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Cost and Environmental Impacts of Leachate Nitrogen/Phosphorus Management Approaches

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COST AND ENVIRONMENTAL IMPACTS OF LEACHATE NITROGEN/PHOSPHORUS MANAGEMENT APPROACHES

by

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ABSTRACT

Landfill leachate is a challenging wastewater to discharge into municipal wastewater treatment plants (WWTPs), the most common approach for leachate management, due to the presence of contaminants that may affect the performance of the treatment plant. Treatment, disposal, and transportation of leachate are expensive and therefore a concern.

Currently, sidestream treatment is becoming increasingly common in WWTPs prior to returning the liquid to the plant influent. For this research, a new treatment scheme is introduced combining centrate and leachate to reduce contaminants, recover phosphorous and nitrogen through struvite precipitation, and reduce energy requirements through anaerobic ammonium oxidation (Anammox). By combining the two waste streams, the respective limited nutrients (nitrogen in centrate and nitrogen in leachate) can be removed in a low cost chemical treatment resources can be recovered. Carbon contaminants and remaining nutrients can be removed in subsequent innovative biological treatment units.

The objective of this thesis is to conduct a cost analysis and environmental assessment of the proposed novel treatment approach and to compare it to more traditional landfill on-site leachate treatment approaches (e.g., membrane bioreactors (MBR) and sequencing batch reactors (SBR)). The study was completed with the use of spreadsheet-based models. Spreadsheets have been developed to evaluate treatment costs (Capital + O&M) for both the proposed nutrient recovery/biological and traditional on-site leachate treatments. Transportation costs of leachate to the WWTP have been studied and analyzed by the use of a spreadsheet model as a function of distance.

Results suggest that treatment using Struvite – Aerobic Granular Sludge – Anammox

(SGA) was higher in cost compared to traditional approaches. However, positive outcomes from this process include: lower N_2O emissions, lower power consumption, struvite fertilizer, and overall recovery of nitrogen and phosphorus with the combination of centrate and leachate.

Dedicated to my parents,

To my younger siblings Amani and Marzouk,

Your constant love and support means the world to me.

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CHAPTER ONE: INTRODUCTION

1.1 Leachate Challenges

Landfill leachate management is becoming an environmental problem for the operation of sanitary landfills (Kulikowska et al., 2008). Due to the increase in waste volume that is growing faster than the world's population, municipal solid waste (MSW) management shapes a major problem worldwide (Renou et al., 2008). Managing leachate is important for various reasons, for instance; it may contaminate surface and groundwater, and may cause a risk to public health if improperly disposed. Treatment, disposal, and transportation of leachate are expensive and therefore a major concern. Landfill leachate is a challenging wastewater to discharge into municipal wastewater treatment plants (WWTPs), the most common approach for leachate management, due to its high concentrations of organic and inorganic contaminants that may affect the performance of the treatment plant (Wiszniowski et al., 2006). Municipal WWTPs cannot always treat the concentrated leachate to acceptable levels and that may cause issues between managers of landfills and local publicly owned treatment works (POTWs).

1.2 Common Sidestream Treatment

Currently, excess sludge produced during wastewater treatment is treated using anaerobic or aerobic digesters which destroy pathogens, reduce up to 50% of the sludge volume (Kotay et al., 2013) and reduce biochemical oxygen demand (BOD) (Holloway et al., 2007). The digested sludge is dewatered, generating a thickened digested sludge and liquid. This liquid removed during sludge dewatering is commonly known as centrate or filtrate depending on what process is used (Kotay et al., 2013). Centrate is produced during centrifugation of sludge and is rich in nutrients such as ammonia nitrogen ($NH_3 - N$) and phosphorus (P), along with high levels of chemical oxygen demand (COD) (Kotay et al., 2013; Wang et al., 2010). In conventional plants, centrate flows by gravity to storage tanks and the liquid stream is recycled back to the head of the plant for treatment. The recycled centrate causes 15-20 % of extra $NH_3 - N$ loading back to the plant (Fux et al., 2002; Holloway et al., 2007). This additional load of $NH_3 - N$ requires both aeration and addition of a readily biodegradable organic substrate (rbCOD), which contribute significantly to the energy and operational costs of the plant (Kotay et al., 2013). Furthermore, high P concentrations in the sidestreams create problems when returned to head of the plant (Münch et al., 2001), and therefore becomes a big problem and the impetus of this research. One of the options to deal with centrate is sidestream treatment. Currently, sidestream treatment (or sidestream returns) is increasingly common in WWTPs prior to returning it to the plant influent. Advantages of centrate sidestream treatment may include nutrient recovery and the reduction of energy and chemical use in the primary treatment process.

For this project, a new treatment scheme combining centrate and leachate to reduce contaminants (ammonia and phosphorous), recover P and nitrogen (N) through struvite precipitation, and reduce energy requirements through Anammox was proposed. The idea of combining both centrate and leachate at the treatment plant not only alleviates the problem of extra P, N and carbon (C) loading and toxicity but also provides a unique opportunity to recover useful resources such as nutrients and carbon. The complete Struvite – Aerobic Granular Sludge – Anammox (SGA) treatment scheme involves three processes as shown in Figure 1 (1) struvite precipitation of N and P, (2) an aerobic granular sludge process for the removal of N and organic carbon, and (3) attached growth anaerobic ammonia oxidation (Anammox) for N removal. The SGA process will be analyzed for cost and effectiveness.

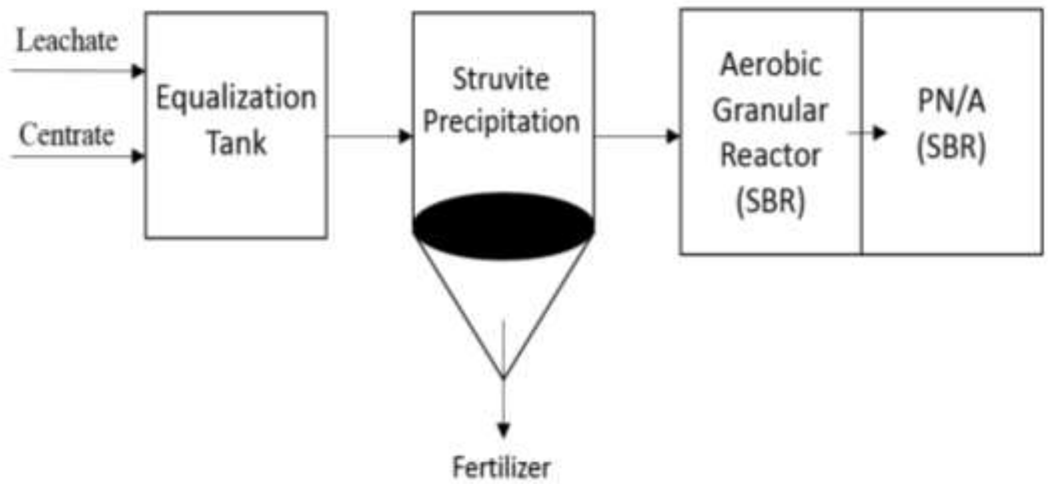

Figure 1: Complete Treatment Scheme for the SGA Process

1.3 Traditional Approaches

A similar analysis will be accomplished for sequencing batch reactors (SBR) and membrane bioreactors (MBR), which are considered more traditional approaches to treatment of leachate. The SBR process is a multiple stage operation including filling, reaction, settling, drawing, and idling (EPA, 1999). SBRs were chosen as one of the traditional approaches due to the fact that they are easy to operate, low in cost, and able to achieve high treatment efficiency. On the other hand, MBRs are a combination of conventional processes (i.e., activated sludge) and membrane filtration. MBRs are reported to achieve a high effluent quality and have smaller footprint than other biological processes, low sludge production, and high mixed liquor suspended solids (MLSS) tolerance (Ahmed et al., 2012).

1.4 Research Objectives

This project focuses on the idea of recovering useful resources and treating residual contaminants using innovative approaches for treating landfill leachate. The objective of this project task is to conduct a cost analysis and emissions assessment for the proposed novel treatment approach in comparison with more traditional on-site approaches (e.g., MBR and SBR). One of the main goals of this novel approach is sustainability through recovery of useful nutrients and reduced energy requirements. This project focuses on determining treatment design parameters, capital and operation and maintenance (O&M) costs, and environmental performance for the proposed treatment scheme aimed at nutrient recovery for combined centrate and leachate. Spreadsheet-based models have been developed to evaluate treatment costs for both the proposed nutrient recovery and traditional on-site leachate treatments as well as the cost of transport of leachate to the WWTP as a function of distance.

1.5 Thesis Outline

This thesis is structured into five chapters and is organized as follows:

Chapter 1 presents a general overview of the research; highlighting the research objective and research goals.

Chapter 2 provides a literature review that describes alternative on-site WWTP sidestream and leachate treatments. This is done to facilitate comparison of the proposed sidestream treatment and traditional on-site processes. Following this, research gaps are described.

Chapter 3 presents the methodology that includes justifying data used for this project and detailed steps of how these data were analyzed.

Chapter 4 presents the results and discussion that includes the main findings in this research.

Chapter 5 summarizes the findings of the research and includes recommendations for future research that can be accomplished. Lastly, supplemental materials will be found at the end of this thesis, with spreadsheet tables, design parameters, and costs.

CHAPTER TWO: LITERATURE REVIEW

In this chapter, an overview of leachate characteristics and recovery of useful nutrients is provided. This chapter was directed towards describing alternative on-site WWTP sidestream and leachate treatments. This was done to facilitate comparison of the proposed sidestream treatment with traditional on-site leachate treatment processes. In addition, this review includes research describing previous studies, along with research gaps at the end of this chapter.

2.1 Introduction

2.1.1 Landfill Leachate Generation

Landfilling is commonly used all over the world for the disposal of MSW and remains until this day the preferred practice of choice in most developed countries due to its economical advantages (Greedy, 2016). Leachate is generated by liquid coming from the disposed waste itself or originating as precipitation and passing through the waste layers. The most common approach for leachate management is the discharge of leachate into municipal WWTPs, however this approach has associated issues relating to impact on treatment efficiency and costs. Therefore, it is crucial to investigate new strategies to treat leachate without affecting municipal WWTP operations, standards, or energy consumption.

2.1.2 Characteristics of landfill Leachate

Many factors affect the quality of leachate including the type of waste (municipal, industrial, or hazardous), landfill condition (age, location, and recirculation), and climate (Renou et al., 2008). Leachate is usually characterized using five-day biochemical oxygen demand

(BOD_5), chemical oxygen demand, total organic carbon (TOC) (Kochany et al., 2009), BOD/COD ratio, $NH_3 - N$, total Kjeldahl nitrogen (TKN), turbidity, or heavy metals content (Foo et al., 2009). Landfilled waste and leachate changes over four different stages as time proceeds; 1) aerobic, 2) hydrolysis and fermentation, 3) anaerobic acetogenic, and 4) methanogenic (Ahmed et al., 2012; Foo et al., 2009). Because of the stages of organic waste degradation, leachate properties mentioned may vary widely. Table 1 shows the ranges for leachate parameters classified by three ages (Table 1 was generated based on ranges by Foo et al., 2009). BOD_5/COD ratio is often used as an indicator of the best treatment method for landfill leachate. For example, leachate containing a BOD_5/COD ratio less than 0.1 is considered stabilized and best treated using physical/chemical processes (Comstock et al., 2010). On the other hand, for young leachate with a BOD_5/COD ratio greater than 0.5, biological processes are the most appropriate treatment method, because higher fractions of biodegradable materials are present in fresh waste and young leachate (Comstock et al., 2010). Landfill leachate contains a combination of microbial, chemical, and physical pollutants (Kjeldsen et al., 2002). Leachate also contains xenobiotic organic compounds (Smith et al., 2013). These contaminants should be removed due to their toxic effect on the environment (Kjeldsen et al., 2002).

Table 1: Ranges for Leachate Parameters Classified by Three Age Stages

Type of Leachate	Young (<5years)	Intermediate (5-10 years)	Old (>10 years)
pH	<6.5	6.5-7.5	>7.5
COD, mg/l	>10,000	4,000-10,000	<4000
BOD_5/COD, unitless	0.5-1.0	0.1-0.5	<0.1
$NH_3 - N$, mg/l	<400	N.A.	>400
TKN, mg/l	0.1-0.2	N.A.	N.A.
Biodegradability	High	Medium	Low

2.1.3 Characteristics of Centrate

Centrate is a municipal wastewater stream that is highly concentrated and its characteristics differ from landfill leachate characteristics. Fattah et al. (2008) completed a study about centrate characteristics, which suggested that the characteristics were for the same WWTP, however since centrifugation is a batch process, the characteristics change every time the tank was filled. First study showed a pH, conductivity, phosphate-P ($PO_4 - P$), ammonium-N ($NH_4 - N$), and magnesium (Mg) concentrations of 7.3, 6.5 mS/cm, 60 mg/L, 780 mg/L, and 5.1 mg/L respectively (Fattah et al., 2008). Second study showed a pH, conductivity, $PO_4 - P$, $NH_4 - N$, and Mg concentrations of 7.6, 6.4 mS/cm, 60 mg/L, 720 mg/L, and 11 mg/L respectively (Fattah et al., 2008). Another study by Yecong et al. (2010) summarized two different types of centrate (raw and autoclaved centrate). Total nitrogen, total phosphorus, COD, ammonia, and TSS characteristics averaged at 120 mg-N/L, 220 mg-$PO_4 - P$/L, 2300 mg/L, and 0.07 respectively (Li et al., 2011).

2.2 Biological Treatment for Landfill Leachate

Many technologies exist for the treatment of landfill leachate including (1) biological processes (SBR, lagoons, and MBR), (2) discharge to municipal WWTPs, and (3) physical and chemical processes (air stripping, adsorption, and flocculation/coagulation) (Torretta et al., 2016). Biological treatment has gained attention due to its relatively low cost and ease of operation. This section focuses on current biological treatment technologies used for leachate treatment that include, SBRs, and MBRs. The application of these processes is for the removal of organics before the leachate is discharged to the environment. In addition, this section describes anaerobic ammonia oxidation, also known as Anammox, low-energy biological treatment for the removal of nitrogen from wastewater.

2.2.1 Sequencing Batch Reactor Process

A SBR utilizes a fill-and-draw activated sludge system to treat landfill leachate and wastewater (Vigneswaran et al., 2009). SBRs consist of a single tank with multiple stage operating processes. SBRs are operated under non-steady state flow conditions, and are considered flexible because they work in a time rather than a space sequence (Laitinen et al., 2006). Main advantages of SBRs include their ease in operation, low cost, and high organic removal efficiencies. Although SBRs have many advantages, there are some challenges accompanied with them, such as high-energy consumption, high level of maintenance (automated switches, and automated valves) required, and the need for equalization after the SBR (Aziz et al., 2013) . SBRs have minimal footprint and are suited for low flows. Usually, treatment systems have more than one SBR tank for redundancy (Vigneswaran et al., 2009).

2.2.1.1 Basic SBR Treatment Process

SBRs operate with a sequence of stages (phases) including filling, reacting, settling, drawing, and idling (Vigneswaran et al., 2009). The treatment process starts with filling the reactors with untreated wastewater, in this case, leachate. In this phase, the feed amount is based on the desired hydraulic retention time (HRT), food to microorganism ratio (F/M), and loading rate (Aziz et al., 2013). Following the completion of the filling phase, the react phase begins. During the react phase, continuous aeration is supplied to remove organic contaminants and to convert ammonium to nitrate (nitrification) and under unaerated conditions, nitrate and nitrite are converted to nitrogen gas (denitrification). The react phase can take up to 50% of the entire cycle time (Aziz et al., 2013). Thereafter the settling phase separates biosoilds from a clear layer known as supernatant. In this phase, the clear supernatant appears on the top, whereas the MLSS is settled to the bottom (Aziz et al., 2013). During the draw and decant phase, the effluent is discharged from the reactor through a withdrawal mechanism (Vigneswaran et al., 2009). The phase between draw and fill is known as idle. The idle time can be used to waste settled sludge to control the sludge retention time. Along with that, the idle phase can be eliminated if more than two SBRs are present and the tanks are operated with staggered fixed cycle times. After a phase of idle, the reactor is filled again with the wastewater (Aziz et al., 2013).

2.2.1.2 Applications of Sequencing of Batch Reactor to Treat Landfill Leachate

Many studies have been conducted on landfill leachate treatment using SBRs. Most of these studies focused on organics and N removal, such as the study by Uygur et al. (2004). In this study powdered activated carbon (PAC) was added to enhance nitrification efficiency in the biological treatment of leachate (Uygur et al., 2004). COD, $NH_4 - N$, and $PO_4 - P$ removals from the pre-treated leachate and domestic wastewater were 75%, 44%, and 44% with the addition of PAC, respectively. On the other hand, COD, $NH_4 - N$, and $PO_4 - P$ removals in the absence of PAC were 64%, 23%, and 26% respectively. Results indicate that the addition of PAC can improve nutrient removal significantly (Uygur et al., 2004).

In a study done by Lo (1996), three treatment trials for methanogenic leachate in Hong Kong, China using SBRs were conducted to study their treatment efficiencies. Leachate samples were taken from two different landfills; two samples from an active landfill site and one from a closed landfill. Both trials were operated with both a HRT of 20 and 40 days. This study showed that with HRTs of 20 and 40 days, high removal efficiencies for both COD and $NH_3 - N$ in SBRs could be possible (Lo, 1996). Therefore, SBRs are considered to be well suited for leachate treatment because they are able to handle leachate high variability in quantity and quality. Table 2, summarizes major removal efficiencies for full-scale on-site landfill leachate treatments using SBRs.

Table 2: SBR Removal Efficiencies from Landfill Leachate

Reactor type	SRT	HRT (Days)	COD (%)	BOD (%)	NH_4-N	NH_3-N	Source(s)
Full-scale SBR	N/S	1.9-5	N/S	63.3-95	98.87	91.5%	(Morling, 2010)
Full-scale SBR	N/S	N/S	N/S	88.4-98	99.4	95%	(Morling, 2010)
Lab-scale SBR	1 day	2.5	90.5	92.6	N/S	N/S	(Perera et al., 2014)
Full-scale SBR	N/S	N/S	60%	N/S	N/S	99%	(Robinson, 2017)

- N/S = not specified

2.2.2 Membrane Bioreactor Process

MBR is a biological process used for leachate and wastewater treatment. It consists of a combination of conventional processes (i.e., activated sludge) and membrane filtration (Kraume et al., 2010). MBRs commonly operate with equipment such as ultrafiltration (UF) or microfiltration (MF) membranes, where hollow fiber, flat sheet, or tubular membranes are most commonly chosen (Ahmed et al., 2012). There are two main MBR configurations, submerged MBRs (immersed) and sidestream MBRs (external) (Ahmed et al., 2012). Submerged membranes are located inside the reactor, whereas sidestream membranes are located in a separate cell. In sidestream MBRs, high velocities should be maintained to overcome flux decline due to fouling. Submerged MBRs are more compact, save energy, and are low in cost because they do not require high-flow recirculation pumps (Ahmed et al., 2012), therefore submerged systems are more frequently used for treatment applications. Kraume et al. (2010) state that MBRs are expected to grow in use; their value was expected to increase from $296 million in 2008 to $488 million by 2013 (Kraume et al., 2010).

2.2.2.1 Applications of Membrane Bioreactor to Landfill Leachate Treatment

A study was conducted by Wilkinson et al.(2010) to pretreat landfill leachate with MBR technology to remove ammonia and total dissolved solids (TDS). The study took place in New Jersey, U.S. (the pollution control financing authority of Warren County) when a WWTP could no longer accept the leachate without pretreatment due to increased production of leachate and ammonia concentration. Technology for leachate on-site pretreatment was evaluated, including nitrifying activated sludge (SBR), ammonia stripping, and MBR. Due to the temperature considerations and future concerns such as TDS accumulation, MBR was the selected technology (Wilkinson et al., 2010). The MBR consists of one anoxic tank (for future denitrification requirements) and two aerobic tanks. Values of $NH_3 - N$ went from 940 mg/L to 34 mg/L during treatment, which means almost 97% reduction was observed (Wilkinson et al., 2010). Furthermore, minimal cleaning was needed for ultrafiltration membranes; they only required cleaning once during a ten-month operating period. Therefore, results reported that the MBR effectively removed ammonia and other permitted constituents with all criteria met and minimal operator attention (Wilkinson et al., 2010).

In a study conducted by Laitinen et al., both a SBR and MBR were evaluated for the treatment of landfill leachate. Both SBR and MBR were operated in a nitrification/ denitrification tank with different operational conditions. The leachate was analyzed for pH, COD, BOD_5, total N and P, and total ammonia-nitrogen. The SBR was operated with a HRT of 4 to 8 days, whereas, the MBR was around 3 days (Laitinen et al., 2006). Effluent from the SBR had high-suspended solids, BOD, and turbidity, which means that sludge was escaping from the SBR unit. It was observed that 94% of BOD_7, 99.5% of ammonia nitrogen, phosphorus up to 82%, and 89% of suspended solids reductions were achieved in the SBR. On the other hand, over 99% of BOD_7, 99% of suspended solids, over 97% of ammonia nitrogen, and over 88% of phosphorus were removed in MBRs (Laitinen et al., 2006). It has been shown in this study that both MBR and SBR can effectively remove $NH_3 - N$.

2.2.3 Partial- Nitritation/Anammox (PN/A)

2.2.3.1 Overview

Conventional plants use nitrification/denitrification for the removal of nitrogenous compounds especially ammonia, which is expensive due to the oxygen needed for nitrification, and carbon source often required for denitrification (Biec et al., 2014; Sun et al., 2017). Compared to conventional nitrification/denitrification processes, the Partial-Nitritation/ Anammox (PN/A) process in recent years has been found to more efficiently remove nitrogen (Biec et al., 2014). Along with that, PN/A consumed less than 50% of oxygen supply, and no organic carbon source was needed (Biec et al., 2014).

Nitrogen removal from wastewater is important for the aquatic environment, because of eutrophication and acidification problems that can develop from excess nitrogen (Sun et al., 2017). Anammox is considered a practical option as a sidestream treatment for the removal of nitrogen from the liquor generated during the dewatering of anaerobically digested sludge (Kotay et al., 2013). In the PN/A reactor nitritation and Anammox conversion of ammonia occur simultaneously in one single process unit. The Anammox process is considered a shortcut to the nitrogen cycle. Ammonium is converted to nitrogen gas with nitrite as the electron acceptor under anoxic conditions, as shown in Equation 1 (Fux et al., 2002; Metcalf et al., 2014):

$$\mathrm{NH_4}^+ + 1.32\mathrm{NO_2}^- + 0.066\mathrm{HCO_3}^- + 0.13\mathrm{H}^+ \rightarrow 1.02\mathrm{N_2} + 0.26\mathrm{NO_3}^- + 0.0666\mathrm{CH_2O_{0.5}N_{0.15}} + 2.03\mathrm{H_2O} \quad (1)$$

As stated by Kotay et al., the Anammox reaction consumes a nitrite to ammonium ($\mathrm{NO_2-N}$ to $\mathrm{NH_3-N}$) ratio of 1:1 to 1.7:1 (Kotay et al., 2013). Therefore, Anammox faces challenges with treating ammonia-rich centrate because of low concentrations of $\mathrm{NO_2-N}$

present in centrate. Furthermore, in order for the Anammox reactor to treat the centrate, an addition of $NO_2 - N$ with the influent, or partial nitritation is needed to generate nitrite by ammonia oxidizers (Kotay et al., 2013). Partially oxidized ammonium to nitrite (partial nitritation) is shown in Equation (2) (Metcalf et al., 2014):

$$2.34\ NH_4^+ + 1.87O_2 + 2.66HCO_3 \rightarrow 0.02C_5H_7NO_2 + NH_4^+ + 1.32NO_2 + 2.55CO_2 + 3.94H_2O \tag{2}$$

2.2.3.2 Single-Stage Partial Nitritation/Anammox Granular Sludge Bioreactor

Sun et al. (Sun et al., 2017) conducted a study of a single stage PN/A process using a sequencing batch biofilter granular (SBBGR) for the treatment of ammonia-rich reject water. The study was conducted for more than 100 days and was divided into two parts: phase 1 where influent ammonia was 100 mg/L, and phase 2 where influent ammonia was 200 mg/L. During phase 1, from day 1 to 36, almost 94% of ammonia removal along with 81% removal efficiency of total nitrogen was achieved. During phase 2 (36-105 days), ammonia removal up to 92% and total nitrogen removal of more than 80% occurred (Sun et al., 2017). These analyses indicated a successful setup for PN/A using SBBGR. Another study by Rodriguez et al. (2016) established a single PN/A granular sludge bioreactor at low temperatures. Results indicated that PN/A granular sludge bioreactor could effectively remove nitrogen at low temperatures (15℃) (Rodriguez-Sanchez et al., 2016).

2.2.3.3 Applications of Partial Nitritation/Anammox for Landfill Leachate

A study by Zhang et al. (2017), investigated the COD and nitrogen removal efficiency of simultaneous partial nitrification, Anammox, and denitrification (SNAD) for landfill leachate treatment in a single SBR. The SNAD process involves ammonia oxidizing bacteria (AOB) (Miao et al., 2018), which oxidize ammonia to nitrite (Zhang et al., 2017), while the remaining ammonia and nitrite was converted to nitrogen gas by denitrifiers (Miao et al., 2018). The SBR was run with intermittent aeration for more than 120 days. Intermittent aeration is considered a promising method for preventing nitrite- oxidizing bacteria (NOB) growth, where nitrite is oxidized to nitrate under aerobic conditions (Zhang et al., 2017). Results showed that the SNAD process achieved 99.3% removal for total nitrogen, and 99.4% removal of NH_4. Dissolved oxygen (DO) parameters were used to control the duration of aeration, and the results showed that a SBR operated under intermittent aeration could improve nitrogen removal from mature landfill leachate (Zhang et al., 2017). PN/A has been regarded a cost-saving alternative technology to conventional biological nitrogen removal via nitrification and heterotrophic denitrification (Miao et al., 2018).

2.2.3.4 Large-Scale Applications of Anammox

Anammox has been widely studied at laboratory scale, but has been limited in full-scale applications. However, nowadays the process is better understood and therefore its use is increasing. In Alexandria, Virginia, US (Alexandria Renew Enterprises), a full-scale sidestream Anammox system is treating centrate. This facility uses the Anammox process to promote short-cut nitrogen removal by bacteria known as red bugs (Riper, 2015). They chose this process because it will reduce supplemental chemical addition and energy consumption. AlexRenew has been operating a centrate pretreatment facility since early 2015, and has shown impressive results of 85% total nitrogen removal at a facility that treats 276,000 gal centrate per day (Riper, 2015). An Anammox reactor has been in operation at the sludge treatment plant at Sluisjesdijk, Rotterdam, NL since 2002 (van der Star et al., 2007). Anammox is used for the treatment of the reject water from sludge digestion. Paques developed the process in cooperation with Delft University of Technology and University of Nijmegen in the Netherlands. Paques states that compared to conventional nitrification/denitrification, Anammox can save up to 60% on operational costs (Paques, 2018).

2.3 Recovery of Useful Nutrients

Conventional biological processes, while effective in removal of nitrogen, do not allow for the recovery of nutrients and also significant energy is consumed. Recently management of landfill leachate and municipal wastewater has been increasingly focused on recovery of nutrients rather than wasting these important resources. Nutrients such as nitrogen and phosphorus are both compounds found in waste streams, which are essential for various life forms. These two nutrients play an important role in both food supply and plant growth and are supplied by the use of synthetic fertilizers (Sengupta et al., 2015). Nitrogen is found in large quantities in the atmosphere (78%) in a highly stable form of gas (N_2), however, it is found in limited quantities in soils (Sengupta et al., 2015).

On the other hand, phosphorus is a non-renewable limited resource that is becoming increasingly scarce and expensive (Sengupta et al., 2015). Research has shown that by early 2035, the lack of phosphorus will lead to increased pricing and global disputes (Batstone et al., 2015). Additionally, eutrophication will result if the discharge of phosphorus and nitrogen into the environment is not controlled (Marti et al., 2017). Recovery of nitrogen and phosphorus, in a form of valuable products, is important and has gained considerable attention. Additionally, phosphorus recovery can generate local supplies of phosphorus fertilizers (Marti et al., 2017).

Prices of phosphorus have gone from $2000/tonne in 2009 to $4000/tonne in 2015 (Batstone et al., 2015).

2.4 Phosphorus Recovery Through Struvite Precipitation

2.4.1 Background

In the early 1960s, a WWTP in Los Angeles, California, was dealing with an extensive operational problem due to the discovery of a white crystalline substance that had deposited in the digested sludge pipes (Stratful et al., 2001). Subsequently, many studies in literature have reported similar problems associated with the white crystalline solid. It was found that the white crystalline substance was an inorganic mineral commonly known as struvite or magnesium ammonium phosphate hexahydrate (MAP, $MgNH_4PO_46H_2O$) (Stratful et al., 2001). Struvite formation is shown in Equation 3 (Kochany et al., 2009):

$$Mg^{2+} + NH_4{}^{+} + H_nPO_4{}^{n-3} + 6H_2O \rightarrow MgNH_4PO_4 * 6H_2O + nH^{+} \tag{3}$$

Struvite deposition was occurring in places with decreased turbulence (Stratful et al., 2001), resulting in clogged pumps and pipes, that led to operational difficulty in the plant. Several remediation options were proposed to deal with the problem; however, processes were either time consuming or too complex to be considered an ideal option.

2.4.2 Struvite Precipitation

When intentionally applied, struvite precipitation is an effective process that Yetilmezsoy et al. (2017) state is easy to implement and is a high-yield physicochemical treatment method for the removal of both nitrogen and phosphorus from wastewater. Struvite precipitation occurs when concentrations of phosphate, magnesium and ammonium ions result in a supersaturated solution (Kochany et al., 2009). Factors that affect the formation of struvite precipitation and should be taken into consideration include pH, temperature, reaction time, and other ions present in the solution (Fattah, 2012). A study conducted by Stratful et al. (2001) illustrated the conditions that influence the precipitation of phosphate. In the study, sodium hydroxide (NaOH) was added to control pH. It was proven that at pH of 8.5, 85% of phosphorus was incorporated into the crystals (Stratful et al., 2001). A trend between pH and the removal of both magnesium and phosphorus was observed. More than 97% of magnesium was removed at a pH of 9, 9.5 and 10, however residual phosphate remained at 12% of its original concentration (Stratful et al., 2001), which ultimately demonstrates that in order to obtain effective struvite precipitation, a pH between 8.5 and 10 is required. Another study performed by Li at al. (1999) reported that struvite precipitation is most effective between pH values 8.5 and 9.0. Wastewater is normally within a pH range of 6 to 8 (Stratful et al., 2001) but based on previous studies, a pH of 8.5 or higher is required for effective struvite removal and thus an additive would be required to adjust the pH levels, such as NaOH or magnesium oxide (MgO).

In Kyoto, Japan, sidestream struvite crystallization from the digested sludge dewatering system, centrate, has been applied on a large-scale (Ueno et al., 2001). The goal of the plant was to recover phosphate by struvite from a WWTP and to produce a phosphorus-rich material in order to sell it as a fertilizer. A pH range of 8.2 to 8.8 was established in the plant, with the addition of NaOH, along with the addition of magnesium hydroxide $(Mg(OH)_2)$ so that the magnesium to phosphate ratio became 1:1 (Ueno et al., 2001). The influent phosphorus concentration was 110 mg/l. After treatment, a concentration of 10 mg/l was achieved (Ueno et al., 2001). Results showed that the plant was capable of removing over 90% of the phosphorus.

Particles ranging in size from 0.5 to 1 mm were achieved with a retention time of 10 days. Fine granular struvite must be recycled to the reactor influent and used seeding material. The final product was sold to a fertilizer company for 27,000 yen/tonne (approximately $258/tonne) (Ueno et al., 2001), which included cost of transportation. In Japan, this fertilizer was used on vegetables, paddy rice, and flowers.

Over the last decade, struvite has become well known as a method of removal and recovery of phosphorus from wastewater. However, recently it has also been proposed that $NH_3 - N$ can be reduced by struvite precipitation from landfill leachate (Kochany et al., 2009). Municipal leachate contains low concentrations of magnesium and phosphorus, compared to high concentrations of ammonium (Di Iaconi et al., 2010). A study was conducted by Di Iaconi et al. (2010) to recover nitrogen from landfill leachate through struvite precipitation. Phosphoric acid was the external source used in this study as well, because phosphoric acid is lower in cost compared to other phosphorus salts. MgO was used as the magnesium source (Di Iaconi et al., 2010). In this study, although ammonia was removed, the addition of chemicals was expensive (Di Iaconi et al., 2010). Furthermore, the addition of phosphorus is not a sustainable practice, since phosphorus is a limited resource that is becoming increasingly scarce and expensive.

2.4.3 Agricultural Use of Struvite

Although struvite can be a problem for WWTPs, over the last decade, struvite has become a well-known fertilizer product. It is well known that phosphorus fertilizers are important for modern agriculture. There is an increasing demand for phosphorus fertilizers in some countries in Africa for example, because of the lack of phosphorus in soils (Shokouhi, 2017). Furthermore, as population increases, the demand for fertilizer is increasing in countries such as China and India. These populated countries consume around 14.6% of the global annual phosphorus fertilizer (Shokouhi, 2017).

Greenhouse gases (GHG) trap heat in the atmosphere. The GHGs carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are key elements for global warming, and currently, global efforts to reduce GHGs are taking place (Rahman et al., 2014). Agricultural soil gas emissions are small, CH_4 and N_2O are major emissions during agricultural practices. The total CO_2, CH_4 and N_2O global emissions from agriculture is 1%, 39%, and 60% respectively (Parris, 1996; Rahman et al., 2014). Nitrogen fertilization is the main emission source of CH_4, N_2O and nitric oxide (NO) from the soil (Rahman et al., 2014). The application of struvite as a fertilizer may reduce the risk of global warming, despite the fact that it only contains 6% nitrogen (Rahman et al., 2014). Struvite fertilizers have slow nutrient releasing characteristics for which minimize N_2O emission from soil (Rahman et al., 2014). Urea is commonly used as a nitrogen fertilizer, and is considered the most prevalent form of nitrogen fertilizer used (Liang et al., 2007). Urea emits large amounts of N_2O, and therefore, struvite can be an alternative to traditional nitrogen fertilizer and can help reduce GHGs (Rahman et al., 2014).

2.5 Research Gaps

Previous literature is missing comprehensive studies of combining two waste streams, centrate and leachate. Studies such as Di Iaconi et al. (2010) recovered struvite from landfill leachate; however, the addition of chemicals such as phosphoric acid and magnesium oxide was expensive (Di Iaconi et al., 2010). Furthermore, the addition of phosphorus is not a sustainable practice, because phosphorus is a limited resource and is becoming increasingly scarce and expensive. On the other hand, centrate contains relatively high concentrations of phosphorus when compared to landfill leachate. By combining the two waste streams, potential economic and environmental benefits may occur.

Struvite precipitation along with PN/A has been widely studied at laboratory scale. However, full-scale applications are limited, and therefore cost data are needed. Furthermore, no studies have investigated the combination of struvite precipitation, aerobic granular sludge process and PN/A process. Previous literature is missing studies regarding cost analysis for these processes as a sidestream treatment. However, in this research these gaps will be addressed.

CHAPTER THREE: METHODOLOGY

3.1 Overview

The purpose of this research is to apply treatment design parameters to determine O&M and capital costs, and environmental performance for the proposed treatment scheme aimed at nutrient recovery for combined centrate and leachate. Spreadsheet-based models have been developed to evaluate treatment costs for both the proposed treatment/nutrient recovery and traditional on-site leachate treatment processes.

This project is a collaborative research with University of Utah, thus, the source of experimental data for this study. A centrate to leachate ratio of 4:1 has been recommended by The University of Utah. For the purpose of this analysis, a study design for a city of 100,000 people was utilized. It is assumed that a city of this size generates around 26500 m^3 (7,000,000 gallons) of wastewater. Centrate from the digested sludge is typically 0.3% to 1.5% of the total WWTP flow (Pedros et al., 2008). The average daily flowrate of centrate was than calculated to be 400 m^3 per day (105,000 gal/day). Given the ratio of leachate to centrate, the average leachate flowrate was than calculated to be 99 m^3/day (26300 gal/day). Spreadsheets were setup as seven separate cost work sheets consisting of: 1) leachate transportation, 2) SBR, 3) MBR, 4) PN/A process, 5) struvite crystallization, 6) aerobic granular sludge process, and 7) an equalization tank for blending centrate and leachate. For each individual cost analysis, capital and O&M requirements were calculated. Capital costs included construction, design, electrical and instrumentation, structural, civil, and piping as well as installation. On the other hand, O&M costs included power, chemical addition, labor and maintenance, in addition to monitoring and testing. Biological treatment, filtration and disinfection are common to all treatment scenarios and therefore costs are not included in this estimate. Capital and O&M costs that are associated with sludge handling and disposal for all three scenarios (SBR, MBR, and SGA) are not included as well. This chapter describes the approaches followed in the development and completion of the spreadsheet models.

3.2 Leachate Transportation Cost

For the purpose of this research, leachate was assumed to be collected from a local landfill and transported to a municipal WWTP by a tanker truck. Transportation costs are important to determine the most cost effective approaches, either by transporting leachate or treating it on-site. A spreadsheet model was developed for estimating the cost per m^3 based on the vehicle capacity. The study analyzed transportation cost using three different truck capacities, water tanker trucks with load capacities of: 7.57 m^3(2000 gal), 15.1 m^3 (4000 gal), and 18.9 m^3 (5000 gallons). Table 3 shows the purchase cost associated with each truck capacity provided by Ledwell Company.

Table 3: Cost of Water Tanker Trucks for Each Capacity ("Ledwell," 2018)

Truck Capacity (m^3)	Cost of Truck *
7.57	$82,000
15.14	$130,000
18.93	$150,000

- 2018 costs

Transportation costs were divided into two categories: fixed cost and variable costs. Fixed cost components included vehicle ownership, insurance, and vehicle registration. On the other hand, the variable cost components included maintenance and repairs, fuel cost, cost of tires, and labor costs (Marufuzzaman et al., 2015). A summary breakdown of the parameters used to calculate unit transportation costs is provided in Table 4.

Table 4: Summary Breakdown of Parameters Used to Calculate Total Transportation Cost

Unit Transportation Cost	Source
• Truck Capacity (m^3)	Table 3
• Work Schedule (trips/day)	Calculated
1) Annualized Fixed cost	
• Equivalent uniform annual cost of truck ownership ($/year)	Equation 5
2) Fixed cost	Section 3.2.1
• Insurance Cost	
• Vehicle registration and fees	
3) Variable Cost	Section 3.2.2
• Fuel ($/year)	
○ Fuel cost ($/liter)	
○ Total distance traveled in (km/day)	
○ Total km per liter	
• Maintenance and repairs ($/year)	
• Tires ($/year)	
• Labor cost ($/year)	
○ Work Schedule (hrs/day)	
○ Total Time at work (hr/day)	
○ Salary ($/hour)	
4) Summary	
• Total Transportation Cost ($/$m^3$)= **Fixed cost + Annualized Fixed cost + Variable Cost**	

3.2.1 Fixed Costs

To determine uniform annual fixed cost, the number of trips needed and the total number of possible trips made by a single truck per day for various distances were calculated. This was done to calculate the number of trucks needed per day as a function of distance. After that, the cost of the truck is obtained by multiplying the costs shown in Table 3 with the number of trucks needed. The possible number of trucks is calculated as shown in Equation 4 based on Figure 2:

$$\text{Daily hours of work} = A + B + (\text{Daily hours of work} \times e) + ((C \times 2) + D) \times X \quad (4)$$

Where,

X= Number of possible trips

A= Time to the parking lot

B= Time it takes from garage to landfill

C= Time to/from landfill to WWTP

D= Loading and unloading time

e= off route time

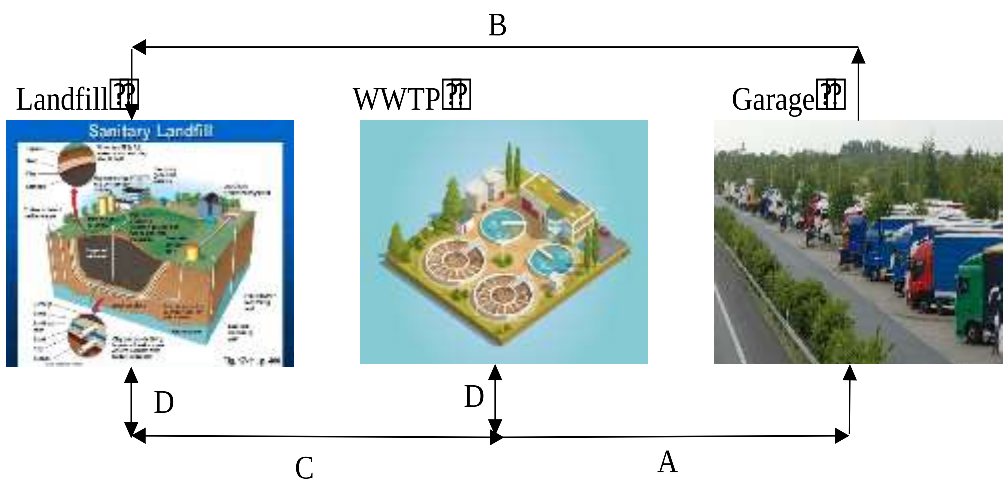

Figure 2: Leachate Hauling Truck Route

The number of trips possible is determined as a function of distance using Equation 4; six different distances were chosen, 15, 20, 25, 30, 60, and 100 km. From these distances and an assumed speed of 65 km/hr, the time to/from the landfill to the WWTP was determined (C). It was also assumed that the off route time was 10% of the daily hours of work and the loading and unloading time was 40 minutes (Driest, 2014), the time to the parking lot was 5 minutes, and also a 15 minute time for the truck to move from the garage to the landfill. Time was obtained from Google maps for the distance between a WWTP and a landfill in Orlando, Florida. Equivalent uniform annual cost of truck ownership ($/year) was calculated for an estimated truck life of ten years with a 5% annual interest rate (van den Boomen et al., 2018) as shown in Equation 5:

$$\mathrm{EUAC} = \mathrm{NPV}\left[\frac{\mathrm{i}(1+\mathrm{i})^{\mathrm{n}}}{(1+\mathrm{i})^{\mathrm{n}}-1}\right] \tag{5}$$

Where,

EUAC= Equivalent Uniform Annual Cost

NPV= Present value ($)

i= annual interest rate,

n= truck life in years.

Insurance and vehicle registration are important factors of fixed costs. Vehicle registration was based on Florida Highway Safety and Motor Vehicles website, which is $251 per year (Rhodes, 2018). These costs include initial registration fee based on the truck weight. Truck insurance can vary widely depending on insurance companies, miles traveled, and vehicle age. Truck insurance cost was determined to be $6500 per year.

3.2.2 Variable Costs

Labor and fuel are the most important variable costs, any changes to them strongly affect the final transportation cost. The first step to calculate variable cost is by estimating labor time using Equation 6:

$$\text{Total work time (hr/day)} = \mathrm{A} + \mathrm{B} + (\text{Daily hours of work} \times \mathrm{e}) + ((\mathrm{C} \times 2) + \mathrm{D}) \times (\text{Actual trips/day}) \tag{6}$$

Labor cost in $/day was then calculated by multiplying labor cost by the driver's assumed salary. Usually labor cost is affected by many factors including the driver's experience and performance

(Hooper et al., 2017). However, a $20/hour-salary was estimated for all scenarios in this research. Fuel cost breakdown includes the cost of fuel per liter, the truck fuel consumption in km per liter, and the total distance the truck travels in km per day. In this study, a fuel price of $0.82/liter obtained from Global Petrol Prices as of May 7, 2018 was used ("Global Petrol Prices," 2018).

Tire costs make up a small percent of total variable cost. According to Marufuzzaman et al. (2015), tire consumption is around 2% of the total variable cost. Tire costs are based on the percent loaded and empty factor, the tire cost and useful tire life, and the number of tires used (Marufuzzaman et al., 2015). Based on literature, the tire cost for trucks is around three times higher than those for passenger vehicles. The range is estimated to be between $0.03 to $0.07 per kilometer (Barnes et al., 2004; Marufuzzaman et al., 2015). A tire cost of $0.06/km was chosen. Maintenance and repair cost depends on many factors, including the truck usage and the truck operating conditions. The maintenance and repair cost were estimated to be $0.3/km (Hooper et al., 2017).

3.3 On-Site Landfill Leachate Treatment Processes Cost Estimate Methodology

3.3.1 SBR Costs

SBR operation is based on fill-draw system as explained in Chapter 2. Figure 3 displays a five-stage sequence SBR in operation. SBR operating cycle consists of a six-hour cycle time associated with four cycles a day with a total HRT of 24 hours. Six-hour operation in an SBR has been found to be the most suitable for wastewater treatment (Davis, 2010). Total time was obtained from the following period times using Equation (7):

$$T_c = = t_f + t_A + t_s + t_d + t_i \quad (7)$$

Where,

T_c= Total cycle time = 6 hours

t_f= Fill time = 3 hours

t_A= React aerate = 1.5 hours

t_s= Settling time = 0.75 hours

t_d= Decant time = 0.5 hours

t_i= Idle time = 0.25 hours

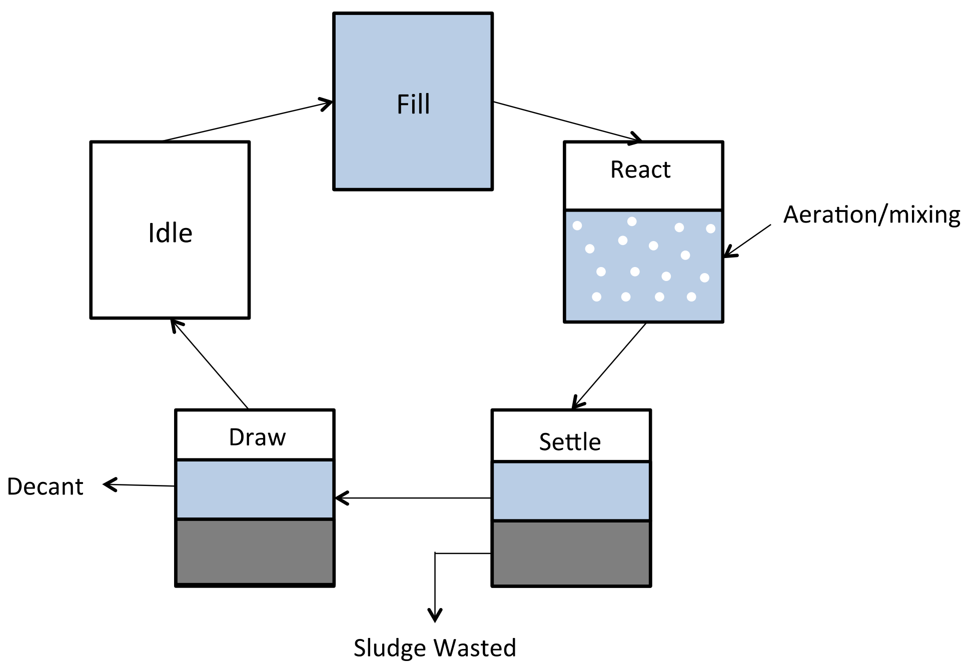

Figure 3: Sequencing Batch Reactors in a Single Tank with Multiple Stages

<u>3.3.1.1 Capital Costs</u>

For this research, it was assumed that two rectangular tanks were needed. Typical design parameters for each SBR tank are 24-hour HRT and 10-day SRT based on ranges obtained from literature (EPA, 1999; Metcalf et al., 2014). Volume of each tank was 99 m^3 and each tank was 7 m in length, 3 m in width, with a water depth of 5 m. Equipment costs included concrete tanks, blowers, aeration equipment, diffusers, and automatic valves. Capital cost of SBRs were based on capital and construction data provided by Torrens (2014). Equipment costs can be correlated with capacity of size using the following Equation (8) (Cooper et al., 2010)

$$\mathrm{Cost_B} = \mathrm{Cost_A}\left(\frac{\mathrm{Capacity\ B}}{\mathrm{Capacity\ A}}\right) \tag{8}$$

Where Cost B is the capital cost of this study, capacity A is the flowrate of Torrens facility in m^3/min, and capacity B is 99 m^3/day. Other construction costs were calculated as follows (Torrens, 2014):

- Piping and installation: 20% of equipment cost
- Electrical and instrumentation: 20% of equipment cost
- Engineering and construction management: 25% of equipment cost
- Structural: 10% of equipment costs
- Civil: 10% of equipment costs
- Contingency: 30% of total capital costs

<u>3.3.1.2 Operation and Maintenance</u>

O&M costs associated with a SBR system are similar to an activated sludge system (EPA, 1999), however, SBRs do not contain return activated sludge (RAS), clarifiers, or clarification equipment and they are operated in one single tank compared to multiple tanks thus reducing labor and maintenance costs. However, the maintenance cost associated with control and switches may be more expensive than conventional activated sludge processes. Cost items that are associated with landfill leachate treatment systems included labor, supplies, maintenance, administration, power, chemicals, safety and training, and laboratory testing. For all processes, labor costs were calculated assuming the workers work 40 hours/week, and 4 weeks/month, with a salary of $20/hour. Chemical additions for SBR include the addition of a carbon source (methanol).

3.3.1.3 Power Cost

This section goes through the approach followed in the calculation of power costs (Metcalf et al., 2014). First, in order to calculate the total power requirement, the effluent substrate concentration was calculated using Equation 9 (Metcalf et al., 2014):

$$S = \frac{K_s[1+b_H(SRT)]}{SRT(\mu_m - b_H) - 1} \tag{9}$$

Where,

S= effluent substrate concentration (BOD), g/m^3

SRT= solids retention time, d

K_s= half-velocity constant, g/m^3

μ_m= maximum specific bacteria growth rate, g biomass/g biomass·d

b_H= specific endogenous decay coefficient, g VSS/g VSS·d

Total oxygen required was calculated using Equation 10. The biomass as VSS wasted ($P_{x,bio}$) was determined using Equation 11. Since oxygen required for nitrification must be considered, nitrogen oxidation was calculated using Equation 12 (Metcalf et al., 2014), where nitrogen mass balance for the system was performed.

$$R_0 = Q(S_0 - S) - 1.42P_{x,bio} + 4.57Q(NO_X) \tag{10}$$

$$P_{x,bio} = \left[\frac{QY_H(S_0 - S)}{1 + b_H(SRT)} + \frac{f_d(b_H)QY_H(S_0 - S)SRT}{1 + b_H(SRT)}\right] \quad (11)$$

$$NO_X = TKN - N_e - \frac{0.12 P_{x,bio}}{Q} \quad (12)$$

Where,

R_0= OTR= total oxygen required, g/d

$P_{x,bio}$= biomass as VSS wasted, g/d

S_0= influent substrate concentration as BOD, g/m^3

NO_x= amount of $NO_3 - N$ produced from nitrification of $NH_4 - N$, g/m^3

Y_H= synthesis yield, g biomass COD/g bCOD removed

f_d= fraction of biomass that remains as cell debris, 0.10-0.15 g VSS/g biomass VSS depleted by decay

TKN= influent TKN concentration, mg/l

N_e= effluent $NH_4 - N$ concentration, mg/l

After the total oxygen required was calculated, the standard oxygen transfer rate (SOTR) was then calculated using Equation 13 (Metcalf et al., 2014).

$$SOTR = OTR \times [\left[\frac{\tau\beta\Omega(C^*_{\infty 20} - C)}{C^*_{\infty 20}}\right] [(\theta^{t-20}](\alpha)(F)]^{-1} \quad (13)$$

Where,

OTR=total oxygen required, kg/h

SOTR = standard oxygen transfer rate, kg/h

α = relative oxygen transfer rate

β = oxygen saturation factor (0.95 to 0.98)

F= diffuser fouling factor (0.65 to 0.9)

C^*_{st}= dissolved oxygen surface saturation concentration at operating temperature, mg/l

C^*_{s20}= dissolved oxygen surface saturation concentration at standard temperature, mg/l

$C^*_{\infty 20}$= DO saturation in wastewater for diffused aeration, mg/l

θ = empirical temperature correction factor (1.024)

τ= temperature correction factor

Ω= pressure correction factor

α = relative oxygen transfer rate

$C^*_{\infty 20}$ can be calculated using the following Equation (Metcalf et al., 2014), as shown in Equation 14.

$$C^*_{\infty 20} = C^*_{s20}\left[1 + d_e\left(\frac{D_f}{P_s}\right)\right] \tag{14}$$

Where,

P_s= standard barometric pressure (10.33 m)

d_e= mid-depth correction factor (0.40)

D_f= diffuser Depth, m

P_b= barometric pressure, m

Motor power from air blowers accounts for a portion of the total plant demand, where air flowrate is a function of SOTR and the diffuser efficiency. Fine bubble diffusers range between 20 to 35% specific oxygen transfer efficiency (SOTE) (Eini, 2012). The air flowrate was found using Equation 15 (Metcalf et al., 2014):

$$\text{Air flowrate}\left(\frac{m^3}{min}\right) = \frac{\text{SOTR}}{E*\frac{(\text{kg air})}{m^3}*\left(60\frac{min}{h}\right)*(0.2318\left(\frac{(\text{kg } O_2)}{\text{kg air}}\right)} \tag{15}$$

Where E is the fine bubble membrane diffusers with an aeration clean water SOTE of 35%. The concentration of oxygen by weight is 0.2318 kgO_2/kg air and therefore the air flowrate was calculated to be 10.7 m^3/min. Shown in Table 5 are all assumptions and values used to calculate the air flowrate in a SBR process.

Table 5: Air Flowrate Cost Parameters

Parameters	Unit	Value
Temperature	°C	25
Ks	g/m^3	8
Maximum specific bacteria growth rate	g/g*d	8.42
b_H	g/g*d	0.146
b_n	g/g*d	0.196
SRT	Days	10
Y_H	g VSS/g bCOD	0.45
NO_X	g/m^3	640
BODL of one mole of cells	mg O_2/mg cell	1.42
f_d	g/g	0.15
MLSS	g/m^3	4800
Average diffuser submergence	m	4.75
α	unitless	0.50
θ	unitless	1.024
Standard temperature	°C	20.00
β	unitless	0.95
C^*_{st}	g/m^3	8.26
C^*_{s20}	g/m^3	9.09
P_b	m	9.71
P_s	m	10.33
d_e	unitless	0.40
$C^*_{\infty 20}$	mg/l	10.76
F	unitless	0.85
Temperature correction factor	uniless	0.91
Pressure correction factor	unitless	0.94
Temperature	K	298.15
Atmospheric pressure	atm	0.94
Density of air	kg/m^3	1.11
Oxygen by weight	kg $O2/m^3$	0.26
Diffuser efficiency	%	35

Power requirement for aeration was then calculated using Equation 16 (Metcalf et al., 2014):

$$P_w = \frac{wRT_1}{28.97ne}\left[\left(\frac{P_2}{P1}\right)^n - 1\right] \tag{16}$$

Where,

P_w = power requirement of each blower, kW

w = air mass flowrate, kg/s

R= universal gas constant for air, 8.314 J/mole*K

T_1= absolute inlet temperature, K

P_1= absolute inlet pressure, atm

P_2= absolute outlet pressure, atm

n= (k-1)/k where k is the specific heat ratio. n = 0.283

28.97= molecular weight of dry air

550= conversion factor from ft*lbs/s to hp

e= efficiency (ranging from 0.70 to 0.9)

Air mass flowrate was calculated by multiplying the air flowrate calculated from Equation 15 by the density of air (kg/m^3). The density of air was calculated at 25 ℃ with a pressure of 95.2 KPa to be 1.12-kg/m^3. Shown in Table 6, are all assumptions and values used to calculate the power requirement in a SBR process. Hourly power costs were determined based on an electricity cost of $0.1 /kW-hr (EIA, 2018).

Table 6: Aeration Power Cost Parameters

Parameters	Unit	Value
Density of air	kg air/m^3	1.204
Air mass flowrate	kg/s	0.21
R	J/mole*K	8.314
Absolute inlet temperature (T_1)	K	293.15
Pressure drop (piping, valves, diffusers)	m	1.22
Absolute inlet pressure (P_1)	m	10.33
Absolute outlet pressure (P_2)	m	15.68
Specific heat ratio (k)	unitless	0.283
Molecular weight of dry air	g/mol	28.97
Conversion factor from	ft*lbs/s to hp	550
Blower mechanical efficiency	%	75
Blower motor electrical efficiency	%	90
Blower overall efficiency (e)	%	68

3.3.2 MBR Costs

3.3.2.1 Capital Costs

MBRs operate with a shorter HRT compared to SBR systems; however MBRs operate with a longer SRT. Figure 4 displays a complete MBR system used for this study. For this research typical design parameters for the MBR include an 8 hour HRT and 25 day SRT (Verrecht et al., 2010). A temperature of 25 °C, RAS recycle of ratio of 6, and membrane flux of 20 $L/m^3 \cdot h$ were obtained from Metcalf et al. (2014). Capital costs include pre-anoxic tank, aeration tank, blowers, aeration equipment, dewatering, chemical pumps, and automatic valves. Capital cost of MBR was based on capital and construction data provided by Torrens (2014). Construction costs were calculated as follows (Torrens, 2014):

- Piping and installation: 20% of equipment cost
- Electrical and instrumentation: 20% of equipment cost

- Engineering and construction management: 25% of equipment cost
- Structural: 10% of equipment costs
- Civil: 10% of equipment costs
- Contingency: 30% of total capital costs

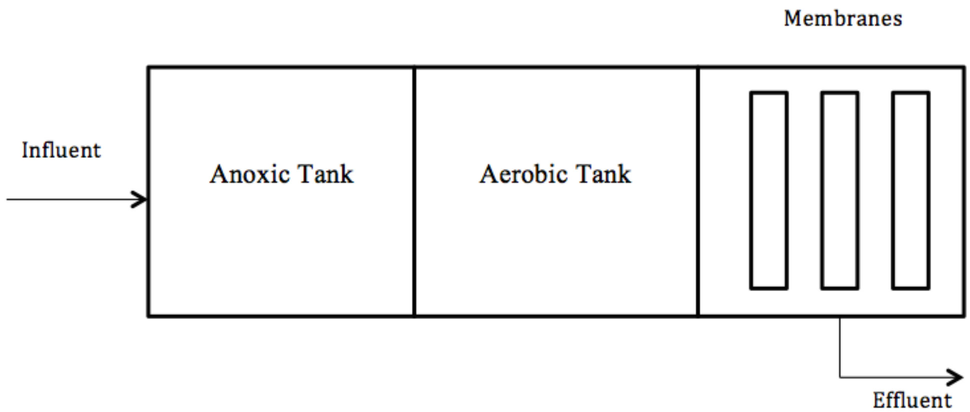

Figure 4: Membrane Bioreactor System

3.3.2.2 Operation and Maintenance

O&M costs associated with an MBR system is similar to an SBR. In order to determine the volume of the membrane separation tank, the membrane surface area is determined by using Equation 17:

$$\text{Membrane surface area} = \frac{\text{Q}\left(\frac{\text{m}^3}{\text{d}}\right)*\left(\frac{\text{d}}{24\text{h}}\right)}{\text{Membrane flux}\left(\frac{\text{L}}{\text{m}^2\text{h}}\right)} \tag{17}$$

The tank volume was calculated by multiplying the membrane surface area by a membrane tank volume to membrane area ratio of 0.025 m^3/m^2 to be 5.16 m^3 (Metcalf et al., 2014). In order to determine the pre-aeration volume, the mass of MLVSS and MLSS in the aeration basin were found using Equations 18 and 19 as follows:

$$P_{x,VSS} = P_{x,bio} + Q(nbVSS) \quad (18)$$

Where,

$P_{x,VSS}$=net waste activated sludge produced each day, kg VSS/day

nbVSS= nonbiodegradable volatile suspended solids

The total mass of solids in the reactor is:

$$P_{x,TSS} = \frac{P_{x,vss}}{0.85} + Q(nbVSS) + Q(TSS - VSS) \quad (19)$$

Where,

TSS= influent wastewater TSS concentration, mg/l

VSS= influent wastewater VSS concentration, mg/l

The volume is than calculated to be 120 m^3. The size of the anoxic zone was than calculated to be 24 m^3. The oxygen requirement is calculated using equations 10-15 shown in Section 3.3.1.3. However, two different SOTR and air flowrates were calculated, one for the preaeration tank and one for the membrane tank. Based on a mixing energy of 8 kW/m^3 used by (Metcalf et al., 2014), the anoxic zone mixing energy was estimated to be 0.19 kW. Membrane fouling control and cleaning chemical and pumping costs are an important factor that affects the overall total MBR system costs. For this research, chemical cost data were provided by Torrens (2014).

3.4 SGA Processes Cost Estimate Methodology

3.4.1 Equalization

WWTPs implement a sidestream treatment process to deal with the resulting loads from sludge dewatering that contain high concentrations of ammonia and phosphorus. To address the impact of batch leachate loads on the sidestream treatment processes, an equalization basin was provided to manage the flow at the beginning of the sidestream treatment. The leachate with varying flow enters the tank before going through the rest of the treatment processes (Goel et al., 2005). It was assumed that a cylindrical equalization tank had a three-day HRT. Volume of the tank therefore was 1500 m^3 and the tank size was 6 m in height, and 9 m in diameter.

3.4.2 Struvite Crystallization

3.4.2.1 Capital Cost

For this research, it was assumed that one cone-shaped struvite crystallization tank was needed. The struvite reactor received influent of centrate from the sludge dewatering combined with leachate transported from a landfill. Munch et al. (2001) studied the effect of HRT on the effluent ortho-P concentration. The study showed that an HRT of 1-2 hour was sufficient, and had no effect on the effluent ortho-P concentration (Münch et al., 2001). For struvite crystallization, an HRT of one hour was chosen for this research (Metcalf et al., 2014), in addition to 30 minutes of settling time to allow the separation of precipitated struvite (Yetilmezsoy et al., 2017). Given an HRT of one hour, the volume of the tank was calculated to be 21 m^3 with a capacity of 500 m^3 per day. Equipment cost includes the struvite tank, chemical tanks for MgO, NaOH and H_2SO_4, automatic valves, building, and chemical dosage pumps. Struvite precipitation has been widely studied at laboratory scale. However, there are few full-scale applications, and therefore capital cost data were limited. Capital costs of struvite crystallization were based on capital and construction data provided in Yetilmezsoy et al. (2017). Other construction costs were calculated similar to a SBR system and are as follows (Torrens, 2014):

- Piping and installation: 20% of equipment cost
- Electrical and instrumentation: 20% of equipment cost
- Engineering and construction management: 25% of equipment cost
- Structural: 10% of equipment costs
- Civil: 10% of equipment costs
- Contingency: 30% of total capital costs

3.4.2.2 Operation and Maintenance

There are principal operational requirements that should be considered for P and N recovery as struvite precipitation including chemical and mixing requirements, and pH control. The chemical requirement included magnesium sources for struvite recovery. The source of magnesium is either magnesium chloride ($MgCl_2$), MgO, or $Mg(OH)_2$. The decision of choosing the best chemical that suits the process is based on cost and availability. For this research, MgO was chosen as the source of magnesium. MgO provides magnesium to the crystallizer, whereas NaOH is used to control pH to the desired level of pH 9. However, in order to feed the effluent of the struvite crystallization process to the influent aerobic granular sludge process, H_2SO_4 was used as a pH controller to maintain the pH within the range of 7 to 7.5. A P:Mg molar ratio of 1:1.2 was obtained from The University of Utah. The weighted average of $PO_4 - P$ concentration for centrate and leachate was calculated to be 10.3 mg/l.

1) The dose of MgO to be added was calculated as follow:

$$\frac{10.32\times10^{-3}\,\mathrm{g/L}}{31\ \mathrm{g/mol}} = 3.3 \times 10^{-4}\ \mathrm{mol/l}$$

2) Since the ratio of P:Mg = 1:1.2, molarity for Mg is:

$0.00033*1.2= 3.96\times 10^{-4}$ mol/l

3) Molecular weight of Mg is 40.3 g/mol, so

= 3.96×10^{-4} mol/l * 40.3 g/mol= 0.016 g/l *1 kg/1000g= $\mathbf{1.6\times 10^{-5}}$ **kg/l of MgO added**.

Mixing power is an important application of P recovery as struvite. The engine power and electricity costs were calculated using Equation 20 (Metcalf et al., 2014):

$$P = \mu G^2 V \tag{20}$$

Where,

P= power requirement, W

G= average velocity gradient, 1/s

μ= dynamic viscosity, N·s/ m^2

V= volume m^2

A velocity gradient of 500 s^{-1} and a mixing time of 15 minutes were chosen (Yetilmezsoy et al., 2017). The water temperature was assumed to be 25 ℃ with a dynamic viscosity of $0.89\times 10^{-3}\ N\cdot s/m^2$. The power requirement was calculated to be 4.7 kW (4700 W). A safety factor of 1.2 was chosen and therefore the adjusted mixing power was 5.6 kW (Yetilmezsoy et al., 2017).

3.4.3 PN/Anammox

PN/A, also known as deammonification, has been described previously in chapter 2. PN/A has a variety of process configurations, such as two-stage Sharon-Anammox process, Terra-N moving bed bioreactor (MBBR) process, and SBR. For this research, a single stage attached growth PN/A reactor operated as an SBR reactor was used. The SBR technology is the most frequently applied reactor type; 88% of PN/A reactors are installed as single-stage configuration (Lackner et al., 2014).

<u>3.4.3.1 Capital Cost</u>

The reactor provided a six-hour cycle time associated with four cycles/day with a total HRT of 48 hours. A HRT of 48 hours was assumed based on overview of full-scale PN/A plants by Lackner et al. (2014). The study evaluated different SBR plants with a variety of operational strategies and an HRT ranging from 45 to 75 hours (Lackner et al., 2014). Volume of each tank for this study was 1000 m^3 and each tank was 15 m in length, 7.5 in width, with a water depth of 8.8 m. The cycle consisted of 5.5 hour reaction period, 0.25 hour decant, and feeding period of 0.25 hours. The reaction period consisted of 5 min of aeration and 10 minutes of anoxic operation to minimize the impact of nitrate on Anammox activity (Metcalf et al., 2014). Equipment costs were based on an SBR system with the addition of attached growth costs. Equipment costs included tanks, blowers, aeration equipment, diffusers, automatic valves, and decanters. Capital cost of Anammox were based on capital and construction data provided by Torrens (2014). Other construction costs were calculated similar to an SBR system and are as follows (Torrens, 2014):

- Piping and installation: 20% of equipment cost
- Electrical and instrumentation: 20% of equipment cost
- Engineering and construction management: 25% of equipment cost
- Structural: 10% of equipment costs
- Civil: 10% of equipment costs
- Contingency: 30% of total capital costs

3.4.3.2 Operation and Maintenance

Similar approaches to calculate the oxygen requirement and power cost for PN/A were used from the SBR process based on Equations (10-15) with these following adjustments:

1) From Equation 2 (p.16) the oxygen requirement for PN/A is 1.9 kg O_2/kg N compared to 4.6 kg O_2/kg N from conventional nitrification/denitrification, which indicates that aeration requirement is reduced by 60%. In order for the Anammox process to achieve optimal performance, an effluent from the PN/A with 50% $NO_2 - N$ and 50% $NH_3 - N$ must be achieved (Eini, 2012).
2) Influent coming into the Anammox reactor is the effluent from the aerobic granular sludge process.
3) The DO concentration is controlled at 0.3 g/m^3 during each aerobic phase compared to a DO of 2.0 g/m^3 for SBR and MBR.
4) The PN/A reactor is operated with a temperature of 34℃ compared to 25 ℃ for both SBR and MBR. Partial nitritation operating temperature usually ranges from 30-35 ℃ and that is to ensure that AOB outcompetes NOB. Since PN/A reactors operate at high temperatures, a heating exchanger is required (Liu et al., 2015).

CHAPTER FOUR: RESULTS AND DISCUSSION

This chapter presents the results from spreadsheet models to evaluate cost effective approaches for the removal of nitrogen and phosphorus from leachate and leachate/centrate mixtures. One of the objectives of this research was to determine capital and O&M costs for traditional on-site leachate treatment processes and compare them to the SGA process. The costs were estimated from various sources (Torrens 2014; Yetilmezsoy et al. 2017; EPA 1999) updated to present value (2018). This chapter is separated into four sections, including the results of costs of transportation, on-site leachate treatment processes, the SGA process, and environmental assessment.

Complete characteristics of leachate and centrate used in this research can be seen in Table 7. Leachate characteristics were obtained from literature (Foo et al., 2009; Smith et al., 2013), whereas centrate characteristics were obtained from the University of Utah in which samples were collected from a local WWTP and analyzed. The mixture characteristics were determined by calculating a weighted average using the following Equation (21):

$$\text{Mixture conc.} = \frac{(\text{Leachate flow} * \text{leachate characteristics}) + (\text{Centrate flow} * \text{centrate characteristics})}{\text{Leachate flowrate} + \text{Centrate flowrate}} \qquad (21)$$

Table 7: Leachate and Centrate Chemical Characteristics

Parameter	Leachate	Centrate	Centrate + Leachate (for side-stream treatment)
BOD, mg/l	1000	79	263.4
COD, mg/l	4000	450	1160
$NH_3 - N$, mg/l	800	485	548
P, mg/l	9.6	10.5	10.3
pH	4.5-9	7.6	-

4.1 Transportation

A spreadsheet model for calculating the cost per m^3 of truck transportation of leachate to a wastewater treatment facility was used. Total transportation cost ($/year) was calculated by using the following Equation (22):

$$\text{Total transportation cost} = \text{Annualized fixed cost} + \text{Total variable cost/year} \quad (22)$$

Total transportation per year costs can be seen in Table 8 for three different water tanker trucks with load capacities of 7.6 m^3(2000 gal), 15.1 m^3 (4000 gal), and 18.9 m^3 (5000 gallons) and for six different distances, 15, 20, 25, 30, 60, and 100 km. Variation of distances gave a better understanding of how transportation is affected by distance.

Table 8: Total Transportation Present Value (2018) Cost in U.S. Dollars per Year

Truck Capacity (m^3)	7.6	15.1	18.9
15 km			
Annualized fix cost ($/year)	$21,300	$16,900	$19,500
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$101,000	$57,200	$48,400
Total Transportation Cost ($/year)	**$129,000**	**$80,800**	**$74,600**
20 km			
Annualized fix cost ($)	$31,900	$33,700	$19,500
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$124,000	$68,400	$57,300
Total Transportation Cost ($/year)	**$162,000**	**$109,000**	**$83,500**
25 km			
Annualized fix cost ($)	$31,900	$33,700	$38,900
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$146,000	$79,500	$66,200
Total Transportation Cost ($/year)	**$185,000**	**$120,000**	**$112,000**
30 km			
Annualized fix cost ($)	$31,900	$33,700	$38,900
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$164,000	$88,400	$73,400
Total Transportation Cost ($/year)	**$202,000**	**$129,000**	**$119,000**
60 km			
Annualized fix cost ($/year)	$53,000	$50,500	$38,900
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$289,000	$151,000	$123,000
Total Transportation Cost ($/year)	**$349,000**	**$208,000**	**$169,000**
100 km			
Annualized fix cost ($)	$74,400	$67,400	$58,300
Fixed Annual ($/year)	$6,800	$6,800	$6,800
Total Variable Cost ($/year)	$454,000	$234,000	$190,000
Total Transportation Cost ($/year)	**$535,000**	**$308,000**	**$255,000**

- Numbers may not total correctly due to rounding

Total transportation in U.S. dollar per year was then converted to U.S. dollar per m^3, using a leachate flowrate of 99 m^3/day as shown in Table 9.

Table 9: Total Transportation Cost in U.S. Dollars per m^3

km	7.6 m^3	15.1 m^3	18.9 m^3	Min cost
15	\$3.6	\$2.2	\$2.1	\$2.1
20	\$4.5	\$3.0	\$2.3	\$2.3
25	\$5.1	\$3.3	\$3.1	\$3.1
30	\$5.6	\$3.6	\$3.3	\$3.3
60	\$9.6	\$5.7	\$4.7	\$4.7
100	\$14.7	\$8.5	\$7.0	\$7.0

The results support that a tanker truck with a capacity of 18.9 m^3 (5000 gal) is the most cost affective way to transport leachate from a local landfill to a WWTP. A tanker truck that size requires fewer trips per day compared to other tanker truck capacities. In addition, it requires a fewer number of trucks ranged from one to three trucks depending on the distance traveled, and as the number of trucks decrease, maintenance, labor, and fuel costs also decrease. Figure 5 displays the cost of transportation as a function of distance.

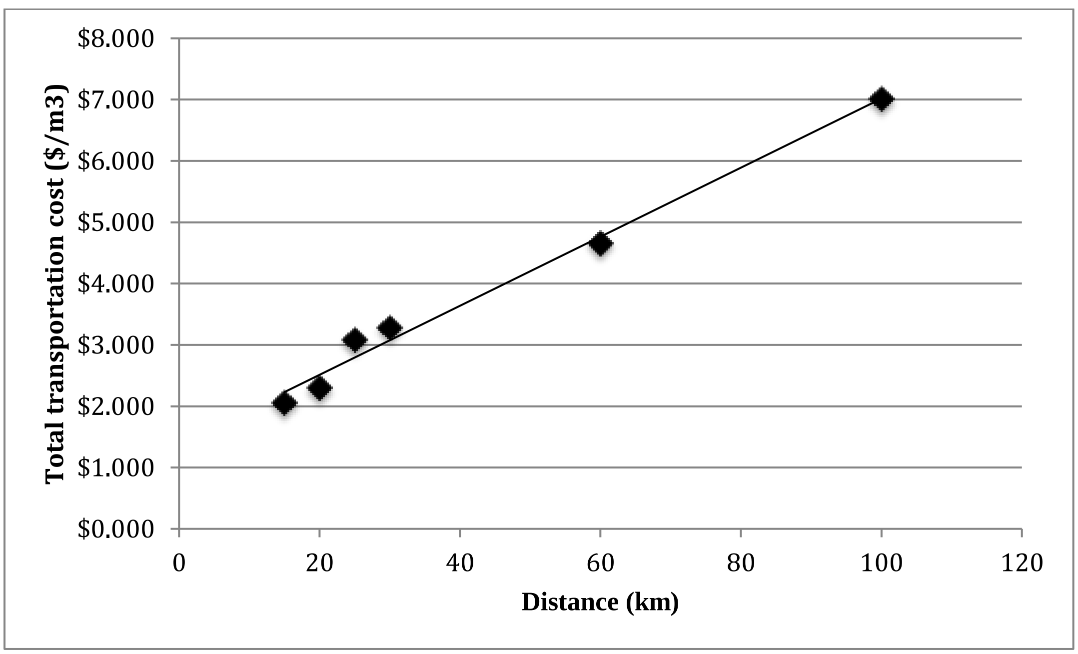

Figure 5: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (km), 18.9 m^3 Truck Capacity

4.2 Comparison of SGA and Leachate Treatment Processes Costs

4.2.1 Treatment Processes Cost Analysis

The SGA process, as previously mentioned, consists of struvite crystallization followed by aerobic granular sludge and PN/A reactors (Figure 1). The results of capital and O&M costs for on-site landfill leachate treatment and the proposed SGA treatment process have been summarized in Tables 10 and 11. Table 10 shows traditional leachate treatment only. The proposed SGA treatment process as shown in Table 11 shows results of the combined leachate and centrate treatment.

Table 10: Total Capital and O&M Cost for MBR and SBR

Process	SBR	MBR
Capital cost ($/m^3) (20 years)	0.90	1.20
Annual operation and maintenance cost ($/m^3)	2.77	3.20

Table 11: Total Capital and O&M Costs for SGA Process

Process	Struvite	Aerobic Granular Sludge	PN/A	Total Cost
Capital cost ($/m^3) (20 years)	0.36	0.86	1.11	2.33
Annual operation and maintenance cost ($/m^3))	0.46	1.15	0.51	2.12

Treatment of struvite crystallization formed was not included in the operation costs, since the struvite was assumed sold as a fertilizer. The total struvite formed was calculated to be 32 kg struvite/day. Using a struvite sale price of $242 per ton (Ueno et al., 2001; Yetilmezsoy et al.,

2017) (2018 dollars) it was estimated that the income from fertilizer was $3000 per year. The total mixing power cost for struvite crystallization was calculated to be $4900 per year using Equation 20.

4.2.2 Capital Cost Analysis

All costs shown in Figure 6 compare the capital cost for each treatment process for a plant design life of 20 years converted to annual cost using an interest rate of 5% and Equation 5 (p.28). The proposed treatment and leachate treatment was than calculated based on the flowrates provided for each system (99 m^3/day for SBR/ MBR and 500 m^3/day for SGA) in order to calculate the cost per m^3 (U.S. dollars/m^3). For the combined SGA process, capital costs were \$2.51/ m^3 compared to SBR at \$0.89/ m^3 and MBR at \$1.19 per m^3. The higher capital cost for the SGA process was expected because of the combination of three processes each including tankage, piping and installation, engineering and construction management etc.

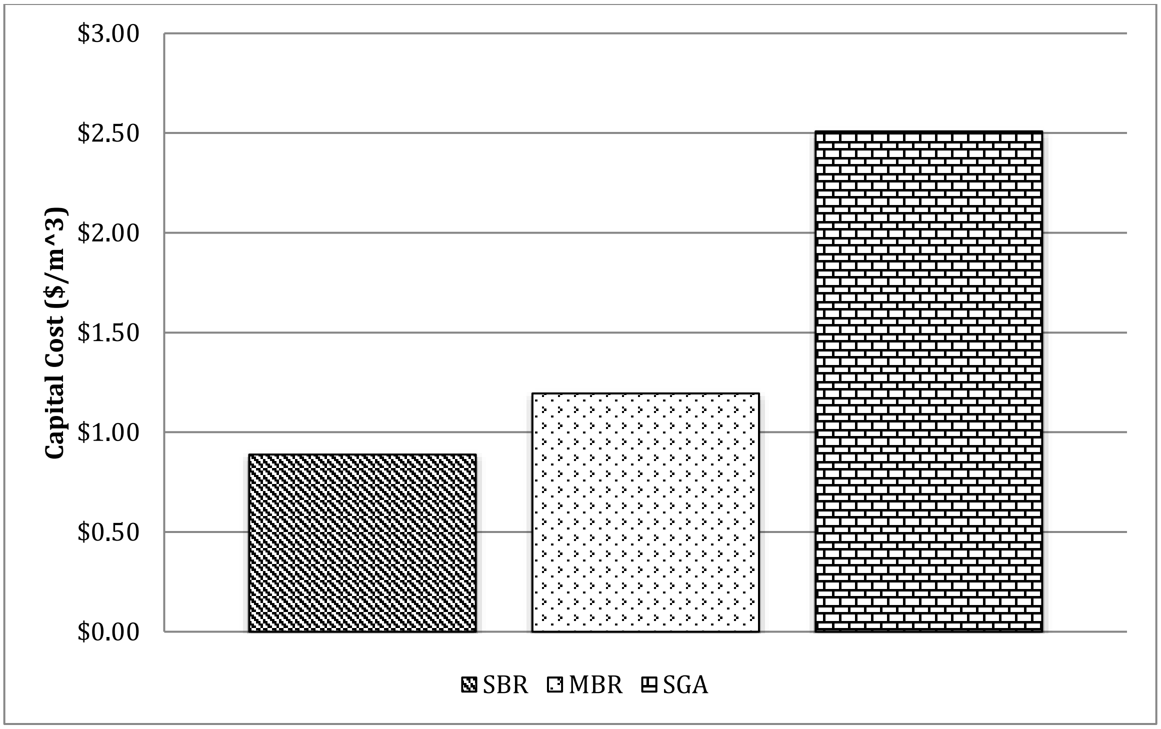

Figure 6: Total Capital Cost in U.S. Dollars per m^3

4.2.3 Operation and Maintenance Cost Analysis

O&M costs were estimated considering power, chemical supplies, maintenance and labor (wages) costs. As can be seen from Figure 7, the combined SGA O&M costs are lower than both conventional on-site landfill leachate treatments. Notably, the lower O&M costs is expected because PN/A requires less oxygen compared to conventional nitrification/denitrification leachate processes. In addition, PN/A does not require any methanol addition, which decreases chemical costs. O&M costs for the SGA process are also offset by the sale of struvite.

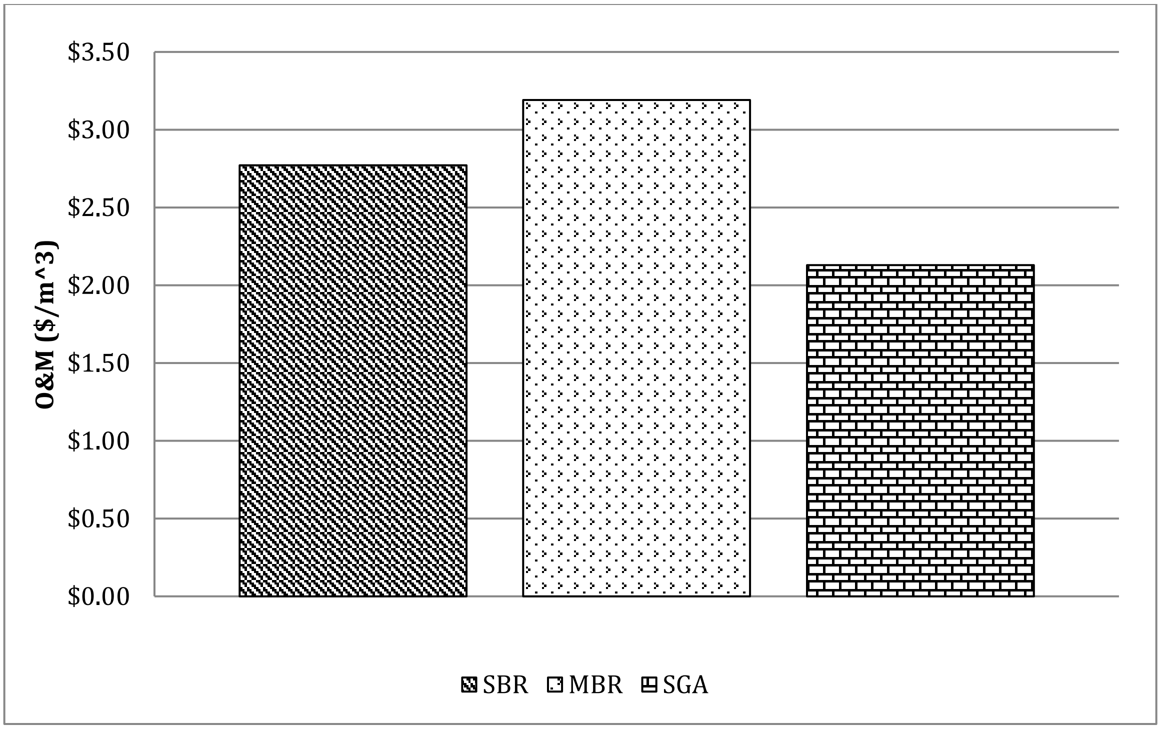

Figure 7: Total O&M Costs in U.S. Dollars per m^3

4.2.4 Power Cost Comparison

Figure 8 compares treatment processes with power cost in ($/m^3). The combined SGA power costs are higher than both conventional on-site landfill leachate treatments. However, in the case of treating older leachate with no biologically removable material, the aerobic granular sludge process can be eliminated and therefore, reduce the overall SGA power cost. The MBR also have high-energy demand due to air scouring of membranes to control fouling.

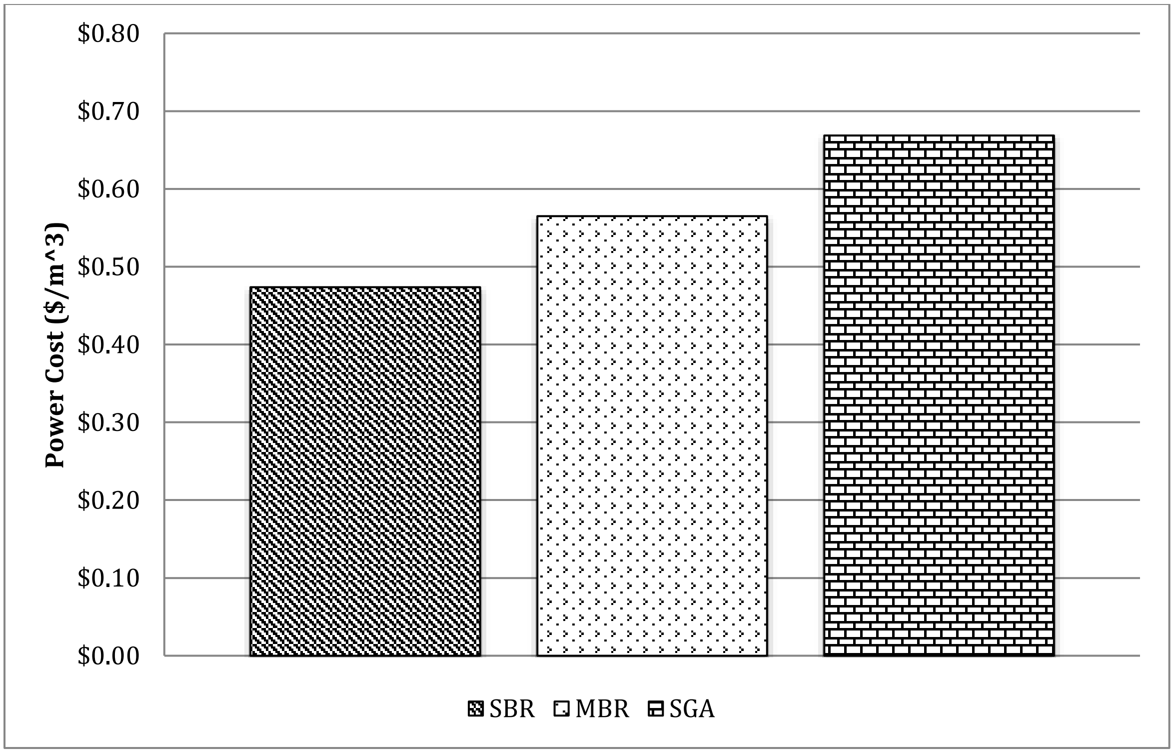

Figure 8: Power Cost in U.S. Dollars per m^3

In order to meet the objective of this study, a comparison of total (Capital + O&M) was made for all processes as shown in Figure 9. Transportation, capital cost, and O&M cost are included to determine the most cost effective approach. As can be seen from Figure 9, the SGA process is significantly higher than both on-site leachate treatment processes. SGA process benefits from selling fertilizer along with reducing the load of phosphorus and ammonia recycled to the head of the plant that may reduce costs. However, the cost of the recycled phosphorus and ammonia into the influent WWTP is recommended as a future study. Figure 9, reflects transportation cost of \$2.05/m^3 for a distance of 15 km. However, with varying distance (km) the total SGA process will increase. For example, for a distance of 25 km, the total SGA cost will increase to \$11.3/m^3.

Table 12 indicates the cost of landfill leachate directly discharged to a WWTP. This calculation was analyzed to compare the price of direct discharge of leachate in a WWTP to the treatment of leachate through sidestream with aerobic granular sludge and PN/A reactor. Cost of discharged leachate was $2.64/m^3 provided by Bolyard (2018). Total costs included transportation, equalization, combined centrate and leachate with struvite crystallization treatment only and POTW (direct discharge) was calculated to be $5.70/m^3. Figure 9 displays this difference in total costs for the two different scenarios.

Table 12: Total Cost of Landfill Leachate Directly Discharged to WWTP in U.S. Dollars per m^3

Process	$/m^3
Transportation	2.05
Equalization	0.18
Struvite Crystallization	0.82
POTW (Cost without trucking)	2.64*
Total	5.70

- Source: (Bolyard, 2018)

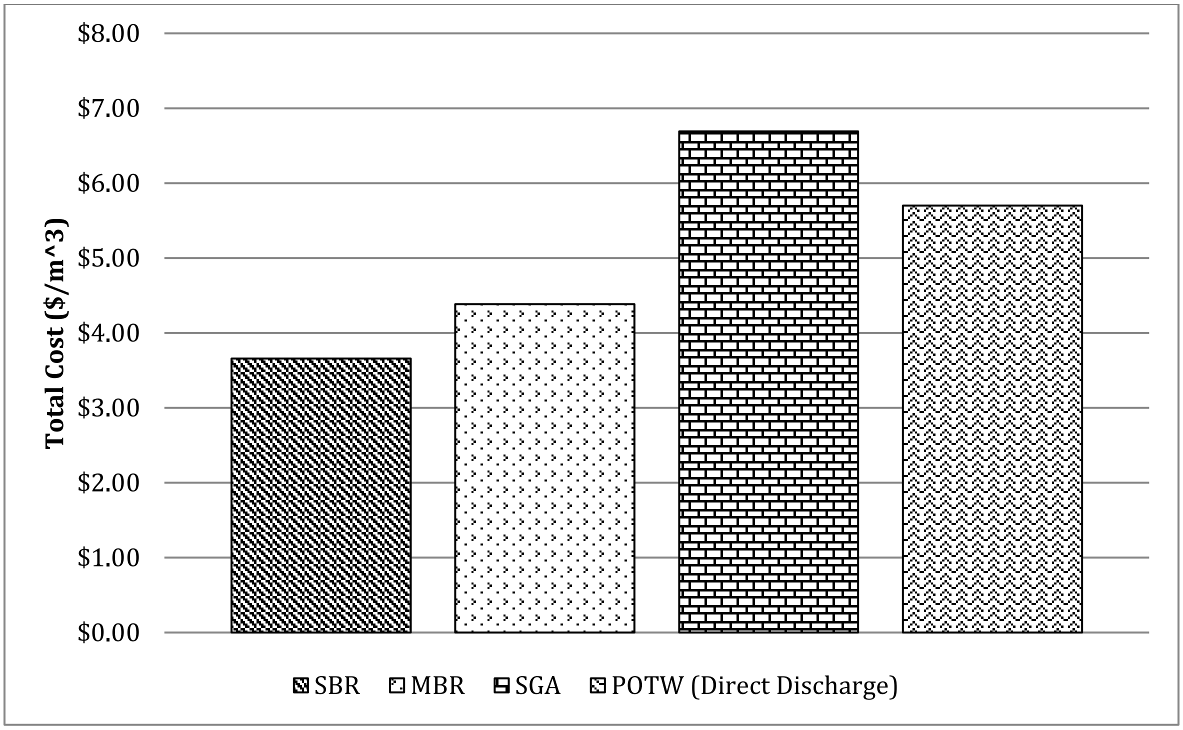

Figure 9: Total Costs in U.S. Dollar per m^3

Figure 10 represents the total costs of the SGA process and compares each individual process to have a better understanding of the reason behind the high SGA costs. As can be seen from Figure 10, transportation cost as a function of distance (15 km) is the highest cost at $2.05/m^3. Following transportation in cost is the aerobic granular sludge process, which is considered the most expensive SGA process. Furthermore, if eliminated the SGA process will therefore decrease dramatically in cost.

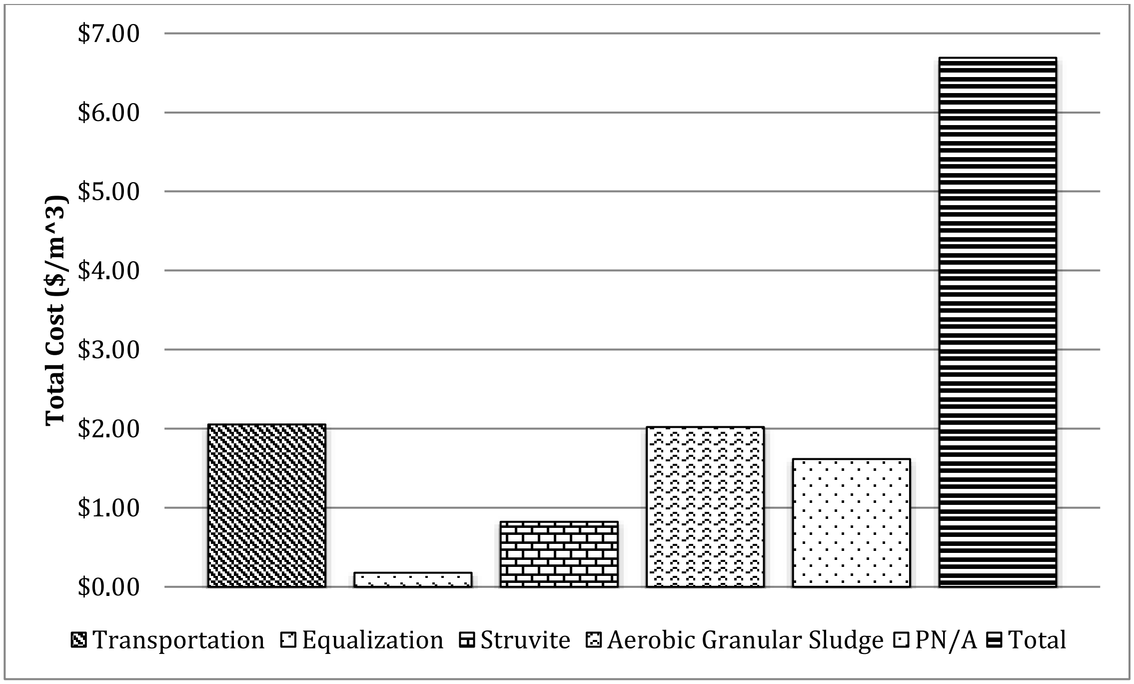

Figure 10: Total SGA Costs in U.S. Dollars per m^3

4.3 Environmental Assessment

The main goal was to lower the phosphorus and ammonia loads going back to the influent main stream in a WWTP. For struvite precipitation, a removal of 78% of P was estimated (Moerman et al., 2009), therefore the SGA effluent was 2.2 mg/l. Without sidestream treatment, the P would be returned to the WWTP influent at 10.3 mg/l as a result of P release in the digester and centrifugation cite. The aerobic granular sludge process was assumed to remove 90% of NH_3, which results in an effluent of 54.4 mg/l. This effluent is than fed to the PN/A reactor, where 90% of the ammonia is removed, resulting in an effluent of 5.4 mg/l compared to an influent concentration of 550 mg/l that would have been recycled back to the plant with no sidestream treatment. It must be noted that this concentration is for the combined leachate and centrate stream. SBR and MBR ammonia effluent concentrations would be 80 mg/l and 160 mg/l, respectively. Table 13 summarizes removal efficiency assumptions for all treatment processes.

Table 13: Effluent Ammonia and Phosphorus Removal Efficiencies

Process	Percent Removal	Source
Phosphorus removed in struvite crystallization	78%	(Moerman et al., 2009)
NH_3 removed in aerobic granular sludge	90%	(Pronk et al., 2015)
NH_3 removed in a PN/A reactor	90%	(Kent et al., 2018; Lackner et al., 2014)
NH_3 removed in a SBR process	>90%	Table 2 (p.12)
NH_3 removed in a MBR process	80%	(Ahmed et al., 2012; Fudala-Ksiazek et al., 2018)

Estimating GHG emissions is an important tool for WWTPs. Nitrous oxide, a GHG, can occur as direct or indirect emissions from wastewater during treatment. Nitrous oxide is usually generated during nitrification and denitrification of the nitrogen present in the form of ammonia. Nitrous oxide emissions can be determined by using the following Equation (23) (Eggleston et al., 2006):

$$N_2O\ \text{Emissions} = N_{\text{Effluent}} * EF_{\text{Effluent}} * N_2O\ MW/N_2\ MW \qquad (23)$$

Where,

N_2O Emissions= emissions in inventory year, kg N_2O/year

N_{Effluent}= nitrogen in the effluent discharge, kg N/year

EF_{Effluent}= emission factor for N_2O emissions

The emissions factor was obtained from (Eggleston et al., 2006) as 0.005 Kg N_2O-N/Kg-N. The comparison of N_2O emissions in Kg per year can be seen in Figure 11. N_2O emissions were calculated to be 7.7 kg N_2O/year compared to 22.7 and 45.4 kg N_2O/year for SBR and MBR respectively. These results indicate that even though SGA process contained higher capital and O&M costs, the process reflect on the environment observed positive outcomes.

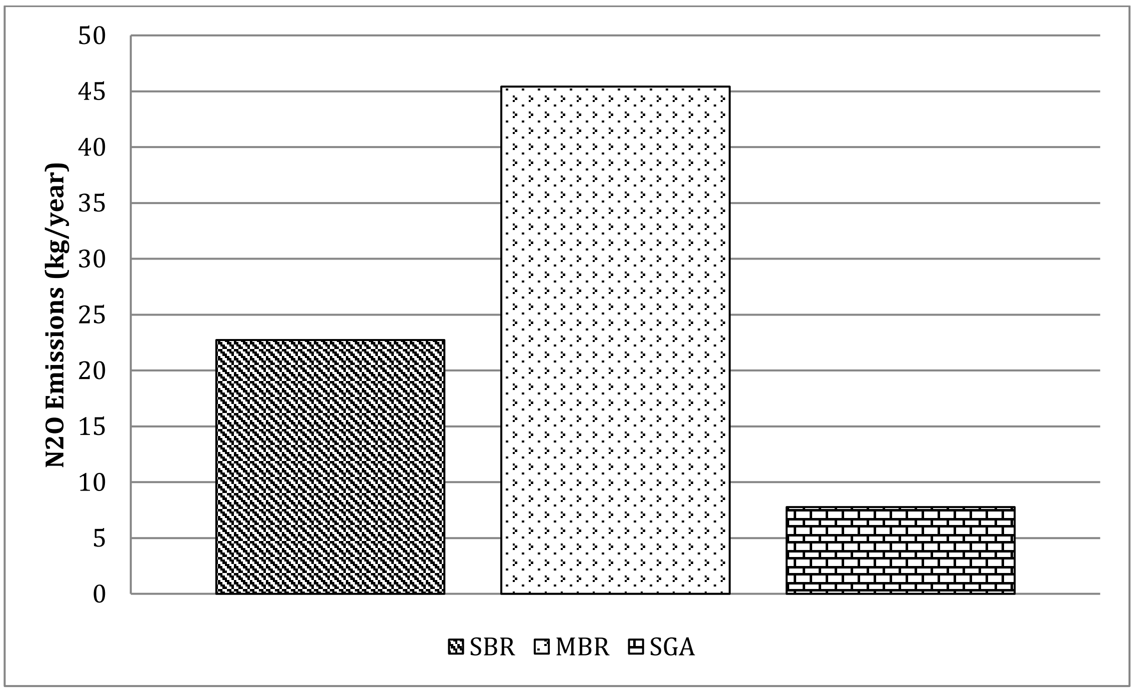

Figure 11: N_2O Emissions in Kg per Year

GHG release will also occur during power generation. Figure 12 compares power consumption for the three treatment scenarios. It can be seen that the SGA power consumption is higher than both traditional on-site treatments. However, as mentioned in Section 4.2.4, aerobic granular sludge process can be eliminated and therefore, reduce the overall SGA power consumption. Furthermore, the power consumption dropped from 6.7 kW-hr/m^3 to 1.1 kW-hr/m^3 and, as a result, would create less GHG emissions assuming fossil fuel is used in the generation of electricity.

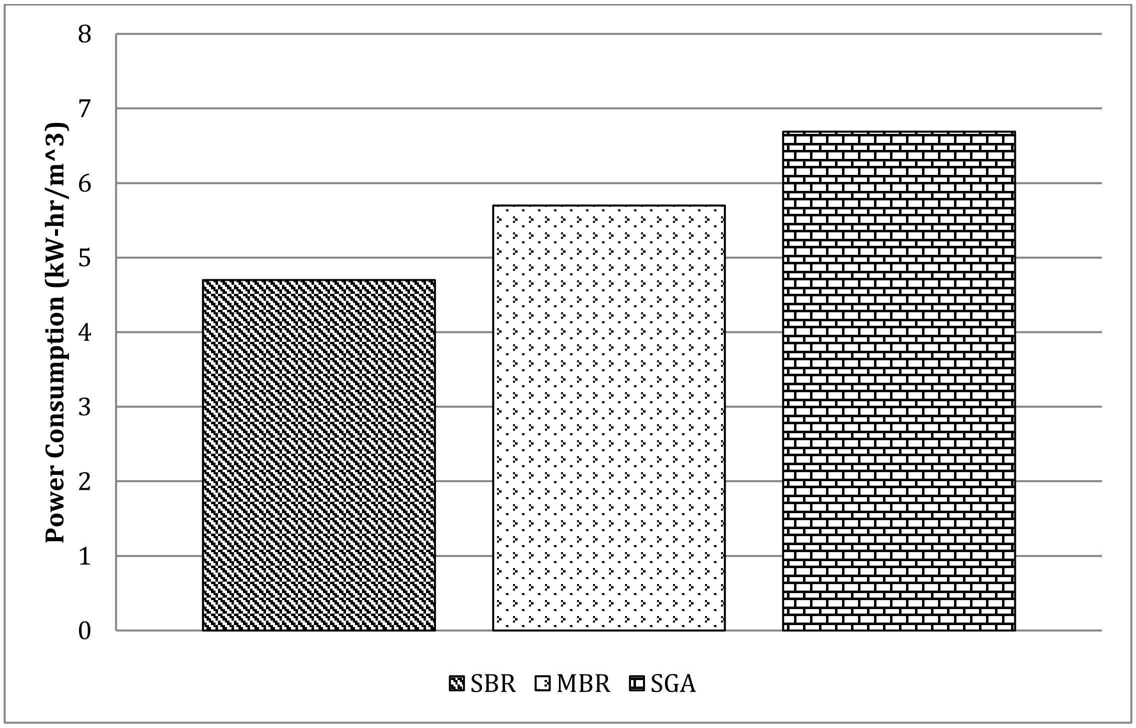

Figure 12: Power Consumption in kW-hr per m^3

CHAPTER FIVE: CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

For this research, a cost analysis and environmental assessment of the proposed novel treatment approach was completed and compared to more traditional landfill on-site leachate treatment approaches (MBR and SBR). The study was completed with the use of spreadsheet-based models. Spreadsheets have been developed to evaluate treatment costs (Capital + O&M) for both the proposed nutrient recovery/biological and traditional on-site leachate treatments. Transportation costs of leachate to the WWTP have been studied and analyzed by the use of a spreadsheet model as a function of distance. Listed below are the key conclusions for this research:

- The results support that a tanker truck with a capacity of 18.9 m^3 (5000 gal) is the most cost affective way to transport leachate from a local landfill to a WWTP.
- For struvite precipitation from the treatment of combined centrate and leachate, it was estimated that the income from struvite fertilizer was $3000 per year for agriculture use.
- Based on figures and tables shown above, total capital and O&M costs of SGA were higher than traditional approaches. However, positive outcomes from this process include lower N_2O emissions, lower power consumption, struvite fertilizer, and overall recovery of nitrogen and phosphorus with the combination of centrate and leachate. In the case of treating older leachate with no biologically removable material, the aerobic granular sludge process can be eliminated and therefore, reduce the overall total costs and power consumption. Operational parameters that include temperature adjustments, WWTP characteristics, and WWTP load are all important parameters to consider. In addition, the

installations of these processes require operator knowledge and training due to monitoring, chemical addition, and pH control.

5.2 Recommendations

Opportunities exist for further research. This research focused on conducting a cost analysis for leachate N/P management approaches, however many aspects of the research are incomplete and should be addressed in the future. Some suggestions include the following:

- Complete a sensitivity analysis to reduce the uncertainty of this project. A sensitivity analysis will determine the most significant inputs to cost. In addition, a sensitivity analysis will point to uncertainty of the plant performance criteria that include sludge production, HRT, treatment efficiencies and energy consumption. Furthermore, The University of Utah lab results will reduce uncertainty for this project.
- For this research, offsetting costs for sidestream treatment of centrate were not accounted for, thus for future research an offset cost should be determined for sidestream treatment.
- There are limitations when comparing full-scale SBR and MBR costs to SGA laboratory results; it is recommended that large-scale studies be conducted to improve accuracy.

APPENDIX A: TRANSPORTATION

Table 14: Equivalent Uniform Annual Costs of Trucks

Number of trucks needed	Cost of Truck	Equivalent Uniform Annual Costs
15 km		
2	$164,000	$21,200
1	$130,000	$16,900
1	$150,000	$19,500
Number of trucks needed	**Cost of Truck**	
20 km		
3	$246,000	$31,900
2	$260,000	$33,700
1	$150,000	$19,500
Number of trucks needed	**Cost of Truck**	
25 km		
3	$246,000	$31,900
2	$260,000	$33,700
2	$300,000	$38,900
Number of trucks needed	**Cost of Truck**	
30 km		
3	$246,000	$31,900
2	$260,000	$33,700
2	$300,000	$38,900
Number of trucks needed	**Cost of Truck**	
60 km		
5	$410,000	$53,100
3	$390,000	$50,600
2	$300,000	$38,900
Number of trucks needed	**Cost of Truck**	
100 km		
7	$574,000	$74,400
4	$520,000	$67,400
3	$450,000	$58,300

Table 15: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (15 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hr/day)	8	8	8
Total Time at work (hr/day)	7.1	4.1	3.5
Salary ($/hour)	20	20	20
Total ($/day)	142	82.2	70.3
Fuel Cost			
Actual fuel cost ($/liter)	0.82	0.82	0.82
Total distance traveled in (km/day)	204	112	94
km/liter	2.41	2.41	2.41
Total ($/year)	25,400	13,900	11,700
Annualized Fixed Cost			
Vehicle Purchase ($/year)	21,300	16,900	19,500
Fixed Annual Cost			
Insurance Cost ($/year)	6,500	6,500	6,500
Vehicle registration ($/year)	251	251	251
Total ($/year)	6,750	6,750	6,750
Total Variable Cost			
Fuel ($/year)	25,400	13,900	11,700
Maintenance and repairs ($/year)	22,400	12,300	10,300
Tires ($/year)	1,900	1,100	900
Labor cost ($/year)	51,800	30,000	25,700
Total Variable Cost ($/year)	101,000	57,200	48,400
Summary			
Annualized fix cost ($/year)	21,300	16,900	19,500
Fixed Annual ($/year)	6,750	6,750	6,750
Total Variable Cost ($/year)	101,000	57,200	48,400
Total Transportation Cost ($/year)	**130,000**	**80,800**	**74,600**
Total Transportation Cost ($/$m^3$)	**3.56**	**2.23**	**2.05**

Table 16: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (20 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hr/day)	8	8	8
Total Time at work (hr/day)	8.0	4.5	3.9
Salary ($/hour)	20	20	20
Total ($/day)	160	91	77.3
Fuel Cost			
Actual fuel cost ($/liter)	0.82	0.82	0.82
Total distance traveled in (km/day)	270	145	120
km/liter	2.41	2.41	2.41
Total ($/year)	33,500	18,000	14,900
Annualized Fixed Cost			
Vehicle Purchase ($/year)	31,900	33,700	19,500
Fixed Annual Cost			
Insurance Cost ($/year)	6,500	6,500	6,500
Vehicle registration ($/year)	251	251	251
Total ($/year)	6,750	6,750	6,750
Total Variable Cost			
Fuel ($/year)	33,500	18,000	14,900
Maintenance and repairs ($/year)	29,500	15,900	13,100
Tires ($/year)	2,500	1,300	1,100
Labor cost ($/year)	58,100	33,200	28,200
Total Variable Cost ($/year)	124,000	68,400	57,300
Summary			
Annualized fix cost ($/year)	31,900	33,700	19,500
Fixed Annual ($/year)	6,750	6,750	6,750
Total Variable Cost ($/year)	124,000	68,400	57,300
Total Transportation Cost ($/year)	**162,000**	**109,000**	**83,500**
Total Transportation Cost ($/$m^3$)	**4.5**	**3.0**	**2.3**

Table 17: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (25 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hr/day)	8	8	8
Total Time at work (hr/day)	8.8	5.0	4.2
Salary ($/hour)	20	20	20
Total ($/day)	176	100	84
Fuel Cost			
Actual fuel cost ($/liter)	$0.82	$0.82	$0.82
Total distance traveled in (km/day)	335	177	146
km/liter	2.41	2.41	2.41
Total ($/year)	41,700	22,100	18,200
Annualized Fixed Cost			
Vehicle Purchase ($/year)	31,900	33,700	38,900
Fixed Annual Cost			
Insurance Cost ($/year)	6,500	6,500	6,500
Vehicle registration ($/year)	251	251	251
Total ($/year)	6,750	6,750	6,750
Total Variable Cost			
Fuel ($/year)	41,700	22,100	18,200
Maintenance and repairs ($/year)	36,700	19,500	16,000
Tires ($/year)	3,100	1,600	1,400
Labor cost ($/year)	64,500	36,400	30,800
Total Variable Cost ($/year)	146,000	79,500	66,200
Summary			
Annualized fix cost ($/year)	31,900	33,700	38,900
Fixed Annual ($/year)	6,750	6,750	6,750
Total Variable Cost ($/year)	146,000	79,500	66,200
Total Transportation Cost ($/year)	**185,000**	**120,000**	**112,000**
Total Transportation Cost ($/$m^3$)	**5.1**	**3.3**	**3.1**

Table 18: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (30 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hr/day)	8	8	8
Total Time at work (hr/day)	9.5	5.3	4.5
Salary ($/hour)	20	20	20
Total ($/day)	191	107	90
Fuel Cost			
Actual fuel cost ($/liter)	0.82	0.82	0.82
Total distance traveled in (km/day)	387	204	167
km/liter	2.41	2.41	2.41
Total ($/year)	48,200	25,400	20,800
Annualized Fixed Cost			
Vehicle Purchase ($/year)	31,900	33,700	38,900
Fixed Annual Cost			
Insurance Cost ($/year)	6,500	6,500	6,500
Vehicle registration ($/year)	251	251	251
Total ($/year)	6,750	6,750	6,750
Total Variable Cost			
Fuel ($/year)	48,200	25,300	20,800
Maintenance and repairs ($/year)	42,500	22,400	18,300
Tires ($/year)	3,500	1,900	1,500
Labor cost ($/year)	69,600	39,000	32,900
Total Variable Cost ($/year)	164,000	88,400	73,400
Summary			
Annualized fix cost ($/year)	31,900	33,700	38,900
Fixed Annual ($/year)	6,750	6,750	6,750
Total Variable Cost ($/year)	164,000	88,400	73,400
Total Transportation Cost ($/year)	**202,000**	**129,000**	**119,000**
Total Transportation Cost ($/$m^3$)	**5.6**	**3.5**	**3.3**

Table 19: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (60 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hr/day)	8	8	8
Total Time at work (hr/day)	14.4	7.8	6.5
Salary ($/hour)	20	20	20
Total ($/day)	$288	$155	$129
Fuel Cost			
Actual fuel cost ($/liter)	$0.82	$0.82	$0.82
Total distance traveled in (km/day)	755	387	314
km/liter	2.41	2.41	2.41
Total ($/year)	93,800	48,200	39,000
Annualized Fixed Cost			
Vehicle Purchase ($/year)	53,100	50,600	38,900
Fixed Annual Cost			
Insurance Cost ($/year)	$6,500	$6,500	$6,500
Vehicle registration ($/year)	$251	$251	$251
Total ($/year)	$6,750	$6,750	$6,750
Total Variable Cost			
Fuel ($/year)	93,800	48,100	39,000
Maintenance and repairs ($/year)	82,700	42,500	34,400
Tires ($/year)	6,900	3,600	2,900
Labor cost ($/year)	106,000	56,900	47,100
Total Variable Cost ($/year)	289,000	151,000	123,000
Summary			
Annualized fix cost ($/year)	53,100	50,600	38,900
Fixed Annual ($/year)	6750	6750	6750
Total Variable Cost ($/year)	289,000	151,000	123,000
Total Transportation Cost ($/year)	**349,000**	**208,000**	**169,000**
Total Transportation Cost ($/$m^3$)	**9.6**	**5.7**	**4.7**

Table 20: Total Transportation Cost in U.S. Dollars per m^3 as a Function of Distance (100 km)

Unit Transportation Cost			
Truck Capacity (m^3)	7.6	15.1	19
Work Schedule (trips/day)	13	7	5
Labor Cost			
Work Schedule (hrs/day)	8	8	8
Total Time at work (hr/day)	21	11	9
Salary ($/hour)	20	20	20
Total ($/day)	418	220	181
Fuel Cost			
Actual fuel cost ($/liter)	0.82	0.82	0.82
Total distance traveled in (km/day)	1,240	630	508
km/liter	2.41	2.41	2.41
Total ($/year)	154,000	78,300	63,100
Annualized Fixed Cost			
Vehicle Purchase ($/year)	74,400	67,400	58,300
Fixed Annual Cost			
Insurance Cost ($/year)	6,500	6,500	6,500
Vehicle registration ($/year)	251	251	251
Total ($/year)	6,750	6,750	6,750
Total Variable Cost			
Fuel ($/year)	154,000	78,300	63,200
Maintenance and repairs ($/year)	136,000	69,000	55,700
Tires ($/year)	11,300	5,700	4,600
Labor cost ($/year)	153,000	80,500	66,000
Total Variable Cost ($/year)	454,000	234,000	190,000
Summary			
Annualized fix cost ($/year)	74,400	67,400	58,300
Fixed Annual ($/year)	6,750	6,750	6,750
Total Variable Cost ($/year)	454,000	234,000	190,000
Total Transportation Cost ($/year)	**535,000**	**308,000**	**255,000**
Total Transportation Cost ($/$m^3$)	**14.7**	**8.5**	**7.1**

APPENDIX B: SBR

Table 21: Total Capital Cost for SBR Process in U.S. Dollars per m^3

Equipment Costs	**2018 Cost (U.S. Dollars)**
Total equipment cost	$167,000 *
Piping and installation	$33,400
Electrical and instrumentation	$33,400
Engineering and construction management	$41,800
Structural	$16,700
Civil	$16,700
Total equipment and construction cost	$309,000
Contingency	$92,700
Total capital cost	**$402,000**
20 year cost	**$32,200**
Total capital cost ($/$m^3$)	**$0.89**

- Two blowers installed
- Automatic valve costs are converted from 2016 cost

Table 22: Total O&M Cost for SBR Process in U.S. Dollars per m^3

Parameters	Value
Oxygen required	
Influent substrate concentration as BOD (S_0) (g/m^3)	1600
$P_{X,bio}$= A+B+C (Equation 11&12) (kg/day)	38.7
A (g/day)	29100
B (g/day)	3200
C (g/day)	6400
NO_X (g/m^3)	670
OTR (kg O_2/hr)	17.3
OTR/SOTR	0.30
SOTR (kg/hr)	57.7
Air flowrate (m^3/sec)	0.18
Aeration Power	
Power requirement for each blower (kW)	11.8
Power requirement for each blower ($/year)	10,300
Aeration energy (kWh/day)	187
Cost of aeration ($/year)	6,800
Total Power Cost ($/year)	**17,200**
Labor ($/year)	**76800**
Chemical cost (methanol) ($/year)	**6600**
Total O&M cost ($/year)	**101,000**
Total O&M cost ($/$m^3$)	**2.8**

- Power costs are calculated at 0.1/kW-hr

APPENDIX C: MBR

Table 23: Total Capital Cost for MBR Process in U.S. Dollars per m^3

Equipment Costs	2018 Cost (U.S. Dollars)
Total equipment cost	$225,000
Piping and installation	$45,000
Electrical and instrumentation	$45,000
Engineering and construction management	$56,100
Structural	$22,500
Civil	$22,500
Total equipment and construction cost	$415,000
Contingency	$125,000
Total capital cost	**$540,000**
20 year cost	**$43,300**
Total capital cost ($/$m^3$)	**1.19**

Table 24: Total O&M Cost for MBR Process in U.S. Dollars per m^3

Parameters	Value
Oxygen required	
$P_{X,bio}$= A+B+C (Equation 11&12) (kg/day)	25.6
A (g/day)	15,400
B (g/day)	8,700
C (g/day)	1700
Preaeration tank	
Oxygen demand in preaeration tank (kg/hr)	9.7
Preaeration power cost ($/year)	5,800
Cost of aeration ($/year)	3900
Membrane tank	
Oxygen demand in membrane tank (kg/hr)	10.8
Membrane tank power cost ($/year)	6,500
Cost of aeration ($/year)	4,300
Mixing power in anoxic zone ($/year)	170
Total power cost ($/year)	**20,500**
Labor ($/year)	**76,800**
Chemical addition	**6,600**
Membrane replacement	**12,000**
Total O&M cost ($/year)	**116,000**
Total O&M cost ($/$m^3$)	**3.2**

APPENDIX D: SGA PROCESS

Table 25: Flow Equalization Tank Cost in U.S. Dollars per m^3

Parameters	Value
Operation/week	3
Operation/day	24
Volume of tank (m^3)	1500
Effluent Flow Rate (m^3/day)	500
Type	Tank
Shape	Cylindrical Tank
Height (m)	15
Base (m^2)	33
Total cost for Tank	**$405,000**
20 year cost	**$32,500**
Cost in U.S. dollars per m^3	**0.18**

Table 26: Total Capital Cost for Struvite Crystallization in U.S. Dollars per m^3

Equipment Costs	2018 Cost (U.S. Dollars)
Total equipment cost	$340,200
Piping and installation	$68,100
Electrical and instrumentation	$68,100
Engineering and construction management	$85,100
Structural	$34,100
Civil	$34,100
Total equipment and construction cost	$629,300
Contingency	$188,800
Total capital cost	**$818,100**
20 year cost	**$65,700**
Total capital cost ($/$m^3$)	**0.36**

Table 27: Total O&M Cost for Struvite Crystallization in U.S. Dollars per m^3

Parameters	Value
Mixing Power	
Dynamic viscosity ($N*s/m^3$)	8.9×10^{-4}
Velocity gradient (1/s)	500
Volume of reactor (m^3)	21
Mixing power (kW)	4.6
Safety factor	1.2
Safe mixing power (kW)	5.5
Cost ($/year)	4,900
Mg Dosage	
P:Mg	1:1.2
Molarity of Mg (kg/l of MgO)	1.6×10^{-5}
Total cost of dosage as MgO ($/year)*	2100
Struvite Fertilizer formed	
MAP effluent $PO_4 - P$ (mg/l)	8.05
phosphate removed (kg-P/day)	4
MAP sludge formed (kg struvite/day)	32
NaOH chemical cost ($/year)	20
Total Power Cost ($/year)	**4900**
Labor ($/year)	**76,800**
Total O&M cost ($/year)	**84,000**
Total O&M cost ($/m^3)	**0.46**

- MgO chemical cost = $700 per ton
- Struvite sale price = $242 per ton
- NaOH chemical cost = $510 per ton
- H_2SO_4 chemical cost = $205 per ton

Table 28: Total Capital Cost for Aerobic Granular Sludge Process in U.S. Dollars per m^3

Equipment Costs	2018 Cost (U.S. Dollars)
Total equipment cost	$809,700
Piping and installation	$162,000
Electrical and instrumentation	$162,000
Engineering and construction management	$203,000
Structural	$81,000
Civil	$81,000
Total equipment and construction cost	$1,500,000
Contingency	$450,000
Total capital cost	**$1,900,000**
20 year cost	**$156,000**
Total capital cost ($/$m^3$)	**0.86**

Table 29: Total O&M Cost for Aerobic Granular Sludge Process in U.S. Dollars per m^3

Parameters	Value
Oxygen required	
$P_{X,bio}$= A+B+C (Equation 11&12) (kg/day)	18.11
A (g/day)	175
B (g/day)	7417
C (g/day)	10500
NO_X (g/m^3)	740
OTR (kg O_2/hr)	78.2
OTR/SOTR	0.3
SOTR (kg/hr)	260
Air flowrate (m^3/sec)	0.8
Aeration Power	
Power requirement for each blower (kW)	69
Power requirement for each blower ($/year)	60,300
Aeration energy (kWh/day)	1090
Cost of aeration ($/year)	39,800
Total Power Cost ($/year)	**101,000**
Labor ($/year)	**76,800**
Chemical cost (methanol) ($/year)	**32,500**
Total O&M cost ($/year)	**210,000**
Total O&M cost ($/$m^3$)	**1.15**

Table 30: Total Capital Cost for PN/A Process in U.S. Dollars per m^3

Equipment Costs	2018 Cost (U.S. Dollars)
Total equipment cost	$1,040,000
Piping and installation	$208,000
Electrical and instrumentation	$208,000
Engineering and construction management	$260,000
Structural	$104,000
Civil	$104,000
Total equipment and construction cost	$1,930,000
Contingency	$577,000
Total capital cost	**$2,500,000**
20 year cost	**$201,000**
Total capital cost ($/$m^3$)	**1.11**

Table 31: Total O&M Cost for PN/A Process in U.S. Dollars per m^3

Parameters	Value
Oxygen required	
Ammonia load per SBR cycle (kg-N/cycle)	6.8
Partial-Nitritation requirement (kg O_2/kg-N)	1.9
Nitrogenous oxygen requirement (kg O_2/cycle)	12.9
OTR (kg O_2/hr)	19.25
OTR/SOTR	0.41
SOTR (kg/hr)	46.4
Air flowrate (m^3/sec)	0.15
Aeration Power	
Power requirement for each blower (kW)	10.3
Power requirement for each blower ($/year)	9,000
Aeration energy (kWh/day)	163
Cost of aeration ($/year)	6,000
Mixing energy (kWh/day)	20.5
Cost of mixing ($/year)	800
Total Power Cost ($/year)	**15,700**
Labor ($/year)	**76,800**
Total O&M cost ($/year)	**92,500**
Total O&M cost ($/$m^3$)	**0.51**

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