Landfill leachate treatment:

Using sequencing batch reactor technology to remove high ammonium concentration

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PHILIP ANDREW R. LU

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Abstract

Landfilling requires various consents and regulations in order to operate most effectively and cause minimal environmental issues to its immediate surroundings. The contaminated liquid produced from precipitation seeping through the layers of waste in landfills are generally called landfill leachate or leachate. Sequencing batch reactors (SBR) are fill-and-draw activated sludge systems that facilitate wastewater treatment in a singular reactor, while utilizing a sequence of different phases to create aerobic, anoxic, and sometimes anaerobic conditions. Multiple studies have been done that demonstrate the capacity of SBRs to treat wastewater for varying strengths of wastewater and concentration of contaminants, including ammonium. The objectives of the research are to: 1. run a lab-scale sequencing batch reactor to treat ammonium concentrations similar to that of landfill leachate; 2. design a full-scale sequencing batch reactor for landfills based on the bench-scale reactor's ammonium removal potential; and 3. analyse the viability and economic feasibility of a SBR for treating landfill leachate.

Two lab-scale SBRs are used in the experiment. SBR 1 was run for 250 days of operation, while SBR 2 was run for only 130 days. These achieved ammonium removal at influent ammonium loading as high as 720 mg/L, and the SBR achieved ammonium removal efficiency of 90% to 100% with a TSS of 17 to 18 g/L during this most stable period of operation. pH control was required at higher influent ammonium loadings above 300 mg/L. Removal rates over the stable periods of operation are calculated to be at 1.65 and 1.55 mg NH_4^+ removed/g TSS/ per cycle for SBR 1 and 2, respectively. An ammonium removal rate of 0.0508 mg NH_4^+/L/min is calculated based on measurements using an ammonium ISE probe. Inhibition in ammonium removal due to low pH is observed. With extended aeration and pH control the ammonium removal continued as long as pH is maintained above 6.5. The oxygen consumption rate is also found to be at 0.01 mg DO/L/min, while the average mass transfer coefficient of O_2 at 0.0144 min^{-1}, and the percentage of oxygen transferred from gas to the mixed liquor calculated to be at 14.8%.

The full-scale SBR design is based on rates calculated from the experimental set-up and approximate leachate flow rate, based on precipitation, in the Waikato region. An area of 49 hectares landfilled is considered, which is similar to one of the class 1 municipal sanitary landfills in the region. A landfill leachate ammonium concentration range of 600 to 1500 mg/L is considered for the full-scale SBR design. The full-scale SBR is calculated to require a working volume of 8168 m^3. The capital and operating cost of which shows high economic feasibility, and is estimated to be cost-effective compared to the cost of offsite treatment.

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1 Introduction

The first chapter discusses a brief background on the study, objectives, and the thesis structure. Included in the brief background are descriptions of the problem being addressed, research gaps, and problem statement of this study. The objective outlines the aims of the study. The thesis structure briefly describes how the following chapters are structured and the flow of discussing this study.

1.1 Background

Solid waste management has continually been improved globally. This includes waste treatment and resource recovery to minimize the amount of waste that end up in landfills. Landfills are ideally mostly composed of the solid waste that have little potential to be recycled or recovered with technology, such as waste-to-energy incinerators. According to the United Nations, the annual rate of solid waste collected is at 11.2 billion tonnes. In 2009, New Zealand also reported that 3.156 million tonnes of its waste was landfilled (*Environment Report Card July 2009*, 2009).

Landfilling requires various consents and regulations in order to operate most effectively and cause minimal environmental issues to its immediate surroundings. Environmental aspects that are of concern with landfills generally include its impact on the air, soil, groundwater, and surface water. The air is affected by gases from the waste piles. The soil and groundwater are susceptible to leaching contaminants from the wide range of material that can be found in a landfill. Nearby surface water can be contaminated by leached contaminants transported by precipitation infiltrating to the groundwater and moving towards nearby water bodies. The contaminated liquid produced from precipitation seeping through the layers of waste in landfills are generally called landfill leachate or leachate.

Landfill leachate is described as contaminated liquid that has percolated through layers of landfilled waste (Foo & Hameed, 2009). This is can be significantly influenced by the waste composition of the landfill that the leachate is coming from (*Environment Report Card July 2009*, 2009). As such, the variation in the waste composition of a landfill tends to generate landfill leachate that are of strong wastewater parameters (Brennan et al., 2017; Capodici et al., 2014), especially compared to typical domestic wastewater (Davis & Cornwell, 2013). Ammoniacal nitrogen is a wastewater parameter of concern, since it is quite toxic to aquatic life if discharged in high concentrations to surface water bodies. Indication of leaching nitrogen into aquatic environments includes eutrophication and acidification (Camargo & Alonso, 2006). Liu and Liptak (2020) categorizes ammoniacal nitrogen as an oxidizing and reducing agent, which causes chemical imbalances in environments it is leaked to in high concentrations, such as those measured from landfill leachate. Typical inorganic nitrogen is referred to as nutrients as a pollutant, which can lead to undesirable growth certain aquatic life causing an imbalance. Due to the large volume of wastes in landfills, ammoniacal nitrogen (NH_4-N) tends to be of high concentration in landfill leachate (Haslina et al., 2021). Landfill leachate is also not always controlled, as it is in sanitary landfills. Unlined landfills and open dumpsites are likely sources for contaminant transport to groundwater and nearby surface water bodies due to leachate seeping through the soil subsurface.

Many wastewater treatment methods currently integrate biological nutrient removal with stricter discharge standards being implemented. One of the more popular methods are activated sludge systems. These systems typically utilize the presence of nitrifying and denitrifying bacteria in a sludge to remove nitrogen from wastewater. Activated sludge processes are used in many wastewater treatment facilities serving wastewater from domestic, industrial, and commercial sources. Domestic wastewater treatment is widely practiced and well-studied. Industrial and commercial sources may require specific technologies to discharge treated water that is up to standard. Compared to these sources, landfill leachate is often more complex as it is full of a wide range of contaminants due to the wide range of wastes in a landfill. Co-treatment of landfill leachate alongside domestic wastewater in wastewater treatment plants have been studied to be effective at a dose of 10% leachate to volume of domestic wastewater being treated, and causes minimal changes to the biological treatment systems being used (Brennan et al., 2017; Capodici et al., 2014; Danley-Thomson et al., 2020).

Thus, treatment of landfill leachate especially for ammoniacal nitrogen is important as the high concentration of it can cause immediate changes to the environment if not managed properly (Camargo & Alonso, 2006; Haslina et al., 2021). Domestic wastewater treatment plants typically take up large footprints with the utilization of different tanks for different phases of treatment. The advantage of a sequencing batch reactor is that most phases of treatment can occur in a single tank potentially requiring a much smaller footprint. Landfill leachate production is also relatively lower in flow rate compared to domestic wastewater production, often following precipitation pattern (Fenn et al., 1975), therefore a simpler treatment system such as an SBR would be preferred. Most large landfills, such as class 1 landfills in New Zealand, are not in close proximity to wastewater treatment plants, which makes piping landfill leachate collection systems to a wastewater treatment plant not feasible.

Laboratory-scale SBRs are usually seeded with sludge from a wastewater treatment plant or granular sludge seeds to establish the activated sludge for facilitating biological nutrient removal. There is a research gap in utilizing organic waste materials in establishing activate sludge, such as cow manure. The potential of which is that leachate treatment plants operate in disposal facilities that do accept a significant amount of organic matter in the disposed waste. In New Zealand, in particular, the high frequency of dairy farms makes cow manure readily available. This study also explores using cow manure to seed and establish an activated sludge system, in the configuration of a SBR, to treat high ammonia wastewater, such as landfill leachate. Therefore, this thesis will examine the development of a lab-scale SBR for removing the high ammonia concentration in landfill leachate, and use the findings to design a full-scale SBR.

1.2 Objectives

The removal of ammonia from landfill leachate is important in ensuring the safety of the aquatic life and health of water bodies receiving the discharge from landfills. This research aims to investigate the feasibility of using a sequencing batch reactor to treat the high ammonia concentration from landfill leachate. The objectives of the study are outlined as follows:

1. To run a bench-scale sequencing batch reactor to treat ammonium concentrations similar to that of landfill leachate;
2. To design a full-scale sequencing batch reactor for landfills based on the bench-scale reactor's ammonium removal potential; and
3. To analyse the viability and economic feasibility of a sequencing batch reactor for treating landfill leachate.

1.3 Thesis Structure

The study starts with reviewing relevant literature in the topics of landfills, landfill leachate, biological nutrient removal, landfill leachate treatment, and sequencing batch reactors, which can be found in Chapter 2. Based on the information from the literature review, the experimental set-up for a laboratory-scale sequencing batch reactor is designed and tests are run on it. Methodology for the experimental set-up and analytical tests are discussed in Chapter 3. The experimental set-up is focused on measuring ammonium in the influent, reactor, and effluent to calculate the removal potential of a SBR at high ammonium loading near those detected in landfill leachate based on literature. The results of which are collated and discussed in Chapter 4. Calculation of ammonium removal potential, rates, and parameters are dicussed in the analysis found in Chapter 5. In the same chapter, the findings from the experimental set-up and literature are then used to design a full-scale SBR to treat landfill leachate. The full-scale SBR is to be validated with typical dimensions of full-scale SBRs. The economic feasibility of the full-scale SBR sizing and operational requirements is assessed to determine if a SBR for landfill leachate treatment is a viable option also discussed in Chapter 5. The summary and conclusions from both the experimental study and design study are presented in Chapter 6. Suggestions and recommendations for further studies on landfill leachate treatment using a lab-scale SBR and full-scale SBR design for landfill leachate are discussed in Chapter 7.

2 Literature Review

The following sections discusses relevant literature to the topics of landfills, landfill leachate, biological nutrient removal, landfill leachate treatment, and sequencing batch reactors. These topics are the most significant in relation to the experimental lab-scale SBR, full-scale design, and economic feasibility assessment.

2.1 Landfill Leachate

This section discusses landfills, leachate production, leachate classes and characteristics, and landfill leachate contamination. The discussion on landfills focuses on the nature and characteristics of a landfill that affect the leachate it produces. Leachate production focuses on literature that discusses the biological, chemical, and physical processes that generate leachate. Literature for leachate classes and characteristics discusses the relevance of landfill age on the classification of landfill leachate and notable characteristics from the landfill's age. Cases where landfill leachate has caused some contamination and its harmful effects if unmanaged is also discussed.

2.1.1 Landfills and Leachate

Landfilling is considered as the most economical way to dispose of solid wastes, with up to 95% of solid waste generated ending up in landfills (Scott et al., 2005). Waste that cannot be diverted for recovery or reuse ideally makes up most of the composition in landfill waste. Due to the large volume of wastes in landfills, the environmental impacts and public health consideration should be accounted in this field (Pazoki & Ghasemzadeh, 2020). Dumpsites differ from sanitary landfills since dumpsites are just areas with piles of waste on top of soil, without any lining to minimize leaching of toxic substance into the soil subsurface and the groundwater. These pose an even higher risk to the environment and public health due to the lack of engineering involved. Pollutant flow is most likely higher with dumpsites (Kumar Tyagi & Ojha, 2023). The procedures of lining and capping, layering, compaction, and leachate recirculation describe the proper management of waste in landfills (Scott et al., 2005).

Landfills generally generate methanogenic leachate quality for most of its operational life (Robinson, 2007). Robinson (2007) studied multiple very large landfills from several countries and found that concentrations of ammoniacal-N often exceeded 1000 mg/L up to 3000 mg/L. In the same study, BOD_5 of less than 1000 mg/L is common along with COD values from 2000 to 8000 mg/L. Relative to domestic sewage heavy metal concentrations are lower in landfill leachate, with the exception of chromium for some landfills (Robinson, 2007).

The following table from Robinson (2007) summarizes the pH, COD, BOD_5, and NH_4-N from different landfills internationally.

Table 2-1. Summary of pH, COD, BOD_5, and NH_4-N from different landfills (Robinson, 2007)

Landfill Leachate Source	pH	COD (mg/l)	BOD_5 (mg/l)	NH_4-N (mg/l)
Hong Kong				
Pillar Point Landfill	8.1 - 8.6	2460 - 2830	135 - 384	1190 - 2700
Ma Yau Tong Landfill	7.6 - 8.1	641 - 873	57 - 117	784 - 1156
Jordan Landfill	7.4	680	43	1240
United Kingdom				
Arpley Walton Phase	8.0	8260	3020	2110
Lapwing Phase	8.1	8510	495	2050
Lalnddulas 02/2000	8.2	3140	112	1240
Winterton 01/2000	8.5	2620	370	1460
Site X 02/2000	8.2	4600	476	1590
Site Y 02/2000	7.4	1760	89	1100
Site Z A 03/2000	7.7	6940	1030	1760
Site Z B 03/2000	7.9	3630	193	1710
South Africa				
Bisasar Road (combined flow)	7.5 - 7.7	1382 - 2488	300	886 - 1919
Bisasar Road (Randles Cell)	7.6 - 7.8	2021 - 3949	-	957 - 1060
Mariannhill	7.6 - 7.8	733 - 1605	300 – 695	387 - 675
Vissershok	8.1 - 8.3	3382 - 5766	627	1998 - 2894
Island of Mauritius				
Mare Chicose	6.08 – 7.76	4031 - 6739	1046 - 1606	1080 – 1436
New Zealand				
Rosedale Road 2 1986 to 1987	7.19	1181	76	-
Hamilton 1 1987	6.98	1969	737	-
Predicted for a North Waikato Landfill Site	7.5 - 8.5	2000 - 4000	400 - 1000	600 - 1000
South Korea				
Site A	7.52	4601	582	1740
Site B	7.93	3989	682	860
Site G	8.29	1840	342	1075
Site L	7.98	4109	1058	1176
Thailand				
Lat Krabang, Bangkok	7.6	2700	-	3032
Kamphang Sein, Bangkok	7.2	1560	-	1424
San Sai, Chian Mai	7.0	1980	-	1350
Indonesia				
Leuwigadja, Bangung	8.4	-	-	2000

It should be noted that lower ranges of ammoniacal nitrogen concentrations are from South Africa, and New Zealand. Mohobane (2008) has done a study on the landfill leachate characteristic of the Horotiu landfill in New Zealand. The ammoniacal nitrogen range from the study is from 227 to 1270 mg/L. The landfill site has been closed since 2006, but bears similarity to leachate from other landfills in New Zealand (Mohobane, 2008). The upper limit of the ammoniacal nitrogen concentration from Horotiu landfill is similar to what is predicted for the North Waikato Landfill Site in the study by Robinson (2007) at the time. This large landfill site is most likely to be already operating, with a landfill in North Waikato currently serving Auckland, Waikato, and Bay of Plenty. Of the five consented class 1 landfills across the country, the Hampton Downs Landfill in the Waikato Region is the biggest in the North Island and is likely the described North Waikato landfill site from Robinson (2007) currently in operation. This serves the regions of Waikato, Bay of Plenty, and Auckland (Tonkin & Taylor Ltd, 2019) receiving almost a 600,000 tonnes of waste annually (*Waste in the Waikato Region*, 2022). The landfill has a total land area of 87 hectares with a total capacity of 30 million cubic metres. The following figure shows the open and closed, both consented and non-consented, landfills in the Waikato Region.

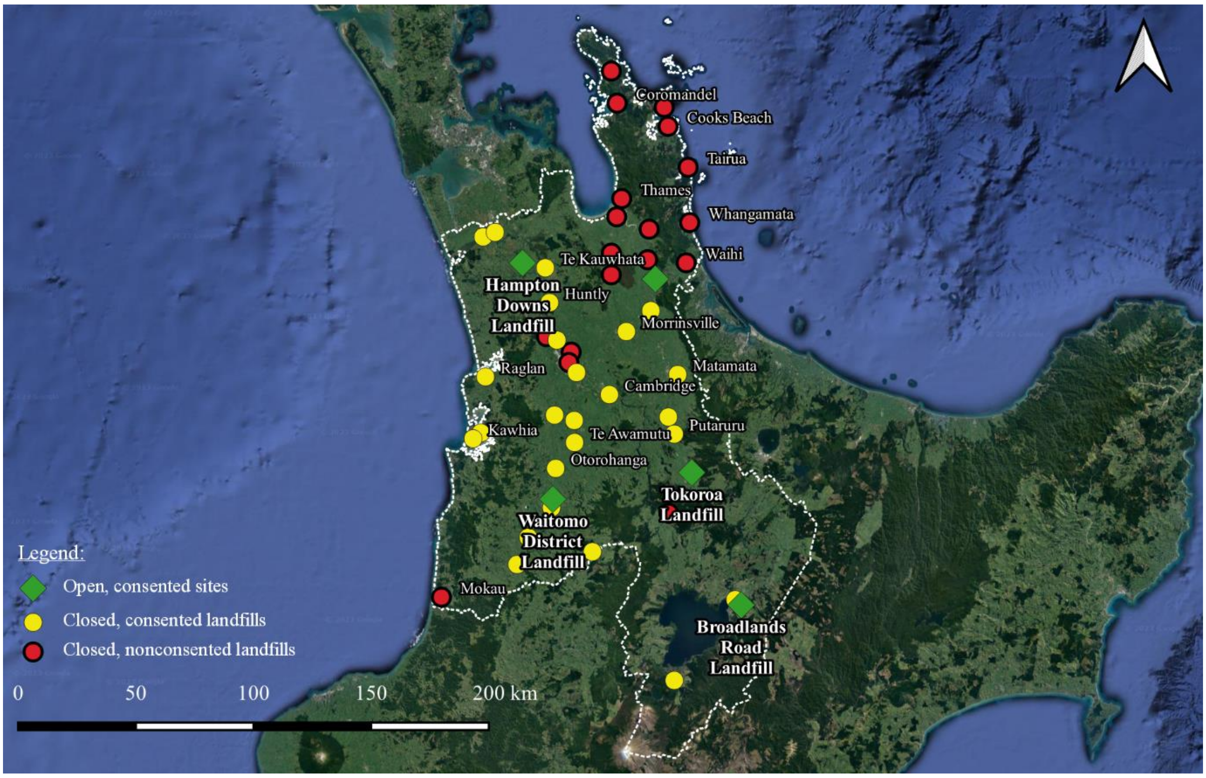

Figure 2-1. Open and closed landfills in the Waikato Region (Google, 2024b; Waste in the Waikato Region, 2022)

There are five consented landfills in the Waikato region. These include the Hampton Downs Landfill, Waitomo District Landfill, Tokoroa Landfill, Broadlands Road Landfill, and Tirohia Landfill. The governing body of the New Zealand government for its landfill management is the Ministry of Environment. Data on landfills in New Zealand is based off self-reported information from the operators. This was started in 2009 with the Waste Minimisation Act 2008 with class 1 municipal landfills starting the self-reporting of data. There are two class 2 landfills and 13 class 3 and 4 landfills in the Waikato Region (Ministry for the Environment, 2023). Class 1 refers to landfills that accept municipal, industrial, commercial, and demolition and construction waste. Class 2 landfills are those that take virgin waste such as clay, rock, and soil from earthworks and land development work. The following table summarizes the landfill classes in New Zealand from the Waste Management Institute of New Zealand.

Table 2-2. *Landfill Classes in New Zealand (WasteMINZ, 2016)*

Class	Waste Accepted
Class 1	Clean fill material, managed fill material, construction and demolition waste, municipal solid waste, household waste, commercial waste, and industrial waste
Class 2	Clean fill material, managed fill material, construction and demolition waste, and non-putrescible industrial waste
Class 3	Clean fill material, and managed fill material
Class 4	Clean fill material

Ever since data reporting on disposed waste in New Zealand was mandated, the total amount of waste disposed in landfills have been measured. Based on the data from the Ministry for the Environment, there has been an increase in waste disposed in landfills since 2009. It should be noted that the reported net tonnage of waste disposed in 2009 was not representative of all reporting landfills in New Zealand. Class 2 landfills also only began reporting in 2022 as shown in Figure 2-2.

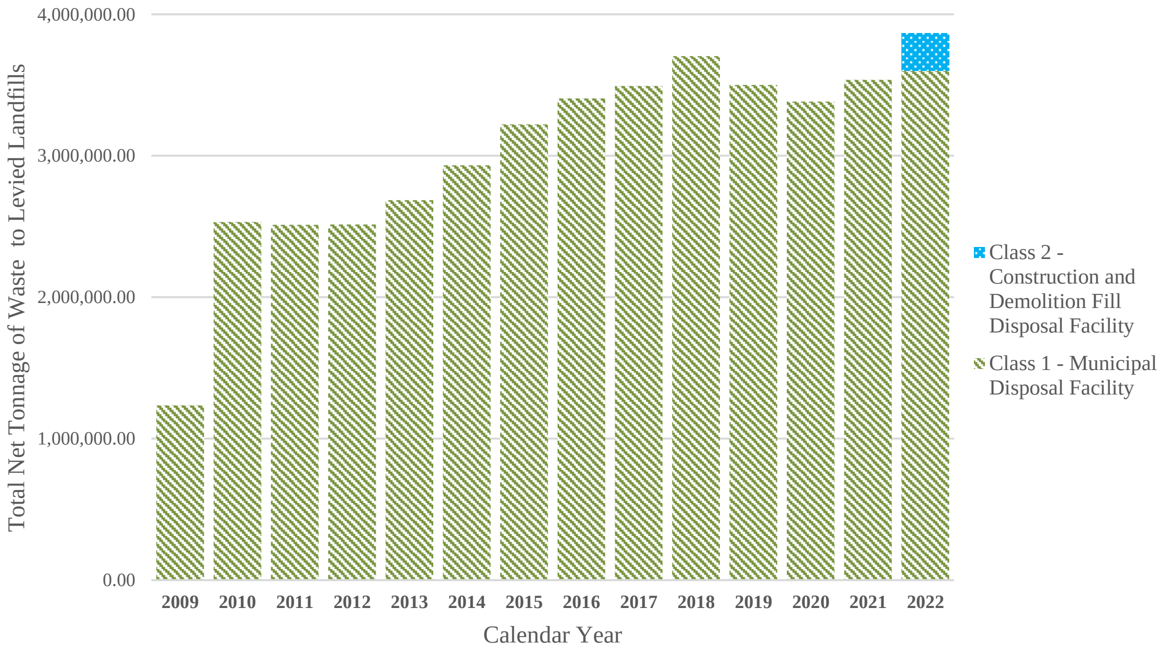

Figure 2-2. *Total net tonnage of waste to levied landfills per year in New Zealand (Ministry for the Environment, 2023)*

With almost more than 3 million net tons of waste disposed at levied landfills in recent years, the per capita waste disposed in class 1 landfills in New Zealand is estimated to be around 706.5 kgs (Ministry for the Environment, 2023). This is contrast to the per capita residential waste generation rate from the National Waste Data Report of 1997 to be at 401 kg. From the same report, 3.18 million tonnes of waste was estimated to have been landfilled in 1995 with around half of this from residential waste and the other from industrial waste. The comparison of the historical data and latest data tells the trend of increasing waste generation and disposal over the past two decades.

This could be an indication of current schemes for waste diversion and recycling may not be able to keep up with higher waste generation (Ministry for the Environment, 1997). This should also be considered in the context of an increasing population over the past two decades. Improvements in waste data reporting from the Waste Minimisation Act of 2008 can also be a factor in the near doubling of waste disposal in landfills as there could have been underreporting occurring before the said act. These, however, do not eliminate the fact that solid waste disposal is a growing issue as the country continues to develop and should be given national importance in policy creation and implementation.

It should also be noted that the dip from 2020 to 2022 compared to 2018 and 2019 could possibly have something to do with the COVID-19 pandemic that affected the whole world. The reduction in waste disposal to landfills could be from the population generally consuming less over this period. A study conducted on the 2020 COVID-19 lockdown in New Zealand found that more New Zealanders were less frequently throwing out food waste, which likely contributes to the lower net tonnage of waste disposed in landfills then (Sharp et al., 2021).

The composition of waste disposed to class 1 landfills, which accepts the most types of waste (Table 2-2), was analysed by New Zealand's greenhouse gas inventory. The waste type found at the highest percentage are inert waste. This type of waste is characterized to non-greenhouse gas emitting waste, including plastics. Food wastes disposed to the class 1 landfills is noted to have decreased over the past few years. The waste composition of landfills is an important factor in the characteristics of the leachate generated. Certain waste types found in high percentages can contribute to higher concentrations of certain contaminants in the landfill leachate generate. The following figure from the Ministry for the Environment summarizes the class 1 waste composition over the past ten years.

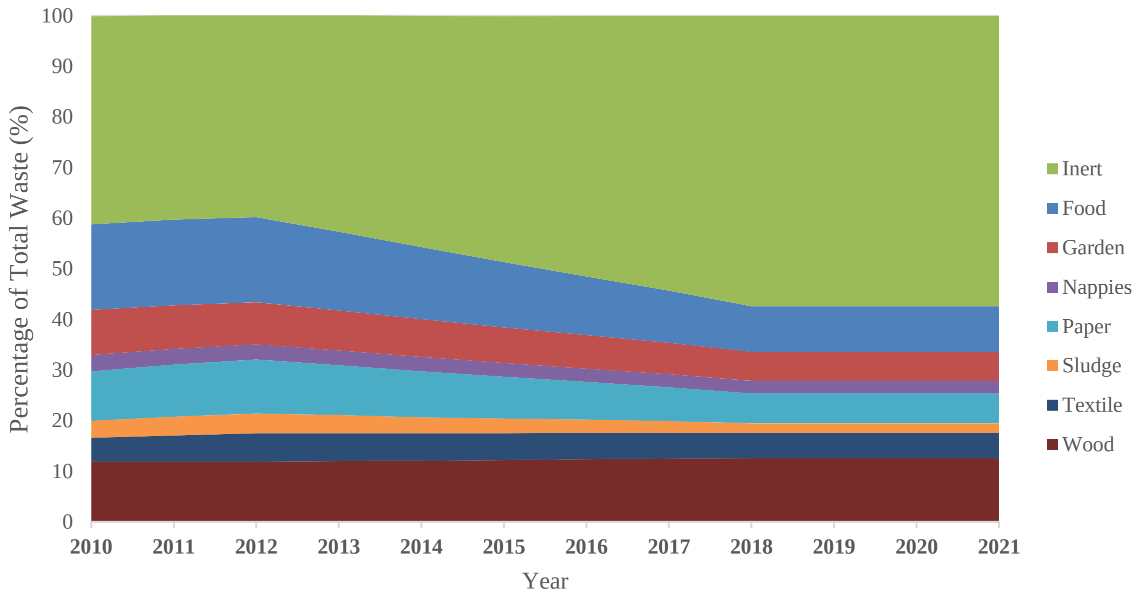

Figure 2-3. Composition of waste disposed to class 1 landfills over time (Environment, 2023)

With the increasing trend in waste disposed to landfills and multitude of landfills in New Zealand, the problem of landfill leachate generated from these must be addressed as the contaminant concentration in leachate may increase along with the trend. Proper treatment infrastructure to treat landfill leachate should be given attention as co-treatment with domestic wastewater treatment plants may cause these plants to run above their design capacity. Aside from this, co-treatment of landfill leachate in these domestic treatment plants, which is the common practice in New Zealand currently, may result in the disruption of biological treatment conditions due to shock loading of certain target contaminants that is higher in landfill leachate compared to domestic wastewater. Organic matter measured as COD and ammonia in landfill leachate usually exceed typical wastewater discharge concentrations (Trabelsi et al., 2013 as cited in A. H. Jagaba et al., 2021). One of these contaminants are ammoniacal nitrogen which is typically only up to 100 mg/L in strong domestic wastewater but beyond this for landfill leachate (Wijekoon et al., 2022).

Leachate is an important factor in the design and construction of landfills. The geological aspect of siting a landfill considers a site subsurface that can contain leachate in the case the lining system fails. Low permeability material is preferred to underlay a landfill, which minimizes the potential for migration of leachate. Along with this, siting a landfill above drinking water supplies, such as aquifers, is undesirable due to the risk of contamination via leachate migration. Typically, landfills are lined to ensure that leachate generated from the landfill is contained, collected, and treated. A leachate management system where the leachate is discharged to a sewerage system, subsurface irrigation, natural water, or recirculation to the landfill (*Landfill Guidelines - Towards Sustainable Waste Management In New Zealand*, 2000). Landfill operation is also significantly based on leachate containment, where typically the leachate is allowed to accumulate along the liner and must be drained out and treated. The cover used by a landfill also affects the potential leachate generated since a cover can also divert rainfall to the landfills stormwater drainage system with minimal contact with the waste. The following figure from Serdarevic (2018) shows typical cover and leachate collection systems for landfills.

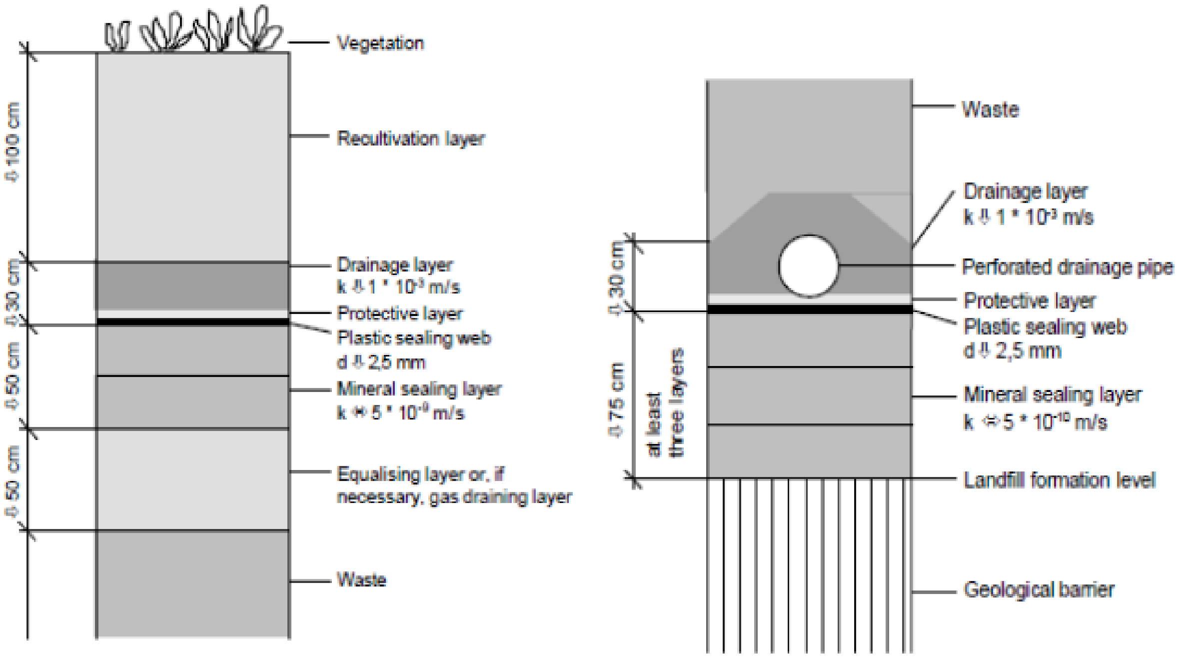

Figure 2-4. Typical landfill cover and base systems (Serdarevic, 2018)

The following section discuss leachate production based on what happens to the waste in a landfill over time and the sources for water to enter the landfill.

2.1.2 Leachate Production

Landfill leachate is the product of water, usually from precipitation, leaching various contaminants from the waste of a landfill and seeping into the subsurface or collection system of a landfill. Other sources for the water that generates leachate can also be from the moisture in landfilled solid waste, liquid waste that may also be disposed of in the site and infiltration of groundwater intro the landfill (Bandala et al., 2021; A. H. Jagaba et al., 2021). Residual water from and biological degradation of organic matter also contribute to leachate generated from landfills (Peng, 2017). Due to the process of leachate generation, leachate is typically formed at the bottom of a landfill (Pazoki & Ghasemzadeh, 2020).

Waste stabilization is an important consideration in the assessment of landfill leachate. This involves the different phases that landfilled wastes undergo in landfills. Pazoki and Ghasemzadeh (2020) describes five phases based from landfill research studies that typically occur in landfills. The first phase is the initial adjustment phase wherein the acclimation of an active microbial community occurs. This is accompanied by the initial placement of solid waste in the landfill with moisture content still accumulating.

This is followed by the transition phase, where oxygen in the landfilled waste is depleted facilitating the transition from an aerobic to an anaerobic condition. Without oxygen, nitrates and sulphates become the electron acceptors in this phase.

As the solid waste continues undergoing hydrolysis with degradation of organic matter, high amounts of volatile organic acids are formed. This third phase is called the acid formation phase and is indicated by significant growth of acidogenic bacteria.

Over time, these acids are consumed by methanogenic bacteria and converted to methane and carbon dioxide. While still in an anaerobic condition, the nitrates and sulphates are turned into ammonia and sulphide, respectively. This is known as the methane fermentation phase. There is also an elevation in pH at this phase that helps the growth of methanogenic bacteria.

The final phase of waste stabilization in a landfill is when there is relatively low or negligible biological activity due to the lack of nutrients and substrate. Leachate quality at this stage is relatively steady and generated at lower quantity. Pazoki and Ghasemzadeh (2020) describes this as the maturation phase, and summarizes constituent parameters of each phase in the following table. These phases can be indicators of expected leachate strength and quality due to leachate generation being dependent on the waste composition and condition of a landfill.

Table 2-3. Parameter ranges for landfills at waste stabilization phases (Pazoki & Ghasemzadeh, 2020)

Parameter	Phase 2: Transition	Phase 3: Acid Formation	Phase 4: Methane Formation	Phase 5: Maturation
COD (mg/L)	4,800 – 18,000	1,500 – 71,000	580 – 9760	31 – 900
Total Volatile Acids (mg/L as acetic acid)	100 – 3,000	3,000 – 18,800	250 – 4,000	0
NH_3-N (mg/L)	120 – 125	2 – 1030	6 – 430	6 – 430
pH	6.7	4.7 – 7.7	6.3 – 8.8	7.1 – 8.8
Conductivity (µS/cm)	2,450 – 3,310	1,600 – 17,100	2,900 – 7,700	1,400 – 4,500

As discussed, landfill leachate is produced due to the different phases the landfill undergoes due to biological and chemical processes occurring. As for the flow rate of leachate, this is mainly driven by the presence and addition of water into the waste in a landfill. Fenn et al. (1975) reported on using water balance to predict leachate generation from landfills. The report found that percolation mainly drives the generation of leachate and, while leachate is unlikely to be in constant flow in a year, generation is likely to follow a similar patten to precipitation. According to Zhang et al. (2023), the leachate yield is primarily dictated by precipitation, initial moisture content, and water holding capacity. The type of waste with the highest initial moisture content is often kitchen waste, which is generally higher in developing countries than in developed ones. The water holding capacity is the amount of water the waste can hold after long periods of drainage by gravity. This means that higher holding capacities would mean longer contact for the water and the waste pile (Zhang et al., 2023). Precipitation infiltrating waste piles is commonly used to correlate leachate flow from waste piles since these are often measured by weather stations and can be reasonably estimated. Krause et al. (2023) studied 36 landfill sites in the USA and found that moisture content in landfills mainly come from incoming waste and precipitation with minimal contribution from leachate recycling.

Ehrig (1983) and Serdarevic (2018) notes that general values for leachate flow can be estimated based on precipitation and level done in the landfill. Landfills that are of low compaction have a leachate flow equivalent to about 25 to 50% of the annual precipitation, while those that are highly compacted have a leachate flow about 15 to 25% of the annual precipitation (Serdarevic, 2018). Based on the description, class 1 landfills in New Zealand may be assumed as highly compacted with the requirements and amount of waste these sanitary landfills are designed for. Based on the literature on leachate production, the study acknowledges that full water balance is a more accurate method to estimated leachate generation from a landfill outside of actual data from landfills (Fenn et al., 1975). In order to simplify leachate flow estimation, the 20% of annual precipitation is used in this study to estimate leachate flow per day per hectare.

2.1.3 Leachate Classes and Characteristics

Landfill leachate can be classified according to the age of the landfill. The three landfill leachate classes are young, middle-age, and mature. Young landfill leachate is from landfills operating for less than 5 years, and is usually characterized by high amounts of volatile fatty acids (VFA), organic compounds, and ammonia less than 400 mg/L with an acidic pH. Middle-age landfill leachate come from landfills of 5 to 10 years of age, and typically have a COD/TN ratio of 3 to 6 with biodegradability that can be described as moderate. Mature landfill leachate is from landfills that have operated for over 10 years, and is characterized by low concentration of biodegradable organic compounds, BOD_5/COD ratio less than 0.1, and ammonia concentration greater than 1000 mg/L. These typical values for the landfill classes varies slightly between studies, however, the general description of lower biodegradability and higher ammonia concentration with age is shared (Haslina et al., 2021; A.H. Jagaba et al., 2021; Tejera et al., 2019). Table 2-4 summarizes the typical parameters and age for the landfill classes. Haslina et al. (2021) reports that ammonia in leachate also varies between regions, citing that Asian countries have been found to have higher ammonia in their landfill leachate.

Table 2-4. Landfill Leachate Classes from Different Studies (Haslina et al., 2021; A.H. Jagaba et al., 2021; Tejera et al., 2019)

Study	**Parameter**	**Class of Leachate**		
		Young	**Intermediate**	**Mature**
A.H. Jagaba et al. (2021)	*Age of Landfill (Years)*	Less than 5	5 – 10	More than 10
	pH	Low	-	-
	BOD/TKN	-	-	Low
	BOD/COD	High concentration of organic compounds	-	Less than 0.1
	NH4-N (mg/L)	Less than 400 mg/L	-	More than 1000 mg/L
	COD/TN	-	3 to 6	-
Haslina et al. (2021)	*Age of Landfill (Years)*	Less than 5	5 – 10	More than 10
	pH	6.5 - 7.5 (7.0)	7.0 – 8.0 (7.5)	7.5 – 8.5 (8)
	COD (g/L)	10 – 30 (15)	3 – 10 (5)	Less than 3 (2)
	BOD/COD	0.5 – 0.7 (0.6)	0.3 – 0.5 (0.4)	Less than 0.3 (0.2)
	NH4-N (mg/L)	500 – 1000 (700)	800 – 2000 (1000)	1000 – 3000 (2000)
	COD/NH4-N	5 – 10 (6)	3 – 4 (3)	Less than 3 (1.5)
Tejera et al. (2019)	*Age of Landfill (Years)*	0 – 10	10 – 20	More than 20
	pH	Less than 6.5	6.5 – 7.5	More than 7.5
	BOD_5/COD	0.5 – 1.0	0.1 – 0.5	Less than 0.1
	BOD_5 (mg/L)	More than 4000	1000 – 4000	Less than 400
	COD (mg/L)	More than 10,000	4000 – 10,000	Less than 4000
	TOC (mg/L)	More than 2500	1000 – 2500	Less than 1000
	NH3-N (mg/L)	Less than 400	-	More than 400
	Heavy metals	Low-medium	Low	Low
	COD/NH4-N	VFA (80%)	VFA (5 – 30%), HA, FA	HA and FA (80%)

The early stages of landfill operation, which is less than 5 years, is characterized by mostly aerobic digestion of the large amount of biodegradable organic matter available and a short period of acid fermentation. This produces a high amount of VFAs contributing to an acidic pH, which are characteristic of young landfill leachate (Anna Tałałaj et al., 2021). As the landfill operates longer and oxygen is depleted in the waste piles, anaerobic digestion dominates and generates methane and ammonia. This is why sanitary landfills are also designed with collection systems for the methane alongside systems capturing leachate. As the landfill continues to operate in this methanogenic phase, biodegradability in the landfill leachate decreases and ammonia increases (Haslina et al., 2021).

2.1.4 Landfill Leachate Contamination

Due to the high concentration of toxic contaminants in landfill leachate, discharging of untreated leachate can cause serious contamination of water bodies through infiltration flow through ground and surface water channels (Aziz et al., 2013).

North et al. (2008) has studied seven New Zealand landfill. One of which is the Middlemarch landfill in Central Otago whose closure was due to contamination of the groundwater that serves a drinking water supply downstream. Contamination of leachate in groundwater is one of the important environmental impacts of landfills. New Zealand, as well as other countries in the world, has operated multiple landfills without lining nor leachate management systems in its history. In Waikato, there has been a study done that indicates that the Horotiu landfill has had any adverse effect on the Waikato river despite being located adjacent to the water body (Macaskill, 2006 as cited in Waikato River Authority, 2014).

Nagarajan et al. (2012) studied three different unlined landfill sites in India that show leachate contaminating the groundwater in the vicinity of the landfills. Ammonium is among the contaminants detected in the groundwater from the study, and the high concentration of the contaminants were deemed to be most likely due to the percolation of landfill leachate. Contamination was lower with distance from the landfills.

Leachate contaminating groundwater and extending past the landfill's subsurface is referred to as a leachate plume. Plumes are studied in order to identify the extend of landfill leachate leakage and possible contamination to the surrounding environment (Kumar Tyagi & Ojha, 2023). Leachate pollution index (LPI) is used to measure the polluting potential of landfill leachate that can be used to aid policy making and selection of treatment method. Included in the leachate parameters for LPI calculation is ammoniacal nitrogen (Wijekoon et al., 2022). The index was created because of the risk that landfill leachate leakage poses to the environment.

2.2 Ammoniacal-Nitrogen

Liu and Liptak (2020) categorizes ammoniacal nitrogen or ammonia nitrogen as an oxidizing and reducing agent, which causes chemical imbalances in environments it is leaked to in high concentrations, such as those measured from landfill leachate. Typical inorganic nitrogen is referred to as nutrients as a pollutant, which can lead to undesirable growth certain aquatic life causing an imbalance (Camargo & Alonso, 2006).

Ammonia nitrogen is called as such since it may be found in the form of either ammonium (NH_4) or ammonia (NH_3). In aqueous solutions, such as water, the form ammoniacal nitrogen takes depends on the pH of the solution. For pH higher than 9.75, NH_3 is more present for water, while for pH less than 8.75 NH_4 becomes the predominant form (Li et al., 2020). The equilibrium is in a pH between 9 and 10 as shown in the following figure.

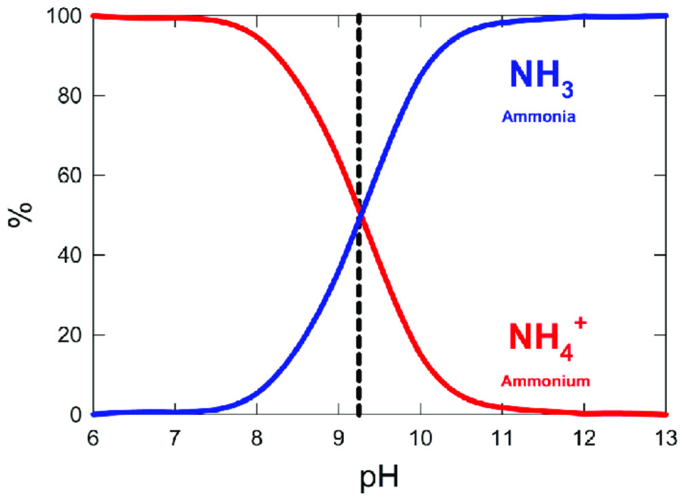

Figure 2-5. Percent abundance of ammonia and ammonium in a closed system (Langenfeld et al., 2021)

Aside from pH, temperature is also a factor in the form that ammonia nitrogen takes in water. Brennan et al. (2017) used the following equation in calculating the free ammonia (NH_3) in water at a given ammoniacal nitrogen (NH_4-N), temperature (T) and pH.

$$Free\ Ammonia = \frac{(NH_4 - N) \times 10^{pH}}{Exp\left[\frac{6334}{273 + T}\right] + 10^{pH}}$$

Equation 2-1. Equation to approximate the concentration of free ammonia (Brennan et al., 2017)

As seen in Table 2-1, the pH of most landfill leachate is generally between 7 to 8.5. This indicates that there is likely presence of ammoniacal nitrogen as both ammonium and ammonia in leachate, with ammonium likely predominating.

Ammonia nitrogen in the environment is more toxic as ammonia compared to ammonium. The ammonia nitrogen can stimulate the unwanted plant and algal growth. Unwanted algae growth in particular is problematic for aquatic systems because the algae compete for nutrients and high number of dead algae cause an increase in dissolved organic matter. This leads to decline in dissolved oxygen, which can lead to anoxic conditions in aquatic ecosystems (Camargo & Alonso, 2006; Li et al., 2020). According to Li et al. (2020) ammonia toxicity becomes an issue for marine fish and invertebrates in anoxic conditions. Although ammonia toxicity is typically considered for ammonia in gas form, there are also risks with ammonia in a liquid for humans. Ammonia dissolved in water forms an ammonium hydroxide solution that is corrosive, similar to ammonia concentration in commercial use solutions. Exposure of ammonia to the skin, eyes, mouth, throat, and stomach causes corrosive damage even in the liquid form. Concentrated solutions of ammonia can also cause pain, inflammation, blisters, and deep burns especially for moist skin areas. Eye irritation is a common effect of ammonia vapor, while concentrated ammonium hydroxide can cause swelling of the eyes that may results to either temporary or permanent blindness. Ingestion of ammonia is commonly followed by nausea, vomiting, and abdominal pain as well (*Medical Management Guidelines for Ammonia*, 2017).

In landfill leachate, ammonia is mainly generated by the degradation of organic matter. However, certain household and commercial solutions may contain 5% to 25% ammonia (*Medical Management Guidelines for Ammonia,* 2017). These mainly include cleaning solutions for household use. Even though disposed containers of these solutions should be empty, the small amounts left behind in these containers can accumulate. Fertilizers are also highly concentrated with ammonia for plant growth (*Medical Management Guidelines for Ammonia*, 2017). Although fertilizers should not typically be found in landfills, trace amounts or leakage into landfills can happen. The variation in direct sources for ammonia in landfills coupled with the degradation of waste material leads to the high concentration of ammonia in landfill leachate, especially for those of mature landfills (Table 2-3).

2.3 Biological Nutrient Removal

Nitrogen removal from wastewater is important as nitrogen is a nutrient that can have adverse effects to ecological balance if discharged at high concentrations, such as those in domestic wastewater and especially landfill leachate. Nitrification and denitrification are two of the most popular methods of biological nutrient removal that is utilized in multiple wastewater systems. These two biological processes can be efficiently utilized in SBR as other biological treatment systems require two separate tanks for each process, while both can occur in an SBR with the adjustment of the different phases. The following subsections discuss the details of each biological process.

2.3.1 Nitrification

Biological nutrient removal is mainly concerned with the removal of nitrogen in the form of ammonia or ammonium in wastewater. The removal of ammonia and/or ammonium is facilitated by nitrifying bacteria. These bacteria are able to turn the ammonia in wastewater to nitrites and/or nitrates in the presence of oxygen. Activated sludge systems ensure that these nitrifying bacteria are present in their sludge such that ammonia removal can be facilitated during periods of aeration. The following equation from Mackenzie L. Davis (2010) summarizes the process of turning ammonia to nitrates.

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$$

Equation 2-2. Nitrification reaction (Mackenzie L. Davis, 2010)

The bacteria responsible for oxidizing ammonia into nitrite (NO2) is called *Nitrosomonas*, while the bacteria that facilitates oxidizing nitrite to nitrate (NO3) is called *Nitrobacter*. *Nitrosomonas* can also be called as an ammonia-oxidising bacteria (AOB), while the *Nitrobacter* a nitrite-oxidizing bacteria (NOB) (Danley-Thomson et al., 2020). Alkalinity is also consumed in nitrification causing a change in pH. The full oxidation reaction requires 4.57 g of O_2/g of nitrogen, most of which comes from nitrite production at 3.43 g O_2/g, and 1.14 g O_2/g for nitrate production. The equivalent of 7.14 g of alkalinity as $CaCO_3$ is required for every gram of ammoniacal nitrogen converted to nitrate nitrogen (Mackenzie L. Davis, 2010).

Heavy metals are contaminants in landfill leachate that may inhibit nitrification. The following table presents heavy metals usually found in landfill leachate and their inhibiting concentration (Brennan et al., 2017).

Table 2-5. Nitrification Inhibiting Value for Heavy Metals (Brennan et al., 2017)

	Cadmium	**Lead**	**Mercury**	**Nickel**	**Arsenic**	**Chromium**	**Copper**
Inhibiting Concentration (mg/L)	1	0.5	0.1	0.5	0.05 – 0.1	1	0.1 – 0.35

According to Haslina et al. (2021), typical concentrations of cadmium and chromium in landfill leachate would be the below the inhibiting values in Table 2-5. However, lead and nickel could be found at inhibiting concentrations in landfill leachate (Haslina et al., 2021). Mohobane (2008) has also reviewed studies that show that concentrations of mercury, arsenic, and copper in landfill leachate can reach inhibiting levels. For this study, inhibiting effects of heavy metals will not be considered in the experiment with the main focus on high ammonia loading from landfill leachate as the target contaminant. With this, it should be considered in further studies the actual effect of these heavy metals in using sequencing batch reactors for landfill leachate treatment given that the heavy metals would tend to settle with the sludge and may accumulate in SBRs with inadequate sludge wasting. The ideal pH range for microbial nitrification is known to be within 7 to 8, with growth of nitrifying microorganisms not occurring in pH under 6.5, as a pure culture (Tarre & Green, 2004). The pH is typically controlled because mixed liquor can become acidic due to the mentioned consumption alkalinity for every gram of ammoniacal nitrogen converted. Although nitrification is sensitive to pH below 7, Tarre and Green (2004) conducted a study using biomass reactors at pH levels around 4 that still exhibited nitrification. Despite this, it is more typical for pH to be maintained above 7 for SBRs that facilitate nitrification (Jeanningros et al., 2010; Liu et al., 2017; Zhou et al., 2020).

Spagni and Marsili-Libelli (2009) studied nitrogen removal from landfill leachate with SBRs through nitrites instead of nitrates, referred to as partial nitritation in literature, to reduce the amount of aeration required and shortcut between the nitrification and denitrification processes. This is due to denitrification usually involving turning nitrates back into nitrites then into nitrogen gas, which will be discussed in the following section.

2.3.2 Denitrification

Denitrification usually occurs following nitrification in nitrogen removal processes. The denitrification process is facilitated by an anoxic condition, wherein there is a lack of free oxygen. The nitrates and nitrites from the previous nitrification process then become oxygen sources, with bacteria stripping oxygen from the nitrates and nitrites generating nitrogen gas that is discharged to the atmosphere (Davis & Cornwell, 2013; Mackenzie L. Davis, 2010). The nitrogen gas can typically manifest as bubble rising from settled sludge in the settling phase of a sequencing batch reactor. The following reaction, also facilitated by denitrifying bacteria, from Davis and Cornwell (2013) summarizes the reaction occurring in the denitrification process.

$$2NO_3^- + Organic\ Matter \rightarrow N_2 + CO_2 + H_2O$$

Equation 2-3. Denitrification reaction (Davis & Cornwell, 2013)

This reaction also produces about alkalinity equivalent to 3.57g $CaCO_3$ per gram of nitrate nitrogen reduced. This means that there is some recovery in alkalinity, and subsequently pH, from denitrification following nitrification (Mackenzie L. Davis, 2010).

Since organic matter is required in the denitrification process, certain systems for nitrogen removal add an organic carbon source during the denitrification phase. The amount of organic matter may also be low due to an aeration phase for nitrification typically preceding the denitrification phase.

One of the challenges of nitrogen removal in landfill leachate is typically low biodegradability, as discussed in Section 2.1, for landfill leachates. This is especially present in landfill leachate that is from landfills that have operated for more than a decade.

2.4 Leachate Treatment

Currently, conventional wastewater treatment methods are challenged by the toxic substances present in landfill leachate. Of the typical treatment processes used today, most do not achieve satisfactory level of degradation for these toxic substances (A.H. Jagaba et al., 2021). Anywhere from 0.05 to 0.2 tons of leachate can be generated by a landfill over its operational life (A.H. Jagaba et al., 2021). Multiple factors affect the composition of leachate, including, but not limited to type of wastes, amount of precipitation, infiltration of groundwater, snowmelt, drainage, erosion, initial moisture content, general soil subsurface characteristics, liner materials, and many others (A.H. Jagaba et al., 2021).

Haslina et al. (2021) has conducted a review of the treatment methods with potential for ammonia removal from landfill leachate. The treatment methods in the review include chemical and physical, biological, and ammonia stripping. The chemical and physical method mainly discussed is coagulation-flocculation, which requires the addition of a coagulant, usually in the form of lime and aluminium sulphate, that causes particle to form larger solids called flocs that can be more easily separated from the mixed liquor. Aluminium sulphate as a coagulant in SBR also results in a shorter aeration phase, and a cleaner and less turbid effluent (Trabelsi et al, 2013 as cited in A. H. Jagaba et al., 2021). This method is often done before biological treatment, and is recommended for mature landfill leachate. Biological treatment is considered to be economical and more feasible than the other methods. However, biological treatment is best for wastewater with high biodegradability, which is not typically the case with mature landfill leachate. The third method is ammonia stripping, which uses aeration to cause a reaction that strips ammonium ions of a hydrogen atom generating ammonia gas. The gas is then treated or converted to ammonium salt by capturing the gas through an acid solution. The difference between aeration for ammonia stripping and biological treatment is that ammonia stripping requires a higher dissolved oxygen concentration and higher pH. Ammonia stripping systems typically maintain a pH of 11 to push the ammoniacal nitrogen balance towards ammonia being the dominant species in an aqueous solution (Haslina et al., 2021). This is in line with Figure 2-5.

Co-treatment of landfill leachate with existing wastewater treatment plants (WWTP) for domestic wastewater is often not preferred due to how it affects the biological processes and effluent quality of the WWTPs (Danley-Thomson et al., 2020). Aside from this, the cost of shipping wastewater from the landfill to a WWTP is also significant. However, Yuan et al. (2016) was able to show that co-treatment of landfill leachate in municipal WWTPs at low mixing ratios can help improve the nutrient removal process without needing pre-treatment. In the same study, the use of raw leachate in an SBR achieved the high total inorganic nitrogen removal compared to SBRs with only municipal wastewater and pre-treated leachate despite having a higher nitrogen concentration in its influent.

The review from A. H. Jagaba et al. (2021) shows that recent leachate treatment plants use a combination of more than one treatment strategy to address the various biodegradable and non-biodegradable substance in landfill leachate. Currently, no single unit process is available to treat leachate with acceptable levels of toxic substance degradation (Bu et al., 2010; D.B. Zhang et al., 2014 as cited in A. H. Jagaba et al., 2021).

For this study, the sequencing batch reactor is considered mainly for its ammonia removing potential with landfill leachate ammonia concentrations. The details of the sequencing batch reactor technology is discussed in the following section.

2.5 Sequencing Batch Reactor

The United States Environmental Protection Agency (US EPA) describes sequencing batch reactors (SBR) as an activated sludge biological treatment that can be used for treating domestic and industrial wastewater at flowrates up about 19,000 m^3/day (*Sequencing Batch Reactors for Nutrification and Nutrient Removal*, 1992). Davis and Cornwell (2013) describes SBRs as a fill-and-draw reactor that utilizes a single reactor where the activated sludge process can be facilitated. Activated sludge processes are where a biological mass is formed and sustained to facilitate biological nutrient removal through the injection of air dissolving oxygen into the wastewater (Mackenzie L. Davis, 2010).

One of the distinct features of the SBR is that the mixed liquor remains in one reactor during the different phases of the treatment. This eliminates the need for a secondary sedimentation tank, since sedimentation is allowed to occur in the SBR (Davis & Cornwell, 2013). Due to the general simplicity of the concept of a sequencing batch reactor, multiple studies have been done that demonstrate the capacity of SBRs to treat wastewater for varying strengths of wastewater and concentration of contaminants, including landfill leachate (Al-Attabi et al., 2017; Anna Tałałaj et al., 2021; Danley-Thomson et al., 2020; Li et al., 2006; Tałałaj, 2022). Configurations for the pH, dissolved oxygen (DO), oxidation-reduction potential (ORP), hydraulic retention time (HRT), sludge retention time (SRT), feed pattern, anoxic-oxic ratio, cycle length, settleability, and temperature in the SBR will depend on the target contaminant and effluent quality required (Rollemberg et al., 2019).

The next subsections discuss the reactor phases, cycle times, sludge retention time (SRT), and laboratory scale SBRs.

2.5.1 Laboratory/Bench-Scale SBRs

A bench-scale study are small-scale systems that handle up to 50 L/d of wastewater. These are used to test if a system can be safely operated and actually treats wastewater sufficiently before scaling-up. High production of extracellular polymers (EPS) or exopolysaccharides, which are viscous and slime-like, could be observed for bench-scale studies. This may cause a decrease in organic matter removal (A. H. Jagaba et al., 2021). The following table summarizes the characteristics of bench-scale SBRs from multiple studies.

Table 2-6. Bench-Scale Sequencing Batch Reactors from Various Studies

Study	Reactor Volume	Cycle Length	Operating Temperature	Sludge Retention Time (SRT)	Experiment Period	Ammonia Removal
Gao et al. (2020)	7.00 L	6 hours	25±3°C	MLSS maintained at 2752 mg/L	40 days	90 to 99%
Spagni and Marsili-Libelli (2009)	24 L	24 hours	20±1°C	20-25 days	300 days	98%
Spagni et al. (2008)	24 L	24 hours	20±0.5°C	-	900 days	98%
Zeng et al. (2003)	4 L	4.8 hours	20±2°C	15 days	-	Not stated but about 99%
Al-Attabi et al. (2017)	5 L	6 hours	20°C	-	60 days	91.5%
Tałałaj (2022)	25 L	20.5 hours	25°C	-	-	95.4% - ammonia nitrogen
Münch et al. (1996)	12 L	6 hours	20±2°C	MLVSS 1620 mg/L	139 days	Not stated
Liu et al. (2017)	3 L and 10 L	8 hours	29±1°C	15 to 30 days	110 days	60 to 80%
Scaglione et al. (2012)	20 L and 2 L	-	-	-	4 and 2 years	80 to 99% and 73 to 93% for sludge source
Zhou et al. (2020)	17 L	10 hours	20 - 29 °C	MLSS 2283 – 4531 mg/L	More than 56 days	96.92%
Danley-Thomson et al. (2020)	8 L	8 hours	-	14 days	More than 6 weeks	20% to 80% - TIN-N removal
Yuan et al. (2016)	3 L	6 hours	-	10 days	117 days	20.6 to 54.6 mg/L
Tarre and Green (2004)	10 L	24 hours	-	-	100 days	Not stated
Capodici et al. (2014)	5 L	24 hours	20±1°C	-	30 days	More than 60% ammonia nitrogen removal
Al-Hazmi et al. (2022)	10 L	24 hours	30±1°C	-	180 days	74% nitrogen removal
Pambrun et al. (2008)	10 L	6 hours	30°C	-	280 days	100%
Rollemberg et al. (2019)	7.2 L	6 hours	28±2°C	MLVSS 2.2 g/L	125 days	More than 90%
Mousavi et al. (2017)	5 L	24 hours	30±0.5°C	MLSS 2 g/L	Over 24 days	41.66 mg NH_4-N/L/h

As shown in Table 2-6, bench-scale SBRs used in previous studies typically do not go below 3L of working volume. The maximum reactor volume from these studies is around 24 to 25 L. Higher volumes can already be considered as pilot-scale studies, if not full-scale. Utilizing bench-scale studies have been able to provide information on nutrient removal rates (Al-Attabi et al., 2017; Tałałaj, 2022; Zeng et al., 2003) and nitrifying bacteria behavior (Münch et al., 1996). Since activated sludge is a hazardous material, minimizing the volume that needs to be handled while studying it should be considered.

Due to the nature of biological reactions, experimental periods with laboratory-scale SBRs can take some time. Spagni et al. (2008) took up to 300 days of experimentation to treat leachate using an SBR through nitritation. Scaglione et al. (2012) even took more than a year. The length of time it takes to experiment with a biological reactor involves innoculation and acclimation of the microorganisms that are needed to remove target contaminants. Multiple parameters such as organic loading rate (Al-Attabi et al., 2017), pH (Tałałaj, 2022), and dissolved oxygen (Al-Hazmi et al., 2022), must be constantly monitored and adjusted to ensure that microoganisms are not shocked and killed. This would then lead to ineffective nutrient removal and very poor effluent quality.

2.5.2 Cycle Time

The cycle time for SBRs refers to the whole duration of all phases in sequence. The complete cycle time can vary depending on the target contaminants and effluent quality. Aziz et al. (2013) has compiled a study on the operational parameters used by SBRs that treat landfill leachate from multiple studies. A good number of articles reviewed in the study have cycle times reaching 24 hours. For the SBR studies at this cycle time, more than 80% of the cycle time is allocated to the react phase. The aeration that occurs in this phase is known to be a significant operational parameter, especially in facilitating nitrification of ammonia and oxidation of organic carbon (Al-Hazmi et al., 2022; Aziz et al., 2013). The lowest cycle time reviewed from the same study is at 3.53 hours.

Table 2-7. Operational Parameters for SBR treating landfill leachate (sources as cited in Aziz et al., 2013)

Reference	Fill (hrs)	React (hrs)	Settle (hrs)	Decant (hrs)	Idle (hrs)	Total Cycle	Aeration
Timur and Ozturk (1999)	0.17	22.5	1.16	0.17		24	
Kennedy and Lentz (2000)	9 - 21		2	1		12 – 24	
Neczaj et al. (2005)	0.15	3	0.3	0.08		3.53	
Klimiuk and Kulikowsk (2006)	0.08	21	2.84	0.08		24	2 – 4 mg/L
Laetinen et al. (2006)	0.25	21.5	2	0.25		24	
Neczaj et al. (2007)	3	20	1			24	> 2 mg/L
Bu et al. (2010)	0.5	21.5	1.5	0.5		24	
Lim et al. (2010)	2	12	2	1	7	24	

The cycle time is significant in determining the amount of time the reactor generally has to acclimate to, as the phases are repeated after each cycle. Due to the length of cycle times, SBR laboratory-scale experimentation period tend to be lengthy, which will be discussed in Section 2.5.1.

2.5.3 Reactor Phases

The SBR treatment process is composed of a sequence of five phases. These five phases are general recognized as fill, react, settle, decant/draw, and idle phase (A. H. Jagaba et al., 2021; *Sequencing Batch Reactors for Nutrification and Nutrient Removal*, 1992). Each phase facilitates a specific step in the treatment process of the SBR. One whole cycle of an SBR should have gone through all five phases, with the react phase typically taking up most of the cycle time. The following table from A. H. Jagaba et al. (2021) presents the typical percentage of the cycle time that each phase takes up.

Table 2-8. Typical percent of SBR cycle time that each SBR phase takes up (A. H. Jagaba et al., 2021)

Sequencing Batch Reactor Process Step	Percent of Entire Cycle Time
Fill	25%
React	50% (can be greater than)
Settle	15% to 20%
Draw	5% to 30% (can be greater than)
Idle	-

Slower aeration rates have the potential to reduce nitrate and nitrite concentrations leading to more carbon sources for the denitrification process. Improvements in biodegradability and reductions of refractory compound toxicity is also possible. High aeration rates results in faster aerobic granulation and inhibitiong of overgrowth of filamentous bacteria. Howerver, high aeration rates are undesirable for its high energy consumption, failure in TN removal, and low phosphorus removal due to the loss of anaerobic conditions. Regulating the high aeration rates also failed due to multiple reasons, including the inhibition of the nitrification due to limited oxygen supply. Reduction in the aeration period is suggested as the best aeration regulatory measure (J.W. Lim, Lim and Seng, 2012).

2.5.3.1 React Phase

The react phase is the phase in which nutrients and biochemical oxygen demand (BOD) are mainly reduced. Ammonia is reduced in this phase due to the oxidation of ammonia to nitrites and nitrates through bacteria, which was discussed in Section 2.3. BOD is also oxidized in this phase. The SBR is aerated through fine or coarse diffusers during this phase, which dissolved oxygen into the mixed liquor inside the reactor (*Sequencing Batch Reactors for Nutrification and Nutrient Removal*, 1992). Mixing is also utilized during the react phase in order to ensure that the wastewater is mixed with the activated sludge in the reactor (Aziz et al., 2013). This phase is typically the longest in the cycle since diffusion of oxygen into water is affected by the size of the bubbles from the aerator (A. H. Jagaba et al., 2021). Carbon in organic matter can also be oxidized in this phase into carbon dioxide reducing the concentration of organics in the mixed liquor.

The react or aeration phase can be adjusted to have lower rate of aeration to limit nitrite and nitrate concentrations if carbon supplies are of concern for a subsequent denitrification phase. This is beneficial in reducing the required amount of energy to run a SBR, since the significant cost of operating one is due to aeration. Higher rates of aeration, however, help in faster aerobic respiration and inhibition of filamentous bacteria growth (A. H. Jagaba et al., 2021). As such, multiple studies have explored various DO setpoints and aeration on/off durations to treat wastewater (Aziz et al., 2013; Liu et al., 2017; *Sequencing Batch Reactor Design and Operational Considerations*, 2005). Studies have generally considered DO less than 2 mg/L would limit nitrification (Aziz et al., 2013; Knox, 1985).

2.5.3.2 Settle Phase

The settle phase is characterized by no aeration, which allows the solids in the mixed liquor to settle to the bottom of the reactor. Some of the solids may have formed large flocs, and cumulatively the solids and biomass that settle to the bottom of the reactor is considered to be the activated sludge. A mixer may or may not be used during the settle phase depending on the settleability of the suspended solids in the reactor (Aziz et al., 2013). Since this phase occurs in the same reactor, the SBR eliminates the requirement for a secondary clarifier where removal of solids by the SBR is sufficient (Al-Attabi et al., 2017; *Sequencing Batch Reactors for Nutrification and Nutrient Removal*, 1992).

Since there is no aeration, this phase creates an anoxic condition in the reactor that can be utilized for denitrification if required. Anoxic denitrification would occur due to bacteria stripping oxygen from nitrates, in the absence of DO, generating nitrogen gas (Davis & Cornwell, 2013). Anaerobic conditions can even be achieved in SBRs with some configurations. Scaglione et al. (2012) achieved removal of dissolved oxygen using an anoxic gas mixture of argon and 5% carbon dioxide.

A supernatant or liquid separate from the settled sludge should be identifiable in the upper region of the reactor (Aziz et al., 2013). This supernatant is drawn out of the reactor in the decant phase, which typically follows the settle phase.

2.5.3.3 Decant Phase

The decant or draw phase is when supernatant, separated from the settled sludge formed during the settle phase, is removed from the reactor (Troyer et al., 1997). This supernatant either proceeds to further treatment processes or discharge as treated effluent should it be within the target effluent quality. The decanting volume varies among literature and matches the amount of wastewater to be fed into the reactor after decanting. Troyer et al. (1997) had a decanting volume of about 3% per cycle. While the *Sequencing Batch Reactor Design and Operational Considerations* (2005) recommends that for full-scale SBRs that only up to one-third of the tank volume should be decanted at a time. Morling (2010) studied a full-scale SBR plants with average decant volumes raining from 10% to 33% of the reactor volume. A. H. Jagaba et al. (2021) has also noted that the decant or draw phase can take anywhere from 5% to more than 30% of the total cycle time.

The wide variation in decant volume, and subsequently feed volume, depends on parameters and conditions being maintained in the reactor. The parameters are often dependent on the wastewater quality being fed into the tank, and will be discussed further in the fill phase subsection.

2.5.3.4 Fill Phase

The fill phase is when the reactor is loaded with the wastewater it is supposed to treat (A. H. Jagaba et al., 2021; Troyer et al., 1997). The amount of wastewater fed into the reactor is often the same amount drawn from it during the decant phase in order to maintain the same volume of mixed liquor in the reactor. In some cases, biogas is also recirculated into the reactor during this phase (Troyer et al., 1997). There are also configurations where filling and drawing occur simultaneously, often called “mixed fill”. This kind of configuration is often applied to high-strength wastewater that may inhibit certain treatment processes by shocking the condition in the reactor if fed into the reactor in large volumes (A. H. Jagaba et al., 2021). Aeration and mixing may also be active at this phase to ensure proper mixing of the fresh wastewater with the mixed liquor (*Sequencing Batch Reactors for Nutrification and Nutrient Removal*, 1992). This phase can take up to 25% of the total cycle time, however this will vary on the configuration and type of wastewater to be treated. Parameters such as amount of organic matter, sludge retention time (SRT), and hydraulic retention time (HRT) are affected by the fill phase. Since biological treatment processes require a certain food to microorganism ratio (F/M), this ratio needs to be considered for the different processes that need to happen in the reactor. At the fill phase, the F/M ratio is typically high since wastewater would have a high amount of organics (Troyer et al., 1997). As such the fill phase, can be configured to either precede an aeration phase or settle phase depending on which aerobic, anoxic, or anaerobic processes would be limited by the F/M ratio. If it is not possible to place the fill phase in an ideal position between the react or settle phase, dosing with a carbon source may be done to supplement certain processes over the cycle time.

2.5.3.5 Idle Phase

The idle phase is the state the reactor is in after the decant phase but before the fill phase. Sludge wasting is often conducted at this point, as well as mixing to allow biomass conditioning (A. H. Jagaba et al., 2021). Mixing and aeration may also be active in this phase depending on the conditions to be achieved in the reactor to prepare for wastewater feeding (*Sequencing Batch Reactor Design and Operational Considerations*, 2005).

2.5.4 Ammonia Removal in Sequencing Batch Reactors

Ammonia removal in SBRs is often through nitrification or some integration of the biological process with physio-chemical processes. Both nitrifying and denitrifying bacteria have been studied to be able to thrive in SBRs under optimal operational conditions (Fernandes et al., 2013; Rodríguez et al., 2011). Parameters that are crucial in facilitating the biological nutrient removal include temperature, pH, and dissolved oxygen. Typically, multiple studies have been able to achieve ammonia removal of more than 50%. However, it is not always clear if ammonia removal is calculated based on the influent or SBR’s initial (after the filling stage) ammonia concentration. Due to the nature of the SBR, dilution of the influent with the reactors mixed liquor would already give a reduction in ammonia concentration when comparing the influent concentration and the reactor’s mixed liquor. The dilution would depend on the amount of influent being loaded into the reactor and the reactor’s performance in ammonia removal from the previous cycle. Spagni and Marsili-Libelli (2009) does not that they have taken into account dilution in their calculation of nitrogen and COD removal. Ammonia removal rates for laboratory-scale SBR studies reviewed are summarized in Table 2-6. For this study, both removal rate between influent and effluent, and SBR before and after react are considered.

Aeration in the react mode is the main driver for ammonia removal in sequencing batch reactors. Nitrification is facilitated by the aerobic condition in the reactor during this phase. Certain studies have also looked into simultaneous nitrification and denitrification (SND) in SBRs with a low DO and intermittent aeration (Iannacone et al., 2019; Münch et al., 1996; Zeng et al., 2003). Other pathways in ammonia removal have also been studied with SBRs, most notably is nitritation. Nitritation can be described as ammonia conversion into nitrites only before denitrification instead of into nitrates. This pathway shortens the process for nitrogen removal because nitrites can be readily converted to nitrogen gas instead of nitrates being converted to nitrites then to nitrogen gas in typical denitrification (Spagni & Marsili-Libelli, 2009).

Multiple studies have also used SBRs for ammonia removal from landfill leachate. Spagni and Marsili-Libelli (2009) used a 24L SBR and achieved around 98% ammonia removal from landfill leachate with an average ammonium nitrogen concentration of 1199 mg/L. Due to the high nitrogen concentration relative to COD, acetate was used as a COD source for denitrification in the study. The study from Aziz et al. (2013) found that SBRs used to treat landfill leachate is able to achieve ammonia nitrogen removal from 44% to 95%. The react time for studies using SBRs to treat landfill leachate ranges from 20 to 22.5 hours. The cycle time for these SBRs is 24 hours, as shown in Table 2-7 (Aziz et al., 2013). The long aeration time could be from high ammonia concentration typical of landfill leachate. The following subsections will briefly discuss the sludge retention time parameter of SBRs and bench-scale SBRs.

2.5.5 Sludge Seeding, Acclimation, and Sludge Retention Time (SRT)

Sludge seeding typically involves the use of already activated sludge from an operational wastewater treatment plant (Jeanningros et al., 2010; Mousavi et al., 2017) or the use of granular biomass (Long et al., 2014). It is basically the step in establishing the mass of microbes that will be facilitating biological treatment in wastewater. However, the same microbes that can thrive and treat wastewater can also be found in certain organic waste material. Junaidi et al. (2020) seeded sludge for nitrification from slaughterhouse grey water. The sludge was seeded by continuous loading of nitrogen to grow nitrifying bacteria from the wastewater. A relatively stable state for COD removal indicated that the sludge had been properly conditioned and may be used for experimentation. The sludge seeding period was 29 days (Junaidi et al., 2020). Rizvi et al. (2015) seeded an upflow anaerobic sludge blanket (UASB) with cow manure and was able to product sludge comparable to mature sludge from an anaerobic digester after 120 days. Other studies do not mention the period of time for inoculation of sludge.

Acclimation or adaptation period involves allowing the microbes to get used to the conditions in a bioreactor. These conditions are significantly impacted by the type of wastewater influent and operational conditions. Sudden changes in influent concentration can result in shock loads that generally cause stress to a bioreactor (A. H. Jagaba et al., 2021). Mousavi et al. (2017) studied acclimatization of nitrifying bacteria, and found that it took 11.5 days for nitrifying bacteria to acclimate at a carbon to nitrogen (C/N) ratio of 0. Biological treatment systems in domestic wastewater treatment plants are typically acclimated to domestic wastewater characteristics, which makes landfill leachate not readily treatable by these typical treatment plants. Co-treatment has been explored to address this issue, and typically about 10% landfill leachate to volume of domestic wastewater is found to be ideal to avoid shocking the biological treatment process (Capodici et al., 2014). Datta et al. (2009) acclimatized biomass for 15 days at half feeding rate, and 20 days at full feeding rate for a total of 35 days of acclimation. Due to the different characteristic of landfill leachate, most notably the inherently low biodegradability to high ammonia concentration, acclimation is important in ensuring that biological systems can handle this type of wastewater. Spagni and Marsili-Libelli (2009) used activated sludge that had been acclimating to landfill leachate for almost two years. In the full-scale scenario, Tilche et al. (1999) took 3 weeks before a full-scale SBR for treating piggery wastewater reached full treatment capacity with sludge from a nearby wastewater treatment plant. The variation in acclimation significantly affects the lengthy experimental period for sequencing batch reactor studies, such as those in Table 2-6. In this study, the sludge is acclimated anywhere from 2 to 4 weeks with each increase in ammonia concentration in the influent wastewater.

The amount of solids maintained inside an SBR is kept by maintaining a solids retention time (SRT). The SRT is defined as the total mass of solids, mostly the suspended solids, in the tank divided by the solids removed from the tank per day, both from decanting and sludge wasting (A. H. Jagaba et al., 2021). An appropriate SRT is significant in ensuring the efficiency of nutrient removal for the SBR. For nitrifying systems, it is recommended that the SRT be based on the aeration time during a cycle, and not the entire cycle time(*Sequencing Batch Reactor Design and Operational Considerations*, 2005).

For the case of nitrifying bacteria, a longer SRT is preferred due to their relatively slower growth rate (Eddy, 2003). Typically, the total suspended solids (TSS) and volatile suspended solids (VSS) concentration in SBRs is used as an estimate of active biomass in the reactor. With the correlation of the SRT to suspended solids concentration, TSS and turbidity levels are often higher at lower SRT (A. H. Jagaba et al., 2021). Barber (2015) found that increased sludge age corresponds to less responsive sludge. As such, a proper length of SRT is crucial in optimising biological nutrient removal. In this study, SRT shall be measured at certain points over the experimental period to provide indication of the amount of biomass in the system.

2.5.6 Supplementation

Since the process of nitrification mainly consume organic matter and alkalinity (Section 2.3.1), high ammonium removal would require a higher amount of organic matter and alkalinity should the concentration in the influent wastewater and reactor be depleted before ammonium if fully removed. Multiple studies note the use of supplementary carbon sources to aid the biological process, including denitrification (Junaidi et al., 2020; Mousavi et al., 2017). This is expected for landfill leachate treatment since the biodegradability is typically low for stabilized landfill leachate with very high ammoniacal-nitrogen concentration (Section 2.1.3). Supplemental carbon sources used in nitrification studies include methanol, glucose dextrose, and an acetate (Al-Attabi et al., 2017; Iannacone et al., 2019; Junaidi et al., 2020). Amounts of each kind of carbon source is varied depending on the target carbon to nitrogen ratio or required carbon to remove more ammonia. However, a balance of which must be considered as high C/N ratio can result in inhibition of ammonium removal due to heterotrophic bacteria dominating over nitrifying bacteria (Mousavi et al., 2017). In line with this, Mousavi et al. (2017) found that increasing C/N ratio slowed down ammonium removal. Al-Attabi et al. (2017) also found that increasing the organic loading rate (OLR) reduces ammoniacal nitrogen removal, however the OLR in this experiment was in the form of potassium nitrate concentrations. A. H. Jagaba et al. (2021) has also found from literature that the addition of a simple carbon source helps in remedying the shock loads. For the purpose of this study, a fixed amount of organic matter shall be utilized in the wastewater with increasing amount of ammonium. Fixing the organic matter should reasonably simulate the relatively low biodegradability to ammonia concentration in landfill leachate. Supplementation with acetic acid shall be implemented as needed if ammonium removal seems to be completely inhibited.

For the purpose of landfills, the supplementation of organic matter through the waste already being disposed in the site may be feasible. This would remove the need for purchasing additional chemicals for an onsite treatment plant. However, this should consider that organic matter will also add ammoniacal nitrogen to the reactor compared using a chemical solution.

2.5.7 Comparison of Lab-Scale to Pilot- and Full-Scale SBR and Scaling-up

Pilot-scale studies are often based on laboratory-scale studies. Typically at this scale, up to half of the total actual flow with a capacity of up to 3000 L. This scale also often serves as a bridge between the laboratory-scale design and the potential full-scale design for SBRs (A. H. Jagaba et al., 2021).

Full-scale SBRs are those operated on site to treat the actual wastewater of concern at the actual flow rate. Design parameters for full-scale SBRs are often from pilot-scale studies (A. H. Jagaba et al., 2021). There seems to be a lack of studies that bridge the laboratory-scale SBR to the design of a full-scale SBR. Typically, laboratory-scale

SBRs are utilized for the purpose of experimenting with biological processes. Design specifications for scaling-up lab-scale to full-scale are not established at the time of writing, which is likely due to the varying wastewater characteristics that affect operational parameters of a SBR. Ketchum (1997) indicates that the phases of the reactor are of higheste concern in full scale design. The configuration of multiple SBR tanks running in parallel are common in municipal use. The following equation relating the number of SBRs (n) in parallel operation to time of each phase is from Ketchum (1997).

$$t_f(n-1) = t_r + t_s + t_d + t_i$$

Equation 2-4. Number of SBRs in parallel based on SBR phase times (Ketchum, 1997)

The time for the fill, react, settle, decant, and idle phase are represented by t_f, t_r, t_s, t_d, and t_i, respectively. With the larger volume of full-scale SBRs, the sludge or settled biomass would take up a significant volume of the reactor. The volume of the treated supernatant is also carefully considered at full-scale as it is directly related to the influent, as is the relation of the decant and fill phases as discussed previously in Section 2.5.3 (Ketchum, 1997).

Typically around three SBRs are used for municipal wastewater treatment plants that utilized SBRs, since the number of SBR tanks would reduct the amount of influent flow per tank and therefore waste strength to be treated in the reactor. The following table summarizes the determination of design cycle times from Ketchum (1997).

Table 2-9. Sequence phases design basis and typical timings (Ketchum, 1997)

Sequence Phase	Basis of Design	Typical Times at Full-Scale
Settle	• Subsidence of sludge layer • Settled biomass concentration and depth	• For shallow tanks, 0.5h • For deep tanks, 0.75h • Conservative design, 1h
Draw	• Hydraulic capacity of decating mechanism • Cost of decanting mechanism	• 1h or more for large systems
React	• Target contaminants • Effluent standards to achieve • Cost of aeration	• 1.5h to 3h • Could be more for extended aeration systems
Idle	• Need for flow equalization • Flow rate fluctuations and uncertainty • Can be 0h for peak design flows, if fluctuations are more predictable	• Can be manipulated to accommodate changes in reactor and influent conditions
Fill	• Dependent on draw volume	• Can be calculated using Equation 2-4

Specific components to consider in a full-scale SBR include the inlet, outlet, mixer, and aeration system. A main controller would be ensuring that all of these work in sync with the planned sequence timings and operational parameters that the SBR should be working with. For municipal treatment, concrete has bee the material of choice for the tanks as it served a conservative approach for long operational life. For industrial settings, steel cylindrical tanks with corrosion protection are preferred as they are less expensive with a relatively shorter operational life expectancy (Ketchum, 1997).

The decanting mechanism are one of the more important components of a SBR as this enables it to discharge treated supernatant, and its efficiency significantly affects the effluent quality. In the event a decanter does not function properly, a significant amount of solids may be discharged to a water body that can cause significant environmental impacts. Ketchum (1997) illustrates three main types of decanters used for full-scale SBRs in the following figure.

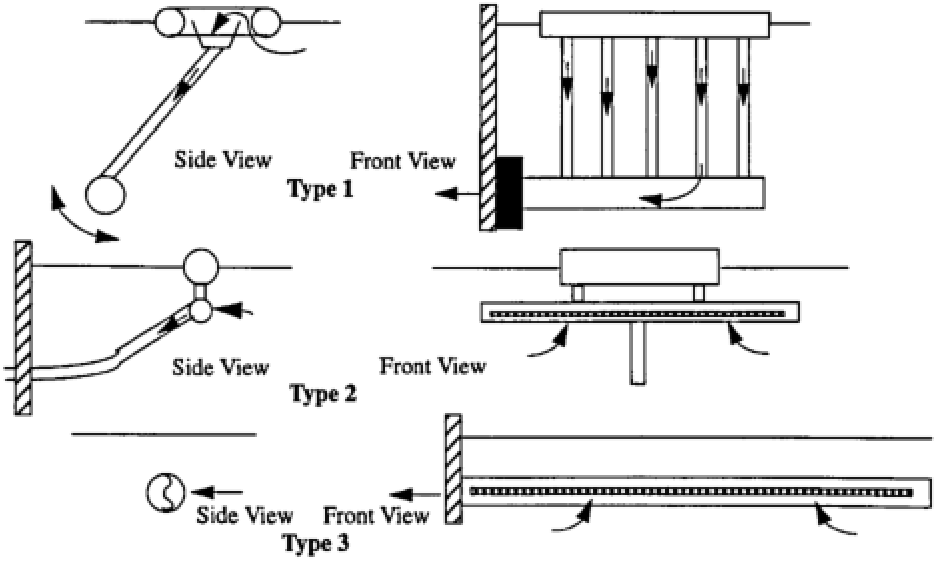

Figure 2-6. *Typical decanter mechanisms (Ketchum, 1997)*

These decanter mechanisms will be further evaluated in the study as a full-scale SBR is designed based on the experimental lab-scale SBR findings. The scale-up would mainly involve multiplying the lab-scale operational parameters by a factor to obtain conditions for treating estimated actual leachate flow. The components, such as aeration and decanting requirements, will be selected as per the volumetric requirements of the scaled-up reactor from the lab-scale findings.

2.6 Conclusion

In conclusion, there is a significant amount of studies on sequencing batch reactors in the laboratory-scale. Many of which have been able to remove ammonia from wastewater, some of which even from landfill leachate. Many of the laboratory-scale bioreactors are of volumes greater than 1 L, which is found to be a research gap this study can contribute to. This study uses a 1 L bioreactor for removing ammonia, which is analyzed and compared to the ammonia removal from studies using wastewater with high ammonia concentration. A cycle of 24 hours is to be used in this study to maximize the period of aeration and remove as much as ammonia as possible per cycle, which is similar to other SBR studies that are studied to treat landfill leachate, as summarized in Table 2-6. The reaction phase is to be controlled to maintain a DO of 2 to 4 mg/L in the bioreactor as much as possible, and last from 18 to 20 hours, which is aligned with the react phase typically taking up more than 50% of a full cycle (Table 2-8). Information on sludge seeding for wastewater treatment are mostly for activated sludge harvested from existing wastewater treatment plants, granular biomass, or from previously studied reactors. The use of raw waste material for sludge seeding is an identified research gap, as the exploration of other material to start-up biological wastewater treatment processes is sparse. Rizvi et al. (2015) was able to use cow manure to generate sludge in an upflow reactor that is comparable to digester, which provides potential for cow manure to also generate activated sludge for an SBR. The use of cow manure in this study to start up an ammonia removing bioreactor is a novelty of this study, and can significantly contribute to potential use cases for cow manure. Given how heterogenous cow manure may be compared to activated sludge from a wastewater treatment plant, this study is also exploring the adaptation length for cow manure to start removing ammonia from start-up.

Standards for scaling-up SBR design from a laboratory-scale are not common, as the typical design flow starts from the laboratory-scale to pilot-scale then full-scale. The experimental results from a 1 L bioreactor is scaled-up in this study to a full-scale, whose performance is compared to studies of full-scale SBR performance to explore the potential of skipping the pilot-scale study in establishing a full-scale SBR for landfill leachate treatment. This could provide economic benefits in going straight to full-scale SBR with reasonable information from laboratory-scale. The design leachate flow is to be estimated from literature relating leachate production to annual precipitation. Operational parameters are to be scaled-up accordingly from the experimental set-up and potential areas that may be found can be ran more efficiently in the full-scale.

The following section will detail the specifics of the methodology used in the laboratory-scale SBR from the set-up to the analytical methods used in measuring target contaminants, mainly ammonium and ammoniacal-nitrogen.

3 Methodology

The following section discusses the experimental set-up for a laboratory-scale sequencing batch reactor, and the analytical methods used to monitor ammoniacal-nitrogen removal. The discussion starts with the laboratory-scale set-up and goes into detail regarding the operational parameters set-up over the start-up and experimental period. This is followed by the discussion on analytical methods used in measuring ammonium, nitrites, nitrates, alkalinity, turbidity, and total and volatile suspended solids in the influent, SBR mixed liquor, and effluent.

3.1 Lab-Scale SBR Set-up

The experimental set-up is located in C3.10 of the C Building in the University of Waikato, Hamilton, New Zealand. Analytical testing methods are performed mostly in C3.11 of the same building, which is the Environmental Engineering Laboratory of the University of Waikato Hamilton Campus. The period of experimentation is from April 2023 to February 2024.

Two laboratory-scale SBRs with a working volume of 1L are monitored using a Winpact Parallel System Controller. The type of vessel the SBR is specifically the FS-V-A Jacket Vessel from Winpact. The SBR is equipped with an aeration rod, stirrer, pH probe, DO probe, and temperature probe. The system controller is able to log the probe readings over time, which can be harvested via USB flash drive. The aeration rod passes through the controller via a gas flow meter up to 5 liters per minute (LPM). The stirrer is also controlled by the controller and can be set at variable RPM. Decanting effluent from and dosing influent to the SBR are handled by two peristaltic pumps. Peristaltic pumps are selected to minimize scouring due to solids in the influent and effluent. The following picture and diagram illustrate the SBR set-up for this study.

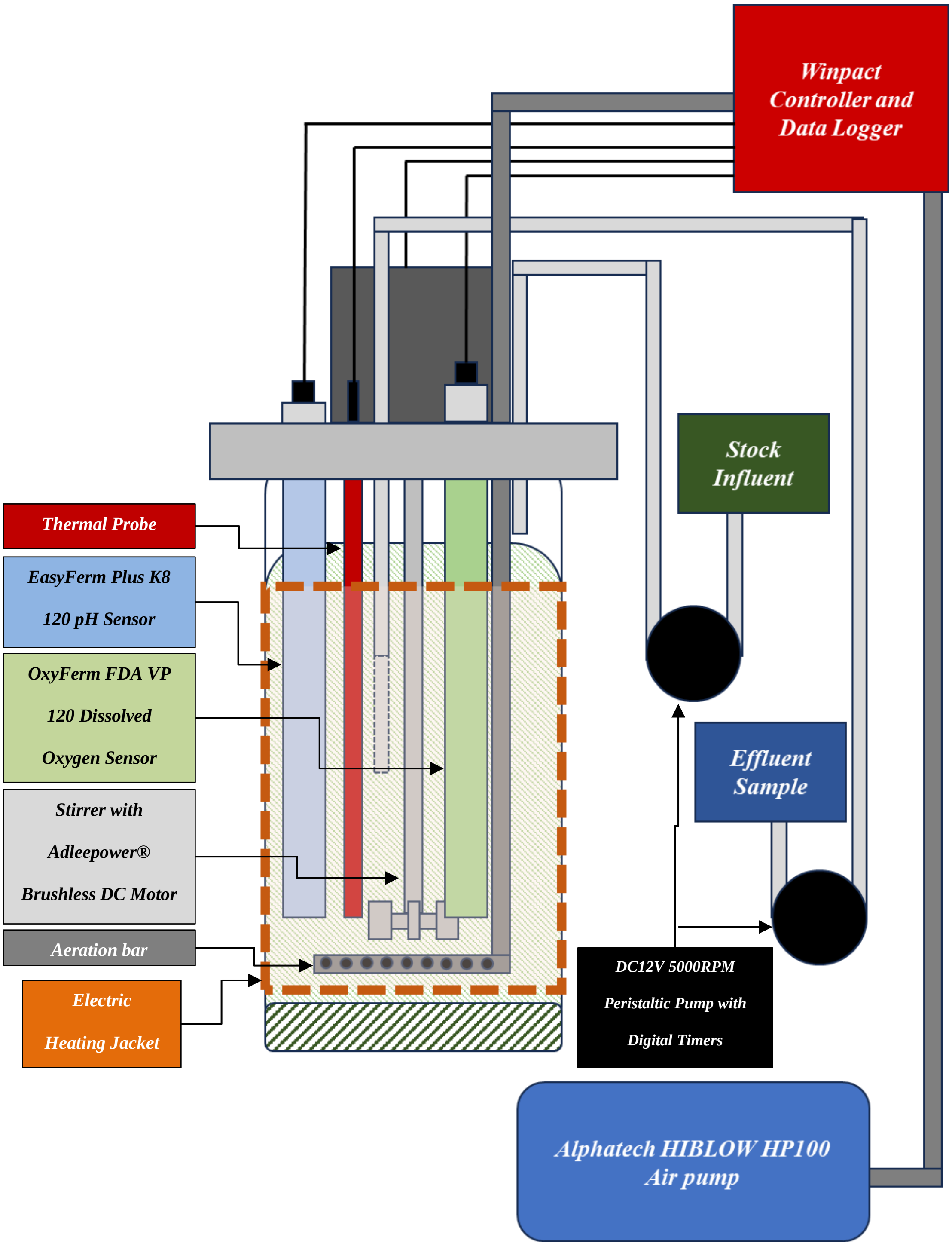

Figure 3-1. Laboratory-scale SBR set-up diagram

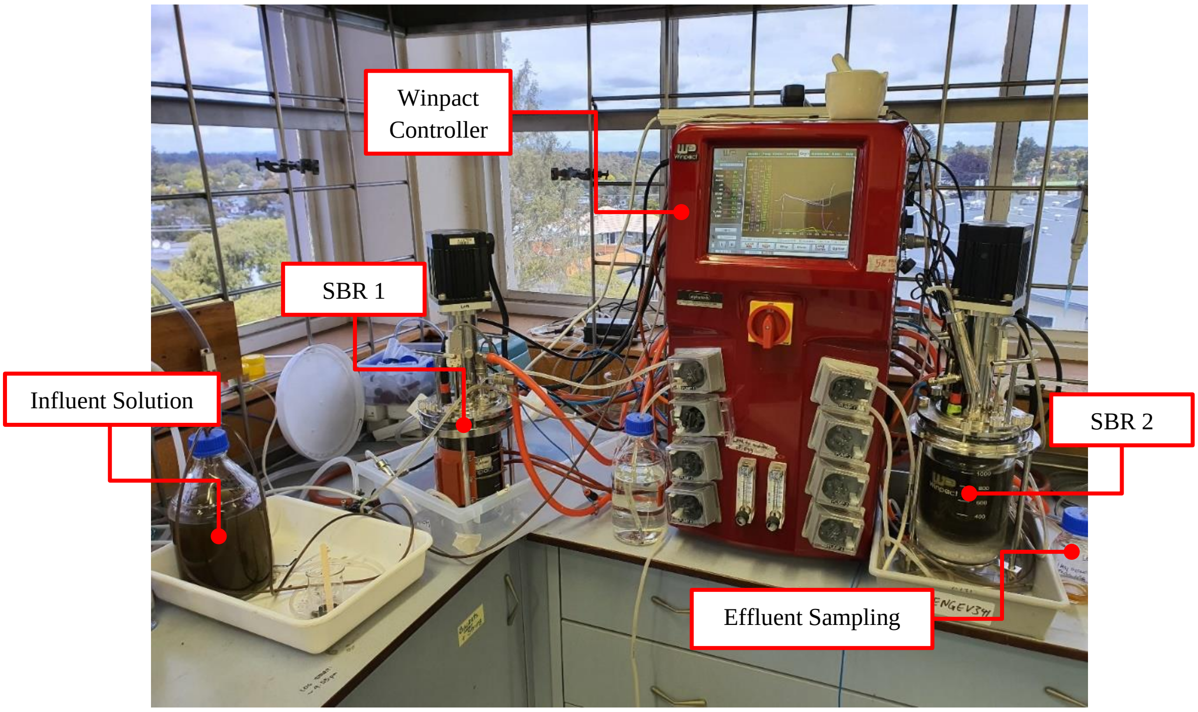

Figure 3-2. *Winpact Controller with SBR 1 (Left) and SBR 2 (Right)*

3.1.1 Cycle Length and Phase Sequence

The cycle length for the lab-scale SBR is selected to be 24 hours as this is the common cycle length for SBR studies on treating landfill leachate (Table 2-7). The react phase is set to take up majority of the cycle since this phase facilitates the ammonia removal in the reactor via nitrification. The react phase in the experimental period started at 12 to 16 hours initially split in two 6 and 8 hour intervals, respectively over the start-up period of the SBR. This is then is increased to 20 hours as the ammonia loading is increased after 60 days. The aeration period is then adjusted to 18 hours to add 2 hours to the settling period to accommodate higher suspended solids concentration and possible denitrification. The aeration is controlled by an HPM slim digital timer for the Alphatech HIBLOW HP100 air pump and is given power during the set aeration periods, since the air pump itself does not have an on of off switch. The following tables summarize the cycle lengths and programs used over the experimental period.

Table 3-1. Cycle and phase lengths used in the operation of the lab-scale SBR

Cyle Program	Phases	Length of Phase (hours)
Start-up	React/Aeration	12 - 16 (Two 6- to 8-hour long intervals)
	Settle	8 - 12 (Two 4- to 6-hour intervals after each react phase)
	Fill	< 6 minutes (after draw phase)
	Draw/Decant	< 6 minutes (after the settle phase)
	Idle	-
Cycle 1	React/Aeration	20 (Continuous)
	Settle	4 (Continuous)
	Fill	< 6 minutes (after draw phase)
	Draw/Decant	< 6 minutes (after settle phase)
	Idle	-
Cycle 2	React/Aeration	18 (Continuous) with Stirring at 25 rpm
	Settle	6 (Continuous) with Stirring at 25 rpm
	Fill	< 6 minutes (after draw phase)
	Draw/Decant	< 6 minutes (after settle phase)
	Idle	-

Table 3-2. Cycle program operated over experimental period

Month	May	June	July	August	Sept	Oct	Nov	Dec	Jan
Cycle Program	Start-up		Cycle 1			Cycle 2			
Days Elapsed	60		90			125			

Stirring is controlled using the Winpact controller and is set at the lowest setting at 25 rpm even during the settle phase. This is done for the Cycle 2 program because setting time steps for the stirrer using the controller was not possible without interfering with the data logging feature. The fill and draw are controlled using HPM slim digital timers, one for each peristaltic pump. The digital timers are times such that the draw pump is engaged and followed by the fill pump once the decant phase is done. No idle phase is utilized for the experiment as there is not sludge wasting done for this study.

3.1.2 pH, temperature, and DO monitoring

The controller uses a Hamilton Company pH (EasyFerm Plus K8 120 pH Sensor) and DO (OxyFerm FDA VP 120 Dissolved Oxygen Sensor) probes for measuring the respective parameters in the SBRs. A Winpact Parallel System Controller (Model: FS -05-220) handles logging of data from the pH and DO sensors. The controller also has an included temperature sensor that is used to measure the temperature in the SBR. Data from the pH, temperature, and DO sensors are logged every 5 minutes. The data can then be harvested in comma-separated values file (csv) that is analysed using spreadsheet software.

The pH sensor is calibrated on the Winpact controller to a pH 7 buffer solution. Based on literature, the ideal pH to maintain for facilitating nitrification is around 6 to 8 pH with optimal range of 7 to 8 pH for nitrifying bacterial growth. This in line with literature that maintains pH for nitrification within the same range, such as Gao et al.

(2020) maintaining a pH at 7.50. The pH is adjusted manually with the addition of 1M NaOH solution since the nitrification process lowers pH and may cause acidic conditions to occur when denitrification cannot make up for the alkalinity consumed during nitrification. Haslina et al. (2021) notes that the advantage of using NaOH for pH adjustment is that it would avoid scaling that occurs with other solutions to increase pH, such as calcium hydroxide.

The temperature probe logs data in degrees Celsius units, and control over the reactor temperature is facilitated by an electric jacket for SBR 1 and water jacket for SBR 2. The temperature over the experimental period is maintained within the 28^{o}C to 30^{o}C. This temperature range is selected in order to best facilitate a favorable environment for the biomass, since their source is the internals of cattle, which is typically a couple degrees above this range. Changes to temperature are limited in order to reduce acclimation period for the biomass with increasing influent ammonium concentration, and to maximize the potential for nitrification.

The DO probe measures DO in percent, which is calibrated to over the experiment to ensure the ideal DO concentration is maintained during the aeration period. Due to lack of automatic control for maintaining the DO in the reactor, a built-in gas flow meter on the Winpact controller is used to control the aeration rate. The gas flow meter is set between 0.5 to 1 liter per minute (LPM) over the course of the study. Multiple studies also consider that at least 2 mg/L DO is ideal to facilitate nitrification. Since the DO probe measures in percent, a DO probe measuring in mg/L is used to help calibrate the SBR's DO probe to what concentration 100% DO is in mg/L of DO. Generally, the DO concentration is kept around 2 to 4 mg/L during the aeration phase, and at most around 5 mg/L due to lack of precise control of DO. It should be noted that the DO probe socket of the Winpact controller for SBR 2 is broken and did not function, therefore DO readings will only come from SBR 1.

3.1.3 Sludge Retention Time (SRT)

For the experiment, sludge wasting is not regularly done due to the small working volume and the significant change in conditions a few milligrams of sludge wasting may do the SBR. The sludge retention time (SRT) is assumed from the measurement of TSS of the effluent against the TSS of the mixed liquor in the SBR. This is calculated by dividing the TSS of the mixed liquor by the TSS of the effluent multiplied by the daily decanted volume to get the number of days the TSS is approximately completely replaced (A. H. Jagaba et al., 2021). The mixed liquor suspended solids is not controlled, but the TSS is measured from both effluent and mixed liquor to check that SRT is around or more than 15 days, which is around the ideal amount of time to grow the nitrifying bacteria required for ammonia removal via nitrification.

$$SRT\ (days) = \frac{TSS\ in\ mixed\ liquor\ g/L}{(TSS\ in\ effluent\ g/L) * Decanted\ volume\ per\ day\ L/day}$$

Equation 3-1. Equation for calculation sludge retention time for the study

The analytical method for measure TSS is discussed in Section 3.4.5.

3.2 Sludge Inoculant

Since there is no known local wastewater treatment plant that biologically treats landfill leachate and the difficulty in handling activated sludge, cow manure is used to inoculate the sludge in the SBR, similar to Rizvi et al. (2015). Cow manure is obtained from a dairy farm in 265 Crawfod Road, Horotiu, Hamilton, Waikato on April 28, 2023 (Figure 3-3). The 1L SBR is seeded with 100mL of cow manure and distilled water. To ensure microorganism growth from the cow manure, the SBR kept at a temperature of 30±2ºC over the inoculation stage for 2 months. This inoculation stage using an upflow reactor from Rizvi et al. (2015) took 4 months before the sludge quality was similar to that of mature sludge. Over this inoculation period, the SBR is fed with influent solution of 1% cow manure in distilled water. This is to ensure minimal shocking of the system, and gradual conditioning of the activated sludge over time.

Figure 3-3. Location map for source of cow manure used for the study (Google, 2024a)

After two months of inoculation, ammonium removal is monitored between the influent and effluent and increase in ammonium concentration of the influent is initiated.

3.2.1 Nitrogen and Carbon Content

The carbon and nitrogen content of the cow manure is analysed using a combustion carbon nitrogen analyser (Elementar Organic Elemental Analyzer) from the Earth Sciences Department, School of Science, University of Waikato. The analyser combusts up to 80 samples. For the analysis of the cow manure used for the sludge seed, three sample are used. Approximately 5 milligram samples of cow manure and three 5 milligram standards, in the form of acetanilide, are prepared using a high precision Mettler Toledo scale (Figure 3-4). The samples are wrapped in tin capsules and places in the elemental analyser and combusted. The following table summarizes the result of the nitrogen and carbon content analysis from the analyser (Table 3-3).

Figure 3-4. *Mettler Toledo high-precision digital scale used for weighing samples*

Table 3-3. *Carbon and nitrogen content of cow manure used for sludge seeding*

Sample	**Sample Weight (mg)**	**% Nitrogen**	**% Carbon**
Cow Manure	5.0540	2.498	41.478
	5.0400	2.527	41.358
	5.0640	2.530	41.375
	Average	2.518	41.404

The average nitrogen and carbon content of the cow manure is at 2.518% and 41.404% respectively. This gives a C/N of 16.44, similar to that of Marañón et al. (2006) found at 18.9 for one of the farms tested in the study. A total nitrogen content of 2.7% is also measured from the same study.

3.2.2 Moisture Content

The moisture content of the cow manure is also measured according to the American Public Health Association (2017) (APHA). Cow manure sample is weighed in a beaker of known weight such that the net wet weight of the sample is known. The sample is placed in an oven set at 105±5ºC for 24 hours, and the container with dry sample is left to cool. The sample is then weighed and recorded, such that the net dry weight can be calculated. The moisture content is measured using the following equation.

$$Moisture\ Content\ (\%) = \frac{Wet\ weight\ (g) - Dry\ weight\ (g)}{Wet\ weight\ (g)} * 100\%$$

Equation 3-2. Equation for calculating moisture content of cow manure

The average moisture content of the cow manure is calculated to be at 87.27% from three samples. This is near the average moisture content measured by Taylor (1917) at 83%.

3.3 Influent or Landfill Leachate Substitute

Due to difficulty in procurement and handling of landfill leachate samples for testing with the SBR, cow manure mixed with NH_4Cl is used as landfill leachate substitute. The solution is referred to as influent in this study. The cow manure is mixed with distilled water up to 5% by mass and the ammonium concentration of the solution is increased with the addition of NH_4Cl of known mass.

Over the experimentation period, the ammonia concentration in the influent is increased to be within the high concentration range for ammonia in landfill leachate, which is at least 500 mg/L. The amount of cow manure mixed with distilled water is only up to 5% by mass to fix the amount of organic matter to be within a range while increasing the ammonia concentration. This is to as closely resemble the characteristic of mature landfill leachate to generally have low biodegradability while still having a high concentration of ammonia. The amount of NH_4Cl added to the solution started from 0.3g to 3g. This is in line with literature such as Spagni and Marsili-Libelli (2009) that also used NH_4Cl for increasing ammonium concentration in synthetic wastewater. The following table summarizes the influent wastewater used in the study, including the composition, days used, volume dosage, and approximate NH_4 concentration.

Table 3-4. Influent wastewater length of use, composition, and estimated NH_4 concentration

Days Used as Influent	Composition	Volume Dosed per Cycle	Approximate NH_4 Concentration
Up to 60 days	1% Cow Manure in Distilled Water	400mL	2 – 6 mg NH_4/L
60 to 90 days	5-10% Cow Manure in Distilled Water	200mL	40 – 60 mg NH_4/L
90 to 120 days	• 5% Cow Manure in Distilled Water • 0.3 g NH_4Cl	200mL	50 – 80 mg NH_4/L
120 to 150 days	• 5% Cow Manure in Distilled Water • 0.5 g NH_4Cl	200mL	100 -120 mg NH_4/L
150 to 180 days	• 5% Cow Manure in Distilled Water • 1.5 g NH_4Cl	100mL	260 – 340 mg NH_4/L
180 to 200 days	• 5% Cow Manure in Distilled Water • 2 g NH_4Cl	100mL	350 – 410 mg NH_4/L
200 to end of period	• 5% Cow Manure in Distilled Water • 3 g NH_4Cl	40 to 100mL	600 – 720 mg NH_4/L

The ammonia concentration of the influent is measured analytically to identify the corresponding influent ammonia load on the SBR at a given time.

The preparation of the influent starts with weighing out 1% or 5% mass of cow manure with distilled water. The weighed cow manure is then mixed with distilled 100 or 20 times the mass of cow manure measured. This mixture is then strained to separate large chunks of solids in the cow manure that may scour the peristaltic pumps. Once the whole mixture is strained, it is mixed with a solution of NH_4Cl, if needed (Table 3-4). This preparation is done for all the influent prepared over the duration of the experiment. The following images show parts of the preparation process for the influent.

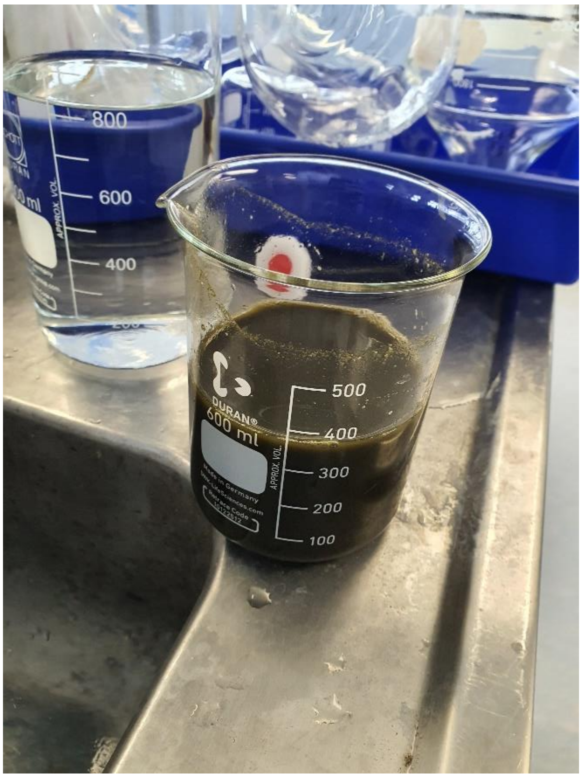

Figure 3-5. Cow manure mixed with distilled water before straining

Figure 3-6. Straining of cow manure and distilled water solution to separate large solids

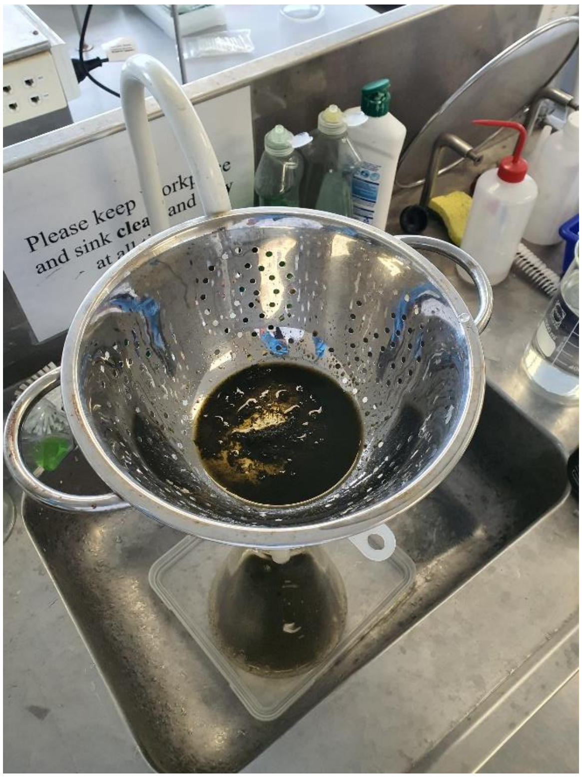

Figure 3-7. Large solids separated from the cow manure solution

Figure 3-8. 2 L of prepared influent solution

3.4 Analytical Methods

The ammoniacal nitrogen, nitrites, nitrates, and alkalinity are measured by colorimetric analysis using Palintest reagents and photometer. The total suspended solids and volatile suspended solids are measured according to the Standard Methods for Examination of Water and Wastewater (American Public Health Association, 2017).

3.4.1 Ammonia/Ammonium

The ammoniacal nitrogen concentration in the influent and effluent of the laboratory-scale SBR is measured by colorimetric analysis using Palintest reagents and photometer for influent and effluent comparisons over 250 days or a little more than 8 months of experimentation. An ion-selective electrode (Vernier Go) is used for measurement of ammonium removal in the bioreactor in a cycle. The following subsections discuss each instrument used for ammonium measurement and the calibration done for both.

3.4.1.1 Colorimetric Analysis using Palintest photometer

The ammonia test using the Palintest photometer follows the Palintest Ammonia Test. This test is based on the indophenol method for measuring ammonia. The reagents are alkaline salicylate and chlorine which react with ammonia that develop a green-blue indophenol complex rapidly. Catalysts are also present in the reagent to enable the rapid colour development. The reagents are labelled as Ammonia Tablet No. 1 and No. 2. The test has a range of measurement from 0 to 1.0mg NH_4-N/L. The procedure starts with the preparation of 10mL of sample in the Palintest tube (Palintest, 2018b).

Figure 3-9. *Palintest Photometer 7500 unit used for the study*

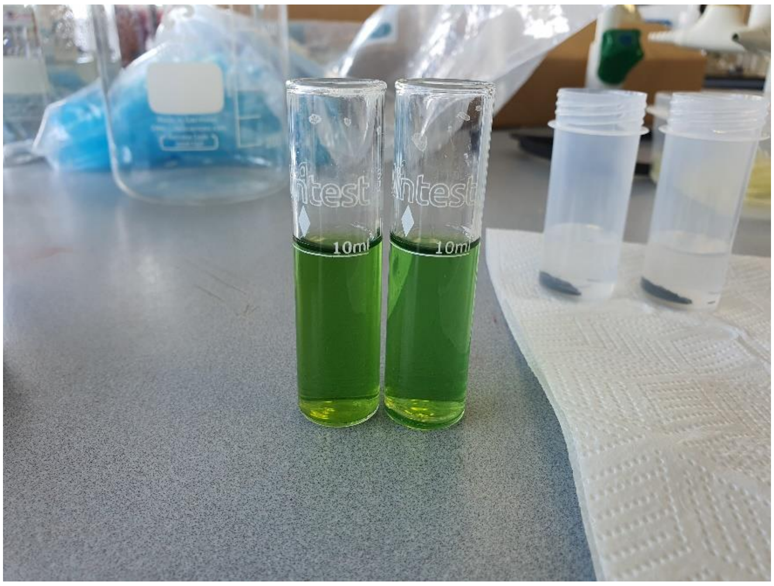

Figure 3-10. *10 mL Palintest tubes with diluted sample with Ammonia 1 and 2 tablets mixed*

Due to the low range of measure of the test, the samples are diluted accordingly to be within the readable range. Each tablet is dropped into the sample solution, and crushed until dissolved. The mixed sample is left to develop colour for 15 minutes. Once the colour is fully developed, the photometer is turned on and the Phot 4 photometer program for ammonium nitrogen is selected. The photometer is blanked with a Palintest tube filled with 10mL of distilled water, and the sample is placed in the photometer with the cover and read. The ammonium nitrogen, ammonium, and ammonia concentration in mg/L is recorded and multiplied by the dilution factor, if needed. The ammonium and ammonia concentrations are calculated by the photometer from the ammonium nitrogen measurement by multiplying the mg NH_4-N/L reading by 1.3 and 1.2, respectively (Palintest, 2018b).

The ammonium of the influent and effluent are measured at least twice a week for a period of 250 days or approximately 9 months. The removal rate of the reactor is calculated both based on the difference of ammonium concentration of the influent and effluent, and calculated SBR concentration after fill phase, which is based on the influent concentration and volume of dosing, against the effluent concentration. The second calculation is more accurate for measuring the ammonia removal potential of the SBR, while the first removal rate calculation is a measurement of the SBRs performance against influent strength.

3.4.1.2 Vernier Go Ion Selective Electrode

Real time ammonium ($[NH_4]^+$) concentration is measured using a Vernier Go Ammonium Ion Selective Electrode. Ion-selective electrodes work by being tuned to respond to a selected ion, such as ammonium, when placed in a solution by measuring a potential between the inside and outside of a membrane (Li et al., 2020). For this study, the ion-selective electrode membrane used is from Vernier and its data is logged via mobile application Graphical Analysis. This electrode uses a Bluetooth receiver for transmitting measurements attached to a probe with a PVC-based membrane tip. The review of ammonium detection methods by Li et al. (2020) found PVC to be the most common type of membrane for ammonium ion-selective electrodes. The following table summarizes the specifications of the ion-selective probe from Vernier.

Table 3-5. Vernier Go ammonium ion-selective electrode specifications (Vernier, 2020)

Range	1 to 18,000 mg/L (or ppm)
Accuracy after calibration (precision)	±10% of full scale (calibrated 1 to 100 mg/L)
Interfering ions	K^+, Li^+, Na^+, Cs^+, Mg^{3+}, Ca^{2+}, Sr^{2+}, and Ba^{2+}
pH range	2 - 7 (no pH compensation)
Temperature range	0 – 40°C (no temperature compensation)
Electrode slope	+56 ±4 mV/decade at 25°C
Standard voltages, typical	High (100mg/L) at 116 mV, Low (1mg/L) at 0mV
Electrode resistance	0.1 to 5 MΩ
Minimum sample size	Must be submerged 1.1 in (2.8 cm)
USB specification	2.0
Wireless specification	Bluetooth 4.2
Maximum wireless range	30 m
Dimensions	EA: 8.5 cm height x 3 cm wide x 1.75 cm depth ISE: 20.5 cm length, 12 mm OD
Battery	300 mA Li-Poly
Battery life (single full charge)	~24 hours
Battery life (long term)	~500 full charge cycles (several years depending on usage)

The probe is calibrated using a high standard and low standard ammonium solution with concentrations of 100 mg/L NH_4-N and 1 mg NH_4-N/L, respectively. Measuring outside of the range of 1 to 100 mg NH_4/L requires calibration with higher standards that can be prepared by dissolving a calculated amount of NH_4Cl in distilled water. The sensor works with an ion-specific (sensing) half-cell and a reference half-cell, wherein the ion-specific half-cell generates a potential that can be measured against the reference half-cell based on the activity of the ammonium. The ammonium ion activity and potential reading of which is correlated to the ammonium concentration. The Nernst equation (Equation 3-2) is used to relate the potential measured in the ISE to the ammonium concentration.

$$E = E_o - 2.303\frac{RT}{nF}\log{(C + C_o)}$$

Equation 3-3. Nernst equation relating the potential measured by the ISE and ammonium concentration

The E is the measured potential, while E_o is the standard potential between the ISE and reference electrode. R in the equation refers to the universal gas constant at 8.314 J/mol*K, while T is the temperature in Kelvin of the sample. F is for the Faraday constant equivalent to 96,485 C/mol and n is the valence of the ion. C and C_o are the concentration of the ammonium and detection limit, respectively (Vernier, 2020). Ammonium ISE's are relatively cheap compared to other methods of measuring ammonium, however are found to be less effective in low-level concentrations, amines tend to affect readings, and is not applicable to sea water samples (Molins-Legua et al., 2006). For this study, the probe is mainly utilized to observe and estimate nitrification rates in the lab-scale SBR over a cycle, and therefore these limitations are not present. The SBR is treating non-saline influent solutions, while the concentration level in the SBR is not low-level most of the time. According to Chen et al. (2004), the amine content in cow manure is around 7.92%, which is taken note of in taking the readings from the ISE.

For this study, the probe is dipped into the SBR over three periods of the regular experimental period. The probe is left in the SBR for a full cycle, 24 hours, and the data on pH, DO, and ammonium concentration data are collected from the controller and Graphical Analysis application, and plotted using spreadsheet software.

The probe is also used to measure how long it takes before the ammonia removal is inhibited outside of pH level. This is conducted by leaving the probe in the SBR with continued aeration until there is an observed stabilization of ammonium concentration reading at the same time stable pH levels even under aeration. The pH, DO, and ammonium concentration data is collected from the controller and Graphical Analysis application, and plotted using spreadsheet software.

3.4.1.3 Calibration

Since ammonium is the target concentration to be measured from the influent, mixed liquor, and effluent, a calibration of the instruments used to measure ammonium is done to verify the accuracy of the available tests for measuring ammonium concentrations.

Calibration is done using distilled water solutions and effluent solutions with NH_4Cl to raise ammonium concentrations in controlled steps. The distilled water solution is made with a 100ppm stock solution using 1L of distilled water mixed with 300mg of NH_4Cl. The same stock solution is diluted to create solutions of concentrations at 20 mg/L, 40 mg/L, 60 mg/L, 80 mg/L, and 100 mg/L of ammonium. The Palintest tests are done according to the Palintest procedure for ammonium testing (Section 3.4.1). The solutions, except the 0 mg/L solution, are diluted 100 times, which means 0.1 mL of the sample solution is mixed in the 10 mL Palintest tube. Three replicates are run per sample and the average of the three are compared to the expected NH_4 concentration in mg/L based on the dilution from the 100 mg/L stock solution. The Vernier Go ammonium ISE is calibrated with the high and low standards that it came with. The Vernier Go ISE is dipped in the 50 mL samples and a reading is taken over 3 minutes. Three readings are done using the ISE, and the average from the three are compared to the expected concentration. The following table (Table 3-6) and graphs (Figure 3-11 & Figure 3-12) summarize the results of the calibration.

Table 3-6. Calibration results with distilled water solution

Step	**% as 100 mg/L stock solution**	**% as distilled water**	**Solution Volume (mL)**	**Expected $[NH_4]^+$ concentration (mg/L)**	**Palintest**				**Vernier ISE**	
					Dilution Factor	***Ave. Palintest Reading (mg NH_4/L)***	***Effective Reading***	***St. Dev.***	***Ave. (mg NH_4/L)***	***St. Dev***
0	0	100	50	0	1	0.02	0.02	0.01	0.06	0.01
1	20	80		20	100	0.23	23	1.73	18.61	1.94
2	40	60		40	100	0.54	54	1.53	44.40	0.24
3	60	40		60	100	0.72	72	9.54	65.28	1.00
4	80	20		80	100	0.91	91	2.65	78.36	0.75
5	100	0		100	100	0.13	125	1.53	95.93	2.36

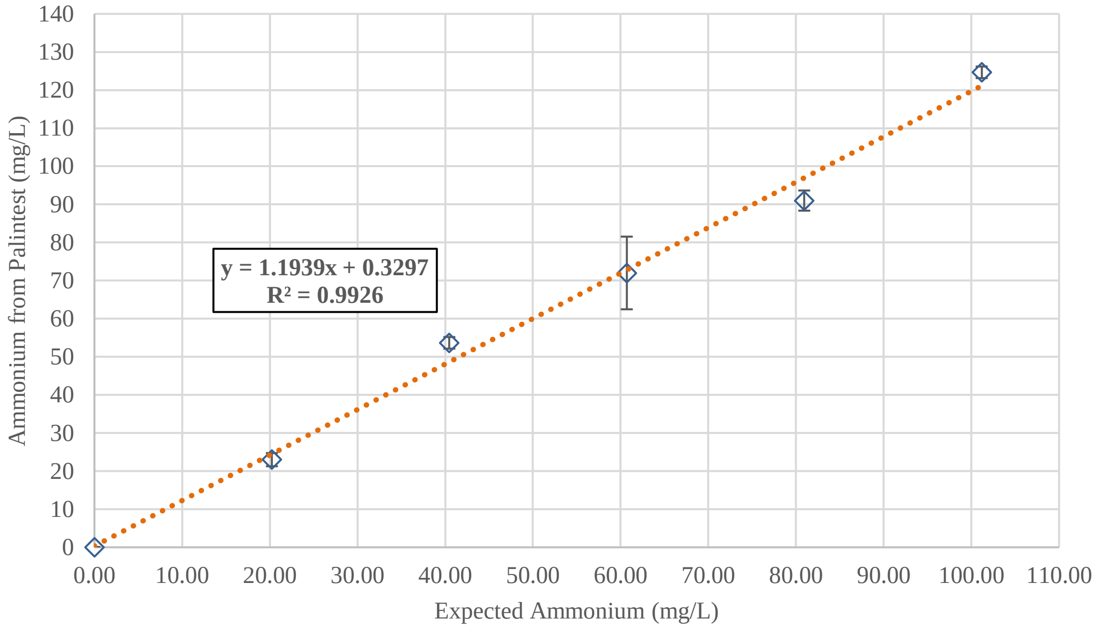

Figure 3-11. Calibration of Palintest ammonium readings with distilled water solutions

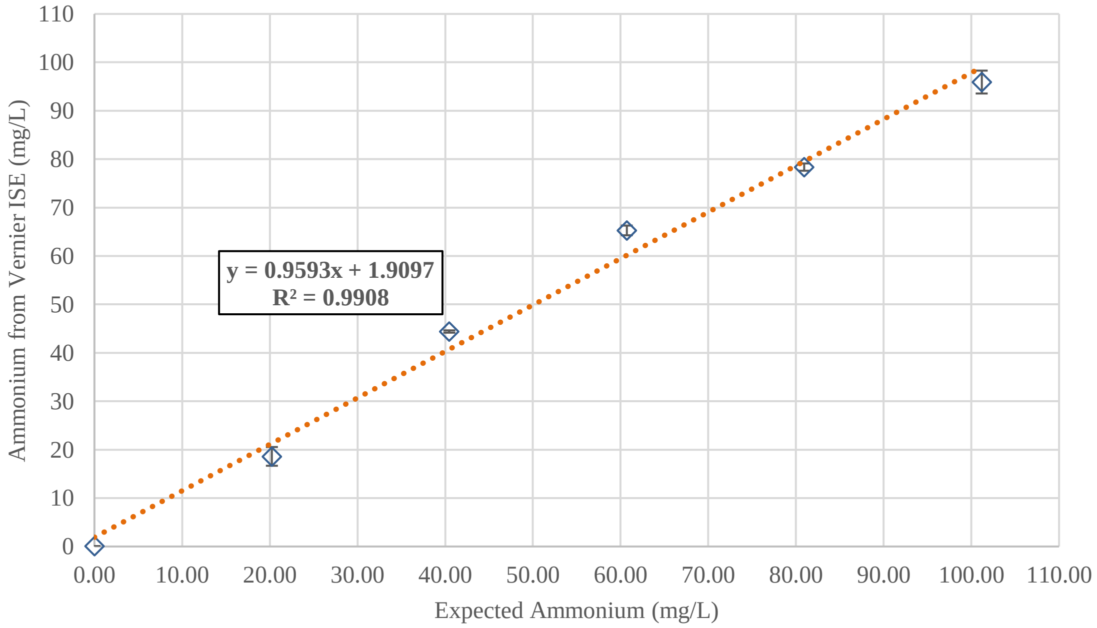

Figure 3-12. Calibration for Vernier Go Ammonium ISE with distilled water solutions

As shown in Figure 3-11 and Figure 3-12, the trendline closely shows a linear trend. This is expected since the increase in ammonium in each step is linear. The results generally show some good correlation between what is expected based on the amount of NH_4Cl added and the reading from both instruments. This indicates that the instruments have an acceptable level of detection for ammonium concentration and generally can detect ammonium with sufficient accuracy within the 0 to 100 mg/L range, even with dilution for the Palintest.

The effluent solution started with a 200 mL sample from the SBR taken after 233 days of operation. The sample is dosed with 4mg of NH_4Cl with each step to increase the ammonium concentration of the sample at an expected 7 mg/L per step. The testing with the Palintest and Vernier ISE are the same for the effluent solution as was with the distilled water solutions. It should be noted that the recorded amount of NH_4Cl in the following table (Table 3-7) is based from the total amount of NH_4Cl added at the step. For example, in step 3 12.50 mg of NH_4Cl is added but this already includes the NH_4Cl added from steps 1 and 2, at 4.30 mg and 4.2 mg respectively. The following table summarizes the added NH_4Cl per step with each kind of solution and the expected ammonium concentration.

Table 3-7. Calibration results with effluent solution

					Palintest				**Vernier ISE**	
Step	**NH_4Cl added (mg)**	**Expected Increase in NH_4 (mg/L)**	**Solution Volume (mL)**	**Expected $[NH_4]^+$ concentration (mg/L)**	***Dilution Factor***	***Ave. Palintest Reading (mg NH_4/L)***	***Effective Reading***	***St. Dev.***	***Ave. (mg NH_4/L)***	***St. Dev***
0	0	0.00	200	21.67	1	0.22	21.67	0.58	9.99	0.03
1	4.30	7.23		28.90	100	0.29	29.00	1.73	13.73	0.23
2	8.50	14.30		35.97	100	0.36	35.67	0.58	17.67	0.21
3	12.50	21.03		42.70	100	0.45	44.67	0.58	21.38	0.08
4	16.80	28.27		49.94	100	0.54	54.33	2.52	25.16	0.10
5	20.80	35.00		56.67	100	0.66	66.33	4.51	28.89	0.06

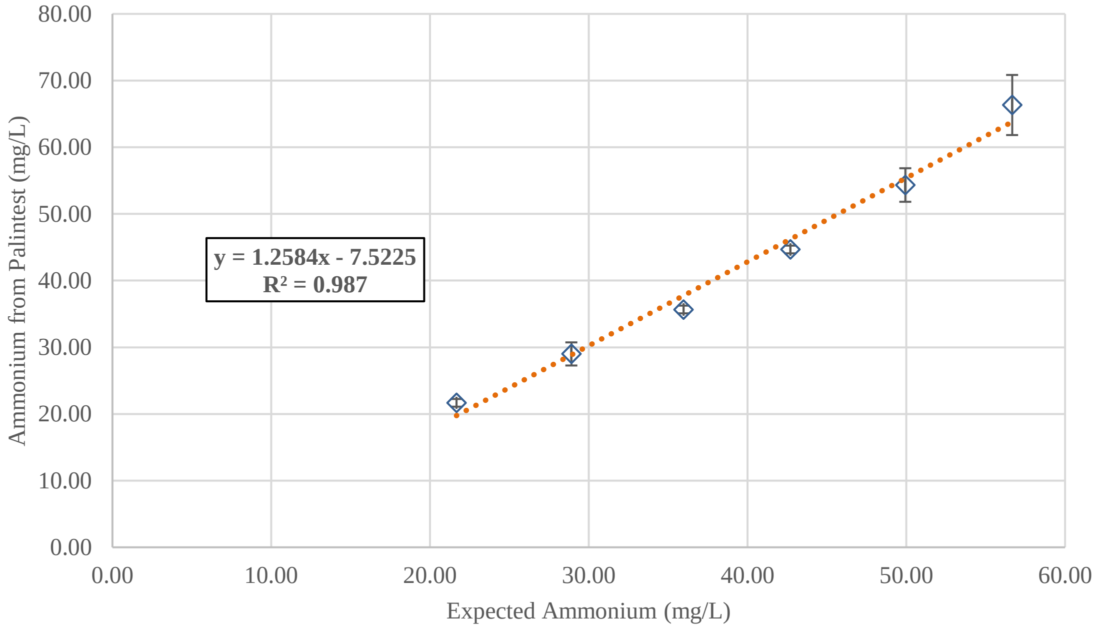

Figure 3-13. *Calibration for Palintest ammonium readings with effluent solutions*

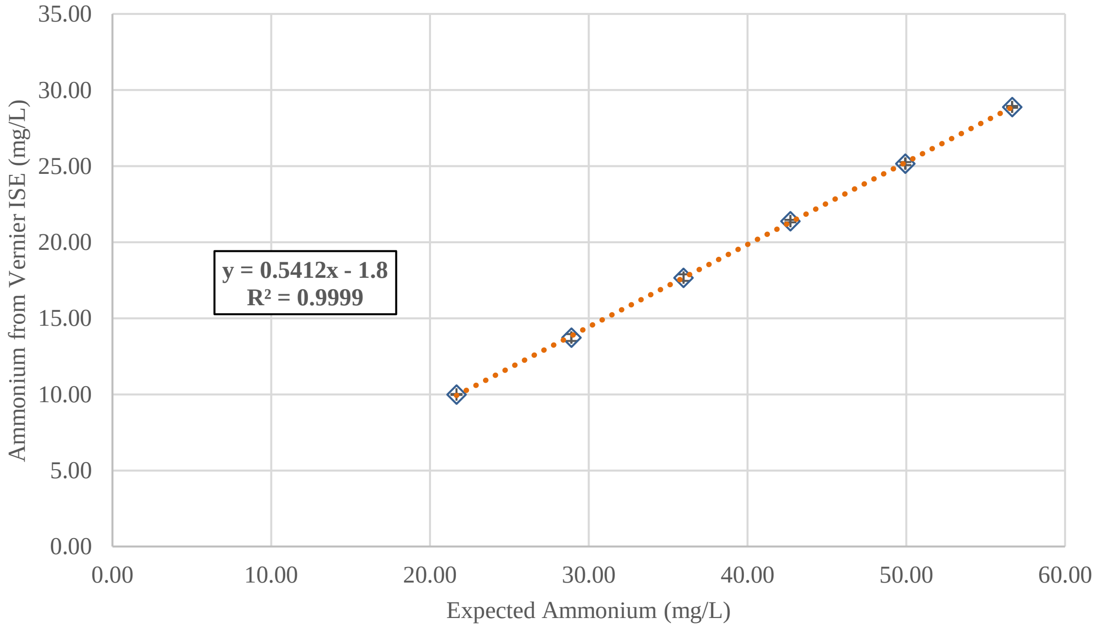

Figure 3-14. *Calibration for Vernier Go Ammonium ISE with effluent solutions*

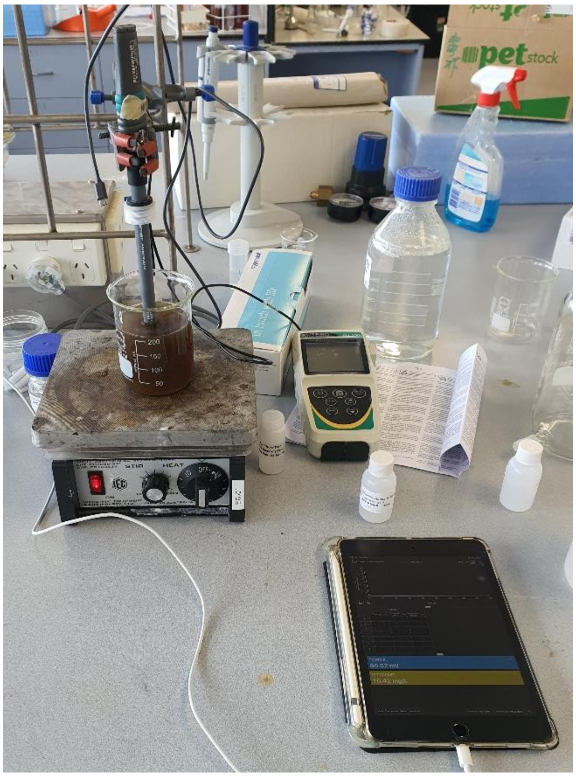

Figure 3-15. Vernier Go ammonium ISE reading effluent solution for the calibration test

As shown in Figure 3-13 and Figure 3-14, a linear trend can be observed comparing the instrument readings with the expected ammonium concentration based on added NH_4Cl. Based on the results of the Palintest readings, the Palintest measures closely to what is expected. It should be noted that the initial expected ammonium reading is based on the Palintest of the raw effluent. With increase with NH_4Cl, the Palintest generally has good correlation with what is expected. The Vernier Go ISE, on the other hand, seems to read lower than what is expected. However, comparing the increase in reading with each step from the ISE is closely correlated to the amount of NH_4Cl added in grams. For example, the addition of 4.2 mg of NH_4Cl in Step 2 from Step 1 has an increase of 3.90 mg/L in reading. This is likely due to the nature of ISE's use of being dipped into the solution instead of being sampled at a fixed volume. The presence of amines in the effluent, which has cow manure contents, might be affecting the concentration readings for the ISE. In effect, the calibration results are considered to have acceptable levels of measurement of the effluent especially for the Palintest. Since the use case of the ISE is to mainly observe the nitrification rate over a cycle and over continued aeration with pH adjustment, this calibration is accepted.

3.4.2 Nitrites

The nitrite concentration in the influent and effluent of the laboratory-scale SBR is measured by colorimetric analysis using Palintest reagents and photometer.

The method for colorimetric analysis of nitrites is facilitated by the reaction of nitrites with sulphanilic acid in an acidic solution. A diazo compound pairs with N-(1-naphthyl)-ethylene diamine that forms a reddish dye. The reagent for this reaction is contained in a single tablet called the Nitricol tablet for this method. This test method can measure in the range of 0 to 0.5 mg NO_2-N/L (Palintest, 2018d).

Figure 3-16. 10 mL Palintest tubes with diluted samples with Nitricol mixed

The low range of measurement requires some dilution for testing the samples from the bioreactor. A 10mL sample or diluted solution is mixed with the Nitricol tablet in the Palintest tube. The mixture is left to develop colour for 10 minutes. Once the colour is fully developed, the photometer is turned on and the Phot 24 photometer program for nitrite nitrogen is selected. The photometer is blanked with a Palintest tube filled with 10mL of distilled water, and the sample is placed in the photometer with the cover and read. The nitrite nitrogen and nitrite concentration in mg/L is recorded and multiplied by the dilution factor, if needed (Palintest, 2018d).

3.4.3 Nitrates

The nitrate concentration in the influent and effluent of the laboratory-scale SBR is measured by colorimetric analysis using Palintest reagents and photometer.

The method for measuring nitrate concentration using the Palintest photometer involves the reduction of the nitrates into nitrites. The reduction stage is facilitated by a zinc-based powder called Nitratest powder and tablet that enables rapid flocculation after one minute called Nitratest Tablet. The reduction step is done with 20mL of sample, which is decanted and measured in the 10mL Palintest tube. The test measures over a range of 0 to 1 mg NO_3-N/L. The colour development occurs after the reduction by measuring the nitrites similarly as in Section 3.4.1.1 with the use of a Nitricol tablet as reagent and development of a reddish hue (Palintest, 2018c).

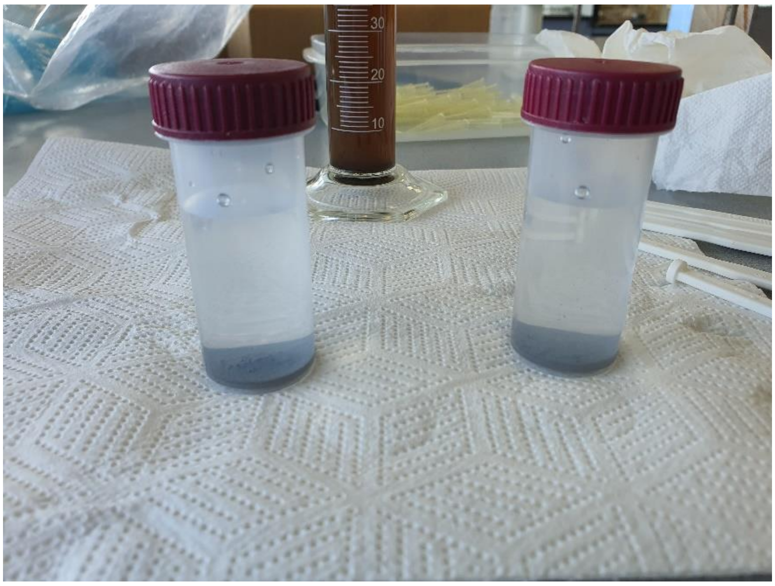

Figure 3-17. *20 mL diluted samples after mixing and settling with Nitratest tablets and powder*

Figure 3-18. *10 mL decanted samples with Nitricol tablets*

Figure 3-19. 10 mL Palintest tubes with decanted samples with Nitricol mixed

The low range of measurement requires some dilution for testing the samples from the bioreactor. A 20mL sample or diluted solution is mixed with a spoonful of Nitratest powder and one Nitratest tablet in a 20mL tube. The mixture is shaken for one minute and left to stand for one minute after. The tube is then inverted three to four times and left to stand for three minutes to allow the flocs formed to settle. The cap is then unscrewed and the rim of the tube is wiped clean with tissue. The solution carefully decanted into a 10mL Palintest tube. One Nitricol tablet is dropped and mixed in with the decanted solution. The mixture is left to develop color for 10 minutes. Once the colour is fully developed, the photometer is turned on and the Phot 23 photometer program for nitrate nitrogen is selected. The photometer is blanked with a Palintest tube filled with 10mL of distilled water, and the sample is placed in the photometer with the cover and read. The nitrate nitrogen and nitrate concentration in mg/L is recorded and multiplied by the dilution factor, if needed (Palintest, 2018c).

3.4.4 Alkalinity

The alkalinity in the influent and effluent of the laboratory-scale SBR is measured by colorimetric analysis using Palintest reagents and photometer.

The method for colorimetric analysis of alkalinity is measured over a range of 0 to 500 mg CaCO3/L with colour development ranging from yellow to green then blue in increasing alkalinity. The reagent for this reaction is contained in a single tablet called the Alkaphot tablet for this method (Palintest, 2018a).

Figure 3-20. 10 mL Palintest tubes with diluted samples with Alkaphot mixed

The low range of measurement requires some dilution for testing the samples from the bioreactor. A 10mL sample or diluted solution is mixed with the Alkaphot tablet in the Palintest tube. The mixture is left to develop colour for 1 minute. Once the colour is fully developed, the photometer is turned on and the Phot 24 photometer program for nitrite nitrogen is selected. The photometer is blanked with a Palintest tube filled with 10mL of distilled water, and the sample is remixed then placed in the photometer with the cover and read. The nitrite nitrogen and nitrite concentration in mg/L is recorded and multiplied by the dilution factor, if needed (Palintest, 2018a).

3.4.5 Total and Volatile Suspended Solids

The total suspended solids of the influent, effluent, and mixed liquor of the SBR is measured according to the American Public Health Association's standard methods for the examination of water and waste water (American Public Health Association, 2017).

For this study, the volume of sample is first measured using a graduated cylinder of appropriate size and gradation. The filter used to separate the suspended solids from the sample is the Whatman^{TM} 110mm diameter ashless filter papers. The paper is weighed and the weight is recorded. The paper is fitted on a ceramic cone with the same diameter over a conical flask with side arm that is attached to a vacuum to aid in filtering. The sample of known volume is then poured through the filter paper on the ceramic cone until all liquid has dripped into the conical flask. Any left debris on the sides of the ceramic cone are wash with distilled water to flow onto the filter paper.

The moist filter paper with the filters solids is weighed and put in an oven set at 100±5 °C for 24 hours. The dried filter paper's weight is measured and recorded.

The TSS measurement in g/L is calculated by taking the difference in weight of the moist filter paper and dried filter paper, then subtract the weight of the filter paper from this difference. This is then divided by the volume of sample filtered. The following equation shows the formula used for calculating the TSS.

The same dried filter paper from the total suspended solids test is weighed and placed in a crucible of known weight. The crucible with the dried filter paper is placed in a furnace set at 550°C and ignited for four hours. The weight of the crucible with the remaining ash is then measured and recorded. The VSS in g/L is taken as the difference of the weight of crucible with dried filter paper and crucible with ash. The weight of the filter paper is then subtracted from this difference then divided by the volume of the sample filtered. The following equation shows the formula used for calculating the VSS.

This is measured once every few months since the small sample from decanting is typically used for measuring ammonia, nitrites, and nitrates.

4 Experimental Results

The results chapter presents the results from testing sample from SBR1 and SBR2. The results to be discussed first are from testing for TSS. This is followed by the testing results for ammonium and effective ammonium removal efficiency between influent and effluent. The total inorganic nitrogen and nitrates concentration tests are shown after as these are expected indicators of nitrification. Lastly, test runs using the Vernier Go Ammonium ISE to measure rate of nitrification over a cycle and extended aeration are shown.

It should be noted that due to the Winpact controller’s automated pH control, SBR 2 was dose with about 40 mL of NaOH after about 130 days of operation which pushed the pH of the mixed liquor to 14. Due to this event, further testing on SBR 2 after 130 days is stopped, and the biomass in the reactor is assumed to be inactive already. The nitrification rate measurements using the Vernier Go Ammonium ISE are all done in SBR 1 since this was operating for a much longer period of time.

4.1 Total Suspended Solids and Sludge Retention Time

The total suspended solids of the SBR 1 was measured after 130 days, 137 days, and 190 days of operation. Due to the small volume of the reactor, and small sample volume obtained from decanting, the testing for TSS in the effluent and mixed liquor were limited to these three period in the experiment as to not heavily change the biomass concentration inside the reactor. The SRT for SBR 1 is calculated using Equation 3-1 and found to be around 12 to 13 days around 130 to 137 days of operation, with the influent ammonium concentration in the 50 to 100 mg/L range at this stage. After 190 days of operation, the SRT is measured to be at 37 days. This is noticeably high, and most likely due to the lower volume decanted at this stage of the experiment since the SBR was being acclimated to a high ammonium concentration at this period, in the range of 200 to 400 mg/L. The TSS in the reactor is noticeably higher at this point compared to around 130 to 137 days, and can be attributed to not having any intentional and regular sludge wasting.

For SBR 2, the SRT is measured to be around 9 to 12 days after 60 to 67 days of operation, with the influent ammonium concentration in the 50 to 100 mg/L range at this stage. The SRT is also found to be much higher after prolonged operation after 120 days at 38 days, similar to SBR 1 after 190 days with the same influent ammonium concentration range of about 200 to 400 mg/L. The same reasons of lower decanting volume, and lack of intentional and regular sludge wasting can be attributed to this increase. The lower decanting volume evidently affects the higher SRT as the TSS in the reactor was actually measured to be slightly lower after 120 days of operation as compared to 60 to 67 days of operation, coming from 17.12 g/L to 15.52 g/L. However, despite a higher TSS concentration in the effluent at 120 days of operation at 10.18 g/L compared to 7.07 g/L and 9.16 g/L at 60 and 67 days of operation, respectively, the SRT is still higher and this is most likely due to the lower decanting volume. The lower decanting volume at this stage is due to lower volume of influent being introduced that is of a higher magnitude of ammonium concentration. The following table summarizes the results from the TSS testing and SRT calculation for SBR 1 and 2 (Table 4-1).

Table 4-1. TSS Measurement for SBR 1 and 2

	SBR 1	**SBR 2**
	After 130 days of operation	*After 60 days of operation*
Average Effluent TSS (g/L)	6.94	7.07
Average Reactor TSS (g/L)	18.39	17.12
Decanted Volume (L)	0.20	0.20
SRT (days)	13	12
	After 137 days of operation	*After 67 days of operation*
Average Effluent TSS (g/L)	7.93	9.16
Average Reactor TSS (g/L)	18.39	17.12
Decanted Volume (L)	0.20	0.20
SRT (days)	12	9
	After 190 days of operation	*After 120 days of operation*
Average Effluent TSS (g/L)	21.64	10.18
Average Reactor TSS (g/L)	32.65	15.52
Decanted Volume (L)	0.04	0.04
SRT (days)	37	38

Due to the small volume of samples and small amount of TSS, measurements of VSS was not obtained for this experiment. The TSS results are used in calculating rates of nitrification and DO consumption in the following sections.

4.2 Ammonium Concentration and Removal

The ammonium concentration of the influent and effluent of the SBR is measured according to the Palintest procedure as discussed in Section 3.4.1. The removal efficiency of ammonium is calculated two ways from the measurements of the influent and effluent. The first calculation is done by a straightforward comparison of the influent and effluent ammonium concentration, which is referred to as percent removal of ammonium ($R_{I\text{-}E}$). The following equation shows how the percent removal of ammonium is calculated.

$$R_{I-E} = \left(\frac{Influent\ NH_4^+ - Effluent\ NH_4^+}{Influent\ NH_4^+}\right) * 100\%$$

Equation 4-1. Equation for calculating percent removal of ammonium ($R_{I\text{-}E}$)

The second calculation for removal efficiency considers the effective concentration of ammonium in the reactor after dosing by a certain volume of influent with a known concentration, and compares this to the ammonium concentration in the effluent. This is referred to as percent removal in reactor ($R_{reactor}$). This second calculation of removal efficiency captures the actual amount of ammonium removed by the biomass in the reactor by measuring the amount of ammonium before aeration, effective reactor concentration, and after aeration, which is the effluent concentration. This calculation requires the reactor concentration before dosing of an influent of known concentration, as such this is calculated from test results from consecutive days. The following equation shows how the percent removal in reactor is calculated.

$$R_{reactor} = \left(\frac{Reactor\ NH_4^+ - Effluent\ NH_4^+}{Reactor\ NH_4^+} \right) * 100\%$$

$$where\ Reactor\ NH_4^+ = (Effluent\ NH_4^+)_{T-1}\left(1L - V_{influent}\right) + (Influent\ NH_4^+)_{T-1}(V_{influent})_{T-1}$$

Equation 4-2. Equations for calculating percent removal in reactor ($R_{reactor}$)

In both equations, it should be noted that influent concentration compared to the effluent concentration of a specific day (T) is from a previous day (T-1) since one cycle is 24 hours. In Equation 4-2, calculating for the reactor concentration before a cycle starts is based on the influent concentration and amount dosed at the fill stage, which is from the day prior to the effluent sampled at the end of the cycle. For example, to get the percent removal in reactor for November 2, the influent concentration, dosing volume, and effluent concentration measured from November 1 are used to calculated the Reactor NH_4^+. This calculated Reactor NH_4^+ from November 1 is compared to the effluent NH_4^+ measured on the cycle ending in November 2. The Reactor NH_4^+ is a basic concentration and volume calculation in the reactor since the reactor has a working volume of 1 L. For measurements not from consecutive days, the Reactor NH_4^+ is estimated from the average of the nearest two previous ammonium concentration measurements, which in most cases is from the week prior.

4.2.1 SBR 1 Results

The ammonium concentration of the influent and effluent for SBR 1 is measured for 250 days. The removal efficiency is calculated based on these measurements and according to Equation 4-1 and Equation 4-2. The R_{I-E}, percent removal of ammonium, is found to be mostly between the 90 to 100% range. The R_{I-E} dipped below this range when the influent ammonium concentration was at 42 mg/L in 200 mL dose, 340 mg/L in 80 mL dose, and

670 mg/L in 40 mL dose. The lowest R_{I-E} calculated was for the concentration at 42 mg/L in 200 mL dose after 108 days of operation. This was about 10 days after the influent ammonium was stepped up from less than 10 mg/L to more than 50 mg/L. The $R_{reactor}$ fluctuates in a wider range than the R_{I-E}, anywhere from 17% to 95% over the experimental period. This value is calculated to be most stable over the period of volume dosing at 200 mL with influent ammonium concentration in the range of 34 mg/L to 124 mg/L. This period is from 115 to 154 days of operation. Both the R_{I-E} and $R_{reactor}$ are generally stable in this same period. The $R_{reactor}$ fluctuated the most once influent ammonium concentration jumped to upwards of 300 mg/L. The removal efficiency for SBR1 is summarized in Figure 4-1.

The influent, mixed liquor, and effluent ammonium concentration is presented in Figure 4-2. The effluent ammonium concentration remained less than 10 mg/L at influent ammonium concentrations less than 300 mg/L. The Reactor NH_4^+ is shown in the diamond data points, and the graph presents the effect of mixed liquor dilution on the actual amount of ammonium that the SBR actual removes over a cycle, since the Reactor NH_4^+ is closer in magnitude to the effluent concentration than it is to the influent, which is expected due to the volume dosing. Generally, effluent concentrations less than the Reactor NH_4^+ is indicative of ammonium removal still occurring in the SBR, and this is shown to have been present over the experimental period. There are two instances where the Reactor NH_4^+ falls below the effluent concentration. One instance is after 177 days of operation with an influent ammonium concentration at 340 mg/L, and another after 219 days of operation at an influent ammonium concentration of 670 mg/L. It is apparent that these two instances show the SBR seemingly recovering and continuing to remove ammonium with the Reactor NH_4^+ being above the effluent concentration after these two instances.

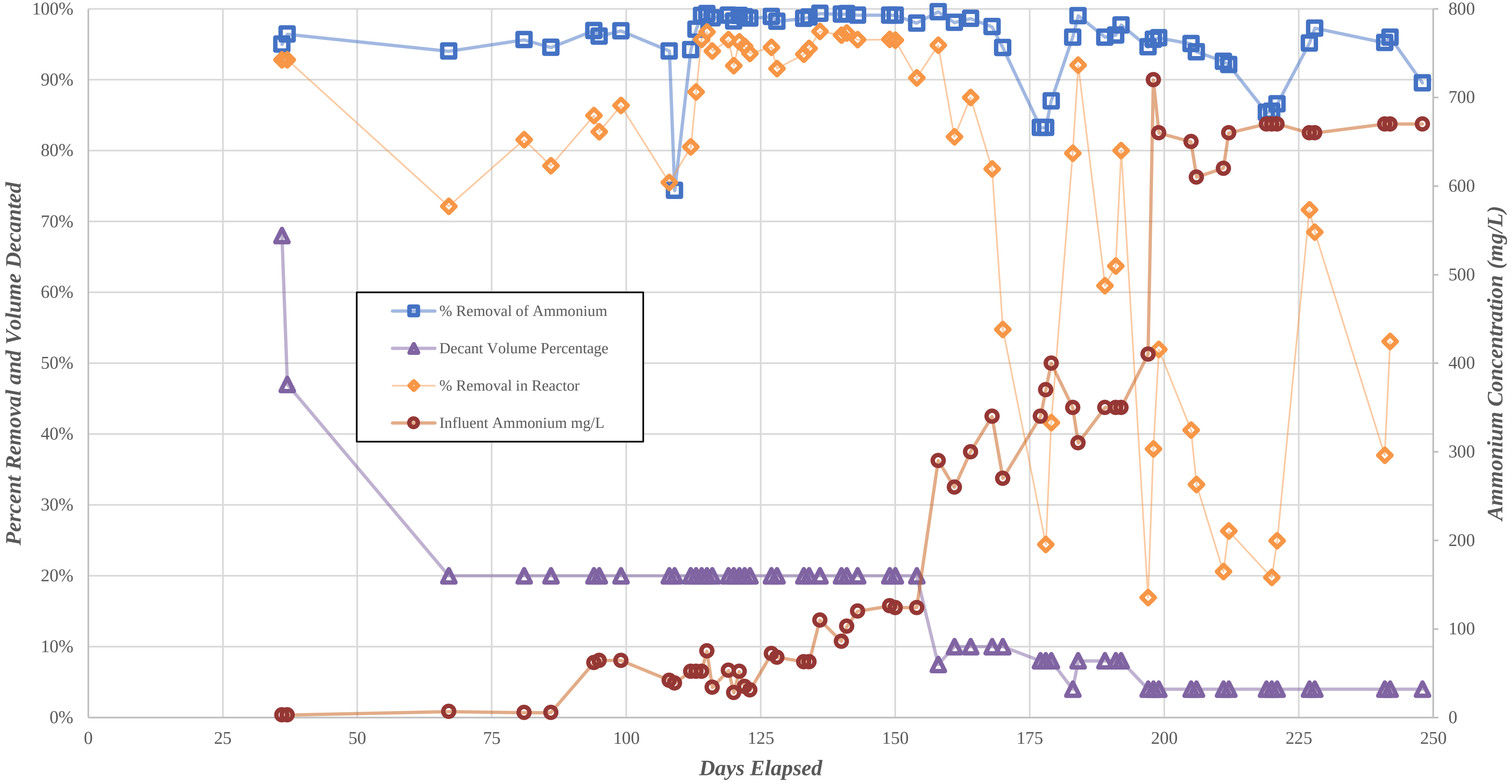

Figure 4-1. *SBR 1 ammonium removal over experimental period*

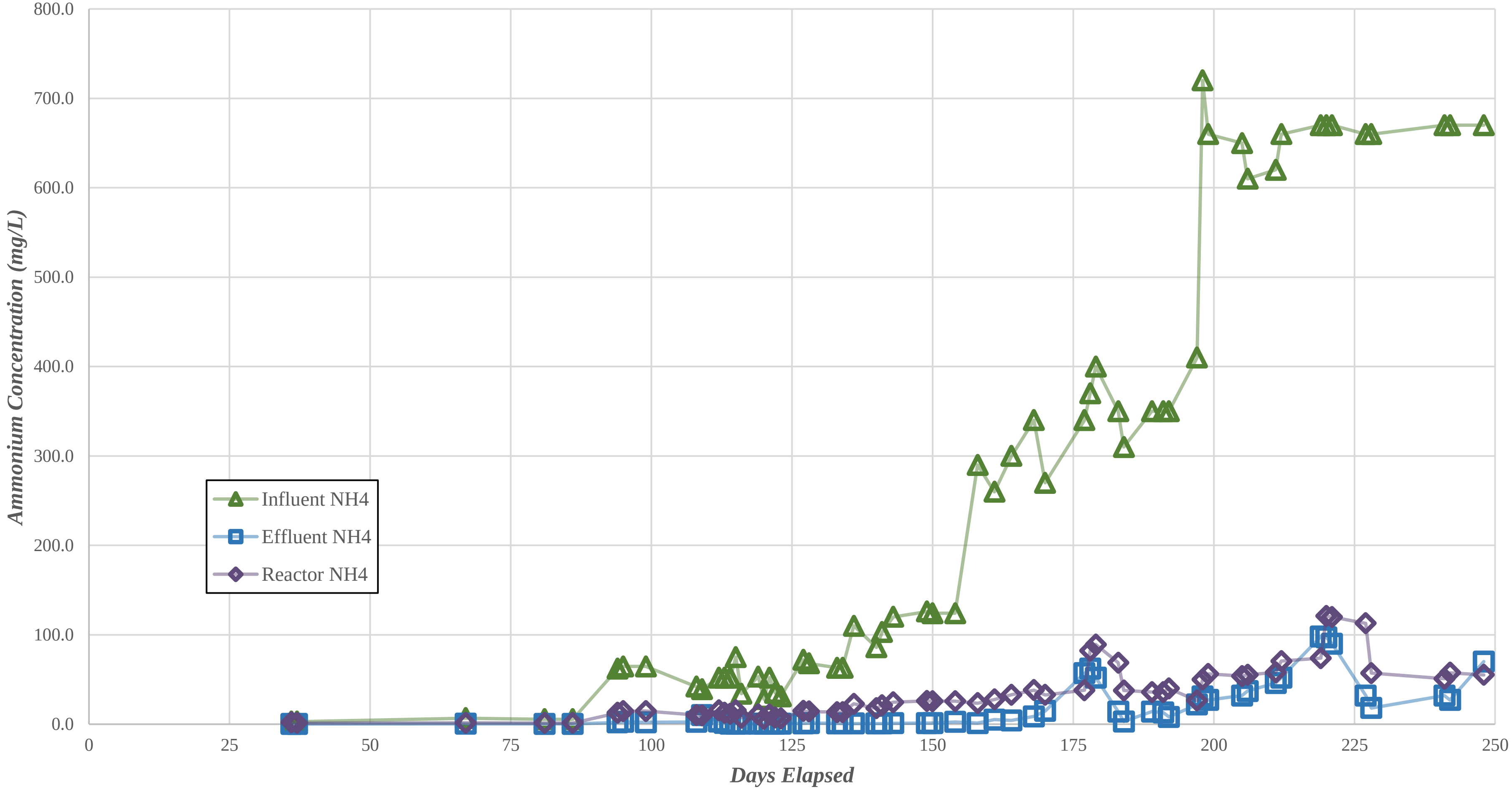

Figure 4-2. *SBR 1 ammonium concentration in influent, reactor before cycle, and effluent*

4.2.2 SBR 2 Results

The ammonium concentration of the influent and effluent for SBR 2 is measured for 142 days. The removal efficiency is calculated based on these measurements and according to Equation 4-1 and Equation 4-2. The R_{I-E}, percent removal of ammonium, is found to be mostly between the 90 to 100% range from 44 to 129 days of operation. The sludge seeding period can be observed by the fluctuating R_{I-E} before 44 days of operation, and also the influent ammonium concentration being dosed during the seeding period for SBR 2 is higher than for SBR 1. The R_{I-E} dipped below this range once after 53 days of operation when the influent ammonium concentration was at 31 mg/L in 200 mL dose. The $R_{reactor}$ fluctuates in a wider range than the R_{I-E}, anywhere from 19% to 98% over the 44 to 129 days of operation. This value is calculated to be mostly stable over the period of volume dosing at 200 mL with influent ammonium concentration in the range of 34 mg/L to 124 mg/L, being above 90% for most of this period from 115 to 154 days of operation. Both the R_{I-E} and $R_{reactor}$ are generally stable in this same period. This is similar to SBR 1. The $R_{reactor}$ fluctuated the most once influent ammonium concentration jumped to upwards of 300 mg/L, but the R_{I-E} remained still remained above 90%. This is indicative of SBR 2 being able to handle the high ammonium loading relatively better than SBR 1. The removal efficiency for SBR 2 is summarized in Figure 4-3.

The influent, mixed liquor, and effluent ammonium concentration is presented in Figure 4-4. The effluent ammonium concentration remained less than 20 mg/L for most of the operational period. Generally, effluent concentrations less than the Reactor NH_4^+ is indicative of ammonium removal still occurring in the SBR, and this is shown to have been present over the experimental period. There was no instance when the Reactor NH_4^+ fell below the effluent concentration over the period from 44 to 129 operational days. SBR 2 shows consistent ammonium removal after seeding (after 44 days of operation), and before the occurrence of overdosing of NaOH after 129 days of operation.

The overdosing of NaOH is due to an unknown error from the Winpact controller dosing SBR 2 with 40 mL of 1M NaOH when set to control pH at 7.5. Due to this, tests for SBR 2 were halted after 142 days as it was becoming more apparent that the biomass was no longer active in SBR 2.

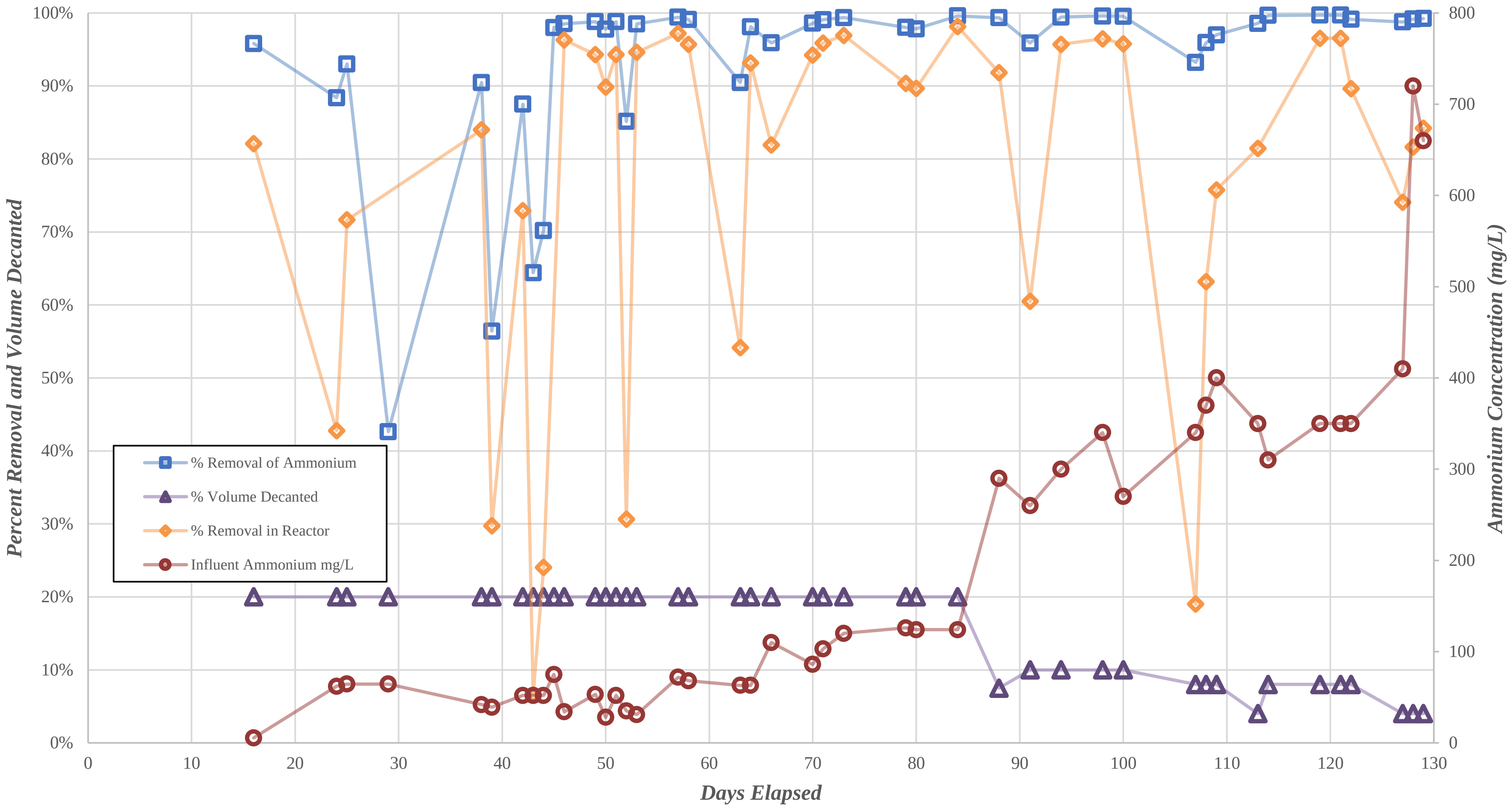

Figure 4-3. *SBR 2 ammonium removal over experimental period*

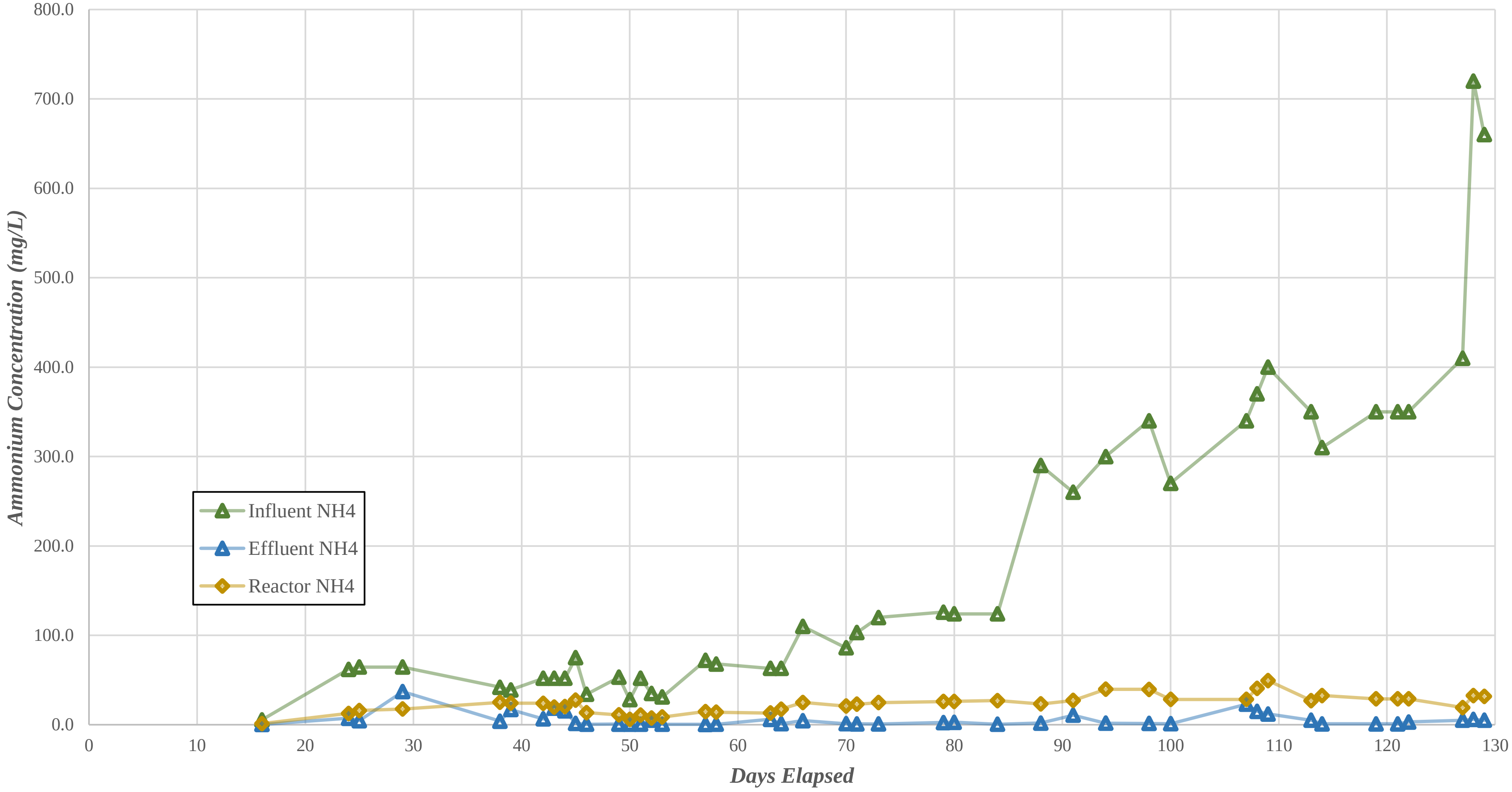

Figure 4-4. *SBR 2 ammonium concentration in influent, reactor before cycle, and effluent*

4.3 Nitrates and Total Inorganic Nitrogen Concentration

As an expected products of nitrification (Section 2.3.1), nitrites and nitrates are measured to check if the ammonium is being converted into these two compounds through ammonia-oxidizing bacteria. The presence of the nitrites and nitrates may be indicative of nitrification being the main mode of ammonium removal in the SBRs, as well as providing insight to any reduction of ammonium removal as the accumulation of either may cause inhibition of nitrification. It should be noted that there is a settle phase in the operation of the SBRs in this study, where the mixed liquor is under an anoxic condition, and denitrification may occur. Measuring nitrites and nitrates can provide indication of denitrification should nitrates are not accumulating in excessive amounts. The reactor nitrates before aeration is calculated in the same way as was for $R_{reactor}$ for ammonium. The following equation (Equation 4-3) shows how this calculation was done.

$$Reactor\ NO_3^- = (Effluent\ NO_3^-)_{T-1}\left(1L - V_{influent}\right) + (Influent\ NO_3^-)_{T-1}(V_{influent})_{T-1}$$

Equation 4-3. Equation for calculating reactor nitrates before start of cycle

The total inorganic nitrogen (TIN) concentration in this study refers to the sum of the ammoniacal-nitrogen, nitrite-nitrogen, and nitrate-nitrogen. This is calculated by adding the measurement of each nitrogen component from the Palintest procedures (Section 3.4). The total inorganic nitrogen concentration before the start of each cycle is calculated similarly to Equation 4-2 and Equation 4-3, which is shown in the following equation (Equation 4-4).

$$Reactor\ TIN = (Effluent\ TIN)_{T-1}\left(1L - V_{influent}\right) + (Influent\ TIN)_{T-1}(V_{influent})_{T-1}$$

Equation 4-4. Equation for calculating the TIN before start of cycle

4.3.1 SBR 1 Results

A similar graph to the ammonium concentration graphs (Figure 4-2 and Figure 4-4) are created but with nitrate concentrations for SBR 1 (Figure 4-5). In general, nitrates concentrations are found to be steady before and after a cycle. Removal of nitrates is not expected since the cycles prioritize the aeration phase to facilitate and maximize ammonium removal. Notably nitrates seem to have accumulated in the SBR despite considering the increase in nitrate concentration from the influent. The influent nitrate concentration measurements seem to have risen with higher ammonium concentration with the use of NH_4Cl. The reason for this is unclear, and may require further analysis of possible reactions between the cow manure and NH_4Cl. The testing shows that effluent nitrate concentrations are generally less than the influent nitrate concentration until after 189 days of operation, where there is a measured jump in nitrate concentration from the reactor and effluent. This jump lagged the jump in nitrate concentration from the influent by 30 days from when the influent started to have nitrates upwards of 300 mg/L. This could be indicative of nitrate accumulation occurring from the nitrification process. Since there is an increase in ammonium concentration during the same period to approximately above 290 mg/L, there would also be an increase in conversion of ammonium to nitrates. With the high influent nitrate concentration at above 500 mg/L, the ammonium in the reactor before cycle and effluent are found to be above the influent nitrate concentration to upwards of 988 mg/L. This could be indication of the mentioned nitrate accumulation, and the graph shows that after 200 days of operation a jump occurs for the nitrate concentration to be above 1500 mg/L of nitrates in both the effluent and reactor. It could be expected that if experimentation were continued that this nitrate concentration would hit a steady state, similarly to how the nitrates of SBR 1 was until 150 days of operation. An extended period of settling could aid in regulating the nitrate concentration in the effluent.

The TIN measurements and calculations are summarized in Figure 4-6. The reactor and effluent TIN generally had a similar concentration in contrast to the influent concentration. This indicates that although there is conversion of ammonium to nitrates the nitrogen from the mixed liquor may not have been getting removed efficiently during the denitrification via the anoxic conditions from the settle phase. This is expected with the react phase prioritized in the SBR operation to facilitate ammonium removal. The slight difference between the reactor and effluent TIN may indicate that some denitrification must be occurring but in general the TIN concentration in the SBR tends to hit a steady state. This is also indicative of ammonium not accumulating in the SBR, and further indication that ammonium removal is being facilitated in SBR 1.

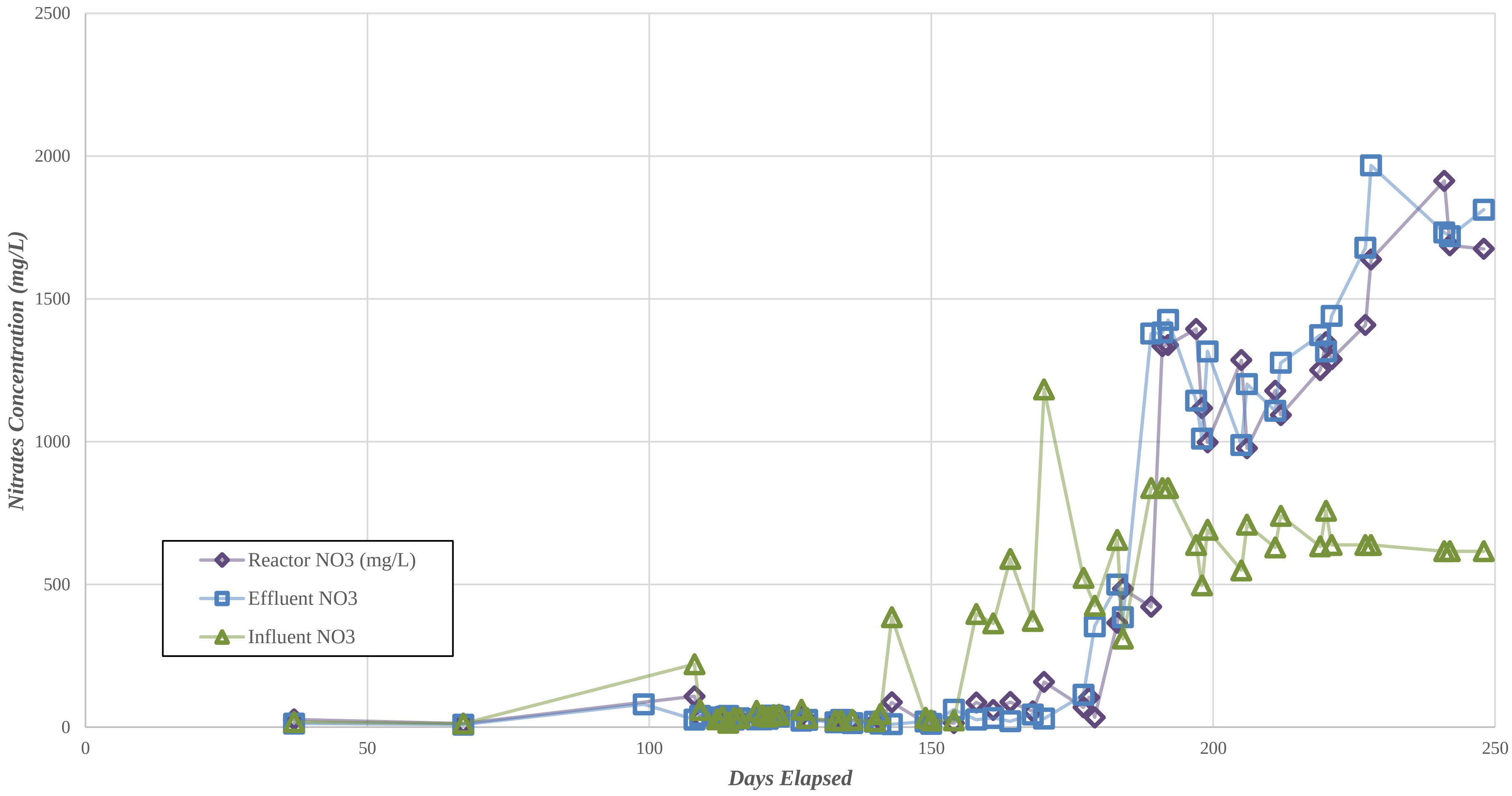

Figure 4-5. *SBR 1 nitrates concentration in influent, reactor before cycle, and effluent*

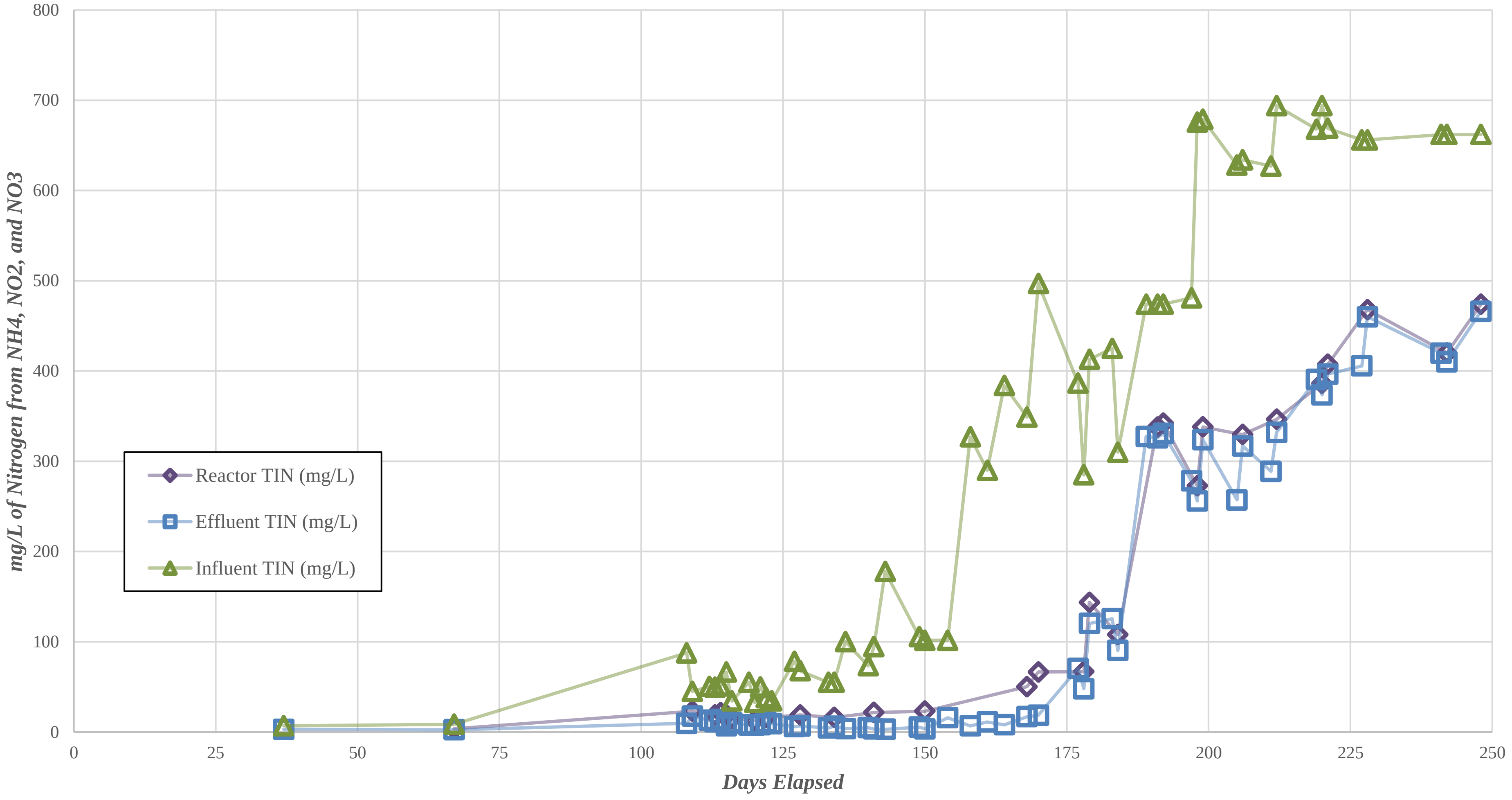

Figure 4-6. *SBR 1 total inorganic nitrogen in influent, reactor before cycle, and effluent*

4.3.2 SBR 2 Results

SBR 2 shows a similar trend to SBR 1 with regards to the increase in nitrates concentration of the effluent and reactor. From 40 to 100 days of operation, the nitrates concentration was well below the influent nitrate concentration. After 100 days of operation, nitrate accumulation in the reactor seems to have occurred with the concentration of the reactor and effluent exceeding that of the influent. Again, since the same influent is used in dosing SBR 2, the nitrates in the influent shows a jump that may require further investigation on the reactions of cow manure with NH_4Cl. Figure 4-7 summarizes the nitrate concentration for SBR 2.

The TIN for SBR 2 is found to be more stable over the experimental period. The calculated reactor nitrate concentration is generally above the effluent concentration, which indicates some nitrogen being removed from the system. This could be from some denitrification occurring during the anoxic conditions of the settle phase in the cycles. This is similar to what is observed with the results from SBR 1, and could be indicative of the presence of denitrifying bacteria in the biomass, although these may not have enough organic matter to utilize for denitrification given that the anoxic condition follows the aeration phase, which may have consumed most of the possible carbon sources for the denitrifying bacteria that may be present. Supplementation with carbon sources during the anoxic condition, and lengthening of settle phase are possible measures to provide better nitrogen removal. The accumulation of nitrates is likely accounting for the increase in TIN over time since there is an increase in influent nitrate and nitrification. Since ammonium removal is present over the experimental period, the conversion to nitrates is likely happening from the aeration phase and both the results of the nitrates and TIN concentration are indicative of this.

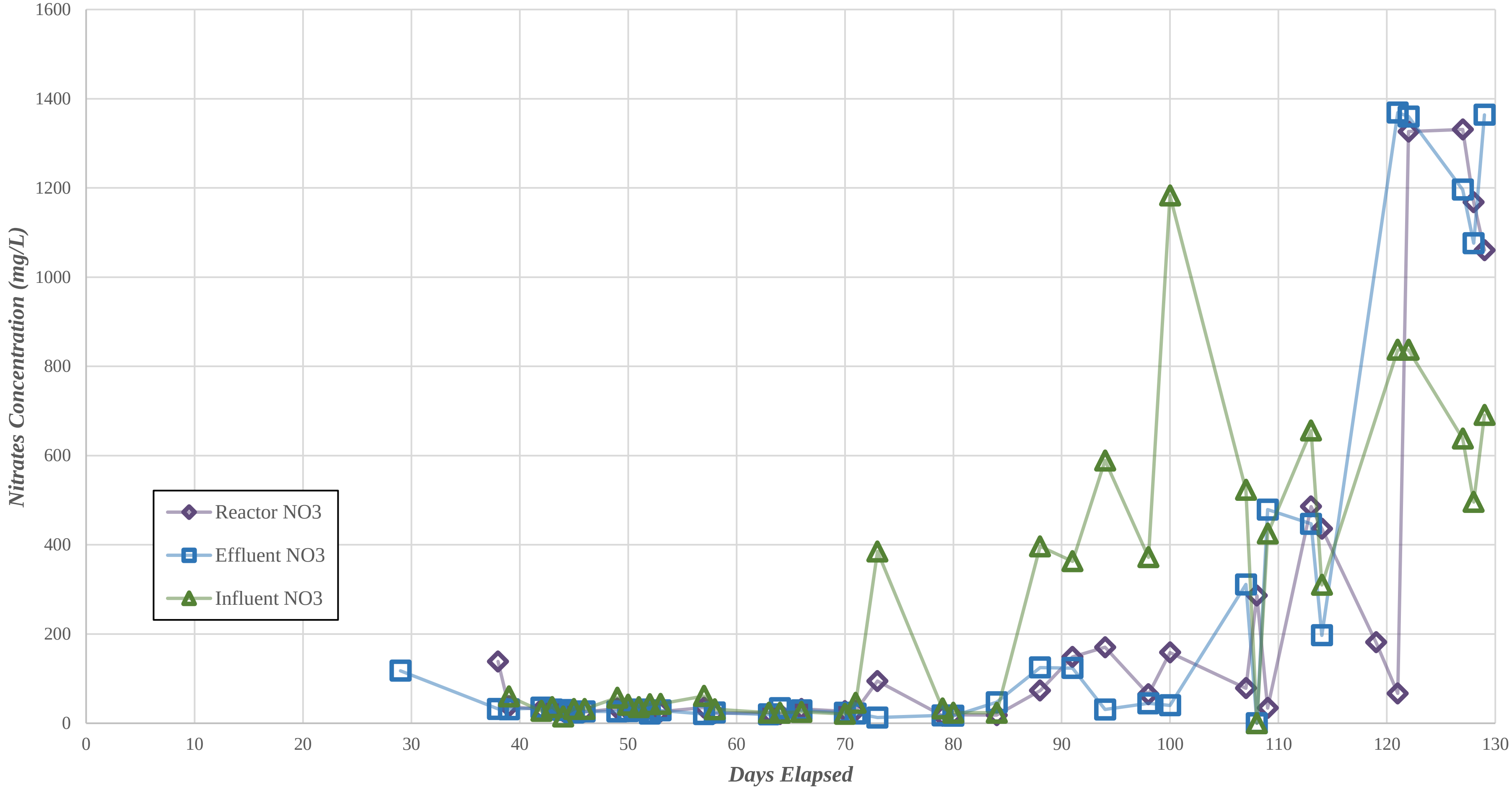

Figure 4-7. *SBR 2 nitrates concentration in influent, reactor before cycle, and effluent*

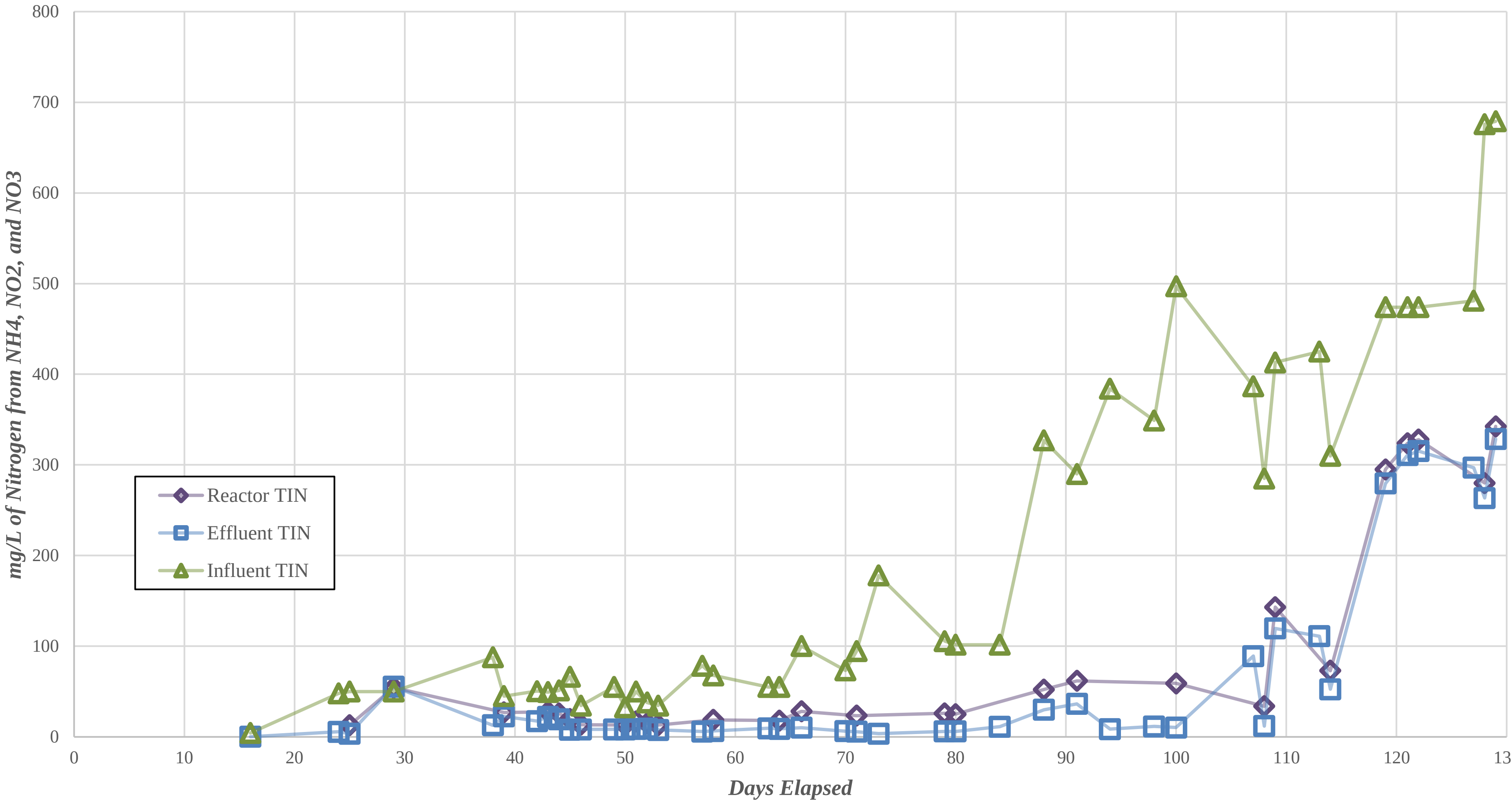

Figure 4-8. *SBR 2 total inorganic nitrogen in influent, reactor before cycle, and effluent*

4.4 Rate of nitrification

The rate of nitrification over a single cycle is measured mainly using the Vernier Go Ammonium ISE, since the probe can log ammonium concentration in real-time and does not require any sampling. The reactor and effluent ammonium concentrations and alkalinity in $CaCO_3$ are measured and calculated using the Palintest as well. The pH and DO data is obtained from the Winpact controller and aligned with the data logs from the Vernier ISE.

4.4.1 Nitrification during a cycle

The following table (Table 4-2) summarizes the four trials for measuring rate of nitrification in a typical cycle.

Table 4-2. Starting and ending ammonium concentration in reactor during nitrification measurement during typical cycles

Date	**Palintest**				**Vernier**		**pH**		**DO (mg/L)**
	Reactor NH_4^+ (mg/L)	***Effluent NH_4^+ (mg/L)***	***Reactor $CaCO_3$ (mg/L)***	***Effluent $CaCO_3$ (mg/L)***	***Starting NH_4^+ (mg/L)***	***Ending NH_4^+ (mg/L)***	***Starting pH***	***Ending pH***	
Oct 16 to 17	33 to 38*	9 to 15*	1144 to 3585*	160 to 650*	51.58	16.82	7.11	5.88	**
Oct 24 to 25	80	54	4800	5000	61.10	30.79	7.77	5.74	4.01
Oct 26 to 27	89	48	1968	400	63.71	40.84	7.91	5.65	1.56
Oct 30 to 31	69	14	916	850	33.16	3.66	9.40	6.56	1.62

**Data from average from October 16 and 18 effluent samples, due to loss of sample from the 17th*

*** DO data from October 16 to 17 was 0 because the probe was scoured during this period*

The Palintest results show a reduction of ammonium concentration ranging from 23 mg/L to 41 mg/L with an average change in ammonium of about 36 mg/L. The Vernier ISE logged ammonium concentration reduction from 22.87 mg/L to 34.76 mg/L with an average of 29.36 mg/L ammonium removed in a cycle. The readings show that the Palintest has relatively higher readings than the Vernier ISE, which is in line with what is found from the calibration of the two methods (Section 3.4.1.3). These readings do indicate that ammonium removal is occurring in the SBR, even at starting ammonium concentrations of upwards of 60 mg/L. An indication of nitrification occurring is the measurable decrease in pH from these runs. This is expected since nitrification consumes alkalinity as discussed in Section 2.3.1. A reduction in alkalinity is measured by the calculating the starting reactor and effluent alkalinity concentration as $CaCO_3$. The reduction in alkalinity is in the range of 66 mg $CaCO_3$/L to 2935 mg $CaCO_3$/L with an average of 1523 mg $CaCO_3$/L consumed over each cycle. It should be noted that there is no reduction in alkalinity measured for the cycle from October 24 to 25, with the calculations based on measurements resulting to alkalinity increasing, however this is not reflected by the reduction in pH still observed during this run. The reduction in pH has an average of 2.09, ranging from 1.23 to 2.84 change in pH. This change in pH shows the reactor mixed liquor becoming acidic after the cycle, which is indicative of nitrification consuming alkalinity in the mixed liquor and effectively reducing the pH. The DO is measured to have been around 4 mg/L during these cycles, and the low DO reading for Oct 16 to 17, Oct 26 to 27, and Oct 30 to 31 can be attributed to the DO probe being scoured solids causing low DO readings for these runs.

The readings from the Vernier ISE for the four runs are graphed in Figure 4-9, Figure 4-10, Figure 4-11, and Figure 4-12. These graphs show a similar trend between the ammonium and pH reduction over time. The two parameters follow a similar trend that can be described to be exponential that decays until it hits steady state. It can be seen that, up until 600 minutes of aeration, the ammonium decreases, after which minimal change in the ammonium concentration is observed. The pH follows a similar trend and does not change much even if aeration extends for another 600 minutes. The run from October 30 to 31 does show ammonium reduction up until the 800 minutes of aeration, which might be due to the relatively high starting pH of this run compared to the others, at 9.40 while the others started below 8 pH. The DO is also presented in the graphs, except for the run from October 16 to 17 (See note on Table 4-2). The DO is seen to be held relatively constant over the period of aeration. It can be seen that even after the point where ammonium reduction and pH change stop, around 600 to 800 minutes of aeration, these two parameters do not change despite continuous aeration. This indicates that lack of aeration may not be inhibiting the nitrification process during the aeration, since the DO from 600 minutes to 1200 minutes of aeration is no longer utilized to remove ammonium. Due to this pH is suspected to be inhibiting ammonium concentrations from further reducing. From this observation, the readings are done but at extended aeration but with manual pH adjustment using 1M NaOH dosed anywhere from 1 to 3mL at times when pH is observed to be at 6 or less, which is observed to be the point where ammonium and pH no longer occurs from these four runs. The following subsection discusses the results from using extended aeration to observed how low ammonium in the SBR can go with pH adjustment.

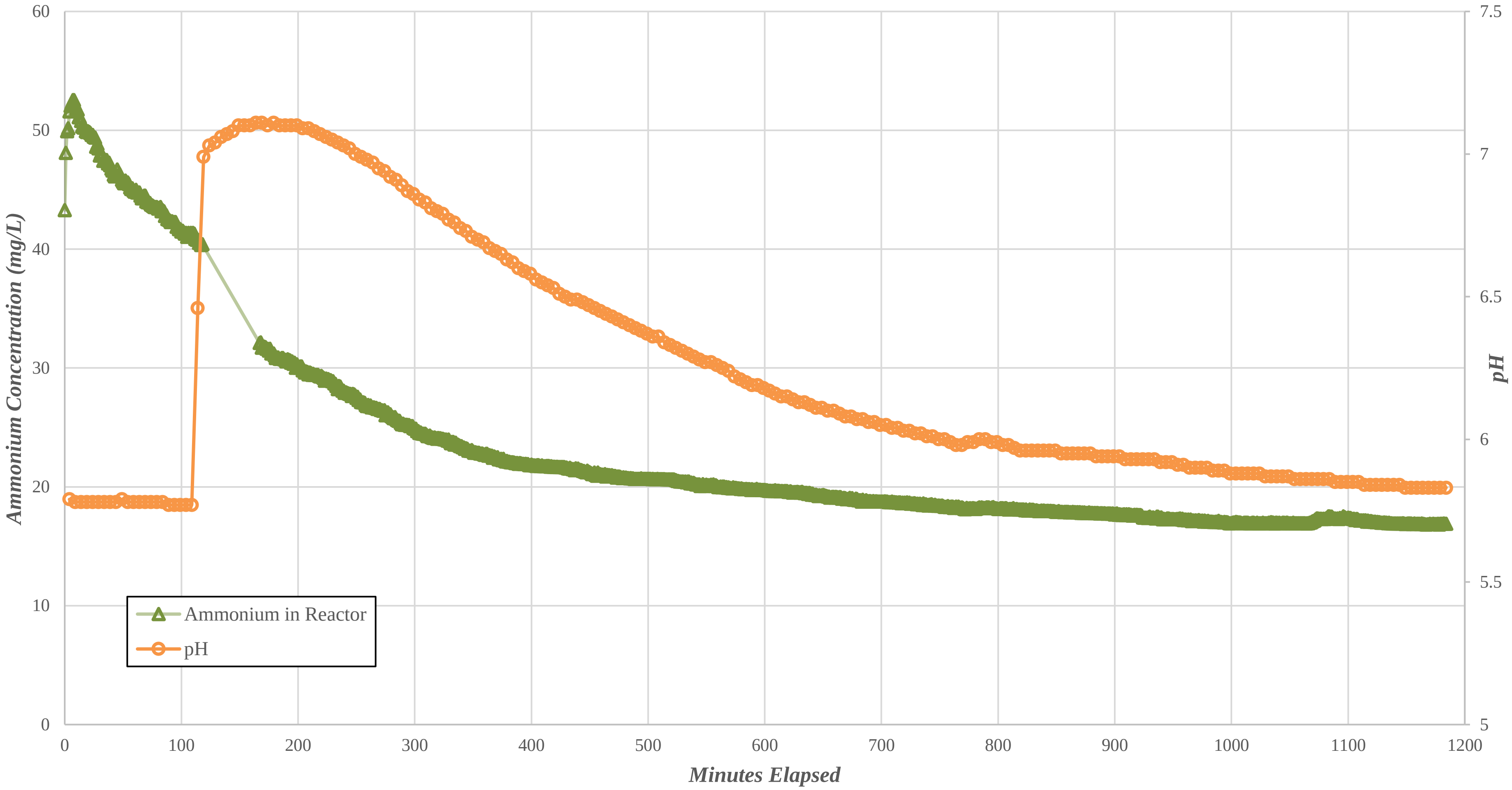

Figure 4-9. Ammonium concentration and pH measurements over SBR 1 aeration phase from October 16 to 17

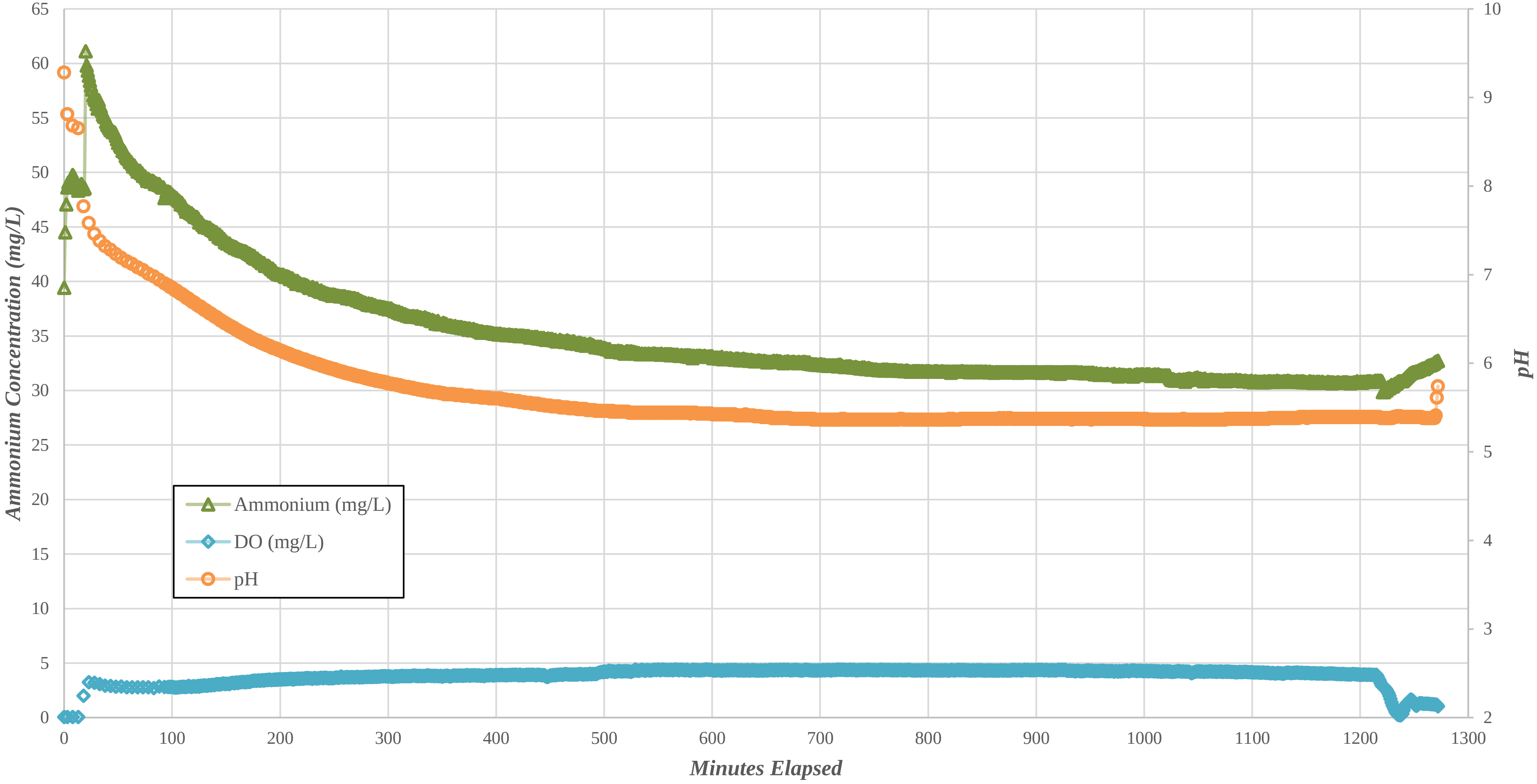

Figure 4-10. Ammonium concentration and pH measurements over SBR 1 aeration phase from October 24 to 25

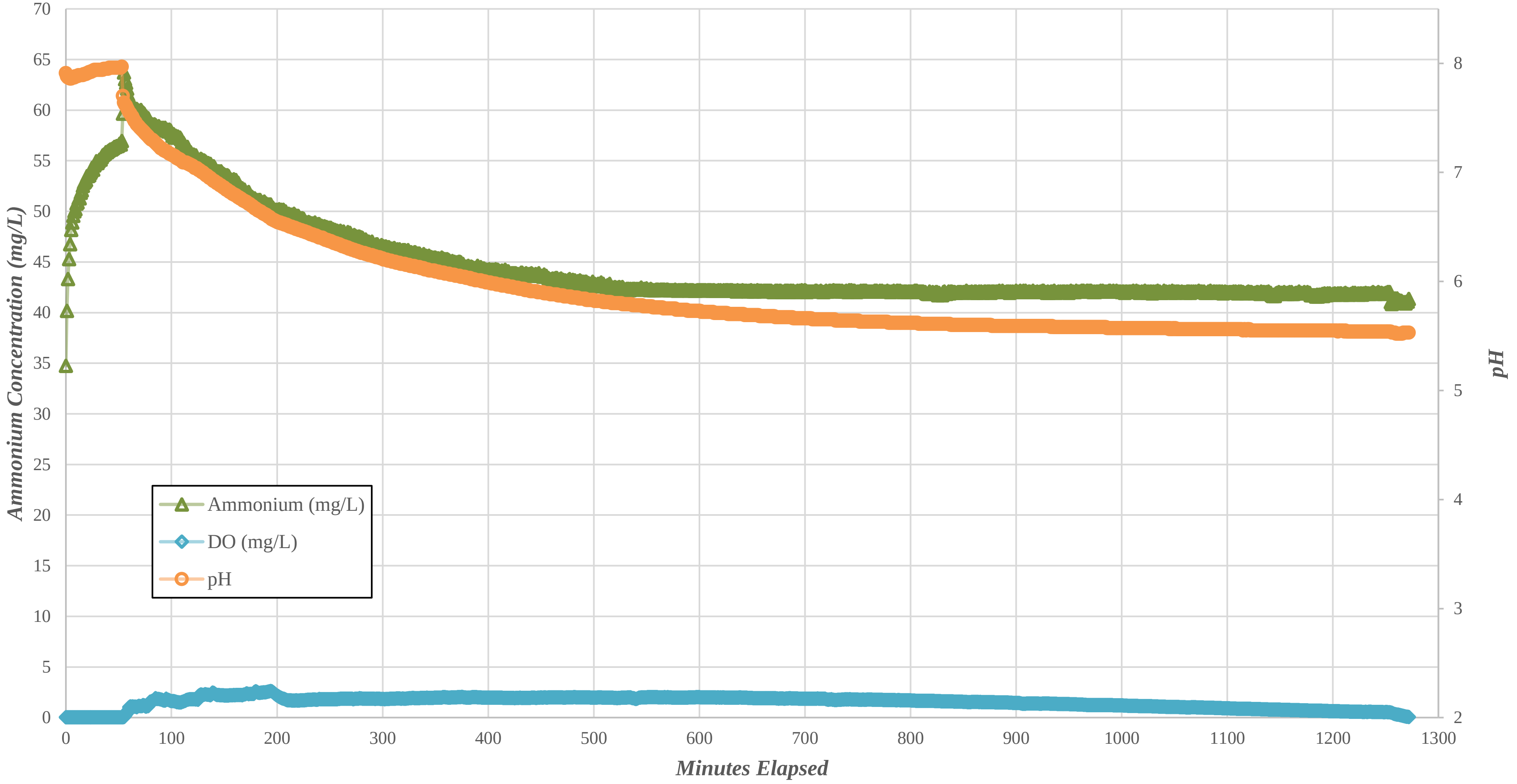

Figure 4-11. Ammonium concentration and pH measurements over SBR 1 aeration phase from October 26 to 27

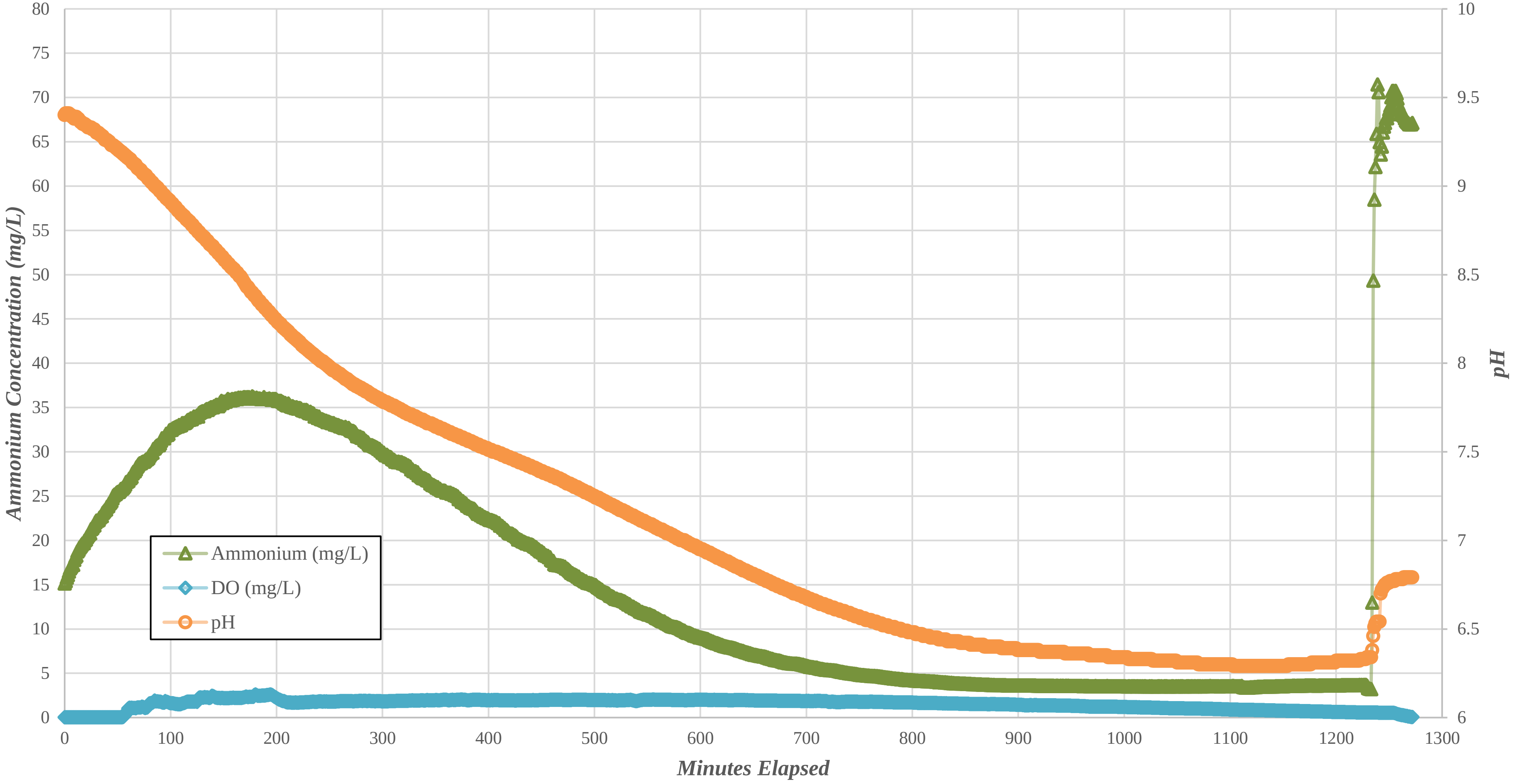

Figure 4-12. Ammonium concentration and pH measurements over SBR 1 aeration phase from October 30 to 31

4.4.2 Nitrification with extended aeration and pH adjustment

Ammonium concentration monitoring is done with extended aeration until inhibition of ammonium removal even with pH adjustment is detected for three runs. Figure 4-13, Figure 4-14, and Figure 4-15 present the results from extended aeration until ammonium removal ceased from January 4 to 8, January 8 to 11, and January 15 to 18, respectively. It should be noted that there are data gaps in the ammonium readings because the Vernier ISE would run out of battery after approximately 24 hours of operation, therefore these gaps are when the ISE receiver ran out of the battery. This was addressed by charging the receiver for 2 to 3 hours using an external power bank during these periods such that the probe is never removed from the reactor. Data logging while charging also gives very erratic readings from the probe, as such data logging was not done while the probe is charging. The pH is adjusted by dosing the reactor with 1 to 3 mL of 1M NaOH solution to get the pH within the range of 7.5 to 8 when it goes below the 6 to 6.5 range. This is the range determined from the readings from a typical operational cycle of the SBR, when ammonium removal stops even with aeration. The aeration during the runs is kept at 1 LPM to maintain a DO of around 2 to 4 mg/L.

There is an observed jump in ammonium reading from the ISE with the addition of 1M NaOH in the reactor, which is manifested by the jumps in pH in the graphs, as well as sudden jumps in value of the ammonia concentration around the same time. This is expected since the NaOH might contain ions that disrupt the probe upon initial dosing into the reactor. The graphs show that the ammonium readings return to stable readings a few minutes after NaOH dosing. It can be seen from both runs that ammonium removal was achieved even past 600 to 800 minutes with pH adjustment.

The run from January 4 to 8 shows that ammonium reduction from around 45 mg/L down to 4 mg/L with continuous aeration. The second trial from January 8 to 11 is measured from a starting concentration of about 95 mg/L down to 4 mg/L after more than 3600 minutes of aeration. The third run from January 11 to 15 reduced the ammonium concentration of about 55 mg/L down to 3 mg/L after about the same amount of aeration. These results show potential for the SBR to perform better at ammonium removal with an automated pH control that allows the mixed to operated at the optimum pH range for nitrification that seems to be around 6.5 to 8. Of the three runs, the second run shows the most continuous reading of ammonium concentration over the whole aeration period. This second run gives a good visualization of how ammonium continues to decrease when the pH is in the preferred range of 6.5 to 8. This run also shows the highest reduction in ammonium concentration of the three runs. All three runs seems to reduce the ammonium concentration to about 3 to 4 mg/L before ammonium removal seems inhibited not by the pH.

It is unlikely that aeration is limiting the ammonium removal at this point since aeration is still continued after 3600 minutes. It is more likely that organic matter or carbon sources may be limiting further ammonium removal past this point. This was considered in the third run with the addition of 1 mL of acetic acid after 3697 and 3747 minutes have elapsed. It should be noted that pH adjustment with NaOH was done with the addition of acetic acid since the pH naturally went down after dosing with acetic acid. However, after a further 250 minutes of aeration no ammonium removal is measured, while still maintaining pH between 6.50 and 8.00. Aeration was then turned off after approximately 4012 minutes of aeration, and the mixed liquor is allowed to settle. Interestingly, the anoxic condition from this run exhibited a significant degree of pH recovery going from 6.72 to 7.70. The pH increase observed to hit a steady state after about 1000 minutes or almost 17 hours without aeration. This is indicative of denitrification occurring, since this reaction involves the production of alkalinity (Section 2.3.2) that should cause the pH to increase. The acetic acid added prior to the settling phase is likely utilized as a carbon source for this reaction since denitrification is often limited by the amount of organic carbon present especially if it follows an aerobic phase. This indication of denitrifying bacteria can be tested further in future experiments in optimizing the SBR to do both nitrification and denitrification. It should be noted that the settling phase used for the typical cycle of the SBR during the experimental period did not exhibit pH recovery similar to what is observed here. Especially at higher ammonium influent loads, the settle phase of 4 to 6 hours (Table 3-2) did not gain pH recovery back into the preferred pH range of 6.5 to 8. The acetic acid dosed in this run is the main component that was not added previously in the experiment, which indicates that the additional carbon source might be needed to facilitate denitrification and subsequently pH recovery during the settle phase. Figure 4-16 shows the settling phase that achieved and increase in pH.

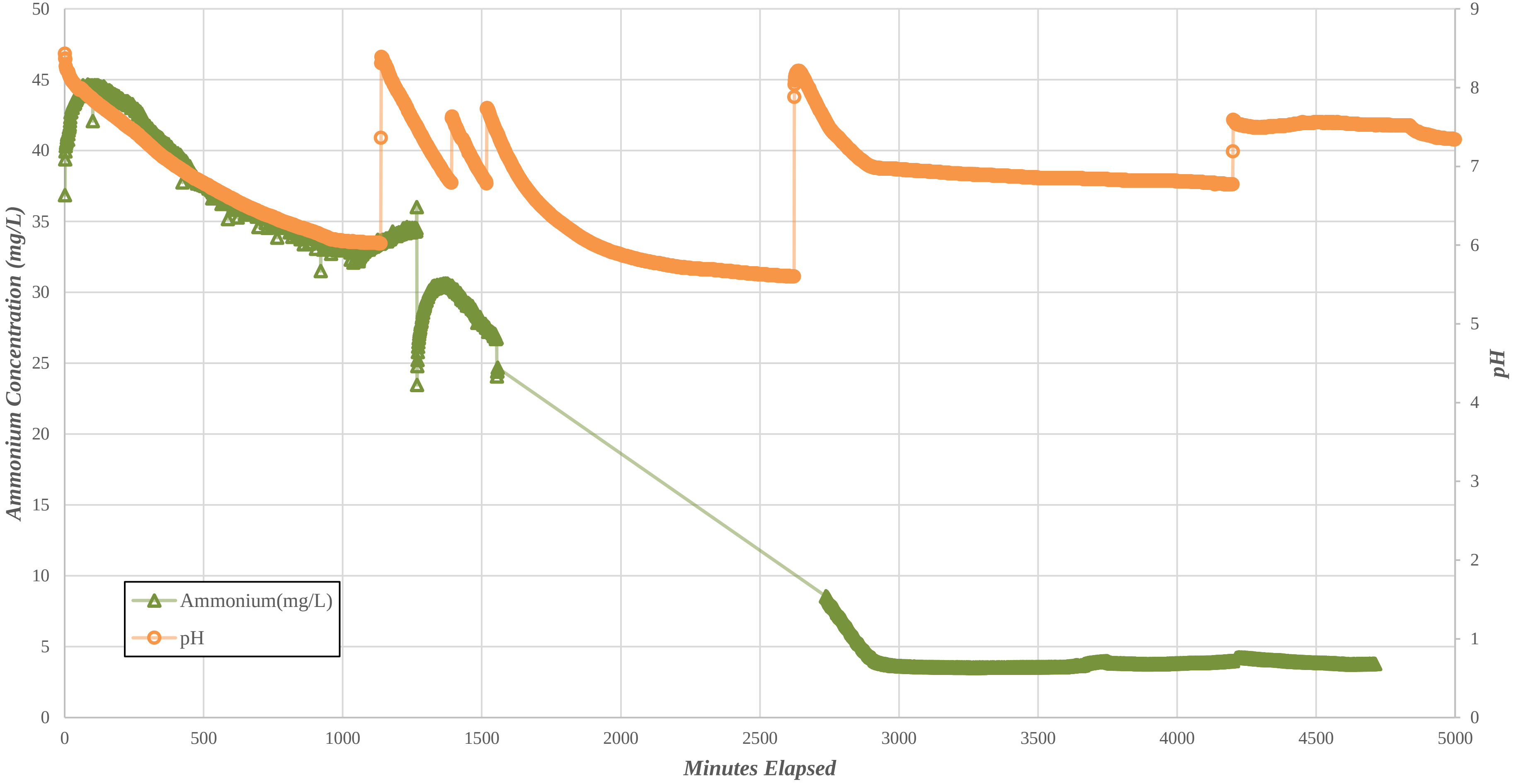

Figure 4-13. Ammonium concentration with continuous aeration and pH adjustment from January 4 to 8

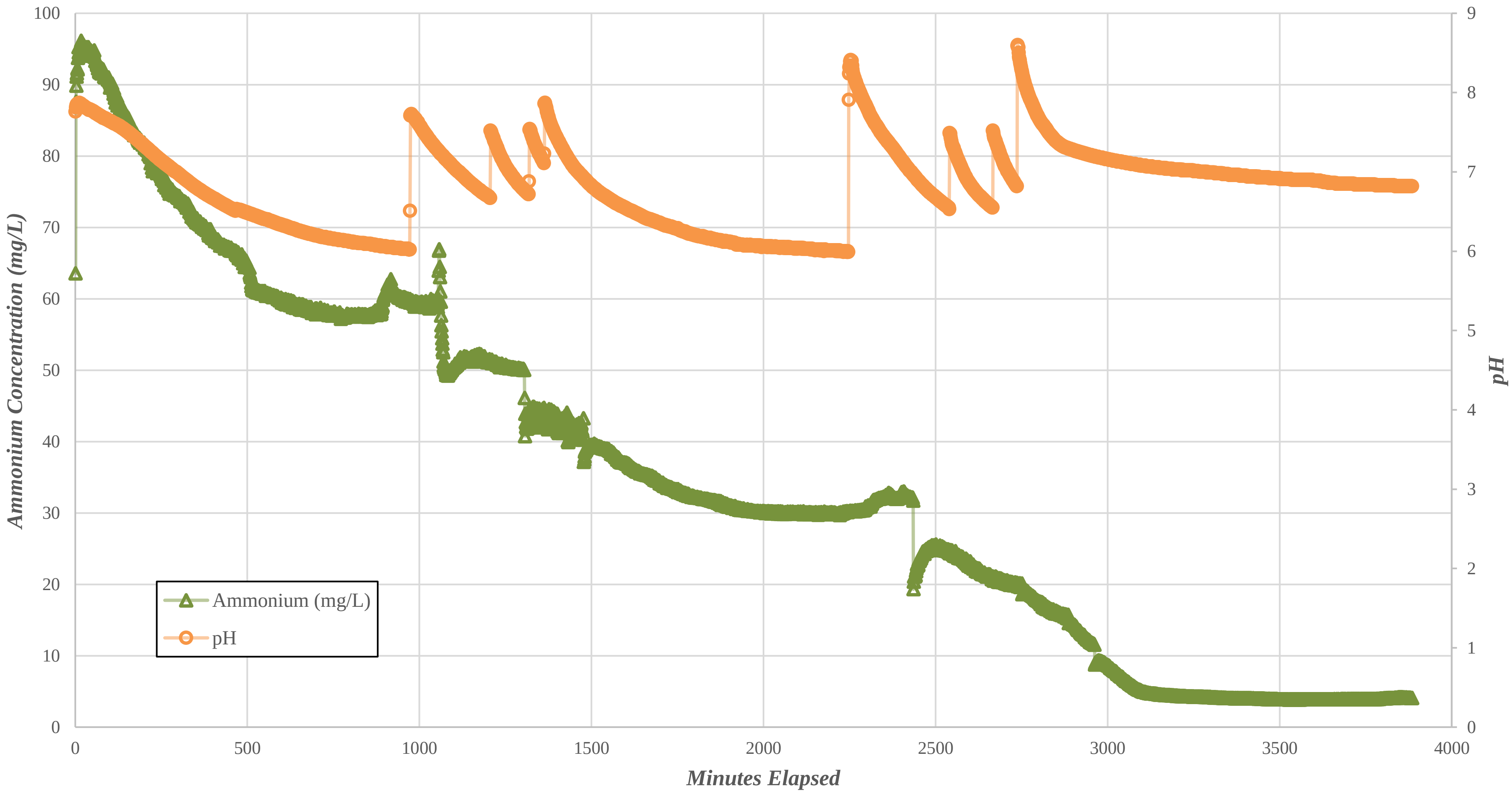

Figure 4-14. Ammonium concentration with continuous aeration and pH adjustment from January 8 to 11

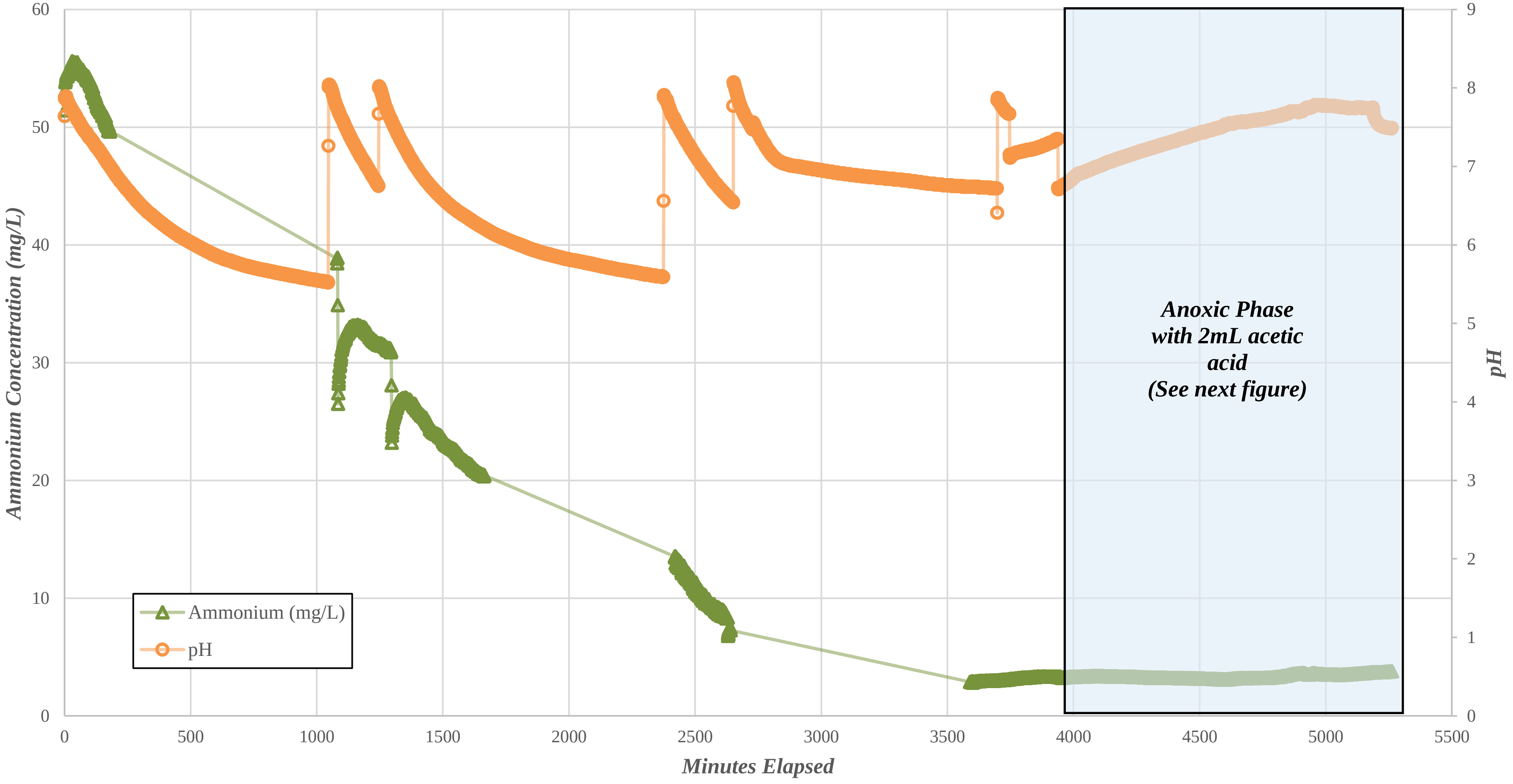

Figure 4-15. Ammonium concentration with continuous aeration and pH adjustment from January 15 to 18

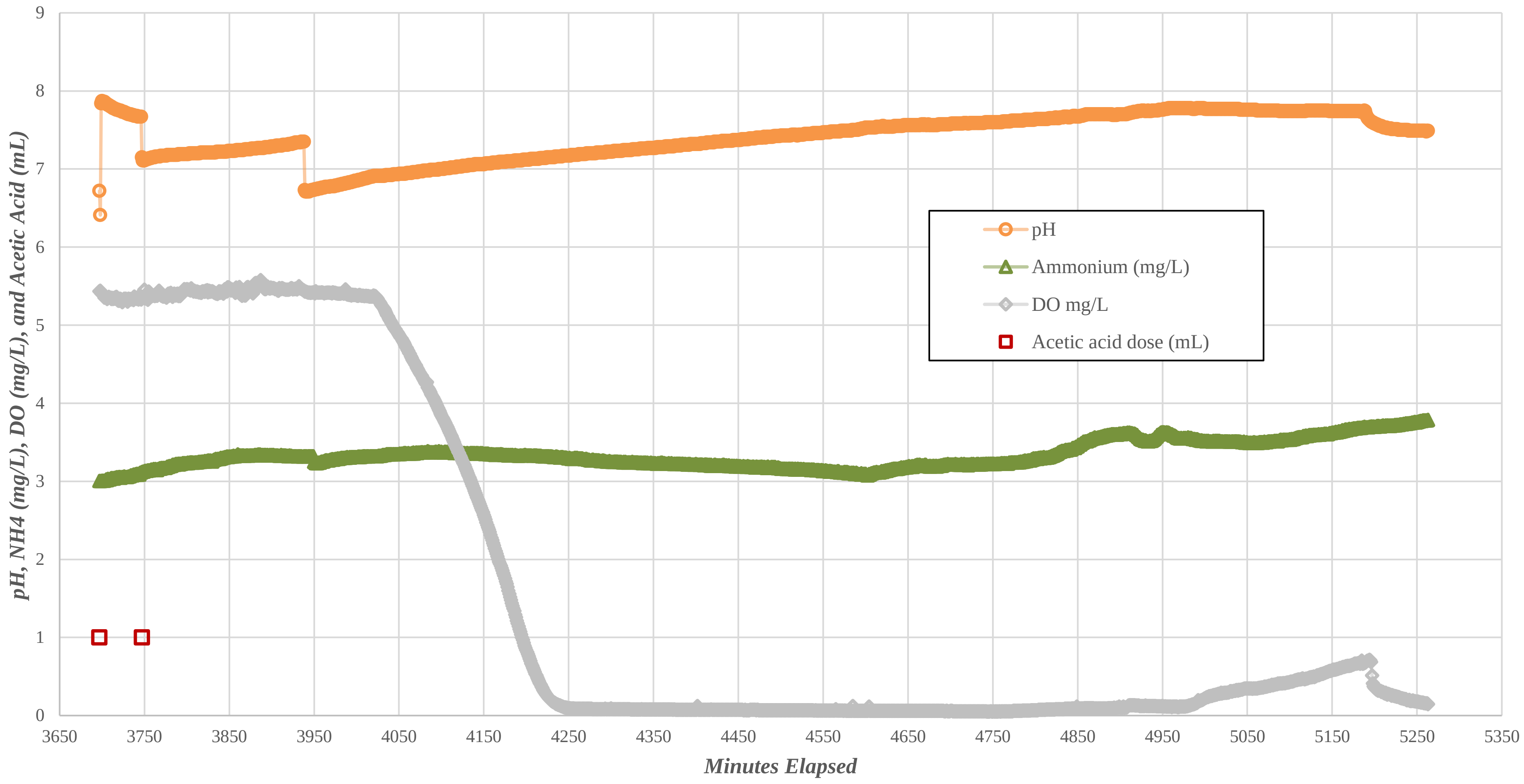

Figure 4-16. Extended anoxic phase after continuous aeration from January 15 to 18

5 Analysis and Discussion

This chapter discusses the analysis of the experimental test results specifically identifying operational parameters and ammonium removal rates to be used for a full-scale SBR. The full-scale SBR is then designed based on the parameters and removal rates considering leachate flow rate analysis based on latest annual precipitation data from weather stations in the Waikato region as a study area. The full-scale SBR design shall then be assessed for feasibility in application for a class 1 municipal landfill in the Waikato region based one actual reported treatment efficiency of full-scale SBRs from similar studies.

5.1 Analysis of Experimental Test Results and Parameters

The analysis of test results calculates for required parameters for the full-scale design that are significant to ammonium removal These include the ammonium removal rate per g of TSS, and ammonium removal rate per minutes. The rates of oxygen consumption are also significant in the full-scale design as aeration is a significant portion of the power consumption of an SBR. The oxygen consumption rate per minute, mass transfer coefficient, and fraction of oxygen transferred from gas to mixed liquor are calculated in the following subsections.

5.1.1 Ammonium removal rates

The experiment mainly focused on the ammonium removal of the SBR. With the data from the experiment, removal rates and efficiency shall be calculated based on periods when the SBR seemed to be most stable at a given ammonium loading rate. The ammonium loading rate per cycle is based on the influent ammonium concentration and volume of dose. The following equation describes the calculation for the ammonium loading rate.

$$Ammonium\ Loading\ Rate\ (mg\ NH_4^+/cycle) = Influent\ NH_4^+(mg/L) * Volume\ Dose\ (L/\ cycle)$$

Equation 5-1. Equation for calculating the ammonium loading rate at mg of ammonium per cycle

It should be noted that since the working volume of both SBR 1 and 2 are 1 L, the concentrations of ammonium in mg/L in the reactor can be considered to just be in mg. Using the ammonium loading rate, the amount of ammonium in the reactor and removed due to nitrification in the reactor can be calculated. Figure 5-1 and Figure 5-2 show the ammonium loading rate, ammonium in the reactor before a cycle, and ammonium removed after a cycle. All three of this being equal or close in value would pertain to the SBR operating most efficiently. This is such since an ammonium loading rate equal or near to the effective amount of ammonium in the reactor before a cycle means that the mixed liquor prior to filling is very low such that most of the SBR's ammonium concentration comes from the incoming influent and no accumulation of ammonium is occurring. The ammonium removed being close to these two other values pertains to high removal efficiency, which means that most of the ammonium being fed into the SBR before a cycle is removed by the end of the cycle. This ideal or efficient operation of the SBR is reflected in Figure 5-1 and Figure 5-2 for SBR 1 and SBR 2, respectively. For SBR 1, this ideal period is found to be around 110 to 164 days of operation. For SBR 2, this is around 44 to 84 days of operation. These periods are characterized by an ammonia loading rate from 11 mg to 34 mg, with nearly the same amount of ammonium removed after a cycle.

Important parameters to take from these periods are the approximate TSS and O2 consumption. The TSS in the reactor for SBR 1 is measured at 18.39 g TSS/L during this period. For SBR 2, the TSS is at 17.12 g TSS/L. The two TSS concentrations are within about a gram of each other. The SRT during this period for SBR 1 and 2 are 12 and 9 days respectively. Knowing the amount of TSS in the reactor during this period, an approximation of the ammonium removal rate based on biomass in the reactor can be calculated. The following equation shows the calculation for this rate.

$$Ammonium\ removed\ per\ gram\ of\ TSS\ (mg\ NH_4^+/g\ TSS) = \frac{Ammonium\ removed\ (mg)}{TSS\ concentration\ (g/L)}$$

Equation 5-2.   Ammonium removal rate based on biomass

This calculation simplifies the biomass to the amount of sludge in g TSS/L at a certain amount of ammonium in mg removed. For SBR 1, using the peak amount of ammonium removed at 30.40 mg, the rate is calculated to be at 1.65 mg NH_4^+/g TSS. For SBR 2, this is calculated at 1.55 mg NH_4^+/g TSS with the highest amount of ammonium removed measured at 26.46 mg for the ideal period. The amount of ammonium removed in mg during the ideal period for the two SBRs is near the 40 mg of ammonia removed by Gao et al. (2020). However, the 40 mg/L removed by Gao et al. (2020) was the peak. Compared to this study, the peak amount of ammonium removed is 48.40 mg for SBR 1 and 38.10 mg for SBR 2, which are nearer to the findings from the study.

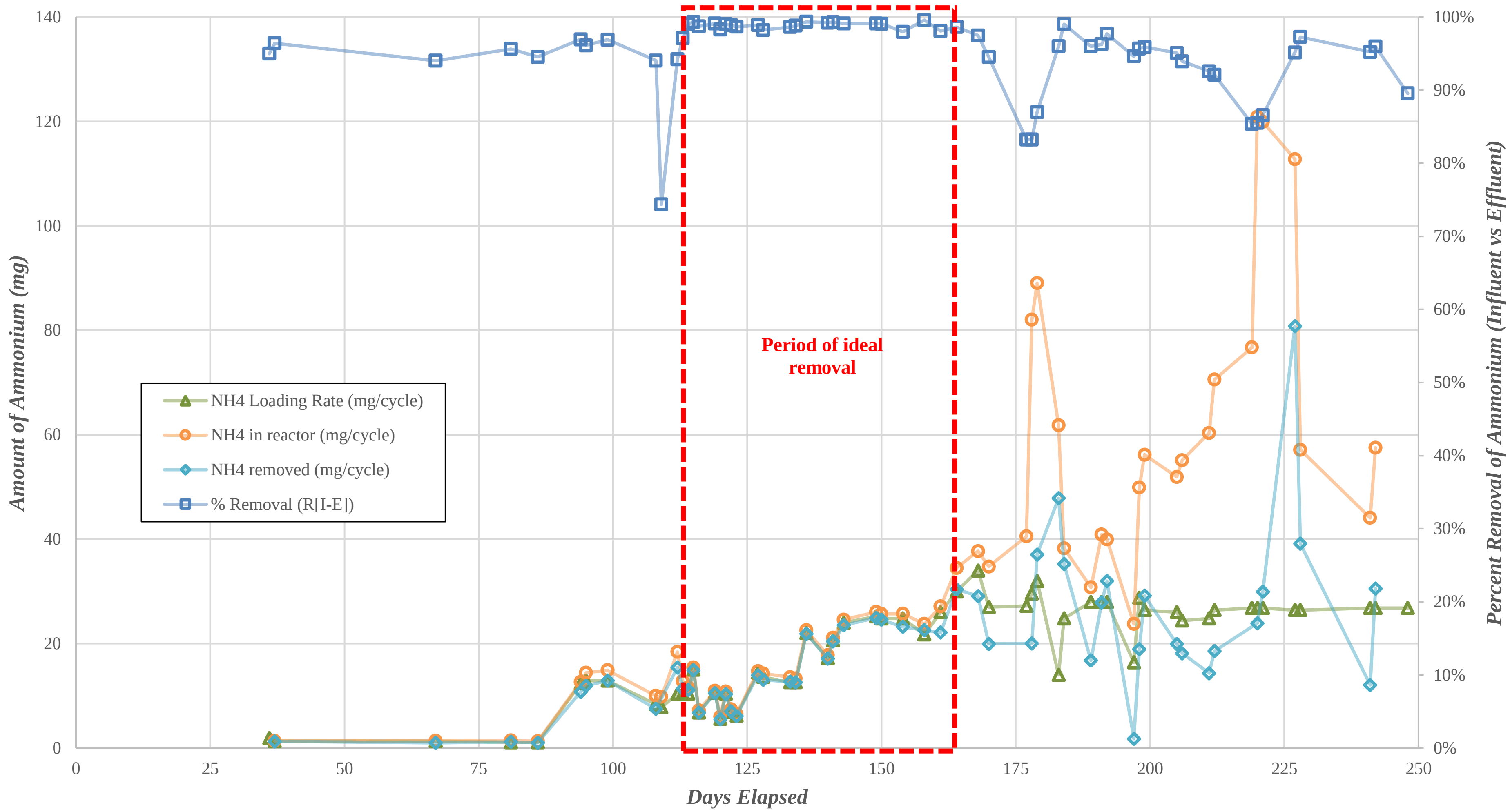

Figure 5-1. Amount of ammonium removed in milligrams for SBR 1

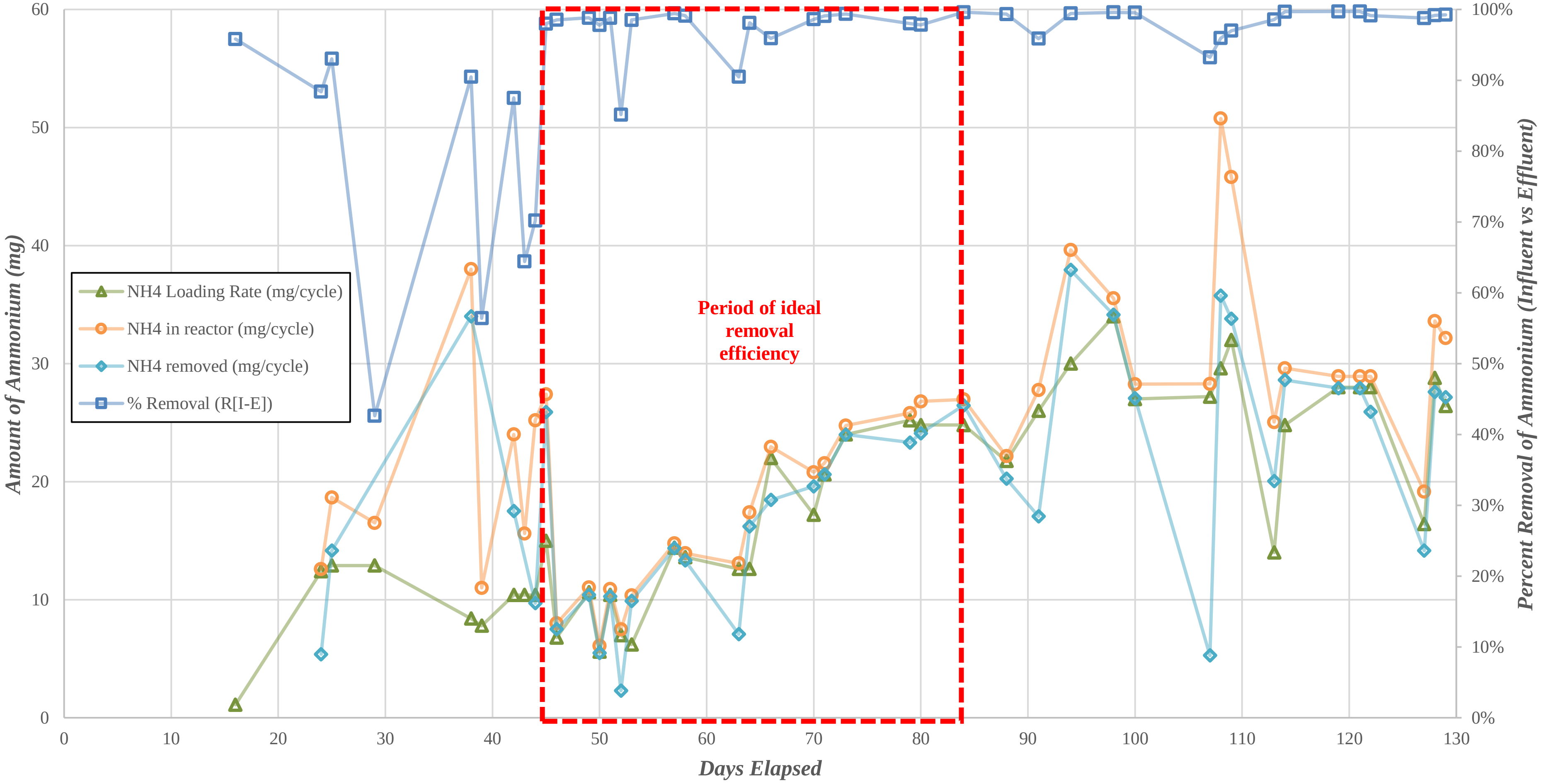

Figure 5-2. *Amount of ammonium removed in milligrams for SBR 2*

Based on the results of the Vernier Go ammonium ISE (Section 4.4), ammonium removal tends to be inhibited once the pH is below 6.5. Whenever the pH was below 6.5, the ammonium removal based on the graphs we decreasing slowly compared to the region of pH 6.5 to 8. The 6.5 pH threshold aligns with Tarre and Green (2004) premise that growth of nitrifying bacteria general does not occur below this threshold as well. With an ideal range of pH to maintain, the ammonium removal is maximized with pH control. Automatic pH control is not achievable with the experimental set-up for this study, the data from the ISE of ammonium concentration per minute can be used to approximate an ammonium removal rate in terms of time. Since the removal rate is found to be best in the pH range of 6.5 to 8, the ammonium removal rate is to be taken from the ammonium concentrations in this pH range from the runs. A linear trendline is fit to the ammonium concentration from the pH at peak concentration up to when pH reaches 6.5. The linear fit is chosen since this portion of ammonium reduction based on the ISE results shows a very steep and almost linear trend in decreasing ammonium. The linear fit also simplifies the ammonium removal rate to the slope of the linear fit in units of mg NH_4^+ removed/L/min. The R^2 value of the trendline is considered in selecting removal rates to be averaged. An average of 0.0508 mg NH_4^+ removed/L/min is the calculated ammonium removal rate. The following table and figures summarize the linear fit equations, ammonium removal rates, and R^2 values from the ammonium ISE runs.

Table 5-1. Ammonium removal rate based on linear fit on ammonium ISE runs

Run Date	Linear Fit Equation	Ammonium Removal Rate (mg NH_4^+ removed/L/min)	R^2
October 16 to 17	$y = -0.0419x + 31.148$	0.0419	0.9707
October 24 to 25	$y = -0.1104x + 56.858$	0.1104	0.9444
October 26 to 27	$y = -0.0771x + 61.132$	0.0771	0.9853
October 30 to 31	$y = -0.0587x + 32.872$	0.0587	0.9673
January 4 to 8	$y = -0.0184x + 44.985$	0.0184	0.9885
January 8 to 11	$y = -0.0674x + 93.149$	0.0674	0.9776
	$y = -0.0470x + 59.791$	0.0470	0.6466
	$y = -0.0285x + 43.675$	0.0285	0.879
	$y = -0.0530x + 33.981$	0.0530	0.6217
	$y = -0.0252x + 24.392$	0.0252	0.9523
	$y = -0.0441x + 20.723$	0.0441	0.9828
January 15 to 18	$y = -0.0399x + 56.299$	0.0399	0.9556
	$y = -0.0261x + 29.497$	0.0261	0.728
	$y = -0.0251x + 13.14$	0.0251	0.9625
Average Removal Rate with R^2 greater than 0.90		***0.0508***	

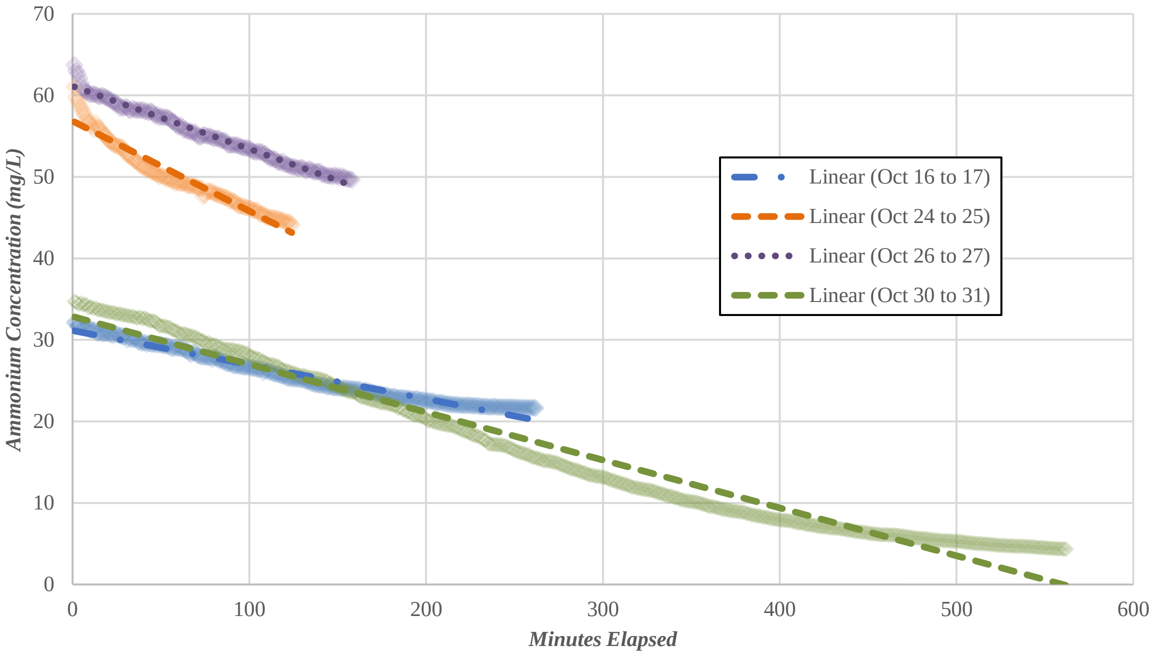

Figure 5-3. Linear trendlines for runs from Oct 16 to 17, Oct 24 to 25, Oct 26 to 27, and Oct 30 to 31

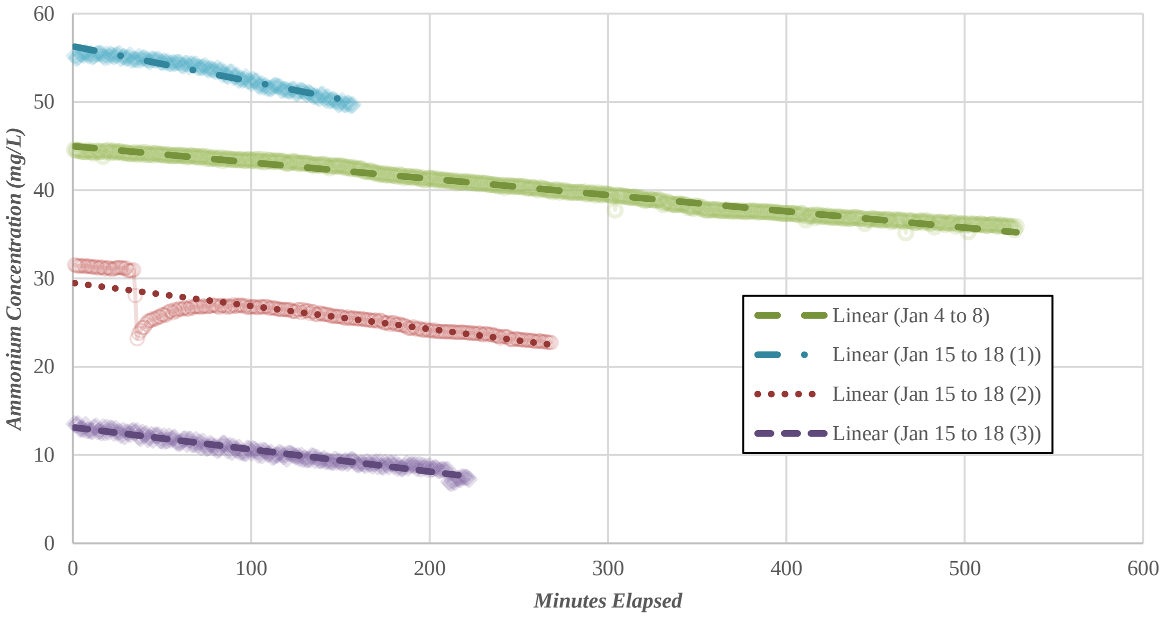

Figure 5-4. Linear trendlines for extended aeration runs from Jan 4 to 8 and Jan 15 to 18

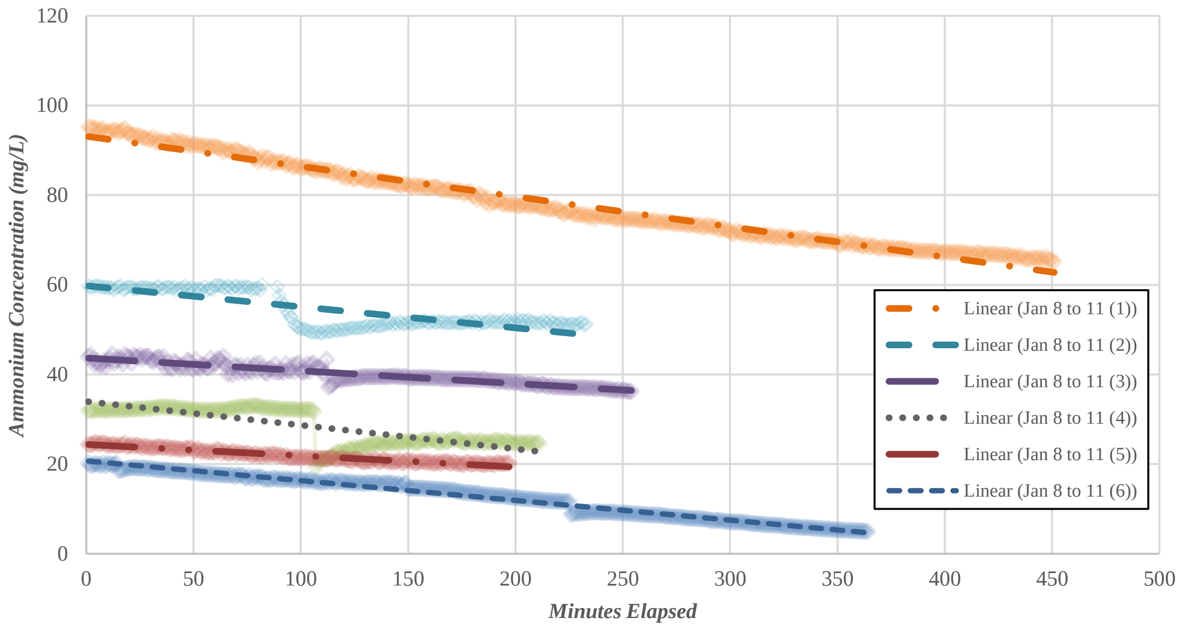

Figure 5-5. Linear trendlines for the extended aeration run from Jan 8 to 11

Based on Table 5-1, the R^2 values for most linear equations are greater than 0.90, which indicates high correlation between the time, subsequently time of aeration, and ammonium concentration. This correlation is analysed as ammonium removal over time, and as such the slope becomes the ammonium removal rate in milligrams of ammonium removed per litre per minute. The R^2 values less than 0.90 are from data logged midway through the extended aeration runs. This is most likely due to the jump in values that is caused by the dosing of 1M NaOH to allow the pH to go up between 7 to 8 (Figure 5-4 and Figure 5-5). As such, the average value calculated is based on the removal rates with an R^2 greater than 0.90.

5.1.2 Oxygen consumption rate

The oxygen consumption of the mixed liquor in the SBRs are calculated based on the dissolved oxygen (DO) reading from the probe of SBR 1 due to the DO probe in SBR 2 not functioning. The calculation simply takes the DO reading at the point when aeration is stopped ($DO_{Initial}$) and subtracts it by the lowest or nearest zero DO reading after (DO_{final}), and divide that by the number of minutes it takes to get to the lowest DO reading at the first instance during the settle phase. Since the DO readings from the controller are in percent, the probe is calibrated to be at 100% at 4 mg/L, which will be referred to as the DO saturation point. The DO readings are multiplied to the DO saturation point to get an approximation of the oxygen consumption in mg DO/L. The consumption rate is in the units of mg DO/L/minute. The following equation shows the formula used for the calculation.

$$DO\ consumption\ rate\ (mg\ DO/L/minute) = \frac{(DO_{Initial} - DO_{Final}) * (DO_{Saturation})}{Minutes\ Elapsed\ to\ DO_{Final}}$$

Equation 5-3. Equation for calculating DO consumption rate

The DO consumption rate to be considered for the full-scale design would be the rate calculated during the period of ideal removal efficiency of the SBR 1, which from 110 to 164 days of operation (Figure 5-1). Based on the available DO data from this period of operation, the DO would reach near zero from 20 to 95 minutes. The $DO_{Initial}$ from the period was in the range of 54% to 104.5%, and for DO_{Final} from 0.6% to 3.8%, but mostly at 0.9%. This indicates that the aeration rate of 1 LPM at this period was keeping the SBR 1 in the DO range from 2 to 4 mg/L, similar to SBR studies from the review by Aziz et al. (2013) . Calculating the consumption rate during this period using Equation 5-3, the average is found to be at 0.007 mg DO/L/minute. This is rounded up to 0.01 mg DO/L/minute since the maximum rate calculated for the period was around 0.011 mg DO/L/minute. It should be noted that not all days from the range of 110 to 164 days was calculated from since a significant number of days showed scouring of the DO probe where DO would be less than 1% even during aeration. These were excluded from the calculation as it would lower the average consumption rate for the period. This can be compared to the DO consumption rate that can be calculated from Figure 4-16, where the end of the aeration phase shows the decrease in DO reading that looks linear. The consumption rate from this is calculated at 0.002 mg DO/L/minute with a $DO_{Initial}$ of 117.5%, DO_{Final} of 7.7%, and duration of consumption at 217 minutes at a $DO_{Saturation}$ of 4 mg/L. This consumption rate is well within the 0.01 mg DO/L/minute to be considered for oxygen requirements to achieve the ammonium removal found from the ideal period.

The mass transfer coefficient of oxygen in the mixed liquor of SBR 1 is also calculated based on the equation from Davis and Cornwell (2013). This coefficient quantifies the rate in which gas dissolves in the liquid, in this case oxygen in the mixed liquor. The equation for calculating the mass transfer coefficient (K_{La}) relates the rate of mass transfer for the difference between the concentration of gas in a liquid (C_t) and the saturation concentration of the gas in the same liquid (C_s). The saturation concentration in the SBR to be around 2 to 4 mg/L. The following equation shows the calculation for this relationship.

$$\frac{dC}{dt} = K_{La}(C_S - C)$$

Equation 5-4. *Equation relating the mass transfer coefficient to gas concentration (C) at a certain time (t) (Davis & Cornwell, 2013)*

Integrating Equation 5-4, and rearranging forms the following equation.

$$\frac{C_S - C_t}{C_S - C_0} = e^{-K_{La}t}$$

Equation 5-5. *Form of equation after integrating and rearranging Equation 5-4 (Davis & Cornwell, 2013)*

In the form of Equation 5-5, plotting the graph for ln(C_s-C_t) over the period of aeration until the liquid reaches saturation and fitting a linear trendline that has linear function whose slope is the -K_{La}. This calculation is applied to the aeration period of cycles last October 24 to 25, October 26 to 27, and October 30 to 31 (Figure 4-10, Figure 4-11 & Figure 4-12). The plot of the ln(C_s-C_t) vs time from these runs are shown in the figures below.

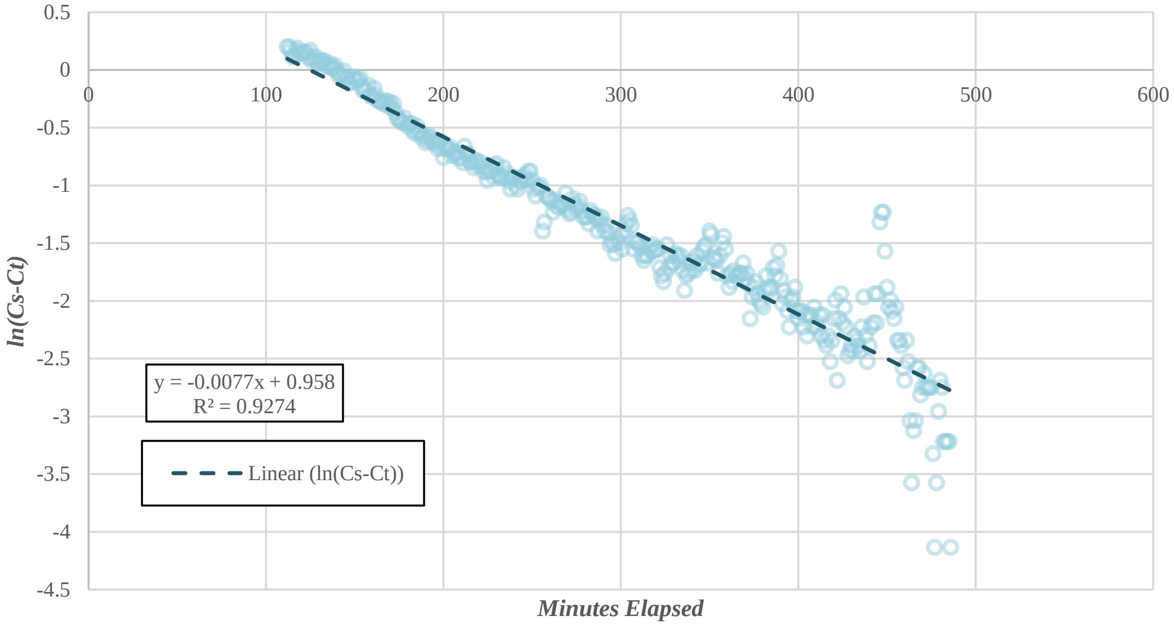

Figure 5-6. *Plot of ln(Cs-Ct) for cycle October 24 to 25 with K_{La}=0.0077 min^{-1}*

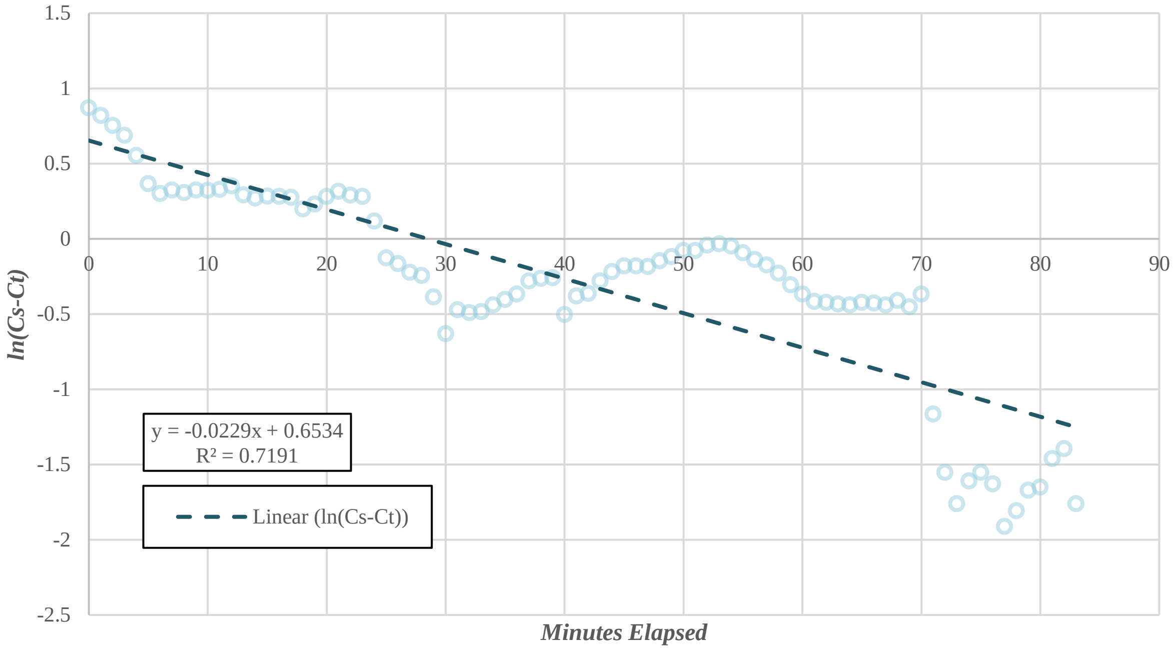

Figure 5-7. *Plot of ln(Cs-Ct) for cycle October 26 to 27 with K_{La}=0.0229 min^{-1}*

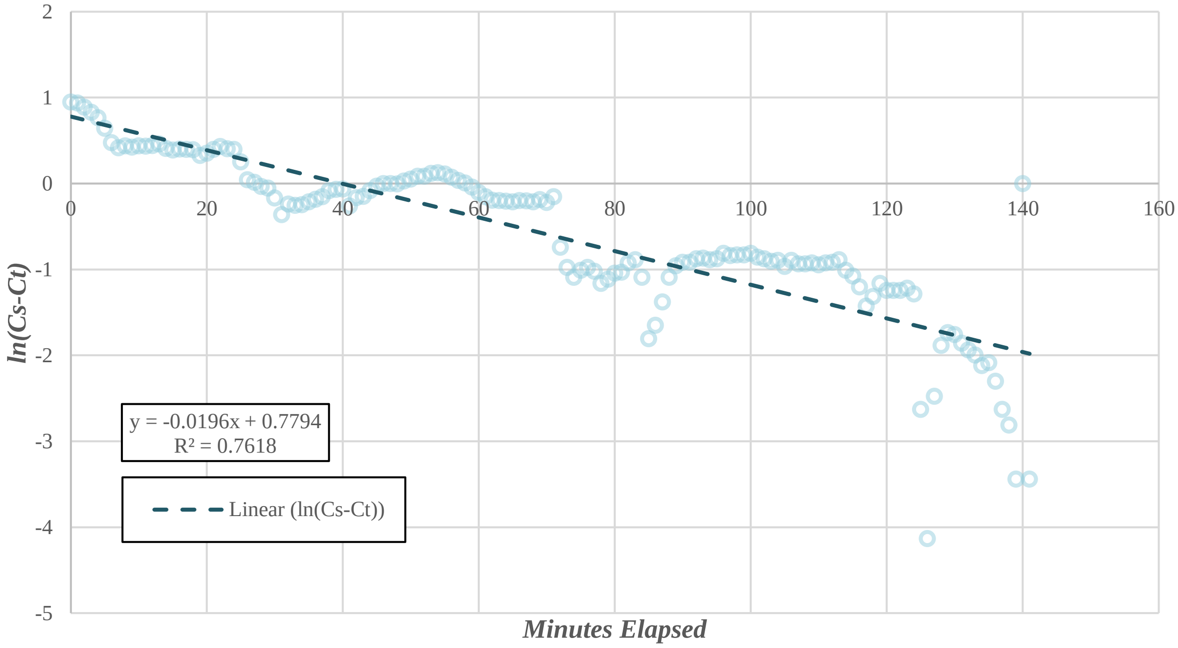

Figure 5-8. *Plot of ln(Cs-Ct) for cycle October 30 to 31 with K_{La}=0.0196 min^{-1}*

The following table summarizes the K_{La} values from the three cycles. An average of 0.0144 min^{-1} for the mass transfer coefficient is calculated and is selected to be used for calculations in the full-scale SBR design.

Table 5-2. Summary of K_{La} values from cycles from October 24 to 25, 26 to 27, and 30 to 31

Cycle Date	K_{La}
Oct 24 to 25	0.0008
Oct 26 to 27	0.0229
Oct 30 to 31	0.0196
K_{La} average	0.0144

The K_{La} is near the calculated O_2 consumption rate of 0.01 mg DO/L/minute, which indicates that DO or oxygen transferred into the mixed liquor is mostly consumed by the biomass. This is ideal for aeration of SBRs because it ensures that the aeration pump is running at an optimal rate that is not less than what the biomass can consume nor does it pump excessive amount of air into the reactor, which creates high operational costs.

Al-Ahmady (2006) provides an equation to calculate the fraction of oxygen transferred from the air into the mixed liquor (δ) using the mass transfer coefficient. This fraction or percentage is useful in the design of a full-scale SBR to identify the required amount of air to be pumped into the system. The following equation shows the calculation for δ from Al-Ahmady (2006).

$$\delta = \frac{K_{La} * (C_{st}) * H}{I * j}$$

Equation 5-6. Equation for calculating the fraction of oxygen transferred from air to water (Al-Ahmady, 2006)

The calculation for δ is a function of the mass transfer coefficient (K_{La}), reactor depth (H), saturation concentration (C_{st}), volumetric air flux per unit area of tank surface (I), and constant for the amount of oxygen in gram per cubic meter of air (j) (Al-Ahmady, 2006). This equation is used to calculate the fraction of oxygen transferred to water based on the experimental set-up parameters.

The volumetric air flux per unit area of tank surface is in units of m/min, and is based on the gas gauge setting of 1 LPM used in the experimental set-up. The calculation for I involves dividing the aeration rate by the area of the tank surface. The following table summarizes the calculation for I using SBR 1 dimensions and parameters.

Table 5-3. Calculation of volumetric air flux per unit area of tank surface based on SBR 1

Parameter	Unit	Value
SBR 1 diameter	mm	100
SBR 1 height	mm	195
SBR 1 area of tank surface	m^2	0.00785
Aeration rate	LPM	1
	m^3/min	0.001
Volumetric air flux per unit area of tank surface (I)	m/min	0.127

The constant *j* is at a value 298 at air temperature 20°C at 1 atm. Since the SBR has a DO saturation kept within 2 to 4 mg/L, the two limits are used as two saturation concentrations (C_t) shall be used to calculate a δ. The following table shows the reactor parameters and the calculated δ.

Table 5-4. Calculation of fraction of oxygen transferred from gas to water

Parameter	Unit	Values	
Saturation concentration (C_{st})	mg/L	2	4
Mass transfer coefficient (K_{La})	min^{-1}	0.0144	0.0144
Air flux per unit area of tank surface (I)	m/min	0.127	0.127
Amount of oxygen in air (j)	g/m^3	298	298
Reactor depth (H)	m	0.195	0.195
δ	%	14.8	29.6

The fraction of oxygen transfer at a saturation concentration of 2 mg/L is calculated to be more conservative at 14.8% compared to 29.6% at a saturation of 4 mg/L. As such, the fraction of oxygen transfer to be used for the full-scale design is 14.8%.

The following table (Table 5-5) summarizes the findings from further analysis of the experimental results from the two lab-scale SBRs. These findings are to be used in designing a full-scale SBR to treat influent ammonium concentrations characteristic of landfill leachate.

Table 5-5. Ammonium removal rates based on experimental set-up

Parameters	SBR 1	SBR 2
Ammonium Removal Efficiency (I-E)	90-100%	90-100%
Ammonium Removal per gram of TSS per cycle	1.65 mg NH_4^+ removed/g TSS	1.55 mg NH_4^+ removed/g TSS
Rate of ammonium removal	0.0508 mg NH_4^+/L/minute	-
O2 consumption (mg DO/L/minute)	0.01 mg DO/L/minute	DO Probe non-functioning
Mass transfer coefficient (K_{La})	0.0144 min^{-1}	-
Percent oxygen transferred from air to water (δ)	14.8%	-

5.2 Landfill Leachate Parameters

Since there is difficulty encountered in acquiring landfill leachate samples for the experiment, the substitute influent used is cow manure diluted in distilled water with a certain amount of NH_4Cl (Section 3.3). However, through literature review, it is found that landfill leachate's characteristic high ammonium concentration can be approximated based on length of operation (Section 2.1.3).

For this study, literature is used as reference for landfill leachate concentrations for a full-scale SBR design. However, flow rate is usually a function of the site location. The leachate flow rate is critical in the full-scale SBR design as this will dictate the appropriate sizing of the tanks and equipment requirements to operate an SBR. As previously mentioned, an approximation of landfill leachate flow can be done based on annual precipitation (Ehrig, 1983; Serdarevic, 2018). For the case of this study a class 1 municipal sanitary landfill is to be designed for. This assumes that the landfill is compacted regularly with soil cover, and is categorized to be highly compacted. Estimated leachate flow is based on 20% of the annual precipitation, which is based on the average percentage of annual precipitation for highly compacted landfills (Ehrig, 1983; Serdarevic, 2018). The following equation shows how the estimation of daily leachate flow based on annual precipitation is calculated.

$$Q\ in\ m^3/hec/day = \left(\frac{Precipitation\ mm/a}{365\ days}\right) * \left(\frac{1\ m}{1000\ mm}\right) * \left(\frac{10{,}000\ m^2}{hec}\right) * 20\%$$

Equation 5-7. Equation for approximate leachate flow rate estimation for highly compacted landfills

Data from the National Climate Database from the National Institute of Water and Atmospheric Research (NIWA) is used to get the annual rainfall from 2023. Weather stations within the Waikato region that have precipitation data from 2023 were selected. The monthly average rainfall is summed up to get the annual rainfall and divided by 365 days to get the average daily rainfall from each weather station for the year of 2023. Since the available data measures precipitation in millimetres, the average daily rainfall is converted to meters and multiplied by 10,000 m^2 to get daily average rainfall per hectare. This is then multiplied by 20%, as mentioned earlier, to estimate the landfill leachate flow due to the precipitation. This data is plotted on QGIS, and the built-in inverse distance weighted (IDW) interpolation function is applied to the landfill leachate flow from the weather station locations. The following heat map is generated from the interpolation to show the approximate daily landfill leachate flow per hectare landfilled per day in the Waikato region for 2023 (Figure 5-9).

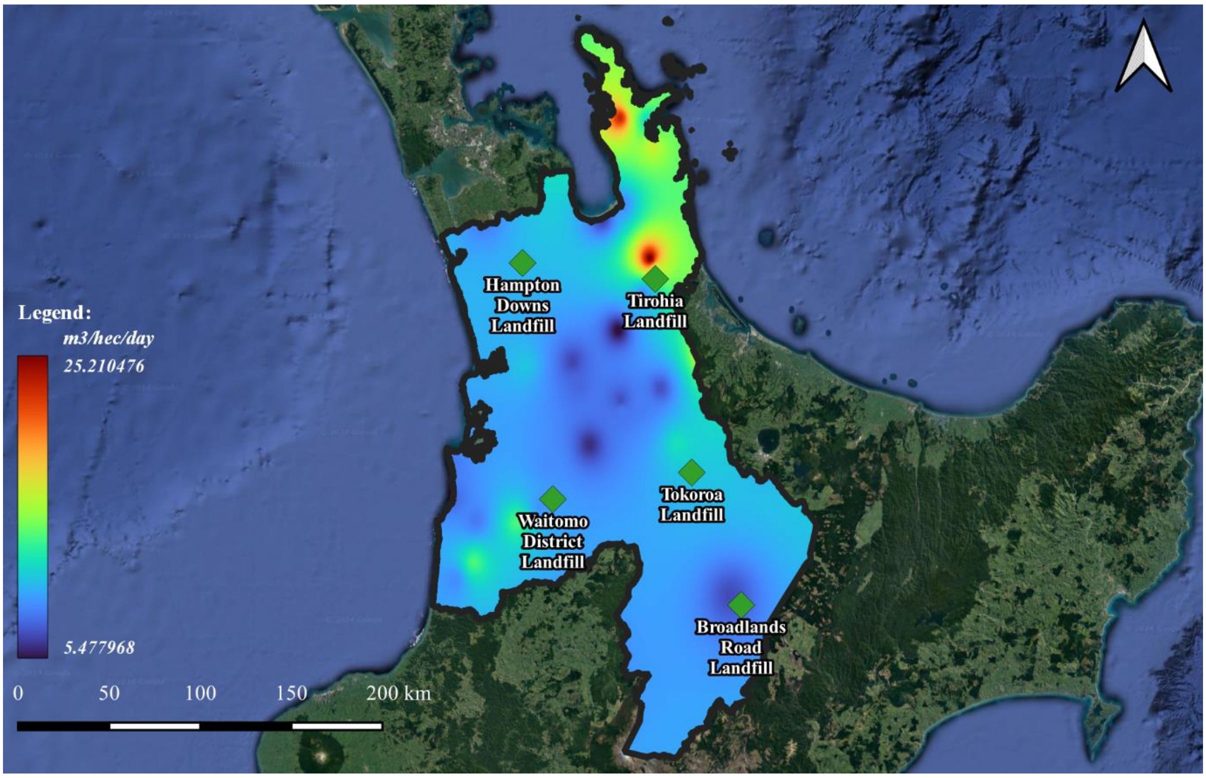

Figure 5-9. Estimated landfill leachate flow based on annual precipitation in 2023 in Waikato (Google, 2024b)

Also shown in Figure 5-9 are the open and consented class 1 sanitary landfills operating in the Waikato region. These landfills are shown to be in areas with a landfill leachate flow around 15 m^3/hec/day at most, which is the case of the Tirohia landfill, and as low as around 6 m^3/hec/day, such as for Broadlands Road Landfill. As such, this is the estimated landfill leachate flow to be used in the design for a full-scale SBR in this study.

According to Ridgeway (2016), in 2016 the North Waikato Regional Landfill (Hampton Downs Landfill) has a landfill leachate discharge rate of 150 to 300 m^3/day. At the time, the leachate is shipped off to the Mangere Wastewater Treatment Plant in Auckland at a cost of NZD 22.00/m^3. Equation 5-7 shall be verified based on the actual landfill discharge rate reported from Ridgeway (2016). Based on a total landfilling area of 87 ha for the Hampton Downs Landfill and a depth of filling of 70 m (Ridgeway, 2016), at total volume of 60.9M m^3 of waste can be fit into the landfill. According to the Waikato Regional Council, the landfill receives 600,000 tonnes of waste per year (*Waste in the Waikato Region*, 2022). Waste in a class 1 sanitary landfill is compacted and, based on the Ministry for the Environment (2021) fact sheet on calculating waste tonnages, compacted waste have a density of 320 kg/m^3. The landfill started operation in 2005 and in 2016 has been operating for 11 years. The percentage of the landfill that is covered is approximated at 34% or 29 ha of the available 87 ha in 2016. Using data from NIWA, the nearest weather station with data from 2016 recorded an annual precipitation of 1303.3 mm. Using Equation 5-7, the approximate leachate flow based on this annual precipitation is 7.14 m^3/hec/day. This daily leachate flow when multiplied by the 29 ha for area landfilled comes to a leachate flow rate of 207.1 m^3/day, which is within the leachate flow rate range of 150 to 300 m^3/day from Ridgeway (2016). This shows that the leachate flow rate estimation used in this study can be a suitable estimate in the absence of available data.

For the case of this study, the estimated percentage of the Hampton Downs Landfill filled by 2023 in hectares shall be used as a reasonable estimate of the size of a large modern-day landfill that receives a large amount of waste regularly. Using the same calculation, 55% or 48 ha of the landfill would be filled by 2023. The following table shows the calculations for verifying the landfill leachate flow estimate in 2016 against Equation 5-7 and the area filled in 2023.

Table 5-6. Estimation of landfill leachate flow of Hampton Downs landfill in 2016 and 2023

Parameter	Units	Value
Total filling area	ha	87
	m^2	870000
Depth of filling	m	70
Total Volume	m^3	60900000
Waste per year (*Waste in the Waikato Region*, 2022)	tonnes	600000
	kg	600000000
Density of municipal waste (Ministry for the Environment, 2021)	kg/m^3	320
Start Year	2005	
Year for estimation	***2016***	
Volume of waste disposed	m^3	20625000
Percentage of landfill used	34%	
Hectares filled	ha	29
Annual Precipitation in 2016	mm	1303.3
Estimated landfill leachate flow using Equation 5-7	m^3/hec/day	7.14
	m^3/day	207.1
Year for estimation	***2023***	
Percentage of landfill used	55%	
Hectares filled	ha	48
Minimum daily landfill leachate flow (Figure 5-9)	m^3/hec/day	6
	m^3/day	288
Maximum daily landfill leachate flow (Figure 5-9)	m^3/hec/day	15
	m^3/day	720

For ammonium concentration in the landfill leachate, literature shows that ammoniacal-nitrogen concentrations tend to increase with the age of a landfill and stabilizes as the landfill reaches a mature state (Section 2.1.3). As such, a mature landfill leachate ammonium concentration should provide a suitable estimate of the ammonium concentration a landfill SBR should be treating for most of its operational period. Robinson (2007) cites that a predicted landfill in the Waikato region would have an ammoniacal nitrogen concentration range of 600 to 1000 mg/L, while Mohobane (2008) found that the Horotiu landfill leachate, which is in the Waikato region, has an ammoniacal-nitrogen concentration ranging from 227 to 1270 mg/L. Haslina et al. (2021) cites that mature landfill leachate has an average ammoniacal-nitrogen concentration of 2000 mg/L, while Tejera et al. (2019) notes a lower threshold from 400 mg/L and higher ammoniacal-nitrogen concentrations. Ridgeway (2016) measured the raw landfill leachate from the Hampton Downs Landfill to be 730 mg/L and 930 mg/L from two occasions. Based on literature, a minimum ammonium concentration of 400 mg/L and maximum of 1500 mg/L is considered for this study, and is well within the ranges mentioned in studies for mature landfills and landfills in the Waikato region. Along with literature, a low biodegradability is considered for landfill leachate, especially from mature landfills. With this, carbon supplementation is to be included as necessary for the full-scale SBR design in the form of acetate, which is a common form of carbon supplementation used in wastewater treatment (Al-Attabi et al., 2017; Iannacone et al., 2019; Junaidi et al., 2020). The following table summarizes the landfill leachate parameters for consideration in the full-scale SBR design to be discussed in the next subsection.

Table 5-7. Landfill leachate parameters for full-scale SBR design

Parameter	Unit	Value
Landfill area	hectares	48
Minimum landfill leachate flow	m^3/day	288
Maximum landfill leachate flow	m^3/day	720
Minimum ammonium concentration	mg/L	400
Maximum ammonium concentration	mg/L	1500
Carbon supplement	Acetic acid or Methanol	

5.3 Full-Scale SBR Design

To apply the findings from the lab-scale SBR experiment, a full-scale SBR shall be designed and compared to typical SBR configurations used for municipal and industrial wastewater settings based on case studies from the US EPA (*Wastewater Technology Fact Sheet Sequencing Batch Reactors*, 1999).

5.3.1 Reactor Sizing

The reactor refers to the main tank where all the phases of a SBR occur. The size of the basin is a function of the daily leachate flow, influent ammonium concentration, cycle time, and ammonium removal rate per gram of TSS.

The daily leachate flow and influent ammonium concentration are selected in Table 5-7. The ammonium removal rate per gram of TSS based on the experiment is at 1.65 mg/g TSS/cycle from SBR 1. The following table show basin sizes for full-scale SBRs in operation from the US EPA.

Table 5-8. Full-scale SBR basin volume and dimensions (Wastewater Technology Fact Sheet Sequencing Batch Reactors, 1999)

Design flow (m^3/day)	**Number of tanks**	**Per Tank Volume (m^3)**	**Total volume (m^3)**	**Flow-to-reactor volume ratio**	**Total Footprint (m^2)**	**Reactor depth (m)**
48	1	84	84	0.57	20	4.2
400	2	276	552	0.72	106	5.2
4800	2	3632	7264	0.66	1175	6.2
4000	2	1916	3832	1.04	618	6.2
5600	2	2712	5424	1.03	874	6.2
5840	2	3640	7280	0.80	1117	6.5
8000	2	3832	7664	1.04	1235	6.2
17000	4	6224	24896	0.68	3056	8.1
20800	4	5436	21744	0.96	2780	7.8

**Volume and length units are in gallons and feet in the original document*
***Volume conversion used: 1 gallon = 4 Litres, 1000 Litres = 1 m^3*
****Length conversion used: 3.3 feet = 1 meter*

The influent ammonium concentration is considered to be inversely related to the flow rate within the selected ranges in Table 5-7. This is due to landfill leachate being expected to be much less concentrated at high flow rates due to higher volume of water present for a relatively stable amount of leaching compound, such as ammonium, in the landfill. With this consideration, the likely amount of ammonium to be dosed in the reactors is calculated by multiplying the lowest concentration by the highest daily flow rate and vice versa. The following table summarizes this calculation done while interpolating three equal intervals between the maximum and minimum flow rates and concentrations to show a convergence to a peak amount of ammonium that may most likely be dosed into the reactors.

Table 5-9. SBR basin volume based on daily ammonium load from landfill leachate

Leachate flow rate (m^3/day)	288	396	504	612	720
Ammonium concentration (mg/L)	1500	1225	950	675	400
Ammonium per day (kg/day)	432	485	478	413	288
Ammonium Removal (%)	100	100	100	100	100
TSS required (kg)	261818	294000	290181	250363	174545
Reactor volume required (m^3)	14545	16333	16121	13909	9697
Flow-to-reactor volume ratio	0.02	0.02	0.03	0.04	0.07

Based on Table 5-9, the ammonium loading reaches a peak somewhere a little above the minimum leachate flow rate and a little less than the maximum ammonium concentration at 396 m^3/day and 1225 mg/L. The target removal for ammonium concentration is set at 100%, based on the discharge consent reported by Ridgeway (2016) for ammonium to be less than 0.5 mg/L. The basin volume based on flow rate, ammonium removal rate per gram of TSS, and TSS required ranges from as low as 9697 m^3 in the most dilute influent to 16333 m^3 corresponding to the highest ammonium load.

The cycle time in the experiment is at 24 hours, however, based on the rate of ammonium removal measured using the Vernier ISE, most of the ammonium removal occurred in the first 600 minutes or 10 hours of aeration without pH control with the removal being most efficient up to a pH of 6.5. With the measurement of ammonium removal over extended aeration with pH control, the 6.5 pH is reached anywhere from 120 to 550 minutes, most of which taking around 200 to 400 minute (Figure 4-13, Figure 4-14 & Figure 4-15). This indicates that, while keeping a react phase to cycle time ratio of 20 hours to 24 hours, as was in the experiment, it can be possible to reduce the cycle time down to 6 to 12 hours having a react phase lasting 5 to 10 hours. The reduction of cycle time to 6 to 12 hours per day, allows a smaller reactor volume since the daily flow rate can be theoretically divided by 2 to 4. This is considered to be achievable since leachate storage tanks in landfills can act as equalization basins to control the influent flow rate to the SBR. The following table and figure shows the effect of cycle times at 6, 8, 10, and 12 hours on the required volume based on the previously calculated steps in Table 5-9.

Table 5-10. Effect of cycle time on volume required for the reactor

Leachate flow rate (m^3/day)	288	396	504	612	720
Ammonium concentration (mg/L)	1500	1225	950	675	400
Ammonium per day (kg/day)	432	485	478	413	288
Cycle time (hrs)			***6***		
Reactor volume required (m^3)	3636	4083	4030	3477	2424
Flow-to-reactor volume ratio	0.08	0.11	0.13	0.18	0.30
Cycle time (hrs)			***8***		
Reactor volume required (m^3)	4848	5444	5374	4636	3232
Flow-to-reactor volume ratio	0.06	0.07	0.09	0.13	0.22
Cycle time (hrs)			***10***		
Reactor volume required (m^3)	6061	6806	6717	5795	4040
Flow-to-reactor volume ratio	0.05	0.06	0.08	0.11	0.18
Cycle time (hrs)			***12***		
Reactor volume required (m^3)	7273	8167	8061	6955	4848
Flow-to-reactor volume ratio	0.04	0.05	0.06	0.09	0.15

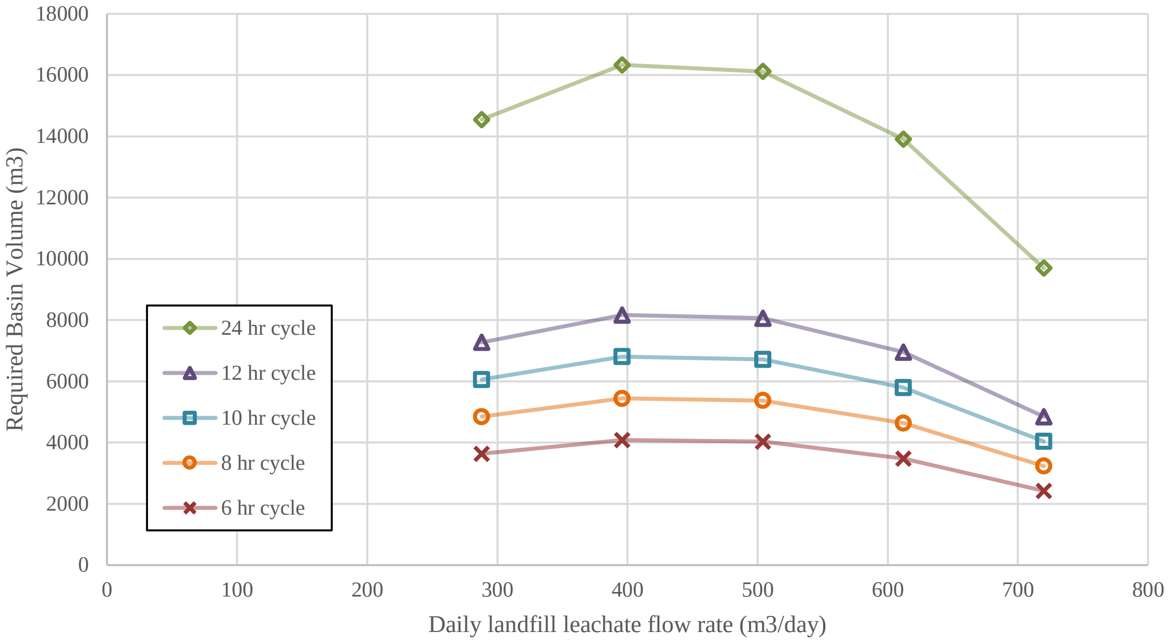

Figure 5-10. *Graph of required reactor volume spread at different cycle times at different leachate flow rates*

Based on Table 5-10 and Figure 5-10, the reactor volumes decrease with decreasing cycle times with the assumption that the daily flow rate can be divided equally to how many cycle times can be fit into each day. The curve shown in Figure 5-10 shows how the required reactor volume is at its highest at a relatively low leachate flow rate that is less diluted and therefore would have a higher ammonium concentration. The spread in the required reactor volumes decreases with reduction in cycle time, which makes the required reactor volume range for the ammonium loadings from Table 5-9 smaller.

The flow-to-reactor volume ratios are presented at each required reactor volume calculated to show that, for the case of such ammonium loading, the daily leachate flow rate is relatively small compared to the reactor size being required. This is due to the high ammonium concentration in landfill leachate, which takes advantage of the effect of dilution in reducing the concentration of ammonium that needs to be treated in the reactor. This is also discussed in the context of the lab-scale experimental set-up wherein the actual amount of ammonium removed in the SBR is based on the ammonium concentration after the influent has been diluted by the mixed liquor that has been treated from the previous cycle (Section 5.1). Comparing the flow-to-reactor volumes obtained for this study's full-scale SBR design to those from the US EPA cases (Table 5-8), the flow-to-reactor volumes are much smaller for this study. The US EPA cases are for domestic wastewater, some are for industrial but are not identified. Comparatively landfill leachate are of much higher ammonium concentrations to domestic wastewater, while at the same time having much less biodegradability. The challenge of balancing available organic resources for nitrifying is present for landfill leachate treatment, especially in SBRs.

From the analysis of reactor volume sizes, a 10-hr cycle is selected for the full-scale SBR, which has a 6806 m^3 basin requirement. This cycle is selected as it provides a reasonable reactor volume size. In actual application of full-scale SBR, redundancy should always be accounted for and two basins of this size would be recommended. Both can be run at the same time for days that follow high rainfall, and maintenance of one reactor does not completely disable the treatment plant's ability to treat landfill leachate. A leachate storage tank that can act as a buffer tank to provide more control on the influent flow rate is highly recommended as well. Under site-specific conditions, the basin size can be further reduced with the presence of the buffer tank, since the SBR may be loaded with ammonium at a more controlled rate.

The following table shows the sizing dimensions for a SBR running in 10-hr cycles (Table 5-11). A cylindrical tank shape is selected to reduce sludge scouring and improve mixing. The basin depth selected is 6.5, which is well within the range of actual full-scale SBRs from Table 5-8. The final working reactor volume is at 8,168 m^3 after rounding up the required tank diameter, and is above the required working reactor volume. It should be noted the working reactor volume refers to the volume of the reactor that is filled with mixed liquor and sludge.

Table 5-11. *SBR sizing dimensions with a 10-hour cycle time*

Volume of reactor required	m^3	6806
Reactor depth	m	6.5
Shape	Cylindrical Tank	
Reactor surface area	m^2	1047
Required reactor diameter	m	37
Final reactor diameter	m	40
Freeboard height	m	2
Final tank height	m	8.5
Final working volume	m^3	8168

5.3.2 Operational Requirements

Operation of the full-scale SBR consists of many factors if studied in great detail. For the purpose of this study, the two biggest factors in ensuring ammonium removal achieved in the full-scale SBR are aeration and supplementation. The aeration is the main facilitator of ammonium conversion through nitrification, while at the same time would contribute the highest costs in terms of power. The proper estimation of required aeration is necessary to optimize this aspect of the operation. Supplementation, specifically carbon supplementation, is important since the bacteria facilitating the nitrogen removal require carbon as an energy source. This may be consumed well before the target amount of ammonium removed is reached. As such, supplementation with external carbon sources is a significant consideration for a full-scale SBR. Dosing of a pH buffer to counteract the decreasing pH with nitrification is considered to be negligible as landfill leachate is typically of high pH, and the filling phase is likely enough to keep the pH within the range of 7 to 8.

5.3.2.1 Aeration requirements

The aeration requirements for the full-scale SBR are based on the dissolved oxygen consumption rate measured from the experiment at 0.01 mg O_2/L/minute. The cycle time selected is at 10 hours, and a react phase to cycle time ratio of 5 to 6, similar to that of the experiment, shall be used. This comes to an equivalent react phase of about 8 hours 20 minutes or 8.33 hours. A simple aeration requirement calculation is done to obtain a power requirement for aeration, which is expected to take up most of the power requirements for operating the SBR. The following table summarizes the calculations done to obtain compressor power requirement of 25 kW consuming around 210 kWhr/day with two 10-hour cycles per day (Table 5-12).

Table 5-12. Aeration requirement calculation

Parameter	Unit	Value
Reactor Working Volume	m^3	8168
O_2 consumption	mg/L/min	0.01
Aeration/React Phase	% of cycle	83
O_2 required per cycle	mg/L/cycle	5
Reactor O_2 required per cycle	kg/cycle	40.84
Percent O_2 transferred from bubble	%	14.8
O_2 required per cycle	kg	276
Mass fraction O_2 in air	%	21%
Air required per day	kg	2628
Density of air	kg/m^3	1.293
Volumetric flow of air	m^3/day	2033
Volumetric flow of air	m^3/hr	121.95
Power required	kW	25
Power consumption with 10-hr cycles	kWh/day	420

The calculation for aeration requirement gets the O_2 requirement based on the consumption rate of 0.01 mg DO/L/min. The O_2 required at the reactor size of 8168.14 m^3 is calculated to be at 40.84 kg/10-hr cycle. Using the calculated percent of O_2 transferred from air to water from Section 5.1.2. The actual supply of O_2 required in the reactor is at 276 kg/10-hr cycle. Since the aeration will use air to pump oxygen into the mixed liquor, the percentage of oxygen in air, which is about 21%, is accounted for to obtain the amount of air needed. The flow rate of the air needed is calculated to be at 2,033 m^3/day, accounting for two 10-hr cycles per day. This flow rate is then converted to cubic meters per hour and the power requirement is correlated to typical values. Typical values for air compressor power requirement at volumetric flow rates of air is obtained from VMAC (2023), which designs and manufactures air compressors. This correlation is used to make a simple approximation of power requirements from aeration, and it should be noted that specific service providers should be consulted for actual designs. Table 5-13 shows the approximate power requirement of mobile air compressors at different volumetric air flows. It should be noted that values from the source where in cubic feet per minute and were converted to cubic meter per hour by the researcher. A conversion factor of 3.3 feet for every meter is used. The nearest power output for the volumetric air flow of 121.95 m^3/hr is at 25 kW. This comes up to a power consumption of about 420 kWh/day with two 10-hr cycles with approximately 8.33 hours of aeration.

Table 5-13. Power output for mobile air compressors at different air flows (VMAC, 2023)

Power Output (kW)	Air flow (m3/hr)
5	25 - 34
10	51 - 68
25	127 - 170
50	255 - 340
100	510 - 680
250	1274 - 1699

5.3.2.2 Carbon supplementation

With the low biodegradability of landfill leachate, biodegradable carbon source available for biomass utilization in the influent can be assumed to require some supplementation. With oxidation of high amount of ammonium, high amounts of nitrogen in the form of nitrates and some in nitrites is expected, and denitrification of these may be desired especially with effluent standards with total nitrogen or total inorganic nitrogen requirements. Removal of nitrates and nitrites are facilitated by denitrification (Section 2.3.2), and although this was not the focus of the experimental set-up it is considered in the full-scale SBR. With an approximate 1.6 to 2 hour of settle phase, an anoxic condition is expected where denitrification may occur. As was observed in the instance of adding acetic acid to SBR 1 followed by an anoxic period yielded pH recovery indicating denitrification might have been occurring. For the full-scale bioreactor, carbon sources are assumed to be depleted by the aeration phase and the settle phase is assumed to require carbon supplementation, which is often the case for many nitrogen-removal systems (Davis & Cornwell, 2013).

The most common carbon source used to supplement biological nitrogen removal is methanol. Other carbon sources include, ethanol, acetate, and acetic acid (WRF (The Water Research Foundation), 2019). For the purpose of this study, methanol and acetic acid are to be evaluated as potential carbon sources to remove nitrogen in the full-scale SBR. The COD equivalence for a gram of methanol is 1.5g of COD (WRF (The Water Research Foundation), 2019) while a gram of acetic acid is 1.067g of COD (Atasoy et al., 2020).

For the calculation of carbon supplementation, the approximation for required amount of readily biodegradable COD (rbCOD) in grams to remove a gram of nitrate-nitrogen (NO_3-N) from Mackenzie L. Davis (2010) is used and shown in the following equation.

$$\frac{g\ of\ rbCOD}{g\ of\ NO_3 - N} = \frac{2.86}{1 - 1.42Y_n}$$

Equation 5-8. *Equation calculating the g of rbCOD required to remove a gram of NO_3-N (Mackenzie L. Davis, 2010)*

In Equation 5-8, Y_n is the net biomass yield in gram of VSS per gram of rbCOD consumed. According to WRF (The Water Research Foundation) (2019), the typical value for biomass yields in post-anoxic tanks is at 0.3 g VSS/g rbCOD. Timur and Özturk (1999) studied anaerobic SBRs for treating landfill leachate and obtained a biomass yield of 0.1 g VSS/g rbCOD. Both biomass yields are selected to calculate a range of amount of carbon supplements required. The NO_3-N is assumed to have all been fully converted from the ammonium-nitrogen from the influent loading. This results to the amount of nitrogen to be removed per day to be at 377.3 kg, which is calculated from the nitrogen content of the peak daily ammonium loading of 485.1 kg per day considered in the reactor sizing (Section 5.3.1). The assumption being made here is that the nitrogen to be denitrified with the assistance of carbon supplementation comes from the landfill leachate ammonium concentration. This study does recognize that in actual design, site-specific testing and analysis is best for estimating amount of carbon supplementation required. The following table shows the calculation for required volume of methanol and acetic acid per day at two different biomass yields.

Table 5-14. Calculation for required volume of methanol and acetic acid per day for nitrogen removal

Parameter	Unit	Values	
Biomass yield	g VSS/g COD	0.1 (Timur & Özturk, 1999)	0.3 (WRF (The Water Research Foundation), 2019)
Required COD	g COD/g NO_3-N	3.33	4.98
Nitrogen	kg/day	377	377
Required methanol per g N	g COD/g NO_3-N	2.22	3.32
Required mass of methanol per day	kg/day	838	1253
Density of methanol	kg/m^3	792	
Required volume of methanol per day	m^3/day	1.06	1.58
Required mass of acetic acid per day	kg/day	1179	1762
Density of acetic acid	kg/m^3	1050	
Required volume of acetic acid per day	m^3/day	1.12	1.68
Strength of acetic acid	%	80%	
Required volume of acetic acid at strength per day	m^3/day	1.40	2.10

The required amount of methanol per day is 1.06 m^3 to 1.58 m^3, while for acetic acid it is from 1.40 m^3 to 2.10 m^3. Similar volumes of both carbon supplements are required per day based on the calculation and only very slight difference should not necessitate any drastic difference in storage requirements. The acetic acid is on average higher by 0.5 m^3 for daily requirements. The selection of appropriate carbon supplement source can be dependent on cost and operator preference. The following subsection discusses the estimated costing for the full-scale SBR design and the operational requirements.

5.3.3 Costing and Economic Feasibility

The costing for the full-scale SBR is estimated based on estimated costs of operation requirements and using the Lang's constant to estimate the capital cost of establishing the full-scale SBR. The operational cost is compared to the cost of transporting leachate to the nearest wastewater treatment plant from Ridgeway (2016). A simple calculation of the capital cost based on the operational costs is done and the economic feasibility of the full-scale SBR is assessed.

Prices calculated in the following costings are adjusted to the latest prices using New Zealand dollar conversion from Inflation Tool (2024) from the year of the costing's source to 2023 valuation.

5.3.3.1 Operational cost

The operational costs are estimated based on the calculated aeration power requirement and carbon supplementation requirements. These are two identified operational requirements identified to be crucial in ensuring the SBR is able to remove ammoniacal-nitrogen from the landfill leachate. Based on the aeration requirements, the power consumption for aerating the SBR is at 420 kWh/day with two 10-hour cycles per day. According to the MBIE (2023), the average cost of electricity in the industrial sector in 2023 was at NZD 0.1560/kWh. The calculated daily power consumption from aeration would be about NZD 65/day, which amounts to NZD 23,725.00/year for power consumed by aeration.

The costing for carbon supplementation is calculated for both methanol and acetic acid. According to Methanex (2024), their price for methanol in the Asia Pacific region at the time of writing is at USD 390/MT. Using a conversion rate of NZD 1.64 is to USD 1.00, which is the conversion at the time of writing, the unit price for methanol is at NZD 639.6/MT or NZD 0.64/kg of methanol. Based on the daily requirement for methanol, the daily cost for methanol would range from NZD 536.27 to NZD 801.60. The annual cost is calculated to range from NZD 195,738.21 to NZD 292,584.30 for methanol. Acetic acid at 80% strength, on the other hand, has a price of NZD 2,830.80/200 L or NZD 14.15/L based on Echochem (2024). This amounts to NZD 19,855.38 to NZD 29,679.30 per day. The annual cost to use acetic acid is estimated to be from NZD 7,247,215.10 to NZD 10,832,945.00. Based on the estimated costings, methanol is significantly more cost effective to be used as a carbon supplement and is selected for the full-scale SBR design. It should be noted that shipping costs are not accounted here, and availability may vary depending on the site location of the landfill. Other advantages of methanol include lower sludge production; however, this is due to slower biomass growth, which is not ideal especially at reactor start-up (WRF (The Water Research Foundation), 2019).

The need for operators of the full-scale SBR is considered for the operational cost. A conservative estimate of three operators is used for the costing calculation. According to Careers.govt.nz (2024), a wastewater treatment plant operator with at least 2 years of experience can make up to NZD 70,000.00 per year. This amounts to an estimated NZD 210,000.00 in wages for operating the SBR.

Adding up the operational cost from aeration, carbon supplementation, and manpower, the annual operational cost for running the full-scale SBR is estimated to be at NZD 526,309.29. This considers the higher daily methanol requirement.

Based on Ridgeway (2016), treatment of landfill leachate from Hampton Downs at the Mangere Wastewater Treatment Plant costs NZD 22.00/m^3 in 2016. Using an inflation conversion of NZD 1.00 in 2016 being equivalent to NZD 1.33 in 2024, the current cost would be at NZD 29.26/m^3. Accounting for increase in landfill leachate flow since then, the landfill leachate flow used in the full-scale SBR design is used to calculate the operational cost. At an average daily leachate flow rate of 504 m^3/day, the daily operational cost for offsite treatment is at NZD 14,747.04. This would then cost NZD 5,382,670.00 per year. The following table summarizes the operational costs of the full-scale SBR and offsite landfill leachate treatment.

Table 5-15. Annual operational cost for a full-scale SBR and offsite treatment

Treatment Method	Full Scale SBR	Offsite treatment
Aeration power consumption	NZD 23,725.00	None
Carbon supplementation with methanol	NZD 292,584.29	None
Manpower	NZD 210,000.00	None onsite
Cost of transport	None	NZD 5,382,669.60
Total operational cost per year	NZD 526,309.29	NZD 5,382,669.60

Comparing the operational costs, the operational cost for a full-scale SBR is significantly smaller and is only about 10% that of the offsite treatment. This indicates that there is good potential that operating a full-scale SBR onsite would translate to savings compared to offsite treatment.

5.3.3.2 Capital cost and economic feasibility

The estimate of capital cost for the full-scale SBR includes the use of Lang factors. Lang factors estimate a capital cost for a processing plant based on the total equipment costs. There are also Lang factors that can be used to estimate various installations in a processing plant based on the total equipment cost (Bouman et al., 2005). For the case of this study's capital cost estimation, the Lang factor based on the type of plant is to be used. Since the full-scale SBR processes landfill leachate, this falls into the category of fluid processing plant with a Lang factor of 5. The following table shows the Lang factors according to type of processing plant (Table 5-16).

Table 5-16. Lang factors according to type of processing plant (Bouman et al., 2005)

Lang Factor	**Type of processing plant**
4.0	Solids processing plant (e.g. cement plant)
4.3	Solid-fluid processing plant (e.g. fertilizer plant)
5.0	Fluid processing plant (e.g. oil refinery)

With a selected Lang factor of 5, the total equipment cost to be considered is the annual operational cost calculated for the full-scale SBR, which is at NZD 526,309.29. The capital cost is then calculated to be at NZD 2,631,546.46, which is still less than the annual cost for offsite treatment. Even if the offsite treatment is calculated at the minimum leachate flow rate for the design full-scale SBR at 288 m^3/day, the annual cost of offsite treatment would still be greater than the capital cost at NZD 3,075,811.20. This indicates that the capital investment on a full-scale SBR to treat landfill leachate for ammonium removal has good economic feasibility as an alternative to offsite treatment, and provides significant savings for the landfill, such as Hampton Downs for the case of this design.

6 Summary and Conclusion

The study focused on the potential for ammonium removal from landfill leachate using the sequencing batch reactor technology. An experiment using a lab-scale SBR and high ammonium influent wastewater is used to measure the ammonium removal potential of a SBR at high ammonium loadings similar to that of landfill leachate. The sludge used in the lab-scale set-up is seeded from cow manure, and measurable ammonium removal is achieved after two months of inoculation. A second SBR seeded using the sludge from the first SBR, and achieved measurable ammonium removal after 30 days of inoculation. Aeration is prioritized in the cycle programs used, since ammonium removal through oxidation and nitrification is the aim of the study. The successful inoculation of ammonium removing sludge from cow manure indicates the potential for the waste material to be used as activated sludge seed in areas with difficult access to wastewater treatment plants, and may require onsite treatment.

The lab-scale SBRs achieved ammonium removal at influent ammonium loading as high as 720 mg/L, and found that the SBR was stable and achieving high removal efficiency around 98% to 99% at loading up to 124 mg/L with a TSS of 17 to 18 g/L during this most stable period. SBR 1 was run for 250 days of operation, while SBR 2 was run for only 130 days due to overdosing of 1M NaOH. A significant change in operation of the SBR at higher influent ammonium concentrations is the need to buffer pH to the 7 to 8 range in between cycles, since the high ammonium load required greater consumption of alkalinity. With no significant pH recovery observed during the regular cycles, it is indicative that the anoxic phase is unable to undergo denitrification due to a lack of available biodegradable sources since it follows the aeration phase. Removal rates are measured over the most stable period of SBR operation with a removal rate of 1.65 mg NH_4^+/g TSS/ per cycle is achieved from the experiment. Using an ammonium ISE probe to be able to take ammonium measurements in real-time during a cycle, an ammonium removal rate of 0.0508 mg NH_4^+/L/min is calculated. An inhibition in ammonium removal due to low pH is observed, and as such period of extended aeration with pH control is done on SBR 1 after 250 days of experimentation. The extended aeration period found that ammonium removal continued as long as pH is maintained above 6.5, with the pH taking around 100 to 500 minutes to fall back to 6.5 or below indicating continuous ammonium removal. The oxygen consumption rate is also found to be at 0.01 mg DO/L/min, while the average mass transfer coefficient of O_2 at 0.0144 min^{-1}, and the percentage of oxygen transferred from gas to the mixed liquor calculated to be at 14.8%.

Since landfill leachate flowrate can be site-specific, due to the influence of precipitation, the design of the full-scale SBR is based on estimated landfill leachate flow in the Waikato region. The approximation of landfill leachate flow is based on 20% of the annual precipitation in line with literature (Ehrig, 1983; Serdarevic, 2018). The estimate found that class 1 municipal landfills in the Waikato region are located in areas with an estimated landfill leachate flow anywhere from 6 to 15 m^3/hec/day. For the site characteristic of hectares landfilled, the Hampton Downs landfill is used for this case with approximately 49 hectares currently landfilled based on calculation of per annum waste landfilled in the area and years of operation. Information regarding the site mainly obtained from publicly available data from *Waste in the Waikato Region* (2022), Ministry for the Environment (2021), and Ridgeway (2016). A landfill leachate ammonium concentration range of 600 to 1500 mg/L is considered for the full-scale SBR design. The full-scale SBR is calculated to required a working volume of 8168 m^3 and a depth of 8 m, including freeboard running two 10-hour cycles with around 8 hours of aeration, based on the ammonium and oxygen consumption rates from the experimental set-up. The full-scale SBR design is comparatively not oversized compared to typical SBR cases reported in the *Wastewater Technology Fact Sheet Sequencing Batch Reactors* (1999) by the US EPA. Comparatively the ratio of influent flow to total volume of SBR is much lower for this study's SBR design, and this is likely due to sizing the tank to a bigger volume to dilute the ammonium concentration and achieve 100% removal at the rates measured from the experiment.

Costing for the full-scale SBR design calculates operational costs mostly coming from aeration power consumption, carbon supplementation requirements, and manpower. A total of NZD 526,309.29 is estimated as the operational cost per year for the full-scale SBR. Based off information from Ridgeway (2016) and inflation rates, the estimated current cost for offsite treatment for the average daily landfill leachate flow is at NZD 5,382,669.60 per year. The operational costs for a full-scale SBR to provide onsite treatment is found to have good potential. The capital cost is calculated using a Lang factor of 5, which results in a capital cost of NZD 2,631,546.46, which is still less than the operational cost for offsite treatment. The economic feasibility of having onsite treatment with a full-scale SBR is high based on this calculation. It should be noted that simple estimations are done for the economic feasibility of the full-scale SBR, and a detailed costing is still more accurate should the actual implementation of a full-scale SBR be done.

7 Recommendations for further study

The current study provides information on the potential of using SBR technology in treating the high ammoniacal-nitrogen content in landfill leachate. Further investigation into the viability of the technology is recommended in order to optimise removal rates, which can further improve on the full-scale SBR design calculated in this study. The following are specific recommendations to improve on the research:

- The experiment on the SBR lasted for 250 days due to time constraints of the research program, however, other studies conduct experiments over longer periods of time. Longer experimentation such as longer acclimation at high influent ammonium concentration is recommended to find out if stability can be observed at these high concentrations.
- The experimental set-up used cow manure for the sludge inoculant and influent wastewater. Acquiring landfill leachate samples to actually test ammonium removal in the SBR is recommended to provide more accurate results on removal considering the actual characteristics of landfill leachate, including low biodegradability.
- This research focused mainly on ammonium removal through nitrification. It is recommended that denitrification can be further studied with the same experimental SBR, since the addition of acetic acid in the extended aeration runs indicated some denitrification occurring in the form of pH recover after aeration was turned off.
- Further research can be done to identify the microbial composition of the sludge used in the research and compare it to activated sludge from domestic wastewater treatment plants to find out the difference in the biomass composition of sludge seeded from cow manure to activated sludge for wastewater treatment.
- Variation in cycle programs can be further studied and tested to achieve alternative modes of biological nutrient removal, such as partial nitritation (Section 2.3.1) and simultaneous nitrification and denitrification (SND).
- Testing for removal potential of a SBR for other common or emerging landfill leachate contaminants can be further studied to better understand the full capabilities of a SBR for landfill leachate treatment.
- Detailed design and costing for a SBR treating landfill leachate can be further studied as there is not much information on full-scale SBR design specifically for landfill leachate treatment.

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