Impact of wastewater temperature & influent flow as the indicators of climate change on wastewater treatment systems

by

Vaibhav Bhate

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Abstract

Wastewater treatment systems are essential for the safety of people and the wellness of the environment. Climate change has caused significant changes in precipitation patterns, surface temperatures, snowmelt and surface runoff events that also change wastewater characteristics. Change in wastewater temperature and influent flow (indicators of climate change) can affect the physical, chemical, and biological processes in wastewater treatment plants (WWTP) and wastewater treatment performance.

This study focused on the impact of wastewater temperatures on secondary / biological activated sludge (AS) systems including conventional AS and Ludzack-Ettinger, using BioWin as a WWTP modeling software. For each treatment system, a wide range of wastewater temperatures and solids retention time (SRT) values were assessed, and removal efficiencies of total chemical oxygen demand (COD), carbonaceous biochemical oxygen demand (cBOD), total suspended solids (TSS), and total ammonia were evaluated. It was observed for the conventional AS and Ludzack-Ettinger systems that the effluent total COD, cBOD, TSS and total ammonia concentrations were higher for temperature (T) < 13 ℃ and lower for T > 13℃. While increasing the SRT values, total COD and TSS concentration showed an increasing trend. cBOD and total ammonia concentration dropped with the increase in SRT values.

In addition, biological nutrient removal (BNR) efficiency under the impact of wastewater temperatures was studied for AS systems including conventional AS, modified Ludzack-Ettinger (MLE), Phoredox, anaerobic/anoxic/aerobic (A2O), and modified Bardenpho. Simulations were conducted at varying SRT and wastewater temperatures and cBOD, total

ammonia and total phosphorous (TP) wastewater characteristics were studied. It was observed that increasing SRT improved the total ammonia removal efficiency for most of the AS systems but the removal efficiency for cBOD and TP deteriorated beyond a certain SRT value. Increasing temperature improved the removal efficiency for cBOD and total ammonia for all the systems. Whereas, increasing temperature impacted the TP removal efficiency for most of the systems except conventional AS and modified Bardenpho system. 5 days SRT and 17 ℃ were the critical parameters for all the AS systems except the conventional AS system.

This study also assessed the combined impact of change in the influent flow pattern and wastewater temperatures on conventional AS and MLE systems. It was observed that wastewater temperature was a critical parameter for removing the total COD, cBOD and total ammonia. Whereas SRT was a crucial parameter for TP removal. The high flow presented better removal efficiency for total COD and cBOD (for the conventional AS system) than at low flow, at 5 ℃. But it reversed at 25 ℃ for the conventional AS system. The results indicated significant changes in treatment performance due to temperature and precipitation changes that may be brought by climate change. The results of this study help to understand how climate change may impact wastewater treatment plants in Canada and how plants can modify their operational parameters to adapt to these changes.

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List of Abbreviations

A2O Anerobic/anoxic/aerobic

ANN Artificial neural networks

AOB Ammonia oxidising bacteria

AS Activated sludge

ASM Activated sludge models

BNR Biological nutrient removal

BOD Biochemical oxygen demand

BPR Biological phosphorous removal

C / °C ° Celsius

cBOD Carbonaceous biochemical oxygen demand

CCCR Canada's climate change report

COD Chemical oxygen demand

CSO Combined sewer overflow

DO Dissolved oxygen

EDCs Endocrine-disrupting compounds

EU European Union

F/M Food / microorganisms

FST Final settling tank

GAOs Glycogen accumulating organisms

GHGs Greenhouse gases

HF High flow

IPCC Intergovernmental Panel on climate change

ISS Inert suspended solids

IWA International Water Association

LF Low flow

MLE Modified Ludzack-Ettinger

MLSS Mixed liquor suspended solids

MVS Multivariate statistical methods

N_2O Nitrous oxide

NCA National climate assessment

NML Nitrifying mixed liquor

NO Nitric oxide

NO_2 Nitrite

NO_3 Nitrate

NOB Nitrite oxidising bacteria

OHOs Ordinary heterotrophic organisms

OPO_4 Orthophosphate

PAOs Phosphorous accumulating organisms

PHA Poly-β-hydroxyalkanoates

PHB Polyhydroxybutyrate

ppm parts per million

PST Primary settling tank

RAS Return activated sludge

SRT Solids retention time

SS Suspended solids

T Temperature

TP Total phosphorous

TSS Total suspended solids

UCT process University of Cape town process

VFA Volatile fatty acids

VOC Volatile organic compounds

VSS Volatile suspended solids

WAS Waste activated sludge

WHO World Health Organization

WSER Wastewater systems effluent regulations

WWTPs Wastewater treatment plant

Chapter 1: Introduction

1.1 Background

Wastewater treatment plants (WWTPs) play an integral role in improving public health and society's life (Elhassan & Eldouma, 2018). WWTPs with different levels of treatment can remove settleable organic solids, suspended and soluble organic matter, nutrients, heavy metals, and inorganic solids (Sperling Von, 2007; Zouboulis et al., 2015). Hence, it becomes important to consider the factors that might affect the performance and operation of WWTPs.

Climate change in Canada is particularly concerning as the surface warming in Canada is higher than the global average. This leads to an ambient temperature rise, irregular precipitation patterns, natural events such as floods and droughts, and sea-level rise (Bush & Lemmen, 2019). All these events that occur because of climate change have direct impact on the WWTPs, and it is essential to understand how climate change is impacting the WWTPs. It was also observed that snowmelt changes the wastewater temperature and inflow. Also, infiltration through the sewers may also occur during the snowmelt (Kaczor & Bugajski, 2012). Due to ageing of the sewers, the water generated due to snowmelt infiltrates in the sewers and mixes with the wastewater. This may change the wastewater temperature and wastewater characteristics. This impacts the operational criteria and treatment efficiency of the wastewater treatment systems.

Wastewater treatment and optimization models are available to predict and optimize the performance of the wastewater treatment processes, dosage of various chemicals, kinetic and stoichiometric parameters, and other process-related parameters. Models were also

developed where specific input and scenarios were studied by conducting simulations and used to optimize WWTPs. BioWin is such a wastewater numerical modeling software developed by the International Water Association (IWA) to forecast the tentative scenarios due to changes in the system input (Gernaey et al., 2004).

Ambient temperature and change in precipitation are major consequences of climate change that impact the wastewater treatment systems. The wastewater temperature and influent flow to the WWTPs are directly impacted by the above-said factors. The wastewater temperature has a significant impact on biological treatment (Langeveld et al., 2013). Wastewater bacteria and nitrifiers are sensitive to temperature changes (Plósz et al., 2009; Zhang et al., 2019). Hughes et al. (2021) mentioned that temperature change could cause odor problems in the treatment system. The activity of phosphorous accumulating organisms is impacted by the increase in the wastewater temperature (Chen et al., 2014; Zhilong et al., 2014). Overall, changes in wastewater temperature play an important role in determining the performance of physical, chemical, and biological wastewater treatment processes.

Mines (2007) mentioned that changes in the influent flow change the strength of the wastewater. The low flow influent (LF) has high strength, while the high flow influent has low strength due to dilution. This change in flow impacts the operational aspects of the WWTPs (VO et al., 2014), and modifications to the solids retention time (SRT), aeration, pH, and chemical dosing may be required to meet the effluent discharge criteria.

Performing pilot scale experiments could be a method to assess the impact of wastewater temperature and influent flow change. But this method could be cost exhaustive and wide of scenarios on various biological activated sludge (AS) systems could not be monitored. Modelling is the best available tool to assess the changes in biological treatment processes for the WWTPs. SRT could be used as a control parameter to optimize the wastewater model and maintain (or improve) the effluent condition (Ekama, 2010). It was also observed that adequate SRT and change in the aeration conditions improved the nitrogen removal process (Elawwad et al., 2019).

Various studies have been performed on various biological AS systems by varying SRT, wastewater temperature, DO, pH and other kinetic and stoichiometric parameters. Various numerical models have been simulated on wide range of scenarios. But this study tries to evaluate combined impact of wastewater temperature change and varying SRT on various biological AS systems. This study also attempts to review the combined impact of influent flow and wastewater temperature change along with varied SRT range on different AS systems. biological nutrient removal (BNR) is very critical for the WWTPs. Various biological systems are studied in this study such as conventional AS, MLE, Phoredox, A2O and modified Bardenpho. The main reason for the selections of these systems was to evaluate the typical removal attribute of each system. For example, carbon removal was a criterion for all the systems; total ammonia removal from MLE system; TP removal for Phoredox system and, total ammonia and TP removal from A2O and modified Bardenpho systems. This is an unique quality of this study that various biological systems were considered and using numerical modeling, simulations were conducted on a wide range of wastewater temperatures. Wastewater temperatures changed during the course of various

climatic events and change in wastewater temperature was a suggestive that climate change impacted the WWTPs. Efforts have been made in this study to optimize the WWTP and avail best efficiency for BNR to mitigate the impact of change in the wastewater temperatures.

1.2 Objective

The objective of this study is to assess the impact of change in wastewater temperature and influent flow (as the parameters impacted due to climate change)on the secondary wastewater treatment systems using the numerical model in BioWin 6.0. Various simulations using wastewater temperatures and influent flow were conducted on a wide range of activated sludge systems such as conventional activated sludge (AS), Ludzack Ettinger, modified Ludzack Ettinger, Phoredox, anaerobic/anoxic/aerobic (A2O) and modified Bardenpho.

The primary objectives of the study are:

- Analyze how change in ambient temperature (due to climatic conditions) change the wastewater temperature range.
- Identify the key indicators of climate change that impact the WWTPS.
- Generate various modifications for the secondary treatment, such as the addition of aerobic and anoxic bioreactors to the secondary system and setting the flow and wastewater characteristics.
- Conduct simulations for the temperature range of 5-25 °C and varying SRT range from 3-11 days for the secondary treatment system model.

- Conduct simulations to assess the BNR efficiency (nitrogen and phosphorous) from the modified secondary treatment system models for 5-25 °C wastewater temperature range.
- Study the trends of influent flow and corresponding wastewater characteristics, which were obtained by studying the historic (5-year) flow data of a treatment plant in Ontario.
- Conduct the simulations for the high flow and low flow conditions for secondary treatment models on varying wastewater temperature (i.e., 5-25 °C).

1.3 Thesis Organization

There are six chapters in this thesis.

- Chapter 1 provides the background information, objective of the study, thesis organization and references.
- Chapter 2 comprises the literature review about climate change, the impact of climate change in Canada, and the impact of climate change on the wastewater systems. This chapter also includes the fundamentals of AS models and configurations.
- Chapter 3 includes a study on the combined impact of temperature and SRT changes on conventional AS and Ludzack-Ettinger systems.
- Chapter 4 studies the combined effect of temperature and SRT changes on conventional AS and other BNR systems such as modified Ludzack Ettinger, Phoredox, anaerobic/anoxic/ aerobic (A2O), and modified Bardenpho.
- Chapter 5 studies the effect of influent flow (high flow/low flow) and wastewater temperature on the secondary treatment systems (conventional AS and MLE).

- Chapter 6 summarizes the overview and significant results of the thesis. It also provides recommendations for future work.

1.4 References

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Chapter 2 Literature Review

2.1 Importance of wastewater treatment

Health and sanitation play a vital role in the efficient lifestyle of human beings. The human lifestyle has evolved since the inception of humans. Sanitation means not only adequate cleanliness and hygiene, but also appropriate disposal of the waste generated by humans. About 2.6 billion of the world's population are deprived of minimal sanitation, leading to nearly 10 % of the global epidemic (Mara et al., 2010).

Large amounts of domestic wastewater is generated every day. The wastewater incorporates sewage, surface runoff from roads for combined systems and wastewater infiltrating through the sewer pipes (Holger et al., 2006). After the treatment of the raw sewage, the effluent is discharged into the nearest water body. The effluent characteristics of the discharged water must be below the assimilative capacity of the water body to avoid the contamination of water sources and eutrophication (Sperling Von, 2007). Hence, the treatment of wastewater is necessary for safe sanitation and the protection of aquatic life.

Wastewater treatment is bifurcated into different levels: preliminary, primary, secondary, and tertiary. The preliminary treatment removes large, suspended materials; primary treatment removes the settleable organic solids; secondary treatment removes the suspended and soluble organic matter and nutrients too; and the tertiary treatment emphasizes the removal of nutrients, heavy metals, and inorganic dissolved solids (Zouboulis et al., 2015; Sperling Von, 2007).

Secondary treatment is a one of fundamental levels in the wastewater treatment system. It is a biological treatment system to remove the organic matter (biochemical oxygen demand

- BOD) and suspended solids (total suspended solids). The domestic sewage comprises solids, organic matter, and nutrients such as nitrogen and phosphorous. The removal of the organic matter in the secondary / biological treatment is boosted by the microbial community, which fast-track the process of decomposition of the organic matter (soluble BOD and particulate BOD). The kinetics of the microbial community is controlled by various physical parameters such as temperature, pH, alkalinity, dissolved oxygen (DO), and contact time. (Sperling Von, 2007).

2.2 Impact of climate change

The impact of climate change is well evident. Some organizations anticipated the consequences of climate change on the developed world. Frequent and severe changes in the climatic setup are projected to damage the infrastructure and various ecosystems (Reidmiller et al., 2017). Intergovernmental Panel on Climate Change (IPCC) predicted that by 2050, significant economies such as the US and European Union (EU) would see an adverse impact on their economic growth (Bush & Lemmen, 2019; Du et al., 2017). Also, according to IPCC assessment report 5, climate change is expected to affect energy consumption (more energy for cooling and less energy for heating) in the coming years (IPCC, 2014). Potential migration is also driven by weather events such as droughts, floods, and temperature rise, which are consequences of climate change (Kaczan & Orgill-Meyer, 2020; Dennis & Fisher, 2018).

2.1.1 Impact of climate change on the environment in Canada

The national policies decided for Canada are based on the scientific assessment conducted by IPCC. Canada's Climate Change Report focuses on the physical science of climate and the possibility of some modification in the future, based on the global analysis of IPCC.

The activities done by humans emit greenhouse gases (GHGs). GHGs are one of the primary reasons for the warming of the climate. In Canada, Canadian Greenhouse Gas Measurement Program handles various stations across Canada to monitor GHGs emissions (Bush & Lemmen, 2019).

Bush & Lemmen (2019) also mentioned that using various climate models, the concentration of carbon dioxide in the atmosphere could reach from nearly 400 parts per million (ppm) to 580 ppm in 2050. It may range from around 380 ppm to 980 ppm, 2100. The study suggests a relative increase in the concentration of atmospheric CO_2 in the future, which would impact global warming. As per the study of various climatic models, the change in average surface temperature could range from approximately 1.5 °C to 2.2 °C in 2050. It may vary from 1.5 °C to 4 °C in 2100. The Paris Agreement has also set the target of the change of temperature to 1.5 °C by 2050.

These projections are concerning and give a direct warning of the climate crisis in Canada (Bush & Lemmen, 2019) as Canada's warming (surface warming) is faster than global change. The warming is due to snow and sea ice loss, which ultimately reduces the capacity to reflect solar rays. The pattern of precipitation is also gradually changing because of climate change. From the study in CCCR, the precipitation pattern in Canada is more likely to get extreme. However, it is not easy to estimate the intensity of the precipitation.

Félio (2017) studied the environmental effects of climate change. They are as follows:

- Availability of basic amenities of potable water distribution and wastewater collection.
- Environmental impact includes the discharge of toxic substances into the atmosphere and the environment.

- Deterioration of water quality and severe harm to various ecosystems.
- The sources of potable water are polluted.

2.2 Impact of climate change on wastewater treatment systems

Due to the high magnitude of rainfall under climate change, there could be a direct impact on the combined sewer overflow (CSOs) and frequent occurrence of flood-like events (Kleidorfer et al., 2009; O'Neill II, 2010). The winters could be wetter, and high rainfall intensity could be observed (Langeveld et al., 2013). Climate change may also impact the pattern of seasons. Extreme rainy conditions would be prevalent in summer and could increase the number of rainy days in summer (Klein Tank & Lenderink, 2009). Langeveld (2013) also mentioned that the combined impact of temperature change and precipitation would give an overall idea of the climate change on the wastewater system. Figure 2.1 represents the overall impact of increased rainfall and increased temperature on the

WWTPs. The WWTP infrastructure is damaged, and higher operation and maintenance costs are occurred, as shown in Figure 2.1.

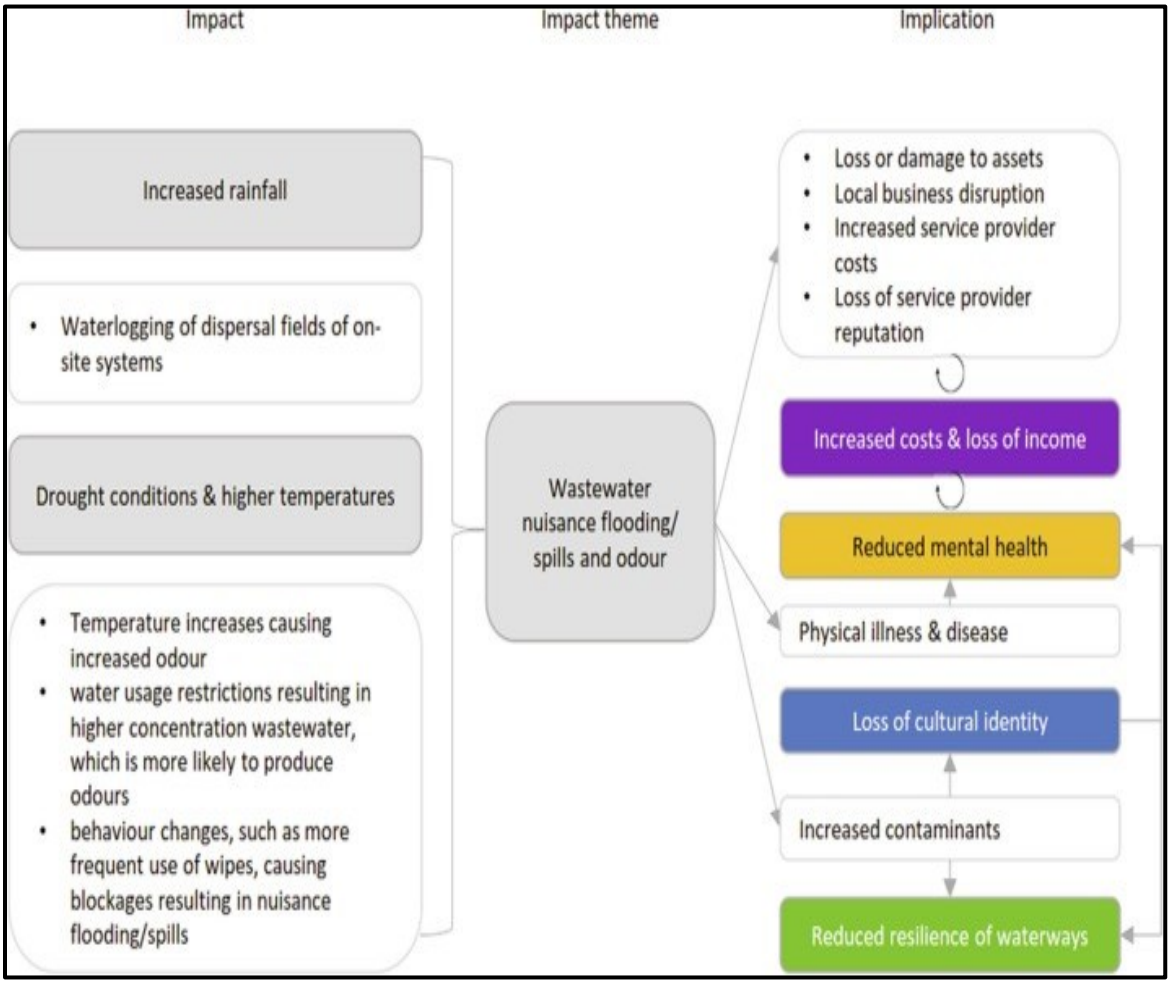

Figure 2.1: Impact and implication of climate change on WWTPs (Hughes et al., 2021)

2.2.1 Impact of temperature on the wastewater treatment system

Abdulla et al. (2020) and Alisawi (2020) reported that temperature is an essential parameter in maintaining the removal efficiency of the BOD from the treatment plant. The study concluded that temperature impacts the removal of Chemical Oxygen Demand (COD) and TSS. Further, it was observed that the BOD removal was maximum in summer, and eventually, the removal efficiency dropped in winter by a maximum of 73.68%. COD removal was also observed to be better in summer, and gradually the efficiency dropped in

winter (47% maximum). TSS was least impacted by the change in temperature. An increase in temperature improved the biological degradation of the biomass. Also, heavy loadings of COD and BOD were observed due to a drop in precipitation because the rainwater did not assimilate into the sewers.

Hughes (2021) specified the impacts of temperature on various wastewater infrastructures. It was evident that the odor problem was increased in the wastewater conveyance structures with the increase in temperature. Also, blockages were observed in pumping stations due to the increased flow of sewage (more water consumption in warm weather). The biological and sludge treatment processes were significantly impacted by the change in wastewater temperature.

The change in wastewater temperature impacts the nitrification rate. Eventually, with increase in the wastewater temperatures, the effluent ammonia concentration drops. The increase in temperatures improves the nitrification rate. This ultimately, accelerates the ammonia removal from the system. Figure 2.2 depicts the drop in the ammonia concentration with respect to the increase in temperature. The aerobic treatment is dependent on temperature as well. With the increase in temperature, the effluent TSS increases with the increase in the growth rate of the microbial community. The increase in the growth rate is evident for a specific temperature range. The efficiency of phosphorous removal is decreased with the increase in temperature. (Alisawi, 2020).

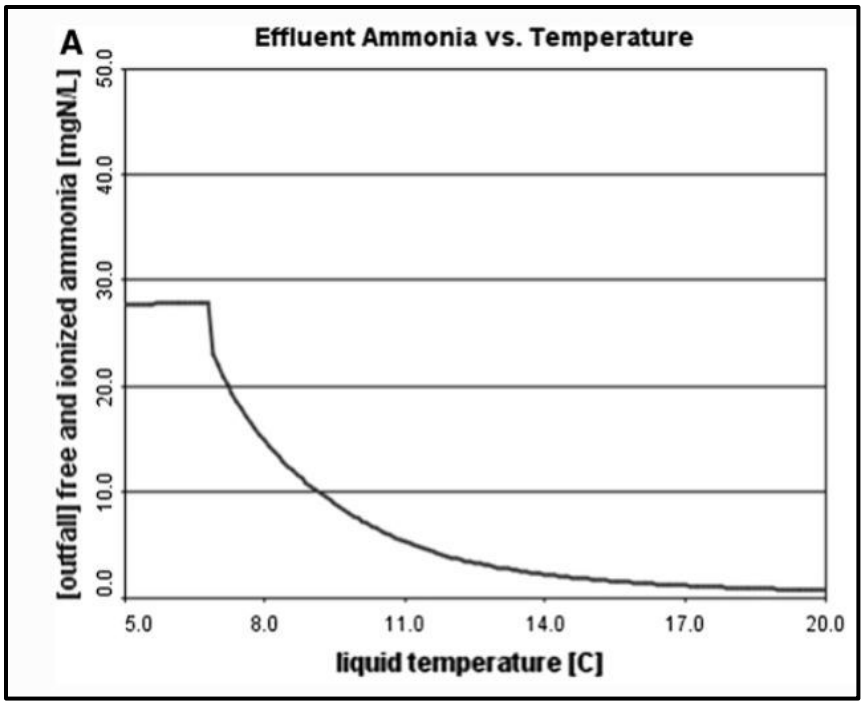

Figure 2.2: Concentration change in effluent ammonia with respect to the temperature change (Alisawi, 2020)

Metcalf et al. (1991) reported specific observations based on studying the impact of temperature on wastewater systems:

- With the increase in temperature as presented in Table 2.1, the concentration of dissolved oxygen decreased. Dissolved oxygen is an essential parameter for biological or secondary treatment. Dissolved oxygen also impacts the growth of microbial community.

Table 2.1: Temperature versus dissolved oxygen concentration (Metcalf et al., 1991)

TEMPERATURE (°C)	CONCENTRATION OF DO (mg/L)
0	14.6
5	13.1
10	11.3
15	10.1
20	9.1
25	8.2

- Anaerobic digestion and the methanogenesis process are highly sensitive to the variation of temperature. The methane production is increased with the increase in temperature (Metcalf et al., 1991).

Zouboulis et al. (2015) observed that warmer temperatures impacted various primary and secondary treatment processes. Higher wastewater temperatures elevated the reaction rate of the microbial community and reduced the density of the settled sludge. The waste activated sludge (WAS) was required to be thickened to initiate proper digestion.

SRT is considered an essential parameter for the secondary plant. Usually, the treatment plant runs at a constant SRT throughout the year. It was observed that temperature changes during the year could impact the SRT variation. Similar effluent characteristics were observed at lower SRT in summer days with respect to higher SRT in winter days. The outcomes of this study could be used to optimize the WWTP's operational cost (Shahzad et al., 2015).

2.2.2 Impact of precipitation on the wastewater treatment system

O'Neill II (2010) mentioned that the wastewater infrastructure is impacted in various ways such as:

- Reduction in water consumption leads to a reduction in water flowing in the wastewater conveyance system. But this reduction in flow does not reduce the waste loading rate. This condition is favourable for corrosion of the wastewater conveyance units. It also deteriorates the effluent quality.
- If the precipitation rate is higher, the water infiltrates into the sewers. This infiltration causes damage to the sewers (Hughes et al., 2021).
- The effluent criteria become stricter if the water flow is less than the base flow through the sewers in a drought-affected region. Baseflow is the minimum flow that the wastewater treatment plant should receive to determine the effluent characteristics.

Mines (2007) also mentioned that the intensity of rainfall determines the concentration of BOD and TSS in the influent. In this study, it was observed that with an increase in the influent loading of BOD and TSS, the effluent values also increase, respectively. With the rise in water flow in the wastewater treatment (due to rainfall), the influent BOD and TSS showed a drop in their values.

Due to heavy rainfall, the concentration of activated sludge (AS) in the aeration tank was reduced by 50 %. In contrast, the loading of sludge was surged up by ten times. Due to increased organic loading due to fluctuation in the rainfall, the process of denitrification in the secondary clarifiers formed a gel-like layer on its surface. The maintenance cost rises to remove the scum layer from the clarifier (VO et al., 2014).

The wastewater infrastructure, supply of hygienic water, basic amenities such as electricity, transport are extensively damaged due to floods. Also, there is a great incursion of salts in the wastewater systems. It increases the salt load on the design, and hence, it becomes more difficult to remove the salts from the system (Aboulnaga et al., 2019).

Due to the high volume of the influent, sometimes the untreated volume of the wastewater is to be discharged. The mixed liquor suspended solids (MLSS) and the dissolved oxygen in the wastewater showed a significant drop due to the increase in the inflow of the wastewater. The SRT and the pumping capacity at the influent were significantly reduced to reduce the burden on the treatment facility and meet the required effluent standards. Also, the aeration requirement needs to be changed according to the change in the characteristics of the inflow (Langeveld et al., 2013).

2.2.3 Impact of sea-level rise on the wastewater treatment system

Sea rise is likely to occur based on climate models from around 0.35 m to 0.9 m by 2081-2100. The groundwater table is also expected to increase with the sea-level rise. The sea-level rise would eventually change the coastal profile. It would impact the wastewater treatment facilities adversely (Amador et al., 2014). Due to the flooding of the WWTP (due to sea-level rise), the untreated waste in the water ecosystem could result in monetary loss for the municipalities and endanger public health (Zouboulis et al., 2015). It may also result in the contamination of the water bodies where it is discharged.

VO et al. (2014) indicated some of the impacts due to sea-level rise as:

- Heavy demolition of the treatment plant units could occur.

- Disruption in the wastewater processes due to power cut-off or the dripping of hazardous chemicals in the treatment units.
- Closure of the discharge of the sewage in the water bodies.

2.3 Other impacts of climate change on wastewater treatment systems

Pocock & Joubert (2017) mentioned the financial impacts. Some of them are as follows:

- If there is flow reduction/change in the wastewater volumes due to variation in the precipitation patterns, energy consumption and the use of chemicals are adversely impacted.
- The significant changes are observed in the WWTP are observed at the pump stations. There is a reduction in energy consumption with a decrease in the hydraulic load. However, the organic load remains the same. Hence, the aeration requirements do not decrease; instead, they increase for lower hydraulic loading.
- The chemical usage may increase or decrease based on the influent wastewater flow pattern. For instance, chlorine consumption for the disinfection process may increase during the low flow condition to improve the disinfection process for high strength wastewater (to meet the effluent criteria).
- The chlorine usage increases with the occurrence of odors and the growth of the filamentous bacteria.
- Certain chemicals such as polymers, lime, alum, and ferric hydroxide are used to enhance settling properties. The consumption of these chemicals is likely to increase.

- Wastewater treatment infrastructure is designed for a minimum of 10 years. If there is flow fluctuation, the infrastructure could have been economized and saved around 40 % of the capital expenditures.
- The SRT could be reduced according to the flow, saving a high operational cost.
- The carbon footprint out of the WWTP could significantly be reduced based on the flow pattern.
- Also, if the occurrence wet weather conditions become frequent, the wastewater infrastructure and plant operation could be modulated accordingly.
- The concentration of contaminants such as pharmaceuticals and endocrine-disrupting compounds (EDCs) is likely to increase with a variation in the wastewater flow.
- In the water bodies, the frequency of algal blooms (eutrophication) increases during the summer season.

2.3.1 GHGs emissions from the WWTP

According to a research study, around 20% of carbon footprint in the WWTP is generated by fossil fuel. There are basically two types of activities responsible for GHGs emissions: Onsite activities such as wastewater and sludge treatment; and offsite activities such as energy requirements, transportation, and chemical usage (Shahabadi et al., 2009). Similarly, other GHGs such as methane and nitrous oxide were also emitted from the WWTP during conveyance and various treatment processes. The reuse of the treated wastewater was costlier than the conventional use and contributed more to the production of GHGs. The anaerobic treatment of the sludge was suggested to reduce the GHGs emission by 23-55% and is helpful in biogas production (VO et al., 2014).

A Norwegian study considered the change in the sewage temperature and simultaneous removal of nitrogen and TSS in the winter months. When we observe the electric consumption cost, the study shows that the price increases by 16% if the ambient temperature dropped below -1.5 C. Furthermore, the increase in the melting period due to an increase in the ambient temperature presented higher consumption of electricity (Plósz et al., 2009).

Nitrous oxide (N_2O) and nitric oxide (NO) are produced by disintegrating nitrogen products such as urea, ammonia, and protein. The aerobic process (nitrification) and anaerobic process (denitrification process) take place during biological nutrient removal. The aerobic process contributes to fewer emissions. At the same time, all the anaerobic processes contribute nearly 50-80% of the total WWTP GHGs emissions (Zouboulis et al., 2015).

Hydrogen sulphide (H_2S) is discharged from the anaerobic decomposition of biomass. The combination of H_2S and CH_4 + CO_2 creates a corrosive environment in the sewer pipes. Another emerging issue is volatile organic compounds (VOCs). It is matter of concern for both industrial and municipal wastewater (Zouboulis et al., 2015). Table 2.2 mentions various GHGs emitted from different treatment processes such as primary, secondary, tertiary, sludge management and effluent discharge.

Table 2.2: GHGs emissions from various treatment processes (Zouboulis et al., 2015)

Process	Expected GHGs Emissions
Primary	None
Secondary	CH_4 from anaerobic treatment (i.e., lagoons)
Tertiary	N_2O from NDN process
Solids Handling	CH_4 from sludge handling such as digestion or from incomplete combustion of digester gas and emissions from offsite operations
Effluent Discharge	N_2O from the denitrification process of nitrogen species originating from wastewater effluent in receiving water

2.4 Modelling of WWTP

It is very costly to perform experiments for various scenarios based on different hydraulic loading conditions and other variable parameters (influent wastewater characteristics, temperature, pH) (Bixio et al., 2002; Butler et al., 1995). Various models are developed for the WWTP to optimize the treatment processes and to predict future scenarios for the efficient operation of the treatment plant. Some models could also determine the operational cost, chemical dosing, electric consumption by each process, aeration requirements, calculation of retention time considering various kinetic and stoichiometric parameters. Risk and uncertainty analysis could be done based on critical parameters such as catchment area, design parameters for retention tanks, along with rainfall and temperature change profile (Martin & Vanrolleghem, 2014).

White modelling and black modelling are different types of modelling that are usually used. White modelling focuses on the design, education of concepts and steps relating to optimizing various processes. This modelling is widely applicable to the community WWTPs. Its emphasis is on the biological nutrient removal in the AS system. International

Water Association (IWA) introduced AS model (ASM) families which includes ASM 1, ASM 2 and ASM 3, which provided a benchmark for the modellers to define and calibrate with other datasets. The Black-box model helps determine the influent loads, estimate the biomass activities, and study the effluent characteristics. There are some steps required to define a WWTP model: Firstly, the purpose of the model is defined. Then, various units of models are selected (AS model or sedimentation model). The next step is to estimate the hydraulic load for the WWTP.

Further, the next step in developing the white box WWTP model is to specify the wastewater parameters and biomass sedimentation. All the data are set up for a steady-state condition. After that, the calibration of all the ASM parameters is done. A check is done after that to confirm that the stated parameters and data are adequate and correct. Hence, the scenarios are simulated. These are the steps required to set up an ASM model (Gernaey et al., 2004).

Gernaey et al. (2004) stated that data acquisition is an essential step for calibrating the model. Different data are required such as reactor volume, pump flow rates, and aeration capacities (design data). Next, the operational data includes flow rates (average or dynamic), influent, effluent, wasting (waste activated sludge), pH, temperatures, and aeration. These data are followed by a tracer test and sedimentation test to characterize hydraulic and settler model parameters, respectively. The concentration of the influent and effluent (along with intermediate process) are categorized as the suspended solids (SS) (further characterized into total suspended solids (TSS), volatile suspended solids (VSS) and inert suspended solids (ISS)), COD, NH_4-N, NO_3-N, PO_4-P and, other wastewater parameters After that, the sludge composition is studied. Reaction rates and reaction

stoichiometry are considered. Data collection considers the sensitivities of the process variables with respect to the model parameters. Hence, data collection is a critical step in the modelling of an ASM.

There were other procedures than white-box methods, such as the black-box model. The black-box method was based on the input and output data and indicated the sludge sedimentation problems in the WWTPs. Other models such as Box Jenkins, artificial neural networks (ANN), multivariate statistical methods (MVS, a black-box model) are also used. The hybrid combination of the white-box model and the other models could resolve many issues and provide process optimization (Gernaey et al., 2004).

BioWin ASM is one of the numerical modelling software, which is used to account for biological nutrient removal, to treat industrial wastewater, and remove heavy metals too. BioWin has many complex features for modelling pH, ammonia stripping process, and conducting simulation for industrial wastewater. The domestic wastewater parameters are automatically considered in BioWin. In contrast, parameters characterization for the industrial wastewater is required to be inputted. BioWin was calibrated by adjusting some stoichiometric and kinetic parameters in the steady-state condition. Validation could be done in the dynamic state. Hence, BioWin optimized the WWTP processes about COD and ammonium in this study (Elawwad et al., 2019).

Xu et al. (2020) observed that the ASM of BioWin includes 78 kinetic parameters and 54 stoichiometric coefficients. Parameters such as maximum specific growth for ordinary heterotrophs (OHOs), growth and decay of nitrite-oxidizing organisms and phosphorous accumulating organisms (PAOs) are required to be calibrated for estimating the biological nutrient removal (BNR). In this study, sampling was conducted to assess the authenticity

of the BNR model of BioWin. Experimental results are quite similar to the simulated results generated from BioWin. The concentrations of NH_4-N and orthophosphate (OPO_4) were studied in this study and, the simulated results were found similar to the field results.

Proper calibration delivers better results. In the biggest WWTP in Egypt, Gabal El-Asfar, adequate calibration, and validation gave positive results for nitrification and denitrification processes. To study the impact in different zones, SRT, DO, return activated sludge (RAS) flows, and the waste activated sludge varied. The increase in SRT and changing the aeration condition (aerobic to anoxic) benefitted the denitrification process. ASMD could predict the dynamic behaviour of the WWTP (Elawwad et al., 2019).

Many studies have been conducted on climate change, impact of wastewater temperature change on the treatment processes, impact of varying SRT on the biological systems and, impact of influent flow on the wastewater treatment systems. This study is unique in a way that it used numerical modelling and assessed the carbon and BNR removal for various biological systems. Research has not been conducted using numerical modelling as a tool and to study the simultaneous impact of varying SRT (as a control parameter) along with varying wastewater temperature and influent flow. This study attempts to address the simultaneous impact of the wastewater temperature and influent flow along with varying SRT on various biological AS treatment systems. This study would also try to deliver the critical operational parameters (SRT) required for the optimization of the WWTPs and better efficiency of the WWTPs. Data from various treatment plant was studied and analyzed. The influent flow conditions were classified from this analyzed data and further, this influent flow was used to assess the combined impact of change in wastewater

temperature and influent flow on the biological treatment systems .From the literature review , it was observed that this kind of analysis was done before.

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Chapter 3: Impact of wastewater temperature and SRT changes on activated sludge processes

Abstract

Wastewater treatment processes are likely to be impacted by projected changes in wastewater temperatures under climate conditions. Warmer temperatures may increase the wastewater temperature in hot seasons but may also lower the wastewater temperature during snowmelt. This study investigated the impact of climate change driven temperature changes on the performance of the conventional activated sludge (AS) and the Ludzack-Ettinger systems and their effluent quality using numerical modelling. The impact of the solids retention time (SRT) was also investigated as a process control and climate change adaptation tool. Treatment performance and final effluent quality were assessed with total COD, cBOD, TSS, and total ammonia concentrations. The kinetics of the microbial community depends on temperature based on Monod's equation. Hence, effluent total COD, cBOD, TSS, and total ammonia concentrations were all higher at colder temperatures and lower at temperatures above 13 ℃. The results showed an overall increase in COD and TSS and a decrease in cBOD and ammonia with increasing SRT values. Results of this study provide insights into the potential impact of climate change on secondary treatment processes and can help with the optimal operation of WWTPs under a changing climate.

Keywords: climate change, biological treatment, wastewater temperature, SRT, effluent quality.

3.1 Introduction

Effective wastewater treatment is fundamental to safeguarding human health and the natural environment. Wastewater treatment plants (WWTPs) are critical infrastructures that remove deleterious substances from wastewater before it is reused or discharged into the environment. In Canada, wastewater systems effluent regulations came into force in 2012 and require secondary treatment to achieve mandatory minimum effluent quality standards (cBOD ≤ 25 mg/L, TSS ≤ 25 mg/L, total residual chlorine ≤ 0.02 mg/L, un-ionized ammonia < 1.25 mg/L expressed as nitrogen (N), at 15 ℃ ± 1 ℃). Previous to the federal regulations, primary and secondary treatment systems were used in most municipalities, with some facilities providing additional tertiary treatment (Environment Canada, 2011).

The Intergovernmental Panel on Climate Change report (Masson-Delmotte et al., 2018), the U.S. Fourth National Climate Assessment Report (Reidmiller et al., 2018), and Canada's Changing Climate Report (Bush and Lemmen, 2019) indicated that the recently observed trends in warming levels are similar to the best estimate of the human-induced global warming. Urban infrastructures face growing challenges such as aging, deterioration, and extreme weather events that can be exacerbated by climate change, which will negatively impact the urban quality of life (Reidmiller et al., 2018). For instance, climate change poses a significant challenge to the urban wastewater treatment systems, which are susceptible to projected climate change-related long-term trends and greater fluctuations in temperatures and precipitation (Anastasios, 2015). Other examples of climate change impacts are risks associated with sea-level rise including flooding, storm

surges, erosion, and saltwater intrusion, which can impact wastewater system reliability and operating costs (Langeveld et al., 2013).

A significant percentage of the epidemics in the world is due to a lack of adequate sewage system and sanitation (Mara et al., 2010). WWTPs play an important role in preventing epidemics and contamination of the water ecosystem. Secondary treatment is an important component of the WWTP. Secondary treatment is the biological treatment in the WWTP, which includes the removal of biomass by the degradation of the biomass by the microbial community. Temperature and pH play a vital role in the biological activity of the microbial community. The removal of organics and biological nutrient removal are the main functions of the secondary treatment (Sperling Von, 2007). Hence, it is essential to investigate the potential impact of climate change on secondary treatment systems.

A warmer climate can accelerate the reaction kinetics in the summer and impact the biological and chemical treatment of wastewater. In spring, faster and more frequent snowmelt can decrease the wastewater temperature and affect the growth of microorganisms in biological treatment (Ghanizadeh et al., 2001). Rainstorms can increase the natural organic matter content in the influent, which can decrease the efficiency of the coagulation and flocculation processes and increase turbidity, while drought conditions can exacerbate the influent quality in WWTP and increase the wastewater strength (Kim et al., 2017). An increase in rainfall intensity can also increase the pollutant concentrations, floatable materials, and sediments in grit tanks and primary settling tanks (Vo et al., 2014). Activated sludge (AS) processes can be affected by climate change driven fluctuations in wastewater influent temperature and discharge (Alisawi, 2020; Butler et al., 1995; Plósz et al., 2009; Shahzad et al., 2015; VO et al., 2014; Zouboulis et al., 2015; Hughes et al., 2021).

Plósz et al. (2009) studied the wastewater influent temperature in wintertime and assessed its impact on the biological nitrogen removal and secondary clarification processes. They demonstrated that the sewer overflows during the melting and the precipitation periods adversely impact the biological processes and the operation of the treatment plant.

As the climate conditions change, there is a need for methodological approaches to understand the climate change driven impacts, assess the magnitude and significance of these impacts, and identify adaptation approaches. This will help to minimize the impact of climate change on wastewater treatment infrastructure and ensure that wastewater treatment goals and discharge limits are met. Modelling, as an integral part of the design of wastewater treatment systems, provides a tool to investigate the potential impacts of climate change on wastewater treatment plants.

The main objective of this study is to investigate the impact of temperature changes on the performance of conventional AS and Ludzack-Ettinger biological treatment systems using numerical modelling. These two processes were chosen as they are the most commonly used AS processes at wastewater treatment plants. The role of the SRT as a climate change adaptation tool for process control, was also investigated. The outcome of this study provides an insight into the operational challenges that WWTP may face under climate change.

3.2 Materials and Methods

BioWin is a widely used numerical model to simulate the wastewater treatment processes based on the AS models developed by the International Water Association (IWA) (Gernaey et al., 2002). Kinetic and the stoichiometric parameters considered for the simulations were adopted from the IWA manual and BioWin manual. These manuals were based on the

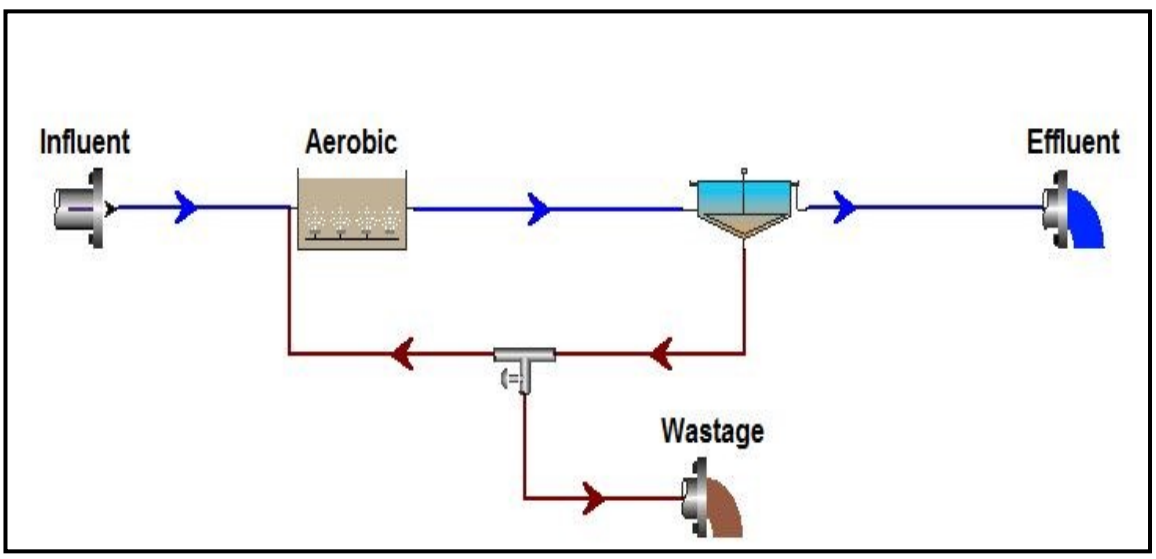
activated sludge numerical modelling book. The values for these parameters were similar as the values are considered in the field observation. Hence, to develop and assess an similar condition as it is observed in the field, these parameters were used to obtain results. All the parameters were well within the range. Table A1 in the appendix specifies the parameters considered for the simulation. For both processes, wastewater temperature range of 5 ℃ to 25 ℃, and SRT range of 3 to 11 days were examined to cover both the low and high values. BioWin, as default for the pilot example, provided values for the total COD (chemical oxygen demand), cBOD (carbonaceous biochemical oxygen demand), TSS (total suspended solids), and ammonia concentrations for the influent to the AS system. The data for the influent parameters were also obtained from a wastewater treatment plant to ensure that they were representative.

In this study, the simulation does not include the primary treatment and focuses on the secondary treatment only. Therefore, the average removal percentages for the wastewater parameters during primary treatment were obtained from the literature (Tchobanoglous et al., 2003). The removal rates in primary clarifiers were estimated to be 48% for COD, 59% for cBOD, and 68.7% for TSS. Correspondingly, the secondary influent was assumed to have 52%, 41%, and 31.3% of the raw sewage COD, cBOD, and TSS concentrations, respectively. The secondary influent wastewater characteristics used for the numerical modelling are presented in Table 3.1.

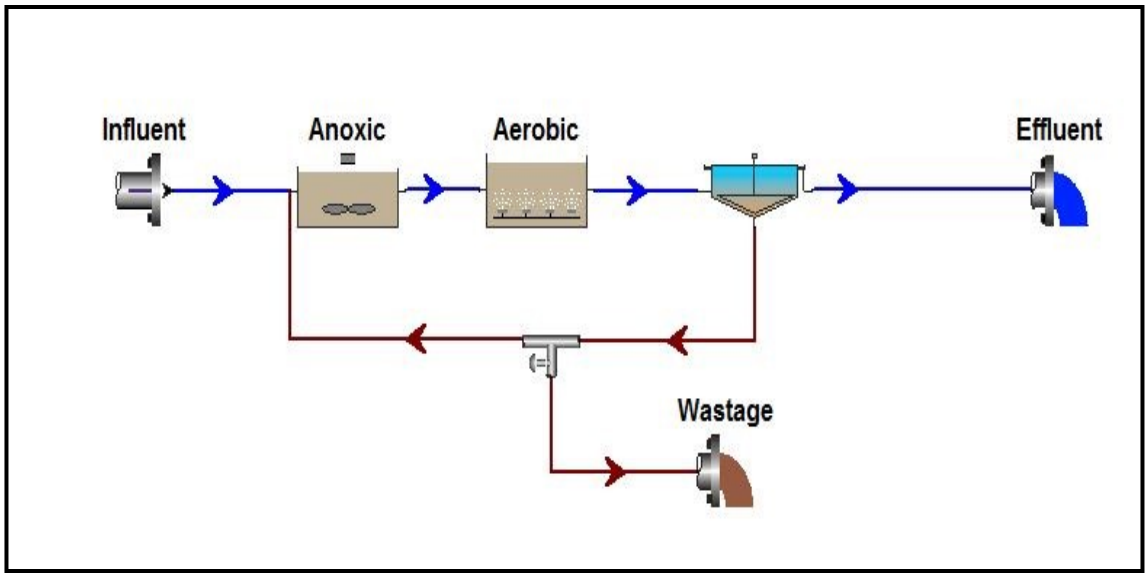
Table 3.1: Influent data for wastewater secondary biological treatment system

PARAMETER	VALUE AND UNIT
Flow	100,000 m^3/day
Total COD	275 mg/L
cBOD	130.60 mg/L
TSS	138.18 mg/L
Total ammonia	26.4 mgN/L

Two configurations of the AS system were considered. The first one is the conventional process for the AS system with an aerobic zone followed by a clarifier. There is a return-activated sludge (RAS) stream from the clarifier to the aerobic tank, and the rest is considered as waste-activated sludge (WAS), which is represented in Figure 3.1(a). The second configuration is the Ludzack-Ettinger process. This configuration comprises an anoxic zone preceding the aerobic zone and the return sludge is directed to the anoxic zone. This configuration is depicted in Figure 3.1(b).



(a)

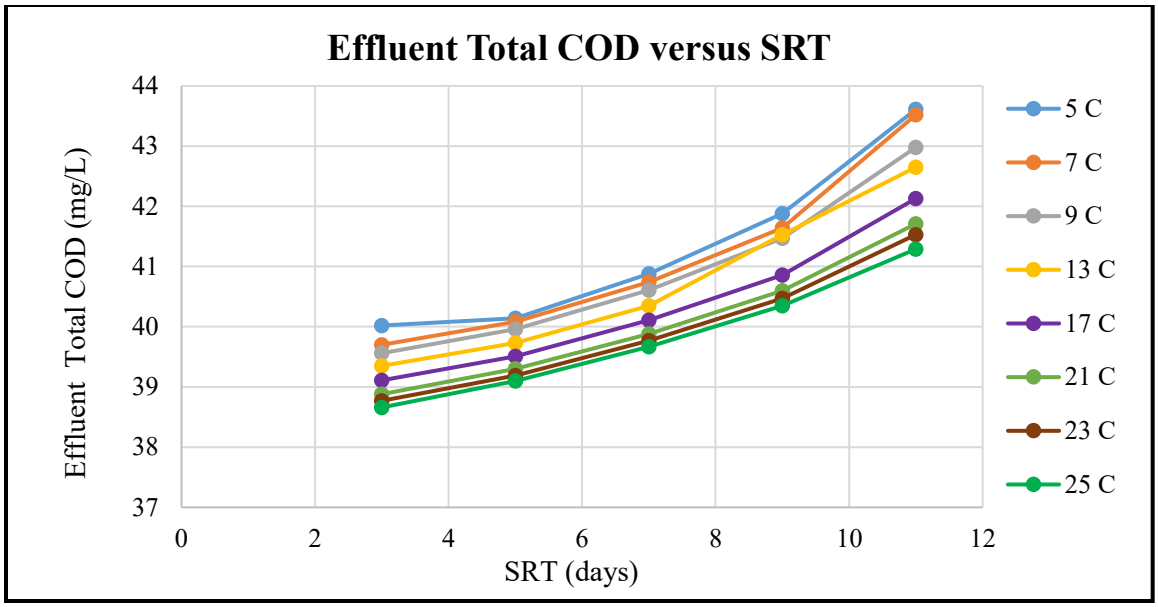


(b)

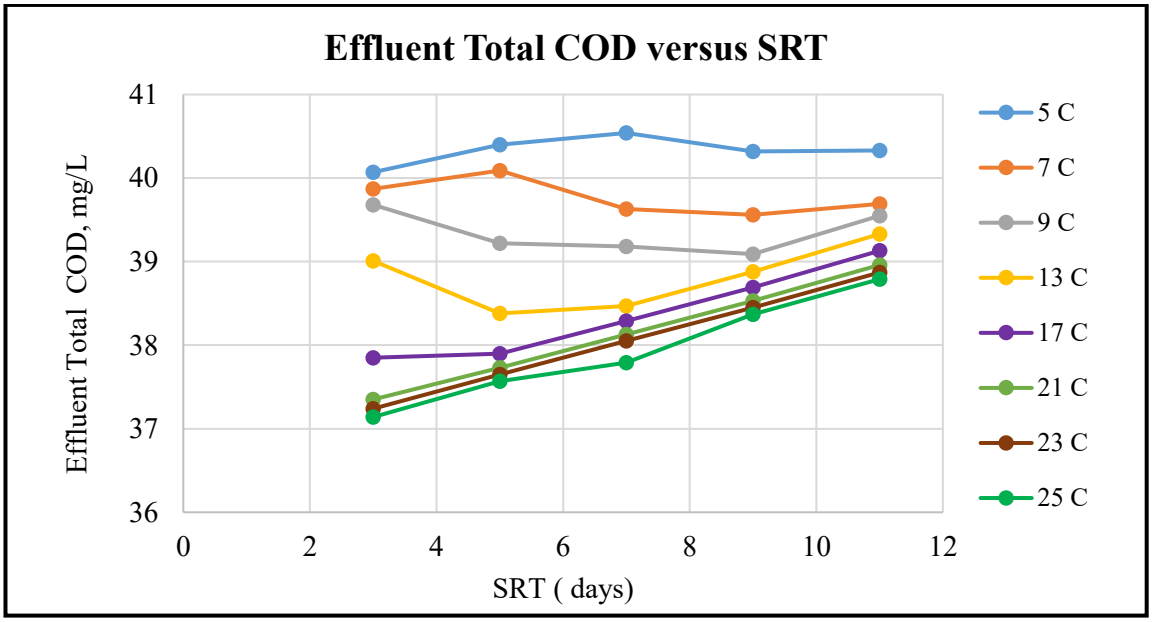
Figure 3.1: Schematic view of two processes considered in this study: (a) conventional AS (b) Ludzack-Ettinger

3.3 Results and Discussion

The simulations for the AS and Ludzack-Ettinger processes were conducted. The AS process mainly removes carbon and the Ludzack-Ettinger process removes carbon and nitrogen. The results are discussed for the effluent concentrations of total COD, cBOD, TSS, and ammonia. Several scenarios were simulated with the temperature range of 5 to 25°℃ and SRT range of 3 to 11 days. The same influent characteristics were used for both processes. The results are presented as total effluent COD, cBOD, TSS, and ammonia concentrations against the SRT range at various temperatures. The effluent parameters are reported in mg/L and SRT in days. The results offer an insight into the variations observed in the effluent parameters with changes in wastewater temperature and SRT.



(a)



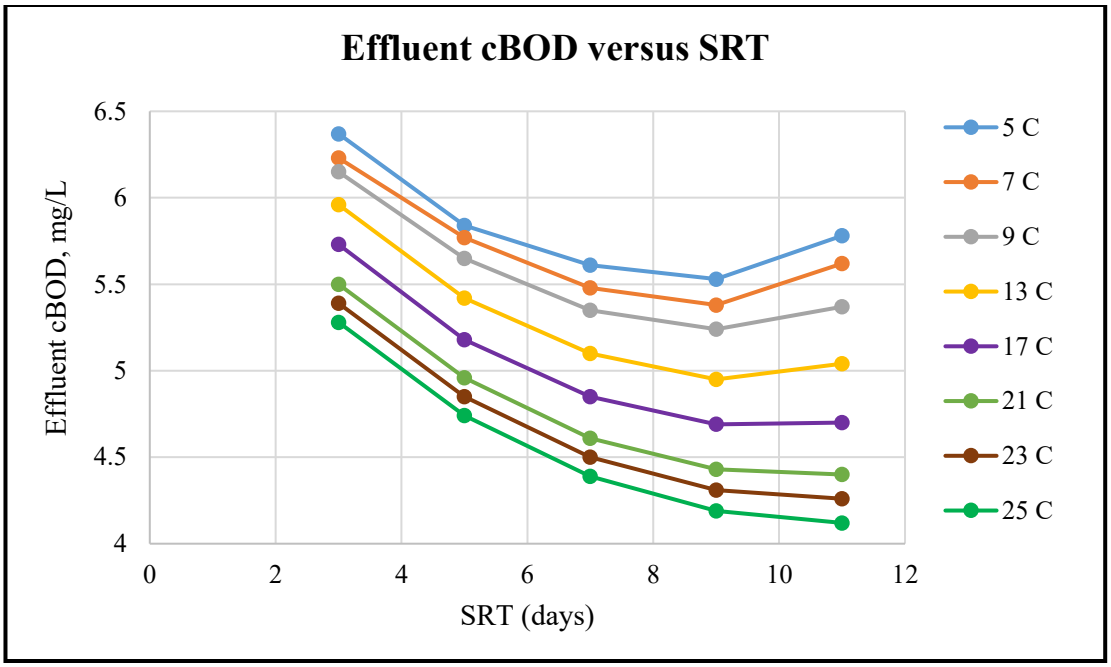
(b)

Figure 3.2: Effluent total COD for (a) conventional AS process and (b) Ludzack- Ettinger process.

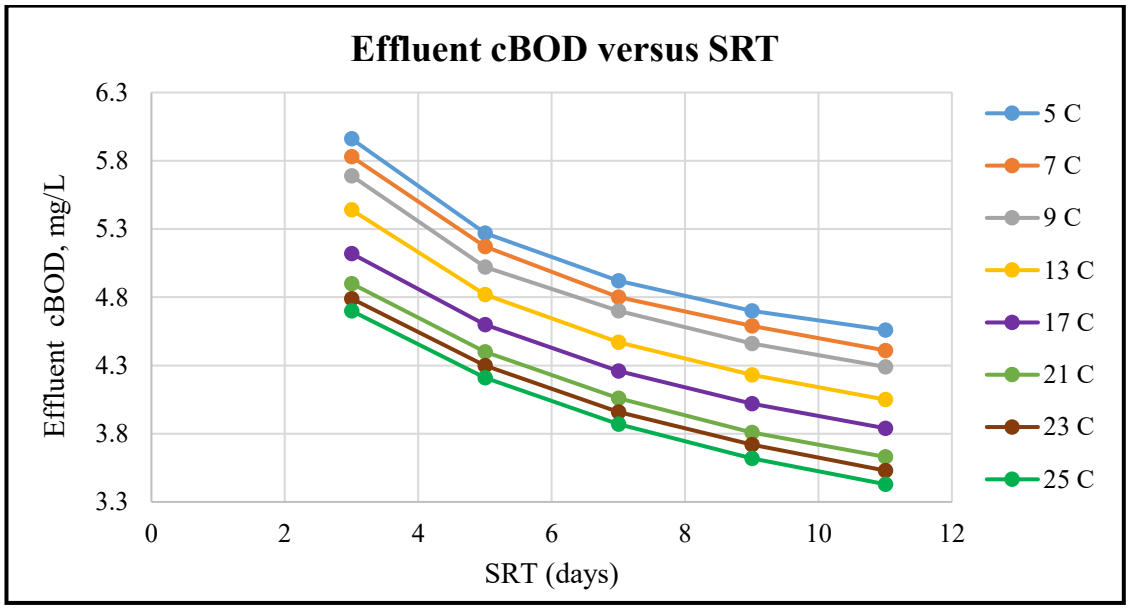
The results from the conventional AS process are presented in Figure 3.2(a). Increasing the wastewater temperature decreased the effluent COD due to the increased biological reaction rates under warmer temperatures and improved treatment performance.

Accordingly, colder temperatures, which may occur due to snowmelt in cold countries, increased the effluent COD concentrations. At an SRT of 5 days, the difference in effluent COD was approximately 1.5 mg/L between 5 and 25 ℃. At an SRT of 11 days, the difference in effluent COD increased to approximately 2.5 mg/L between 5 and 25 ℃. Increasing the SRT of the AS process increased the effluent COD concentrations. At an SRT of 3 days, the effluent COD was 38.5 mg/L increased to 41.3 mg/L at 25 ℃. In an experimental study with laboratory-scale AS reactors, an increase in SRT was reported to lead to an increase in effluent polysaccharides due to the release of extracellular polymers, which increased the effluent COD (Phillips, 1998). These results indicate that temperature caused changes in effluent COD can be compensated by adjusting the SRT values.

The effluent COD concentrations for the Ludzack- Ettinger process in Figure 3.2 (b) showed similar trends when wastewater temperature was warmer than 17 ℃ but deviations were observed at colder temperatures (<13 ℃). Nitrifiers are known to be very sensitive to cold temperatures and these deviations can be explained by the system perturbations in cold temperatures (Zhang et al., 2019). At an SRT of 5 days, the effluent COD was around 37.5 mg/L at 21, 23 and 25 ℃, but quickly increased up to 40 mg/L at colder temperatures down to 5 ℃. Increasing the SRT linearly increased the effluent COD at higher temperatures, but in the low temperature range (5, 7, and 9 ℃), the linearity was lost and in fact the COD values remained relatively the same regardless of the SRT. Cold temperatures also inhibit the biological activity of the heterotrophic bacteria, which play a key role in COD removal.



(a)

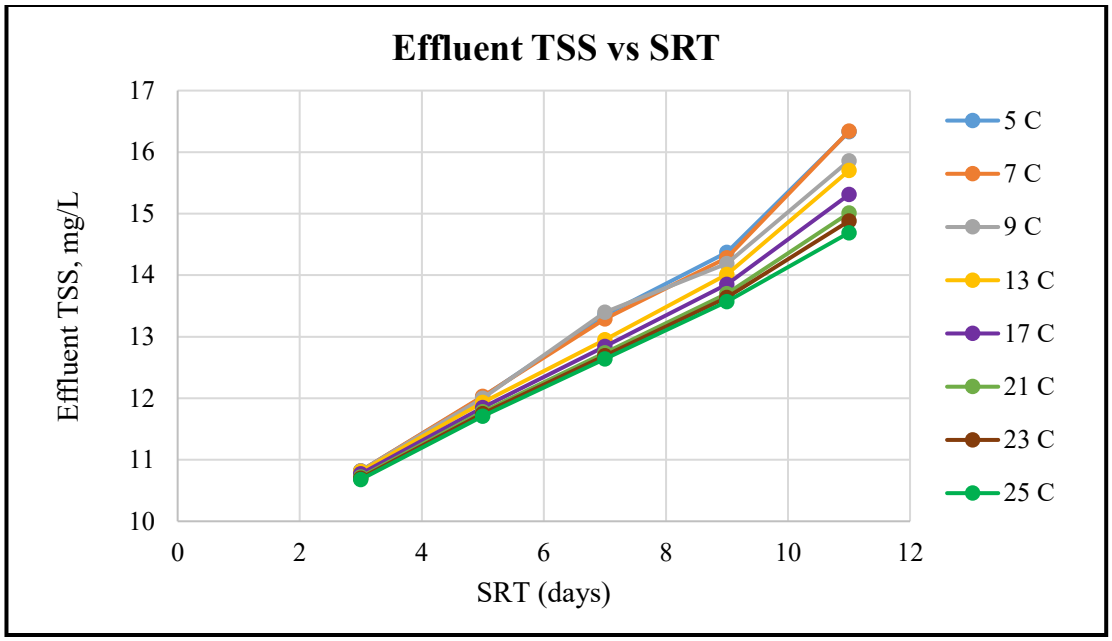


(b)

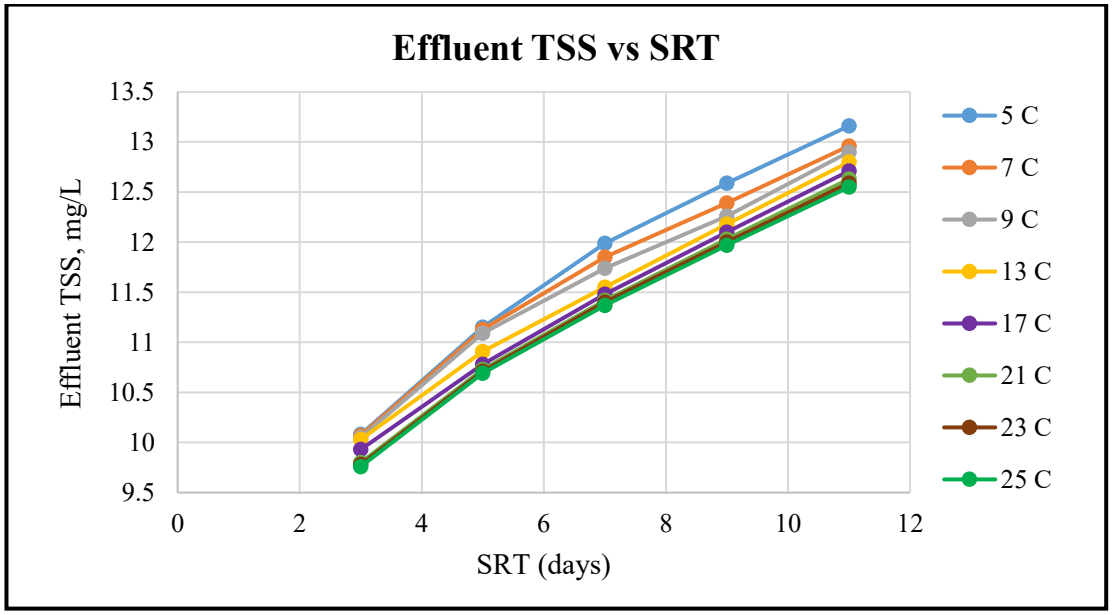
Figure 3.3: Effluent cBOD for (a) conventional AS process and, (b) Ludzack-Ettinger process.

Figure 3.3 shows the cBOD concentrations in the secondary effluent. In the AS and Ludzack- Ettinger processes, increasing the SRT resulted in a decrease in the effluent

cBOD despite the observed increased in COD values. This was also seen in other studies. Phillips (1998) also observed the same phenomenon and reported that the increase in effluent COD was not observed in effluent BOD mainly due to the resistance of the released extracellular polymers to biodegradation. For both systems, increasing the wastewater temperature decreased the cBOD concentration in the effluent due to the increased biological activity in wastewater. For the AS system, at an SRT of 5 days, the effluent cBOD increased by approximately 1 mg/L when the temperature decreased from 25 ℃ to 5 ℃ (Figure 3.3 (a)). For the same temperatures and at an SRT of 11 days, the cBOD difference increased to approximately 1.5 mg/L. A slight increase in cBOD was observed at cold temperatures likely due to the release of intracellular and extracellular materials that were resistant to biodegradation. This increase was not observed in the Ludzack- Ettinger process (Figure 3.3 (b)).



(a)

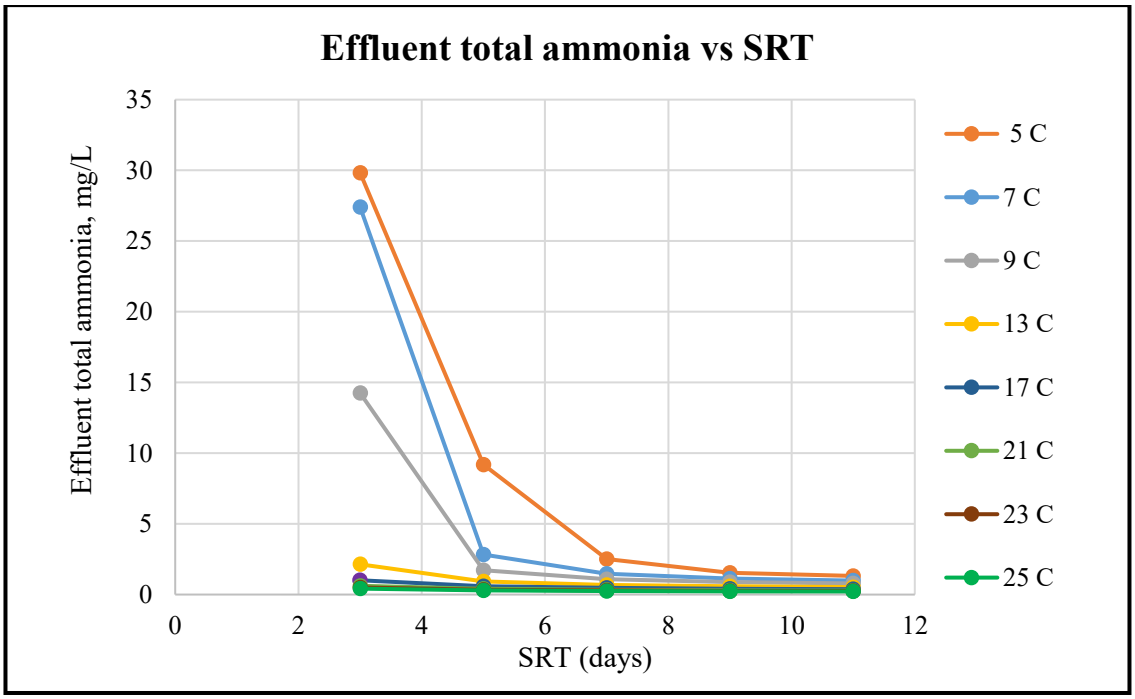


(b)

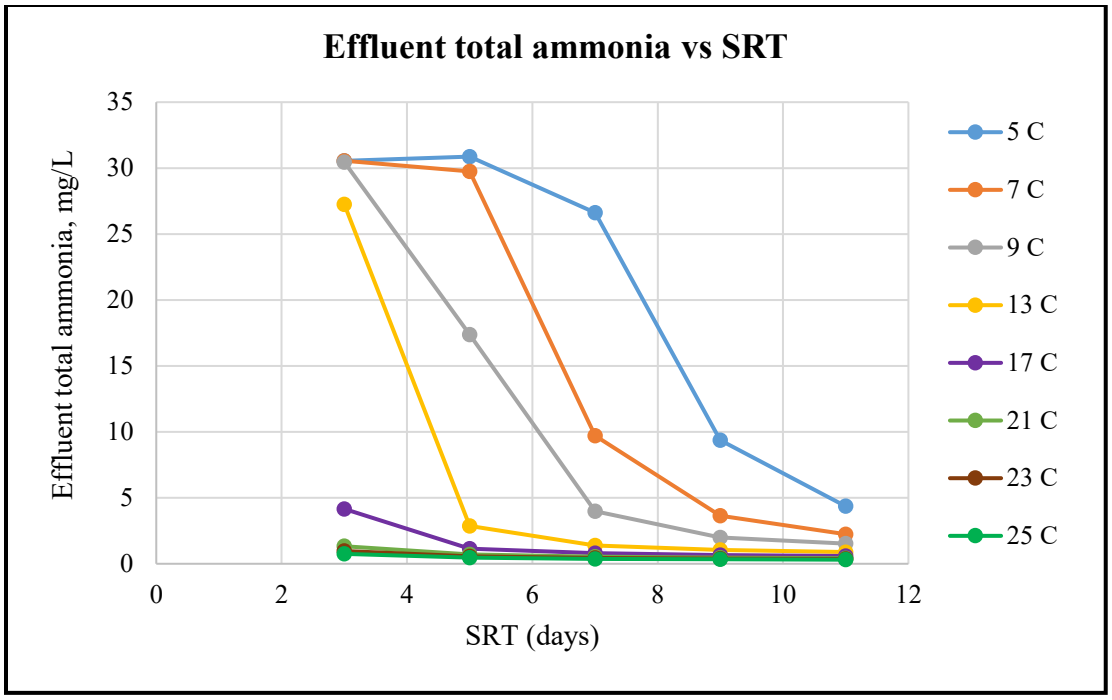
Figure 3.4 Effluent TSS in (a) conventional AS process and (b) the Ludzack-Ettinger process.

Colder temperatures resulted in higher effluent TSS concentrations for the AS and the Ludzack- Ettinger processes. Cold temperatures decrease the density and viscosity of water

and adversely impact the settling velocity of the wastewater particles. For the AS system, at an SRT of 5 days, the effluent TSS concentrations increased by 0.3 mg/L when wastewater temperature decreased from 25 ℃ to 5 ℃ (Figure3.4(a)). At an SRT of 11 ℃, the difference in TSS concentrations increased to 1.7 mg/L. For the Ludzack- Ettinger system, the TSS concentration increased by 0.4 mg/L and 0.6 mg/L at SRTs of 5 and 11 days under the same temperature decreased (Figure 3.5(b)). Extended SRT times result in filamentous bulking that adversely impacts the settling of particles in the secondary clarifiers (Smith et al., 2015). This resulted in more particles escaping the sedimentation tank and increased the TSS in the effluent.



(a)



(b)

Figure 3.5 Effluent total ammonia for (a) conventional AS process and (b) Ludzack-Ettinger process.

Temperature is an important parameter for nitrogen removal efficiency. It not only affects the activity, specific growth rate, and community structure of nitrifying and denitrifying microorganisms but also impacts the concentration of dissolved oxygen, thereby the nitrification and denitrification processes (Zhang et al., 2019). The growth of nitrifying bacteria is greatly impacted by temperature. As the growth of nitrifying bacteria is slower in colder temperatures, the nitrification process is also slower compared to higher temperatures (Zhang et al., 2019). SRT also plays an important role in nitrification and denitrification. An SRT of 4-8 days was required for nitrification and lower SRTs result in elevated ammonia in the effluent (Smith et al., 2015). Denitrification requires longer SRT times of 10-15 days and sufficiently short SRTs may result in incomplete denitrification.

Figure 3.5 shows the effluent total ammonia concentrations and how both processes were impacted by temperature and SRT changes.

For the AS system, temperatures lower than 9 ℃ resulted in high total ammonia concentrations particularly at low SRTs (Figure 3.5(a)). At an SRT of less than 4 days, nitrifiers would be washed out at a faster rate than they regenerate resulting in a lower rate of nitrification (Smith et al., 2015). At 5 ℃ and at an SRT of 3 days, the total ammonia concentration reached 30 mg/L. At low temperatures, the nitrification rate was very low or negligible as the nitrifying bacteria are inactive. When the SRT was sufficiently high, 1 mg/L of total ammonia was achieved even at 5 ℃. At temperatures greater than 17 ℃, ammonia was successfully removed especially when SRT was higher than 5 days. Disruption to nitrification and accumulation of total ammonia in the effluent adversely impacted the denitrification process. For the Ludzack-Ettinger process, effluent ammonia concentrations remained high at temperatures colder than 17 ℃ and 11 days of SRT were required to substantially lower the effluent total ammonia at 5 ℃ (Figure 3.5(b)).

3.4 Conclusion

This study investigated the effect of changes in wastewater temperature and SRT on the treatment performance of AS and Ludzack-Ettinger processes. Warmer temperatures may increase the wastewater temperature in hot seasons but may also lower the wastewater temperature during snowmelt in cold countries. The results from Bio-Win modelling showed an overall increase in total COD and TSS and a decrease in cBOD and ammonia with increasing SRT values. The rate of reaction depended on the wastewater temperature, based on Monod's equation. The colder wastewater temperatures reduced the rate of the activity of the microbial community. Hence, the effluent total COD, cBOD, TSS, and total

ammonia concentrations were all higher at colder temperatures and lower at temperatures below 13 ℃. The results of this study demonstrated that climate change driven temperature changes can significantly impact the removal efficiency of AS systems. However, these changes can be compensated by adjusting the SRT to achieve the desired effluent concentrations as illustrated in this study. Climate change will impact the operation of wastewater treatment plants in the future, and even though some of these changes can be managed by adjusting the operational parameters, there will be an associated increase in energy, chemical use, and cost. This study provides an insight into the operational challenges that WWTP may face under climate change.

3.5 Acknowledgments

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Chapter 4: Impact of wastewater temperature as an indicator of climate change on the biological nutrient removal systems

Abstract

Biological nutrient removal (BNR) is an important process in removing nutrients (nitrogen and phosphorus) from wastewater that cause eutrophication in surface waters. Configurations of secondary wastewater systems such as modified Ludzack-Ettinger (MLE), Phoredox, modified Bardenpho and anaerobic/anoxic/aerobic (A2O) were designed for individual or combined removal of nitrogen and phosphorus compounds. This study examines the impact of climate change, particularly wastewater temperature change, on the efficiency of BNR processes using the modelling tool BioWin. Various simulations were conducted for the temperature range of 5℃-25℃ and by varying solids retention time (SRT) from 1.5-13 days. The results of the study showed that the efficiency of modified Bardenpho system was better than other systems. Higher SRTs improved the total ammonia removal but deteriorated the carbonaceous biochemical oxygen demand (cBOD) and total phosphorous (TP) removal for most of the systems. 5 days SRT and 17 ℃ were critical parameters for all systems except the conventional activated sludge (AS) system. This study also provides an insight for the optimization of various temperature-related biological treatment processes affected by wastewater temperature under the impact of climate change.

Keywords: AS, SRT, MLE system, A2O system, Phoredox system, modified Bardenpho, temperature, F/M ratio, A2O, phosphorous accumulating organisms (PAOs),

4.1 Introduction

The municipal wastewater consists of different forms of carbon, nitrogen, and phosphorous, which can negatively impact the aquatic ecosystem in receiving water bodies. Eutrophication is when biological nitrogen and phosphorous accumulate in the water bodies and promote algal blooms in the freshwater bodies (Hilton et al., 2006). The algal blooms take up the dissolved oxygen (DO), which results in inadequate DO for the aquatic flora and fauna (West et al., 2016). This process adversely harms marine and human ecosystems (Smith et al., 1999).

It has been shown that greenhouse gases (GHGs) emissions are the primary driver of climate change (IPCC 2014; Bush et al.,2019; Reidmiller et al., 2017). The ambient climatic conditions are projected to change due to the changing climate. According to West (2016), freshwater was one of the significant contributors of CH_4, N_2O and CO_2. As the concentrations of DO decrease in freshwater, it promotes anaerobic conditions favorable for methane production (West et al., 2016). These processes promote the emissions of greenhouse gases, contributing to climate change (Li et al., 2021). As conventional AS systems are often inadequate to completely address the changes due to increased organic and nutrient loading, additional modifications and processes are required to meet effluent limits (Massara et al., 2017). Ludzack-Ettinger process, modified Ludzack-Ettinger (MLE) process, modified Bardenpho process, Phoredox process, Anaerobic/Anoxic/Aerobic (A2O) process and University of Cape Town (UCT) process are examples of different configurations in the activated sludge (AS) systems designed for the removal of COD, cBOD, as well as ammonia, phosphorous or combined removal (Esfahani et al., 2018).

Climate change and wastewater treatment systems are directly related, with climate change affecting wastewater influent characteristics and ambient climatic conditions that can affect operations of the wastewater treatment facility (Anastasios, 2015). Investigation of the impacts of climate change on wastewater characteristics, as operational units, conveyance units and all the units concerning biological processes (enhancing wastewater quality) can be impacted, is crucial to improving wastewater treatment (Hughes et al., 2021a). According to the Canadian Climate Change Report (Bush and Lemmon 2019), the average warming across Canada is twice the global warming, with even more significant warming rates in Northern Canada. In addition, due to the warmer winters, a shift from snow to rain is projected. These changes, in turn, can influence the wastewater influent temperature and flow patterns.

SRT is one of the main parameters regulating BNR performance as it determines the mixed liquor suspended solids (MLSS) concentration, nitrification rate and phosphorous removal (Ekama, 2010). In some studies, aeration rate is regulated, and its impact on carbon removal and BNR was studied (Liu et al., 2017). Specific methods focused on nitrogen removal by controlling the dissolved oxygen (DO) in the denitrification process (Capodaglio et al., 2016). Capodaglio (2016) also discussed the impact of changing the food/microorganism (F/M) ratio on the denitrification process. Other studies revealed the influence of having very short SRT from 0.5 – 4 days on carbon and phosphorous removal (Shao et al., 2020). However, previous studies have not assessed the performance of various AS systems considering different wastewater temperatures and SRT scenarios that can be induced by climate change.

This study investigated carbon and nutrient removal performance from various BNR systems under different wastewater temperature and SRT scenarios. We considered five AS systems: (i) conventional AS system, (ii) MLE system, (iii) Phoredox system, (iv) A2O system and (v) modified Bardenpho system. We employed BioWin 6.0 to numerically simulate the wastewater treatment processes under various wastewater temperature and SRT scenarios. To assess the performance of different AS systems, a wide range of wastewater temperatures 5℃ - 25℃ was considered, together with an appropriate range of 1.5-13 days SRT for different systems. This study examined the optimum usage of all treatment units to deliver the required effluent criteria. The combined effect of temperature change and varying SRT on carbon removal, nitrification, denitrification, and phosphate uptake was assessed.

4.2 Methodology

Specific numerical models such as BioWin and GPSx were used to simulate the wastewater treatment processes in WWTPs. In this study, we employed BioWin 6.0, a widely used tool developed by International Water Association (Gernaey et al., 2004). Influent kinetic and stochiometric parameters specified in Table A1 (in appendix) were considered. The kinetic and the stochiometric parameters were well within the range, mentioned in the IWA manual. Based on the climatic conditions, the wastewater temperature may vary widely during the year. The wastewater temperature range and its pattern vary across Canada due to different climatic regions. For instance, according to the data obtained from WWTPs in Ontario, a range of 6-25 ℃ was observed. The range and trend of the wastewater temperature throughout the year may change under climate change.

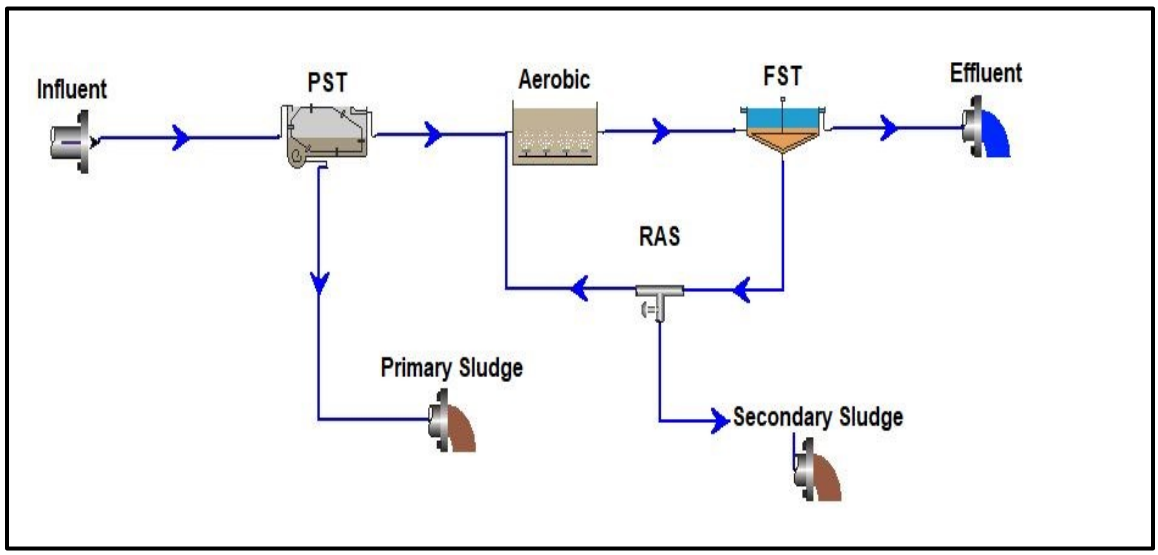
In this study, we used BioWin 6.0 to conduct simulations on five different AS systems, namely the conventional AS, MLE, Phoredox, A2O and modified Bardenpho. MLE and Phoredox systems were specifically designed to remove nitrogen compounds and phosphorous from the mainstream. At the same time, A2O and modified Bardenpho systems were designed to remove both nitrogen and phosphorous from the system. The conventional AS system comprised of a primary settling tank (PST), aerobic bioreactor, final or secondary settling tank (FST) and the effluent was ejected, as shown in Figure 4.1 (a). The return activated sludge (RAS) was recirculated in the conventional AS system. In the MLE system, the aerobic bioreactor was preceded by an anoxic bioreactor, where nitrifying mixed liquor (NML) was recycled from the effluent of the aerobic bioreactor, presented in Figure 4.1 (b). Figure 4.1 (c) shows that the Phoredox system had the same configuration as the conventional AS system, except the aerobic bioreactor was preceded by an anaerobic bioreactor. The SRT for anoxic and anaerobic bioreactors ranges from 1.2-4.37 days. Figure 4.1 (d) shows that the A2O system had anaerobic, anoxic, and aerobic bioreactors. The local SRT of each bioreactor ranged from 0.67-2.90 days. Lastly, as shown in Figure 4.1 (e), the modified Bardenpho system comprised an anaerobic bioreactor followed by two alternative anoxic-aerobic bioreactors. The local SRT for all the bioreactors in the modified Bardenpho system ranged from 0.42-2.04 days. The overall system SRT range considered for the conventional AS system was 1.5-11 days. Whereas the SRT range considered for complete MLE, Phoredox and A2O systems were 3-11 days. And the SRT range for the modified Bardenpho system was 3-13 days. SRT range of 3-13 days was selected for the modified Bardenpho system to understand the system's efficiency for higher SRT. BioWin provided the local SRT values for all the systems. Parameters such

as carbonaceous biochemical demand (cBOD), total ammonia and TP were observed and measured in mg/L.

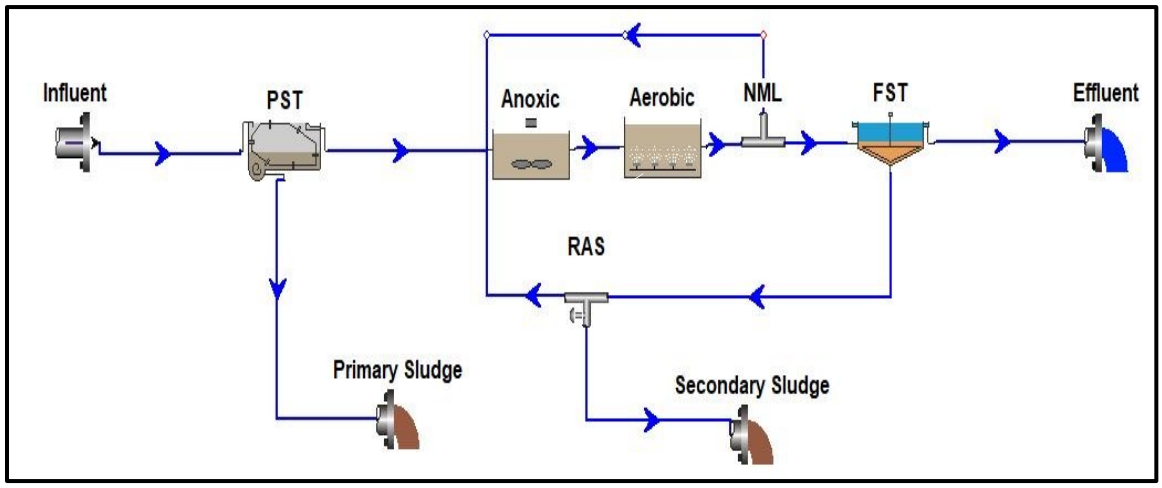
As presented in Table 4.1, the default raw influent characteristics from the BioWin software were used for the simulations. The PST was included in the simulation to account for the impacts of temperature changes on the primary treatment processes that determine the influent of the secondary treatment processes.

Table 4.1: Influent characteristics of the raw wastewater

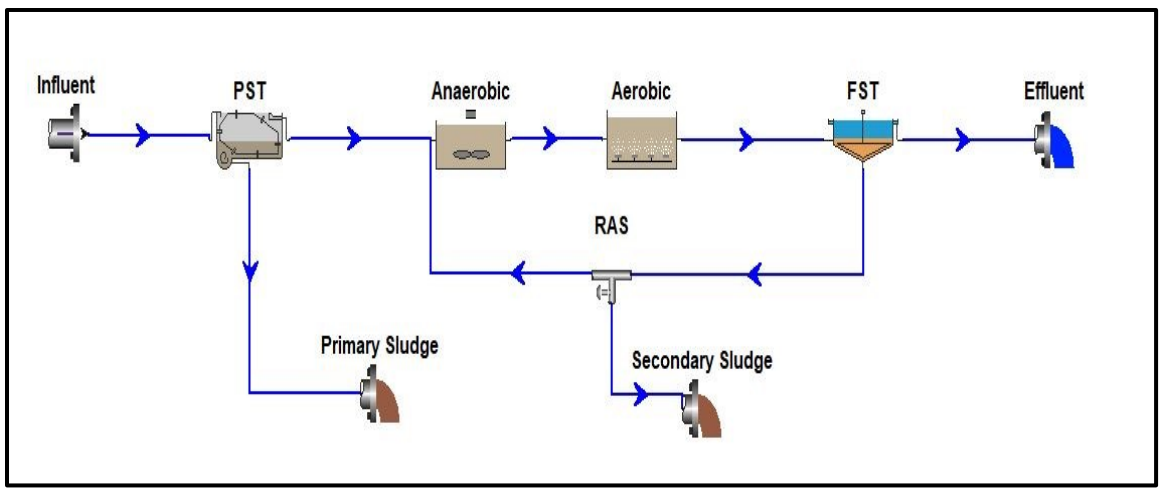
BIOLOGICAL PARAMETER	VALUE
Flow	100,000 m^3/d
Total COD	500 mg/L
cBOD	245.21 mg/L
TSS	242.71 mg/L
Total ammonia	26.4 mg/L
TP	6.5 mg/L



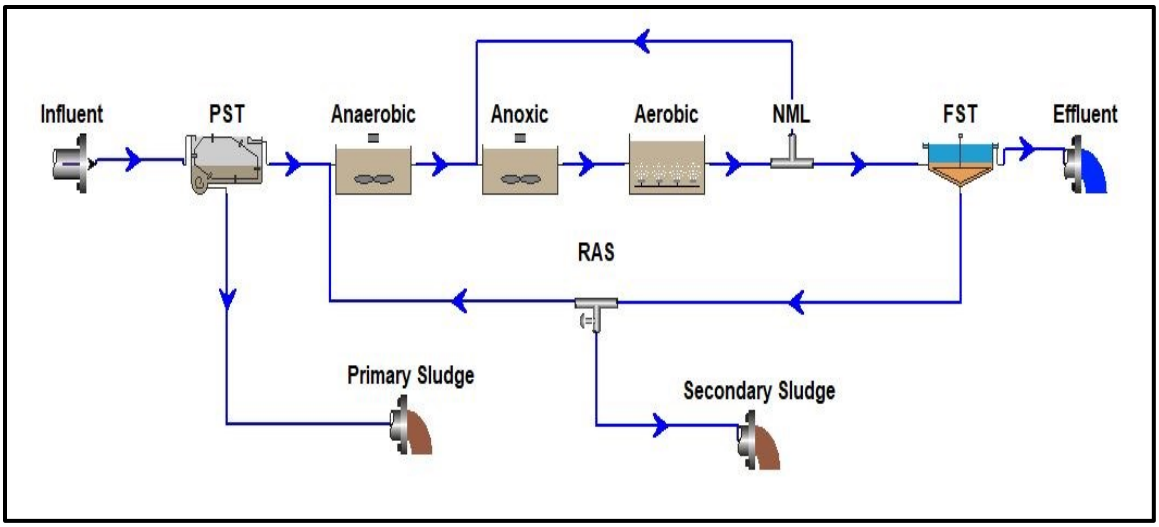
(a)


Influent
PST
Anoxic
Aerobic
NML
FST
Effluent
RAS
Primary Sludge
Secondary Sludge

(b)


Influent
PST
Anaerobic
Aerobic
FST
Effluent
RAS
Primary Sludge
Secondary Sludge

(c)



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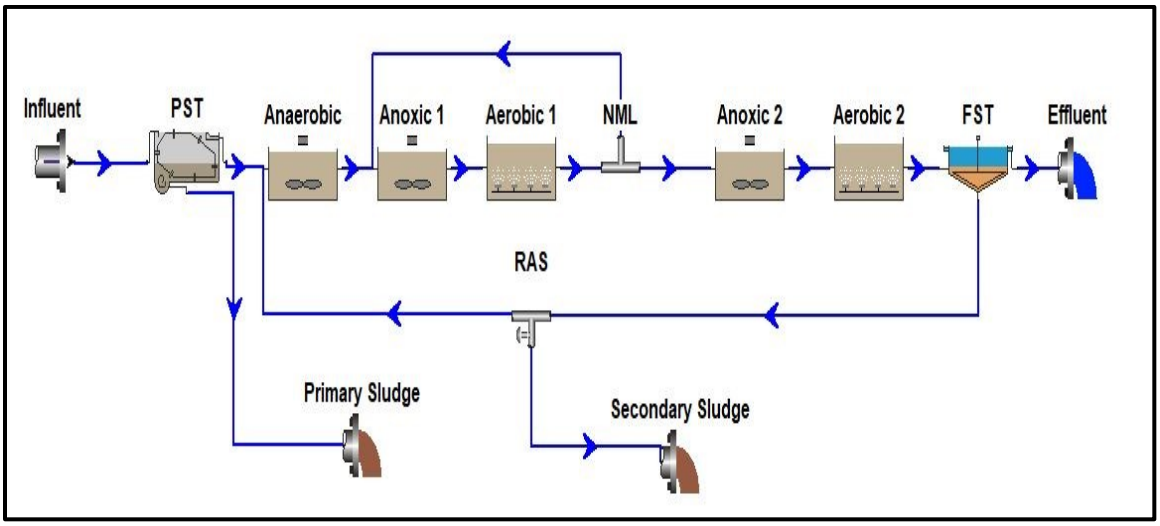


(e)

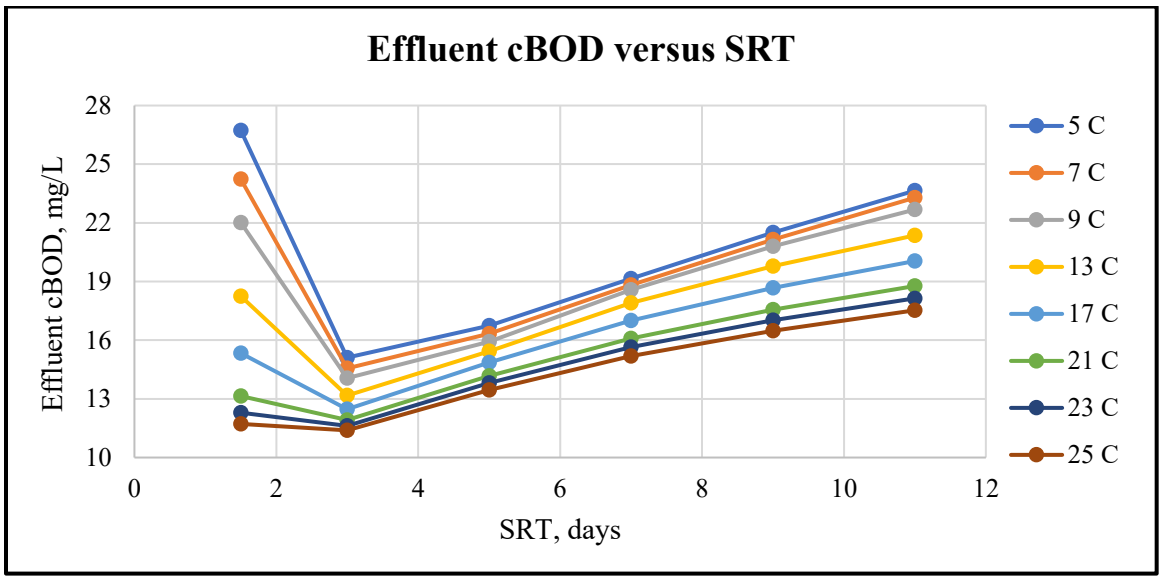
Figure 4.1: Schematic view of five systems considered in this study: (a) conventional AS (b) MLE (c) Phoredox (d) A2O (e) Modified Bardenpho

4.3 Results and Discussions

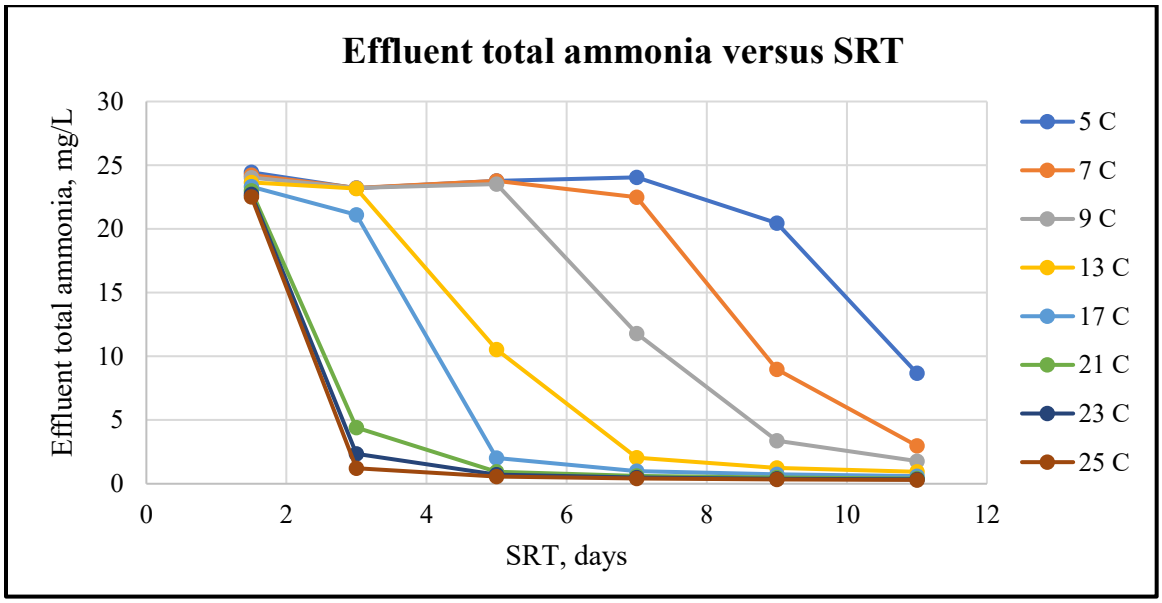
4.3.1 Conventional AS system

The conventional AS system is designed to remove TSS, cBOD and total COD from the mainstream. The effluent parameters such as cBOD, total ammonia and TP were measured in terms of mg/L and SRT in days. Conditions such as aeration rate, food/microorganisms

(F/M) ratio, specific growth rate and all the kinetic parameters were kept the same during all the simulations.



(a)



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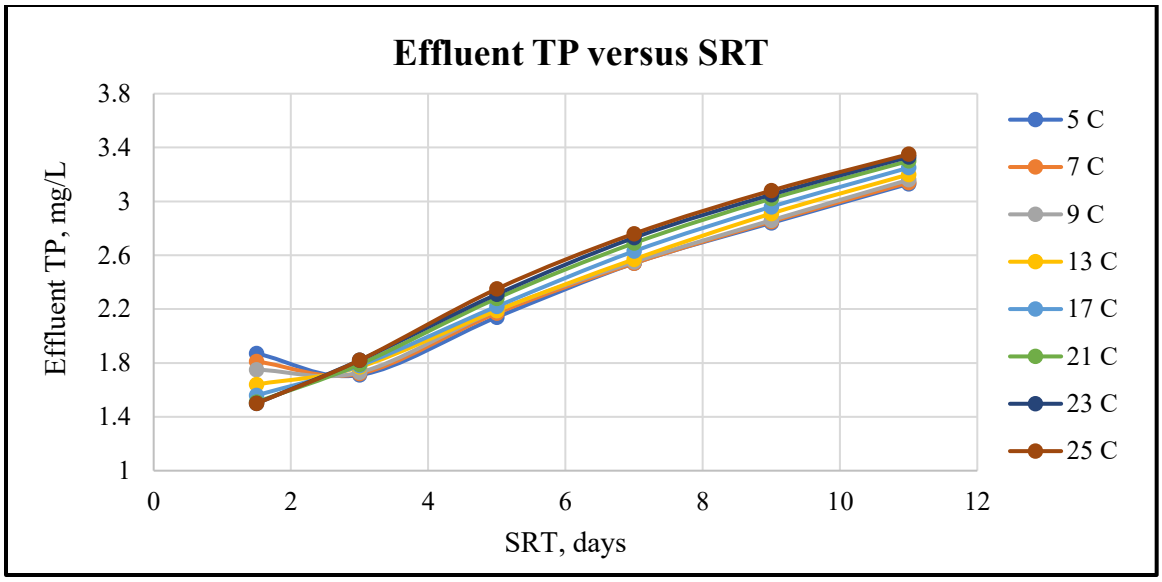


(c)

Figure 4.2: Effluent characteristics for conventional AS system (a) cBOD (b) total ammonia (c) TP

As shown in Figure 4.2 (a), the effluent cBOD at 1.5 days was relatively high due to the low settleability of the flocs. Hence, 3 days SRT improved the effluent quality compared to 1.5 days SRT. But, gradually, when the SRT was increased from 3-11 days, the effluent quality deteriorated as the F/M ratio decreased. The mixed liquor suspended solids (MLSS) increased at the same dissolved oxygen (DO) level. This potentially led to increased effluent cBOD concentrations (Liu et al., 2015). The effluent concentration for cBOD was less at 25℃ compared to at 5℃, due to augmented biological phenomenon. Around 30-33% cBOD removal was improved when the temperature was raised 5-25℃.

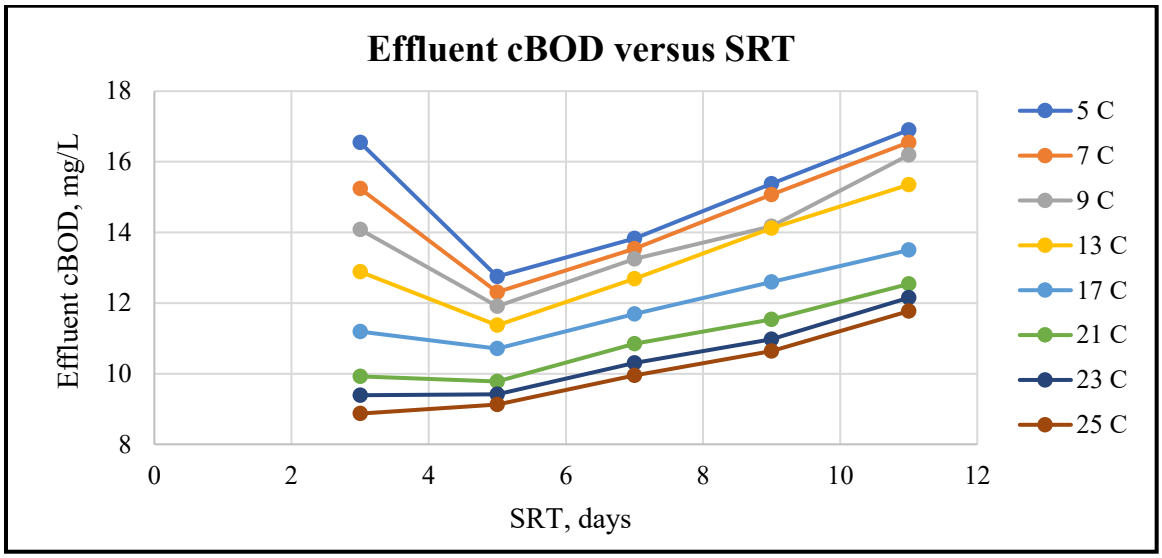
The conventional AS system could be operated at elevated SRT to initiate the nitrification process for the deammonification from the system. The nitrification process requires aerobic conditions. Also, at colder temperatures (T < 12˚C), the nitrifying bacteria were inactive, and the nitrification process was hindered (Plósz et al., 2009). It is shown in Figure 4.2 (b) that the temperature played an essential role in the removal of total ammonia from

the system (M. Chen et al., 2018). Plosz (2009) also mentioned that nitrifiers initiate their activity at warmer temperatures. Figure 4.2 (b) shows that 13℃ was the critical temperature at which the total ammonia removal improved. SRT also played an essential role in the nitrification and denitrification process. While increasing SRT was beneficial for all temperature ranges, it was most beneficial for temperatures 13˚C and above. An SRT of 5 days was essential for the nitrification process (Ekama, 2010). Ekama (2010) mentioned that 5 days or longer SRT was mandatory for nitrogen removal.

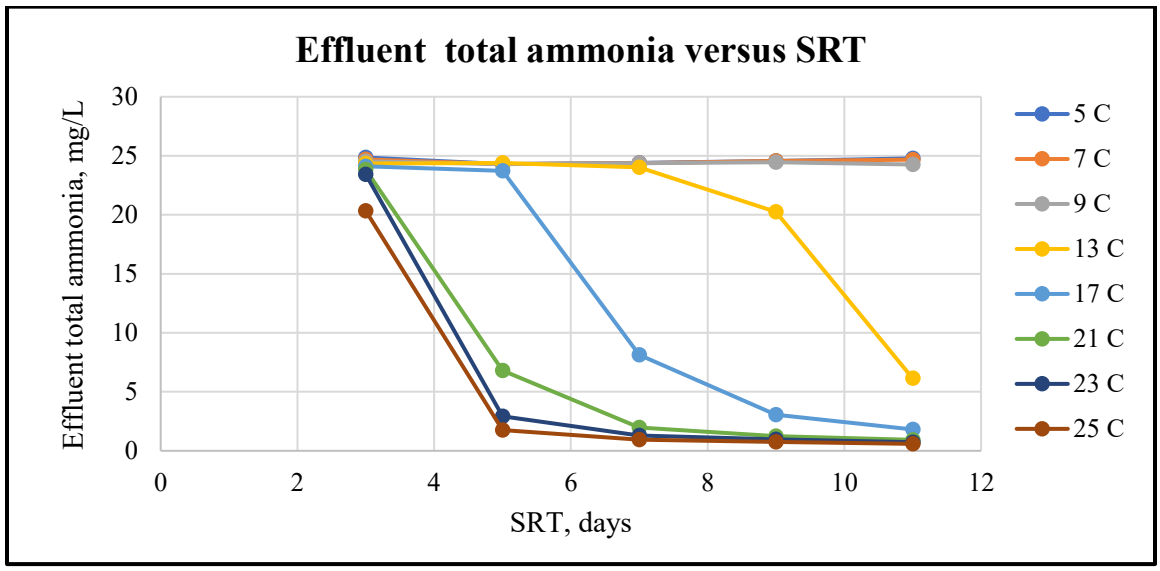
Initially, the effluent TP was high at low SRT for all temperature ranges, as shown in Figure 4.2 (c). Increasing SRT was beneficial in the conventional AS system as PAOs would consume the orthophosphate with the energy stored in their cells. However, increasing SRT was not beneficial under aerobic conditions as nitrite-oxidizing bacteria (NOB) and ammonia-oxidizing bacteria (AOB) may prove more dominant than PAOs. This could have increased the effluent phosphorous in the system. The temperature affected the phosphorous uptake rate in the aerobic zone (Brdjanovic et al., 1998). In the aerobic bioreactor for the conventional AS system, increasing the temperature from 5-20°C increased the phosphorous uptake rate by the PAOs (Chen et al., 2014; Zhilong et al., 2014). As the uptake of phosphorous was increased, it resulted in a drop in the effluent TP concentration at 3 days SRT. The biological phosphorous removal (BPR) efficiency decreased by around 10% for extreme temperature range values (5˚ C and 25˚ C), represented in Figure 4.2 (c). As the temperature and SRT increased, Figure 4.2(c) depicted the dominance of AOB, NOB, and other microbial communities over PAOs. Hence the effluent TP concentrations continued to rise.

4.3.2 MLE system

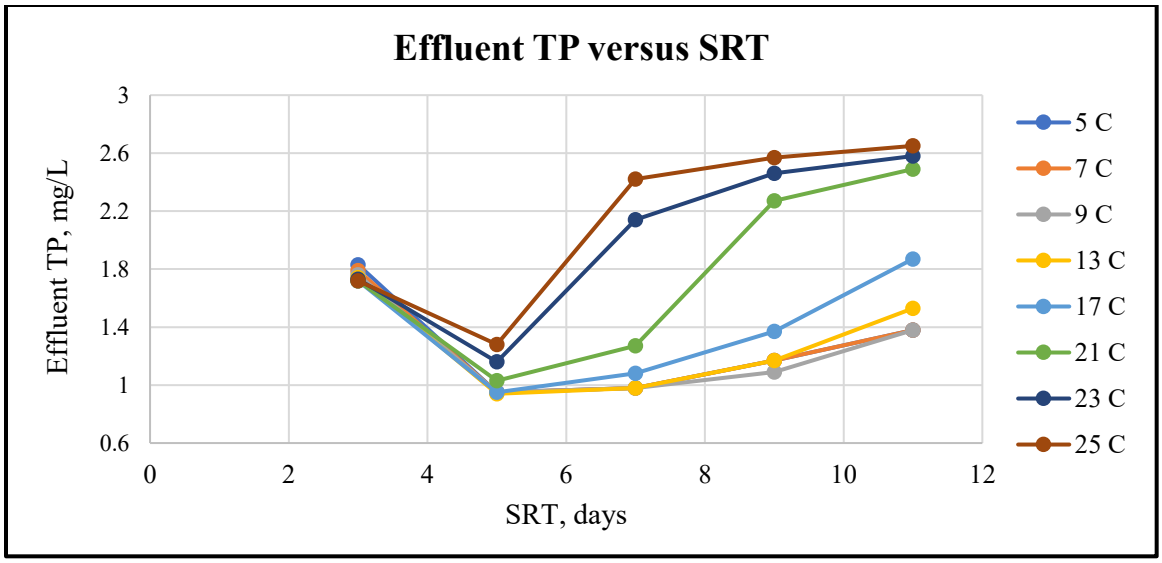
In the MLE system, simultaneous denitrification (in an anoxic zone) and nitrification (in an aerobic zone) occurred. There is a return of nitrifying mixed liquor (NML) to the anoxic zone reducing nitrites and nitrates to nitrogen gas in the anoxic chamber.



(a)



(b)



(c)

Figure 4.3: Effluent characteristics for MLE system (a) cBOD (b) total ammonia (c) TP

The cBOD followed a similar trend in the Figure 4.3 (a) as it was represented in Figure 4.2 (a), in terms of SRT and effluent cBOD pattern. Initially, the values were high at 3 days SRT due to the low settleability of the AS flocs. Then, there was a dip in the effluent cBOD at 5 days, and it again rose due to increased MLSS concentration (Liu et al., 2015). cBOD removal improved by around 84% when the temperature was raised from 5˚C- 25˚C, at SRT for 3 days. For other SRT values, removal ranged from 40-43%.

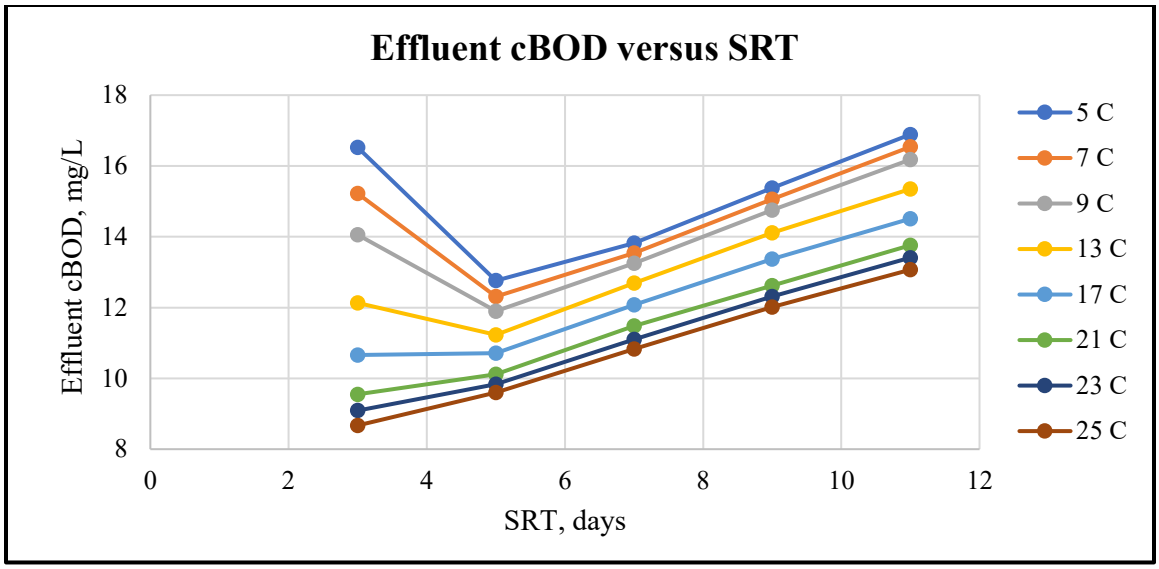
The effluent total ammonia concentration was constant for colder temperatures (T < 13˚ C), with nitrifiers increasing biological activity at 13℃. The performance improved for warmer temperatures, and increased SRT enhanced the MLE process (Sayi-Ucar et al., 2015). The NML recycle recirculated NO_3 and NO_2 in the anoxic bioreactor. Furthermore, the denitrification of the nitrates and nitrites, eventually resulted in ammonia removal from the system. Initially, in Figure 4.3 (b), at SRT 3 days, the effluent total ammonia removal was improved by 24% for extreme temperature (25℃), increasing to 92% for SRT 11 days. Compared to the influent ammonia concentration, there was up to 93% reduction in the

total ammonia concentration at warmer temperatures and elevated SRTs, shown in Figure 4.3 (b).

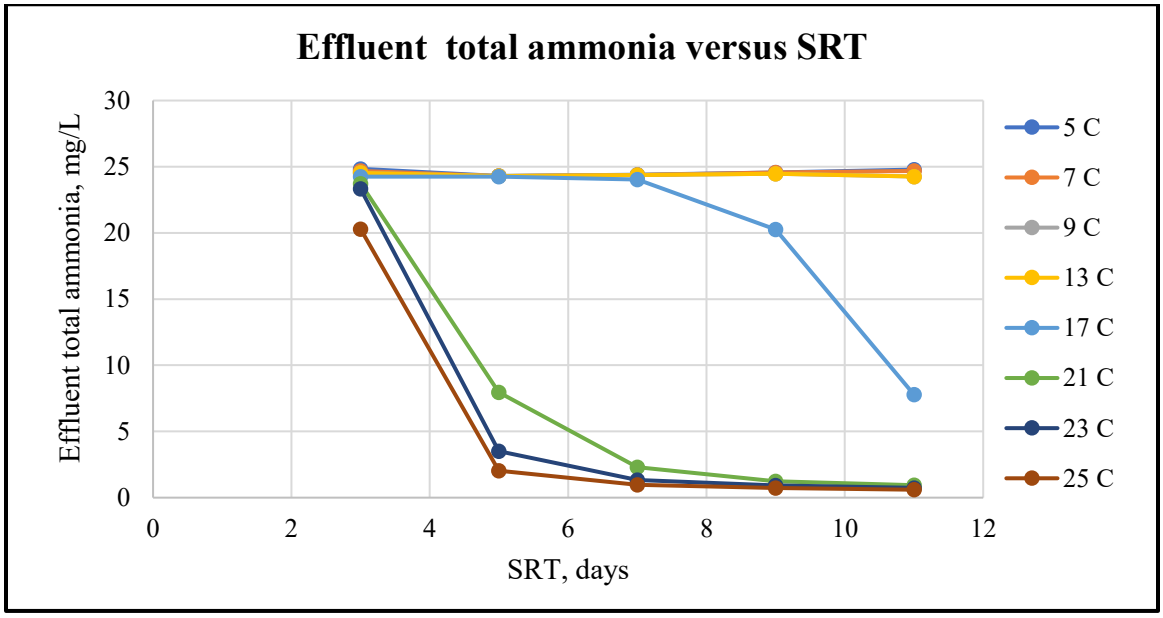
In anoxic and aerobic bioreactors, nitrate and oxygen acted as electron acceptors in the MLE process (Esfahani et al., 2018). The PAOs consumed poly-β-hydroxyalkanoates (PHA) and polyhydroxy butyrate (PHB), converted orthophosphate to polyphosphate and stored it (Esfahani et al., 2018). Removal through sludge prevented the assimilation of phosphorous in the effluent. But, increasing SRT was not advantageous, as shown in Figure 4.3 (c). Furthermore, increasing SRT by more than 5 days, might have benefitted the GAOs and nitrifiers. Additionally, when the temperature was raised, it was observed that COD would have been consumed by the biomass. As at higher temperatures biomass consumed the COD (carbon), lower phosphorous was released in the system. Hence, the removal of TP was reduced. (Sayi-Ucar et al., 2015). As Sayi-Ucar (2015) discussed, GAOs can store volatile fatty acids (VFA) and be in competition with PAOs at higher temperatures. Compared to influent, this system removed up to 86% of phosphorous at 5 days SRT and for colder temperatures ($T \leq 13^{\circ}C$), shown in Figure 4.3 (c).

4.3.4 Phoredox system

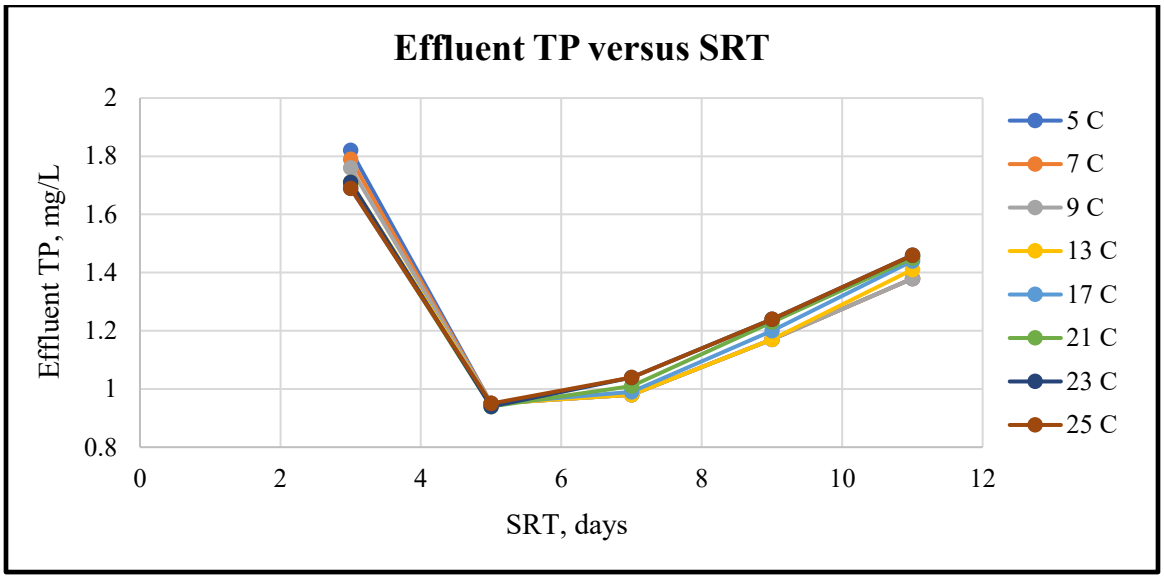
There is an anaerobic zone preceding the aerobic bioreactor in the biological treatment in this system. This system aimed for better phosphorous removal efficiency as the anaerobic bioreactor was added to the system, favoring phosphate release.


Effluent cBOD versus SRT
Effluent cBOD, mg/L
18
16
14
12
10
8
0
2
4
6
8
10
12
SRT, days
5 C
7 C
9 C
13 C
17 C
21 C
23 C
25 C

(a)


Effluent total ammonia versus SRT
Effluent total ammonia, mg/L
30
25
20
15
10
5
0
0
2
4
6
8
10
12
SRT, days
5 C
7 C
9 C
13 C
17 C
21 C
23 C
25 C

(b)



(c)

Figure 4.4: Effluent characteristics for Phoredox system (a) cBOD (b) total ammonia (c) TP

The cBOD removal from the system was similar to the trend of the MLE system. A minor increase of 1 mg/L in the effluent cBOD at the warmest temperature (25℃) in the Phoredox system compared to the MLE system at maximum SRT is displayed in Figure 4.4 (a). The removal efficiency of this system was 96.5%, compared to the influent cBOD at SRT 3 days at the warmest temperature 25˚C, represented in Figure 4.4 (a), suggesting that the elevated temperatures were beneficial for the cBOD removal in all the systems.

Figure 4.4 (b) demonstrates the delay in simultaneous nitrification and denitrification compared to MLE. Total ammonia removal began at 17˚C. With increasingly warmer temperatures above 17˚C, total ammonia removal increased and plateaued at 7 days for all T > 17˚C. The denitrification of the nitrates and nitrites was hindered in the anaerobic zone as PAOs dominated in the anaerobic bioreactor. In the anoxic zone, usually the DO ranges upto 0.2-0.4 mg/L (Capodaglio et al., 2016), and in the anaerobic bioreactor, the DO was nearly 0 mg/L. The deficit of DO in the anaerobic bioreactor may have impacted the

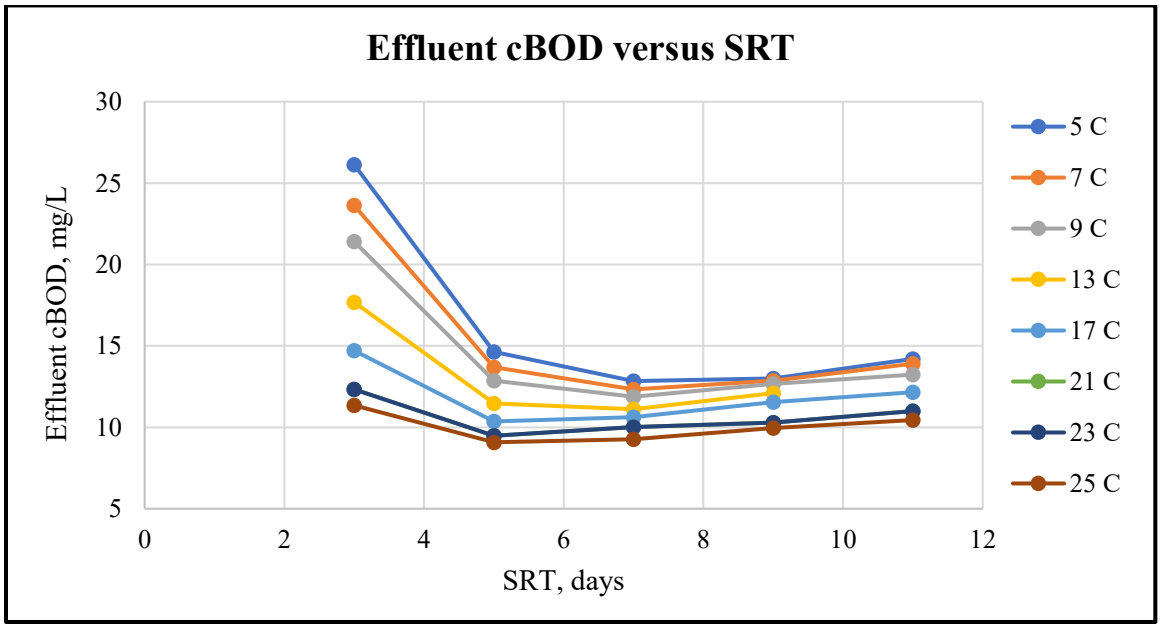
denitrification process, as some DO was available for the denitrification process in an anoxic zone. But in the Phoredox system, the DO was nearly 0 mg/L in the anaerobic bioreactor, and the growth of NOB was hindered as the dominance of PAOs was observed over NOB. Therefore, the total ammonia removal dropped in this system.

The Phoredox system was designed for BPR because of the anaerobic zone in the biological treatment (Curtin et al., 2011). PAOs consumed volatile fatty acids (VFAs) in the anaerobic zones and converted the orthophosphate to polyphosphate. Later, the polyphosphate could be taken through the sidestream during the wasting in the sludge and phosphorous could be recovered. The effluent values for the TP were nearly 0.95 mg/L at 5 days SRT for all temperature ranges, with almost 86% removal efficiency, shown in Figure 4.4 (c). The increase in the effluent for TP was more gradual than in the MLE process. The lowest removal efficiency with increased SRT is 78.5%, shown in Figure 4.4 (c), and the general duration for the MLSS in the anaerobic zone was 1.5-2 days (Curtin et al., 2011). The increased temperature created a competition between GAOs and PAOs and other metabolic activities at elevated temperatures, which resulted in deteriorating effluent quality for TP (Brdjanovic et al., 1998; Sayi-Ucar et al., 2015).

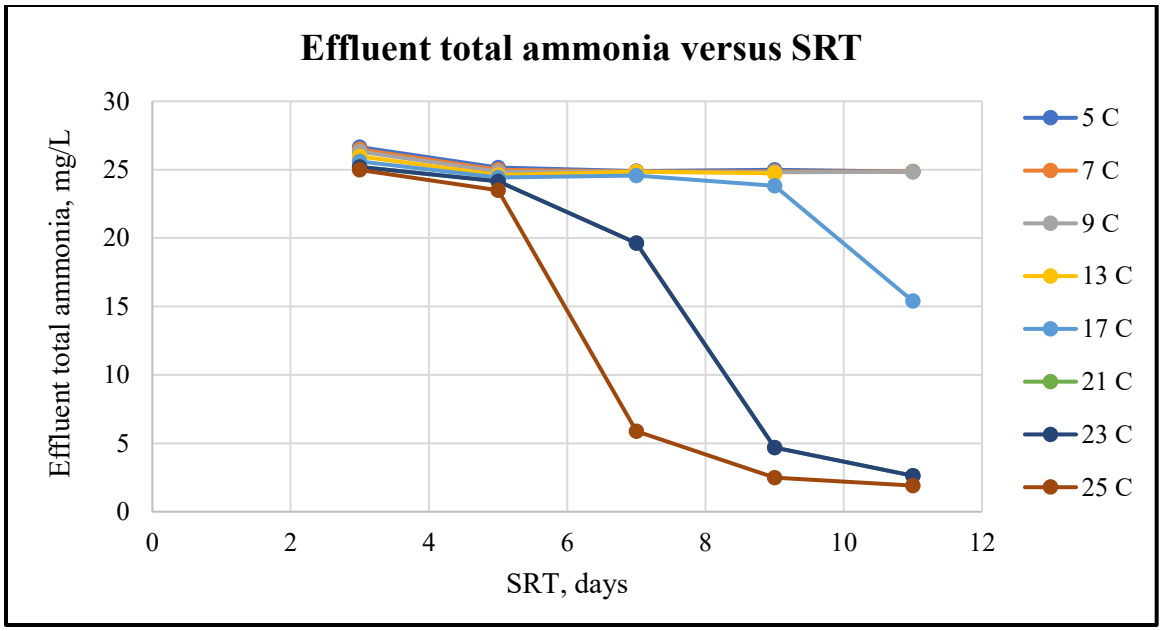
4.3.5 A2O system

A2O is an essential system for biological nitrogen and phosphorous removal. As shown in Figure 4.1 (d), this system consists of three bioreactors: anaerobic, anoxic, and aerobic. The RAS was recirculated from the final settling tank (FST) to the anaerobic bioreactor. Also, there of nitrifying mixed liquor (NML) was recirculated from the aerobic bioreactor to the anoxic bioreactor. Anoxic and aerobic bioreactors helped in the denitrification and nitrification process, eventually leading to the removal of total ammonia from the

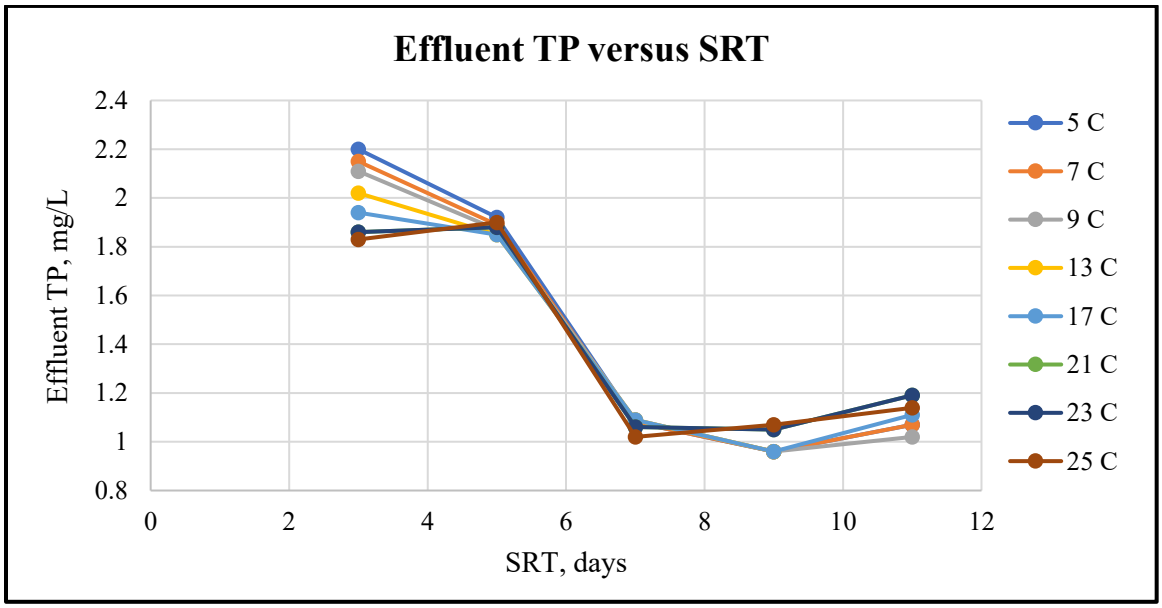
mainstream. Anaerobic bioreactor also aids the denitrification of the nitrate to nitrogen gas (Hamad, 2014). Anammox, known as anaerobic oxidation of ammonium and nitrite to nitrogen gas, could also occur under specific conditions in the anaerobic bioreactor (Jin et al., 2012). This causes a reduction in the total ammonia concentration from the mainstream. Hamad (2014) also mentioned that aerobic conditions aid in the growth PAOs. PAOs convert their organic matter to poly-β-hydroxyalkanoates (PHA) in anaerobic conditions. PAOs consumed the energy generated by the disintegration of polyphosphate molecules to generate PHA. Phosphorous is hence released, and under aerobic conditions, PAOs use the energy stored in them via PHA disintegration (obtained during anaerobic conditions) to take up phosphorous from the mainstream (Hamad, 2014). The low oxygen demand in the A2O system gives it an upper hand compared to other systems. It is efficient in carbon, total ammonia, and TP removal.



(a)



(b)



(c)

Figure 4.5: Effluent characteristics for A2O system (a) cBOD (b) total ammonia (c) TP

Figure 4.5 (a) presents the trend for the removal of cBOD in the A2O system. At 3 days SRT, the effluent SRT was comparatively high compared to higher SRT (5-11 days) for 5-25 °C range. This could be due to the low settleability of the AS flocs at lower SRT (Smith et al., 2015). It was seen increasing SRT from 3-5 days to be advantageous for the cBOD

removal. Further increasing SRT from 5-11 days, the cBOD concentrations did not rise significantly. The concentrations were almost stagnant. This could be because cBOD might be used as a carbon source for ammonia and phosphorous removal. With the limited F/M ratio and DO, the values rose at 11 days SRT for 5-25 °C. As the temperature increased from 5-25 °C, biological activity enhancement occurred. This led to a reduction in effluent concentrations. At 3 days SRT, the effluent concentration at 25 °C was 57.6 % lower than the effluent concentration at 5 °C. While at 11 days SRT, the effluent concentration at 25 °C was nearly 25% lower than the effluent concentration at 5 °C. Also, the effluent cBOD concentration (at 11 days) dropped by approximately 44.2% at 5 °C, increasing SRT from 3-11 days. At 25 °C, with an increase in SRT from 3-11 days, the effluent concentration at 11 days was almost the same. Hence, an increase in temperature was advantageous for cBOD removal in the A2O system. While increasing SRT initially proved beneficial for this system.

Shorter SRT inhibited the nitrification process (Smith et al., 2015). Nitrifiers require adequate temperature and DO conditions for the growth of their community. Figure 4.5 (b) showed that the nitrification ceased for T < 17 °C. Though the SRT was increased from 3-11 days for 5-13 °C, the removal of total ammonia from the mainstream was negligible. But at 17 °C, as the SRT increased from 3-11 days, it depicted a drop in the total ammonia concentration. This could be indicative of the initiation of the nitrification process. The NML was circulated to the anoxic zone, reducing nitrate to N_2 gas. Hence a reduction in total ammonia was observed. As the temperature increased, the total ammonia concentration dropped at 5 days SRT (21-25 °C). At 11 days SRT for 5 °C, the effluent total ammonia was almost the same as at 3 days SRT. At the same time, the effluent total

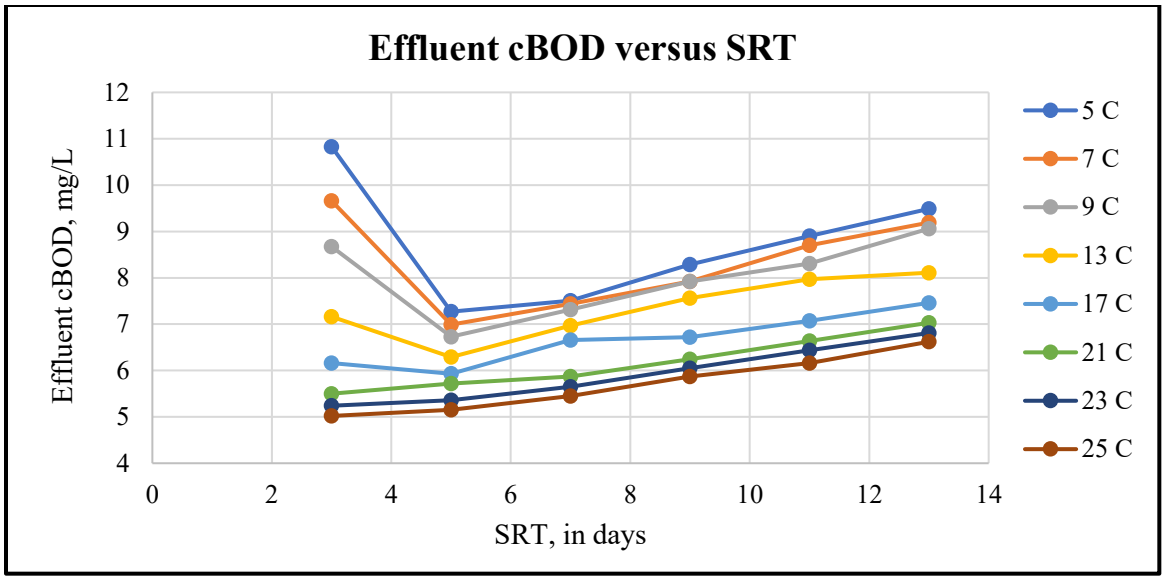
ammonia concentration for 25 °C at 11 days dropped by 84 % compared to that at 3 days SRT. 17 °C was considered critical for the simultaneous nitrification and denitrification process.

Figure 4.5 (c) represents that at 3 days SRT, TP effluent concentration at 5 °C was nearly 18.18% higher than that of the TP effluent concentration at 25 °C. This could be because of the lower activity of the PAOs at lower temperatures. The phosphorous release could have occurred in the anaerobic zone. But due to the low activity of PAOs at 5 °C, the effluent TP concentration was higher. With the increase in SRT from 3-5 days for 5-17 °C, there was a drop in the TP effluent concentration due to the simultaneous release and uptake of phosphorous in the anaerobic and the anaerobic zone. Figure 4.5 (c) also represents that for T > 17 °C, there was a slight increase in the TP concentrations at 5 days SRT compared to 3 days SRT. This could be because the nitrification process is accelerated for T > 17 °C. Hence, the effluent TP concentration might have risen due to limited DO in the aerobic bioreactor. With a further increase in SRT (overall SRT) from 5-7 days, the mean residence time in the anaerobic bioreactor increased. Curtin et al. (2011) studied that 1.5-2 days mean residence time in the anaerobic bioreactor, proliferates the phosphorous release in the anaerobic zone. With the increase in overall SRT, the mean residence time in the anaerobic bioreactor increased. There was a deficit of carbon sources for the phosphorous release. At higher overall SRT, nitrifiers were also recycled to the anaerobic bioreactor. The nitrifiers inhibited the phosphorous release and offer competition to the PHA to release phosphorous. Curtin et al. (2011) also observed that with the increase in SRT, GAOs offered competition to the PAOs for the carbon source and eventually inhibited the phosphorous release. Hence, Figure 4.5 (c) observed that with the increase in SRT from 7-11 days in the A2O system,

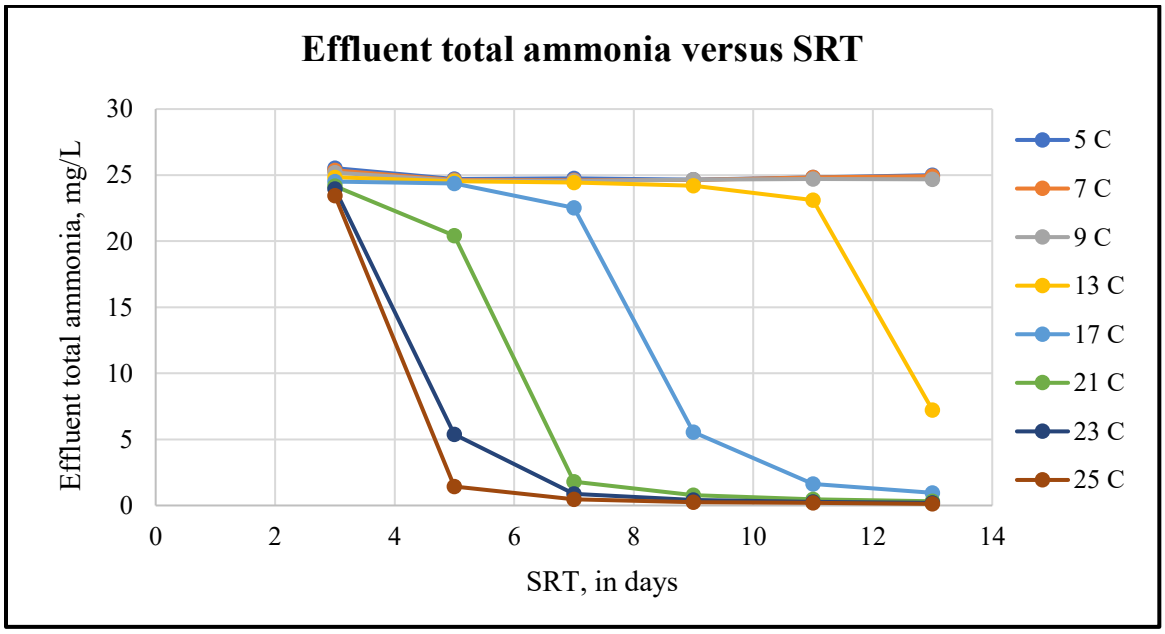
there was a slight increase in the effluent TP concentration. At 11 days SRT, the effluent at 25 °C was nearly 15 % higher than that at 5 °C. The effluent quality improved for 5 °C and 25 °C by 54.54% and 34.44%, respectively when SRT increased from 3-11 days.

4.3.6 Modified Bardenpho system

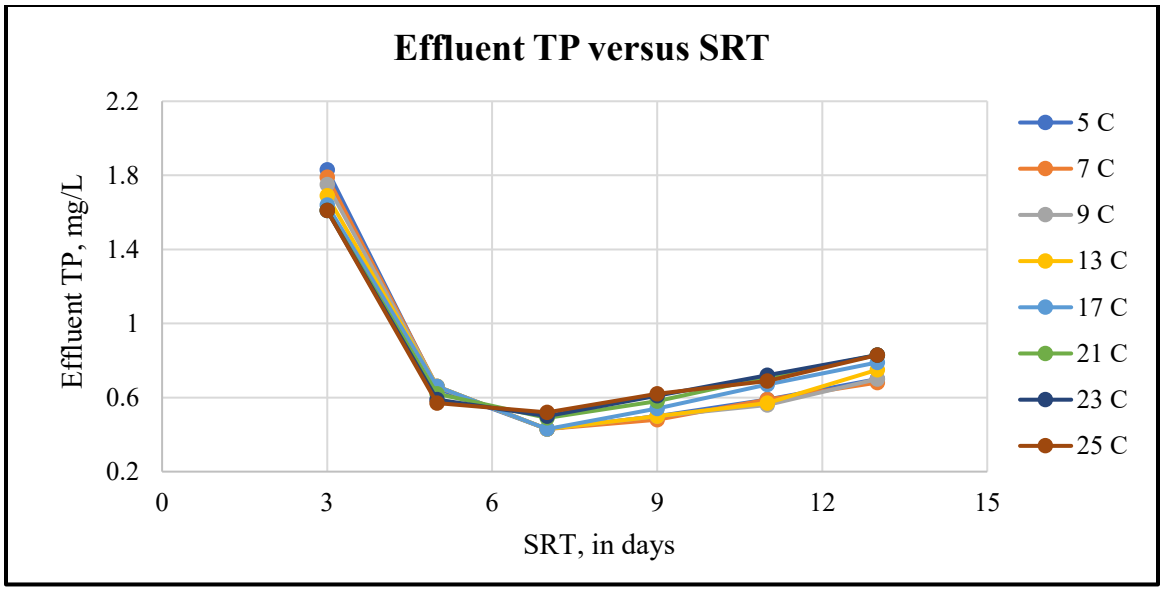
Figure 4.1 (e) represents the schematic for the modified Bardenpho system. This system incorporates five bioreactors: anaerobic, anoxic, aerobic, anoxic, and aerobic. The RAS was recirculated from the FST to the anaerobic bioreactor. Due to continuous anoxic and aerobic bioreactors, minimum NML was recirculated to the anaerobic bioreactor. This improved the phosphorous release in the anaerobic bioreactor. Also, the carbon source recycled was consumed mainly by the PAOs. The series of anoxic and aerobic bioreactors aids in the simultaneous denitrification-nitrification process. There was a recycle of NML from the aerobic 1 bioreactor to the anoxic 1 bioreactor. This process enhanced the denitrification of the nitrate in the anoxic 1. Due to this arrangement, the effluent cBOD, total ammonia and TP concentrations were significantly low compared to other BNR systems. This system could provide efficient results in lower operation duration. Also, this system could reduce the chemical usage for the removal of nutrients. Modified Bardenpho system was modeled for 3-13 days SRT. Higher SRT duration was chosen for this system to understand the efficiency of SRT for the series of bioreactors. Each bioreactor (anaerobic, anoxic and aerobic) had a mean residence time of 0.42-2.04 days for the overall SRT range of 3-13 days. Esfahani et al. (2018) mentioned that the modified Bardenpho system was highly capable of removing total COD, cBOD, TSS, and other heavy metals. The operation of this system required precise conditions for every bioreactor.


Effluent cBOD versus SRT
Effluent cBOD, mg/L
12
11
10
9
8
7
6
5
4
0
2
4
6
8
10
12
14
SRT, in days
5 C
7 C
9 C
13 C
17 C
21 C
23 C
25 C

(a)


Effluent total ammonia versus SRT
Effluent total ammonia, mg/L
30
25
20
15
10
5
0
0
2
4
6
8
10
12
14
SRT, in days
5 C
7 C
9 C
13 C
17 C
21 C
23 C
25 C

(b)



(c)

Figure 4.6: Effluent characteristics for Modified Bardenpho process (a) cBOD (b) Total Ammonia (c) TP

Figure 4.6 (a) represents the effluent cBOD concentrations over the SRT of 3-13 days. It was observed from Figure 4.6 (a) that for T < 17 °C, the effluent cBOD at 3 days SRT was higher than effluent SRT at 5 days SRT. The biological activity was significantly low for the colder temperatures (5-13 °C). This resulted in higher cBOD concentrations. Also, for T < 17 °C, low settleability of the AS flocs might result in higher effluent cBOD concentrations at 3 days SRT than 5 days SRT. For T ≥ 17 °C, when the SRT was increased from 3-13 days, the effluent cBOD values continued to rise. This could be due to inhibition caused by the nitrifiers. Also, nitrifiers might have catered competition to the other microbial community for limited DO. Whereas, for T < 17 °C, there would have been little competition for the nitrifiers and the AS microbial community. Hence, a drop was observed at 5 days SRT for T < 17 °C. Thereby increasing SRT (5-13 days) for T < 17 °C, the effluent cBOD depicted a gradual increase. At 3 days SRT, the effluent cBOD concentration at 25 °C was nearly 54.54 % lower than that of effluent cBOD at 5 °C. For 13 days SRT, the

effluent cBOD concentration at 25 °C was around 28.42 % lower than that of effluent cBOD at 25 °C. The effluent cBOD dropped by 13.63 % for 5 °C, when the SRT was increased from 3-13 days. In contrast, the effluent cBOD values rose by 36 % for 25 °C when SRT was increased from 3-13 days.

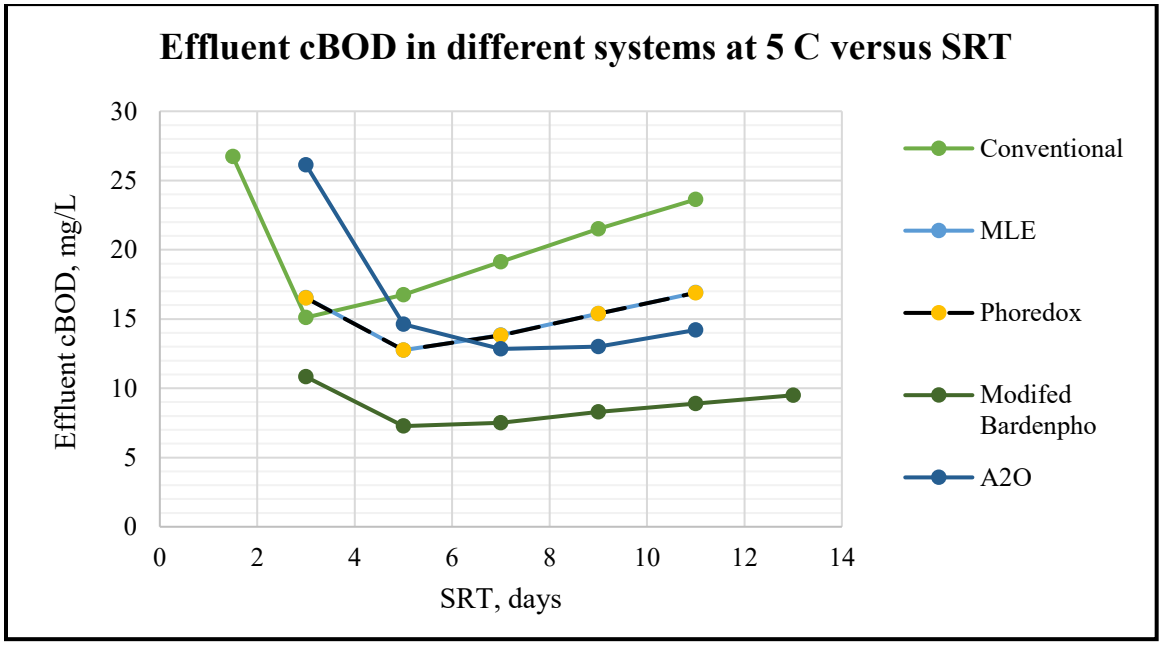
Effluent total ammonia concentrations for the modified Bardenpho system were represented in Figure 4.6 (b). It was observed that negligible/minor removal for total ammonia was observed for T < 13 °C. This could be due to the inactivity of nitrifiers at colder temperatures (5-13 °C). For T < 13 °C, increasing SRT and having a series of anoxic and aerobic bioreactors did not aid in the removal of total ammonia. At 13 °C, as the SRT was increased from 3-13 days, nitrification was initiated. As the NML was recycled from aerobic 1 to anoxic 1, the denitrification was boosted. Also, the successive anoxic and aerobic bioreactors accelerated the simultaneous denitrification and nitrification process. The effluent quality at 13 °C at 13 days improved by 72 % compared to the effluent at 3 days. As the temperature rose more than 13 °C, the effluent total ammonia concentration dropped even for 7-11 days SRT. The activity of nitrifiers was significantly boosted with the elevation of temperature. As shown in Figure 4.6 (b), the effluent total ammonia concentration dropped by nearly 79 % and 94 % at 23 °C and 25 °C, respectively, when the SRT was increased from 3-5 days. At 21 - 25 °C, the effluent total ammonia concentration almost reached below 1 mg/L for 7-13 days. Figure 4.6 (b) presents that effluent total ammonia concentration for T > 17 °C dropped by 97.91 % at 13 days compared to 3 days SRT. Whereas, for T < 17 °C, the effluent total ammonia concentration did not decrease significantly (except at 13 °C at 13 days, the concentration dropped by 72%).

The effluent TP concentration at 3 days SRT at 25 °C was 11.11 % lower than at 5 °C. The activity of PAOs is lower at low temperatures and is accelerated by the increase in temperature (Chen et al., 2018). As the overall SRT increases from 3-13 days, the mean residence time in the anaerobic bioreactor also increased from 0.42-2.04 days. Figure 4.6 (c) explains the drop in the effluent of TP at 5 days SRT. The concentration of TP at 7 days SRT for 5-25 °C were very close to that of 5 days SRT. The PAOs stored energy from PHB during the anaerobic bioreactor and used that energy in the aerobic bioreactor and up took phosphate. This resulted in a drop in TP concentration at 5-7 days SRT. With the further increase in SRT, due to the limitation in carbon source, the release of phosphate in the anaerobic zone was inhibited. This resulted in a rise in the TP concentration for 5-25 °C. The concentration for TP at 25 °C was slightly higher than that of 5 °C because of the increased activity of the nitrifiers. As shown in Figure 4.6 (c), the effluent TP values continued to rise for 5-25 °C, increasing SRT from 7-13 days. Approximately 0.4 mg/L effluent TP was observed at 7 days SRT. A maximum of 76.47 % drop in the effluent TP concentration compared to 3 days SRT was observed at 7 days SRT. The effluent quality improved for 5 °C and 25 °C by 63.88 % and 53.33 %, respectively when SRT increased from 3-13 days.

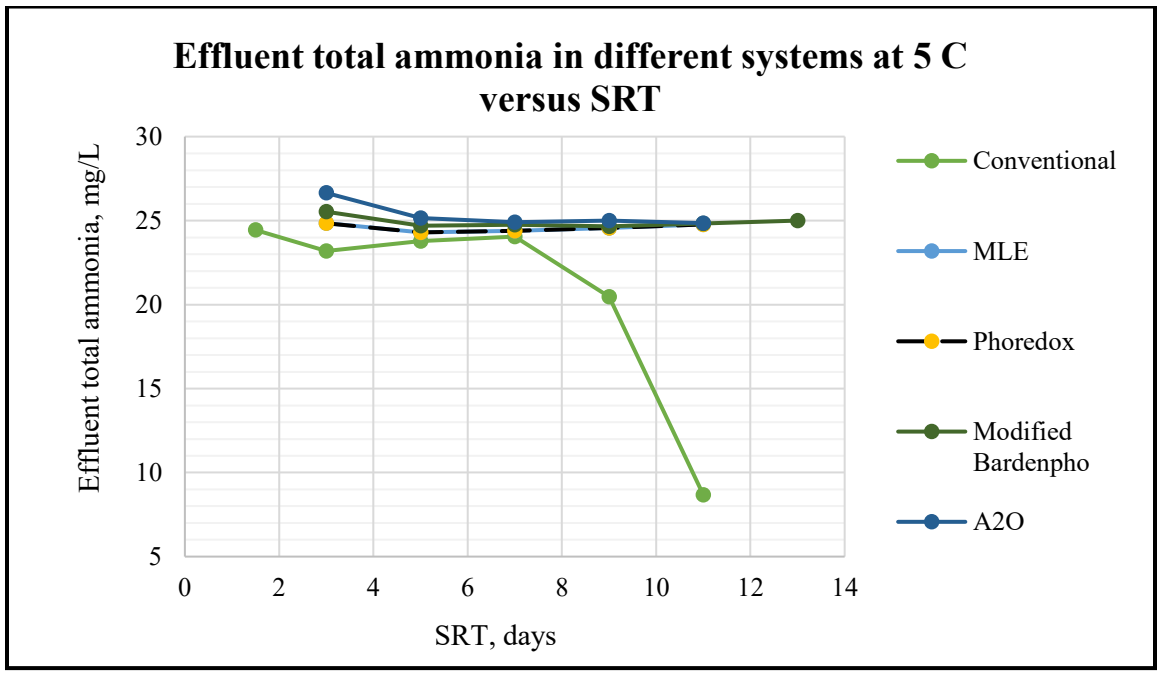
4.4 Comparison between conventional AS, MLE, Phoredox, A2O and modified Bardenpho systems

Temperature and SRT played a critical role in carbon and biological nutrient removal. The conventional AS system was initially designed to remove carbon and suspended solids. MLE and Phoredox system was developed for efficient nitrogen and phosphorous removal, respectively. At the same time, A2O and the modified Bardenpho systems were

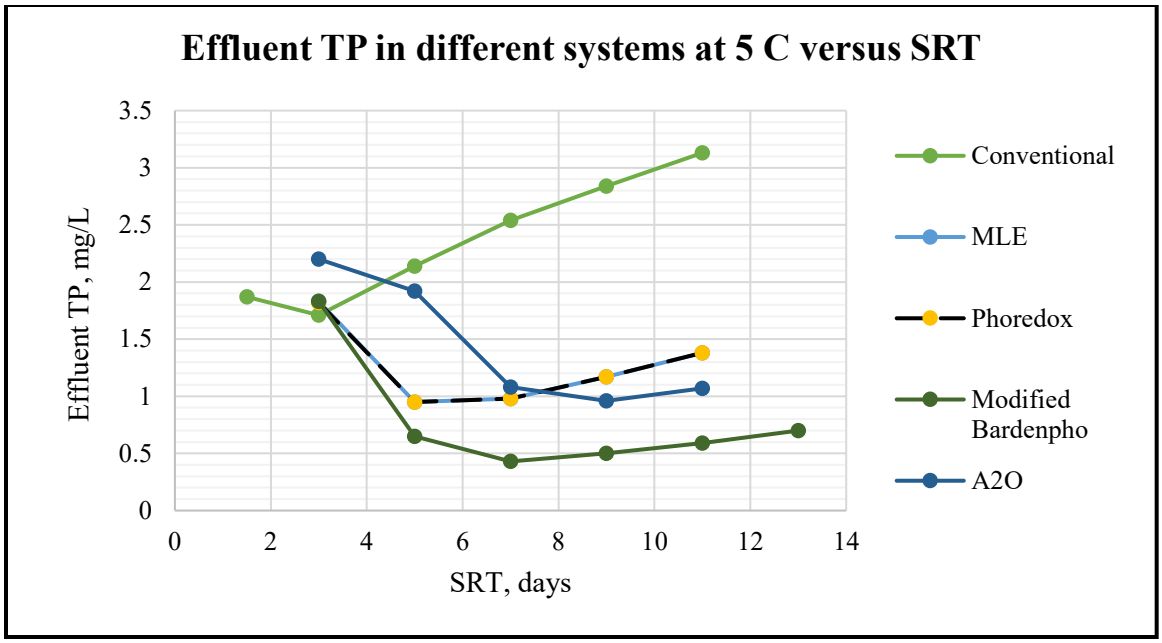
simultaneously designed for efficient nitrogen and phosphorous removal. In addition to the nitrogen and phosphorous removal, carbon, TSS, and heavy metals were removed from the modified Bardenpho systems (Esfahani et al., 2018).



(a)



(b)



(c)

Figure 4.7: Effluent characteristics for different systems at 5 ℃ (a) cBOD (b) total ammonia (c) TP

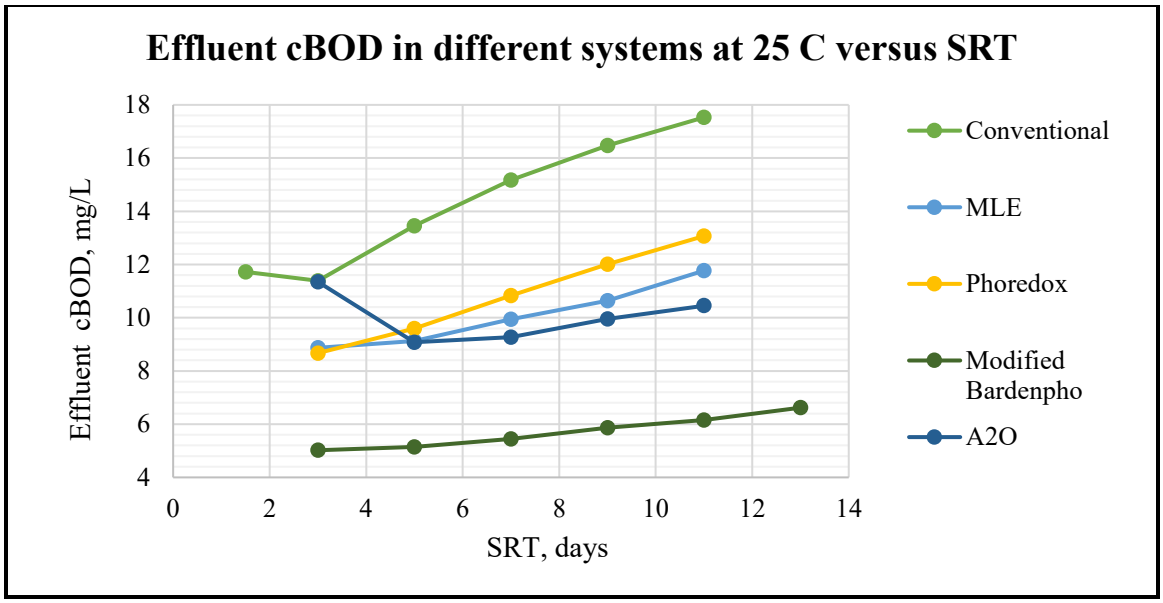
Figure 4.7 and Figure 4.8 represent the effluent concentrations for cBOD, total ammonia and TP at 5 ℃ and 25 ℃, respectively. Figure 4.7 (a) presents that at 3 days SRT, the effluent cBOD was around 15 mg/L. Compared to that, the effluent cBOD was nearly 10 % higher (~16.5 mg/L) for both Phoredox and MLE systems. There was a recycle of carbon source in the anaerobic and anoxic bioreactors of Phoredox and MLE system. Due to low activity, carbon was not taken up by denitrifying bacteria nor by PAOs at 5 ℃, due to low activity. This might have increased the effluent cBOD value for MLE and Phoredox systems. The effluent cBOD was around 76.66 % higher for the A2O system than the conventional AS system. This could be because of the lack of uptake in both anaerobic and anoxic bioreactors. The lower SRT could also explain the lower settleability of the AS flocs.

In contrast to this, as shown in Figure 4.7 (a), the effluent cBOD concentration for the modified Bardenpho system was 26.66 % lower than the conventional AS system. Though there are five bioreactors consisting of one anaerobic and two pairs of successive anoxic and aerobic bioreactors, the effluent cBOD concentration was lower because of two aerobic bioreactors. At 5 days SRT, all the systems presented a drop in the cBOD concentrations except the conventional AS system. Elevation in SRT favored carbon consumption as a source for the nitrogen and phosphorous removal systems nitrogen and phosphorous removal. After 5 days, the effluent cBOD increased for all the systems except the A2O system. Effluent cBOD decreased from 3 to 7 days for the A2O system. As mean residence time in the anaerobic and anoxic bioreactor increased, it might have resulted in the reduction of cBOD concentration in the A2O system. The lowest cBOD concentration was observed at 5 days SRT (~7 mg/L) for the modified Bardenpho system. Figure 4.7 (a) depicted that an increase in SRT above 5 days deteriorated the effluent quality for most systems.

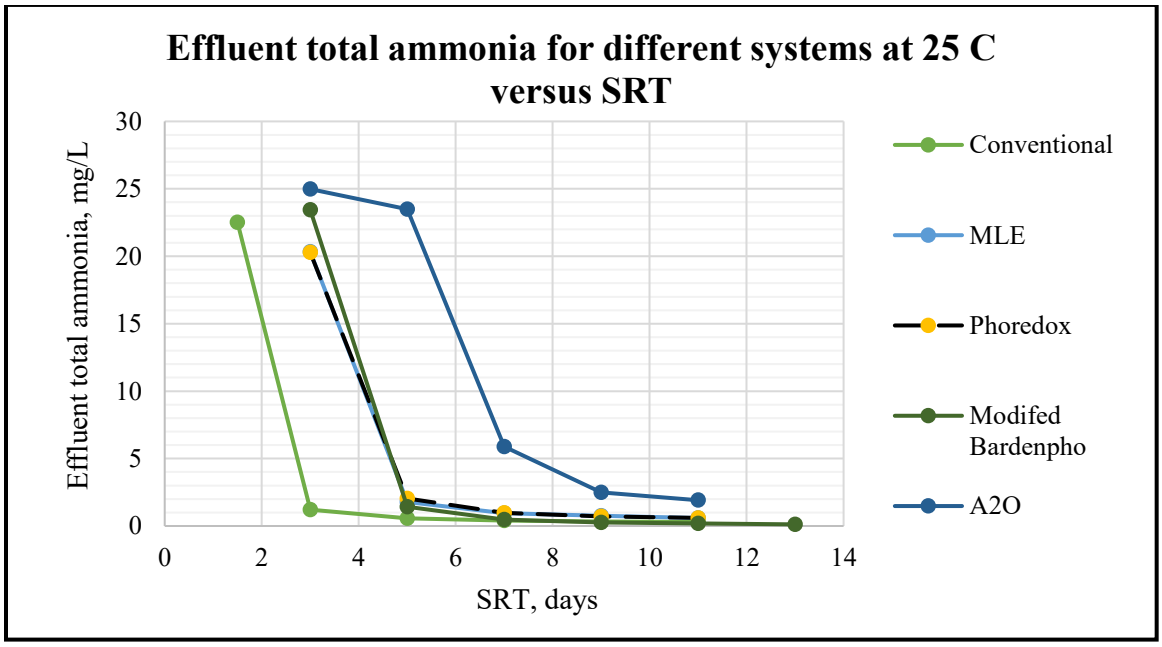
As discussed earlier, nitrifiers are very sensitive to temperature. Figure 4.7 (b) depicts this behavior for all the systems. With an increase in SRT from 3-11 days for MLE, Phoredox and A2O system, effluent total ammonia concentration did not drop. But for the conventional AS system, as shown in Figure 4.7 (b), at 7 days SRT, the nitrification process was initiated. This led to a drop in the effluent total ammonia concentration. For other systems, no significant removal was observed with an increase in SRT. Instead, the effluent concentration of total ammonia for the A2O and modified Bardenpho system was higher than the influent total ammonia concentration for shorter SRT (3 days). There might be competition between PAOs and other AS microbial communities for limited DO. At 11

days SRT, the effluent total ammonia dropped by 65.30 % (~8.5 mg/L) compared to 3 days SRT for the conventional AS system. This explained that an increase in SRT was not at all beneficial for the removal of total ammonia for all systems except conventional AS.

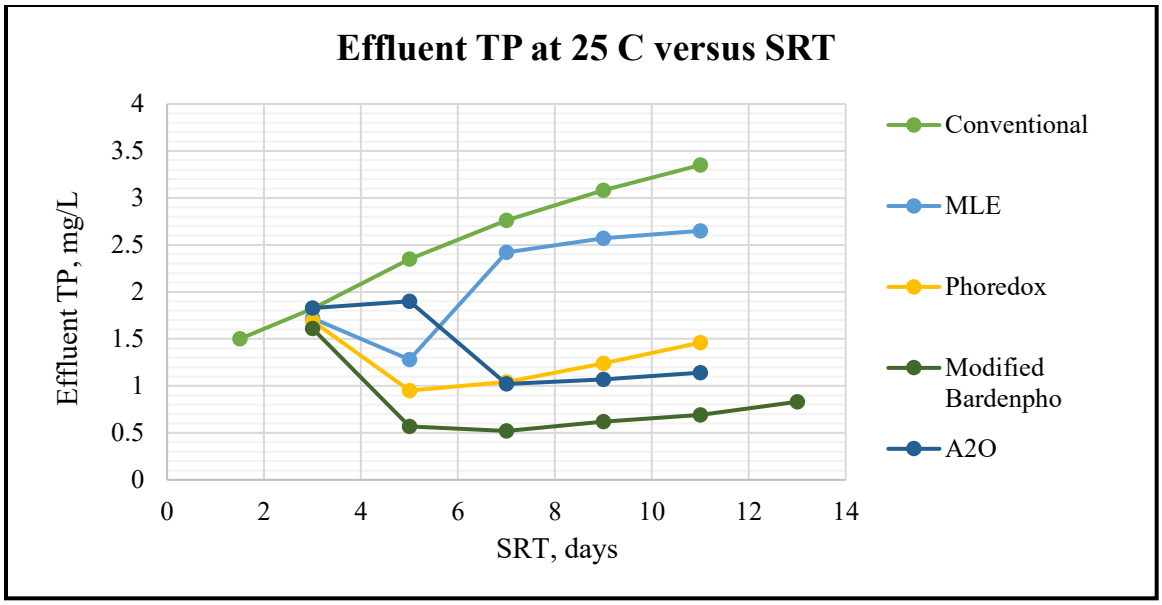
The release of phosphate in the anaerobic zone is not sensitive to temperature change. But the growth of PAOs in the aerobic zones was a factor of temperature (Curtin et al., 2011). Effluent TP concentration was around 1.7 mg/L for the conventional AS system at 3 days SRT. While the TP concentration for MLE, Phoredox and modified Bardenpho system was about 1.85 mg/L, as shown in Figure 4.7 (c). This might be because of the short residence time in the anaerobic or anoxic bioreactor. MLE and Phoredox systems acted likewise at 5 °C due to the inactivity of the nitrifiers. PAOs consumed the carbon source and eventually reduced the effluent TP concentration. In Figure 4.7 (c), it could be observed that at 3 days, the effluent TP concentration for the A2O system was 22.72 % (~2.2 mg/L) higher than that of the conventional AS system. The effluent TP could be higher at 3 days SRT for the A2O system because of the lag in the growth of PAOs in the aerobic bioreactor (as anaerobic and anoxic bioreactors precede aerobic bioreactor). As the SRT increased from 3-11 days, the effluent TP concentration dropped by 50 % (~1.1 mg/L). Also, as the SRT was raised from 3-11 days, the effluent TP reached the lowest at 7 days (~0.4mg/L). Thereby, increasing the SRT and limited carbon source led to increased effluent TP concentration for most of the system. Figure 4.7 (c) explains the significance of SRT as a critical parameter at 5 °C for the TP removal for all the systems.


Effluent cBOD in different systems at 25 C versus SRT
Effluent cBOD, mg/L
18
16
14
12
10
8
6
4
0
2
4
6
8
10
12
14
SRT, days
Conventional
MLE
Phoredox
Modified Bardenpho
A2O

(a)


Effluent total ammonia for different systems at 25 C versus SRT
Effluent total ammonia, mg/L
30
25
20
15
10
5
0
0
2
4
6
8
10
12
14
SRT, days
Conventional
MLE
Phoredox
Modifed Bardenpho
A2O

(b)



(c)

Figure 4.8: Effluent characteristics for different system at 25 ℃ (a) cBOD (b) total ammonia (c) TP

Comparing Figure 4.7 (a) and Figure 4.8 (a), it was observed that the effluent cBOD concentrations at 25 ℃ were significantly lower than that at 5 ℃. The lowest cBOD concentration at 25 ℃ and 5 ℃ were 5 mg/L and 7 mg/L for the modified Bardenpho system. The highest cBOD concentration at 25 ℃ and 5 ℃ were nearly 17.8 mg/L and 27 mg/L for the conventional AS system. Figure 4.8 (a) represents that effluent cBOD at 3 days for MLE, Phoredox and modified Bardenpho system was 23.72%, 25.42 % and 57.62 % lower than the conventional AS system. The cBOD concentration at 3 days for A2O and conventional AS system were almost similar. Later, as shown in Figure 4.8 (a), the effluent cBOD concentration dropped (~9 mg/L) for A2O at 5 days SRT. From 3-11 days, the cBOD concentration continuously increased for conventional AS, Phoredox, MLE, and modified Bardenpho systems. With limited DO and low f/m ratio, the effluent cBOD might have risen for the above-said systems. It could be observed from Figure 4.8 (a) that temperature increase improved the cBOD removal and increase in SRT above 5 days

deteriorated the effluent quality for all the systems except conventional AS (for conventional AS 3 days).

Figure 4.8 (b) explains the substantial impact of temperature on the nitrification process and total ammonia removal for all the systems. It was observed that at 3 days SRT, the conventional AS system depicted the effluent total ammonia concentration of ~1.5 mg/L at 25 °C. Compared to the conventional AS system, the effluent total ammonia concentrations for MLE and Phoredox (for both), modified Bardenpho, and A2O were significantly high (~20.5 mg/L, 23.5 mg/L and 25 mg/L, respectively) at 3 days SRT. This explained the inefficiency of these BNR systems as short SRTs (3 days), though the temperature rose to 25 °C. As the mean residence time in anaerobic, anoxic and aerobic bioreactor increased, substantial removal of total ammonia for MLE, Phoredox and modified Bardenpho was observed at 5 days SRT. Approximately 93.6 %, 90.24 % and 90.25 % drop in the effluent total ammonia at 5 days SRT were observed for modified Bardenpho, MLE and Phoredox system, compared to 3 days SRT. MLE and Phoredox systems almost exhibited the same behaviour because, at higher temperatures, PAOs were less dominant than the nitrifiers and GAOs. Hence, total ammonia removal in these systems exhibited similar total ammonia removal behaviour. While as shown in Figure 4.8 (b), the effluent total ammonia for A2O was very high at 5 days SRT (~23.5 mg/L). The DO was limited for phosphorous and ammonia removal in the aerobic bioreactor in the A2O system. Hence due to this, it might have presented a slower removal rate. Further, increasing SRT from 3-11 days for the A2O system, effluent total ammonia concentration dropped by 92 % (~2 mg/L). Figure 4.8 (b) showed that the other systems performed quite efficiently with an SRT increase from 3-7 days. It was observed that further rise in SRT for the systems

except A2O was not adding any value. The effluent total ammonia concentration almost reached 1 mg/L and below for most systems. The effluent total ammonia for all systems except A2O was plateaued after 7 days. Comparing Figure 4.7 (b) and Figure 4.8 (b), it was observed that as the SRT increased from 3-5 days, a considerable drop in the total ammonia concentrations was observed for all the systems. The maximum influent removal efficiency of 98.48 % and 68.18 % was observed at 25 °C and 5 °C.

Comparing Figure 4.7 (c) and Figure 4.8 (c), it could be observed that the effluent TP concentrations for the conventional AS system and modified Bardenpho system showed almost similar trends and effluent TP concentrations. The minimum and maximum concentrations at 5 °C were 0.4 mg/L (modified Bardenpho system) and 3.5 mg/L (conventional AS system). In comparison, the maximum and minimum concentrations at 25 °C were 0.5 mg/L (modified Bardenpho system) and 3.45 mg/L (conventional AS system), respectively. Figure 4.8 (c) showed that MLE and Phoredox portrayed a quite different trend at 25 °C than at 5 °C. At 25 °C, the nitrifiers dominate the PAOs. Therefore, in the MLE system in Figure 4.8 (c), the effluent concentration was higher than in the Phoredox system. Also, NML inhibited the phosphate release in the anoxic bioreactor in the MLE system. As the SRT increased from 3-11 days for the MLE system, the TP concentration dropped from 1.7 mg/L (3 days) to 1.3 mg/L (5 days) and then after 5 days, it significantly rose. It continued to grow for 11 days for the MLE system (~2.6 mg/L). At the same time, the TP concentrations for the Phoredox system were lower because there was no return of NML in the anaerobic bioreactor. At 3 days, the effluent TP concentration for the Phoredox system was nearly 1.7 mg/L which dropped to 1 mg/L at 5 days. After 5 days, the effluent TP concentrations increased gradually until 11 days (~1.5 mg/L). As

shown in Figure 4.8 (c), there was a slight increase in TP concentration from 3-5 days of 5.55 % for the A2O process. After that, a sharp fall of 47.36 % (~1 mg/L) in the TP concentration was observed, as the SRT increased from 5-7 days for the A2O system. After that, the TP concentration rose gradually for higher SRT (9-11 days). It was observed that higher SRT (7days) is required for the A2O system of efficient TP removal. SRT was a critical parameter in the TP removal at 25 °C for all the systems. Eventually, higher SRT prevailed deficit of carbon source and DO. Temperature played a crucial role in the removal mechanism of the MLE and Phoredox system.

4.5 Conclusion

This study explored the impacts of climate change associated with wastewater temperature change on various AS system processes, such as conventional AS, MLE, Phoredox, A2O and modified Bardenpho. The combined effect of temperature change from 5-25 °C over SRT of 1.5-11 days for conventional AS, 3-11 days SRT for MLE, Phoredox and A2O system and, 3-13 days SRT for modified Bardenpho system was studied. Three parameters were studied for these systems: cBOD, total ammonia and total phosphorus. Following are the conclusions for each system:

- Conventional AS system: Temperature increase reduced the effluent cBOD and total ammonia concentration. But the temperature did not play an essential role in the TP removal in this system. However, the concentrations at 3 days SRT for 5 °C were higher than at 25 °C and vice versa at 11 days. It could be concluded that temperature had an indirect impact on TP removal in the conventional AS system. An increase in SRT from 3-11 days elevated cBOD and TP concentrations but

reduced total ammonia concentrations substantially. 13 °C and 5 days SRT were critical parameters for the total ammonia removal in this system.

- MLE system: cBOD concentrations in this system were lower than in the conventional AS system because nitrifiers consumed carbon. For T < 17 °C, there was a drop in the cBOD concentration for SRT increase from 3-5 days. After that, cBOD gradually increased for T > 17 °C. Also, better total ammonia removal was at T ≥ 17 °C. Temperature played a vital role in the TP removal at higher SRT. At 11 days SRT, effluent TP at 25 °C was around 85.71 % higher than at 5 °C. This could explain the higher biological activity of nitrifiers at 25 °C. 5 days SRT was a critical parameter for cBOD, total ammonia and TP removal.
- Phoredox system: Effluent cBOD concentration showed a similar trend and concentration as the MLE system. PAOs also consumed carbon for the phosphorous release in the anaerobic bioreactor. The total ammonia removal was initiated at 17 °C. This could be because there was no recycling of NML and at higher temperatures, nitrifiers presented dominance over PAOs. 5 days SRT was significant for TP, total ammonia and cBOD removal. This could have availed adequate residence time in each bioreactor.
- A2O system: Higher SRT (7-11 days) was advantageous for cBOD, total ammonia and TP removal. There is a lag in the duration in this system because of the anaerobic, anoxic and aerobic zones. Adequate mean residence for each bioreactor was achieved at higher SRT. Hence, efficient cBOD, total ammonia and TP removal were observed at 7 days SRT. For TP removal, at 3 days SRT, the

temperature change impacted substantially (effluent TP at 5 °C was 18.18 % higher than effluent TP at 25 °C).

- Modified Bardenpho system: Maximum removal efficiency for cBOD, total ammonia and TP was observed for this system. Due to five bioreactors, the removal efficiency of this system was observed at 5 days SRT. 17 °C was essential for total ammonia removal in this system. An increase in SRT after 5 days increased cBOD concentrations for 5-25 °C. At the same time, an increase in temperature and SRT was observed as beneficial for total ammonia removal. The temperature did not significantly impact the TP removal in this system.

Overall, the modified Bardenpho process was the most efficient for cBOD, total ammonia and TP removal for 5-25 °C. However, at 5 °C, the conventional AS system presented the best total ammonia removal compared to all other systems at 11 days SRT. 25 °C showed better removal efficiency for all the systems than at 5 °C for cBOD and total ammonia removal. MLE and Phoredox depicted an almost similar removal trend at 5 °C and 25 °C. However, there was an exception for cBOD and total ammonia at 25 °C for MLE and Phoredox system. A2O performed efficiently at higher SRT (≥ 7 days). In the conventional AS system, the effluent TP was not varied by temperature change (5-25 °C). Future research could study the impact of change in other parameters such as DO, pH, COD/N ratio or COD/P ratio. Also, it could incorporate the effects of high flow and low flow influent conditions. The increase in the wastewater temperature improved the cBOD and total ammonia removal but did not prove beneficial for the TP removal. Hence, ambient temperature change because of climate change may aid the removal of influent wastewater

characteristics. But extreme increase in the ambient temperature may cease the biological processes and pose disruption in the treatment facility.

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Chapter 5: Impact of change in influent flow and wastewater temperature on the secondary wastewater treatment systems

Abstract

Climate change is expected to impact ambient air temperature as well as the precipitation magnitude, frequency, and patterns, which in turn can affect the raw wastewater influent temperature and flow for the wastewater treatment plants (WWTPs). The seasonal variation of wastewater influent flow and its characteristics can also be impacted by changes in snow melting periods and drought/flood situations under climate change. As the flow and temperature variations of the raw wastewater influent can have implications for the efficacy and effectiveness of wastewater treatment processes, it is essential to study their impacts on the treatment processes in WWTPs. This study is focused on the secondary treatment processes and investigates the removal efficiency of the conventional activated sludge (AS) system and modified Ludzack-Ettinger (MLE) system under low and high flow influent conditions. BioWin is used as a numerical model tool to simulate the secondary treatment processes under various low and high flow scenarios. The change in temperature, solid retention time (SRT) and influent flow are used to design various simulation scenarios. The results indicated that the wastewater temperature can significantly impact the removal of the total chemical oxygen demand (COD), carbonaceous biochemical oxygen demand (cBOD), and total ammonia for both systems. At the same time, SRT was shown as a critical parameter for total phosphorous (TP) removal. The influent removal efficiency for total COD and cBOD for the conventional AS system was higher for high flow (HF) at 5 ℃ and vice versa at 25 ℃ for the conventional AS system. In comparison, the removal efficiency for total COD and cBOD

for the MLE system for low flow was higher than the high flow (5-25 ℃). This study also suggested that increasing SRT may not always improve effluent quality. This study provides insights into the potential impact of influent flow and wastewater temperature on the secondary treatment systems.

Keywords: influent flow, high flow (HF), low flow (LF), SRT, wastewater temperature, removal efficiency, conventional AS system, MLE system

5.1 Introduction

Wastewater treatment plants (WWTPs) are vital elements of the urban infrastructure to protect public health. In Canada, Wastewater Systems Effluent Regulations (WSER), established in 2012 under the Fisheries Act, is a federal body that sets national baseline effluent quality standards that could be achieved by secondary wastewater treatment (WSER, 2012). As per guidelines of WSER, the minimum standard for cBOD 25 mg/L, total suspended solids (TSS) 25 mg/L, total residual chlorine 0.02 mg/L and unionized ammonia at 15 ℃ $\pm$ 1 ℃, should be less than 1.25 mg/L.

Bush & Lemmen (2019) mentioned in Canada's climate change report that based on various climatic models, the average surface temperature may rise by 1.5 ℃-2.2 ℃ by 2050. By 2100, the average surface temperature in Canada may increase by 4 ℃. Changes in climatic conditions are expected to heavily damage infrastructure and various ecosystems (Reidmiller et al., 2017). Heavy rainfall due to climate change could lead to an overflow of untreated wastewater from the combined sewer systems to the local water bodies (Shahabadi et al., 2009). This eventually contaminates the rivers and lakes.

Climate change may severely impact the urban wastewater infrastructure due to more variation in the temperatures and precipitations (Zouboulis et al., 2015). Langeveld (2013) predicted that due to climate change, the winter would tend to be wetter because of heavy rainfall. Hughes (2021) suggested that extreme climatic conditions such as heavy rainfall could increase contaminants. Loss of infrastructure and advanced treatment plant service are a few of the implications of climate change for wastewater treatment systems. Hughes (2021) also suggested that an increase in temperature could increase odors and frequent flooding events at treatment plants.

Abdulla et al. (2020) mentioned that temperature is an important parameter to maintain and improve the BOD removal efficiency from the system. Sperling Von (2007) noted that the temperature and pH play an essential role in secondary or biological treatments, which are vital for removing the majority of soluble deleterious materials such cBOD, ammonia and phosphorous. Rapid snowmelt due to an increase in ambient temperature can increase the influent flow (for combined sewers) and decrease the wastewater temperature (Ghanizadeh et al., 2001). Alisawi et al. (2020) mentioned that increasing temperature reduces the concentration of dissolved oxygen (DO). In addition, the growth of the microbial community is affected by the decrease in the DO (Metcalf et al., 1991). The increase in temperature may also result in an increase in the effluent TSS concentration as the total COD increases. However, the increasing temperature does not directly increase the effluent TSS. The nitrification rate is improved by rising temperature, which enhances the total ammonia removal (Alisawi et al., 2020). Alisawi et al. (2020) also suggested that the TP removal is decreased with the increase in temperature.

Kim et al. (2017) specified that the drought-like condition (low flow) can worsen the wastewater influent characteristics and elevate the strength of wastewater. While climate events like rainstorms and floods, the rise in the influent flow, wastewater characteristics such as BOD and TSS concentrations dropped due to the dilution of wastewater. Low flow conditions increased the wastewater strength, and hence, the increasing problem of corrosion and clogging in the sewer was observed (O'Neill II et al., 2010).

The main objective of this study is to investigate the impact of variation in temperature and influent flow on two biological systems: conventional AS system and the MLE system. Numerical modelling is used as an inspection tool to simulate various flow scenarios for different SRT and temperature ranges. Numerical modelling accounts for the input wastewater characteristic and is based on the biological system and its set stochiometric and kinetic parameters, and it provides the effluent wastewater characteristics. In addition, we investigated how the change in SRT corresponding to different temperatures and influent flow impacts the removal efficiency. The outcome of this study provides an insight into the removal efficiency of the two biological systems impacted under different flow and temperature conditions.

5.2 Materials and methods

BioWin, developed by International Water Association, is a numerical modelling tool based on various AS models to study different wastewater systems and various simulation scenarios (Gernaey et al., 2004). Due to the snowmelt, the wastewater inflow and temperature were observed impacted. Data were analysed from the WWTP across Ontario for the wastewater temperature range, and hence, 5-25 °C was considered. The SRT range considered for this study was 3-11 days.

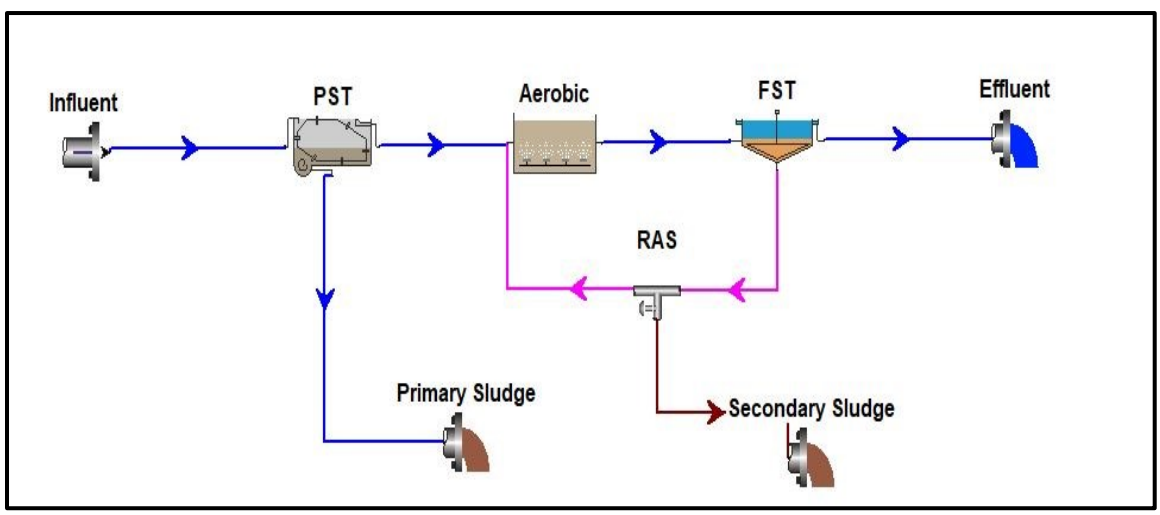
To formulate different influent flow conditions, the six-year influent flow of a municipality in Ontario was analysed. The municipality considered for this study had a separated wastewater collection system for sewage and stormwater. Also, the wastewater inflow was monitored, and no infiltration occurred in the system. Three influent flow conditions with low, average, and high flows were identified. To obtain the initial condition for the raw influent characteristics in the model, the average flow scenario with known wastewater characteristics was considered as the baseline, and the wastewater characteristics for the low and high flow scenario was obtained based on the percentage change in the influent flow relative to the average flow condition. The influent flow, as well as the wastewater characteristics corresponding to the respective flow condition, are presented in Table 5.1. The wastewater influent characteristics were significantly higher in the low flow condition compared to the high flow condition. For the high flow condition with respect to average flow condition, Wastewater characteristics such as total COD, cBOD, TSS and total ammonia were lower comparatively.

This study considered the conventional AS system and the MLE system under low flow high flow conditions. Kinetic and stochiometric parameters were modified based on the strength of the wastewater. Hence, these parameters were set for low flow and high flow conditions to obtain the required influent wastewater characteristics. The default range acceptable for these parameters was mentioned in the IWA manual. The kinetics and reactivity of the microbial community behave differently for low and high flow condition. Table A2 and Table A3 specify the wastewater parameters considered for the numerical simulations. Several simulation scenarios were conducted where the temperature (5-25 °C) and SRT (3-11 days) varied for each flow condition (high flow and low flow). The

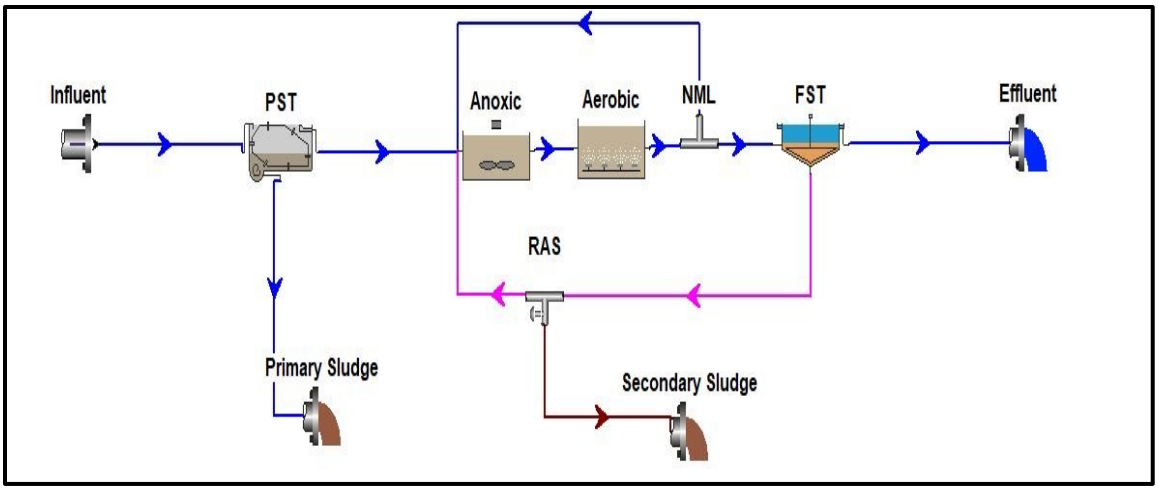
conventional AS system comprises of primary settling tank (PST), followed by an aerobic bioreactor, and lastly, the final settling tank (FST), which is shown in Figure 5.1 (a). The DO level is set at 2.0 mg/L. The return activated sludge (RAS) was recirculated from FST to the aerobic bioreactor. The rest of the activated sludge was considered as waste activated sludge (WAS) from the RAS. MLE has the same configuration as the previous system with the exception that it has an anoxic bioreactor preceding the aerobic bioreactor. Figure 5.1 (b) shows that there is a return stream of nitrifying mixed liquor (NML) after the anoxic and aerobic bioreactors (before the FST). Simultaneous denitrification (anoxic bioreactor) and nitrification (aerobic bioreactor) occur in this system.

Table 5.1: Influent wastewater characteristics

Type of flow	Flow m^3/d	Total COD, mg/L	cBOD, mg/L	TSS, mg/L	Total ammonia, mg/L	TP, mg/L
Low	42600	1205.00	518.59	769.34	69.40	18.33
Average	100000	500.00	245.21	242.71	26.40	6.50
High	123600	357.14	159.64	190.5	24.88	7.94



(a)



(b)

Figure 5.1: Schematic view of (a) conventional AS system, (b) MLE system

5.3 Results and Discussion

5.3.1 Conventional AS system

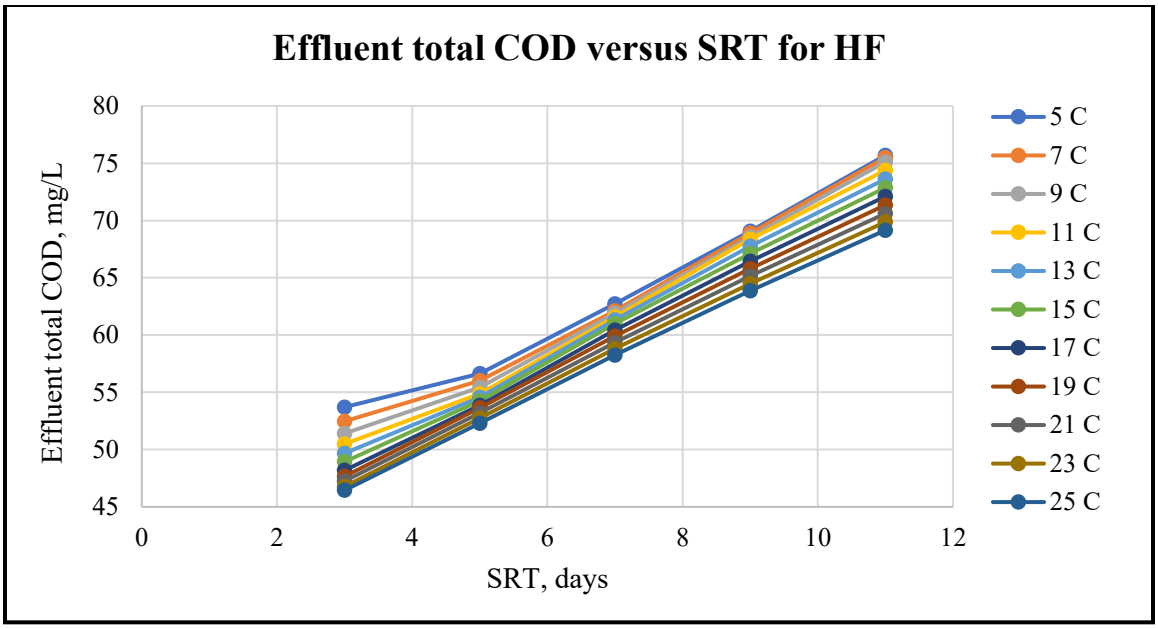
While the flow, temperature and SRT varied for all the scenarios, all other parameters such as food / microorganism (F/M) ratio, DO, specific growth rate and all the kinetic and stoichiometric parameters were kept consistent. total COD, cBOD, TSS, total ammonia and TP were the wastewater parameters studied for both secondary treatment systems.

5.3.1.1 Total COD

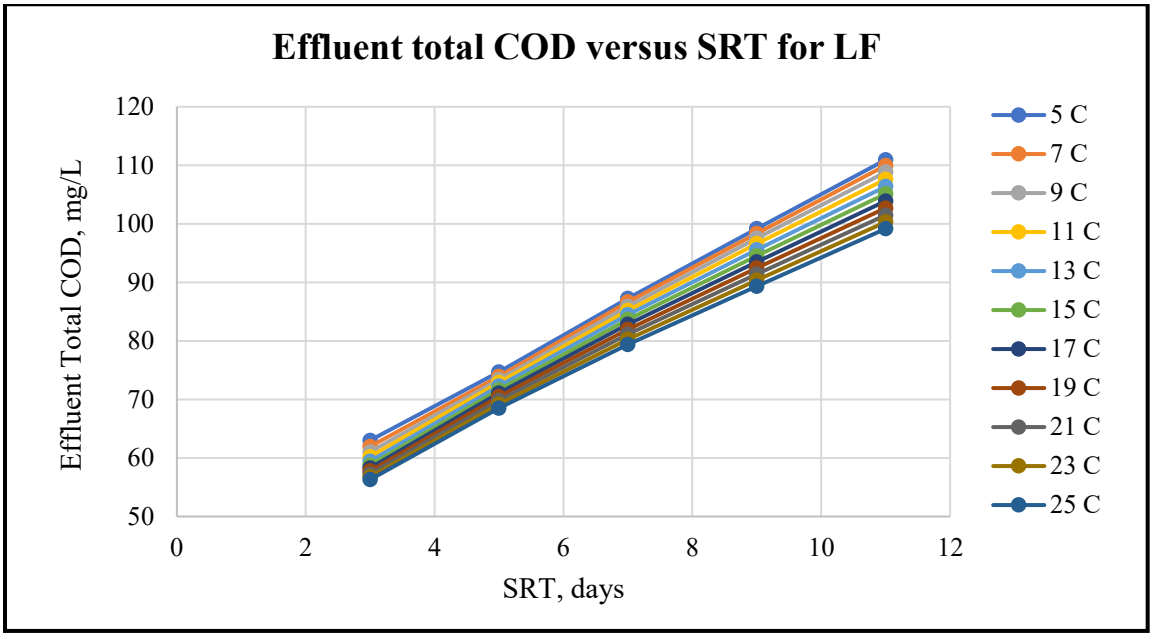
A conventional AS system was adopted to remove carbon (cBOD and total COD) and TSS. As shown in Figure 5.2, with the increase in SRT, the F/M ratio decreased. Phillips (1998) mentioned that the increase in SRT led to the rise in polysaccharides, which resulted in the release of the extracellular polymers and, furthermore, increased the effluent total COD. Also, the DO was kept the same throughout for all the simulations for high flow and low flow. Hence, this increases effluent total COD subsequently (Liu et al., 2015). It was also evident that an increase in temperature increases the biological activity of the microorganisms.

The increase in the total COD was observed for both high flow and low flow with the rise of SRT. In Figure 5.2 (a), with the rise in temperature from 5-25 °C for high flow, the removal efficiency improved by 11.6% for 11 days SRT. For SRTs as short as 3 days, the effluent total COD was 54 mg/L and 46 mg/L for 5 °C and 25 °C respectively. For 3 days SRT, the removal efficiency was improved by almost 14.8% while elevating the temperature from 5-25 °C, in Figure 5.2 (a). Hence, elevated temperature projected better removal efficiency.

The low flow effluent total COD is presented in Figure 5.2 (b). It showed the same trend as high flow. An increase in SRT deteriorated the effluent quality. Cold temperatures constrain the biological activity of the heterotrophic bacteria. Hence, increasing the temperature has proven advantageous for the removal efficiency. For instance, effluent total COD at 11 days of SRT was nearly 112 and 98 mg/L for 5 °C and 25 °C, respectively. The removal efficiency was improved for 25 °C by 12.5% compared to 5 °C. However, for 3 days, the removal efficiency was improved by approximately 14.06 % at 25 °C compared to 5 °C. There was a minor drop in removal efficiency for low flow compared to high flow. The initial wastewater parameter concentrations were significantly high for low flow conditions than the high flow conditions. This could be the reason for reduced removal efficiency for different temperature and SRT ranges.



(a)°C



(b)

Figure 5.2: Effluent total COD for conventional AS system (a) HF, (b) LF

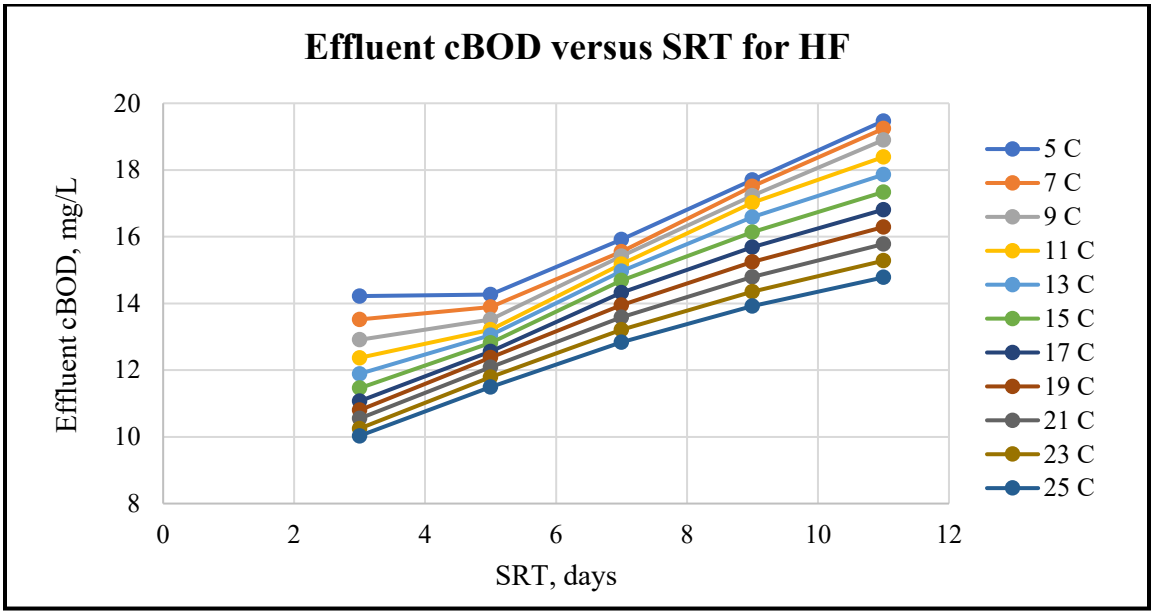
5.3.1.2 cBOD

For high flow, cBOD removal for 5 °C was almost constant initially due to the inactivity of microorganisms because of low temperature. With the increase in temperature, the

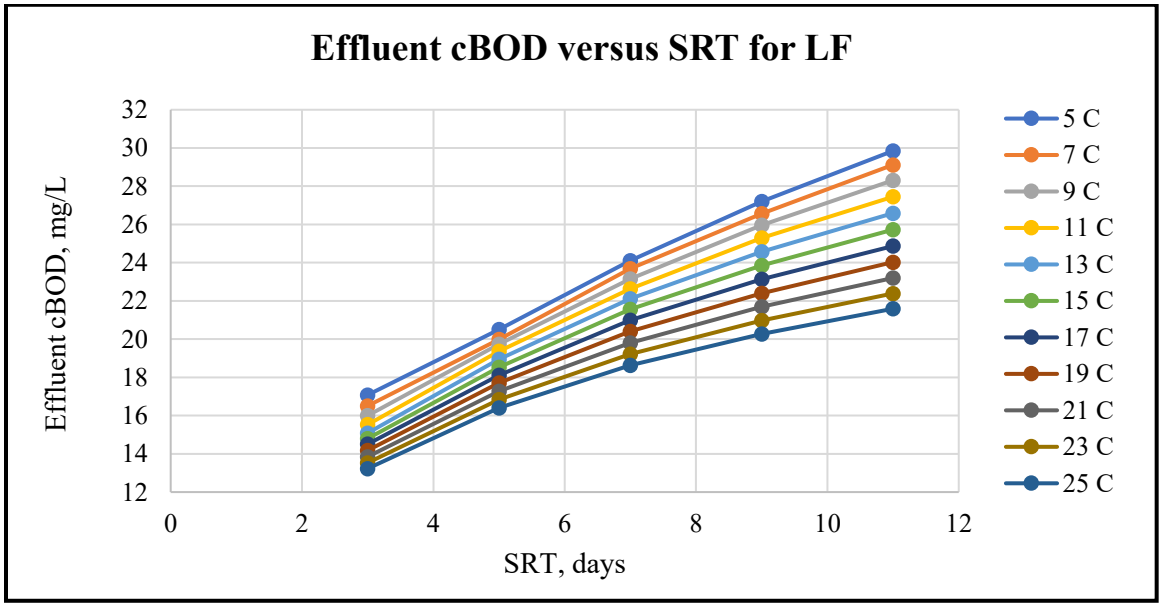
biological activity of the microbial community proliferated. It led to better removal efficiency of cBOD. As the temperature elevated to 25 °C, the removal efficiency was improved by around 29.5% at 3 days SRT. At 11 days, the removal efficiency for 5 °C dropped by 25.5% compared to 25 °C, which is shown in Figure 5.3 (a). Figure 5.3 (a) also presents that with the continuous increase in SRT from 3-11 days, cBOD removal efficiency was decreased by 38% and 46% for 5 °C and 25 °C, respectively. Liu et al. (2015) mentioned that as the F/M ratio decreased and DO been the same, effluent wastewater characteristics shall be deteriorated with the increase in SRT.

Figure 5.3 (b) shows the effluent cBOD for LF. A similar trend for the removal of cBOD was observed for low flow in Figure 5.3 (b), as high flow shown in Figure 5.3 (a). For 5 °C, with an increase of SRT, cBOD continuously increased. This was true for all temperature ranges given. At 3 days, with the rise in temperature from 5 °C to 25 °C, removal efficiency jumped by 15.38%. At 11 days, the cBOD removal efficiency for 25 °C was 38% higher than 5 °C. Figure 5.3 (b) also presented that with the increase in SRT from 3-11 days, the effluent cBOD removal efficiency dropped by nearly 76.4% and 68.4% for

5 °C and 25 °C respectively, for 11 days SRT compared to 3 days SRT. Also, effluent cBOD values were lower for high flow compared to low flow conditions.



(a)



(b)

Figure 5.3: Effluent cBOD for conventional AS system (a) HF, (b) LF

5.3.1.3 TSS

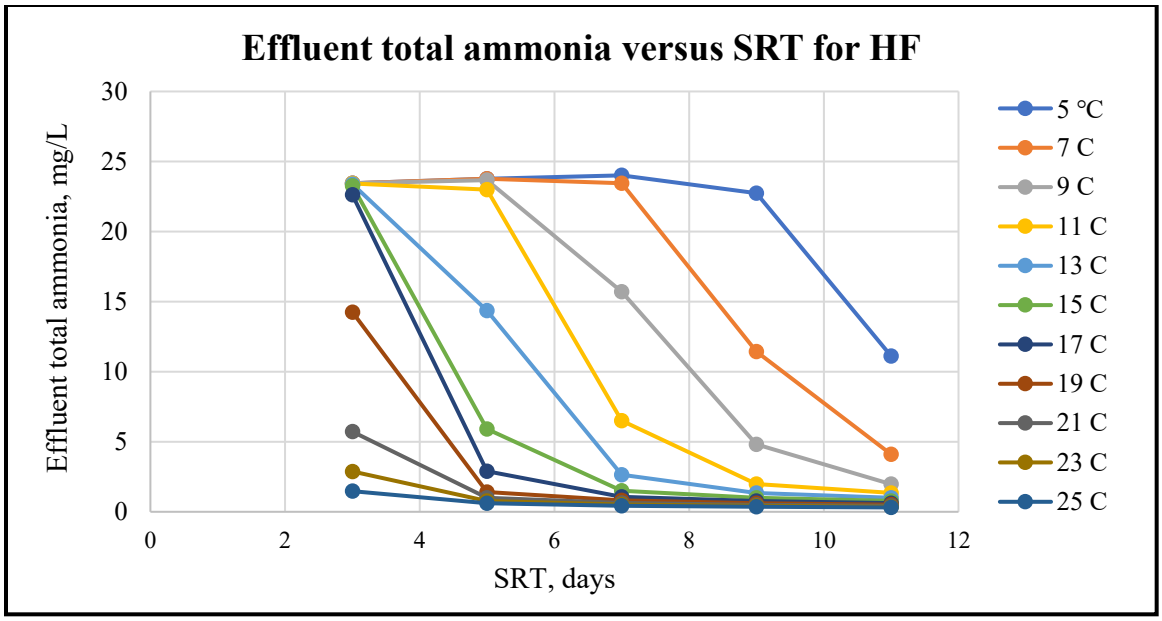
The results of TSS for the conventional AS system for high flow and low flow are presented in the appendix (see Figures A1). It was observed that an increase in SRT in both high flow and low flow deteriorated the effluent TSS quality. Increase in temperature did not significantly impact high flow and low flow removal efficiency. An increase in the total COD concentration as discussed in Figure 5.2 earlier, explained the increase in TSS concentration. The influent TSS concentration was higher for low flow compared to high flow because the strength of wastewater was much higher for low flow than the high flow conditions. Due to this, the effluent TSS values were higher for low flow than high flow, for the conventional AS system. Significant impact of temperature and flow change on the TSS removal efficiency was not observed.

5.3.1.4 Total ammonia

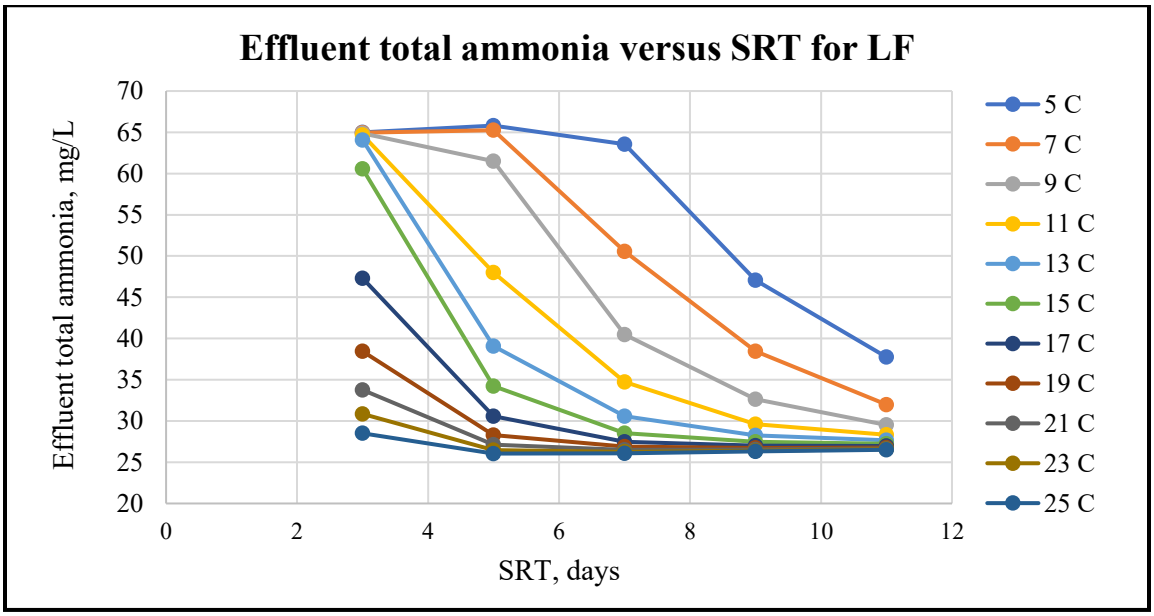
Nitrification occurs in the aerobic bioreactor. Nitrifying bacteria are sensitive to temperature and show inactivity for colder temperatures (T < 12 °C) (Zhang et al., 2019; Chen et al., 2018; Plósz et al., 2009). Chen et al., (2018) suggested that temperature plays an important role in reducing total ammonia from the mainstream. As the nitrification process started, it resulted in the reduction of ammonia in the effluent. In Figure 5.4 (a) for high flow, effluent total ammonia showed a minimal increase compared to the influent. This could be due to the constant DO level and no nitrification occurrence. According to research conducted by ECOS Environmental Consultants (2013), soluble cBOD is an inhibitor for the growth of nitrifying bacteria. Hence, high SRT was required for the growth of nitrifying bacteria. As shown in Figure 5.4 (a), at 7 days SRT, the nitrification was initiated for the colder temperature (5-7 °C). Smith et al. (2015) reported that with the

increase in temperature, nitrification occurs at shorter SRT Figure 5.4 (a) presents the same trend that at 11 °C, nitrification proliferated at 5 days SRT. Ekama (2010) reported that minimum 5 days SRT was required for the nitrification process. At 13 °C, the effluent total ammonia (5 days SRT) significantly dropped by 36.9% compared to shorter SRT (3 days SRT). For T > 19 °C, the effluent total ammonia was reduced by around 79 % compared to T < 19 °C for 3 days SRT. For T > 19 °C, with the increase in SRT from 3-11 days, the effluent total ammonia value almost plateaued. Comparing 5 °C and 25 °C, the removal efficiency of total ammonia for 25 °C was improved by 95.8 % and 90.9 % at 3 and 11 days SRT, respectively.

Figure 5.4 (b) presents the results of total ammonia removal for low flow scenario. For 5 °C, at shorter SRT (3-7 days), the nitrification rate was negligible or minimal. Hence, initially, for 5 °C, the effluent was higher than the total influent ammonia for 5 days SRT. With the increase in SRT, the nitrification rate increased and eventually, the total effluent ammonia showed a downward trend. At 11 °C, the nitrification initiated at 3 days SRT. As shown in Figure 5.4 (b), for 13 °C, the effluent total ammonia at 5 days SRT was nearly 39% lower than effluent total ammonia at 3 days SRT. Also, for T > 19 °C, the effluent total ammonia (3 days SRT) was nearly 52% less than T < 19 °C. Comparing 5 and 25 °C for low flow, the removal efficiency of total ammonia was improved by 56.9 % and 42.5 % at 3 and 11 days SRT, respectively.



(a)



(b)

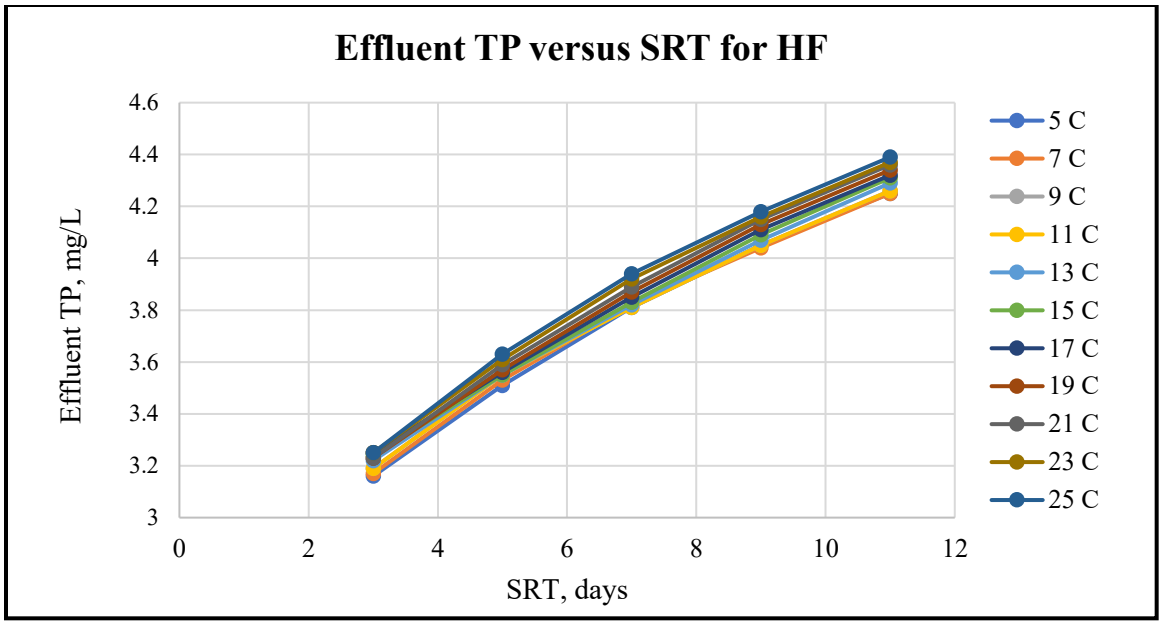
Figure 5.4: Effluent total ammonia for conventional AS system (a) HF, (b) LF

5.3.1.5 TP

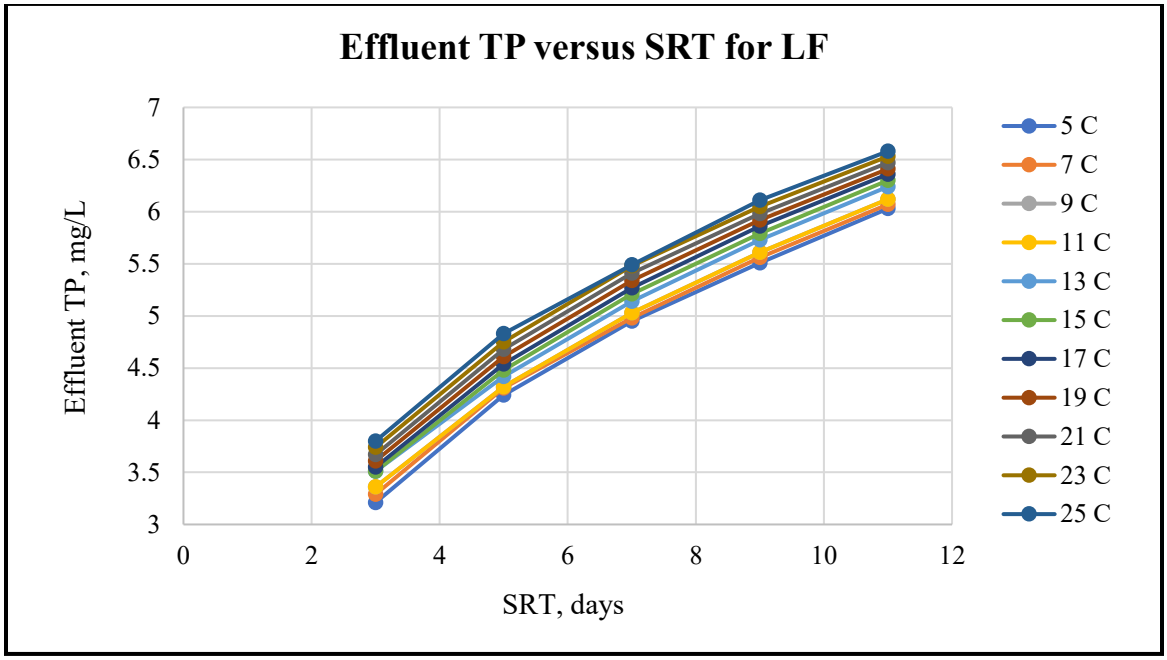
The removal of phosphorous is initiated in an anaerobic bioreactor. Usually, the absence of oxygen in the bioreactor (anoxic/anaerobic) prevails an environment for the biological

phosphorous removal (BPR). Figure 5.5 (a) represents effluent TP concentration for a conventional AS system for the high flow condition. As shown in Figure 5.5 (a), effluent TP concentration for 5-25 °C had minimal difference (3.1-3.3 mg/L). As there was an aerobic bioreactor in the conventional AS system, the BPR deteriorated with an increase in SRT from 3-11 days for all temperature ranges (5-25 °C). According to Sayi-Ucar et al. (2015), temperature drop does not aid in the removal of phosphorous. Polyhydroxyalkanoates (PHA) were consumed by the phosphorous accumulating organisms (PAOs) and converted the orthophosphate to phosphate. This prevented the assimilation of phosphorous in the effluent (Esfahani et al., 2018). It was also Chua et al. (2003) that mentioned PHA was produced more at shorter SRT compared to higher SRT (10 days). This may be a reason for the removal of TP at a shorter SRT. At 11 days SRT, the removal efficiency compared to 3 days SRT for 5 °C and 25 °C dropped by almost 33.33% and 37.07 % for high flow, respectively.

The temperature increase deteriorated the removal of TP, as shown in Figure 5.5 (b). At 3 days SRT, the effluent TP concentration was nearly 3.3-3.9 mg/L for 5-25 °C. As the SRT increases, the effluent TP concentration increased for all temperatures. As shown in Figure 5.5 (b), at 11 days SRT, the TP effluent removal efficiency, compared to 3 days SRT, dropped by nearly 93.5 % and 73.6% for 5 °C and 25 °C, respectively.



(a)



(b)

Figure 5.5: Effluent TP for conventional AS system (a) HF, (b) LF

5.3.2 MLE system

In the MLE, there is an anoxic bioreactor before the aerobic bioreactor. The nitrifying mixed liquor (NML) was recirculated from the splitter to the anoxic bioreactor, shown in

Figure 5.1 (b). Simultaneous denitrification and nitrification in an anoxic and aerobic bioreactor removed nitrogen gas (converted from total ammonia) from the system. MLE is designed explicitly to improve the removal efficiency of total ammonia (Kutty et al., 2011), while the anoxic bioreactor aids in the removal of phosphorous.

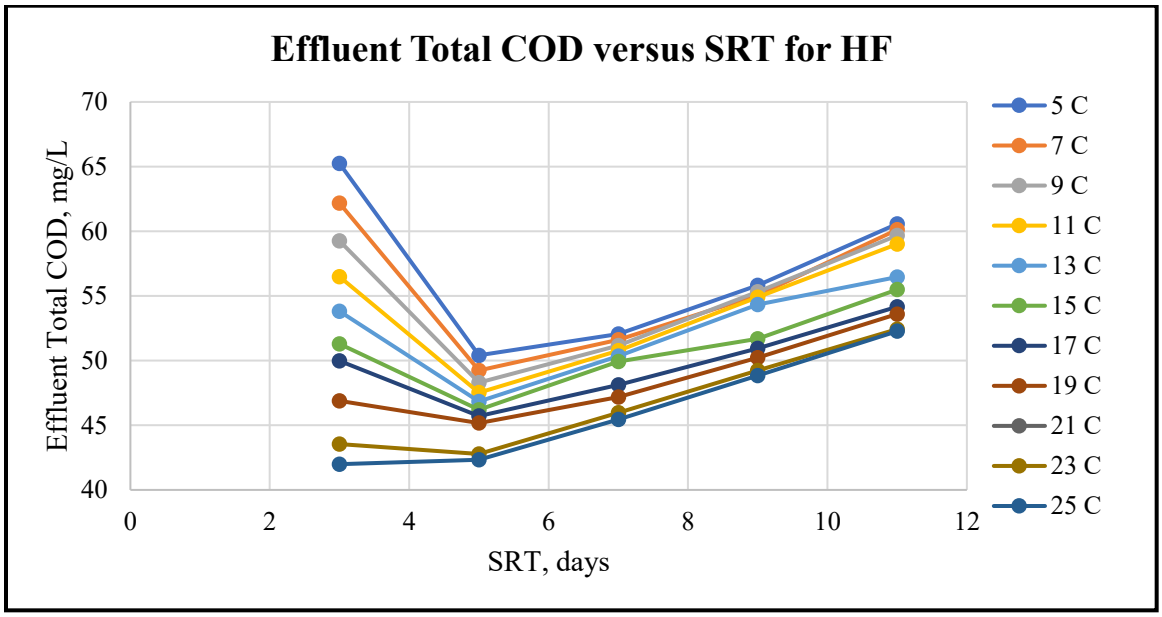
5.3.2.1 Total COD

Figure 5.6 (a) presents the removal of total COD from the MLE system. The biological activity of the microbial community is dependent on temperature. Fluctuation in temperature hampers their biological activity. In Figure 5.6, the effluent total COD was lower for warmer temperatures than colder temperatures. For instance, the effluent total COD concentration at 3 days SRT for 25 ℃ was 30.76 % lower than the concentration at 5 ℃. At low SRT, the settleability of the flocs was low. Therefore, as the SRT was increased from 3 to 5 days, there was a drop in the effluent concentration for total COD for the complete temperature range. But, as the temperature increased, subsequently it also increased the biological activity and eventually increased the settleability of the flocs.

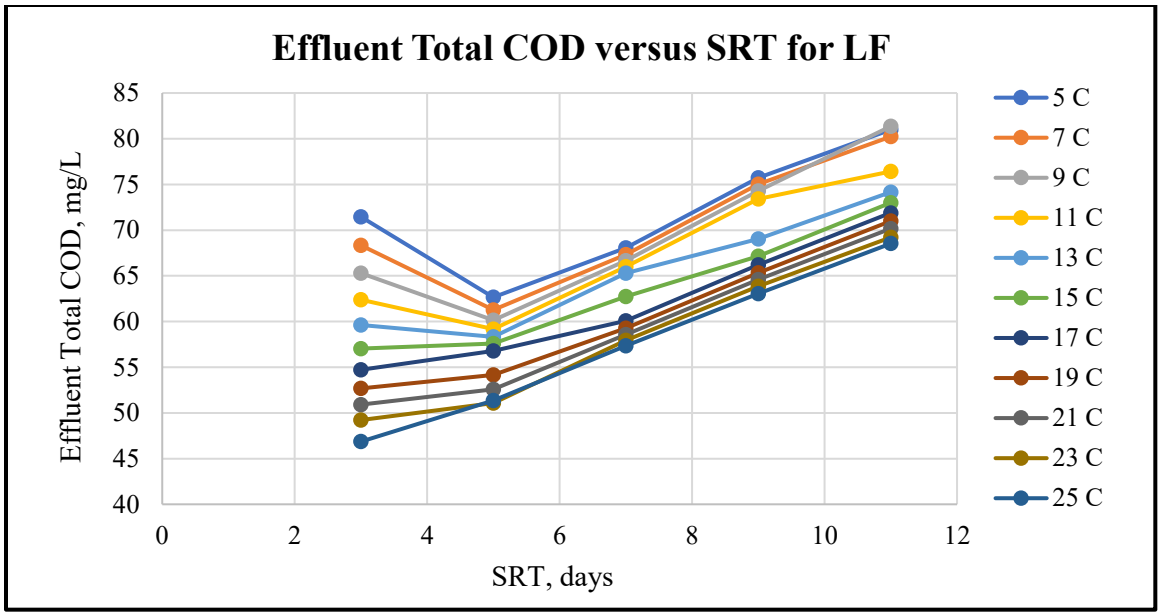
Furthermore, as the SRT increased, the effluent total COD increased with the decrease in F/M ratio, and the same DO level (Liu et al., 2015). At 11 days SRT, the effluent total COD for 25 ℃ was 14.75 % lower than effluent total COD for 5 ℃. Also, as shown in Figure 5.6 (a), the effluent total COD for 5 ℃ at 11 days was 6.15 % lower than effluent total COD at 3 days SRT. Conversely, the effluent total COD for 25 ℃ at 11 days SRT was 23.8 % higher than effluent total COD at 3 days SRT.

For low flow, as shown in Figure 5.6 (b), a similar trend as in Figure 5.6 (a) was observed. Initially, at 3 days SRT the effluent total COD concentration was relatively higher than the effluent total COD concentration at 5 days SRT. Also, at 5 ℃, the concentration was

somewhat higher than at 25 ℃, for 3 days SRT. With the increase in SRT from 5 -11 days, the effluent total COD gradually increased due to limited F/M ratio and same DO. At 3 days, the effluent total COD at 5 ℃ was nearly 53.19% higher than effluent total COD at 25 ℃. Similarly, the effluent total COD at 11 days SRT for 5 ℃ was 22.3% higher than effluent total COD concentration at 25℃. Figure 5.6 (b) showed that the effluent total COD at 11 days SRT for 5℃ and 25℃ was 13.9% and 42.5 % higher than the effluent total COD at 3 days SRT, respectively.



(a)



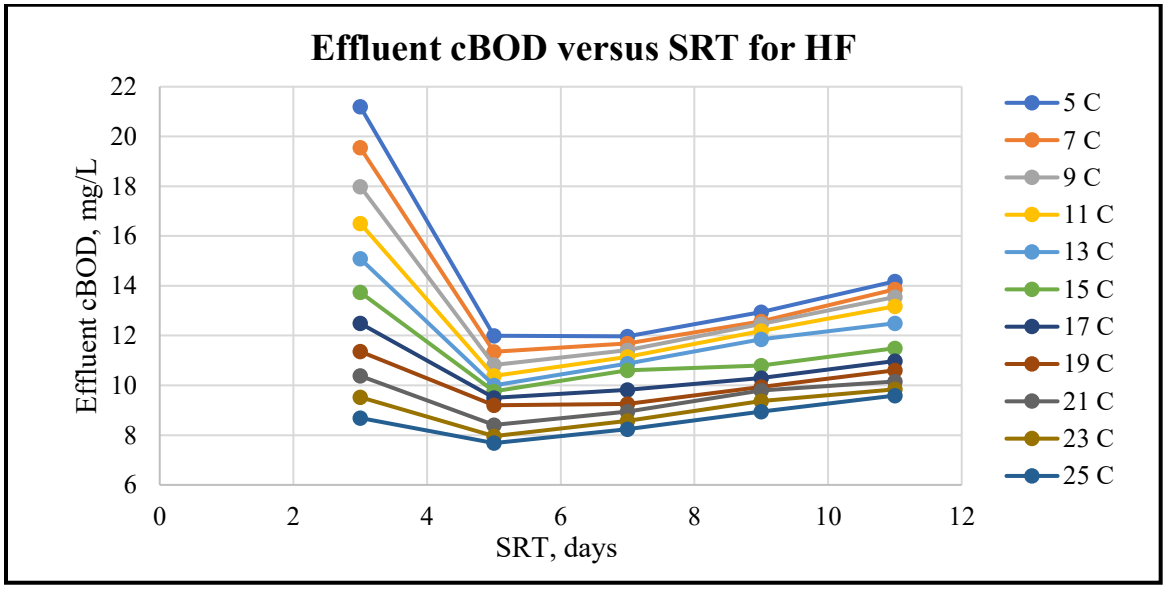
(b)

Figure 5.6: Effluent Total COD for MLE system (a) HF, (b) LF

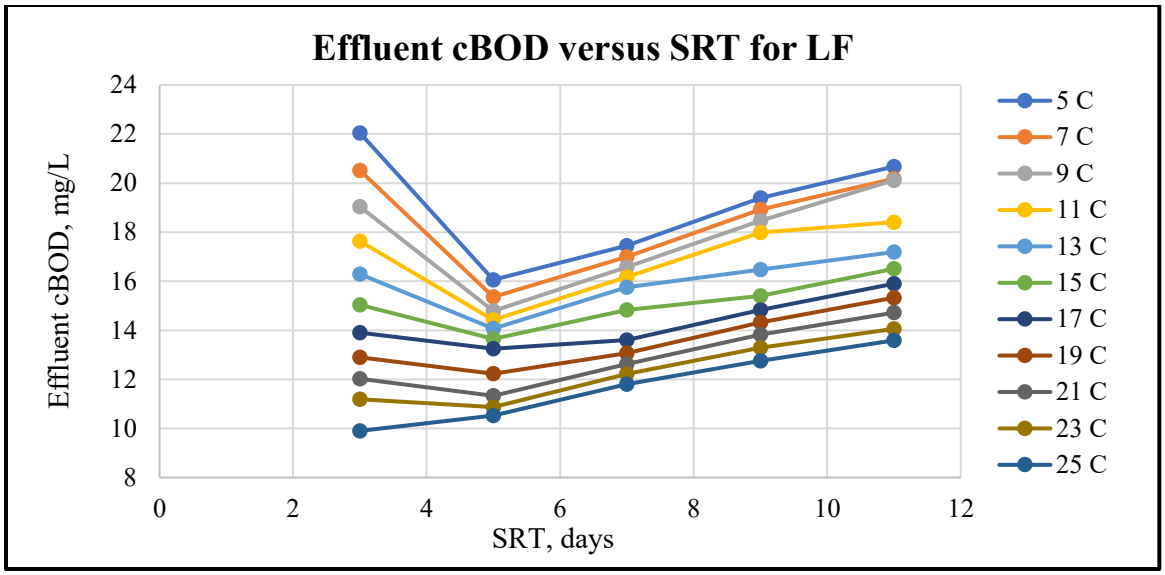
5.3.2.2 cBOD

High flow condition effluent cBOD for the MLE system is represented in Figure 5.7 (a). The Figure 5.7 (a) shows, the cBOD removal followed a trend like the total COD in Figure 5.6 (a). Initially, at 3 days SRT, the effluent cBOD was relatively higher for 5-25 ℃ than 5 days SRT for the same temperature range. With the increase of SRT, initially, the settleability increased, which resulted in the drop of effluent cBOD. But, with limited F/M ratio and same DO, eventually, the effluent cBOD increased. An increase in temperature improved the effluent quality, as shown in Figure 5.7 (a). At 3 days SRT, effluent cBOD at 25 ℃ was 59.6% lower than effluent cBOD at 5 ℃. Similarly, at 11 days SRT, effluent cBOD at 25 ℃ is 30 % lower than effluent cBOD at 5 ℃. For 5 ℃, the effluent at 11 days SRT was improved by 35.7 % than the effluent cBOD at 3 days. Likewise, for 25 ℃, the effluent cBOD at 11 days was deteriorated by 19.5 % than effluent cBOD at 3 days.

Effluent cBOD for low flow in Figure 5.7 (b) showed a trend like total COD in Figure 5.6 (b). For 5-21 ℃, the effluent at 3 days SRT was higher than the effluent at 5 days SRT. But at 25 ℃, the effluent gradually increased with an increase in SRT from 3 to 11 days. At 3 days SRT, effluent cBOD at 25 ℃ was 22 % lower than effluent cBOD at 5 ℃. Similarly, at 11 days SRT, as shown in Figure 5.7 (b), effluent cBOD at 25 ℃ was 34.28% lower than effluent cBOD at 5℃. At 5 ℃, the effluent at 11 days SRT was improved by 4.5 % than the effluent cBOD at 3 days. Likewise, for 25 ℃, the effluent cBOD at 11 days deteriorated by 19% than effluent cBOD at 3 days.



(a)



(b)

Figure 5.7: Effluent cBOD for MLE system (a) HF, (b) LF

5.3.2.3 TSS

Hollas et al. (2019) reported that in the MLE system there was rise in TSS concentration with the increase in SRT in the anoxic and aerobic bioreactor. Hence, for both high flow and low flow for the MLE system, there was a continuous rise with the increase in SRT. Also, with the increase in temperature, the effluent values at 25 °C were lower compared to effluent TSS concentrations at 5 °C. This trend could be explained with respect to the COD concentrations. The biological activity increased, and hence the removal efficiency was improved. The figures depicting the effluent TSS concentrations for high flow and low flow for the MLE system are provided in the appendix (Figure A.2).

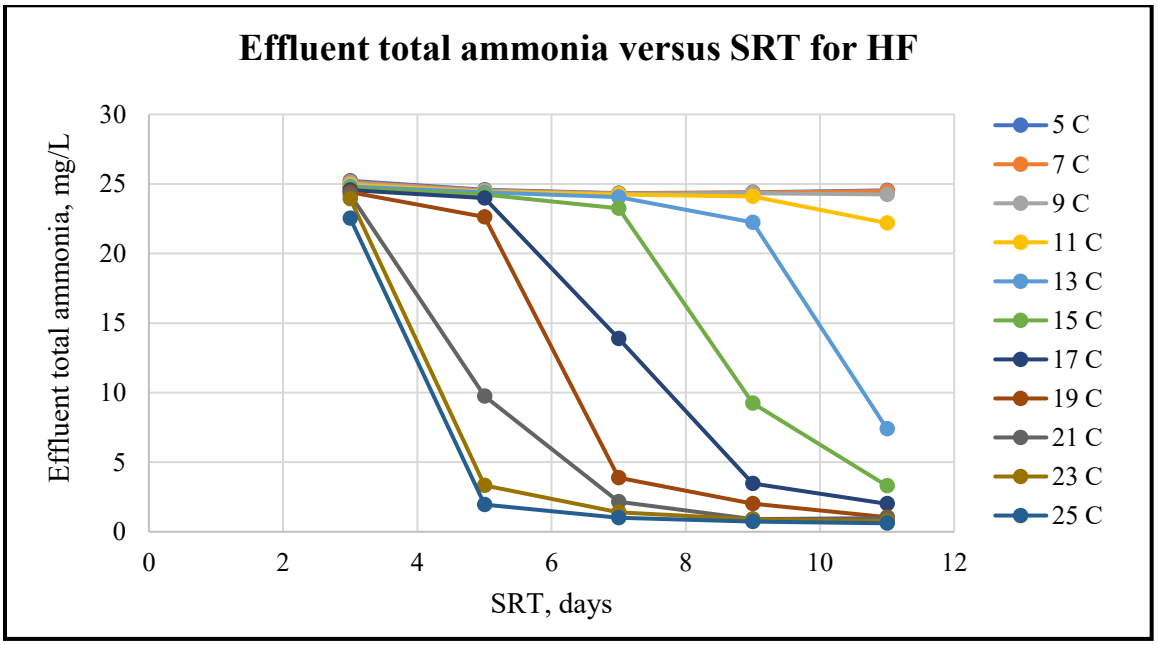
5.3.2.4 Total ammonia

The MLE system involves simultaneous denitrification and nitrification in the anoxic and aerobic bioreactor. Organic carbon is the main parameter that initiates the denitrification process in the anoxic bioreactor (He et al., 2018). Organic carbon could be used as an

electron donor in the denitrification process (Hollas et al., 2019). Further, the oxidation of nitrate to nitrogen gas reduced effluent total ammonia from the system. In the nitrification process, ammonia is oxidized to nitrite and nitrate. The final product of nitrification is nitrate. In the MLE system, this nitrate was recycled to anoxic bioreactor and denitrification occurs. Figure 5.8 (a) presents that for colder temperatures (5-11 °C), nitrifiers were inactive even with 11 days SRT. Figure 5.8 (a) shows that 13 °C was a critical temperature for the nitrification process. It led to the reduction of effluent total ammonia from the mainstream. Temperature and SRT are essential parameters for nitrifiers to proliferate. Nitrogen oxidizing bacteria (NOB) performed well at higher SRT and warmer temperatures (T > 13 °C). Soluble cBOD could act as an inhibitor in the process of nitrifiers because nitrification occurs in the cell. Soluble cBOD may enter the cell and disrupt the nitrification process. Also, there was a competition for uptake between cBOD and nitrifiers for limited DO. For T < 19, the drop in effluent total ammonia from 3 days to 5 days SRT was nearly 4 %. Whereas for T > 19 °C, the decrease in effluent total ammonia concentration from 3 to 5 days SRT was almost 84 %. Furthermore, increasing SRT (3-11 days) for T > 19 °C, the curve for effluent concentration of total ammonia almost flattened. Comparing 5 °C and 25 °C, the removal efficiency of total ammonia at 25 °C was better by 8 % and 95.50 % at 3 and 11 days SRT, respectively.

For low flow for the MLE system, Figure 5.8 (b) showed a trend similar to Figure 5.8 (a). With the increase in SRT and temperature, effluent total ammonia concentration dropped. At 13 °C, the nitrification process was evident for 7 days SRT. After that, for T > 13 °C, nitrification initiated at 5 days SRT. The drop in the effluent total ammonia (for 5 days SRT compared to 3 days) for T < 19 °C was 9.85 %. Whereas, for T > 19 °C, the drop in

effluent total ammonia as the SRT increased from 3 to 5 days, was around 85.83%. As shown in Figure 5.8 (b), for T > 19 °C, as the value of SRT increased, effluent total ammonia concentration became stagnant. If 5 °C and 25 °C are compared, the removal efficiency of total ammonia at 25 °C was improved by 58.33 % and 94.11 % at 3 and 11 days SRT, respectively.



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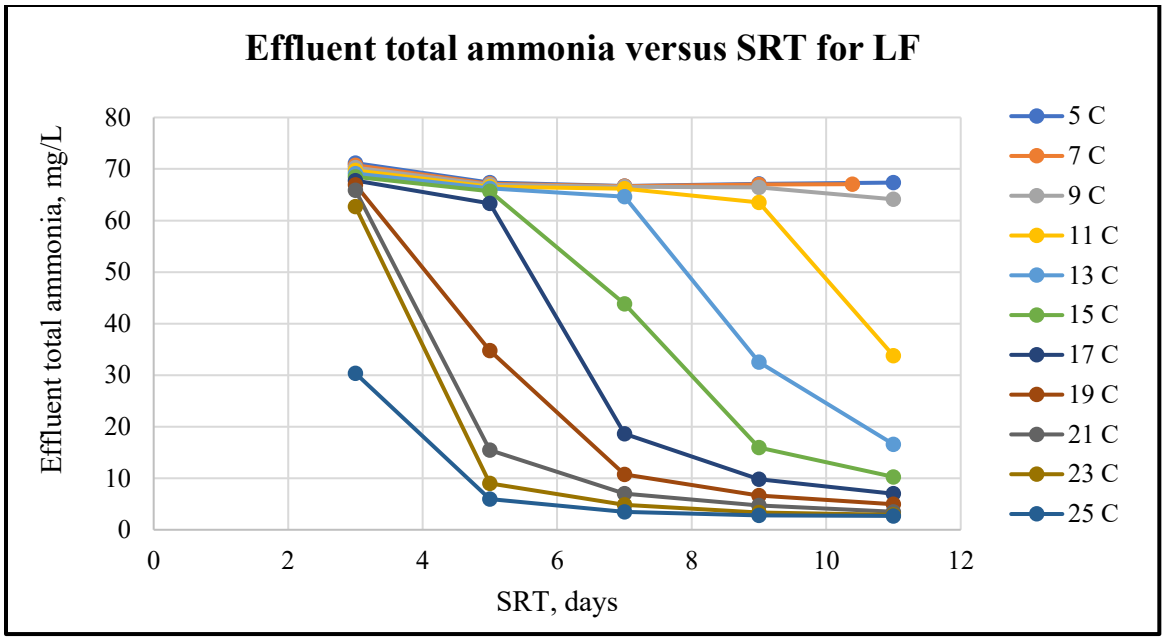


(b)

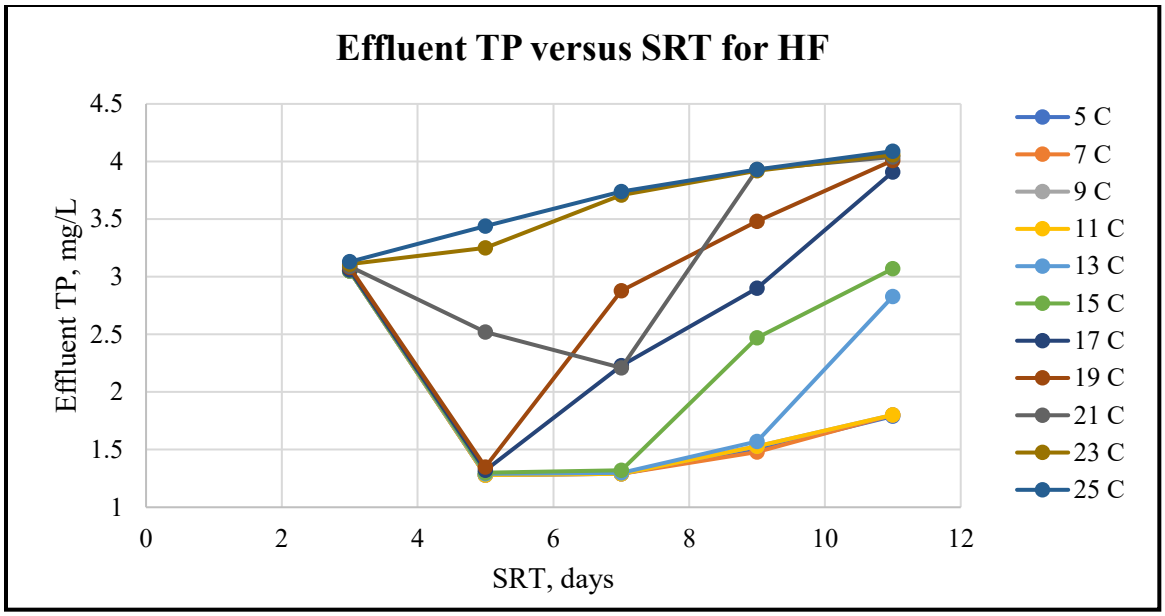
Figure 5.8: Effluent total ammonia for MLE system (a) HF, (b) LF

5.3.2.5 TP

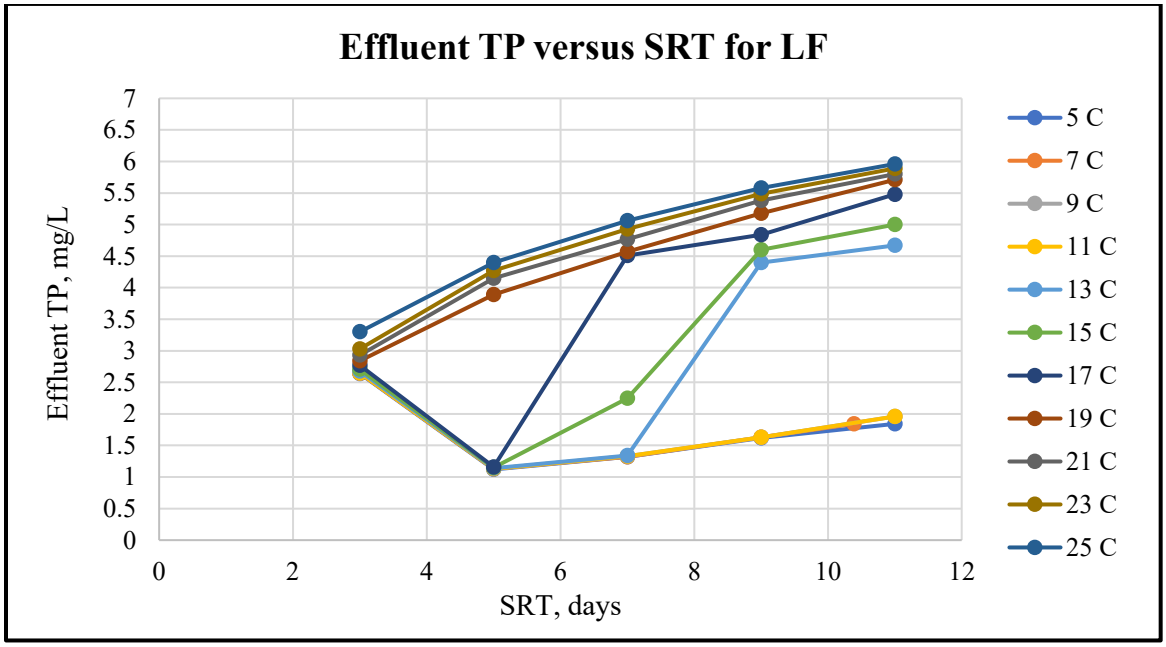
The nitrate and oxygen in the anoxic and aerobic bioreactor act as the electron acceptor in the MLE system (Esfahani et al., 2018). As shown in Figure 5.9, for lower temperatures (T < 13 °C), the nitrifiers were inactive. Hence, the PAOs dominated nitrifiers and anoxic zone. It becomes a favourable condition for the PAOs to consume poly-β-hydroxyl-alkanoates (PHA) and polyhydroxy butyrate (PHB). It converts orthophosphate to polyphosphate and stores them (Esfahani et al., 2018). This, in return, reduced the effluent TP concentration. An increase in SRT aids the nitrifiers and glycogen accumulating organisms (GAOs) to grow. Sayi-Ucar (2015) discussed that GAOs could store volatile fatty acids (VFA) and compete with PAOs at higher temperatures. Hence, PAOs were being dominated by GAO at higher temperatures. Initially, the increase in overall SRT increased the local SRT for the anoxic zones. Consequently, for T < 19 °C, with the rise in the overall SRT from 3-5 days, the effluent TP for 5 days had dropped by nearly 56.67%,

compared to 3 days SRT. Also, with an increase in temperature T > 21 °C, as evident in Figure 5.9 (a), the effluent TP increased by around 3.12 % at 5 days compared to effluent TP at 3 days. At 5 °C, as the SRT was increased from 3-11 days, the effluent TP concentration dropped by nearly 40 %. Similarly, as shown in Figure 5.9 (a), at 25 °C, with the increase in SRT from 3-11 days, the effluent TP increased by 21.80%.

For low flow for the MLE system, in Figure 5.9 (b), showed a similar trend as presented for high flow in Figure 5.9 (a). for T < 17 °C, with the increase in SRT from 3 to 5 days, a significant reduction in effluent TP was observed. As shown in Figure 5.9 (b), approximately 59.25 % of the drop in the effluent TP was observed for T < 17 °C, when SRT was increased from 3-5 days. For T > 17 °C, increase in SRT from 3-5 days increased the effluent TP by nearly 51.85 %. At 5 °C, when SRT was increased from 3-11 days, the effluent TP dropped by 29.6 %. While at 25 °C, an increase in SRT from 3-11 days, the effluent TP increased by 21.8%. Hence, it was observed that temperature increase deteriorated the effluent quality. Also, the initial increase in SRT for warmer temperatures improved the effluent TP quality.



(a)



(b)

Figure 5.9: Effluent TP for MLE system (a) HF, (b) LF

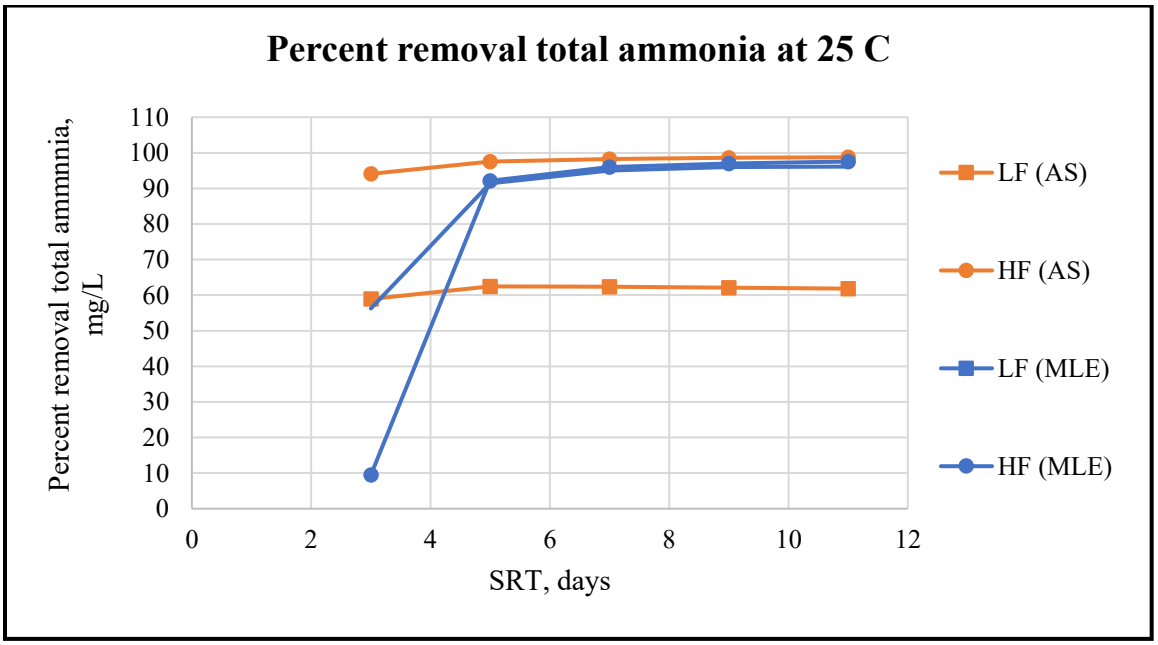
5.4 Comparison of conventional AS and MLE systems for different flow conditions

In this study, by comparing conventional AS system and MLE system, it was observed that change in influent flow (high flow or low flow), the removal efficiency of total ammonia

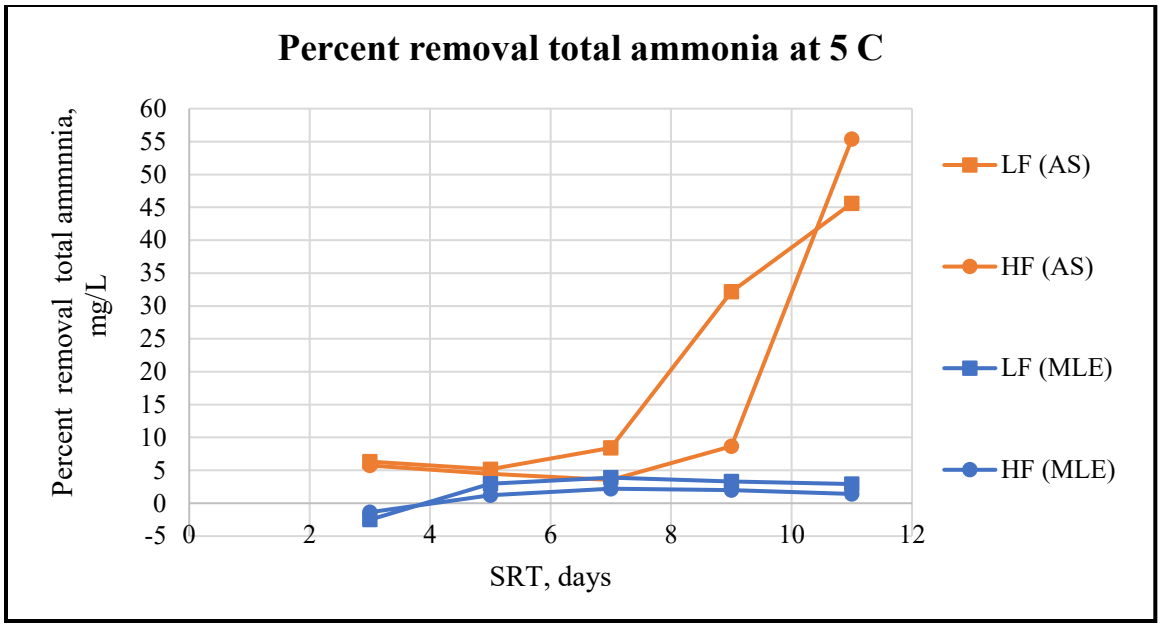
and TP were significantly impacted. Figures 11 and 12 represented the comparison of removal efficiency for total ammonia and TP, for conventional AS system and MLE system at 5 ℃ and 25 ℃, for high flow and low flow conditions. Figure 5.10 (a) represents the total ammonia removal efficiency at 25 ℃ for both the systems (for high flow and low flow conditions). In Figure 5.10 (a), at 25 ℃, the removal efficiency for total ammonia for low flow (conventional AS system) was nearly 35 % lower than high flow (for conventional AS system). The influent concentration for total ammonia for low flow was significantly higher than high flow. Hence additional modifications in the treatment system could subside this change. As shown in Figure 5.10 (a), for the MLE system, at 3 days SRT, the removal efficiency for high flow was around 10 %. Whereas for the MLE system for low flow, it was approximately 60 % at 3 days SRT. Shorter SRT could inhibit the growth of nitrifiers. After that, as the SRT increased from 3-11 days, high flow and low flow almost exhibited a similar removal efficiency trend. It shows that at 25 ℃, the MLE system for both high flow and low flow showed at par efficiency as conventional AS system high flow condition.

Figure 5.10 (b) represents the removal efficiency for total ammonia at 5 ℃ for both systems. As discussed previously, the nitrifiers are inactive at colder temperatures. In Figure 5.10 (b), for the conventional AS system, both high flow and low flow conditions exhibited the same trend at shorter SRTs. As the SRT increases from 3-9 days, low flow has better removal efficiency than high flow for the conventional AS system. But at 11 days SRT, high flow represents 10% higher removal efficiency than low flow for the conventional AS system. Whereas, for MLE system in Figure 5.10 (b), at 3 days SRT, it showed negative removal efficiency. This represented the increase in the effluent compared

to the influent (Figure 5.10 (b)). For the MLE system at 5 ℃, the nitrification process was ceased. The recycle of NML to the anoxic bioreactor in the MLE system may have increased the total ammonia in the effluent. With the increase in SRT from 5-11 days, the removal efficiency was almost the same (below 5%) for both high flow and low flow for the MLE system. Figure 5.10 (b) also shows the maximum removal efficiency at 5 ℃ was 55%, whereas maximum removal efficiency at 25 ℃ was around 96% for high flow (for both the systems). It was also observed in Figure 5.10 at 25 ℃ that the high flow showed higher removal efficiency for both systems. Also, low flow for MLE at 25 ℃ showed similar efficiency as high flow for the MLE system. At 5 ℃, the overall removal efficiency for both high flow and low flow for both the systems was significantly low compared to 25 ℃. The MLE system for high flow and low flow exhibited lower than 5 % removal efficiency, at 5 ℃. The increase in SRT improved the removal efficiency for high flow and low flow for the conventional AS system.



(a)



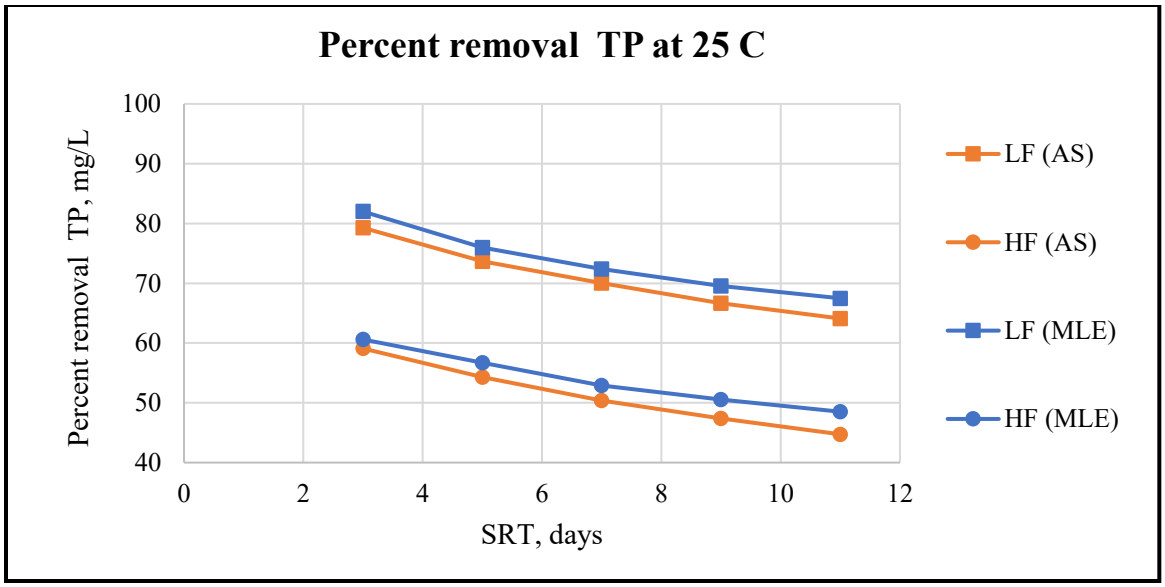
(b)

Figure 5.10: Comparison of removal efficiency of total ammonia for different systems and different flow scenarios for wastewater temperature of (a) 25 ℃ and (b) 5 ℃.

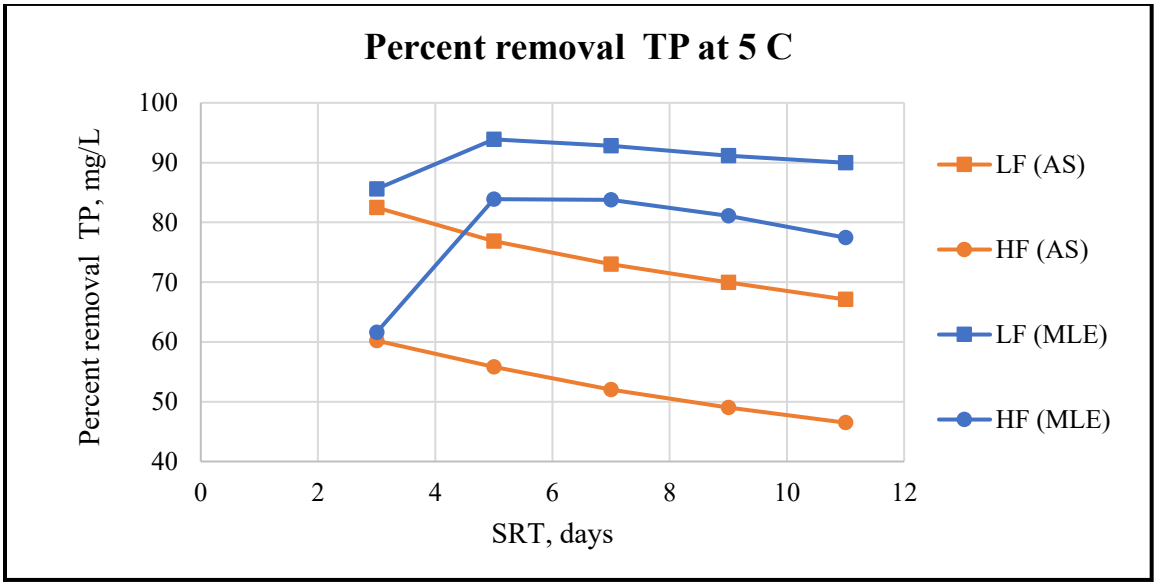
Figure 5.11 represents the removal efficiency for TP at 5 ℃ and 25 ℃ for conventional AS and MLE systems for high flow and low flow. Figure 5.11 (a) shows that at 25 ℃, low flow for the conventional AS system had around 21 % higher removal efficiency than the high flow at 3 days SRT. Gradually with an increase in SRT, the removal efficiency for both high flow and low flow dropped for the conventional AS system. Also, the removal efficiency for low flow in the MLE system was nearly 4 % better than low flow in the conventional AS system. The anoxic bioreactor in the MLE system hence improved the removal efficiency. The maximum efficiency was around 82 % for low flow for the MLE system at 3 days SRT. An increase in SRT at 25 ℃ led to a drop in the effluent removal efficiency for both the systems (for both high flow and low flow). High flow in the MLE system showed better removal efficiency for TP than high flow in the conventional AS system.

In Figure 5.11 (b), at 5 ℃, the biological activity of the heterotrophic bacteria essential for carbon removal and the nitrifiers for nitrification (total ammonia), was ceased. This aided in the removal of phosphorous removal. Also, aerobic bioreactor alone did not favour TP removal. Therefore, an increase in SRT in the conventional AS system degraded the removal efficiency for high flow and low flow for the conventional AS system. The maximum removal efficiency for low flow and high flow (3 days SRT) for the conventional AS system at 5 ℃, was around 81% and 60%, respectively. It dropped gradually with the increase of SRT. Whereas, for MLE in Figure 5.11 (b), the removal efficiency increased for high flow and low flow, increasing SRT from 3-5 days. The rise in the overall SRT increased the residence time in the anoxic bioreactor too. This initially aided in the reduction of effluent TP. An increase in SRT may have increased the total COD. This may have led to an increase in the effluent TP concentration, thereby decreased the removal efficiency due to the rise in SRT. The removal efficiency increased to 94 % and 85 % for the low flow and high flow, respectively, in the MLE system. After that, the efficiency gradually declined with the increase in SRT for high flow and low flow.

It was evident in Figure 5.11 that the conventional AS system under the low flow and high flow scenarios at 25 ℃ and 5 ℃ presents almost the same removal efficiency. It could be deduced that the temperature has a negligible impact on the TP removal in the conventional AS system. For the MLE system, however, the temperature may impact the dominance of GAOs over PAOs, at higher temperatures. The anoxic bioreactor in the MLE system improved the removal efficiency for high flow and low flow by 23% and 12%, respectively, compared to the conventional AS system in Figure 5.11 (b). It could also be concluded that temperature may indirectly impact the TP removal.



(a)



(b)

Figure 5.11: Comparison of removal efficiency of TP for different systems and different flow scenarios for wastewater temperature of (a) 25 ℃ and (b) 5 ℃.

5.5 Conclusion

This study investigated the combined impact of wastewater temperature and SRT on conventional AS system and MLE under various flow conditions. Due to seasonal variation and changes in the ambient air temperature and precipitation magnitude and patterns under

climate change, the influent flow and wastewater temperature may vary. In this study, a six-year influent flow data of a municipality was studied and analyzed statistically. The average flow and corresponding wastewater characteristics were considered as baseline. Based on the percent change in the influent flow relative to average flow, low flow and high flow conditions were assumed. Similarly, corresponding wastewater characteristics for low and high flow were derived considering the percent change for the wastewater characteristics for the average flow. SRT was considered as a control parameter and simultaneously wastewater temperature and influent flow were varied. Simulations were conducted for conventional AS and MLE systems. It was observed that the removal efficiency of TSS for both systems was not significantly impacted by the change in the influent flow (low / high flow) and wastewater temperature change (5-25℃). Total COD and cBOD for both systems were significantly impacted by the change in temperature and high flow and low flow conditions. For the conventional AS system at 5 ℃, the removal efficiency for total COD and cBOD for high flow was nearly 10% and 5%, respectively, higher than the low flow. At 25 ℃ for the conventional AS system, the removal efficiency for total COD and cBOD for high flow was 10% and 4% lower than low flow. However, for the MLE system for both 5 ℃ and 25 ℃ , the removal efficiency for low flow was higher than high flow. For the removal efficiency for total ammonia for both the systems, it was observed at 25 ℃ that the removal efficiency for high flow was higher than low flow. The removal efficiency for total ammonia reached a maximum nearly 96 % at 25 ℃. But at 5 ℃ , the removal efficiency for total ammonia for both systems were relatively lower than the high flow and low flow at 25 ℃. Adding to this, the removal efficiency of TP (for both the systems) at 25 ℃ for low flow was higher than the high flow. Maximum

removal of 94 % was observed at 5 days for low flow (MLE). At 5 ℃, high flow and low flow for the conventional AS system depicted almost the same trend for the removal efficiency as they showed at 25 ℃. Therefore, it could be concluded that temperature had a negligible impact on the removal of TP in the conventional AS system.

Future research could investigate the trend change in the influent flow. Consumption of chemicals and electric power could be accounted with the change in influent flow throughout the year. Changes in wastewater parameters such as pH may also impact the biological processes. These all could give a better insight for the proper optimization of WWTP during adverse influent flow conditions.

5.6 Reference

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Chapter 6: Conclusion and Recommendations

6.1 Conclusion

This study showed that climate change may disrupt the WWTPs. Change in wastewater temperatures and influent flow were identified as the key parameters that may occur because of climate events (rise in ambient temperature, snowmelt, heavy or low precipitation patterns). Change in wastewater temperatures and influent flow also may impact the treatment efficiency of various secondary treatment systems studied. It was also observed that SRT played a critical role in the BNR. Too long or too short SRT did not yield adequate nutrient removal. The modification in the secondary treatment system proved beneficial for the carbon and BNR removal. Also, due to the addition of anaerobic / anoxic bioreactors, the system could achieve better efficiency at shorter SRT.

The following are the main conclusion drawn from each chapter.

- Chapter 3 included the results for the conventional AS and Ludzack- Ettinger systems. The primary effluent was considered as the influent to the secondary system. The activity of the microbial community was sensitive to temperature, according to Monod's equation. Hence, it was observed that effluent total COD, cBOD, TSS and total ammonia were higher at colder temperatures ($T < 13$ ℃) and lower for warmer temperatures ($T > 13$ ℃). The effluent concentrations could be controlled by changing the SRT. Better results were obtained for warmer temperatures at shorter SRTs.

- Chapter 4 comprised the results for the conventional AS, MLE, Phoredox, A2O and modified Bardenpho systems. The primary treatment was incorporated into the model.
    - For the conventional AS system, warmer temperature improved the cBOD and total ammonia removal. Whereas TP removal was not much impacted by temperature in this system. 3-days SRT was critical for cBOD and TP removal. Whereas, 5 days SRT and 13 ℃ were important for total ammonia removal in this system.
    - In the MLE system, the cBOD concentrations were lower than in the conventional AS system, as nitrifiers and PAOs consumed it as a carbon source. Eventually, a further increase in SRT after 5 days deteriorated cBOD and TP removal efficiency. Also, the temperature played an important role in TP removal. At higher temperatures, nitrifiers were more active than PAOs, thereby decreasing the TP efficiency at higher temperatures.
    - For the Phoredox system, the removal of total ammonia was initiated at 17 ℃, because there was minimal / negligible recycle of NML in the anaerobic bioreactor. Also, 5-days of overall SRT provided adequate residence time in the anaerobic bioreactor.
    - For the A2O system, better efficiency was achieved at 7 days SRT because of the presence of three bioreactors (anaerobic, anoxic and aerobic). Adequate residence time in each bioreactor improved cBOD, total ammonia and TP removal.

- For the modified Bardenpho system, the best removal efficiency was obtained for cBOD, total ammonia and TP, compared to all other systems. 5-days SRT was critical for cBOD, total ammonia and TP removal efficiency. It was also observed that temperature did not play a significant role in the TP removal.

- Chapter 5 included the results of the impact of change in influent flow (high / low flow) and temperature for conventional AS and MLE systems.
  - It was observed that TSS concentrations did not vary directly by the change in wastewater temperature and influent flow. Changes in the concentrations of total COD impacted the TSS concentrations and, followed a similar trend as the total COD concentration.
  - For the conventional AS system, the removal efficiency for total COD and cBOD for high flow and at 5 ℃ was higher than at low flow. Whereas it was vice versa at 25 ℃ (removal efficiency at low flow > removal efficiency at high flow. For the MLE system, at 5 ℃ and 25 ℃, the removal efficiency for total COD and cBOD for low flow was higher than at the high flow.
  - For total ammonia removal efficiency for both the systems, it was observed that at 25 ℃, the removal efficiency at high flow was higher than at low flow. But the removal efficiency at 5 ℃ for both systems was lower than the removal efficiency at 25 ℃ (for both high flow and low flow). It was also observed that at 5 ℃ for the MLE system, at 3-days SRT, negative removal efficiency for total ammonia was observed. This indicated an increase in the effluent total ammonia concentration than the influent

concentration. It also explains the inactivity of nitrifiers at cold temperatures and short SRTs.

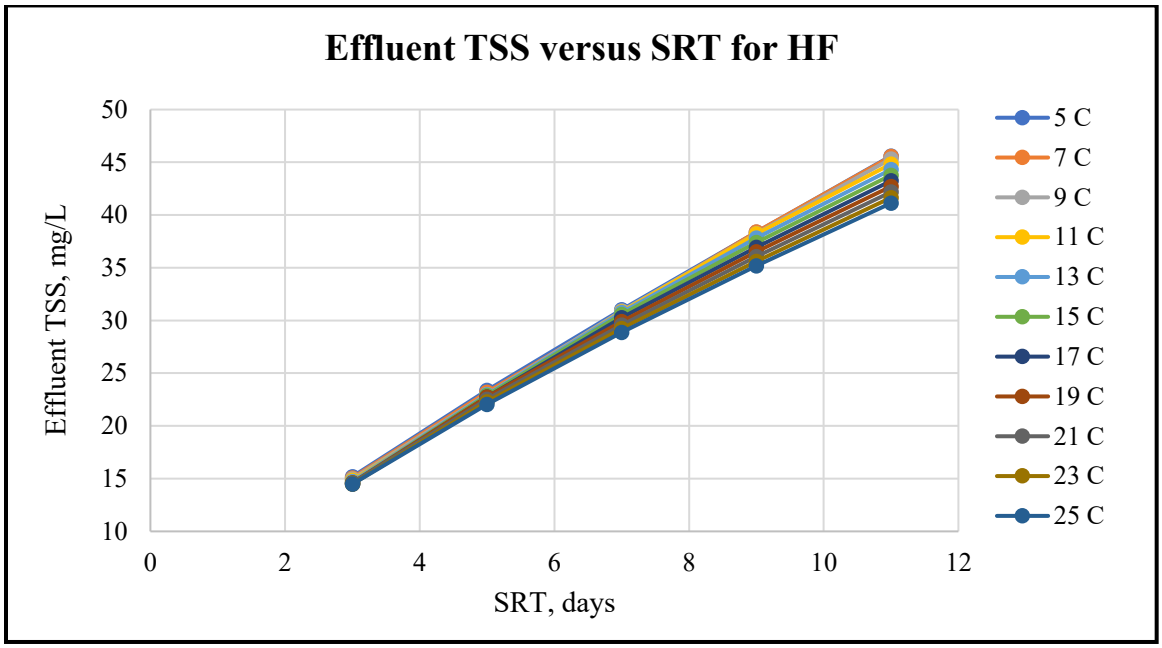
- For TP (for both the systems), at 25 ℃, the removal efficiency for TP for low flow was higher than at high flow. At 5 ℃, the removal efficiency at high flow and low flow depicted the same trend as it showed at 25 ℃ for the conventional AS system. This was indicative that the temperature had a negligible impact on TP removal in the conventional AS system.

6.2 Recommendations

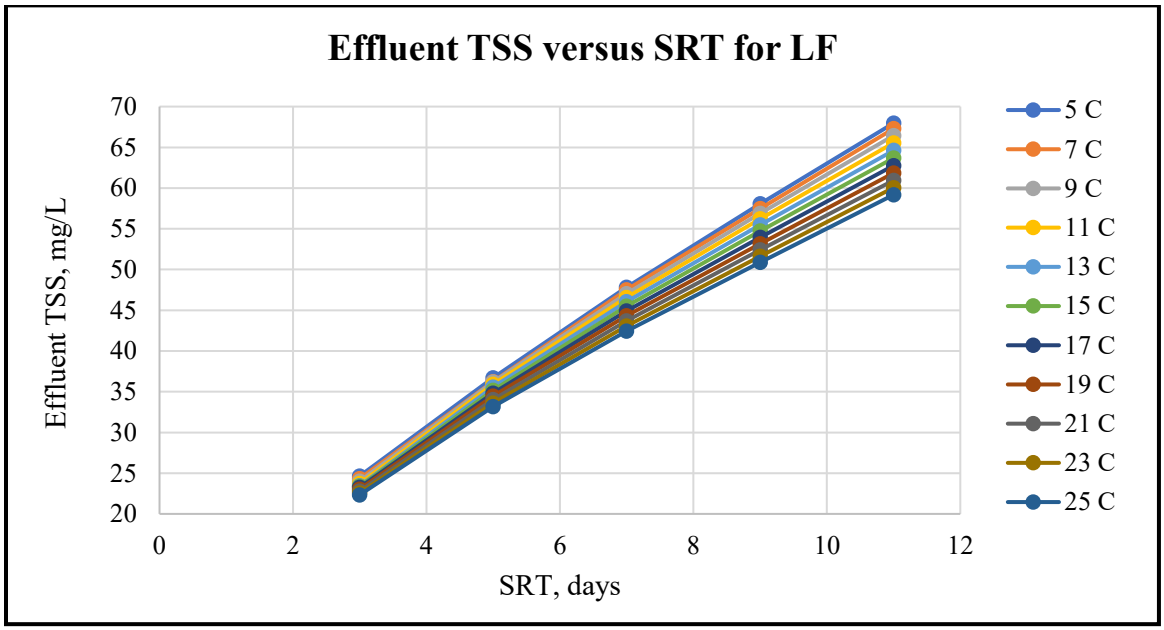
Future studies could be conducted by modulating other parameters such as pH, DO, and specific growth rate and studying their impact on the biological systems. Also, the chemical dosing and energy requirements could be measured in future studies as an account of climate change. Hence, the climate change adversely impacts the processes of the WWTPs and the operational component as well.

Appendix

TSS for conventional AS system



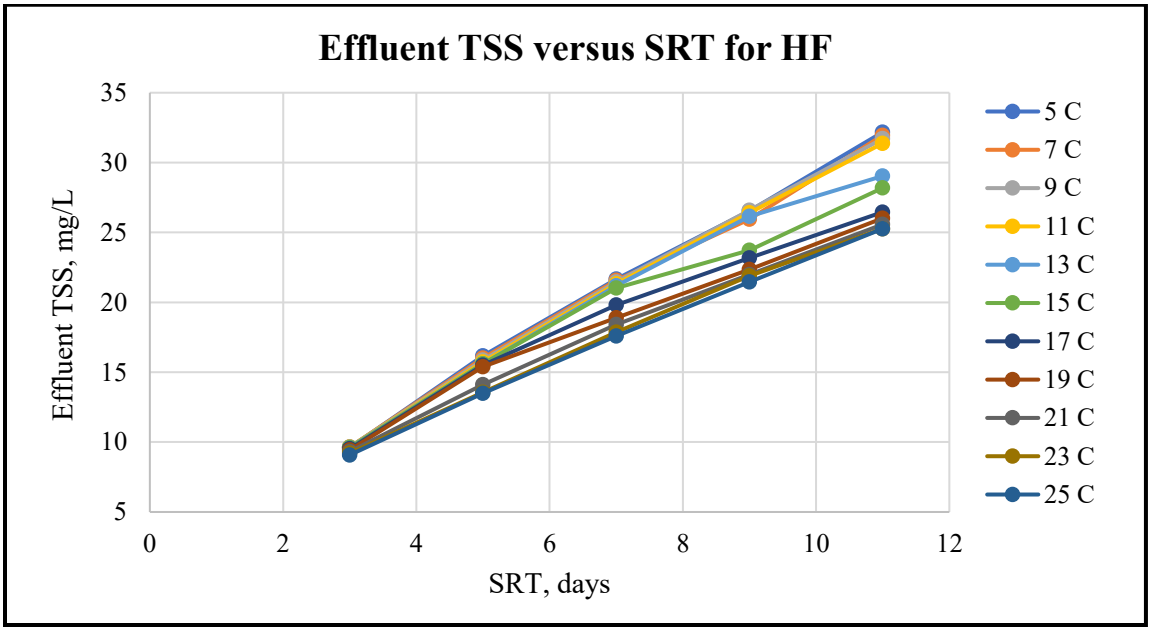
(a)



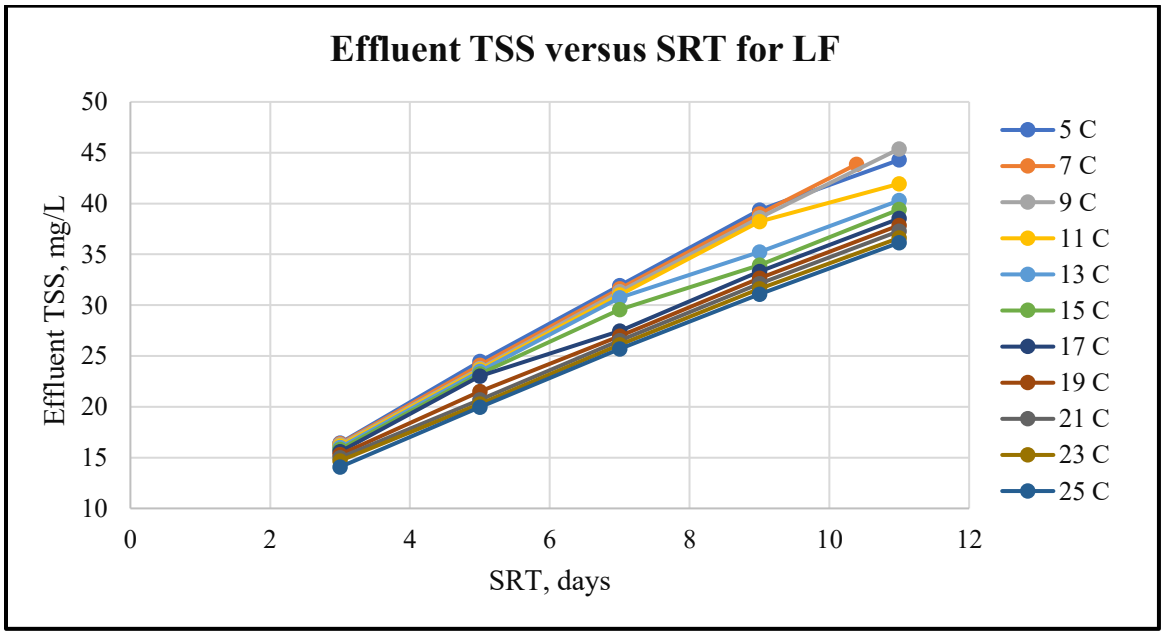
(b)

Figure A.1: Effluent TSS for conventional AS system (a) HF, (b) LF

TSS for MLE system



(a)



(b)

Figure A.2: Effluent TSS for MLE system (a) HF, (b) LF

Influent wastewater parameters considered for conventional AS and Ludzack-Ettinger systems

Table A1: Influent kinetic and stochiometric parameters for conventional AS and Ludzack-Ettinger systems

Name	Raw Defaults	Value
Fbs - Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.1600	**0.2000**
Fac - Acetate [gCOD/g of readily biodegradable COD]	0.1500	**0.2182**
Fxsp - Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.7500	**0.7490**
Fus - Unbiodegradable soluble [gCOD/g of total COD]	0.0500	**0.0909**
Fup - Unbiodegradable particulate [gCOD/g of total COD]	0.1300	**0.1267**
Fcel - Cellulose fraction of unbiodegradable particulate [gCOD/gCOD]	0.5000	0.5000
Fna - Ammonia [gNH3-N/gTKN]	0.6600	0.6600
Fnox - Particulate organic nitrogen [gN/g Organic N]	0.5000	0.5000
Fnus - Soluble unbiodegradable TKN [gN/gTKN]	0.0200	0.0200
FupN - N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.0700	0.0700
Fpo4 - Phosphate [gPO4-P/gTP]	0.5000	0.5000
FupP - P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.0220	0.0220
Fsr - Reduced sulfur [H2S] [gS/gS]	0.1500	0.1500
FZbh - Ordinary heterotrophic COD fraction [gCOD/g of total COD]	0.0200	0.0200
FZbm - Methylotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZaao - Anaerobic ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZppa - Phosphorus accumulating COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZpa - Propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZam - Acetoclastic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZhm - Hydrogenotrophic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZso - Sulfur oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrpa - Sulfur reducing propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsra - Sulfur reducing acetotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrh - Sulfur reducing hydrogenotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZe - Endogenous products COD fraction [gCOD/g of total COD]	0.0000	0.0000

Influent wastewater parameters considered for conventional AS and MLE systems for high flow condition

Table A2: Influent kinetic and stochiometric parameters for the high flow condition for conventional AS and MLE systems

Name	Raw Defaults	Value
Fbs - Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.1600	**0.1320**
Fac - Acetate [gCOD/g of readily biodegradable COD]	0.1500	**0.2546**
Fxsp - Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.7500	**0.6812**
Fus - Unbiodegradable soluble [gCOD/g of total COD]	0.0500	**0.0700**
Fup - Unbiodegradable particulate [gCOD/g of total COD]	0.1300	**0.1400**
Fcel - Cellulose fraction of unbiodegradable particulate [gCOD/gCOD]	0.5000	**0.6000**
Fna - Ammonia [gNH3-N/gTKN]	0.6600	**0.6937**
Fnox - Particulate organic nitrogen [gN/g Organic N]	0.5000	0.5000
Fnus - Soluble unbiodegradable TKN [gN/gTKN]	0.0200	0.0200
FupN - N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.0700	0.0700
Fpo4 - Phosphate [gPO4-P/gTP]	0.5000	**0.4093**
FupP - P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.0220	0.0220
Fsr - Reduced sulfur [H2S] [gS/gS]	0.1500	0.1500
FZbh - Ordinary heterotrophic COD fraction [gCOD/g of total COD]	0.0200	0.0200
FZbm - Methylotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZaao - Anaerobic ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZppa - Phosphorus accumulating COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZpa - Propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZam - Acetoclastic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZhm - Hydrogenotrophic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZso - Sulfur oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrpa - Sulfur reducing propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsra - Sulfur reducing acetotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrh - Sulfur reducing hydrogenotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZe - Endogenous products COD fraction [gCOD/g of total COD]	0.0000	0.0000

Influent wastewater parameters considered for conventional AS and MLE systems for low flow condition

Table A3: Influent kinetic and stochiometric parameters for the low flow condition for conventional AS and MLE systems

Name	Raw Defaults	Value
Fbs - Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.1600	**0.1813**
Fac - Acetate [gCOD/g of readily biodegradable COD]	0.1500	**0.0549**
Fxsp - Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.7500	**0.6861**
Fus - Unbiodegradable soluble [gCOD/g of total COD]	0.0500	**0.0207**
Fup - Unbiodegradable particulate [gCOD/g of total COD]	0.1300	0.1300
Fcel - Cellulose fraction of unbiodegradable particulate [gCOD/gCOD]	0.5000	0.5000
Fna - Ammonia [gNH3-N/gTKN]	0.6600	**0.6980**
Fnox - Particulate organic nitrogen [gN/g Organic N]	0.5000	0.5000
Fnus - Soluble unbiodegradable TKN [gN/gTKN]	0.0200	0.0200
FupN - N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.0700	0.0700
Fpo4 - Phosphate [gPO4-P/gTP]	0.5000	**0.1773**
FupP - P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.0220	0.0220
Fsr - Reduced sulfur [H2S] [gS/gS]	0.1500	0.1500
FZbh - Ordinary heterotrophic COD fraction [gCOD/g of total COD]	0.0200	0.0200
FZbm - Methylotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZao - Ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZno - Nitrite oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZaao - Anaerobic ammonia oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZppa - Phosphorus accumulating COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZpa - Propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZam - Acetoclastic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZhm - Hydrogenotrophic methanogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZso - Sulfur oxidizing COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrpa - Sulfur reducing propionic acetogenic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsra - Sulfur reducing acetotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZsrh - Sulfur reducing hydrogenotrophic COD fraction [gCOD/g of total COD]	1.00E-04	1.000E-04
FZe - Endogenous products COD fraction [gCOD/g of total COD]	0.0000	0.0000