine the present worth of each assembly (Figure 5.3) which allowed for direct comparison of overall costs. The ABR is the least expensive technology at a present worth of $16,190,000 which is approximately 36 percent lower than the Trickling Filter, 32 percent lower than the ABR + Trickling Filter, and 14 percent lower than the ABR + Constructed Wetland. The Trickling Filter Results also indicate that the present worth of capital costs alone for the Trickling Filter and ABR + Trickling Filter exceed the total present worth (capital and operating) for the ABR.

In terms of present worth, the capital costs for all assemblies were approximately three times greater than the operating costs which reflect the low energy requirements of the technologies assessed. The results are in agreement with Smith et al. which found that the present worth of capital costs for conventional activated sludge with anaerobic digestion and an anaerobic membrane reactor were approximately 2.0 and 2.5 times greater than the present worth of operating costs when land application was used for disposal of solids although values varied when incineration or landfill was used in lieu of land application (Smith et al. 2014).

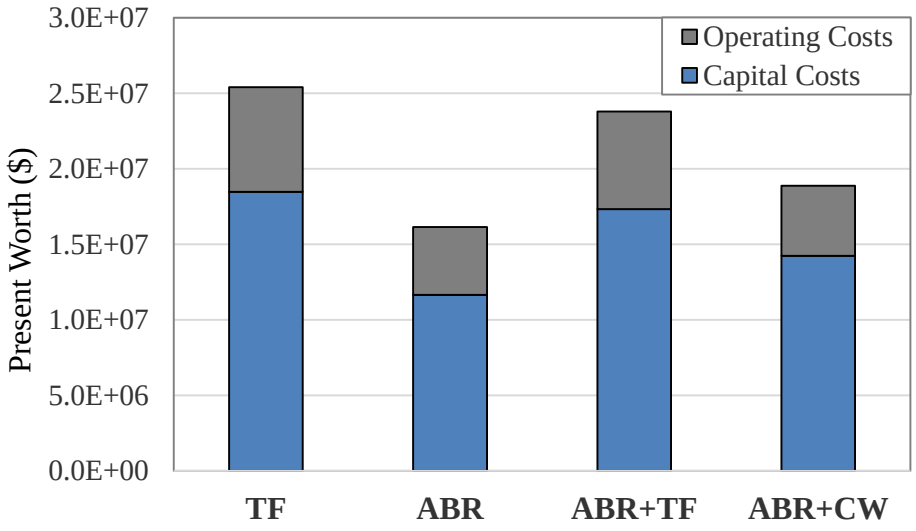


Figure 5.3. Present worth for 2 MGD treatment with and an aggregate method for disposal of solids (66% land application, 14% landfill, 20% incineration) at a discount rate of 3.375%.

5.3.3 Profitability Analysis

A final analysis was conducted which assessed the economic feasibility of the assemblies by determining their payback periods at a 3.375 percent interest rate and a 35 percent tax rate. The tax rate was applied to profits assuming the wastewater treatment plants were operated by a private contractor. The resulting payback periods for the ABR, Trickling Filter, ABR + Trickling Filter, and ABR + Constructed Wetland were 6, 8, 8, and 7 years, respectively. Assuming a zero percent tax rate reduced the payback period for the ABR + Trickling Filter by two years and all other assemblies by one year. Since each of these payback periods is less than the 30 year plant life, each assembly is economically feasible – although the ABR is the most preferable from an economic perspective since it will break even on the cost of investment in just 6 years.

5.4 Conclusion

All treatment systems evaluated in this study are economically viable, although the ABR is the most economically advantageous having a present worth of $16,190,000 and a payback period of 6 years. However, if the ABR were to require a post-treatment process, implementing constructed wetlands in lieu of trickling filter combined with anaerobic digestion would result in savings of approximately $2,051,000 in capital costs and $101,700 per year in operating costs.

The most costly processes are anaerobic digestion which contributes 35 percent and 23 percent of total capital costs in the Trickling Filter and ABR + Trickling Filter assemblies, respectively. Cogeneration is relatively inexpensive (<5 percent of total capital costs) to implement in all scenarios due to its low capital costs ($186,000-$307,000) and annual savings in operating costs ($28,000-$47,000). Therefore, anaerobic treatment systems, like the ABR, which eliminate the need for anaerobic digesters can significantly reduce capital costs while simultaneously reaping the benefit of electricity production from biogas.

The results of this study are limited in that the determined costs reflect planning-level estimates which have and uncertainty of approximately ±15 percent for capital costs and ±10 percent for operating costs (US EPA, 1982). Furthermore, the costs presented in this study may not accurately reflect implementation in alternate locations since costs vary significantly with geographic location.

Future work should conduct a more detailed economic analysis which customizes the assemblies presented in this study by including location-specific unit costs, labor costs, and land costs to improve the accuracy of cost estimates. In addition, more work should be done to evaluate the sensitivity and uncertainty of the costs of the assemblies due to variability in operating conditions (e.g. influent flow rate, temperature, influent organics concentration,

electrical efficiency) to obtain a more comprehensive understanding of results presented in this study.

CHAPTER 6

CONCLUSION

6.1 Conclusion

This research coupled environmental and economic analysis to evaluate the life cycle environmental impacts and economic viability of wastewater treatment technologies. A unique aspect of this research was the use of measured data acquired during treatment of municipal wastewater with a laboratory-scale ABR as a basis for full-scale modeling of ABR treatment. Additionally, the life cycle assessment (LCA) model constructed as part of this research accounted for the effects of dissolved methane present in anaerobic effluent, electricity generation from the methane-rich biogas produced via anaerobic digestion, and included a comprehensive analysis of uncertainty.

All evaluated technologies—ABR, Trickling Filter (with anaerobic digestion used to treat solids), ABR + Trickling Filter, and ABR + Constructed Wetland— had payback periods of less than 30 years indicating that they were economically viable, but the ABR had the shortest payback period and lowest capital and operating costs. The low-energy requirements of each system coupled with bio-electricity generation allowed all assemblies to achieve net beneficial impacts in terms of human health, resource depletion, and ecosystem quality. Uncertainty analysis, however, showed that these beneficial impacts may be in question when uncertainty of parameter values are accounted for. The ABR, however, has a life cycle impact on climate change nearly five times greater than the beneficial impact from electricity production due to dissolved methane released with anaerobic effluent. These results reflect the average dissolved

methane concentration of 28 mg/L and average biogas production of 0.17 m^3/kg COD removed measured for the ABR at 15-20°C treatment. This indicates that, from an environmental perspective, the benefits of biogas production from anaerobic treatment are diminished by the high GWP of methane that volatilizes from the effluent if left unchecked.

The four treatment scenarios benefited, in terms of economic and environmental performances, from cogeneration due to electricity production, which indicates that implementation of combined heat and power is advantageous for 2 MGD treatment. In fact, the energy produced by the ABR was approximately 2.7 times greater than the average energy consumed by conventional wastewater treatment (EPA, 2009). In addition, cogeneration provides a significant benefit in the ABR scenario, resulting in nearly $37,000 worth of electricity annually.

Overall, the study demonstrated that no single technology was optimal from both an environmental and economic perspective, since the ABR and ABR + Constructed Wetland had the lowest present worth, whereas the Trickling Filter and ABR + Trickling Filter had the lowest environmental impact (Figure 6.1). These results validate the importance of considering multiple facets of sustainability.

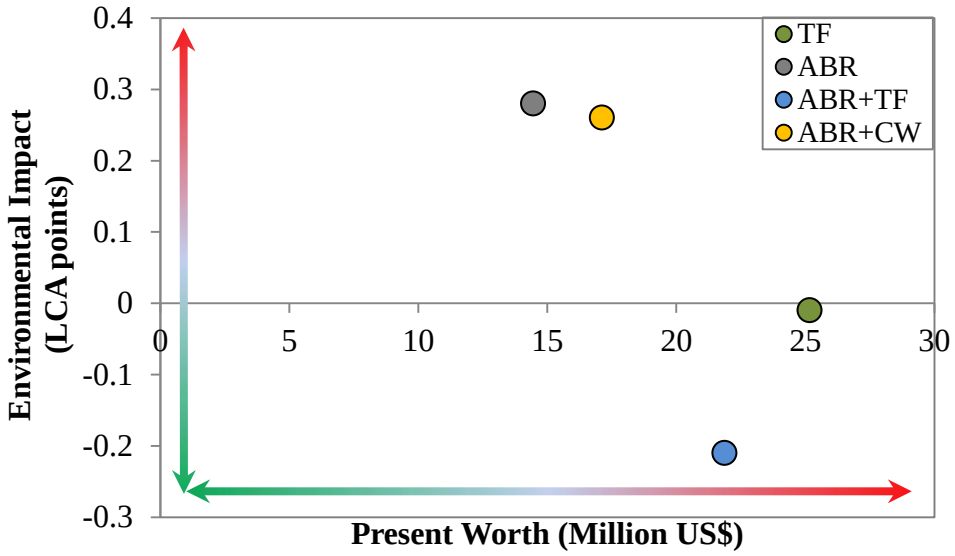


Figure 6.1. Total environmental impact versus present worth for treatment assemblies. Environmental impacts were determined using IMPACT2002+ (human health, ecosystem quality, resources) and IPCC 2013, 100 year (climate change) and were normalized to LCA points for comparison. Present worth represents the sum of the present worth of operating and capital costs assuming an interest rate of 3.375 % and 30 year payback period.

This analysis accentuated the importance of accounting for dissolved methane in LCA of anaerobic treatment systems since its inclusion dictates which treatments generate the lowest environmental impact. When dissolved methane is included, the ABR + Trickling Filter assembly yields the lowest environmental impact. However, if dissolved methane impacts were mitigated or eliminated, the ABR and ABR + Constructed Wetlands would likely be the most preferable technologies from both an environmental and economic perspective.

6.2 Future Work

Future work should focus on the extraction of dissolved methane into the gaseous phase prior to discharge of treated effluent, since results indicated that could increase energy production

from the ABR to approximately 9.7 MJ/kg COD and reduce its climate change impact by nearly 95 percent. In addition, work should be done to quantify the uncertainty of the economic models and to explore varying operating conditions. Specifically, variability in influent and effluent characteristics, solids production, hydraulic retention time, and location should be incorporated into models to fully understand the range of possible economic and environmental impacts of each treatment system. LCA modeling and impact assessment methods should be improved to account for effects of pathogens in treated effluents on ecosystem quality and, especially, human health. Finally, the social impacts associated with each wastewater treatment technology should be evaluated to obtain a more comprehensive understanding of the sustainability of each system – particularly for regions which are poorly developed and do not have reliable access to electricity.

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Appendix A: Correlation Analysis using R

Figures A1-A4 show the results of the correlation analysis of ABR operating parameters to treatment performance for synthetic and municipal wastewater.

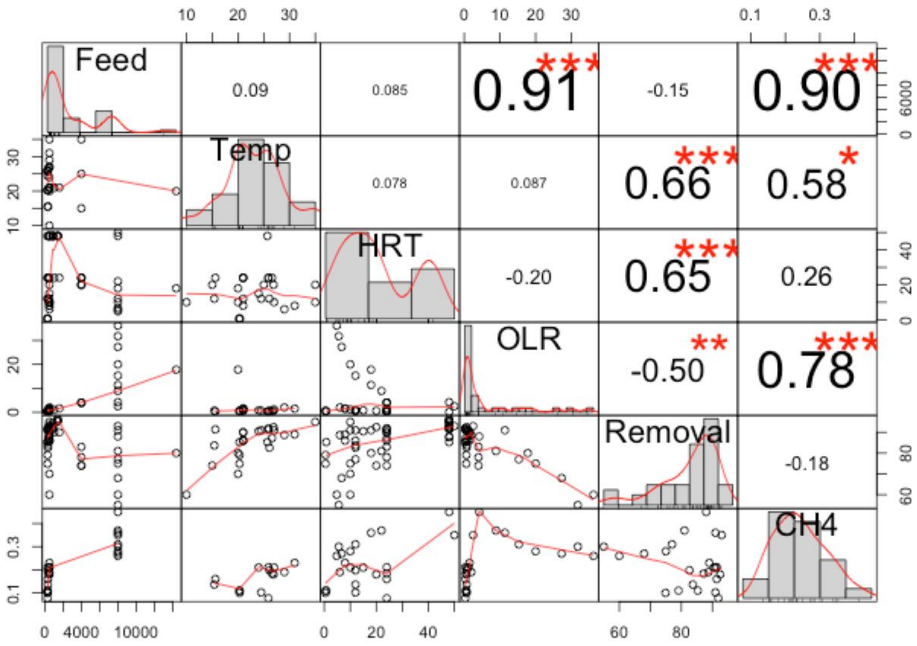

Figure A.1 Correlation analysis results for synthetic wastewater using the Spearman method.

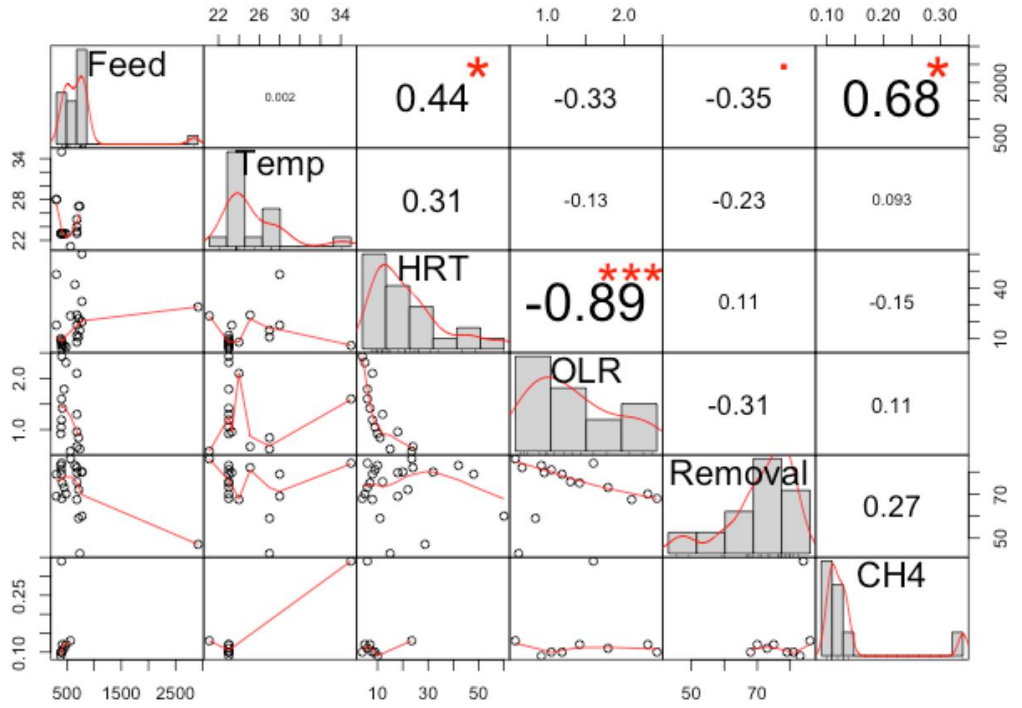

Figure A.2. Correlation analysis results for municipal wastewater using the Spearman method.

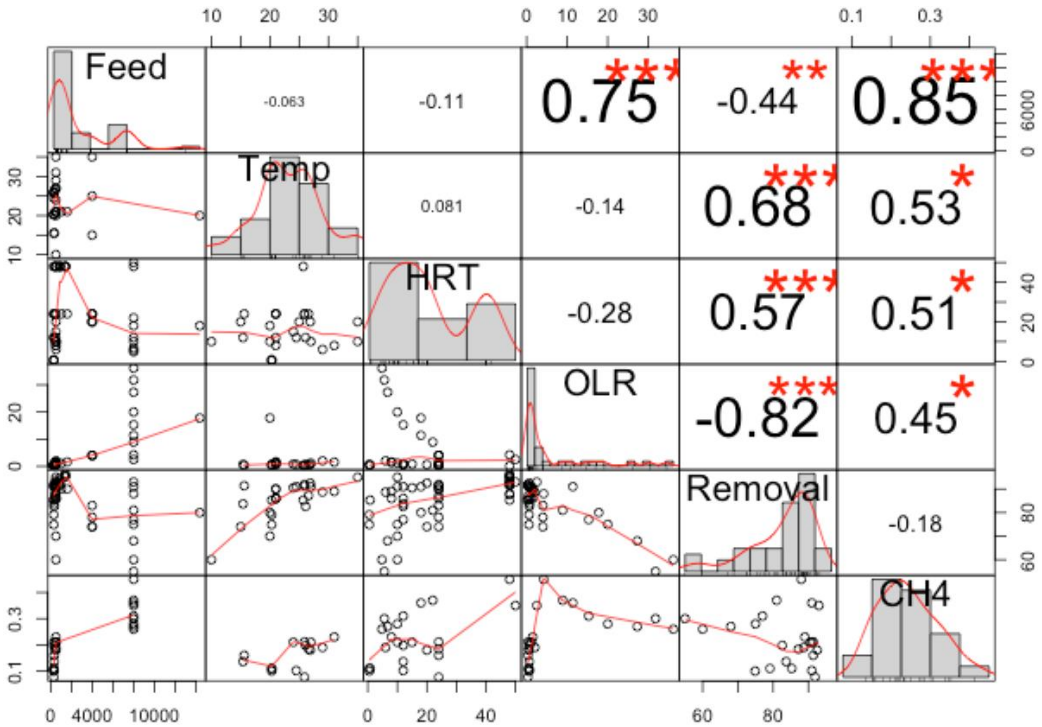

Figure A.3. Correlation analysis results for synthetic wastewater using the Pearson method.

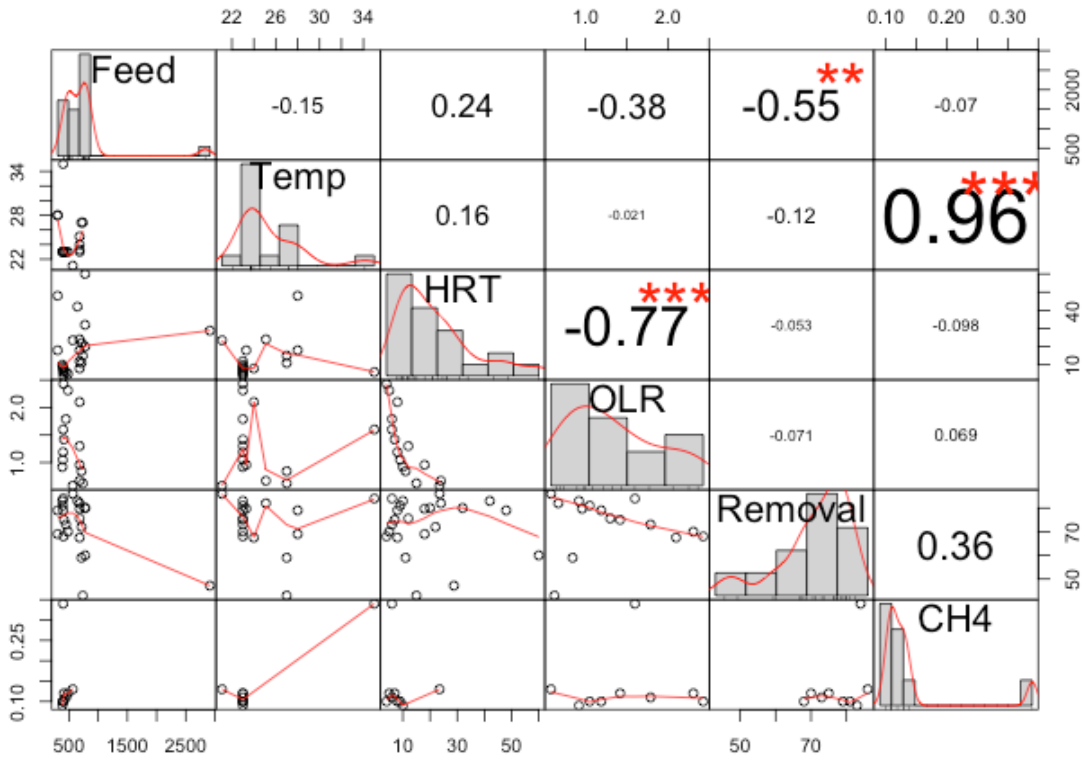

Figure A.4. Correlation analysis results for municipal wastewater using the Pearson method.

Appendix B: Experimental Data Collection

Table B.1 describes the testing schedule used during operation of the ABR.

Table B.1. Anaerobic baffled reactor (ABR) performance testing schedule.

Parameter	Days measured per week			
	Feed	**Chambers - liquid**	**Chambers - biogas**	**Effluent**
Dissolved methane	-	-	-	7
TCOD	7	-	-	7
SCOD	7	-	-	7
TSS	3	-	-	3
VSS	3	-	-	3
TS	-	2 (from Ch1 only)	-	-
VS	-	2 (from Ch1 only)	-	-
pH	3 to 4	3	-	-
VA/PA ratio	3 to 4	3	-	-
Alkalinity addition	3 to 4	-	-	-
Gas production	-	-	7	-
Biogas composition	-	-	7	-
Flow rate	-	7	-	-
VFA	-	3	-	-

Figure B.1 shows the standard curve used to quantify ABR effluent dissolved methane concentration.

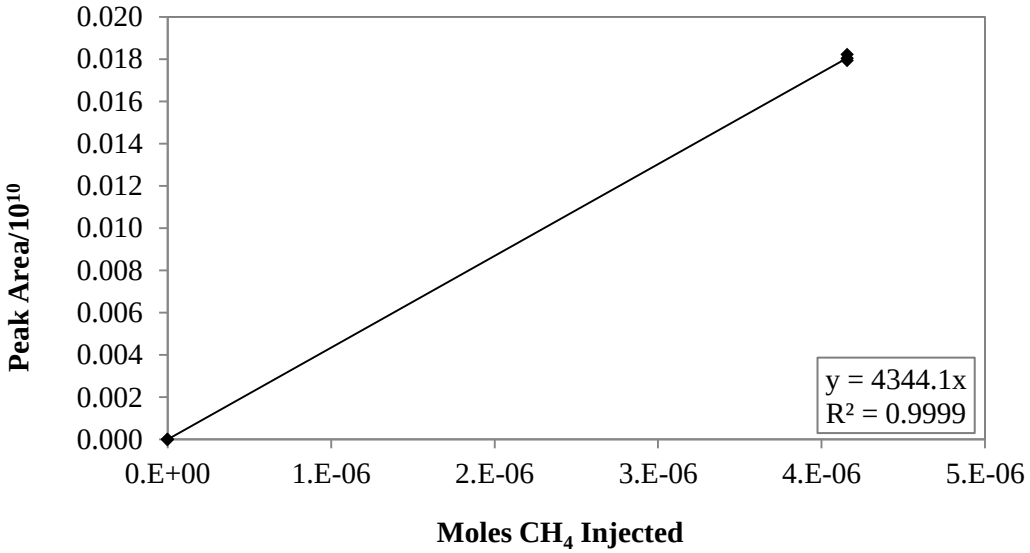

Figure B.1. HP5890 GC standard curve for conversion of peak area to moles of methane present in 0.1 mL injection.

Figure B.2 shows the standard curve used to quantify the percentage of methane in the headspace of each ABR chamber.

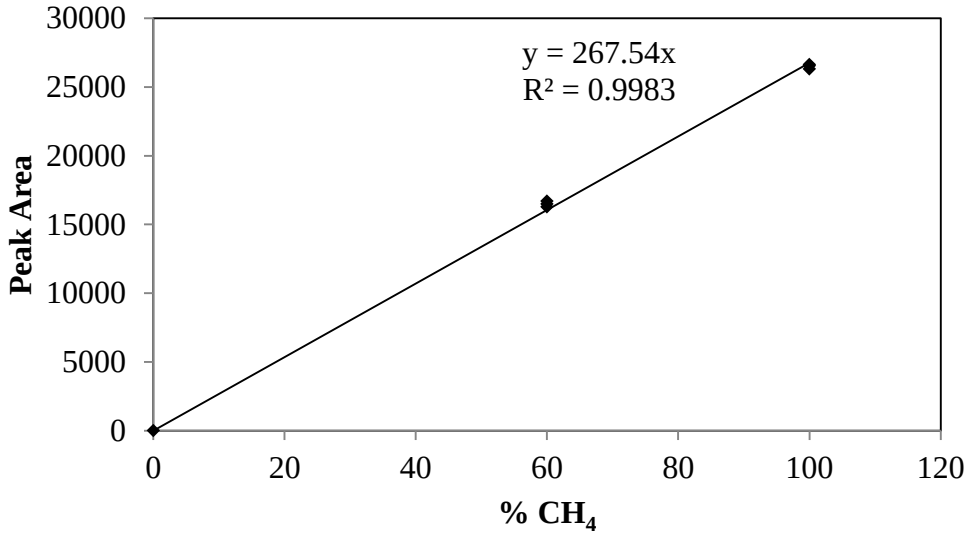

Figure B.2. HP6890 GC standard curve for conversion of peak area to percent methane in 0.3 mL sample injection

The minimum detection limit (MDL) for each VFA was determined by analyzing seven samples of diluted VFA standard solution (Supelco, Bellefonte, PA) with the HP6890 GC. The standard deviation of the seven peak areas determined by the GC was multiplied by the t-statistic (3.143) for a seven sample population with n-1 (6) degrees of freedom to calculate the MDL at a 99% confidence level. The MDL was then multiplied by a safety factor (SF) of two to determine the detection limits shown in Table B.2.

Table B.2. Minimum detection limits for each VFA

	Peak Areas (25 µV*s)								
	HAc	HPr	I-HBu	HBu	I-HVa	HVa	I-Hex	Hex	Hept
MDL (mg/L)	2.5	9.4	18.8	29.5	31.7	25.9	26.9	16.1	33.3
MDL (mg/L) with SF (2)	4.9	18.9	37.5	59.0	63.4	51.9	53.7	32.2	66.7

The quotient of Equations B.1 and B.2 were used to calculate the VA/PA ratio.

$$VA = \frac{V_{t,4.00} * C_t * 50000 \frac{mg\ CaCO_3}{eq.}}{V_s} \quad \text{(B.1)}$$

$$PA = \frac{V_{t,5.75} * 50000 \frac{mg\ CaCO_3}{eq.}}{V_s} \quad \text{(B.2)}$$

$V_{t,4.00}$: titrant volume required to lower the pH from 5.75 to 4.00 [mL]
V_s : sample volume [mL]
$V_{t,5.75}$: titrant volume required to lower initial pH to 5.75 [mL]

Table B.3 contains the measured feed characteristics for the ABR.

Table B.3. Feed characteristics at 15°C and 20°C (average [number of samples] (95% CI)).

Operating Temperature	TCOD (mg/L)	SCOD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	VSS (mg/L)
15°C	576 [56] (522,630)	230 [56] (214,246)	262 [56] (239,284)	233 [23] (162,305)	211 [8] (148,275)
20°C	531 [43] (465,597)	203 [43] (183,223)	241 [43] (212,271)	171 [18] (112,230)	147 [18] (102,192)

Table B.4 contains the measured effluent characteristics for the ABR.

Table B.4. Characteristics of ABR effluent at 15°C and 20°C. (average [number of samples] (95% CIs)).

Operating Temperature	TCOD (mg/L)	SCOD (mg/L)	BOD_5 (mg/L)	TSS (mg/L)	VSS (mg/L)
15°C	136 [56] (129,144)	86 [56] (81,92)	81 [56] (77,85)	11 [20] (9,14)	11 [8] (8,13)
20°C	100 [42] (93,108)	70 [43] (66,74)	60 [42] (56,64)	8 [17] (6,9)	7 [17] (5,9)

Figure B.3 shows influent TCOD concentration versus percent of total produced methane trapped in the dissolved phase.

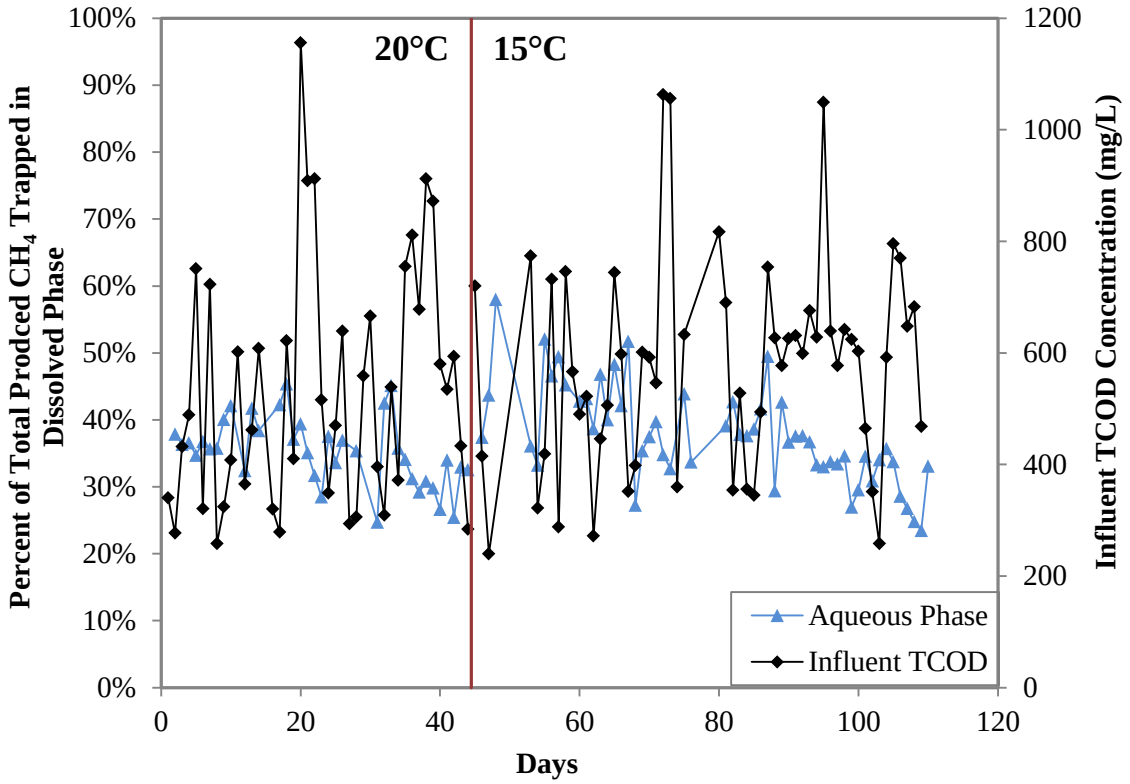

Figure B.3. Methane and influent TCOD concentration data measured at15°C and 20°C

Alkalinity was added every other day, so the ABR feed may or may not contain alkalinity. Figure B.4 shows the pH measurements for the feed and primary influent. "Primary influent" represents the collected municipal wastewater samples (no alkalinity added). "Feed" represents the ABR influent for a given day (may or may not contain alkalinity).

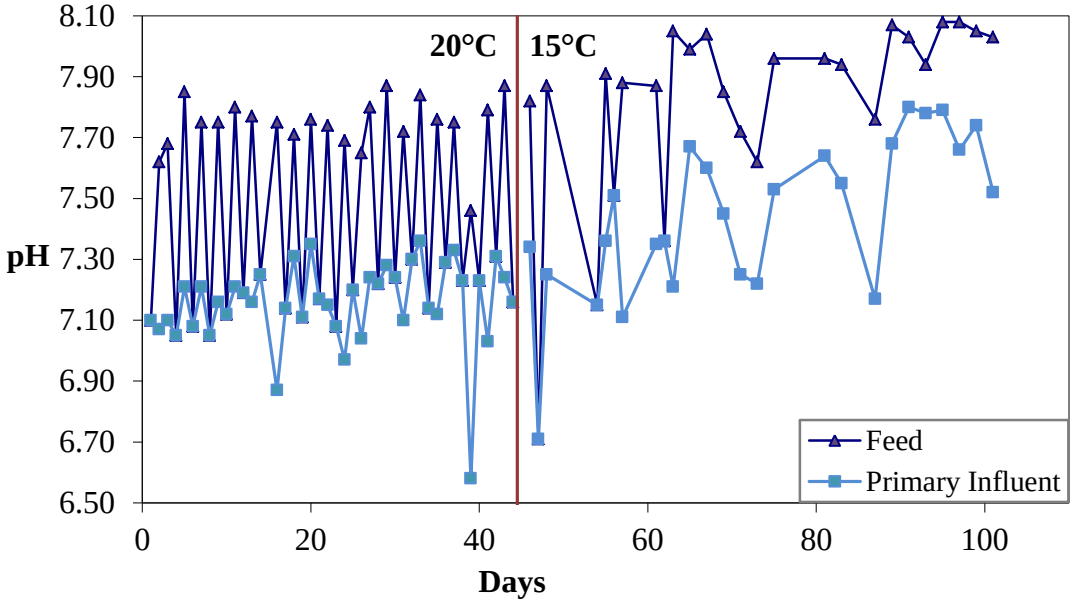

Figure B.4. pH data for feed and primary influent measured at 15°C and 20°C

Figure B.5 shows the pH measurements for each ABR chamber.

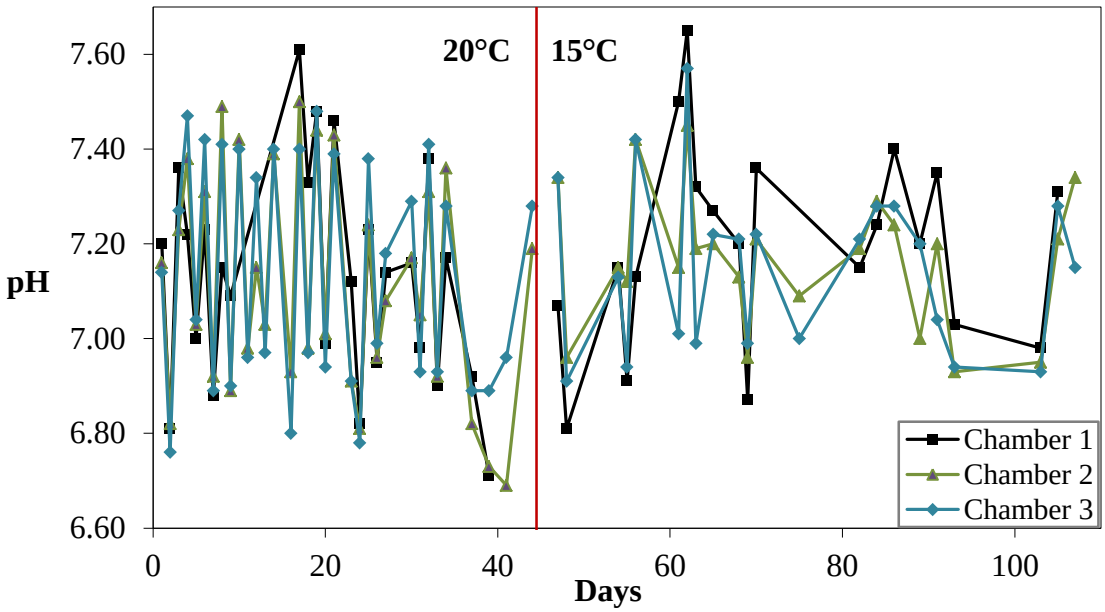

Figure B.5. pH data measured for chambers 1-3 at 15°C and 20°C

Figure B.6 shows the VA/PA measurements for the feed and primary influent.

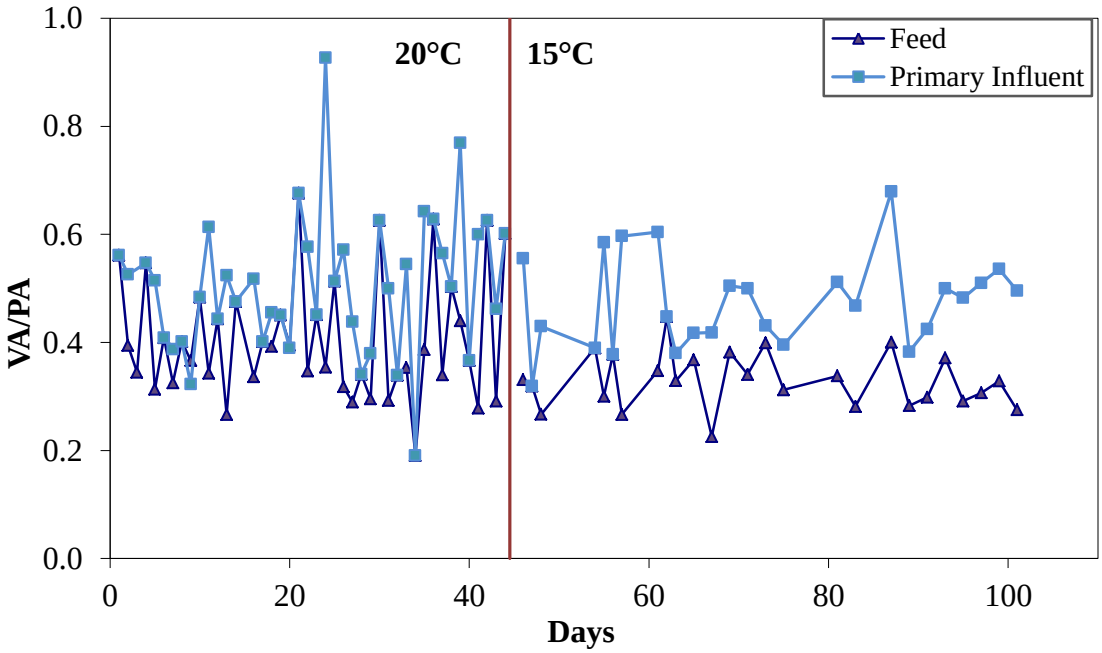

Figure B.6. VA/PA data for feed and primary influent measured at 15°C and 20°C

Figure B.7 shows the VA/PA measurements for each ABR chamber.

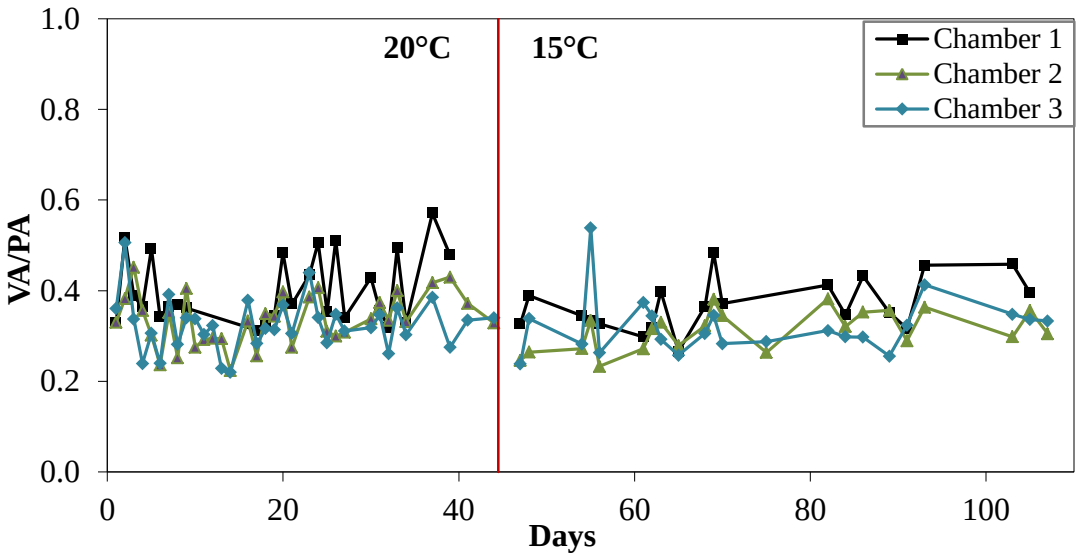

Figure B.7. VA/PA data for chambers 1-3 measured at 15°C and 20°C

Figures B.8 and B.9 show the acetate and propionate concentrations measured for each ABR chamber.

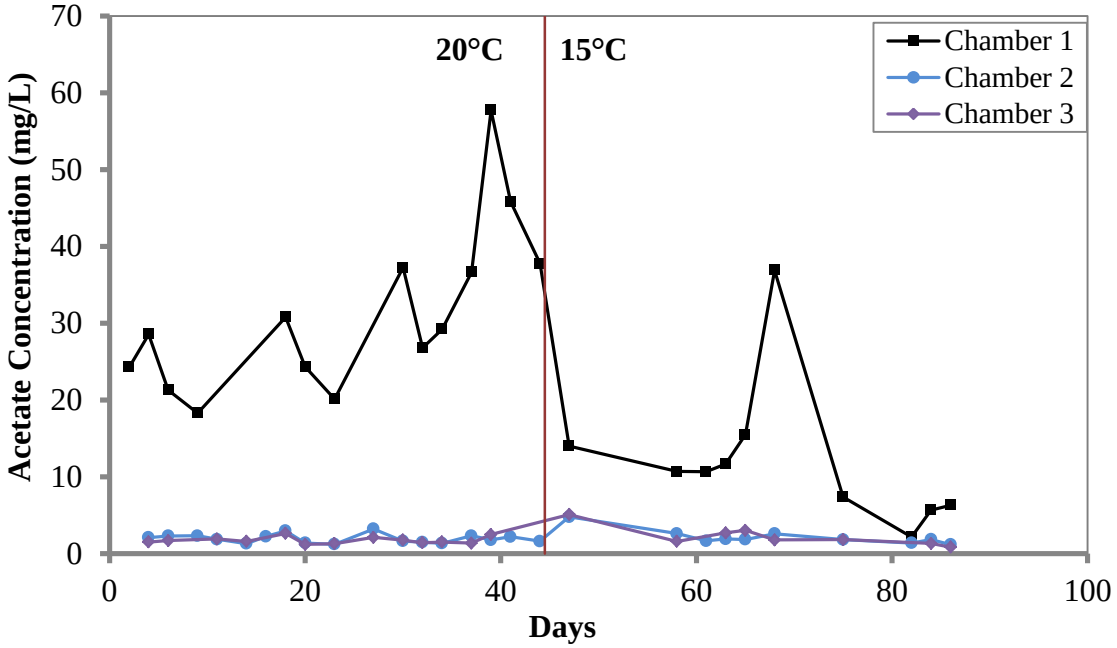

Figure B.8. Acetate concentrations in chambers 1-3 measured at 15°C and 20°C

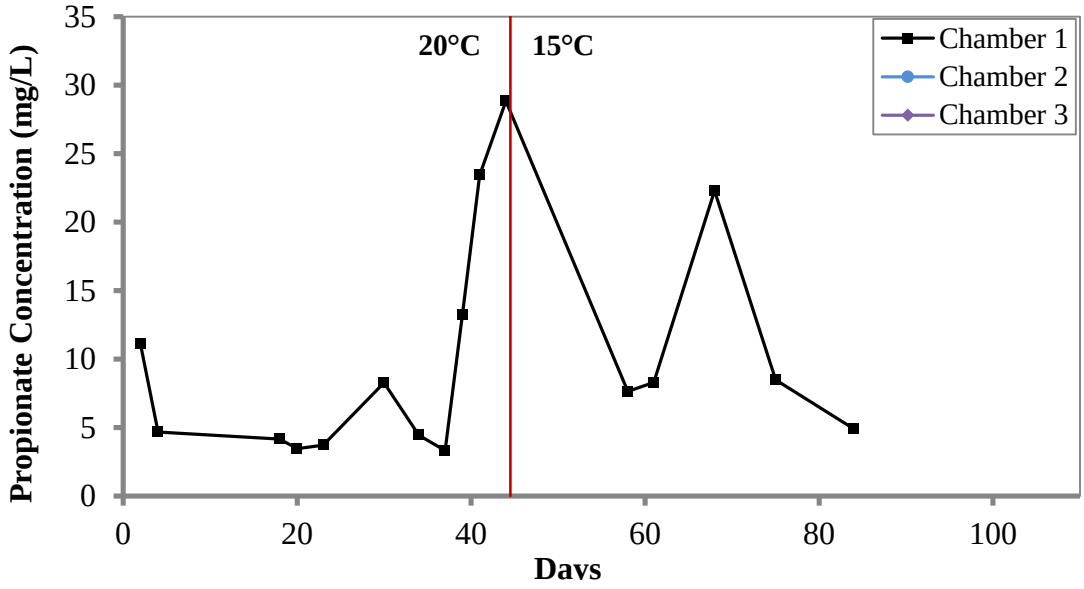

Figure B.9. Propionate concentrations in chambers 1-3 measured at 15°C and 20°C

Dissolved methane concentration sample calculations:

The following equations and sample calculations were used to determine the dissolved methane concentration of a given ABR effluent sample.

$$M_{g,s} = \frac{(moles\ CH_4)_{inj}}{0.0001\ L} \quad \text{(B.3)}$$

$$M_{aq,s} = K_H * M_{g,s} \quad \text{(B.4)}$$

$$M_{CH4,sb} = M_{g,s} * V_g + M_{aq,s} * V_l \quad \text{(B.5)}$$

$$M_{ABR} = \frac{M_{CH4,sb}}{V_e} \quad \text{(B.6)}$$

$$C_{ABR} = M_{ABR} * 16{,}000\ \frac{mg\ CH_4}{mole\ CH_4} \quad \text{(B.7)}$$

$M_{g,s}$: molar concentration of CH_4 in the gaseous portion of the serum bottle [mole/L]
$(moles\ CH_4)_{inj}$: moles of CH_4 present in injected headspace sample [moles]
$K_{H,T}$: dimensionless Henry's constant [M_{aq}/M_g]
$M_{aq,s}$: molar concentration of CH_4 in the aqueous portion of the serum bottle [mole/L]
$M_{CH4,sb}$: total moles of CH_4 present in serum bottle [moles]
V_g: volume of gaseous portion of serum bottle [L]
V_l: volume of liquid portion of serum bottle [L]
M_{ABR}: molar concentration of CH_4 in ABR effluent sample [mole/L]
V_e: volume of ABR effluent injected into serum bottle [L]
C_{ABR}: concentration of CH_4 in ABR effluent sample [mg/L]

Statistical analysis method description:

The mean value and standard deviation of the two data sets were known, therefore, the t-statistic for each parameter was calculated using Equation B.8

$$t = \frac{\bar{x}_1 - \bar{x}_2}{\sqrt{\frac{\sigma_1^2}{n_1} + \frac{\sigma_2^2}{n_2}}} \quad \text{(B.8)}$$

where t is the t-statistic, $\bar{x}_1$ is the mean of the first data set, $\bar{x}_2$ is the mean of the second data set, σ_1 is the standard deviation of the first data set, σ_2 is the standard deviation of the second data set, n_1 is the number of values in the first data set, and n_2 is the number of values in the second data set.

The sample size and variance of the two data sets are not equal. Therefore, degrees of freedom for each parameter were calculated using Equation B.9

$$df = \frac{\left(\frac{\sigma_1^2}{n_1}+\frac{\sigma_2^2}{n_2}\right)^2}{\left(\frac{\sigma_1^4}{n_1^2(n_1-1)}+\frac{\sigma_2^4}{n_2^2(n_2-1)}\right)} \quad \text{(B.9)}$$

where df is the degrees of freedom, n_1 is the number of values in the first data set, n_2 is the number of values in the second data set, σ_1 is the standard deviation of the first data set, and σ_2 is the standard deviation of the second data set. The degrees of freedom were rounded up to the nearest whole number for analysis. The rounded degrees of freedom and the calculated t-statistic were then used in conjunction with a t-test distribution table to either accept or reject the null hypothesis. Specifically, if the calculated t-statistic was greater than the corresponding t from the distribution table, the null hypothesis was rejected and the alternate hypothesis – the parameter value for the 15°C and 20°C data sets are significantly different at a 95% confidence level – was accepted.

Report detailing dissolved methane method comparison (from CENG 645 class project submitted to Professor Matthew Higgins):

The following report details experimentation conducted which compares the dissolved methane measurement method used in a previous study conducted by Wacker to the method used in this study (Wacker 2014).

INTRODUCTION

Anaerobic treatment of wastewater is often considered an environmentally preferable alternative to conventional aerobic wastewater treatment because it does not require electricity for aeration and, therefore, produces less carbon dioxide. However, anaerobic treatment effluent contains dissolved methane – a potent greenhouse gas – which is released into the atmosphere when the effluent is discharged to waterways. The presence of dissolved methane in anaerobic effluent could potentially negate the benefit of reduced carbon dioxide emissions. Consequently, the quantity of dissolved methane in anaerobic effluent must be determined to obtain a more comprehensive understanding of the environmental implications of anaerobic wastewater treatment systems.

Quantification of anaerobic reactor effluent requires an understanding of the fundamentals of chemistry. The anaerobic reactor is a closed system which contains methane in both the gaseous and aqueous state, according to Equation B.10:

$$CH_{4(g)} \leftrightarrow CH_{4(aq)} \tag{B.10}$$

The degree to which methane partitions between the gas and liquid phase is determined by Henry's Law for methane, which is described by Equation B.11:

$$K_H = \frac{[CH_{4(aq)}]}{P_{CH4}} = 0.00136 \frac{mol}{L-atm} \tag{B.11}$$

where K_H is Henry's constant for methane at standard temperature and pressure (STP), $[CH_{4(aq)}]$ is the molar concentration of methane in the aqueous phase, and P_{CH4} is the partial pressure of methane in the gas phase.

There are two primary goals of this research: (1) use two methods for dissolved methane quantification to determine the time required for methane concentrations to reach equilibrium, and (2) assess each method's precision and accuracy by comparing calculated aqueous methane concentrations.

METHODS

Two methods were utilized in this research. The procedure used for the first method, hereby referred to as Method 1, consisted of the following:

Fill an 11.5 mL serum bottle with deionized (DI) water and seal the bottle with a rubber septum and aluminum crimp cap. Insert a two-way needle into the septum. Attach tubing to the exposed end of the two-way needle and place the free end of the tube into a beaker of water.

Draw 20 mL of liquid effluent sample and 40 mL of 100% gaseous helium into a 60 mL syringe. Insert the syringe needle into a rubber stopper to close the system. Shake for 60 seconds to allow the methane contained in the liquid sample to equilibrate. Remove the syringe needle from the rubber stopper and insert the full length of the syringe needle into the prepared 11.5 mL serum bottle. Invert the syringe and bottle to force the helium and methane gas mixture to the top of the syringe and flush the serum bottle with the gas mixture until all the water has exited through the two-way needle. Remove the syringe needle and two-way needle from the bottle. Inject a 100 µL headspace gas sample from the serum bottle on the HP 6980 Gas Chromatography System with Flame Ionization Detector (GC-FID) to obtain a peak area value for the sample.

The alternative method, hereby referred to as Method 2, used the following procedure:

Seal an 11.5 mL serum bottle with a rubber septum and aluminum crimp cap. Inject a 3 mL sample of reactor effluent into the serum bottle and shake for 60 seconds. Inject a 100 µL

headspace gas sample into the HP 5890 GC-FID to obtain a methane peak area value for the sample.

RESULTS & DISCUSSION

Figures B.10 and B.11 depict the peak area values corresponding to various mixing times using Method 1 and Method 2, respectively. The figures indicate that both methods achieve an equilibrium state after approximately 30 seconds of mixing.

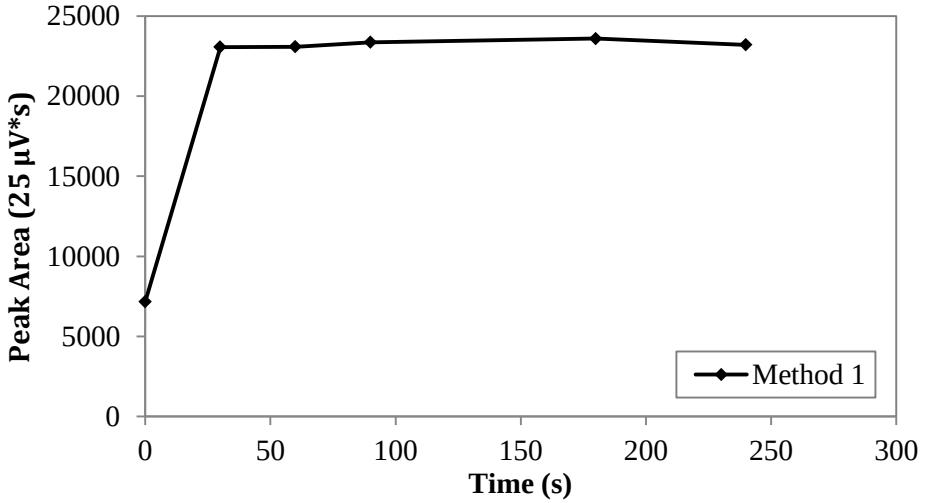

Figure B.10 Peak area vs. mixing time for Method 1

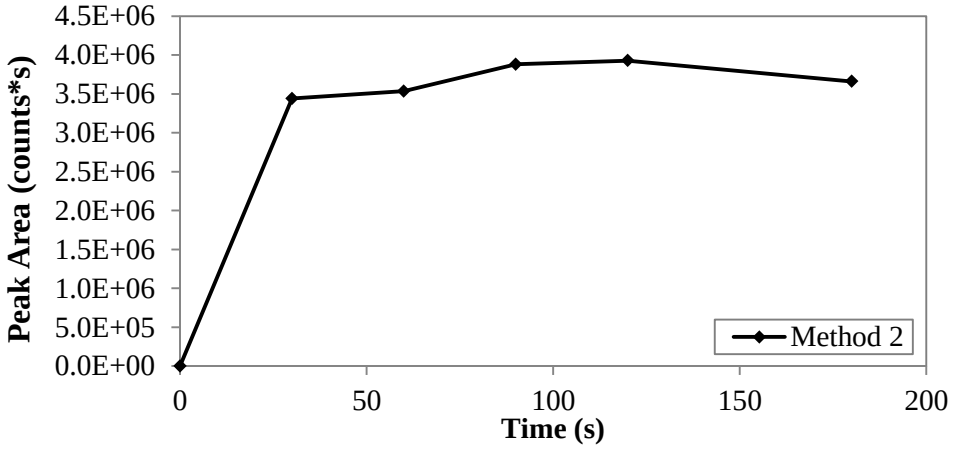

Figure B.11. Peak area vs. mixing time for Method 2

Both systems required a minimum of 30 seconds to reach equilibrium. Therefore, a safety factor of two was applied and a 60 second mixing time was used for the remainder of testing.

The standard curves depicted in Figures B.12 and B.13 were developed for Method 1 and Method 2, respectively. The standard curves were used to convert the peak area values obtained via the GC into units of moles. The R^2 values indicate that the standard curve for Method 1 is more precise than the standard curve for Method 2.

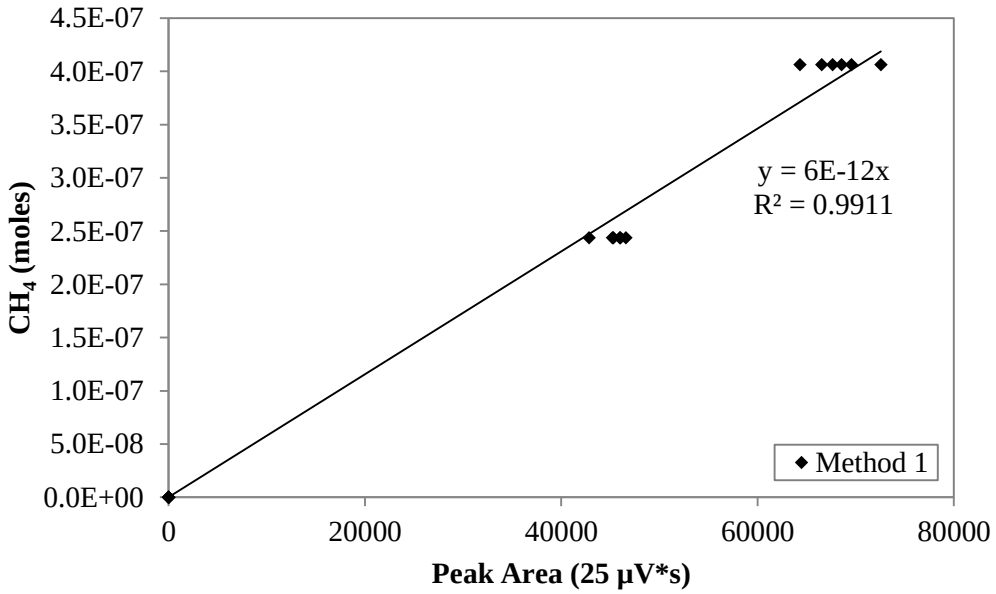

Figure B.12. Standard curve for Method 1

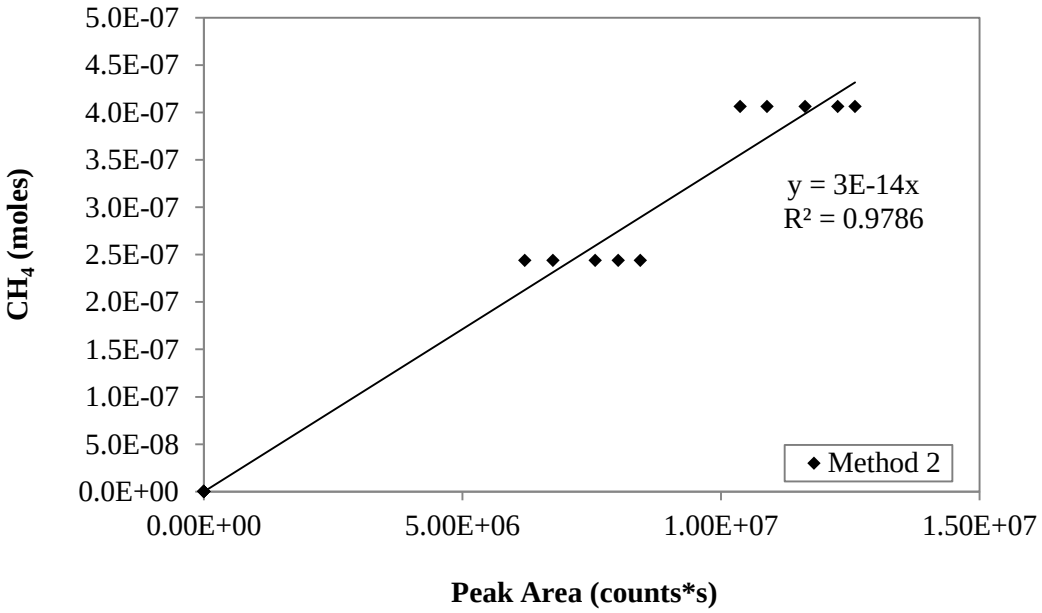

Figure B.13. Standard curve for Method 2

Once the amount of moles present in each sample was determined, the methane concentrations in the anaerobic reactor effluent were calculated. The calculations used to determine the aqueous methane concentrations in Method 1 assumed that all of the dissolved methane contained in the reactor effluent was transferred to the gaseous phase when mixed with helium. The validity of this assumption was testing by running Method 1 three times, using the same effluent sample for the three tests. Four trials of this experiment were run. The results of the experiment are depicted graphically in Figure B.14.

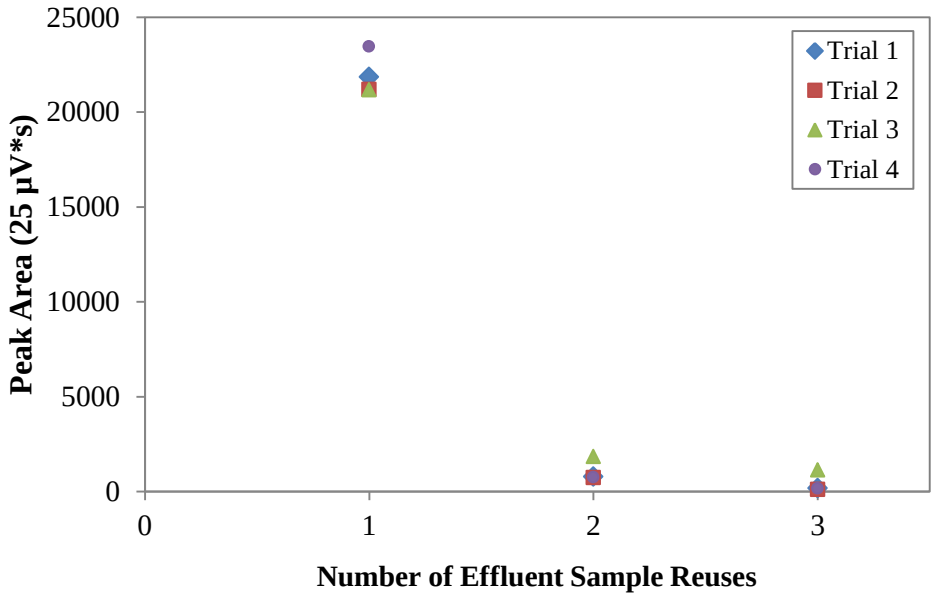

Figure B.14. Peak area vs. number of effluent sample reuses for Method 1

Figure B.14 indicates that a portion of the total methane remains in the effluent samples after the first effluent run. Table B.5 contains the peak area values for the trials shown in Figure B.14 and average estimates of the percent of the total methane in the effluent sample that is extracted with each reuse. On average, approximately 6 percent of total dissolved methane present in the effluent is not accounted for when using Method 1. Note that the "% of Total CH_4 in Sample" column is an over-estimate of the true percentage of methane extracted in each trial. The calculated value assumes that all methane is removed from the sample after the third trial. This is not true because the peak area values for Reuse 3 are greater than zero. It can be concluded that there is at least 6 percent of the total methane remaining in the effluent after the first extraction when using Method 1.

Table B.5. Experimental values obtained for peak area vs. number of effluent sample reuses for Method 1

Sample Reuses	Peak Area					% of Total CH_4 in Sample
	Trial 1	Trial 2	Trial 3	Trial 4	Average	
1	21844	21195	21171	23461	21919	93.9
2	787	732	1857	757	1033	4.4
3	178	107	1134	163	396	1.7

Table B.6 contains the calculated values for the concentration of dissolved methane in the anaerobic effluent determined using Method 1. Note that these values were determined utilizing the assumption that all of the methane present in the effluent sample is transferred to the gas phase at equilibrium, which was determined to be incorrect. The average dissolved methane concentration calculated using Method 1 was approximately 44.5 mg/L.

Table B.6: Calculated dissolved methane concentrations determined using Method 1

Time (s)	Peak Area (25 μV*s)	Injected CH_4 (moles)	moles CH4/mL He-CH_4 mixture	mg CH_4/mL liquid sample	CH_4 Conc. (mg/L)
30	23065	1.38E-07	1.38E-06	4.43E-02	44.3
60	23081	1.38E-07	1.38E-06	4.43E-02	44.3
90	23360	1.40E-07	1.40E-06	4.49E-02	44.9
180	23591	1.42E-07	1.42E-06	4.53E-02	45.3

Table B.7 contains the calculated values for the concentration of dissolved methane in the anaerobic effluent determined using Method 2. Note that STP conditions were assumed and that a Henry's constant value of 0.00136 mole L^{-1} atm^{-1} was used in relevant calculations. The average dissolved methane concentration obtained using Method 2 was approximately 49.9 mg/L.

Table B.7. Calculated dissolved methane concentrations determined using Method 2

Time (s)	Peak Area (counts*s)	Injected CH_4 (moles)	Gaseous CH_4 Conc. (M)	Partial Pressure of CH_4 (atm)	Aqueous CH_4 Conc. (M)	Total CH_4 in Serum Bottle (moles)	Effluent Aqueous CH_4 Conc. (mg/L)
30	3.44E+06	1.03E-07	1.03E-03	2.52E-02	3.43E-05	8.88E-06	47.3
60	3.54E+06	1.06E-07	1.06E-03	2.59E-02	3.52E-05	9.12E-06	48.6
90	3.88E+06	1.16E-07	1.16E-03	2.84E-02	3.87E-05	1.00E-05	53.4
180	3.66E+06	1.10E-07	1.10E-03	2.68E-02	3.65E-05	9.44E-06	50.4

Note that the average calculated dissolved methane concentrations determined using Method 1 are approximately 90% of the calculated dissolved methane concentrations determined using Method 2. Based on these results, Method 2 is a more accurate method than Method 1 for determining dissolved methane concentrations in anaerobic reactor effluent. However,

modifications may be made to Method 1 to reduce the error introduced by assuming that all methane is extracted from the anaerobic effluent during the mixing process.

CONCLUSION

The results of this experiment indicate that for Method 1 and Method 2 a minimum mixing time of 30 seconds is required for the methane gas-liquid system to reach equilibrium. Furthermore, it can be concluded from the R^2 values of the standard curves for the two methods that the HP 6980 GC-FID produces slightly more precise results than the HP 5890 GC-FID. However, it was determined that Method 1 is a less accurate method for quantification of dissolved methane than Method 2 due to its assumption that all methane is extracted from the anaerobic reactor effluent sample during helium mixing. Modifications, such as increasing the number effluent reuses, may be made to Method 1 to reduce the error resulting from the previously stated assumption and produce more accurate dissolved methane concentration values.

Appendix C: Input – Output Model Construction

Biosolids production from microbial growth calculations:

Equations C.1 and C.2 were used to quantify the volatile solids produced in the trickling filter via microbial growth. Table C.1. details the values used in the equations.

$$VS_{microbe} = \frac{Y*1.5\frac{g\ BOD_L}{g\ BOD_5}*(S_o-S_e)*Q*\left(\frac{1\ kg}{10^6 mg}\right)}{1+b_T\theta_x} \quad \text{(C.1)}$$

$$b_T = b_{ref}(1.07)^{T-T_{ref}} \quad \text{(C.2)}$$

$VS_{microbe}$: rate of biosolids production from microbial growth [kg VS/ d]
Y: true yield [g VS/g BOD_L]
S_o: influent BOD_5 concentration [mg/L]
S_e: effluent BOD_5 concentration [mg/L]
Q: influent flow rate [L/d]
b_T: endogenous decay rate for temperature T [d^{-1}]
θ_x: SRT [d]
b_{ref}: known endogenous decay rate for temperature T_{ref} [d^{-1}]
T: temperature of interest [°C]
T_{ref}: known temperature corresponding to b_{ref} [°C]

Table C.1. Values used in trickling filter biosolids production calculations. SRT values are presented as mean (minimum, maximum).

Parameter	Units	15°C Value	20°C Value	Source
SRT	d	4.2 (4.1,4.3)	4.2 (4.1,4.3)	Tchobanoglous et al. 2003
Y	g VSS/g BOD_L	0.45	0.45	Rittmann and McCarty, 2001
b	d^{-1}	0.10	0.14	Rittmann and McCarty, 2001
S_o	mg BOD_5/L	200	200	MRSA primary effluent data for corresponding ABR operating period
S_e	mg BOD_5/L	30	20	EPA limit (15°C); Calculated, (Tchobanoglous et al. 2003) (20°C)

Equation C.3 was used to quantify the volatile solids produced in the ABR via microbial growth. Table C.2 details the values used in the equations.

$$VS_{microbe} = Q * Y * (S_o - S) \quad \text{(C.3)}$$

$VS_{microbe}$: rate of biosolids production from microbial growth [kg VS/ d]
Y: observed yield [g VS/g BOD_r]
S_o: influent BOD_5 concentration [mg/L]
S_e: effluent BOD_5 concentration [mg/L]
Q: influent flow rate [L/d]

Table C.2. Values used in ABR biosolids production calculations. S_o and S_e values are represented as mean (minimum, maximum)

Parameter	Units	15°C Value	20°C Value	Source
Y	g VSS/g BOD_r	0.051	0.051	Shin et al. 2014; Wacker, 2014; Rittmann and McCarty, 2001
S_o	mg BOD_5/L	262 (109,483)	241 (117,525)	Average ABR influent data for corresponding operating period
S_e	mg BOD_5/L	81 (49,116)	60 (14,93)	Average ABR effluent data for corresponding operating period

Cogeneration equations and sample calculations:

The following sample calculations and equations were used to determine the heat and electricity produced via cogeneration of biogas.

$$E_{total} = \frac{Y_{CH4} * (1 - L_f) * LHV_{CH4}}{3.6 \frac{MJ}{kWh}} \quad \text{(C.4)}$$

$$E_{gen} = E_{total} * \varepsilon_{elec} \quad \text{(C.5)}$$

$$E_{heat} = E_{total} * 3.6 \frac{MJ}{kWh} * \varepsilon_{therm} \quad \text{(C.6)}$$

$$\varepsilon_{therm} = \varepsilon_{overall} - \varepsilon_{elec} \quad \text{(C.7)}$$

E_{total} : total kWh generation via biogas (pre-application of efficiencies) [kWh/d]
Y_{CH4}: total methane yield day [m^3/d]

L_f : fugitive methane loss [%/100]
LHV_{CH4}: lower heating value of methane at 20°C and 1 atm [MJ/m^3]
E_{gen}: electricity generated by CHP (post-application of electrical efficiency) [kWh/d]
ε_{elec}: CHP electrical efficiency [%/100]
E_{heat}: thermal energy generated by CHP (post-application of thermal efficiency) [MJ/d]
ε_{therm}: CHP thermal efficiency [%/100]
$\varepsilon_{overall}$: overall CHP efficiency [%/100]

Appendix D: Solids Mass Balance

Solids Mass Balance

In this section, we describe the mass balances conducted on the solids in each assembly to determine the total and volatile solids entering the digester (for the Trickling Filter and ABR + Trickling Filter scenarios) and the total solids that require disposal for each model. Mass balances were conducted for two types of solids: solids in the influent and solids produced as a result of microbial growth. Effluent solids accounted for include solids, which are collected, treated, and disposed, as well as solids that are discharged with effluent. Effluent TSS concentrations were less than or equal to the 30 mg/L TSS EPA limit for all models.

Solids Production Values

Table D.1 summarizes the determined values for TS entering the anaerobic digester, VS entering the anaerobic digester, and TS which require disposal for each scenario. The values in Table D.1 were used to determine impacts relating to disposal of solids and electricity generated from solids destruction in the anaerobic digester. Results show that the trickling filter produces nearly five times more solids for disposal than the ABR. In the Trickling Filter scenario, nearly 55 percent of total solids entering the digester come from the primary clarifier sludge. The remainder of the difference between the solids produced by the ABR and Trickling Filter can be attributed to quicker rates of microbial growth in the aerobic trickling filter than in the ABR.

Table D.1. Results of solids analysis for each model at 15°C and 20°C

	Solids Entering Digester (kg TS/d)	Solids Entering Digester (kg VS/d)	Solids for Disposal (kg TS/d)
ABR 15°C	-	-	244
ABR 20°C	-	-	252
TF 15°C	1818	1267	1109
TF 20°C	1790	1247	1092
ABR+TF 15°C	742	431	501
ABR+TF 20°C	639	350	443
ABR+CW 15°C	-	-	244
ABR+CW 20°C	-	-	252

Refer to Tables C.1 and C.2 for influent BOD5 values used to calculate biosolids production.

ABR Model: Solids

The diagram presented in Figure D.1 depicts the basis of the mass balance on solids in the reactor. There are two sources of solids accumulation in this model: (1) fixed solids trapped in the reactor bed and (2) microbial growth. The fixed solids were determined using influent and effluent solids concentrations from TSS and VSS data measured during steady-state ABR operation (see Chapter 3) and volatile solids produced via microbial growth were determined using calculations detailed in the appendix. The total solids required for disposal were determined by assuming that 90% of the biosolids were organic (Gossett, n.d.).

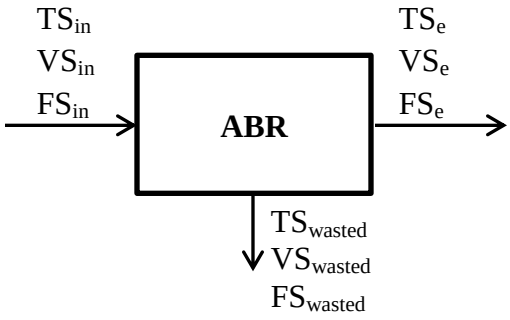

Figure D.1. Block flow diagram used for solids mass balance for ABR

The solids parameters of interest for the ABR were determined using the following calculations and equations where

Q: influent flow rate [L/d]
TS_{in}: total solids in ABR influent [kg TS/d]
VS_{in}: volatile solids in ABR influent [kg VS/d]
TSS_{in}: total solids concentration of ABR influent [mg/L]
VSS_{in}: volatile solids concentration of ABR influent [mg/L]
FS_{in}: fixed solids in ABR influent [kg FS/d]
TS_e: total solids in ABR effluent [kg TS/d]
VS_e: volatile solids in ABR effluent [kg VS/d]
TSS_e: total solids concentration of ABR effluent [mg/L]
VSS_e: volatile solids concentration of ABR effluent [mg/L]
FS_e: fixed solids in ABR effluent [kg FS/d]
$VS_{microbe}$: rate of volatile solids production from microbial growth [kg VS/ d][e]
TS_{wasted}: total solids requiring disposal [kg TS/d]

$$TS_{in} = TSS_{in} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.1)}$$

$$VS_{in} = VSS_{in} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.2)}$$

$$FS_{in} = TS_{in} - VS_{in} \quad \text{(D.3)}$$

$$TS_e = TSS_e * Q * \frac{kg}{10^6\ mg} \quad \text{(D.4)}$$

$$VS_e = VSS_e * Q * \frac{kg}{10^6\ mg} \quad \text{(D.5)}$$

$$FS_e = TS_e - VS_e \quad \text{(D.6)}$$

$$TS_{wasted} = (FS_{in} - FS_e) + \frac{VS_{microbe}}{0.9\frac{kg\ VS}{kg\ TS}} \quad \text{(D.7)}$$

[e] This parameter was calculated using Equation C.3.

Trickling Filter Model: Solids

Figure D.2 shows the basis of the mass balance on solids in the trickling filter model. The average primary clarifier effluent TSS concentrations were provided by Milton Regional Sewer Authority. It was assumed that the primary clarifier achieved 60 percent TS removal – which is within the typical range for primary clarifiers undergoing settling without coagulation (Vesilind, 2003). Average VSS/TSS ratios of 0.675 and 0.825 were assumed for primary clarifier sludge and effluent, respectively, which are averages of the typical ranges for these values (WEF, 2006). Similarly, the secondary clarifier sludge was assumed to have a VSS/TSS ratio of 0.725 which is the average of the typical range reported by Vesilind (Vesilind, 2003). The assumption was made that the trickling filter removed 72.5 percent of TSS from the primary clarifier effluent, which is an average value of a reported typical range of 65-80 percent (Spellman, 2000). It was also assumed that 90 percent of the solids produced via microbial growth were organic (Gossett, n.d.). Details regarding the microbial growth calculation can be found in the appendix. Finally, the assumption was made that 56 percent of VS was destroyed via anaerobic digestion (Rittmann and McCarty, 2001).

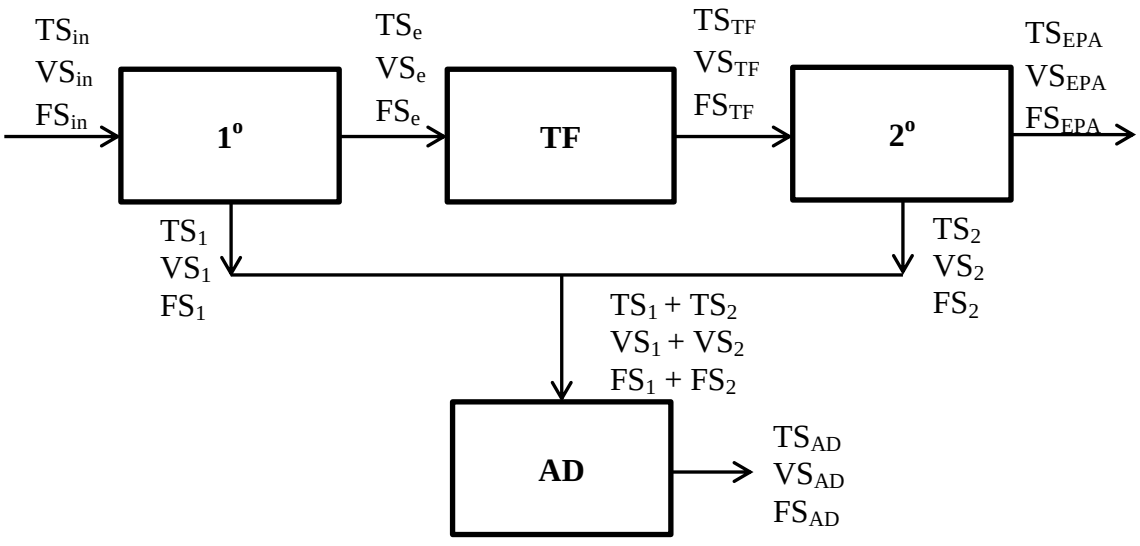

Figure D.2. Block flow diagram used for solids mass balance for the Trickling Filter

The solids parameters of interest for the Trickling Filter were determined using the following calculations and equations where

Q: influent flow rate [L/d]
TS_e: total solids in primary clarifier effluent [kg TS/d]
VS_e: volatile solids in primary clarifier effluent [kg VS/d]
TSS_e: total solids concentration of primary clarifier effluent [mg/L]
FS_e: fixed solids in primary clarifier effluent [kg FS/d]
TS_{in}: total solids in primary clarifier influent [kg TS/d]
TS_1: total solids in primary clarifier sludge [kg TS/d]
VS_1: volatile solids in primary clarifier sludge [kg VS/d]
FS_1: fixed solids in primary clarifier sludge [kg FS/d]
TS_{EPA}: maximum total solids discharged to meet EPA limit [kg TS/d]
TS_{TF}: total solids in trickling filter effluent [kg VS/d]
$VS_{microbe}$: rate of volatile solids production from microbial growth [kg VS/ d][f]
TS_2: total solids in secondary clarifier sludge [kg TS/d]
VS_2: volatile solids in secondary clarifier sludge [kg VS/d]
FS_2: fixed solids in secondary clarifier sludge [kg FS/d]
TS_{AD}: total solids in anaerobic digestion effluent [kg TS/d]
VS_{AD}: volatile solids in anaerobic digestion effluent [kg VS/d]
FS_{AD}: fixed solids in anaerobic digestion effluent [kg FS/d]

$$TS_e = TSS_e * Q * \frac{kg}{10^6\ mg} \quad \text{(D.8)}$$

$$VS_e = TS_e * 0.825 \frac{kg\ VS}{kg\ TS} \quad \text{(D.9)}$$

$$FS_e = TS_e - VS_e \quad \text{(D.10)}$$

$$TS_{in} = \frac{TS_e}{(1-0.6)} \quad \text{(D.11)}$$

$$TS_1 = TS_{in} * 0.6 \quad \text{(D.12)}$$

[f] This parameter was calculated using Equations C.1 and C.2.

$$VS_1 = TS_1 * 0.675\ \frac{kg\ VS}{kg\ TS} \quad \text{(D.13)}$$

$$FS_1 = TS_1 - VS_1 \quad \text{(D.14)}$$

$$TS_{EPA} = 30\frac{mg}{L} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.15)}$$

$$TS_{TF} = (1 - 0.725) * TS_e \quad \text{(D.16)}$$

$$TS_{TF} = \frac{VS_{microbe}}{0.9\frac{kg\ VS}{kg\ TS}} + TS_{TF} + FS_e \quad \text{(D.17)}$$

$$TS_2 = TS_{TF} - TS_{EPA} \quad \text{(D.18)}$$

$$VS_2 = TS_2 * 0.725\frac{kg\ VS}{kg\ TS} \quad \text{(D.19)}$$

$$FS_2 = TS_2 - VS_2 \quad \text{(D.20)}$$

$$VS_{AD} = (VS_1 + VS_2) * \left(1 - 0.56\frac{g\ VS\ destroyed}{g\ VS\ applied}\right) \quad \text{(D.21)}$$

$$FS_{AD} = FS_1 + FS_2 \quad \text{(D.22)}$$

$$TS_{AD} = FS_{AD} + VS_{AD} \quad \text{(D.23)}$$

ABR + Trickling Filter Model: Solids

Figure D.3 shows the basis of the solids mass balance for the ABR + Trickling Filter model. As with the ABR scenario, influent and effluent solids concentrations were obtained from TSS and VSS data measured during steady-state ABR operation and "VS_{ABR}" represents only the volatile solids produced via microbial growth in the ABR. The total fixed solids exiting the ABR with each bulk wasting was calculated as the sum of the fixed solids trapped in the reactor plus the fixed solids associated with microbial growth, assuming that the microbial growth biosolids were 90 percent volatile (Gossett, n.d.). Detailed calculations used to determine biosolids production from microbial growth can be found in the appendix.

As in the Trickling Filter scenario, it was assumed that the trickling filter removed 72.5 percent of VS_e, the TS/VS ratio of the secondary clarifier sludge was 0.725, and 56 percent of VS were removed via anaerobic digestion (Spellman, 2000; Vesilind, 2003; Rittmann and McCarty, 2001). It was assumed that the microbial solids were 90 percent organic (Gossett, n.d.).

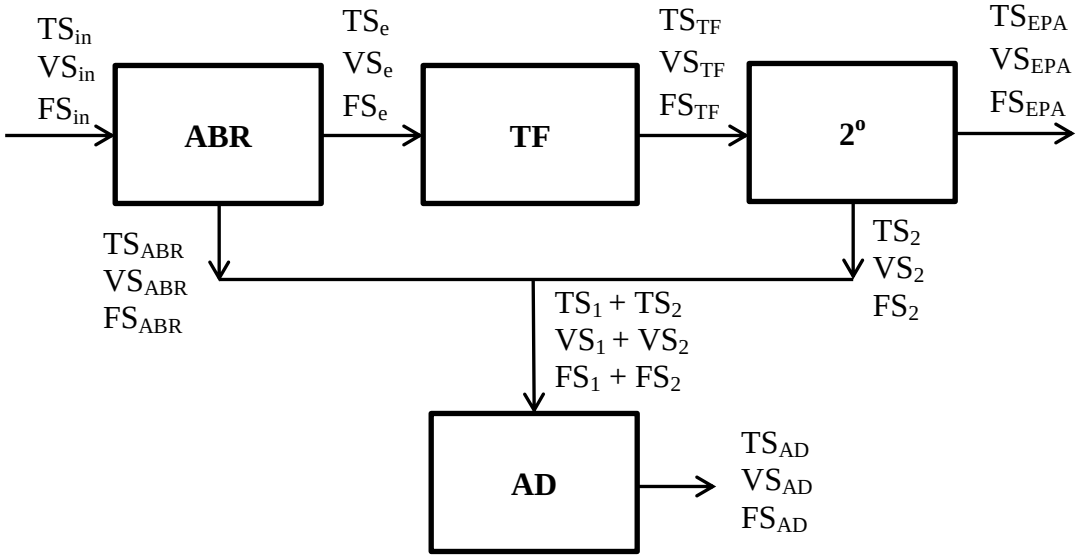

Figure D.3. Block flow diagram used for solids mass balance of the ABR + Trickling Filter assembly

The solids parameters of interest for the ABR + Trickling Filter were determined using the following calculations and equations where

Q: influent flow rate [L/d]
TS_{in}: total solids in ABR influent [kg TS/d]
VS_{in}: volatile solids in ABR influent [kg VS/d]
TSS_{in}: total solids concentration of ABR influent [mg/L]
VSS_{in}: volatile solids concentration of ABR influent [mg/L]
FS_{in}: fixed solids in ABR influent [kg FS/d]
TS_e: total solids in ABR effluent [kg TS/d]
VS_e: volatile solids in ABR effluent [kg VS/d]
TSS_e: total solids concentration of ABR effluent [mg/L]
VSS_e: volatile solids concentration of ABR effluent [mg/L]
FS_e: fixed solids in ABR effluent [kg FS/d]
VS_{ABR}: rate of volatile solids production from microbial growth in the ABR [kg VS/ d]
TS_{TF}: total solids in trickling filter effluent [kg VS/d]
$VS_{microbe,TF}$: rate of volatile solids production from microbial growth in the TF [kg VS/ d][g]
TS_{EPA}: maximum total solids discharged to meet EPA limit [kg TS/d]
TS_2: total solids in secondary clarifier sludge [kg TS/d]
VS_2: volatile solids in secondary clarifier sludge [kg VS/d]
FS_2: fixed solids in secondary clarifier sludge [kg FS/d]
TS_{AD}: total solids in anaerobic digestion effluent [kg TS/d]
VS_{AD}: volatile solids in anaerobic digestion effluent [kg VS/d]
FS_{AD}: fixed solids in anaerobic digestion effluent [kg FS/d]

$$TS_{in} = TSS_{in} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.24)}$$

$$VS_{in} = VSS_{in} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.25)}$$

$$FS_{in} = TS_{in} - VS_{in} \quad \text{(D.26)}$$

$$TS_e = TSS_e * Q * \frac{kg}{10^6\ mg} \quad \text{(D.27)}$$

[g] This parameter was calculated using Equations C.1-C.3.

$$VS_e = VSS_e * Q * \frac{kg}{10^6\ mg} \quad \text{(D.28)}$$

$$FS_e = TS_e - VS_e \quad \text{(D.29)}$$

$$FS_{ABR} = (FS_{in} - FS_e) + \left(\frac{0.1\ kg\ FS}{kg\ TS} * \frac{kg\ TS}{0.9\ kg\ VS} * VS_{ABR}\right) \quad \text{(D.30)}$$

$$TS_{ABR} = FS_{ABR} + VS_{ABR} \quad \text{(D.31)}$$

$$TS_{EPA} = 30\frac{mg\ TS}{L} * Q * \frac{kg}{10^6\ mg} \quad \text{(D.32)}$$

$$TS_{TF} = \frac{VS_{microbe,TF}}{0.9\frac{kg\ VS}{kg\ TS}} + \left(TS_e * (1 - 0.725)\right) + FS_e) \quad \text{(D.33)}$$

$$TS_2 = TS_{TF} - TS_{EPA} \quad \text{(D.34)}$$

$$VS_2 = TS_2 * 0.725\frac{kg\ VS}{kg\ TS} \quad \text{(D.35)}$$

$$FS_2 = TS_2 - VS_2 \quad \text{(D.36)}$$

$$VS_{AD} = (VS_1 + VS_2) * \left(1 - 0.56\frac{g\ VS\ destroyed}{g\ VS\ applied}\right) \quad \text{(D.37)}$$

$$FS_{AD} = FS_1 + FS_2 \quad \text{(D.38)}$$

$$TS_{AD} = FS_{AD} + VS_{AD} \quad \text{(D.39)}$$

ABR + Constructed Wetland Model: Solids

The basis for the mass balance on solids in the ABR+CW model is shown in Figure D.4. Constructed wetlands do not require disposal of solids (Mannino et al. 2008) which makes the solids calculations procedure identical to that described for the ABR model with the exception that it was assumed that the wetland removed 75 percent of TSS (Vymazal and Kröpfelová, 2008). The additional further increased effluent quality.

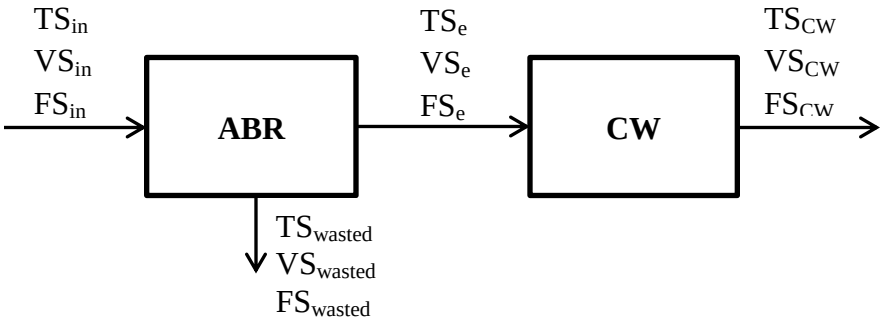

Figure D.4. Block flow diagram used for solids mass balance of the ABR + Constructed Wetland

The solids parameters of interest for the ABR + Constructed Wetland were determined using Equations D.1-D.7. The concentration of solids in the wetland effluent was determined using Equation D.40

$$TS_{CW} = (TS_e) * (1 - 0.75) \quad \text{(D.40)}$$

where TS_{CW} [kg TS/d] is the concentration of total solids in the wetland effluent.

Appendix E: Life Cycle Inventories at 15°C and 20°C

The life cycle inventories for 15°C and 20°C treatment (Table E.1 and Table E.2) were used to determine the differences in eutrophication impacts for the ABR, Trickling Filter, ABR + Trickling Filter, and ABR + Constructed Wetland.

Table E.1. Life cycle inventory for 15°C treatment of 2 MGD domestic wastewater.

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
ABR	Effluent BOD_5	mg/L	81	-	-	-
	Effluent dissolved CH_4	mg/L	29	-	-	29
Trickling filter	Pump, electricity	kWh/d	-	307	307	-
	Fan, electricity	kWh/d	-	24	24	-
	Effluent BOD_5	mg/L	-	30	30	-
Constructed wetland	Sequestered carbon dioxide	kg	-	-	-	343
	Land occupation	ha	-	-	-	3.8
	Polypropylene liner	kg	-	-	-	3.60 E+04
	Effluent BOD_5	mg/L	-	-	-	30
Anaerobic digestion	Mixing, electricity	kWh/d	-	58	20	-
	Heat losses	MJ/d	-	4.82 E+02	2.21 E+02	-
	Heat consumed	MJ/d	-	2.13 E+03	7.23 E+02	-
	Construction, AD plant for sewage sludge	piece	-	5.91 E-02	2.01 E-02	-
Cogeneration	Electricity generated	kWh/d	1.35 E+03	1.08 E+03	1.72 E+03	1.35 E+03
	Heat generated	MJ/d	7.26 E+03	5.56 E+03	9.15 E+03	7.26 E+03
	Construction, heat and power cogeneration unit, 160 kWe	piece	3.54 E-01	2.83 E-01	4.50 E-01	3.54 E-01
Dewatering, sludge drying bed	Land occupation	m^2	741	-	-	741
	Sand	kg	1.82 E+04	-	-	1.82 E+04
	Gravel	kg	3.81 E+05	-	-	3.81 E+05
Dewatering, belt filter	Electricity	kWh/d	-	89	36	-
	Acrylonitrile (polymer)	kg/d	-	10	4	-

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
Sludge disposal (incineration only)	Sludge treated, municipal incineration	kg/d	244	1109	501	244
	Sludge transport, freight, lorry	kg*km/d	1.22 E+04	5.55 E+04	2.51 E+04	1.22 E+04
Sludge disposal (landfill only)	Solid waste treated, sanitary landfill	kg/d	244	1109	501	244
	Sludge transport, freight, lorry	kg*km/d	2.44 E+04	1.11 E+05	5.01 E+04	2.44 E+04
Sludge disposal (land application only)	Nitrogen fertilizer avoided	kg/d	4	20	9	4
	Phosphorus fertilizer avoided	kg/d	4	16	7	4
	Quicklime	kg/d	49	222	100	49
	Electricity, consumed	kWh/d	14	106	43	14
	Diesel, from crude oil, consumption mix	kg/d	0.2	1.3	0.5	0.2
	Solids for land application	kg/d	244	1109	501	244
	Sludge transport, freight, lorry	kg*km/d	1.22 E+04	5.55 E+04	2.51 E+04	1.22 E+04

Table E.2. Life cycle inventory for 20°C treatment of 2 MGD domestic wastewater.

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
ABR	Effluent BOD_5	mg/L	60	-	-	-
	Effluent dissolved CH_4	mg/L	27	-	-	27
Trickling filter	Pump, electricity	kWh/d	-	307	307	-
	Fan, electricity	kWh/d	-	24	24	-
	Effluent BOD_5	mg/L	-	20	15	-
Constructed wetland	Sequestered carbon dioxide	kg/d	-	-	-	343
	Land occupation	ha	-	-	-	3.8
	Polypropylene liner	kg	-	-	-	3.60 E+04
	Effluent BOD_5	mg/L	-	-	-	20
Anaerobic digestion	Mixing, electricity	kWh/d	-	57	16	-
	Heat losses	MJ/d	-	3.62 E+02	1.65 E+02	-
	Heat consumed	MJ/d	-	1.57 E+03	4.41 E+02	-
	Construction, AD plant for sewage sludge	piece	-	5.82 E-02	1.64 E-02	-
Cogeneration	Electricity generated	kWh/d	1.35 E+03	1.07 E+03	1.65 E+03	1.35 E+03
	Heat generated	MJ/d	7.26 E+03	5.48 E+03	8.79 E+03	7.26 E+03
	Construction, heat and power cogeneration unit, 160 kWe	piece	3.54 E-01	2.78 E-01	4.32 E-01	3.54 E-01
Dewatering, sludge drying bed	Land occupation	m^2	741	-	-	741
	Sand	kg	1.82 E+04	-	-	1.82 E+04
	Gravel	kg	3.81 E+05	-	-	3.81 E+05
Dewatering, belt filter	Electricity	kWh	-	88	31	-
	Acrylonitrile (polymer)	kg/d	-	10	4	-
Sludge disposal (incineration only)	Sludge treated, municipal incineration	kg/d	252	1092	443	252
	Sludge transport, freight, lorry	kg*km/d	1.26 E+04	5.46 E+04	2.22 E+04	1.26 E+04
Sludge disposal (landfill only)	Solid waste treated, sanitary landfill	kg/d	252	1092	443	252
	Sludge transport, freight, lorry	kg*km/d	2.52 E+04	1.09 E+05	4.43 E+04	2.52 E+04

Process	Parameter	Unit	Average value, total			
			ABR	TF	ABR+ TF	ABR+ CW
Sludge disposal (land application only)	Nitrogen fertilizer avoided	kg/d	5	20	8	5
	Phosphorus fertilizer avoided	kg/d	4	16	6	4
	Quicklime	kg/d	50	218	89	50
	Electricity, consumed	kWh/d	15	105	37	15
	Diesel, from crude oil, consumption mix	kg/d	0.2	1.3	0.5	0.2
	Solids for land application	kg/d	252	1092	443	252
	Sludge transport, freight, lorry	kg*km/d	1.26 E+04	5.46 E+04	2.22 E+04	1.26 E+04

Appendix F: Comparison of TRACI 2.1 and IMPACT 2002+

TRACI 2.1 was compared to IMPACT 2002+ to gain insight into the potential limitations of and differences between the two methods. The midpoint impacts obtained for the ABR and Trickling Filter models at 15-20°C treatment were determined using IMPACT 2002+ (Figure F.1) and TRACI 2.1 (Figure F.2) and were plotted to show the relative magnitudes of the impacts for the ABR and Trickling Filter. The plot considers the sum of the ABR and Trickling Filter impacts to be 100 percent and presents the total as the percentages of the total impact corresponding to the ABR and Trickling Filter models. This approach allows for comparison of midpoint categories (i.e. carcinogens, non-carcinogens, respiratory effects) that are reported in different units by TRACI 2.1 than by IMPACT 2002+. Note that negative values indicate that the assembly achieved a net beneficial environmental impact for the impact category.

The comparison demonstrates two important points: (1) impact categories vary depending on impact assessment method and (2) impact weights may differ depending on impact assessment method.

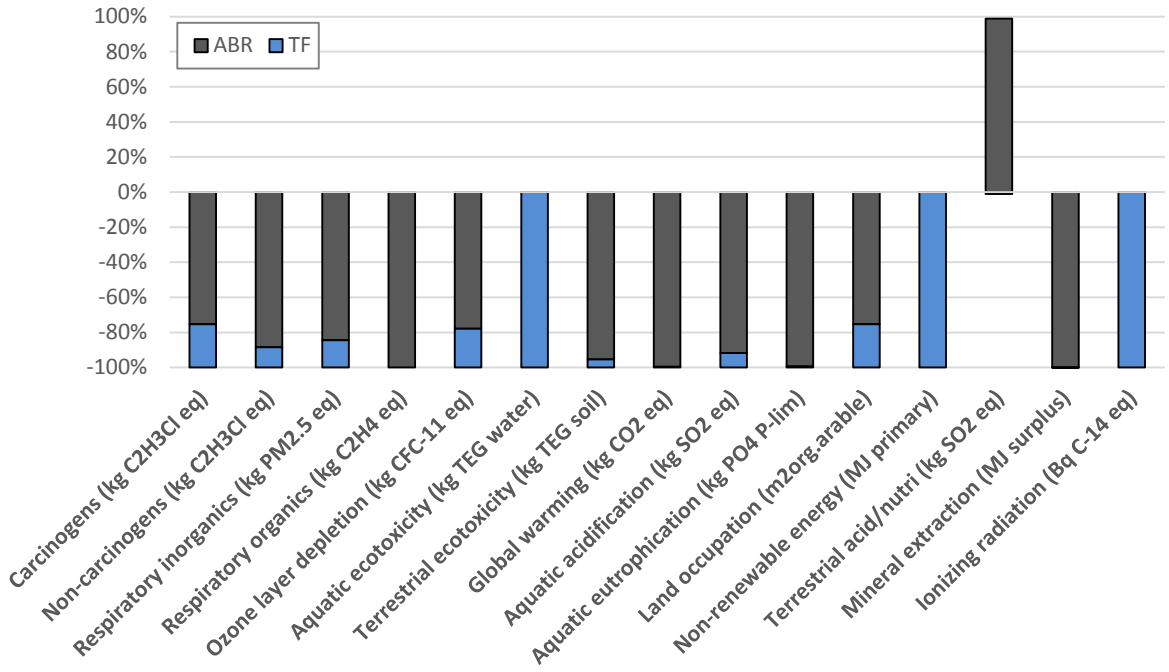

Figure F.1. Relative magnitudes of IMPACT 2002+ midpoint impacts for the ABR and Trickling Filter

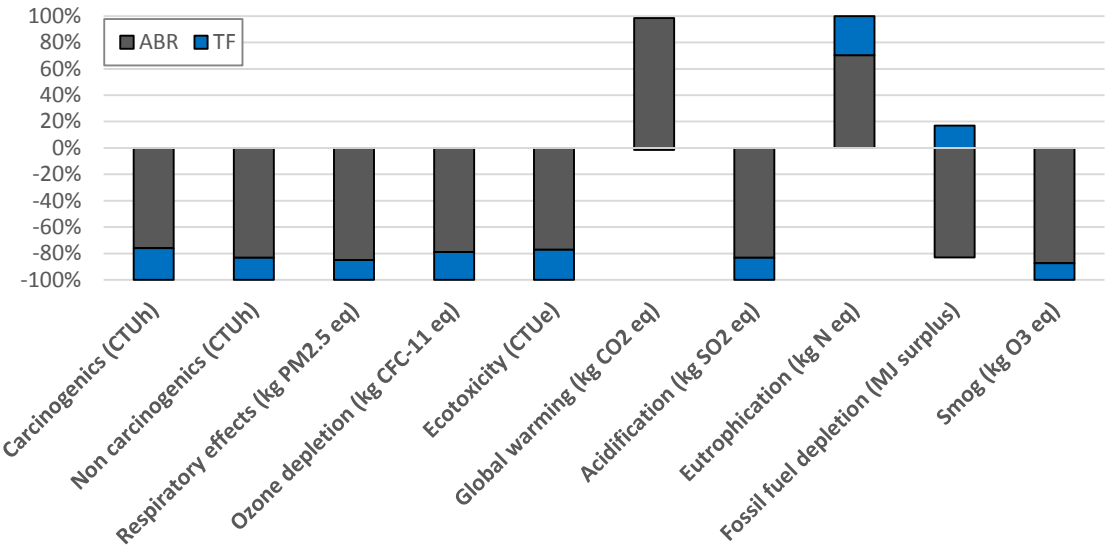

Figure F.2. Relative magnitudes of TRACI midpoint impacts for the ABR and Trickling Filter

IMPACT 2002+ calculates impacts that are not calculated in TRACI 2.1, such as land occupation, mineral extraction, and ionizing radiation. In addition, the comparison showed that a number of categories shared by the two methods (e.g. carcinogens, non-carcinogens, respiratory inorganics, ozone depletion) show similar relative magnitudes, which indicates that these impacts are calculated similarly for the two methods. The relative impacts for aquatic eutrophication and global warming, however, differ for TRACI and IMPACT2002+, suggesting that the two methods calculate the impacts differently. Further analysis showed that TRACI does not account for the impact of biogenic methane and that IMPACT 2002+ does not account for the impact of BOD_5, which explains the discrepancies between reported impact values for global warming and eutrophication.

Appendix G: Economic Modeling

Several design criteria (Table G.1) were adjusted from default values in CapdetWorks to more accurately describe the treatment detailed in Chapter 3 and input-output models described in Chapter 4.

Table G.1. Design criteria used to model costs with for CapdetWorks modeling software package.

Component	Model	Design Criteria	Value	Units
Influent	All	Average Flow	2	MGD
		Minimum Flow	2	MGD
		Maximum Flow	2	MGD
		Suspended Solids	233	mg/L
		Volatile Solids	91	%
		BOD	262	mg/L
		COD	576	mg/L
		SCOD	230	mg/L
		pH	7.4	-
		Average Temperature	15	°C
UASB (ABR)	ABR, ABR+TF, ABR+CW	Reactor Operating Temperature	15	°C
		COD Removal Efficiency	0.75	-
		Influent Solids Retention	0.2[a]	-
		Influent Sulphate	30	g/m^3
		Yield of Acidogens	0.16	-
		Yield of Methanogens	0.035	-
		Yield of Sulphidogens	0.057	-
		Reactor Volume	4164	m^3
		Construction Cost Multiplier	217[b]	$\$/m^3$
Drying beds	ABR, ABR+CW	Depth Applied	10	in
		Time to Drain	10	d
		Final Solids	60	%
Primary clarifier	TF	Design Basis	Average Flow	-
		Surface Overflow Rate	1000	$gal/(ft^2 \cdot d)$
		Sidewater Depth	14	ft
		Weir Overflow Rate	12700	$gal/(ft \cdot d)$
		Suspended Solids	61	%
		BOD	20	%
		COD	40	%
		Diameter	50	ft
Trickling filter	TF, ABR+TF	Solids Production Rate	0.45	lb VSS/lb BOD
		Hydraulic Loading Rate	30	$L/(m^2 \cdot min)$
		Specific Area of Media	90	m^2/m^3
		Diameter	16	m
		Tower Depth	6.1	m
		Number of Filter Towers	2	-

Component	Model	Design Criteria	Value	Units
Secondary clarifier	TF, ABR+TF	Design Basis	Average Flow	-
		Surface Overflow Rate	1.36	$m^3/(m^2 \cdot hr)$
		Sidewater Depth	4.3	m
		Max Weir Overflow Rate	15000	gal/(ft·d)
		Effluent Suspended Solids	30	mg/L
		Diameter	56	ft
Gravity thickener	TF, ABR+TF	Depth	3.5	m
		Based On	Mass Loading	-
		Mass Loading	2	$kg/(m^2 \cdot hr)$
Anaerobic digestion	TF, ABR+TF	Specific Gravity	1.02	-
		Percent Volatile Solids Destroyed	56	%
		Minimum Detention Time	15	d
		Raw Wastewater Temperature	15	°C
		Digester Temperature	35	°C
		Diameter	7 (TF), 4.1 (ABR+TF)	m
		Sidewater Depth	21.8	ft
Belt filter press	TF, ABR+TF	Cake Solids Content	22	%
		Hydraulic Loading	2.3	L/s
		Polymer Dose	0.55	% dry wt
Landfill (100%)	All	Travel Distance	100	km
		Vehicle Life	9	yrs
		Sludge Disposal	60	$/ton
Land application (100%)	All	Travel Distance	50	km
	All	Vehicle Life	9	yrs
	ABR, ABR+CW	Lime Addition	7863	$/yr
	TF	Lime Addition	32059	$/yr
	ABR+TF	Lime Addition	14483	$/yr
	ABR, ABR+CW	Revenue from Biosolids Fertilizer[c]	-3972	$/yr
	TF	Revenue from Biosolids Fertilizer[c]	-18053	$/yr
	ABR+TF	Revenue from Biosolids Fertilizer[c]	-8155	$/yr

[a]Value was modified to obtain solids production characteristic of an ABR
[b]Determined using: Multiplier=1783*Volume$^{-0.2528}$ (Hempseed, 2015)
[c]"Revenue from Biosolids Fertilizer" was represented by entering negative "costs" into "Annual Charge of Landfill" category since the landfill unit process was modified to reflect land application.