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Effects of wastewater heat recovery on nitrogen removal in Finnish wastewater treatment plants

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Abstract

Decentralized wastewater heat recovery is recognized to be a possible environmentally sustainable energy source, even though it is not currently widely applied in housing or in wastewater networks. The use of wastewater heat as an energy source decreases wastewater temperature, if the heat is recovered prior to the wastewater treatment plants (WWTPs). In WWTPs, lower wastewater temperature causes deterioration of the nitrogen removal process in the activated sludge process (ASP). No work has yet been done to connect wastewater heat recovery and the functioning of ASP in Finnish conditions. This thesis determines and assesses the effects of decreasing wastewater temperature on the nitrogen removal and ASP in Finnish WWTPs as well as the feasibility of applying wastewater heat recovery in these conditions. The impacts are assessed by performing data analysis and process simulations. The process simulations were able to show a clear trend between temperature and nitrogen removal. The influence of the decrease of 1 °C in the wastewater temperature intensifies as the process temperature drops to cold climate range. This effect can be seen especially under 10 °C. Process simulations also illustrated, in 15 °C, the ASP process volume would need to grow by 25% for no additional nitrogen leaving compared to 17 °C. The data analysis supported the process simulation results, but also provided an additional point of view into the intricacy of observing the impacts of decreasing temperature in the real world. These results help to assess the potential of wastewater heat recovery in Finnish conditions and illustrate the needed considerations to implement wastewater heat recovery in an environmentally sustainable manner. Based on this thesis, further research concerning the feasibility of using innovative energy storages as well as management efforts against decreasing temperature are recommended. This thesis is a part of project coordinated by HSY called “The energy balance of wastewater heat recovery in a city and its effects in the wastewater treatment”. This project has received funding from the Ministry of the Environment.

Keywords Wastewater heat recovery, nitrogen removal, nitrification, activated sludge process

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Työn nimi Jäteveden lämmön talteenoton vaikutus typenpoistoon ja aktiivilieteprosessiin suomalaisilla jätevedenpuhdistamoilla

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Tiivistelmä

Jäteveden lämmön talteenotto on potentiaalisesti ympäristöystävällinen energian lähde, jota ei vielä ole otettu laajasti käyttöön kiinteistöillä taikka jätevesiverkostoissa. Jäteveden lämmön talteen ottaminen ennen jätevedenpuhdistamoa laskee jätevedenpuhdistamoille saapuvan jäteveden lämpötilaa. Jäteveden lämpötilan lasku hankaloittaa typenpoistoa sekä aiheuttaa monia muita vaikutuksia aktiivilieteprosessissa. Nykyisin on saatavilla kattavasti tietoa lämpötilan vaikutuksista typenpoistoon, mutta juuri jäteveden lämmön talteenoton vaikutuksia typenpoistoon ja aktiivilieteprosessin toimintaan ei ole tutkittu suomalaisissa olosuhteissa.

Tämä diplomityö analysoi ja arvioi laskevan lämpötilan vaikutuksia typenpoistoon ja aktiivilieteprosessiin suomalaisilla jätevedenpuhdistamoilla sekä jäteveden lämmön talteenoton soveltuvuutta kyseisiin olosuhteisiin. Laskevan lämpötilan vaikutuksia analysoitiin ja arvioitiin data-analyysin sekä prosessisimulointien avulla. Prosessisimuloinneilla saatiin selville, että jokaisen alenevan lämpötila-asteen vaikutus typenpoistoon korostuu eritoten, kun jäteveden lämpötila laskee alle 10 °C. Simulointien perusteella selvisi myös, että typenpoiston heikkenemisen kompensointi aktiivilieteprosessin tilavuuden suurentamisella on haastavaa, sillä esimerkiksi jo muutaman asteen jäteveden lämpötilan lasku 17 °C:sta vaatii aktiivilieteprosessin tilavuuden kasvattamista neljänneksellä. Data-analyysin tulokset tukivat prosessisimulaatioiden tuloksia.

Diplomityön tulokset auttavat jäteveden lämmön talteenoton toteuttamiskelpoisuuden arvioinnissa suomalaisessa kontekstissa sekä valottavat toimia, joiden avulla jäteveden talteenottoa voitaisiin harjoittaa ekologisesti kestävällä tavalla. Diplomityön on osa HSY:n koordinoimaa hanketta ”Lämmöntalteenoton energiatase kaupungissa ja vaikutus jätevesien käsittelyyn”, joka on saanut ympäristöministeriön myöntämää valtionavustusta Ravinteiden kierrätyksen ja jätevesien käsittelyn energiatehokkuuden hankkeiden avustushaussa vuonna 2020.

Avainsanat Jätevesilämpö, typenpoisto, nitrifikaatio, aktiivilieteprosessi

Preface

This thesis is part of a larger project by HSY called “The energy balance of wastewater heat recovery in a city and its effects in the wastewater treatment”. The project consortium consists of HSY, Turun seudun puhdistamo Oy, Turun Vesihuolto Oy, Turun Seudun Vesi Oy, Helen Oy, Fortum Power and Heat Oy, and Turku Energia. The project is managed by HSY, and it has received funding from the Ministry of the Environment. I am thrilled and thankful that I got to take part in this project.

First and foremost, I want to thank my advisors Kristian Sahlstedt and Maija Vilpanen for all the time, energy, and valuable comments I received from them. It felt like a privilege to have two such great advisors. I also want to thank Anna Mikola for being my supervisor and for being supportive throughout the writing process.

I want to thank Anna Kuokkanen from HSY and Jarkko Laanti from Turun seudun puhdistamo Oy for providing me important material. Anna was always kind to share her vast knowledge which helped me significantly.

I am also grateful for my friends and family who have believed in me through my whole life. I also want to thank *my* Kristian, for continuously supporting me throughout this writing process. It is due to all of you, that I can treasure these exceptional times and remember them fondly in the future.

Kumpula, 22.11.2021

Maija Ahonen

Contents

Symbols and abbreviations

Symbols

θ = Temperature activity coefficient

η_{NO3} = Anoxic hydrolysis reduction factor

μ_{AmT} = Maximum specific growth rate constant

μ_{AUT} = Autotrophic maximum growth rate of autotrophic nitrifying biomass

μ_H = Heterotrophic maximum growth rate on substrate

K_A = Saturation coefficient for nitrate

K_{ALK} = Saturation coefficient for alkalinity

K_{NH4} = Saturation constant for ammonium

K_{NO3} = Saturation constant for nitrate and nitrite

K_{nT}= Half saturation constant for nitrifiers at certain temperature

K_{O2}= Saturation constant for dissolved oxygen

K_P = Saturation coefficient for phosphorus

k_T= Reaction-rate coefficient in certain temperature

N_A = Bulk liquid ammonia concentration

N_n = Bulk liquid nitrate concentration

S_A = Acetate as fermentation product

S_{ALK} = Alkalinity (bicarbonate)

S_F = Readily biodegradable substrate

S_{NH4} = Ammonium

S_{NO3} = Nitrate and nitrite

S_{O2} = Dissolved oxygen

S_{PO4} = Phosphate

T = Temperature

X_{AUT} = Autotrophic nitrifying biomass

X_{BA} = AOB concentration

X_H = Heterotrophic biomass

Y_A = Nitrifier yield coefficient

Abbreviations

AOB = Ammonia oxidizing bacteria

ASM = Activated sludge model

ASP = Activated sludge process

COD = Chemical oxygen demand

DO = Dissolved oxygen

GHG = Greenhouse gas

IFAS = Integrated fixed film activated sludge process

IWA = International water association

MBBR = Moving bed biofilm reactor

MLSS = mixed liquor suspended solids

NOB = Nitrate oxidizing bacteria

SRT = Sludge retention time

SS = Suspended solids

TKN = Total Kjeldahl nitrogen

WWTP = Wastewater treatment plant

1 Introduction

1.1 Background and motivation

Over the last decades, wastewater heat recovery has received attention due to its potential as a new, environmentally sustainable energy resource. Wastewater contains energy in the form of heat, which is not currently widely taken advantage of in a decentralized way. Decentralized wastewater heat recovery means that the heat of the wastewater is collected right where the wastewater is produced or in the network prior to the wastewater treatment plants (WWTPs). It is recognized that decentralized wastewater heat recovery has various effects on WWTPs, which pose a possible threat to the usability of this energy source. When performing decentralized wastewater heat recovery, the lower temperature of influent wastewater in WWTPs can cause complications, especially in the biological processes which include some of the core functions in wastewater treatment. (Kretschmer, et al., 2016; Kordana, 2017)

The importance of wastewater treatment is easy to see through removal of nutrients and organic matter. Nitrogen removal is vital for keeping the recipient aquatic ecosystems in a good condition, as additional nitrogen load leads to eutrophication of said waterbodies. Nitrogen removal is especially important in Finland, as many surface waters, including Baltic Sea, are highly sensitive ecosystems.

Although the connection between wastewater temperature and removal in WWTPs has been widely acknowledged in the previous research on wastewater heat recovery (Wanner, et al., 2005; Delatolla, et al., 2012; Johnston, et al., 2019), no research has connected it to the performance of WWTPs in Finnish conditions. The kinetics of the biochemical reactions of nitrogen removal have also been well studied (Henze, et al., 2008; Tchobanoglous, et al., 2003; Wanner, et al., 2005). Currently, widely used constants for determining the temperature dependency of biological activated sludge processes (ASPs) are often researched using warm water (>12 °C) (Tchobanoglous, et al., 2003; Coskuner & Jassim, 2008), which does not often correspond to the state of Finnish WWTPs. As this thesis focuses on Finnish conditions, insight into the temperatures below 12 °C is received. Although this thesis utilizes data acquired from three different WWTPs in southern Finland, the results can also be applicable to WWTPs in the northern parts of Finland, as well as any other area with similar topology, climate, urban infrastructure, and consumption culture.

Therefore, the aim of this thesis is to determine the magnitude and assess the effects of low temperature on the biological processes in Finnish WWTPs as well as the feasibility of applying wastewater heat recovery in an environmentally sustainable manner in these conditions. This thesis forms part of a larger project called "The energy balance of wastewater heat recovery in a city and its effects in the wastewater treatment". The project consortium comprises of Helsingin seudun ympäristöpalvelut, Turun seudun puhdistamo Oy, Turun Vesihuolto Oy, Turun Seudun Vesi Oy, Helen Oy, Fortum Power and Heat Oy, and Turku Energia. The project is managed by HSY, and it has received funding from the Ministry of the Environment. The project aims to connect various viewpoints to create a holistic approach for applying wastewater heat recovery in Finland. The other studies in this larger project include mapping the options for wastewater heat recovery in Finnish cities, the effects of wastewater heat recovery on the wastewater network, determining options for wastewater heat storage, the legal framework concerning wastewater heat recovery, and finally its effects on nitrogen removal in WWTPs.

1.2 Objectives

The main objective of the thesis is to assess the effect of lower ASP temperature on the nitrogen load (ammonium load and total nitrogen load) discharged into the recipient waterbody, as well as to determine other effects of lower process temperature. These other effects include changes in finances and other used assets (chemicals, energy), changes in greenhouse gas (GHG) emissions of the WWTP and overall functionality of the ASP. The main research questions can be formulated as follows:

1. How will the nitrogen load of the recipient waterbody change in response to changes in the wastewater temperature when other attributes are constant?
2. What is the process volume needed to avoid an increase in the nitrogen load to the recipient water body as the temperature of the ASP decreases?

To accomplish these objectives, data from three Finnish WWTPs is analysed and ASP simulations are performed on a dynamic activated sludge process model. The combination of data analysis and process simulations provides an opportunity to obtain a more in-depth analysis of the effects of temperature on the nitrogen removal processes in WWTPs. These two methods support each other and verify the results of this thesis.

1.3 Structure

From hereafter this thesis is divided into six chapters. Chapter 2 provides background information on the removal of nitrogen in WWTPs. The chapter also explains the basics of wastewater heat recovery. Chapter 3 describes the nitrogen removal process and the effects of temperature on ASP in greater detail, as well as the mechanics of the way temperature affects nitrification and denitrification. Chapter 3 also reviews the literature covering possible other effects of lower temperature in WWTPs. Chapter 4 provides a description of the materials and methods used in data analysis and process simulations. Chapter 5 presents and analyzes the results from data analysis and simulations. This chapter presents the observed effect of temperature on the ammonium and total nitrogen removal as well as the effect of temperature when sludge age, mixed liquor suspended solids (MLSS) and process volume remain constant. Chapter 6 further analyzes and discusses the results obtained. This chapter also provides discussion on the influence the previous results have on the GHG emissions and financial investments of the WWTPs. This chapter also includes analysis on the possible differences of the data analysis and process simulation results, sensitivity analysis, as well as estimates of the uncertainties related to this thesis. Chapter 8 concludes the thesis by summarizing the contribution of this work and highlighting the significance of the results for implementing wastewater heat recovery in Finnish wastewater treatment plants.

2 Wastewater treatment and heat recovery

2.1 Water treatment process and nitrogen removal

Proper wastewater treatment is a step towards achieving good ecological and chemical status of water bodies for EUs member nations. This common goal has been set as a directive by European parliament and of the council (Establishing a framework for Community action in the field of water policy 2000/60/EC). Additional nitrogen entering the recipient waterbodies increases their biological oxygen demand which might lead to eutrophication. Eutrophication can further cause anoxia. (Henze, et al., 2008) Nitrate can cause eutrophication in surface waters, while nitrite is toxic for eukaryotes and it can prohibit bacterial growth already in WWTPs. (Daims, et al., 2016) Nitrogen entering aquatic ecosystem as ammonia can have direct toxic effects on aquatic life (Henze, et al., 2008). When assessing the level of needed nitrogen removal, qualities of the recipient waterbody, e.g. the ability to restrain nutrients and sensitivity to nitrogen, are observed (Laitinen, et al., 2014).

The generic modern WWTP consists of various main unit processes including pumping station, screening, sand and grit removal, primary settling, and ASP (aeration tank and secondary settling) (Laitinen, et al., 2014). These units can be seen in *Figure 1*. The wastewater is screened before entering the activated sludge process to avoid the suffocation of pumps by debris as well as the deterioration of the treatment processes, (Tchobanoglous, et al., 2003).

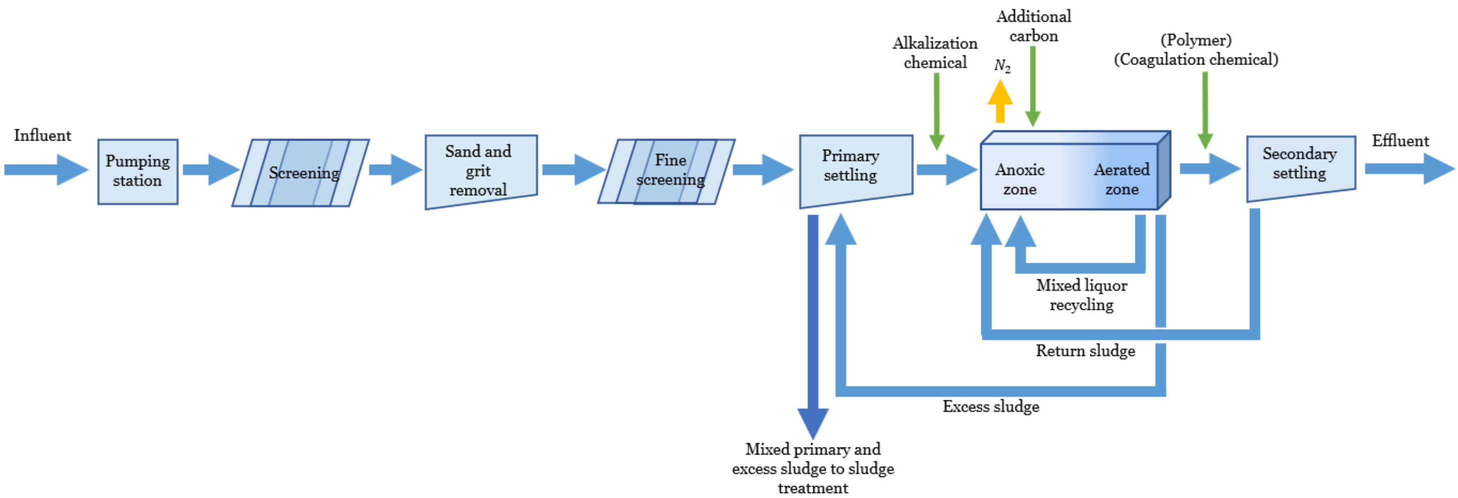

Figure 1. An illustration of common flows and components of a wastewater treatment plant in Finland. The route of the wastewater is illustrated with blue arrows, the inflows are illustrated with green arrows and the outflow with dark blue one. N2 flow is indicated with yellow arrow.

Nitrogen is removed from wastewater in WWTPs with biological processes. Commonly in WWTPs, nitrogen removal is integrated in activated sludge process. (Tchobanoglous, et al., 2003) In activated sludge processes, the process conditions are engineered to facilitate the natural nitrogen cycle in the favor of water treatment. In

WWTPs naturally occurring bacteria are harnessed to perform on an elevated level of effectiveness, as the system lacks natural limitations. Process attributes such as oxygen and pH can be controlled to a favorable level for the microorganisms. (Henze, et al., 2008)

Nitrogen removal consists of aerated zone, where nitrification happens, and anoxic zone where the denitrification takes place. Typically, return sludge is recycled from settling tank to the first anoxic reactor. Aerated wastewater with nitrate is also recycled from the aerated zone back to the anoxic zone, as can be seen in *Figure 2*. (Henze, et al., 2008) Generally, the aerated and anoxic zones are located in the same tank as separate areas. Often so called "switch zones" are also included in the anoxic zone so that part of it can be transformed into aerated zone when needed. With switch zones the nitrification capacity can be maintained e.g. in low temperatures.

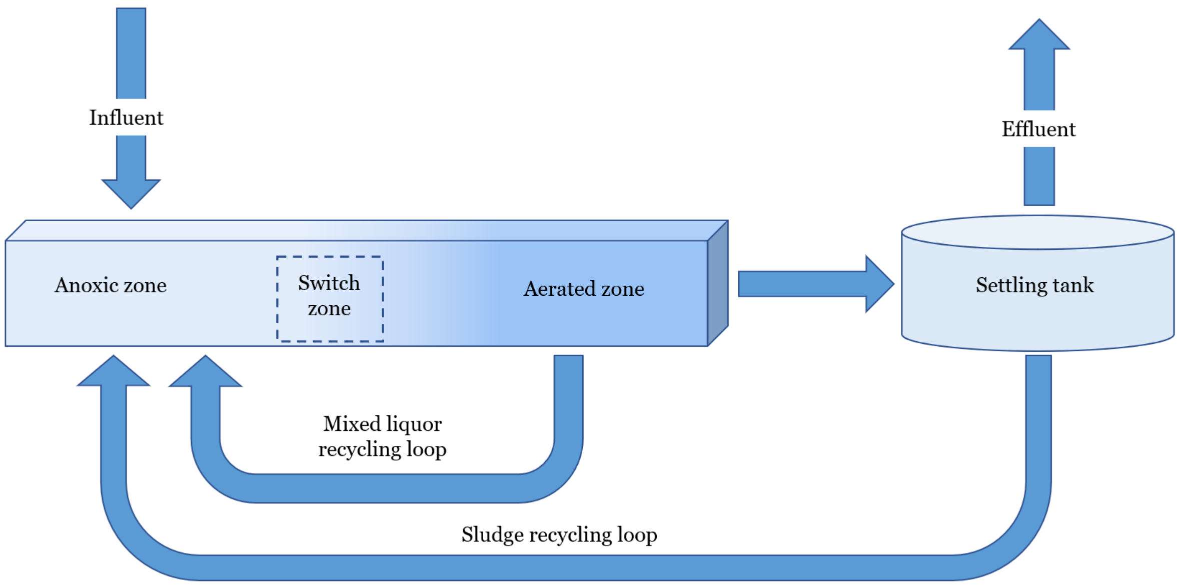

Figure 2. Activated sludge process. The flow of the wastewater is illustrated with blue arrows.

Nitrogen removal in WWTPs is both so called substrate driven nitrogen removal and endogenous driven nitrogen removal. In substrate driven, the electron donors for the reduction reactions originate from the influent wastewater, but in endogenous nitrogen removal they are formed from the decay of the sludge itself. On top of the main nitrogen removal processes in ASP, nitrogen is also utilized in assimilation, meaning that the nitrogen is utilized in growth of the microbes. (Tchobanoglous, et al., 2003)

2.2 Wastewater heat recovery

In Finland, the average amount of water used per day per person is 113 liters and 35% of this water is heated water. (Korhonen, et al., 2020) This means that the energy used to purposefully heat the used water before domestic use as well as the heat energy stored in the raw water escapes as the wastewater is flushed down the drain. A great share of the heat in the wastewater originates from the raw water, which partly leads to temperature of the wastewater depending of the time of the year. (Tchobanoglous, et al., 2003) The amount of heat stored in the raw water depends on its source, e.g. ground water has often more stable temperature, while surface water temperature can vary greatly (Andriamirando, et al., 2007). The temperature changes throughout the day depend on the warm water usage trends.

It has been recognized that wastewater heat recovery can play and important role in more environmentally sustainable energy production, as there are vast quantities of it in sufficient temperature (Gabor, et al., 2018). It has been estimated that decentralized wastewater recovery could produce 3,5 kWh of energy per person per day in domestic circumstances (Gabor, et al., 2018). Decentralized wastewater heat recovery can have vast effects in the biological wastewater treatment processes in WWTPs. *Figure 3* showcases the possible locations for wastewater heat recovery. Locations 1 and 2 have effects on the functioning of the WTWP, but location 3 does not. In this thesis, attention is focused on the first two locations. Locations 1 and 2 are considered to be decentralized, and location 3 centralized.

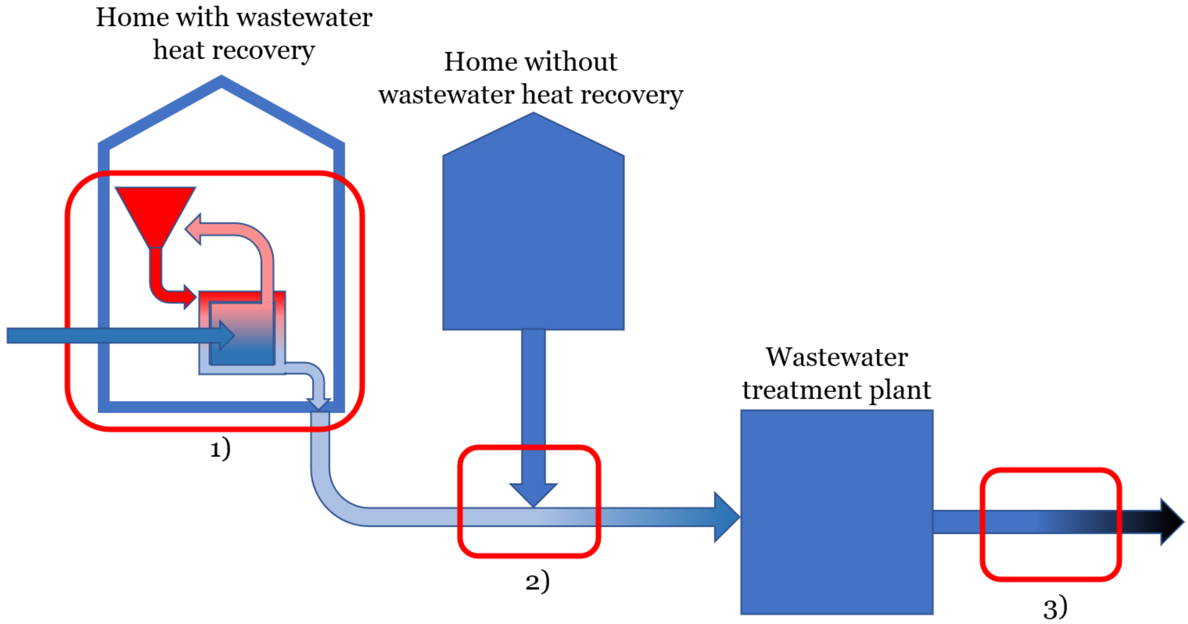

Figure 3. Possible heat recovery system locations. Location 1) is right at the wastewater producing building, location 2) is in the wastewater network and location 3) is located after the WWTP. The same principle for heat recovery illustrated in location 1) can be applied in other locations also. (Hepbasli, et al., 2014)

It has been estimated that 40% of the produced heat escapes cities as wastewater. (Hepbasli, et al., 2014) Wastewater heat recovery was recognized to possibly cover 7% - 15% of the heat demand in cold conditions in United Kingdom. (Abdel-Aal, et al., 2018). Some studies have shown that on top of reducing GHG emissions from heating, wastewater heat recovery also has the possibility to be financially feasible. (Kwiatek, et al., 2019) The feasibility of decentralized wastewater heat recovery systems seems to be less researched subject compared to wastewater heat recovery in and after WWTPs.

Heat recovery from wastewater is performed with heat exchanger and a heat pump. A heat transferring liquid is circulating in both of these systems, which transfers the heat from the wastewater to the heat pump. The heat pump can the further increase the temperature of the liquid so that it can be better utilized. (Kretschmer, et al., 2016) During warm periods, the wastewater can then also be utilized in cooling. (Hepbasli, et al., 2014)

The location of the decentralized wastewater heat recovery is important, as the possibility to recover heat decreases as the distance between the wastewater recovery location and the heat utilizing entity grows. (Hepbasli, et al., 2014) Centralized wastewater heat recovery has already been utilized for some years in Kakolanmäki WWTP as well as in Helsinki from the Viikinmäki WWTP treated water (VALOR Partners Oy, 2016). The spatial aspect of wastewater heat recovery reduces its applicability (Abdel-Aal, et al., 2018). A strength of wastewater heat recovery from the wastewater networks and right in the housing units is the short distance between the heat source and the potential customers. Factors influencing the wastewater temperature arriving to a WWTP are the length traveled in the sewer system as well as the surrounding soil temperature and air temperature in the sewer system. (Kretschmer, et al., 2016)

By careful planning of the placement of the wastewater heat recovery site, the effects at the wastewater treatment plant can be minimized. By aiming to maintain the temperature as much as possible over the set lower limit for the inflowing wastewater temperature, many negative side effects in WWTPs can be avoided. The higher the wastewater temperature is in the possible heat recovery site compared to the temperature at WWTPs, the better chance there is that the heat recovery does not greatly influence the temperature at WWTPs. (Kretschmer, et al., 2016) Kretcshmer et alia (2016) also state that the details of how temperature changes and acts in the sewer needs more research.

On top of these spatial aspects, temporal ones are also important when optimizing the wastewater heat recovery systems. These temporal aspects refer to e.g. the lack of heat recovery occurring during nights and the need for heated water in the morning. During these times more conventional heating methods are still needed to support the wastewater heat. (Spriet, et al., 2020) A lag is observed after high wastewater heat demands, which is also a reason to incorporate alternative heating systems next to the wastewater heat recovery system. Spriet and McNabola (2019) estimated that only slightly above 8% of the heat demand is met with wastewater heat recovery if no system such as hot water tank is incorporated. Already adding a 300-liter hot water tank increases the share of met heat demand up to over 40%. (Spriet & McNabola, 2019)

Larger capacity and new innovations in heat storages would be beneficial in mitigating the temporal mismatches.

Other temporal complications occur during wintertime. Since there is likely more demand for heat in cold seasons, more heat would be needed to be recovered from the wastewater. This could potentially create issues as during winters the wastewater temperature can be very low even without the heat recovery and the biological processes in WWTPs are already greatly affected by the decrease in the wastewater temperature.

3 Nitrogen removal

3.1 Nitrification

The natural nitrogen cycle is utilized in the nitrogen removal process. The whole nitrogen cycle can be seen in *Figure 4*. Nitrogen is removed from wastewater by nitrification combined with denitrification. Nitrification is a microbiological process which happens as a part of certain bacteria's cellular respiration. Nitrifying organisms are aerobic autotrophic micro-organisms. They receive the needed carbon from dissolved CO2 and their energy from oxidizing ammonia and nitrite, so they do not utilize organic matter as food. This makes them grow slower than organisms that do. Nitrification requires ammonium, ammonium oxidizing bacteria (AOB), nitrogen oxidizing micro-organisms (NOB), oxygen, inorganic carbon, and water to occur. Nitrification is also depended on high oxygen levels, right temperature, retention time, and pH. (Henze, et al., 2008)

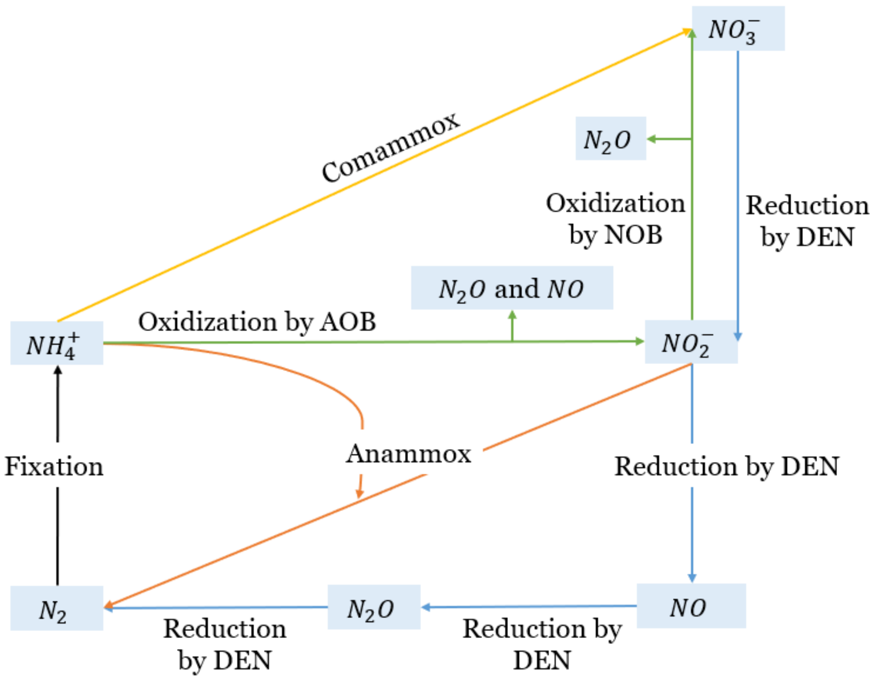

Figure 4. Full nitrogen cycle. Illustration based on (Tchobanoglous, et al., 2003)*,* (Kartal, et al., 2010)*, and* (Daims, et al., 2016)*. DEN means denitrification, AOB ammonia oxidizing bacteria AOB and NOB nitrate oxidizing bacteria.*

Activated sludge of sufficient sludge age contains these micro-organisms which create a favorable, aerated environment for nitrification to happen. Nitrification consists of two reactions, which are ammonium oxidizing into nitrite and then this nitrite oxidizing into nitrate. These nitritation (equation 1) and nitratation (equation 2) reactions are presented below. (Henze, et al., 2008)

$$2\ \mathrm{NH_4^+} + 3\ O_2 \rightarrow 2\ NO_2^- + 2\ H_2O + 4\ H^+ \qquad (1)$$

$$2\ NO_2^- + O_2 \rightarrow 2\ NO_3^- \quad (2)$$

Nitrogen removal requires facilitating these reactions in the activated sludge process.

Nitrification is conducted by a variety of aerobic autotrophic bacteria (Tchobanoglous, et al., 2003). 17 different phyla of bacteria got identified in a study of 5 Finnish WWTPs, with over 200 species. General nitrifying bacteria in Finnish WWTPs are *Nitrosomanas, Nitrosovibrio* and *Nitrosospira* (AOB) and *Candidatus* 'Nitrotoga arctica' *and Nitrospira* (NOB). (Kruglova, et al., 2020) The autotrophic micro-organisms that oxidize ammonia or nitrate are not phylogenetically closely related, even though they often exist in a close approximation of each other. Microbe-microbe interactions such as tight crossfeeding and co-aggregation are often observed with AOB and NOB. (Daims, et al., 2015)

The pH optimum for bacteria in WWTP ranges between 8 and 9 for different genus of bacteria. pH might decrease into unfavorable level, if the alkalinity in the sludge decreases for example due to improper total Kjeldahl nitrogen (TKN) and chemical oxygen demand (COD) share. (Henze, et al., 2008)

The cold condition (10 °C - 17 °C) preferring bacteria, *Candidatus* "Nitrotoga", has been only recently discovered in WWTPs. The uncovering of this bacteria illustrates the co-existence of various bacteria genera in WWTPs. The diverse set of bacteria can dynamically react to changing environmental factors like temperature and aeration conditions. (Daims, et al., 2016) The most diverse group of NOB is *Nitrospira* genus, that has been observed all around on earth from terrestrial, aquatic, high temperature as well as manmade environments. *Nitrospira* is effective nitrate oxidation bacteria and can utilize a range of organic compounds on top of nitrite in its metabolism, which could explain its great distribution. (Daims, et al., 2015) Nitrospiras I and II are the commonly met microbes in WWTPs. (Daims, et al., 2016)

For a long time, only ammonia-oxidizing bacteria and archaea were though to oxidize ammonia and nitrite-oxidizing bacteria oxidize nitrite. Only during 2015 so called comammox bacteria were found. (Daims, et al., 2015; van Kessel, et al., 2015) After the comammox bacteria genes were discovered, they were recognized vastly around the globe, and the discovered comammox turned out to be the most common ammonia oxidizer in WWTPs. (Daims, et al., 2016) Comammox has not yet been properly mapped in Finnish WWTPs. On top of the conventional nitrogen removal reactions, anammox reaction can also happen in ASP. Anammox process happens in aerated environment, where bacteria concomitantly oxidize ammonia and reduce nitrate. (Tchobanoglous, et al., 2003)

3.2 Denitrification

After ammonium has been transformed into nitrate in the nitrification process, denitrification is utilized in order to remove the nitrate as nitrogen gas. This stops nitrogen from entering the aquatic ecosystems and causing eutrophication. (Henze, et al., 2008) Complete denitrification can result in less than 5% of TKN remaining in the effluent. Denitrification is not only a core part of nitrogen removal, but it also has other positive effects on the wastewater treatment process, such as increasing alkalinity and reducing oxygen demand. (Laitinen, et al., 2014) Denitrification occurs in the anoxic areas in the WWTPs where the heterotrophic bacteria are made to utilize nitrate instead of oxygen. (Henze, et al., 2008)

Denitrification is a heterotrophic process, which means that the energy for denitrification originates from organic compounds, which can be measured as COD. COD can be of internal or external sources, meaning that either the COD is internally originated in the wastewater or that it is added from external sources to the process. If there is not enough COD in the wastewater, complete denitrification will not occur. To achieve high level of total nitrogen reduction, additional carbon sources might be needed. (Henze, et al., 2008)

Denitrification consists of reduction from nitrate to nitrite, then to nitric oxide, then to nitrous oxide, and finally to nitrogen gas. (Tchobanoglous, et al., 2003) According to Jones et al. (2008), only some denitrifiers are able to perform the full chain on reduction reactions mentioned above. Therefore, it is assumed, that successfully transitioning nitrate into nitrogen gas is a common effort of a variety of heterotrophic bacteria. The stoichiometry of denitrification is presented in equation 3. (Tchobanoglous, et al., 2003).

$$5\ CH_3OH + 6\ NO_3^- \rightarrow\ 3\ N_2 + 5\ CO_2 + 7\ H_2O + 6\ OH^- \tag{3}$$

Out of many denitrifiers the most frequent denitrifying bacteria is *Pseudomonas* genera, which can utilize a variety of organic compounds. On top of *Pseudomonas* genera, bacteria such as *Achromobacter, Acinetobacter, Agrobacterium, Alcaligenes, Arthrobacter, Bacillus* and *Chromobacterium* are often recognized denitrifiers. Denitrifying bacteria have the ability to utilize nitrate and nitrite in the lack of oxygen in anoxic conditions. Therefore, for nitrate respiration (denitrification) to occur, anoxic conditions are required. (Tchobanoglous, et al., 2003)

4 Effects of temperature on wastewater treatment

4.1 General

In this chapter, a look into the way temperature influences them is provided. AOB and NOB are strongly influenced by temperature as their activity decreases as the temperature decreases. While this happens, the necessary reactions for nitrogen removal (see equations 1, 2, and 3) slow down. (Tchobanoglous, et al., 2003) Drastic decreases in temperature can also lead to inactivity of the micro-organisms (Delatolla, et al., 2012). The magnitude and time period of the temperature change also influence the recovery of the nitrification capacity. (Rantanen, 2010)

Decreasing temperatures occur often during springtime, when high flows of cold water are observed due to snow melt. High flows pose a risk for failure of ASP as well as for sludge escaping from the secondary settling tank. (Tchobanoglous, et al., 2003) Other effects include the increasing GHG emissions, deterioration of sludge settling properties, effectiveness of process chemicals and how well the sludge can be dried. As heating the inflowing wastewater is not an option in combatting the seasonal nitrification failure, WWTPs have tried to find alternative ways to manage low wastewater temperatures. Nitrification can be kept ongoing despite the decreasing temperatures by controlling MLSS and aeration volume. (Johnston, et al., 2019) Already a reduction of 5 °C in the inflowing wastewater almost doubles the required theoretical minimum sludge age (see *Figure 5*) (Laitinen, et al., 2014).

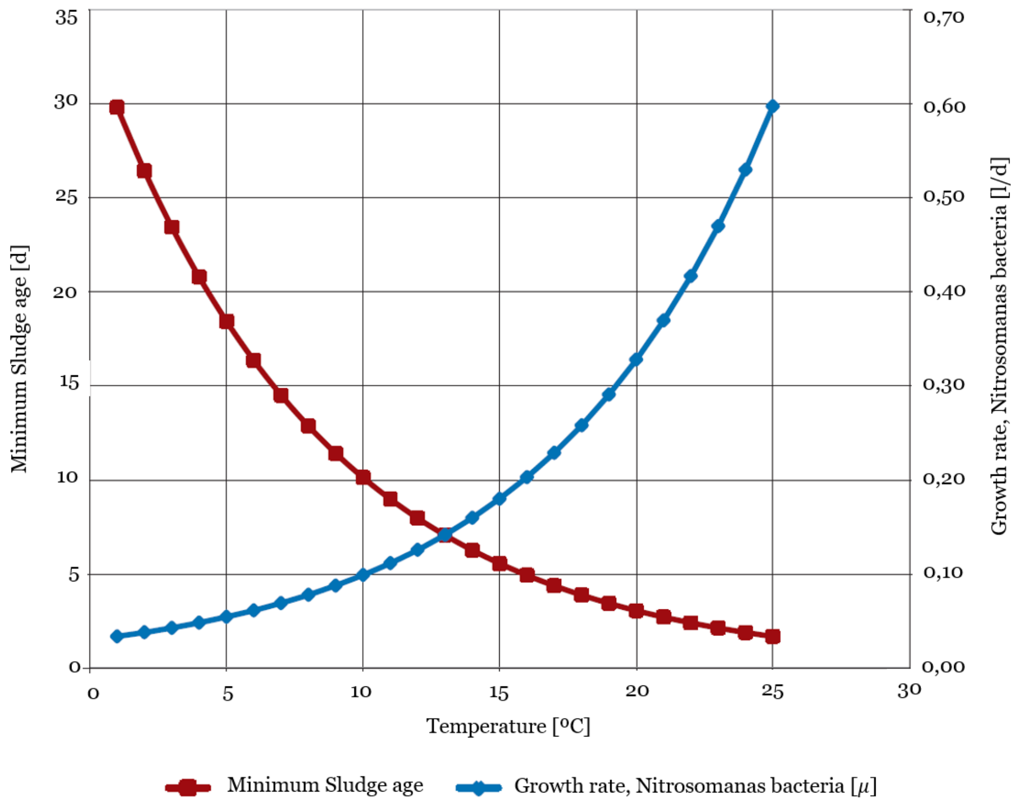

Figure 5. Illustration of temperatures effect on bacterial growth rate and required sludge age by Matti Valve (Laitinen, et al., 2014).

4.2 Effects of temperature on bacteria

It is rather common for WWTPs in temperate climates to experience an extreme drop in nitrogen removal capacity, which is called seasonal nitrification failure. Even during these nitrification failures caused by the dropping temperatures of the inflowing sewage, the plant might still retain its ability to process other contaminants. (Johnston, et al., 2019) This illustrates the temperature dependency of nitrification as well as the nitrogen removal process.

The optimal nitrification temperature range is relatively narrow for bacteria. Mesophilic micro-organisms function optimally in the range of 15 °C – 40 °C (Henze, et al., 2008), but some research studies showcase that nitrification can occur even in very low temperatures, such as 1 °C – 2 °C (Rantanen, 2010). Many suggestions of how AOB and NOB react to changes in temperature have been proposed, but the greatly varying results illustrate the intricacy in predicting the behavior of those organisms in real life. It should also be noted that temperature influences nitrification and denitrification differently due to the growth rates of these bacteria being so different. Denitrifiers are heterotrophic they naturally grow faster than autotrophic nitrifiers (Tchobanoglous, et al., 2003). This is also why denitrifiers are not as sensitive to cold temperatures as the nitrifiers.

Research done by Johnston et al. (2019) states that that the compositions of identified nitrifying bacteria stayed even during seasonal changes in wastewater temperature. They suggested that the fluctuations in nitrogen removal efficiency could be due to changes in nitrifier population. The idea of existing ammonia oxidizers which are unbothered by the seasonal temperature shifts is also introduced. (Johnston, et al., 2019) Lotti et al. (2015) discovered that anammox bacteria that were of low temperature origin reacted differently to the temperature changes than ones which were of warm climate origin. The ways temperature influences the bacteria participating in nitrogen removal has been illustrated in *Figure 6*.

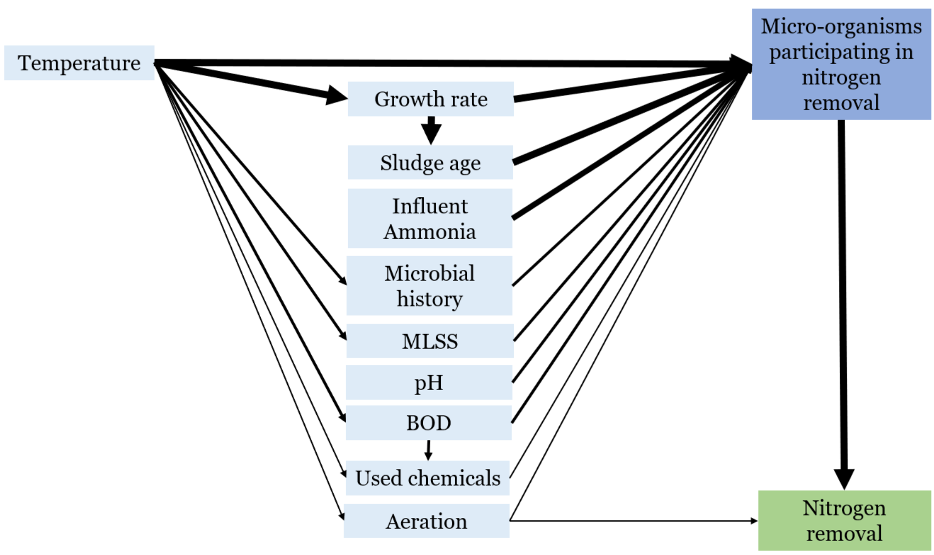

Figure 6. The various ways temperature influences bacteria. The weights of the effects are identified with the thickness of the arrows.

The sludge age can be either expressed as total sludge age or aerobic sludge age. Total sludge age illustrates the theoretical retention time of a sludge particle in the ASP, and aerobic sludge age tells how long the sludge remains in aerobic areas of the ASP. (Henze, et al., 2008) Often repeated connection between wastewater temperature and nitrification is that 1 °C decrease in wastewater temperature leads to 10% decrease in the nitrifying bacteria, therefore leading to 10% longer sludge retention times when aiming to the same level of nitrogen removal (Wanner, et al., 2005). As the dynamics of how microbial communities react to temperature decreasing are unknown, the calculated values like growth rate in regards of temperature can be questioned. Also, many of the bacteria participating in nitrogen removal are unclassified and uncultured (Daims, et al., 2016; Kruglova, et al., 2020) and so there is not yet a change to observe them in a controlled manner in laboratory conditions.

4.3 Nitrification kinetics and temperature

ASP process models are useful in designing, optimizing, and managing the treatment process. Common models include activated sludge models (ASM) 1, 2, 2d, and 3. ASM2d was introduced in 1999 by International water association (IWA) task group (Gujer, et al., 1999). Simulations with ASM require first the inserting the plant definitions, such as sludge retention time (SRT), in the mathematic formulas. Also kinetic, and stoichiometric parameters for both heterotrophic and autotrophic organisms and mean influent concentrations are inserted. (Henze, et al., 1987) To characterize the influent, ASM2d uses fraction-based approach. Fraction-based approach uses fixed ratios for the state variables instead of specific state variables. (Rieger, et al., 2012)

The Monod equation is often used when describing nitrification rates. The rate of nitrite formation equation which bases on the Monod equation is presented below in equation 4. (Henze, et al., 2008)

$$\frac{dN_n}{d_t} = - \frac{dN_a}{d_t} = \frac{1}{Y_A} \frac{\mu_{AmT} N_a}{K_{nT} + N_a} X_{BA} \tag{4}$$

Where

- N_A is bulk liquid ammonia concentration,
- N_n is bulk liquid nitrate concentration,
- μ_{AmT} is maximum specific growth rate at specific ammonia concentration in a specific temperature in certain wastewater,
- Y_A is nitrifier yield coefficient (net organism mass produced per unit mass nitrogen utilized),
- X_{BA} is AOB concentration
- and K_{nT} is half saturation constant in specific temperature.

Equation 4 assumes that the nitrification rate is the same as ammonia conversion rate. It should be also noted that the nitrification rate equation overestimates the rate slightly, as it does not consider the ammonia used to cell mass building by the AOB. (Henze, et al., 2008)

Nitrification rate is described in ASM2d with the following equation 5 (Henze, et al., 1999)

$$\mu_{AUT} \frac{S_{O2}}{K_{O2}+S_{O2}} \frac{S_{NH4}}{K_{NH4}+S_{NH4}} \frac{S_{PO4}}{K_P+S_{PO4}} \frac{S_{ALK}}{K_{ALK}+S_{ALK}} X_{AUT} \tag{5}$$

where

- μ_{AUT} is autotrophic maximum growth rate of autotrophic nitrifying biomass (X_{AUT}),
- S_{O2} is dissolved oxygen,
- $K_{O2}$is saturation coefficient for oxygen,
- S_{NH4} is ammonium,
- K_{NH4} is saturation coefficient for ammonium,
- S_{ALK} is alkalinity (bicarbonate),
- K_{ALK} is saturation coefficient for alkalinity,
- K_P is saturation coefficient for phosphorus,
- S_{PO4} is phosphate,
- and X_{AUT} is autotrophic nitrifying biomass.

Denitrification rate is modelled with the following equation 6 in ASM2d model (Henze, et al., 1999),

$$\mu_H \eta_{NO3} \frac{K_{O2}}{K_{O2}+S_{O2}} \frac{S_{NO3}}{K_{NO3}+S_{NO3}} \frac{S_A}{K_A+S_A} \frac{S_A}{S_F+S_A} \frac{S_{NH4}}{K_{NH4}+S_{NH4}} \frac{S_{PO4}}{K_P+S_{PO4}} \frac{S_{ALK}}{S_{ALK}+S_{ALK}} X_H \tag{6}$$

where

- μ_H is heterotrophic maximum growth rate on substrate, η_{NO3} is anoxic hydrolysis reduction factor,
- $K_{O2}$is saturation constant for oxygen,
- S_{O2} is dissolved oxygen,
- S_{NO3} is nitrate and nitrite,
- K_{NO3} is saturation constant for nitrate,
- S_A is acetate as fermentation product,
- S_F is readily biodegradable substrate,
- K_A is saturation coefficient for nitrate,
- S_{NH4} is ammonium,
- K_{NH4} is saturation coefficient for ammonium,
- S_{PO4} is phosphate,
- K_P is saturation coefficient for phosphate,

- X_H is heterotrophic biomass
- and S_{ALK} is bicarbonate alkalinity.

The temperature dependency of the nitrification rate is due to the factors μ_{AmT} and K_{nT}. To explain it simply, temperature decreases growth rate and increases half saturation constant for nitrifiers. When temperature drops, so does μ_{AmT}, which means that the growth rate of the bacteria slows down. This leads to the fact that the sludge age needs to increase in order to have enough bacteria in the sludge. When there is a need to increase the sludge age, MLSS also increases. On the contrary, when temperature decreases, so does K_{nT}, which leads to decrease of the ammonium content of effluent. The μ_{AmT} rate also decreases as the temperature decreases, which ultimately will deflect the effect of changing K_{nT} value and raise the ammonium content of the effluent. Ammonium content illustrates the efficiency of the nitrification process, as it tells how much ammonium is still left. This means that as the temperature rises, the effectiveness of nitrification also increases to certain degree. After certain tipping point, increasing temperature will not work as an intensifying factor, but rather prohibit the process due to the bacteria denaturizing. This also means, that as the temperature decreases, so does the effectiveness of nitrification. (Henze, et al., 2008) The optimum temperature for nitrification WWTPs is somewhere between 15 °C and 35 °C (Shammas, 1986).

The effects of temperature on biological reaction rate can be put as follows:

$$k_T = k_{20}\theta^{(T-20)} \tag{7}$$

where

- k_T is the reaction-rate coefficient in certain temperature,
- k_{20} is the reaction-rate coefficient at 20°C,
- θ is the temperature coefficient,
- and T is the temperature.

This equation 7 bases on van't Hoff-Arrhenius relationship, which describes the reaction rates relationship with temperature. (Tchobanoglous, et al., 2003) In ASM2d a similar Arrhenius-type equation is used to express the effect of the temperature on the reaction rates. (Rieger, et al., 2012) ASM2d is recommended to be used in temperatures between 10 °C and 25 °C. (Henze, et al., 1999)

Again, as presented in the previous section, the equations describing temperature dependency of nitrification can be questioned due to inherent differences in the differences between laboratory environment and WWTPs. As the composition and dynamics of the microbial communities in WWTPs are still not fully discovered (Daims, et al., 2016) can the equations and values defined in laboratory conditions be considered as sophisticated estimates of the actual values. As even in recent years the understanding of the microbes participating in nitrogen removal has grown enormously (Daims, et al., 2015; van Kessel, et al., 2015), the suspicions in the values defined before this knowledge also increase respectively. Some of the values found in literature are gathered in table 1. The values seen in table 1 can be applied in equations 5 and 6.

Table 1. Various values related to nitrification and denitrification found in literature for different temperatures.

		5 ºC	10 ºC	14 ºC	15 ºC	20 ºC	22 ºC
Sludge age **[d]**	(Laitinen, et al., 2014)	18	10	-	6	3	-
	(Tchobanoglous, et al., 2003)	-	10-20	-	-	4-7	-
	(Randall, et al., 1992)	-	-	5	5	-	-
Nitrification							
KAm **Maximum nitrification rate [mgFSA-N/mgANOVSS.d]**	(Henze, et al., 2008)	-	-	0,034	-	1	0,0425
μ_{AUT} **Maximum growth rate** **[d^{-1}]**	(Laitinen, et al., 2014)	0,05	0,10	-	0,19	0,32	-
	(Henze, et al., 2008)	-	-	0,224		0,45	0,568
	(Randall, et al., 1992)	-	0,10–0.29	-	0,18-0,47	0,26-0,77	-
	(Gujer, et al., 1999)	-	0,35	-	-	1	-
Denitrification							
K_{dmax} **Maximum denitrification rate [mgNO3-N/mgOHOVSS.d]**	(Henze, et al., 2008)	-	-	0,04-0,241	-	0.048-7,20	0,051-1,036
K_{dmax} **Maximum denitrification rate [mgN/gVSS.h]**	(Cherchi, et al., 2009)	-	2,3	-	-	6,1	-
μ_H **Maximum growth rate** **[d^{-1}]**	(Cherchi, et al., 2009)	-	0,3	-	-	1,3	-
	(Gujer, et al., 1999)	-	1	-	-	2	-

4.4 Effects of temperature on investment and operation costs

In Finland, nitrogen removal will add 15% - 30% to financial investments compared to a WWTP without nitrogen removal. Approximately 25% of total operational costs of a WWTP can be attributed to nitrogen removal. The majority of operational cost of nitrogen removal is caused by aeration and sludge circulation. (Laitinen, et al., 2014) When building a WWTP, the ASP reactor size is designed keeping in mind the sufficient nitrification reduction level that is required of the plant. As the wastewater temperature is lower, larger reactor sizes are required which lead to larger building investments.

Factors contributing to operational costs of the ASP are alkalinity chemicals, and additional sources of carbon. Sometimes the long sludge age can cause problems with the settling properties of the sludge. This may require additional human resources compared to treating wastewater without total nitrogen removal. (Laitinen, et al., 2014) As a large part of the operational costs consist of aeration system and pumping costs, the costs of mitigating the effects of decreasing temperature will have a relatively large effect on the total finances of WWTPs.

One way to retain nitrification capacity in low temperature is to apply carriers, or other fixed-film technology, which can be used when there is no possibility to increase reactor sizes. The carriers help to retain nitrification bacteria and nitrogen removal on a higher level in challenging times. (Laitinen, et al., 2014) Reactors with carriers like moving bed biofilm reactor (MBBR) and integrated fixed film activated sludge process (IFAS) make nitrification more efficient in the same reactor size than reactors without carriers. These reactors also help to recover the nitrification rates after drops in process temperature by maintaining high levels of nitrification bacteria. (Rantanen, 2010) Incorporating such technology to balance the effects of wastewater heat recovery would naturally increase the financial investments of a WWTP.

4.5 Effects of temperature on sludge properties

Temperature influences settling characteristics of biological solids. (Tchobanoglous, et al., 2003) The settling rate is highly dependent on the temperature of the sludge. Study by Hayet et al. (2010) suggests that increase in the settling rate can be noticed when temperature increases from around 20 °C to past 30 °C. Settling rate is also assumed to decrease as temperature decreases. (Tchobanoglous, et al., 2003)

The way temperature influences settling is assumed to be through a variety of factors, such as turbidity, viscosity, and density of the wastewater. Increase in the temperature increases the turbidity of the wastewater. (Hayet, et al., 2010) In cold climates, the viscosity is increased to a level that interferes with the flocculation and settling of the particles. The time for settling is 1,38 times higher in 13 °C than in 20 °C. Temperature also influences the operation of settling tank greatly. If the range of wastewater temperature is too drastic (more than 1 °C) in the tank, a density current will form and hinder the efficiency of the settling tank. The great variety of size, shape, and flexibility of the particles in wastewater make the estimation of their actual behavior difficult. (Tchobanoglous, et al., 2003)

Microflocculation, meaning the flocculation of small particles, is less efficient in lower temperatures, as it is mostly driven by Brownian motion. Brownian motion stems from the movement heat generates in small particles, therefore lower temperature leads to less microflocculation. Microflocculation is vital for performing macro flocculation, which is a key part of settling. Floc formations also aids in the dewatering of the sludge (Tchobanoglous, et al., 2003); the dewatering of the sludge becomes more difficult in cold conditions than in warm conditions. Flocs formed in low temperatures are also more irregularly structured than ones in warm conditions, which weakens the settling capabilities of the flocs. (Xiaoa, et al., 2009)

Another problem caused by temperature in regards of settling, is the growth of filamentous bacteria. Filamentous bacteria are known to affect settling and floc formation. (Mielczarek, et al., 2012) Low temperatures are observed to cause increase in the growth of filamentous bacteria. (Fan, et al., 2018) The most common filamentous bacteria in WWTPs is seen to be Microthrix, which was especially large community during cold periods. (Mielczarek, et al., 2012) Settling is a vital part of WWTPs and the poor settling performance causes issues with the overall level of water treatment. Presence of filamentous bacteria causes also issues in other unit processes of the WWTP, e.g. in anaerobic digestion plants where they cause excessive foaming (Ganidi, et al., 2009).

Abundance of filamentous bacteria is also connected with difficulties in sludge-water separation, which lead to issues with drainability. (Fan, et al., 2018) Flocs formed in warm conditions are more compact than ones formed in cold conditions. This together with a lower floc strength during the winter conditions affects the dewatering properties of the sludge. (Mielczarek, et al., 2012) Dewatering is a process, which can have great effects on not only the GHG emissions of the WWTPs but also the finances through the transportation volumes of the dried sludge and through polymer consumption. These effects are gathered and illustrated in *Figure 7*.

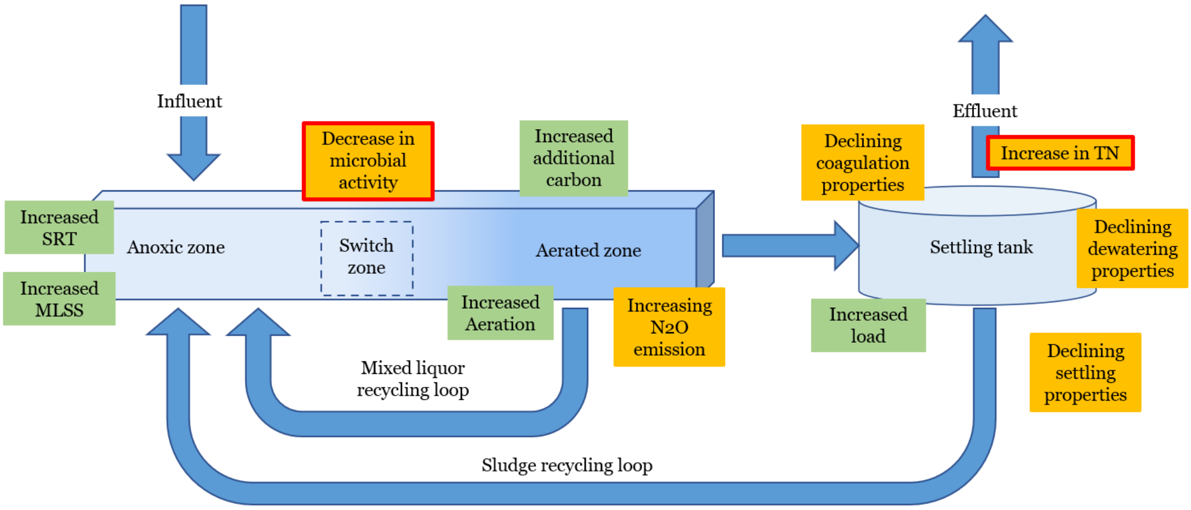

Figure 7. Effects of decreasing wastewater temperature on the ASP. Effects that occur as direct consequence of the drop in temperature are illustrated with orange, and effects that come secondary from the necessary mitigation actions are illustrated with green. The main effects are highlighted with a red line.

4.6 Effects of temperature on GHG emissions

GHG emissions of the WWTPs mainly consist of N_2O emissions (Foley, et al., 2010). Other contributing factors mainly include energy and chemical consumption. Temperature changes can influence the nitrous oxide emissions of the WWTP. Nitrous oxide is a powerful GHG, it has around 300 times the global warming potential of CO_2. (IPCC, 2007) Nitrous oxide emissions are released in WWTPs in the biological nitrogen removal process in the denitrification as well as ammonia oxidization reactions and might contribute to a large part of total GHG emissions from WWTPs (Law, et al., 2012). In nitrification N_2O can be released through two different paths in nitrification, called hydroxylamine oxidation and nitrifier denitrification. These can be seen in *Figure 4*. In denitrification N_2O is an intermediate product of the reaction, and so can also be released from this process. (Wunderlin, et al., 2012) N_2O emissions are mostly released from the aerated part of the ASP, as the aeration air efficiently conveys the N_2O in the liquid to the gas phase (Law, et al., 2012). In decreasing temperatures less energy is needed for gas transfer, which is why aeration could be seen to work more efficiently (Frijns, et al., 2013).

Abruptly changing conditions could be seen to cause stress for the nitrifying and denitrifying bacteria and so cause changes in their behavior. This has been noted to lead to N_2O accumulation especially from AOB. This is why N_2O levels can elucidate the possible failures e.g. in aeration and nitrification. The stress the bacteria are experiencing could also be many transitions between aerated and anoxic zones. Enough time in the anoxic zone should be provided for the sludge, so that the N_2O emissions could be managed before they escape into air in the aerated zones. Other factors resulting in low N_2O are long solids retention times, high sludge recycling rates and large biological process size. (Law, et al., 2012)

Nitrite accumulation has been recognized as a possible factor leading to high N2O emissions. Other factors that can lead to increase in N2O emissions are low levels of dissolved oxygen (DO) and COD and abrupt changes in NH4. pH and DO are good ways to try to control N2O emissions in Finnish conditions. (Kuokkanen, et al., 2021) Research on N2O emissions of WWTPs demonstrates that aiming towards low level of emissions does not compromise the nitrogen removal level. The amount of N2O emissions in relation to nitrogen load vary from 0% to 25% between different WWTPs. (Law, et al., 2012)

4.7 Effects of temperature on consumption of chemicals

Temperature also effects on the chemicals used in WWTPs. Added chemicals often include external carbon, pH control chemicals, coagulant, and polymers. Lime and sodium carbonate are added to recover alkalinity after nitrification had decreased it. (Tchobanoglous, et al., 2003) Often when WWTPs struggle with low temperatures and nitrification efficiency is decreasing, nitrification is prioritized over denitrification. This leads to a possible switch zone being used as an aerated zone instead of anoxic zone, and as a result denitrification does not have as long time to recover alkalinity in the ASP. This is why more alkalinity chemicals might be needed to recover alkalinity in increased amounts during cold time periods.

Temperature of wastewater also influences the use of coagulation chemicals, such as ferrous sulphate and a variety of polymers. Coagulation and flocculation become slower in when temperature decreases. (Xiaoa, et al., 2009) As the issue of slow coagulation in cold conditions stems from hindered microflocculation, the coagulation chemicals can only offer limited help.

External carbon, such as methanol, can be added to the process to increase the rate of denitrification and achieve a higher level of total nitrogen reduction. (Tchobanoglous, et al., 2003) The need for an external carbon source can increase for denitrification to be as efficient as possible in a shorter time than in warm conditions. Cherchi et al. (2009) also observed the temperature dependency of bacteria that is acclimated to using methanol as a carbon source, and they noted that methylotrophs had relatively long SRT in low temperatures compared to bacteria utilizing different carbon sources. A great decrease of 73% in denitrification was observed when temperature decreased from 20 °C to 10 °C while nitrite was simultaneously accumulating in ASP. (Cherchi, et al., 2009)

5 Research material and methods

5.1 Data analysis materials

The aim of the data analysis is to find out the way temperature and nitrogen removal interact in real life in three Finnish WWTPs. The results will support the simulations and allow a deeper the discussion in the following chapters. The materials used were from three WWTPs in southern Finland, Suomenoja (HSY), Viikinmäki (HSY) and Kakolanmäki (Turun Seudun Puhdistamo Oy). These three treatment plants have an activated sludge process for nitrogen removal and the details of these treatment plants can be seen in Table 2. It is important to note that as the used material is from Finnish treatment plants, the results only represent that context. These WWTPs are large facilities in Finland and are also participating in the larger wastewater heat recovery project this thesis is also a part of.

Table 2. Details of the three wastewater treatment plants used in the study. Information gathered from and based on (Leino, 2020), (HSY, 2020) and (HSY, 2020).

	Kakolanmäki	Suomenoja	Viikinmäki
Established	2008	1963	1994
Population connected to wastewater network	275 000	390 000	860 000
% of industry flow	7%	8%	15%
Av. flow [m^3/d]	80 000	100 000	280 000
Rainwater inflow & infiltration	40%	No data	No data
Combined sewer %	2,5%	No data	<10%
ASP volume [m^3]	60 000	36 000	103 500
Max flow [m^3/d]	240 000	250 000	700 000
BOD7 load [kg/d]	22 000	16 800	69 000
P load [kg/d]	580	670	2 100
N load [kg/d]	4 400	3 800	15 500
Nitrogen population equivalent*	293 000	253 000	1 033 000
Activated sludge lines	4	10	9
Post denitrification filter	No	No	yes

*Nitrogen population equivalent was calculated by assuming that one person produces 15 g of nitrogen per day.

The attributes of the WWTPs are important to note as they effect how the plants are operated. Suomenoja WWTP is on the other hand old compared to the two other WWTPs and will not be working for much longer, as it is nearing the end of its lifecycle. Viikinmäki has a post denitrification filter as well as pre-aeration as additional processes to the common WWTP processes. Especially post denitrification filter influences the way the plant is run. Suomenoja WWTP also has pre-aeration, which is located before the primary settling in the wastewater treatment process. (HSY, 2020) Kakolanmäki has sand filtration after the ASP process and physico-chemical process for by-pass water treatment (Leino, 2020), which the two other WWTPs do not have. On top of the differences with the plants themselves, the quality of water varies between cities also due to different populations, infrastructure, and weather conditions.

The material used represents a five-year period from 1.1.2016 to 1.1.2021. Some unusual disturbances in measuring had occurred during these years, which is why some values are disqualified to be used in this data analysis. Most of the laboratory values were not measured daily (nitrogen flows, suspended solids (SS), COD) but instead they were measured two to three times a week. This measuring frequency can be assumed to give an accurate picture of the real functioning of the WWTPs in the chosen time scale. The online data (flows and temperatures) is daily averages throughout the five-year period.

Some additional preparations were done in order for the data to be utilized in the analysis. For example, some data of incoming wastewater temperature was missing from one of the WWTPs due to its discontinued measurement. As knowing this information was vital for investigating the relationship between influent wastewater temperature and nitrogen removal efficiency, the missing values were estimated by using the existing data of two and half years and the full data set of sand and grit removal process temperature.

5.2. Data analysis methods

To analyze the relationship between influent temperature and nitrogen removal efficiency, the interactions between wastewater temperature and nitrogen removal, ammonium removal, BOD removal, aeration air, chemicals, and N_2O emissions are researched. Calculating the amount of total nitrogen, ammonium and BOD removed by cubic meter of ASP is done to understand the load WWTPs are experiencing.

The values used to calculate the nitrogen removal related values are strictly of the ASP of the WWTPs, meaning that the inflowing nitrogen amount is the nitrogen amount entering ASP and leaving nitrogen value is the one which leaves the ASP. This is done in order to observe the nitrogen dynamics inside the ASP rather than observing the dynamics of the whole plant.

The main data analysis methods are mathematical calculations to find out the wanted values based on the data received, and the graphic representation and statistical study of the calculated values. The statistical study includes examination of the Pearson correlation coefficients to further observe the relationship between wastewater temperature and nitrogen removal. Calculations to find out average values for summer and winter periods are done in order to highlight the differences between cold and warm conditions. In this thesis, summer and winter periods are divided according to the influent wastewater temperature. The limit is set at monthly average of 12 °C, below which the water is considered cold. 12 °C was chosen as a limit as it was used in the relevant Finnish legislation. In Finland, the required nitrogen removal efficiency is determined in the environmental permit of the WWTP (Valtioneuvoston asetus yhdyskuntajätevesistä 888/2006). It should be noted that the chosen temperature division influences the received results.

5.3 Simulation materials and methods

The aim of the simulations performed is to find out what are the effects of temperature on nitrogen removal efficiency when sludge age, MLSS and process volume is constant. Also, the needed increase of process volume for the WWTP to maintain a sufficient nitrogen removal level when the wastewater temperature decreases is researched. The goal of these simulations is not to model exact treatment plant to full extent and intense detail, but to give insight to the way ASP reacts to temperature in a controlled manner.

The influence of wastewater temperature on the total nitrogen removal was done by simulating the 24-day period dynamically in temperature between 17 °C and 5 °C. The 24-day dataset consisted of a 7-day intensive monitoring period data being repeated to achieve 24-day data period. Rejected sludge share, sludge age and sludge pumping were not manipulated in the process simulations. After the simulations, the total nitrogen load in the influent and in the effluent was analyzed to understand the effects on total nitrogen removal efficiency. When performing process simulations in order to find out the needed process volume increase, a series of simulations were done. The needed process size was estimated by performing numerous simulations at each temperature and by analyzing the total nitrogen effluent in each scenario. The volume of aerated zones in the ASP model were set by zone-based approach. This is how only the size of each ASP line could be increased in the model while the ASP process stayed similar.

A model prepared and calibrated for the Kakolanmäki WWTP in 2016 was used in the simulations. The model comprises primary settling reactor and activated sludge process of the Kakolanmäki WWTP. It has been calibrated with data from a 7-day intensive monitoring period from 16.9.2015 to 23.9.2015, during which process temperature varied between 16,3 °C and 18,1 °C. (Pöyry Finland Oy, 2016) The model was prepared, and the simulations were conducted with the software GPS-X v.8.0.1. In this model, simulated wastewater comes from fine screening, then goes from pre settling to aeration and finally to secondary settling. The model used when simulating the ASP is a biological model and bases on ASM2d. Secondary settling tank was simulated with Simple 1D -model, which is a Takacks -model with 10 layers. Secondary settling tank also included simulated biological reactions.

The effect of temperature is considered with temperature coefficients which can be seen in table 3. These used temperature coefficients seem to be very similar to the found literature values. The main temperature coefficients that influence the nitrogen removal efficiency that have been defined in the model are the temperature coefficient for maximum heterotrophic growth rate and the temperature coefficient for maximum autotrophic growth rate.

Table 3. Temperature coefficients in the model and in the literature.

	Model value	Literature value, (Henze, et al., 2008)
Maximum heterotrophic growth rate	1,072	1,029-1,20
Heterotrophic lysis and decay rate	1,072	-
Maximum autotrophic growth rate	1,111	1,123
Autotrophic decay rate	1,111	-
Hydrolysis rate	1,401	-

Other constants used in the model which influence nitrification are presented in table 4. When comparing the constants to the ones seen in literature, it can be seen that the model values seem to fit into the frame of the literature values.

Table 4. Kinetic parameters used in the model and in literature in 20 °C.

	Model value	Literature value (Henze, et al., 1999)	Literature value For references, see table 1.
Maximum heterotrophic growth rate (substrate) $[d^{-1}]$	2	6	1,3-2
Maximum heterotrophic growth rate (fermentation) $[d^{-1}]$	2	3	1,3-2
Maximum autotrophic growth rate $[d^{-1}]$	1	1	0,26-1

The simulations performed in this thesis are performed between temperatures 5 °C and 17 °C. This temperature scale was chosen due to the importance of paying attention to low temperatures, as the nitrification becomes challenging in cold scenarios. The extension to 17 °C is needed to be able to compare the effects of low temperature to more favorable conditions in regards of biological water treatment.

The aeration control strategy applied at the WWTP was implemented in the process model. Online measurement of ammonium nitrogen is utilized for cascade control of dissolved oxygen level and airflow. The simulated ASP, there is first zones 1 and 2, which are anoxic, after which the wastewater flows into zone 3. Zone 3 is the switch zone. Zones 4 to 6 are always aerated. The aeration of the switch zone (zone 3) is on, when the ammonium level in zone 5 exceeds 4 mg/l and switched off when the ammonium concentration in zone 5 decreases again under 3 mg/l. This is done in order to have successful nitrification. *Figure 8* and *Figure 9* below further illustrate the aeration controls used in the simulations. As can be seen from the *Figure 8*, as the ammonia nitrogen concentration rises above 4 mg/l in the simulations, the switch zone turns on. In *Figure 9*, the switch zone is mostly on and only turns off when the simulated ammonium concentration is below 3 mg/l.

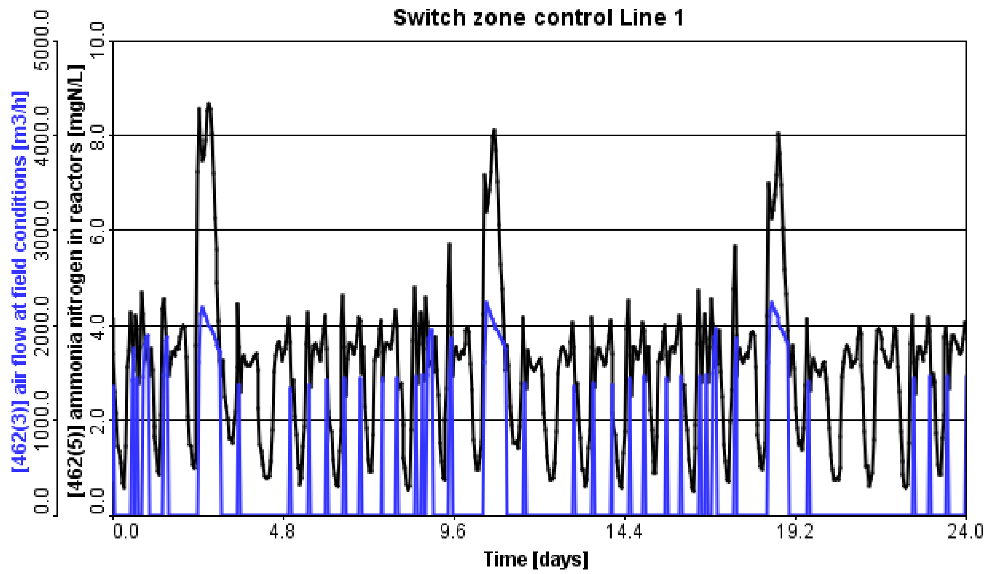

Figure 8. Switch zone aeration when the inflowing wastewater temperature is 15 °C.

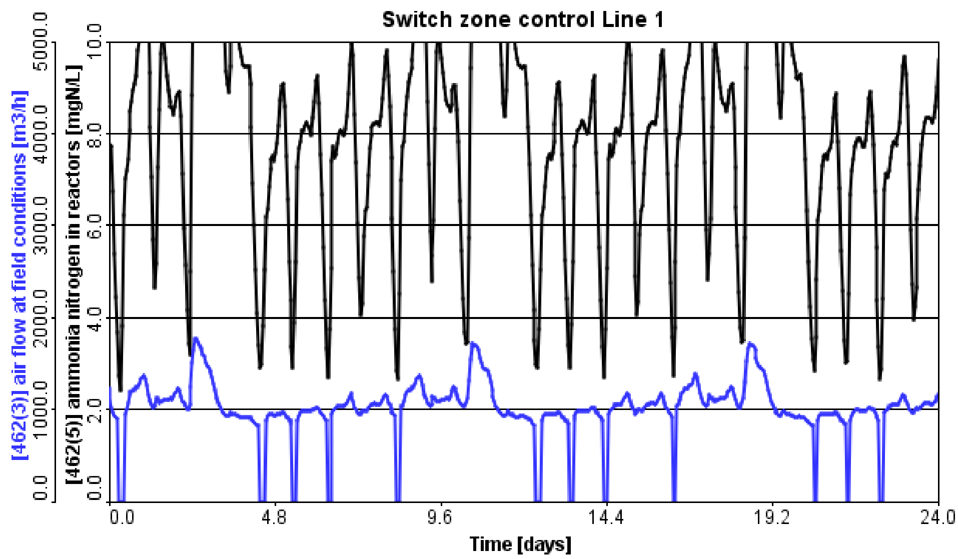

Figure 9. Switch zone aeration when the inflowing wastewater temperature is 5 °C.

6 Results

6.1 Data analysis results

6.1.1 General

Figure 10 illustrates the changes in the inflowing wastewater temperature throughout the five-year data-sampling period. As can be seen from the data, the wastewater temperature dynamics of the three WWTPs are rather similar and the monthly averages vary between 9 °C and 20 °C. The highest temperatures occur yearly during August and September, and lowest temperatures during March and April. The wastewater flows peak during December and early springtime. During springtime the wastewater temperature decrease is due to snow and ice melting and the cold water entering the wastewater network through inflow and infiltration and combined sewers. It is difficult to separate the effect the growing flow has on nitrogen removal from the wastewater temperature when analyzing real data, as these phenomena are often connected to each other. This also supports combining data analysis and simulations as the research methods in this thesis. The simulations complement the data analysis, since in simulations the effects of wastewater flow and temperature can be separated which cannot be done in the data analysis.

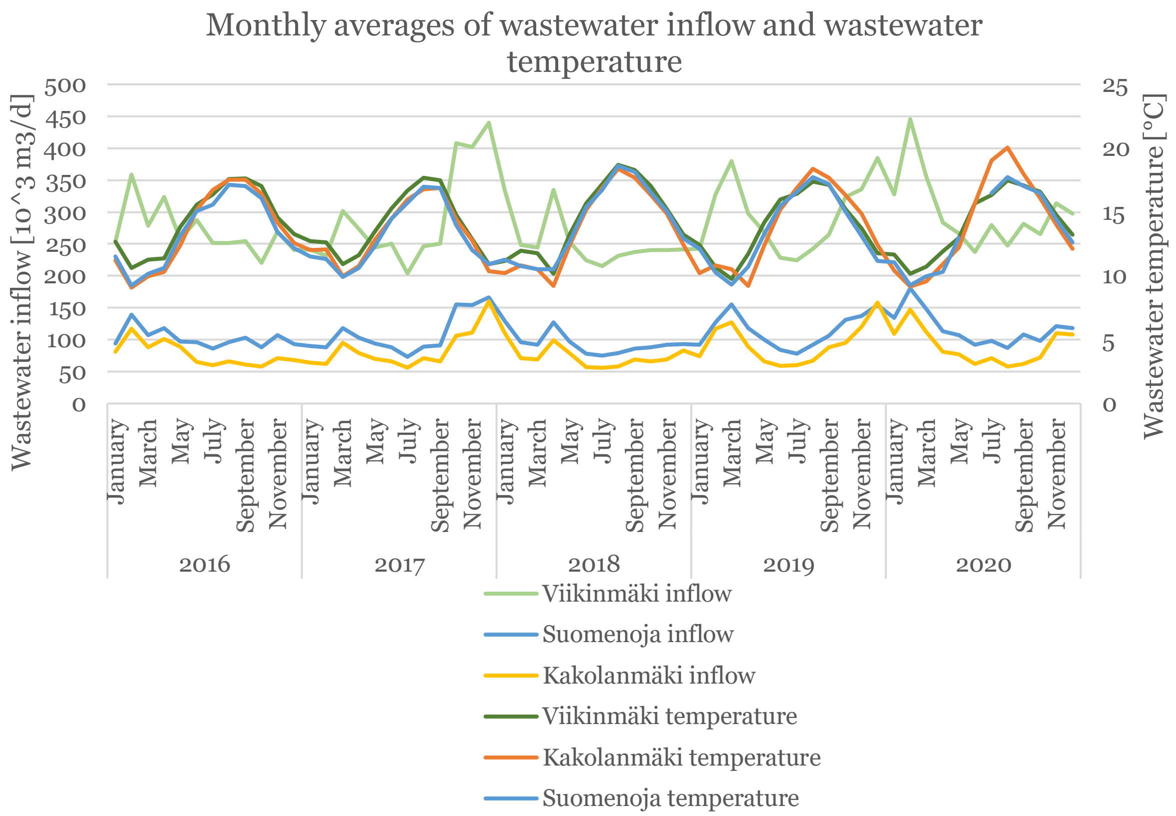

Figure 10. Timeline of the process temperature and wastewater flows of the three wastewater treatment plants for five-year period.

Combined sewers share is important to know as the type and quality of network influences results of the analysis. As seen on table 2, Suomenoja does not have any mixed drainage, meaning that the rain events should not influence the wastewater flow in such a scale as it does on Kakolanmäki and Viikinmäki. Mixed drainage also leads to wastewater heat recovery effecting the wastewater temperature arriving in WWTP in a more complex and ambiguous ways than in non-mixed drainage networks. This is due to cold rainwater entering the networks during rain events more effectively than it enters networks which do not have mixed drainage areas. This is the case also with a network that is in a bad condition, which allows more seepage to happen. For example, most of the rainwater arriving in Kakola enters the network through seepage, not due to mixed drainage (Leino, 2020). *Figure 11* and *Figure 12* illustrate the monthly averages of rainfall and volume of inflowing wastewater in the three WWTPs. Viikinmäki and Suomenoja WWTP are in the same graph, as they are geographically close to each other so that the rainfalls are close to identical.

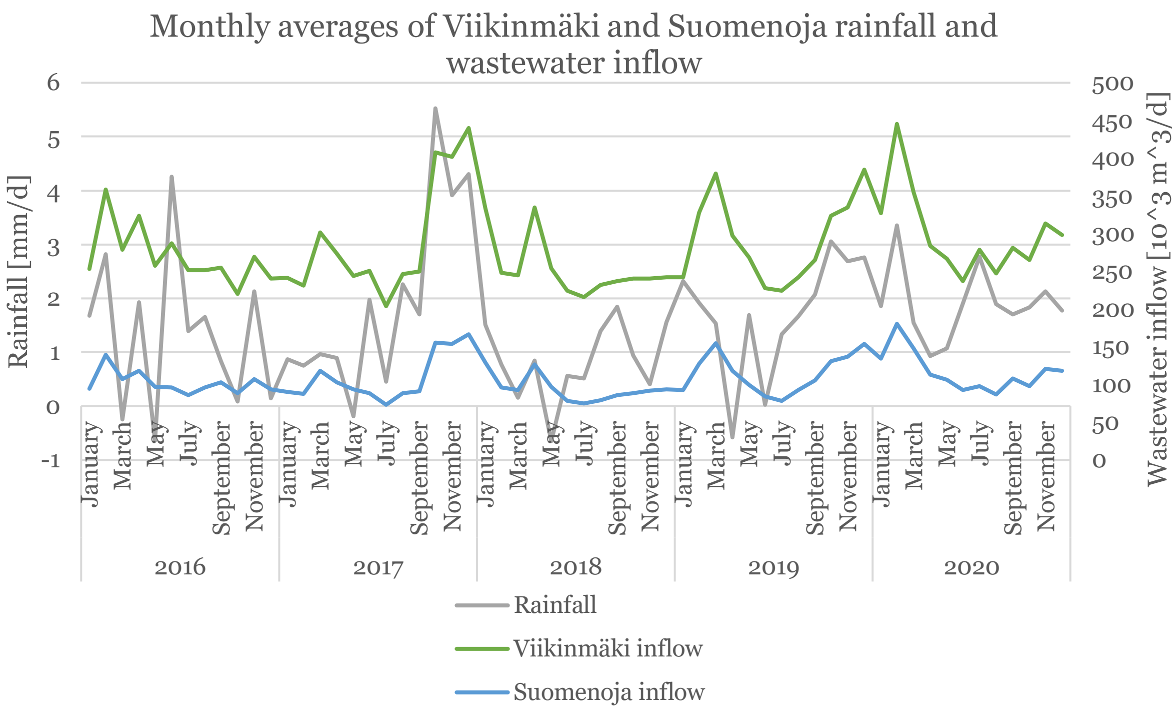

Figure 11. Timelines of average rainfall and wastewater flows in Viikinmäki and Suomenoja WWTP.

Figure 11 shows how the wastewater flow have been impacted by the rainfall. Autumn 2017 and late 2019 exemplify a great increase in rainfall that can be seen in the wastewater flows of Viikinmäki and Suomenoja. The same phenomenon can be seen in *Figure 12* during the same time periods. As the rainfall influences the wastewater flow, a straightforward relationship between the nitrogen content in the wastewater and the quantity of wastewater cannot be assumed. Rainfall can also have direct effects in the possibility to remove nitrogen in the WWTPs. As great amounts of wastewater arrive into the WWTPs due to rain events, the sudden increase in the inflowing wastewater may result in the need to bypass treatment processes especially if there is not a lot of free capacity in the WWTP. The wastewater treatment result does not only depend on the functioning of ASP but also on the amount of bypass during extreme rain events.

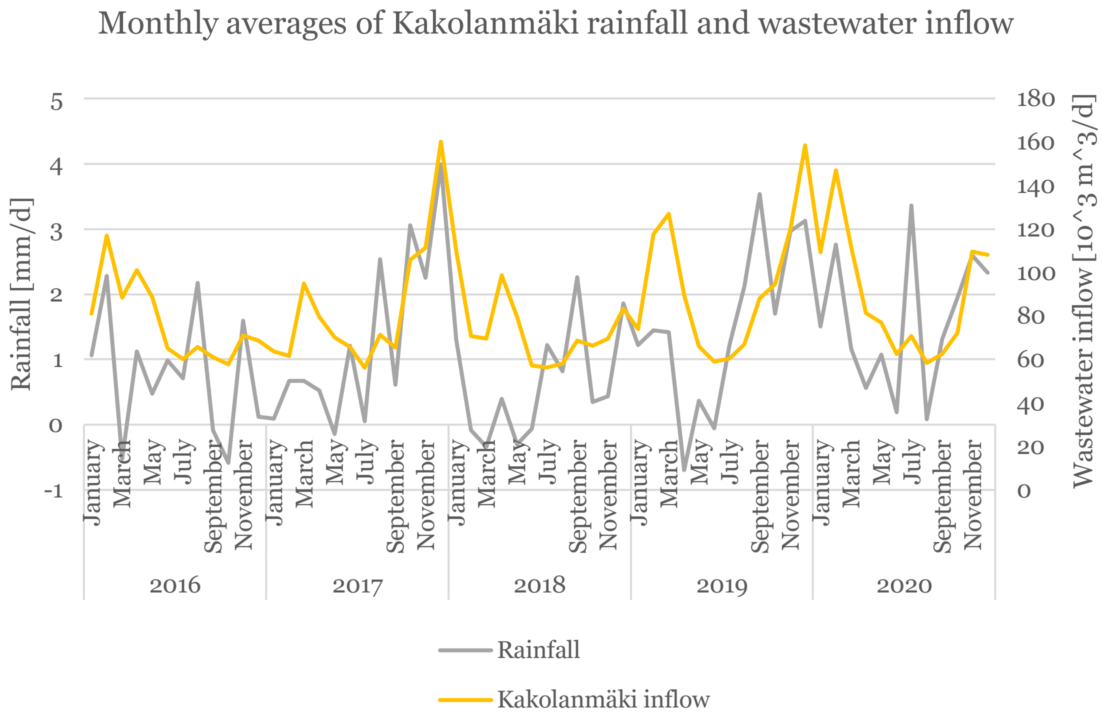

Figure 12. Timelines of rainfall and the wastewater flows in Kakolanmäki WWTP.

The high flows experienced might also result sludge escaping the ASP and therefore loss of nitrifying biomass. As high flows seen in *Figure 11* and *Figure 12* are often a result of snow and ice melt in the spring, the great flows also come in low temperatures. This low temperature can be seen to amplify the effect the large flows have on the ASP and especially nitrogen removal.

Figure 13 displays a scatter dot graph between wastewater temperature and used total sludge age. All of the WWTPs use slightly longer sludge age when the wastewater temperature is low compared to the situation with warmer wastewater temperature. Viikinmäki uses relatively short sludge age and Kakolanmäki relatively long one. Based on the data of these WWTPs, long sludge age does not seem to directly result in higher nitrogen removal efficiency. This is expected, as long sludge age is often used in WWTPs to manage the decreases in nitrification and keep it going despite challenging conditions.

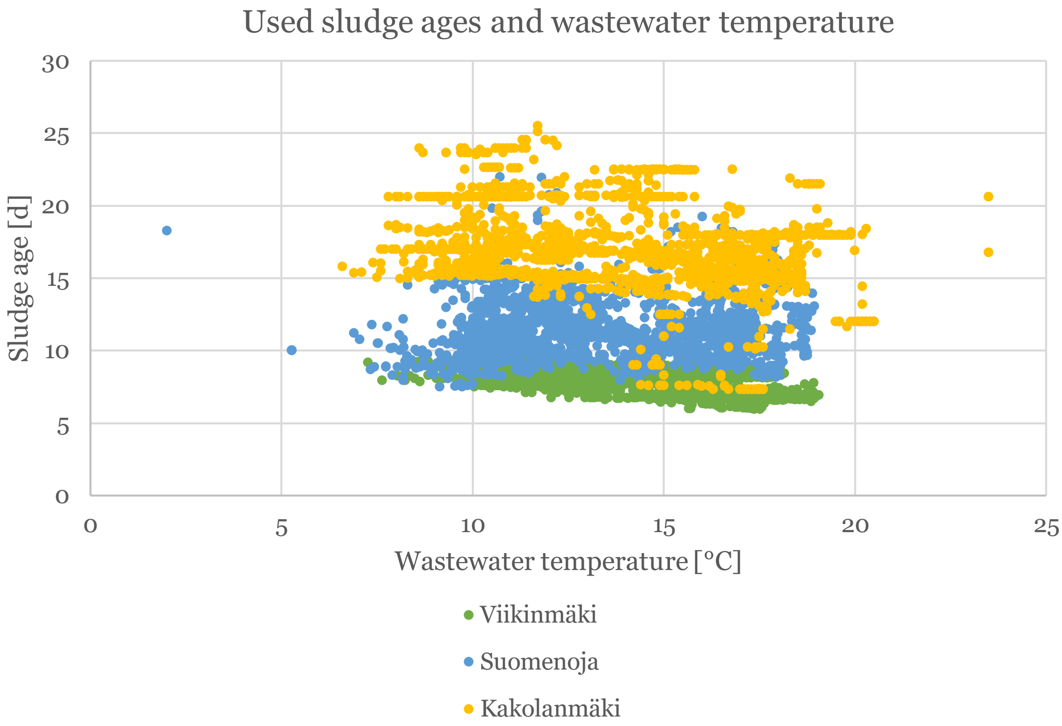

Figure 13. Used sludge ages and wastewater temperature on three WWTPs.

6.1.2 Temperature and nitrogen removal

This chapter presents the results of the data analysis, which answer the previously set research question of "What is the observed effect of temperature on the ammonium and total nitrogen removal in Finnish wastewater treatment plants". First the timelines of total nitrogen removal and ammonium nitrogen removal are observed. As was seen from *Figure 10*, the temperatures of the three WWTPs are very similar to each other throughout the five-year period. This is why in *Figure 14* and *Figure 15* the average of these temperatures is used to simplify the figures. *Figure 14* and *Figure 15* illustrate the timeline of ammonium nitrogen removal and total nitrogen removal on the three wastewater treatment plants as percentages.

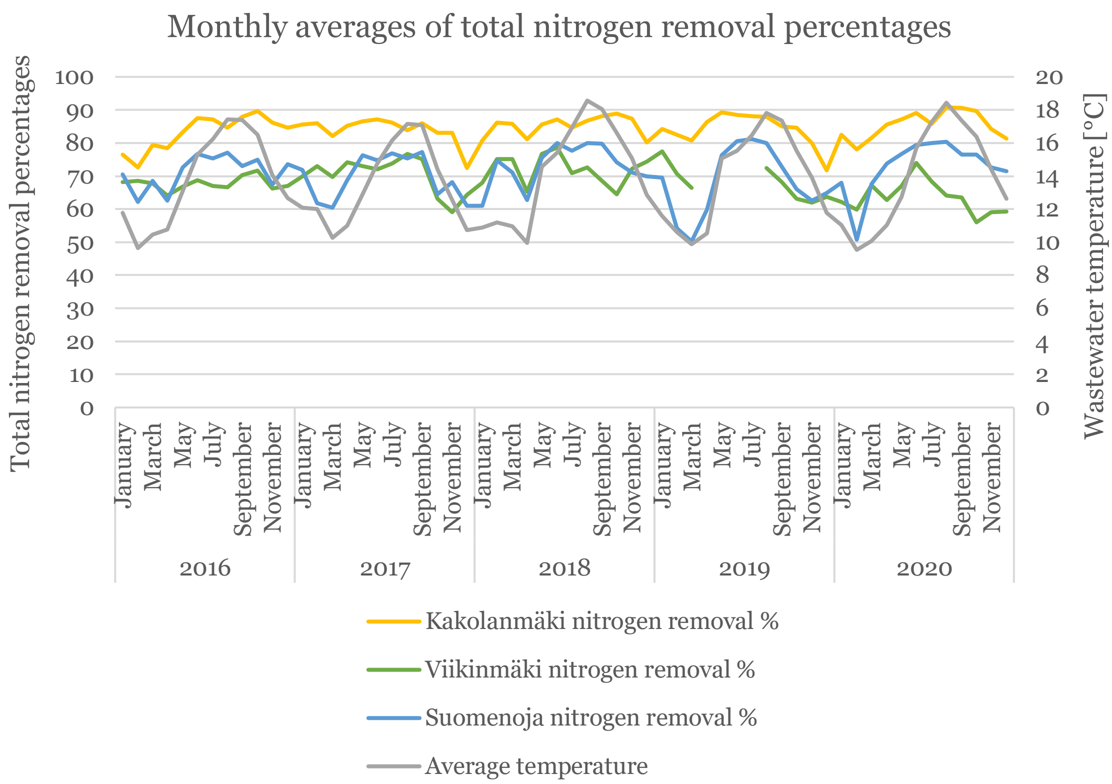

Figure 14. Total nitrogen removal percentages presented for a five-year period as monthly averages.

When observing *Figure 14* and *Figure 15*, it is evident that a drop occurs during the colder periods of each year during the five-year period in total nitrogen as well as ammonium removal efficiency on each of the WWTPs. All of the three WWTPs seem to have very similar ammonium removal efficiency, but with total nitrogen removal efficiency more variance can be seen. When compared to the rainfall and flows showcased in *Figure 11* and *Figure 12*, the great rain events in late 2017 and 2019 can be seen in *Figure 14* and *Figure 15* also as a drop in the ammonium and total nitrogen removal percentages.

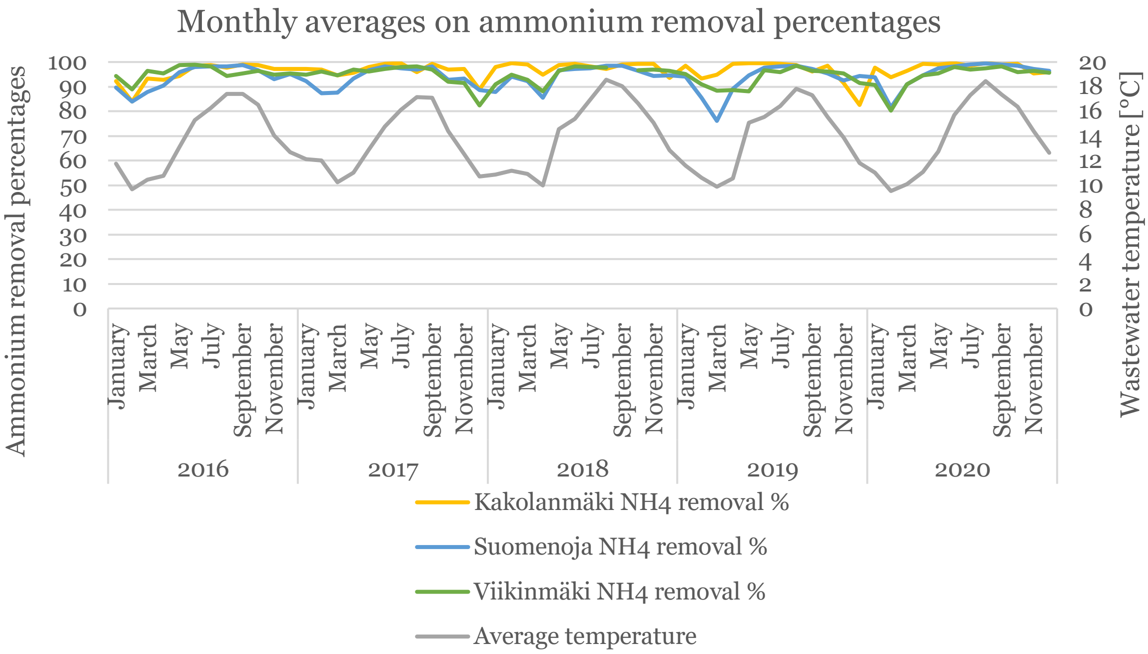

Figure 15. Ammonium removal percentages for the five-year time period as monthly averages for three WWTPs.

Figure 16 illustrates how the incoming nitrogen is related to the nitrogen removal efficiency. The nitrogen removal percentage seems to increase as the incoming nitrogen concentration increases. This can be seen as an upwards trend in all of the WWTPs in *Figure 16*. The Pearson correlation coefficients for incoming nitrogen concentration and nitrogen removal efficiency are 0,65 in Suomenoja, 0,52 in Viikinmäki, and 0,62 in Kakolanmäki.

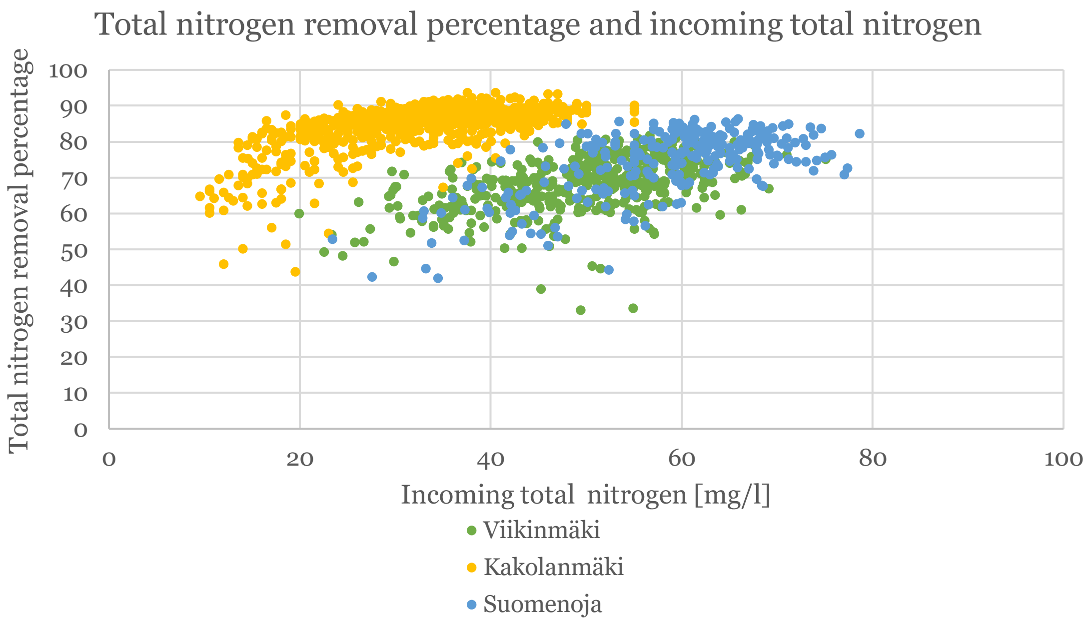

Figure 16. Incoming nitrogen concentration plotted against the nitrogen removal percentage.

Having a positive correlation with incoming nitrogen concentration and slightly negative with incoming nitrogen kilograms is explained by the incoming wastewater flows. When the incoming nitrogen concentration is high, there is likely smaller volume of incoming wastewater. During low flows, a WWTP can have a long SRT and therefore remove total nitrogen efficiently.

To further observe the ammonium and total nitrogen removal efficiencies, table 5 and table 6 are provided. In table 6 we can see the average nitrogen removal percentages based on the five-year data for summer and winter periods in the three WWTPs. In this study, winter period is defined by the arriving wastewater temperature monthly average. If it is 12 °C or below, it is considered to be cold and so winter period. By this definition, on average there was four months of winter period during each year, with Viikinmäki having the shortest winter periods.

Table 5. The average total nitrogen removal percentages for the three WWTPs ASP during winter and summer periods.

	Total nitrogen removed, summer average	Total nitrogen removed, winter average	Share of total nitrogen removal percentage in winter of the summer percentage
Viikinmäki	69%	68%	98%
Suomenoja	75%	64%	86%
Kakolanmäki	86%	78%	91%

As can be seen from table 5, the winter total nitrogen removal percentage is consistently lower than the summer total nitrogen removal percentage. Based on these calculations, on average the winter periods' total nitrogen removal efficiency is 92% of the summer nitrogen removal efficiency, when the winter temperature limit is 12 °C.

In table 6. the ammonium removal percentages can be seen to be consistently lower during winter periods than during summer periods. The ammonium removal percentages are higher than the total nitrogen removal percentages, which means that the ammonium nitrogen is removed more efficiently than total nitrogen and some of the nitrogen escapes all of the ASP as nitrate. The winter ammonium removal efficiency is on average 94% of the summer ammonium removal efficiency.

Table 6. The average ammonium removal percentages for the three WWTPs ASP during winter and summer periods.

	Ammonium removed, summer average	Ammonium removed, winter average	Share of ammonium removal percentage in winter of the summer percentage
Viikinmäki	96 %	89 %	95 %
Suomenoja	97 %	89 %	92 %
Kakolanmäki	98 %	91 %	94 %

Based on tables 5 and 6, Kakolanmäki ASP seems to perform the best in total nitrogen removal and in ammonium removal. Suomenoja ASP seems to have the strongest reaction with both ammonium and total nitrogen removal to the colder wastewater influent temperature, as during winters the efficiencies are experiencing a greater drop than the two other WWTPs. When ASP is struggling to remove ammonium, it means that the nitrification is not working properly. If ammonium removal is efficient but total nitrogen removal is not, it means that the denitrification reaction is not able to perform as good as it should. The analysis of the total nitrogen and ammonium nitrogen removal percentages is not always straightforward, as they are also dependent on how much of the ASP is aerated. To keep the nitrification going despite the decreasing of the inflowing wastewater temperature, a higher aerated volume of the ASP is required. On the contrary this leads to the decrease of anoxic denitrifying volume of the ASP, which leads to increasing total nitrogen emissions. This means, that the increase of total nitrogen emissions might not occur due to problems with the denitrification reaction itself, but rather due to the manner the WWTP is run.

To better understand the differences between the efficiencies of the three WWTPs, the ammonium and total nitrogen load to the ASP is calculated. *Figure 17* and *Figure 18* illustrate these reductions by each cubic meter of the available ASP space. *Figure 17* showcases how Suomenoja removes the most nitrogen per each ASP cubic meter. In *Figure 18* similar order of the three WWTPs can be observed. The temperature does not seem to have clear effects on the total nitrogen and ammonium removed by each ASP m^3. MLSS influences the total nitrogen removed by one cubic meter of ASP greatly, which is why the relationship between total nitrogen reduction and temperature is difficult to observe.

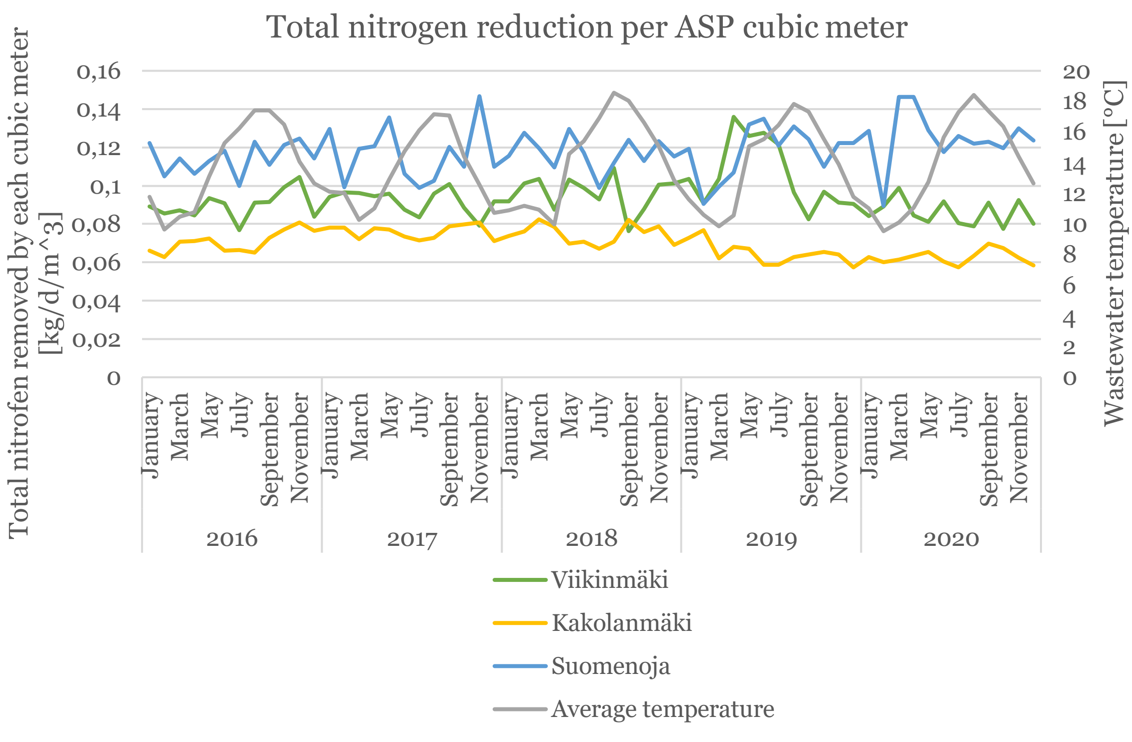

Figure 17. Total nitrogen removed by one cubic meter of ASP in three different Finnish WWTPs.

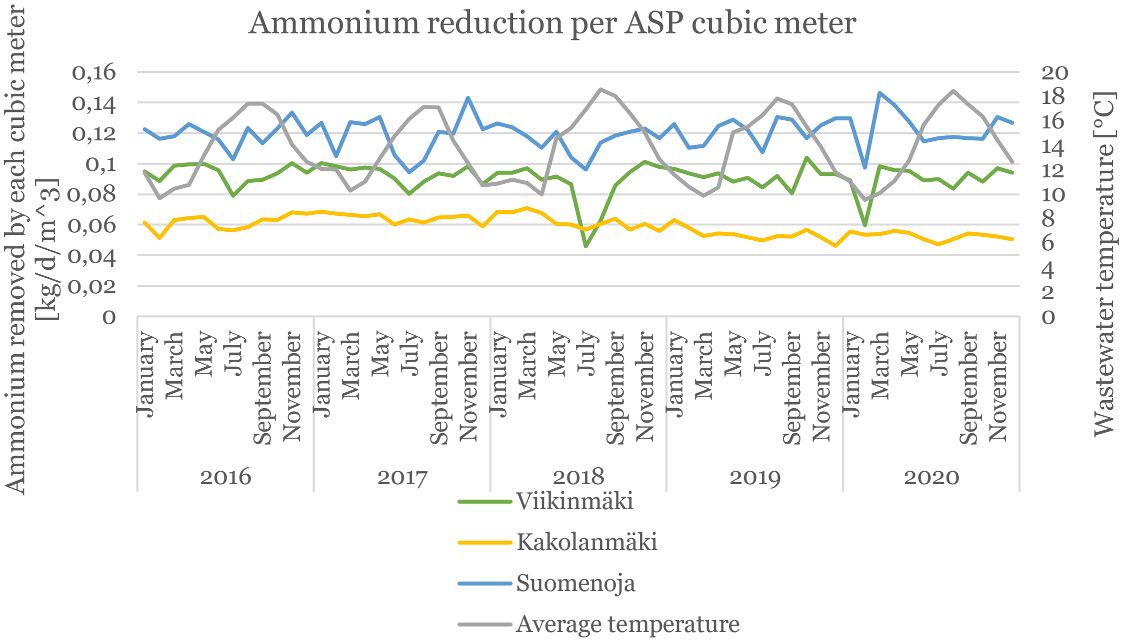

Figure 18. Ammonium removed by one cubic meter of ASP in three different Finnish WWTPs.

To further analyze the relationship between nitrogen removal and temperature, a scatter dot chart is made for each of the WWTPs (see *Figure 19*, *Figure 20*, and *Figure 21*). In the scatter dot chart both nitrogen removal by ASP cubic meter as well as the removal percentage are showcased. In *Figure 19*, the effect of temperature on Kakolanmäki nitrogen removal percentage and the nitrogen reduction per ASP cubic meter is illustrated. The nitrogen removal percentage datapoints seem to represent an increasing trend as the temperature increases, while the nitrogen reduction by each ASP cubic meter seems not to show a strong trend. The linear average fit into the nitrogen reduction datapoints even seems to indicate a decrease as the temperature increases. Kakolanmäki ASP has a Pearson correlation coefficient of 0,47 with nitrogen removal percentage and the wastewater temperature and -0,04 correlation coefficient between temperature and nitrogen reduction per ASP cubic meter. This means that the nitrogen removal percentage and temperature are positively correlated, but the nitrogen reduction per ASP m^3 is not.

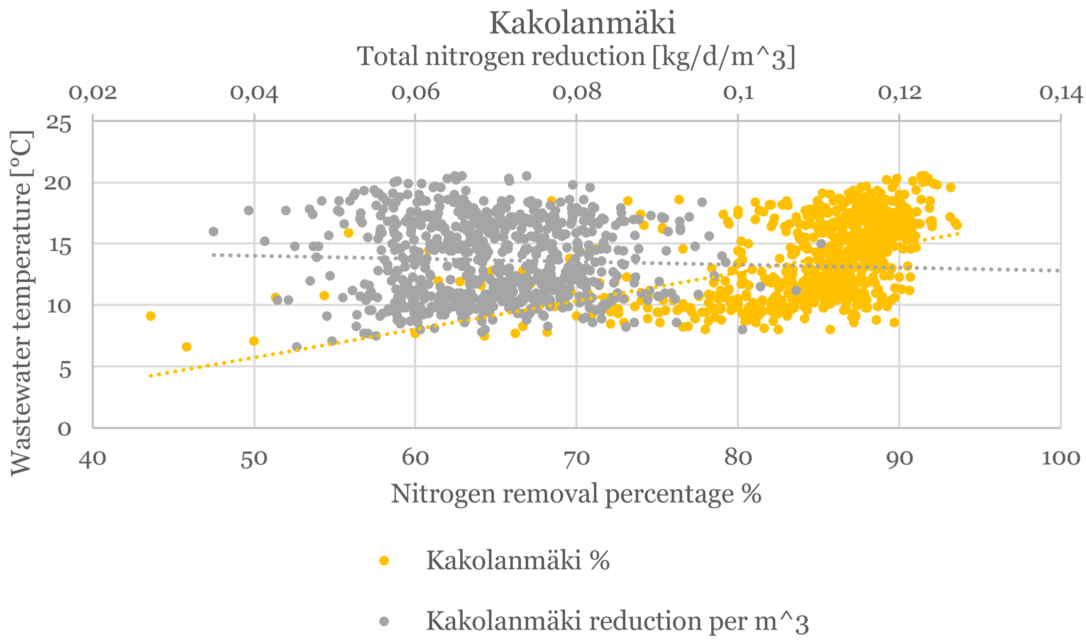

Figure 19. Kakolanmäki ASP nitrogen removal and temperature.

In *Figure 20*, the connection between Viikinmäki ASP nitrogen reduction efficiency and temperature is illustrated. The Pearson correlation coefficients for temperature and nitrogen removal percentage as well as temperature and nitrogen reduction per ASP m^3 do not indicate a strong positive nor negative correlation, as the correlation values are respectively 0,17 and -0,06.

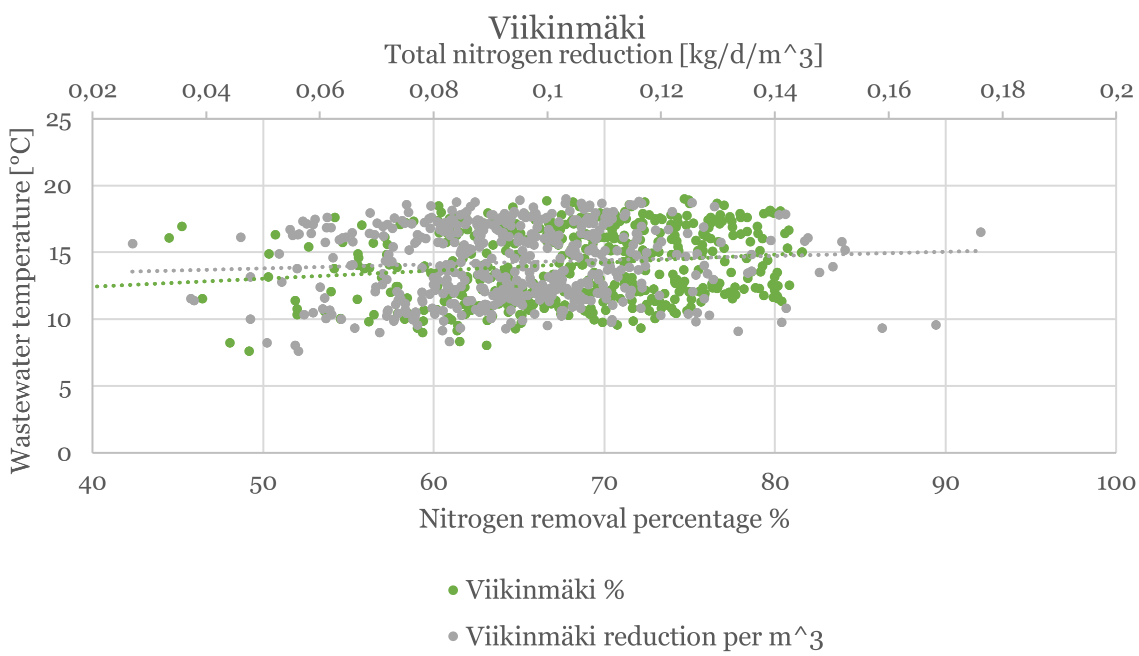

Figure 20. Viikinmäki ASP nitrogen removal and temperature.

Suomenoja ASP nitrogen removal connection with temperature is illustrated in *Figure 21*. Suomenoja seems to exhibit similar trends as Kakolanmäki ASP. The nitrogen removal percentage is positively correlated, but the nitrogen reduction by ASP cubic meter seems to be quite scattered. Suomenoja ASP nitrogen removal percentage and wastewater temperature seem to correlate positively with each other, as they have a Pearson correlation coefficient of 0,67. On the other hand nitrogen removed by Suomenoja ASP m^3 and wastewater temperature have a correlation coefficient of 0,02 which does not indicate positive nor negative linear correlation between these factors.

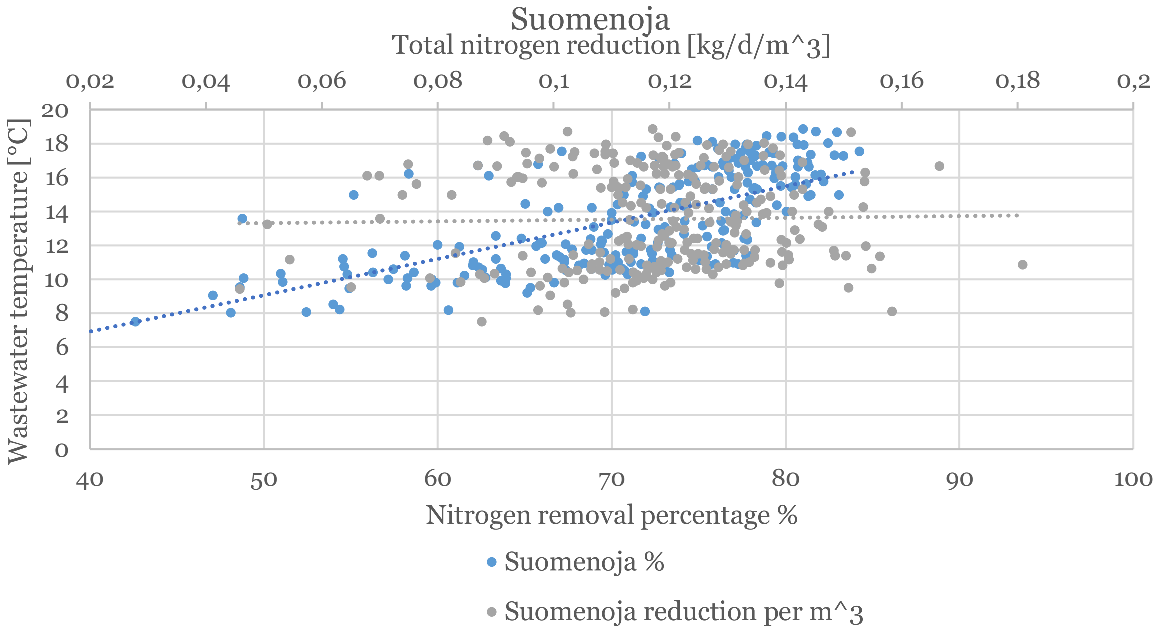

Figure 21. Suomenoja ASP nitrogen removal and temperature.

The fact that temperature does not seem to influence the nitrogen removed by each m^3 as much as it influences the nitrogen removal percentage can be explained by the influence of the incoming wastewater flow and MLSS. When looking at the amount each cubic meter of the ASP has removed nitrogen, the nitrogen is observed as kilograms. The amount of removed kilograms does not necessarily correlate with the nitrogen removal percentage, as the inflowing concentration of nitrogen varies greatly. When the removal percentage is very high, it mostly is due to slow flow and little incoming nitrogen. These sort of conditions also often occur during relatively warm wastewater temperatures. During these times the overall removed nitrogen in kilograms might still be relatively low. During larger flows there are high loads of influent nitrogen, which can still lead to high amounts of nitrogen being removed in kilograms even though the nitrogen removal percentage might be low due to the high arriving amount of nitrogen. The variations in the way the WWTPs react to the changing temperatures of the wastewater might also be due to the loads the WWTPs are experiencing as well as the possible free capacity that these plants have. As a clear relationship between nitrogen removal efficiency and wastewater could not be found in this data-analysis, the need for process simulations is highlighted.

6.1.3 BOD load for the three WWTPs

To better understand the differences between the load of the different WWTPs, the amount of BOD removed by each cubic meter of ASP is examined in *Figure 22*. *Figure 22* illustrates that BOD removal percentages seem to stay relatively high (above 90%) almost all the time during the five-year period, but still some slight drops can be observed during the colder periods of the year.

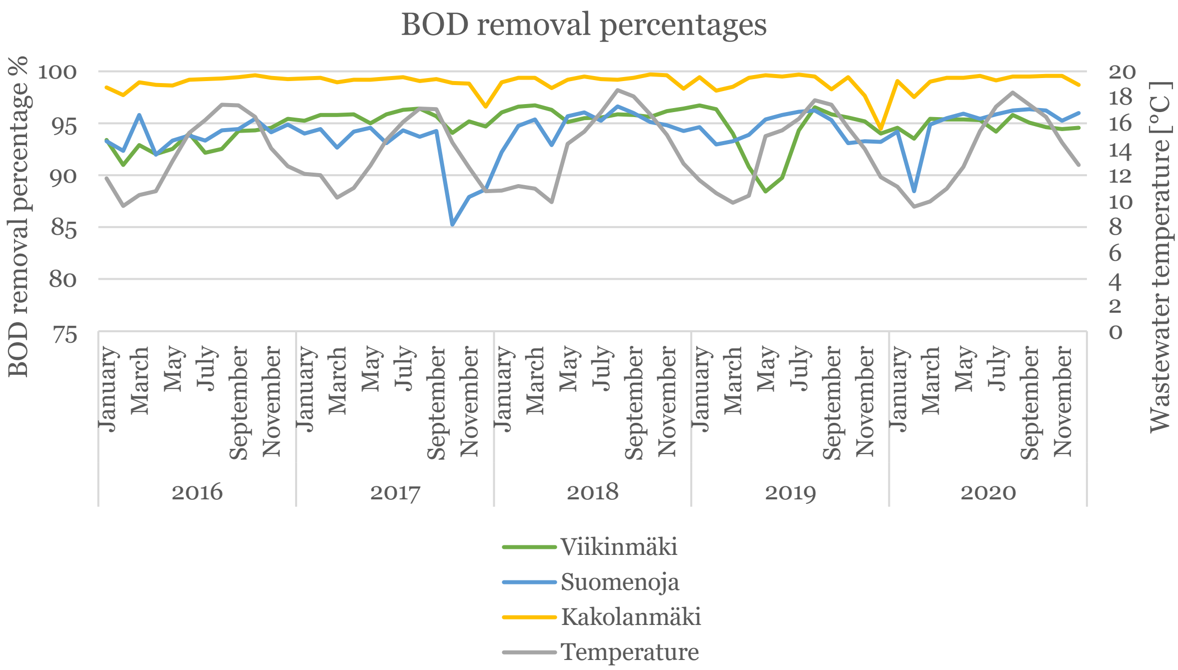

Figure 22. Timeline of monthly BOD removal averages and wastewater temperature.

6.1.4 Aeration and nitrogen load

In this section, aeration air consumption is analyzed against nitrogen removal efficiency. *Figure 23* illustrates how Suomenoja WWTP uses more aeration air per ASP m^3 than Viikinmäki and Kakolanmäki to remove nitrogen. Even though the upward trends can be seen in *Figure 23*, the Pearson correlation coefficient do not indicate a strong linear correlation in Viikinmäki (0,34) and Suomenoja (0,14). The strongest correlation can be seen in Kakolanmäki, where the correlation coefficient between removed nitrogen and used aeration air is 0,58.

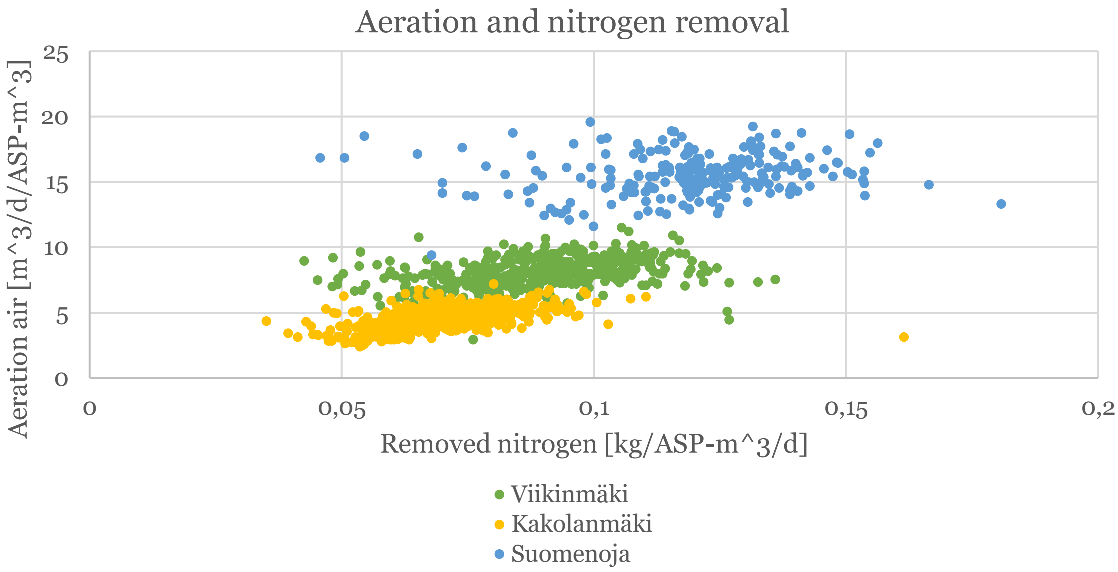

Figure 23. The relationship between volume of air used aeration and amount of total nitrogen removed illustrated in a scatter dot graph.

6.1.5 Temperature and N2O emissions

Viikinmäki WWTP has measured its N2O emissions. The way N2O emissions have reacted to temperature can be seen in *Figure 24*, *Figure 25* and *Figure 26*. *Figure 24* does not seem to indicate a clear relationship between the amount of N2O emissions and wastewater temperature. It still seems, that there is more N2O emissions when the wastewater temperature is high than when its low. In these figures, the N2O emissions are from the ASP, which is why the nitrogen removal values considered are also only from the ASP. This leads to the values presented in this chapter of N2O shares of N2 being smaller when the whole plant is considered.

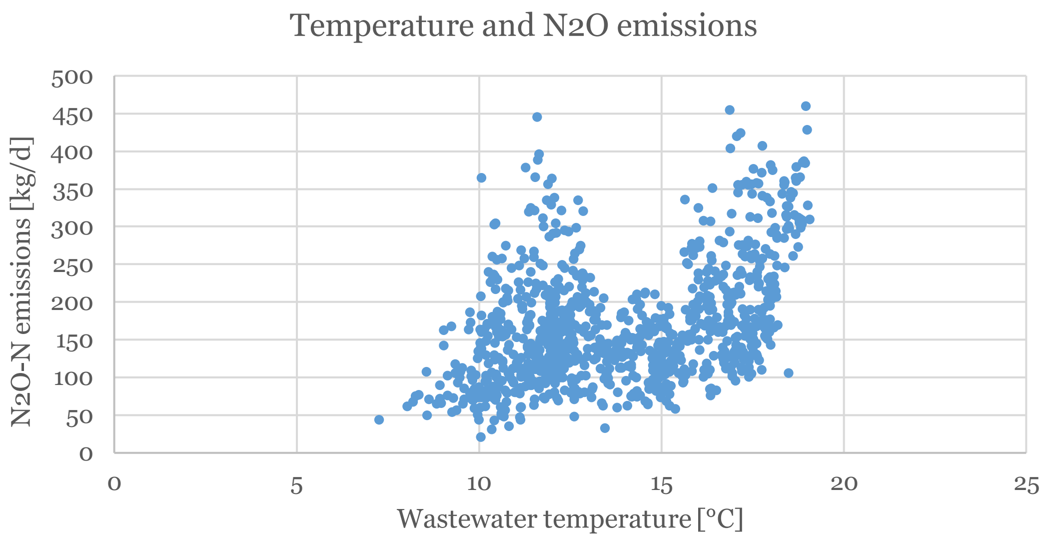

Figure 24. N2O emissions plotted against wastewater temperature.

Figure 25 illustrates that the share of N2O of the total removed nitrogen seems to also rise as the temperature rises. This is interesting, as N2 emissions also increase as the wastewater temperature increases. This means that the growth of N2O emissions is more drastic than the increase in N2 emissions. This observation is also supported by *Figure 26*.

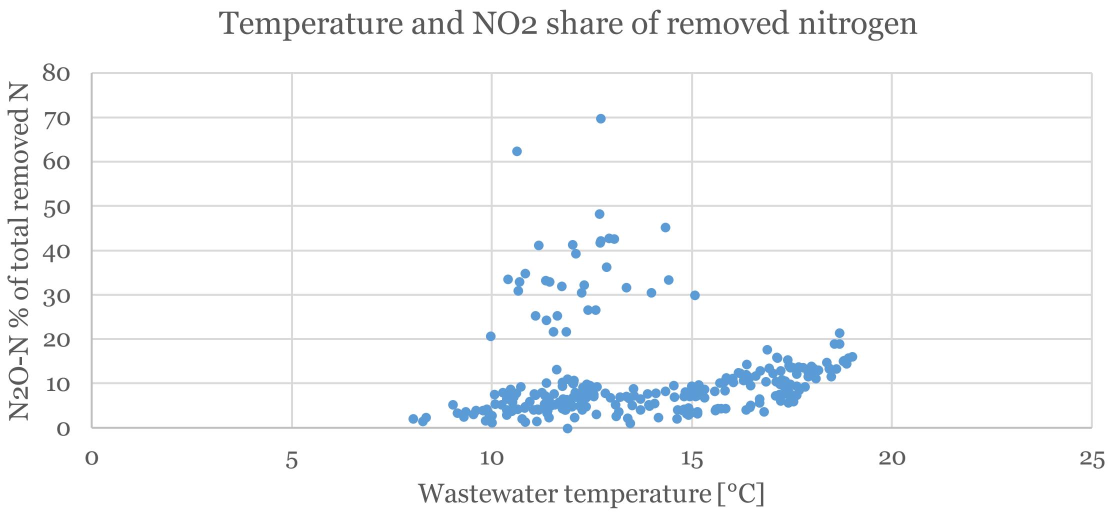

Figure 25. N2O share of removed nitrogen plotted against wastewater temperature.

Even though *Figure 25* and *Figure 26* show a slight upward trend, there is no clear linear correlation between inflowing wastewater temperature and share of N2O of removed nitrogen, and between nitrogen removal percentage and share of N2O of removed nitrogen. *Figure 26* below illustrates how the N2O share of the removed nitrogen seems to rise as the nitrogen removal efficiency of the WWTP increases. This means that as a larger share of the arriving nitrogen is being removed, a larger part of it leaves the plant as N2O instead of N2.

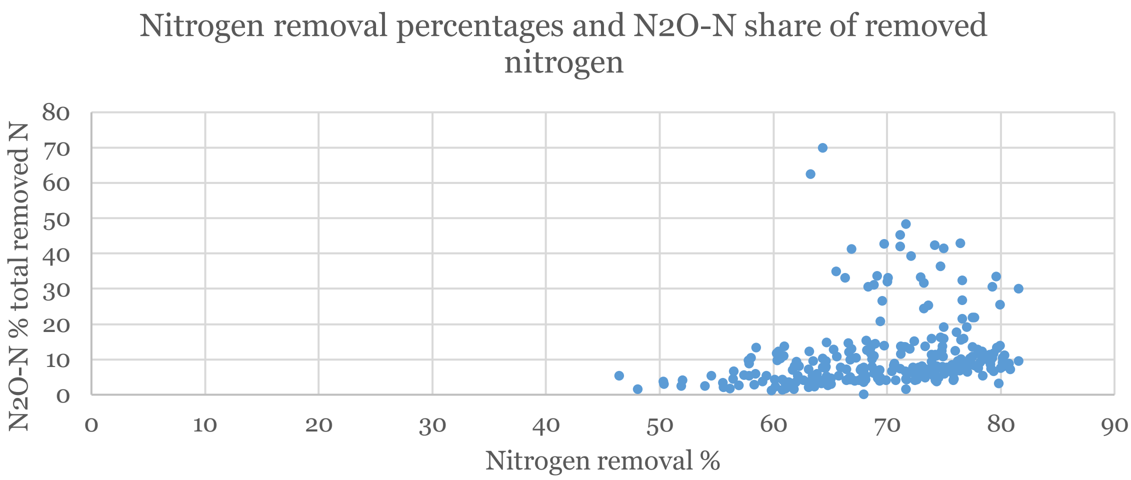

Figure 26. N2O share of removed nitrogen plotted against nitrogen removal percentages.

6.1.6 The use of chemicals

In this chapter, the influence of wastewater temperature on chemical consumption is observed. When observing chemical usage in Viikinmäki, the post denitrification filters are is considered as methanol is used there. The amount of used methanol varies between 0,75 CH3OH kg/removed N kg and 2,75 CH3OH kg/removed N kg in Suomenoja, and 0,5 CH3OH kg/removed N kg and 1,5 CH3OH kg/removed N kg in Viikinmäki. In Suomenoja WWTP, methanol is used in ASP and in Viikinmäki methanol is used in the post denitrification filter. The Pearson correlation coefficients between used amount of methanol and the wastewater temperature does not illustrate a linear relationship in either Viikinmäki WWTP (-0,03) or in Suomenoja WWTP (0,15).

Lime and sodium carbonate are used to control alkalinity in the ASP. Lime is used in Kakolanmäki WWTP and Viikinmäki WWTP. The amount of lime used in the ASP per 1 removed kg of nitrogen varies between 0,1 kg lime/removed N kg and 1,4 kg lime/removed N kg in Kakolanmäki, and in Viikinmäki it varies between 0,3 lime-kg/ removed N-kg and 1,6 kilograms of lime used per removed nitrogen kg. The Pearson correlation coefficients between the wastewater temperature and used lime amount to remove 1 kg of nitrogen are 0,24 for Kakolanmäki and 0,29 for Viikinmäki, which do not illustrate a relationship between temperature and lime consumption.

Sodium carbonate is used in Suomenoja WWTP. *Figure 27* illustrates how much the amount of used sodium carbonate to remove one kg of nitrogen changes throughout the five-year data period. The values vary between 0,5 to 1,7 sodium carbonate-kg/removed N-kg. The Pearson correlation coefficient between wastewater temperature are used sodium carbonate per each removed nitrogen kg is -0,44, which somewhat indicates a negative relationship between these two factors. This can be also seen in *Figure 27* as higher amounts of used sodium carbonate in the cold time periods.

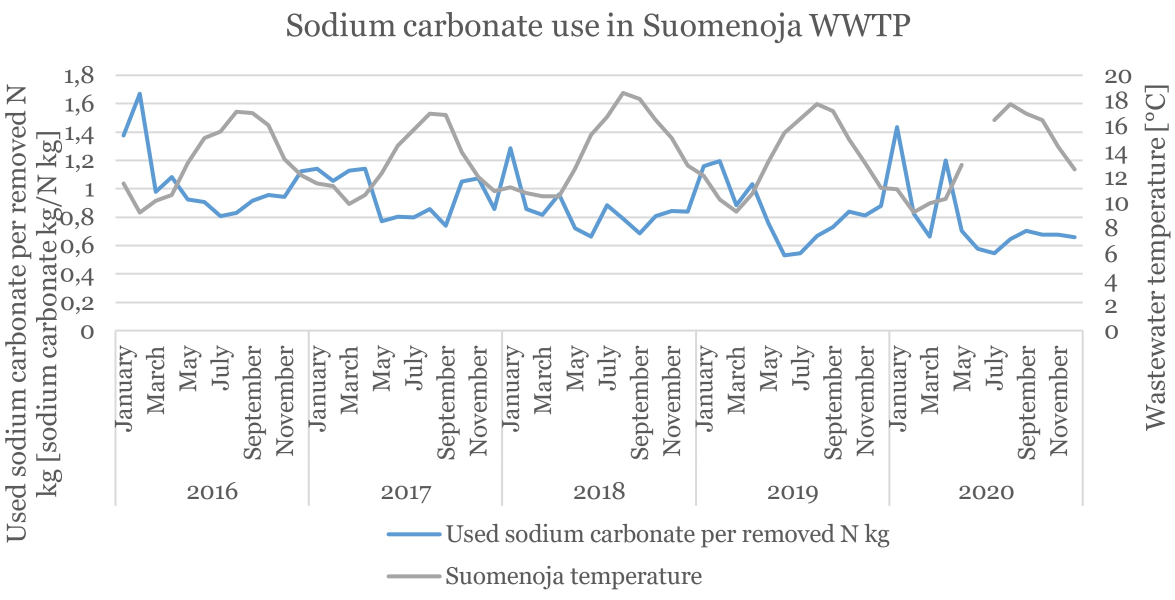

Figure 27. Used amount of sodium carbonate per removed nitrogen kg in Suomenoja WWTP.

Sodium carbonate and lime used in Suomenoja and Viikinmäki are plotted in *Figure 28* with wastewater alkalinity before entering ASP. The alkalinity seems to variate in a similar manner in both Suomenoja and Viikinmäki WWTP, with having the lowest levels on early spring, but otherwise staying quite even throughout the year. There seems to be less sodium carbonate used in the summer times than the rest of the year. The lime usage seems to vary greatly with no clear connection to alkalinity in Viikinmäki.

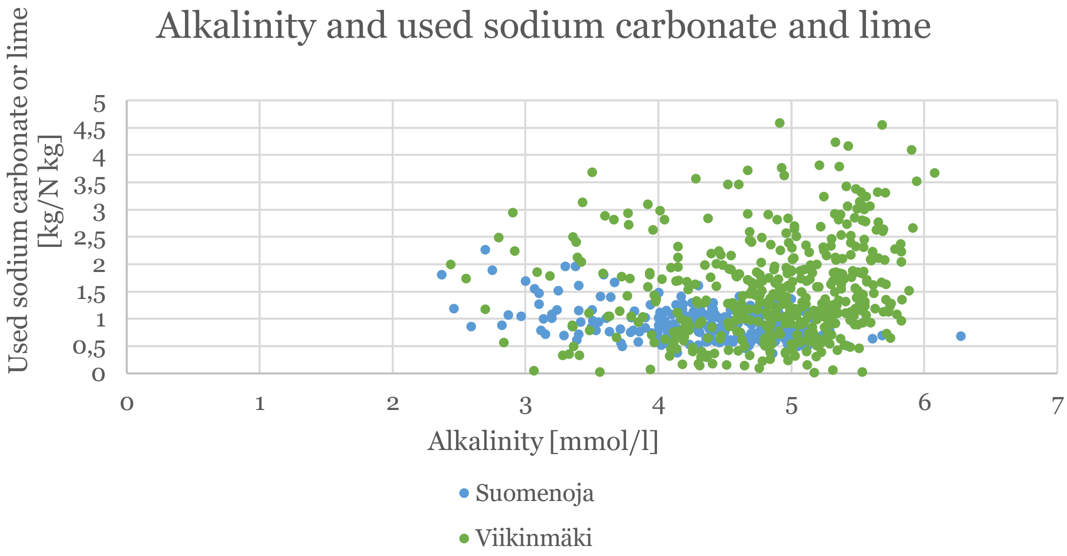

Figure 28. Alkalinity and used sodium carbonate and lime per removed kg of nitrogen in Suomenoja and Viikinmäki WWTP.

Limitations in alkalinity could be affecting the functioning of ASP, and so create uncertainties in the results. It does not seem likely that the process has been significantly limited by alkalinity, as the alkalinity leaving the WWTPs was not below 1 mmol/l in Suomenoja and Viikinmäki WWTP. If the alkalinity is too low, the ASP process would be affected by this. This is of interest, as the inflowing alkalinity is often very low in Finnish conditions.

Polymers are only used in the ASP process Kakolanmäki WWTP. Its usage has varied between 0,025 and 0,08 polymer-kg/removed N-kg. There does not seem to be significant linear correlation between wastewater temperature and the used amount of polymer per each removed nitrogen kg, as the Pearson correlation coefficient is -0,19.

6.2 Process simulation results

6.2.1 Effect of process temperature on nitrogen removal

This section presents the results of how the temperature influences the nitrogen removal efficiency when sludge age and process volume are constant. The simulated time period was 24 days. *Figure 29* showcases how the nitrogen load and nitrogen removal percentage change when wastewater temperature decreases from 17 °C to

5 °C. In 17 °C, the total nitrogen load into receiving waterbodies is 721 kg/d and 85% of total nitrogen is removed from wastewater. In 5°C 1218 kg/d of total nitrogen is leaving the WWTP, while 75% of total nitrogen is removed from the wastewater. As temperature decreases from 17 °C to 5 °C almost 500 additional nitrogen kilograms per day is leaves the ASP. This amount of nitrogen equals 33 133 additional PE of nitrogen in the treated wastewater. Here the additional nitrogen population equivalent was calculated by assuming that one person produces 15 g of nitrogen each day.

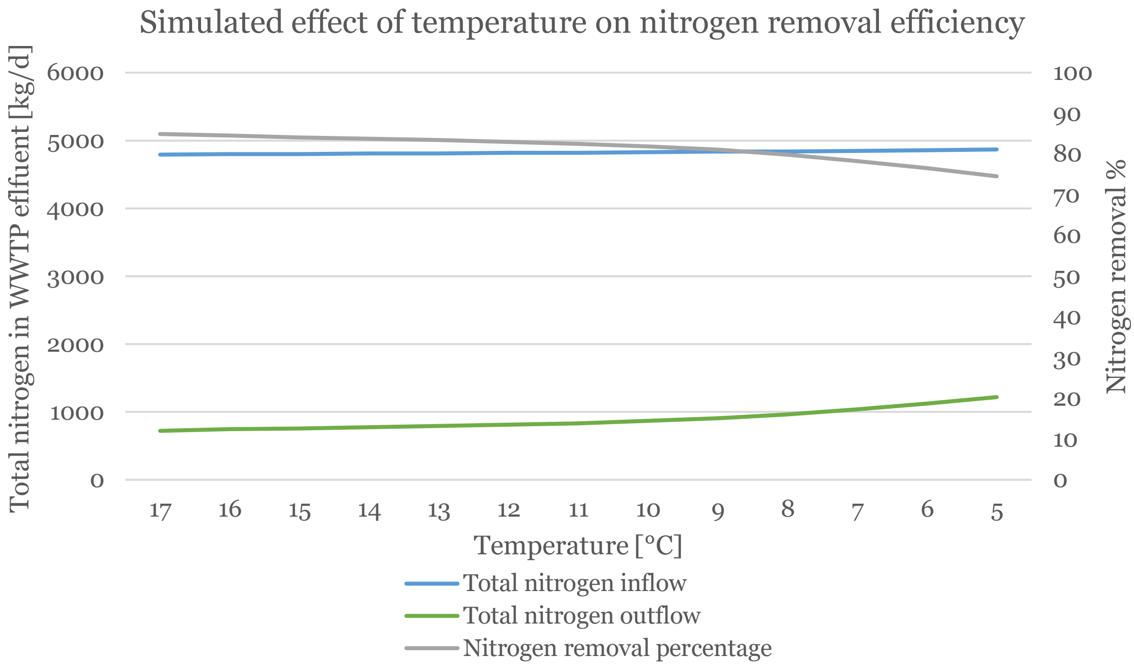

Figure 29. Simulated effect of decreasing inflowing wastewater temperature on total nitrogen load in WWTP effluent.

Figure 30 illustrates how the starting wastewater temperature influences the effect of decreasing wastewater temperature. For example, a decrease of 1 °C in 7 °C has greater effect on the nitrogen removal, than a similar decrease of 1 °C in 17 °C. The additional total nitrogen can be seen to increase as the temperature decreases. The *Figure 30* illustrates that when the temperature decreases especially below 10 °C, the additional total nitrogen concentration starts to increase greatly. Between 16 °C and 11 °C the additional nitrogen per each 1 °C decrease of temperature adds less than 0,3 mg/l, but when the temperature decreases from 11 °C, the concentration of additional total nitrogen raises quickly up 1 mg/l in 6 °C. In 5 °C, the additional total nitrogen concentration compared to 6 °C is already 1,2 mg/l on top of the 14 mg/l nitrogen already leaving the WWTP. Above 11 °C the average total nitrogen concentration therefore stays below 11 mg/l, but after that the total nitrogen concentration leaving the WWTP steadily increases to 15 mg/l in 5 °C.

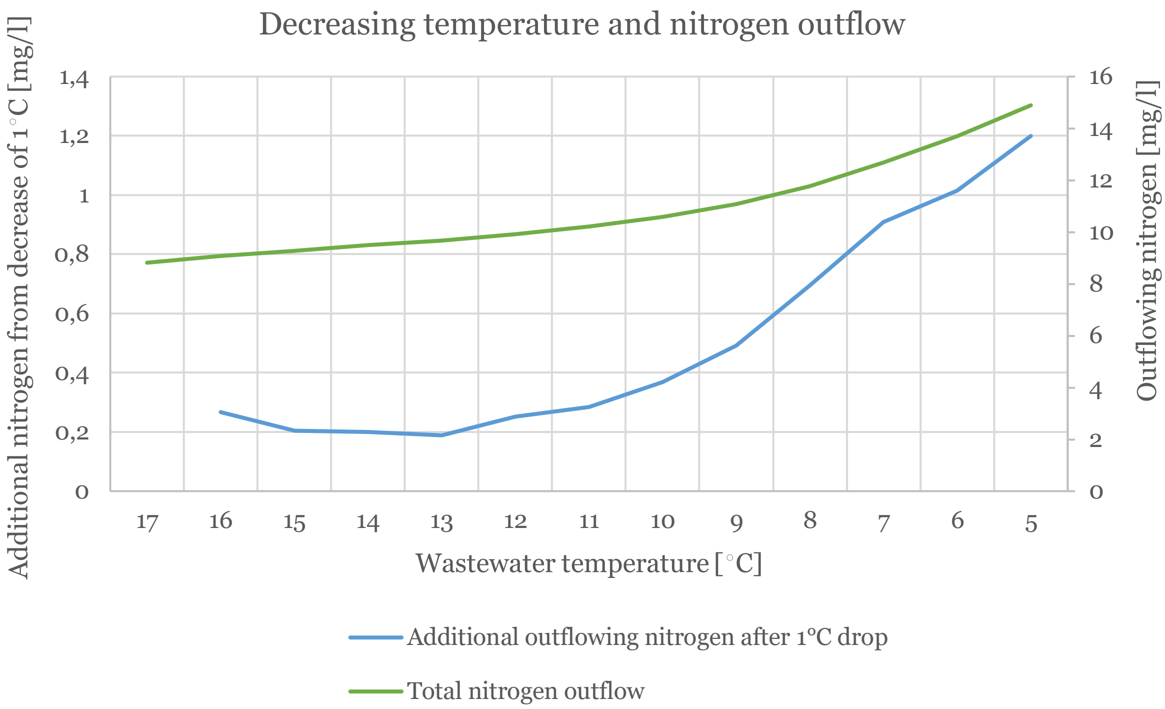

Figure 30. The simulated effect of 1°C drop in temperature in on the outflowing total nitrogen concentration in different temperatures.

As the temperature decreases 1 °C when the temperature is above 10 °C, the effect of the decrease is between additional 1000 and 2000 nitrogen PE, meaning the total nitrogen of untreated wastewater of up to 2000 people ends up in the effluent of the WWTP. This translates to additional 20 to 30 nitrogen kg per each decrease of 1 °C. Under the temperature of 10 °C the additional total nitrogen load starts to increase more dramatically. When the temperature drops from 6 °C to 5 °C, the additional nitrogen load on top of the already great load is 6540 nitrogen population equivalents, which equals an additional load of 100 kg/d of nitrogen, into the recipient water body.

6.2.2 Effect of process temperature on nitrogen balance

The total nitrogen balance of the simulated WWTP is illustrated as Sankey diagrams in 15 °C, 10 °C, and 5 °C. The diagrams can be seen in appendices 1, 2, and 3. These scenarios are produced with all the factors other than wastewater temperature being constant. As seen from the figures, the nitrogen inflow in all these scenarios is the same, so 7100 kg/d, but already by the time the wastewater arrives to the ASP differences can be seen in the nitrogen flows between the temperatures. Biggest difference in the nitrogen balance can be seen in the outflowing total nitrogen. Amount of total outflowing nitrogen in 15 °C is 920 kg/d, in 10 °C 970 kg/d, and in 5 °C 1320 kg/d. The total nitrogen released in gaseous form is 5270 kg/d when the wastewater temperature is 15 °C, 5240 kg/d when the wastewater temperature is 10 °C, and 4940 kg/d when the wastewater temperature is 5 °C.

6.2.3 Management of the effects of temperature on nitrogen removal

The effect of decreasing temperature could be compensated by increasing the ASP volume. This chapter answers the second research question, of what is the process volume needed to avoid an increase in the nitrogen load to the recipient water body as the temperature of the ASP decreases. The way increasing the ASP size can be seen in *Figure 31*. In this simulated WWTP, 721 kg of nitrogen escapes the WWTP in 17 °C. To achieve the same level of nitrogen removal as the ASP does in 17 °C, requires about 7,5% more process volume with every Celsius grade the temperature decreases. This means that according to the simulations, the needed ASP volume doubles as the temperature decreases from 17 °C to 10 °C. As one ASP line is 15350 m^3 in this particular simulated WWTP, would a decrease of 2,5 °C require a new ASP line to keep the nitrogen load from increasing. The lower the temperature is, the more decrease of one Celsius influences the needed ASP volume.

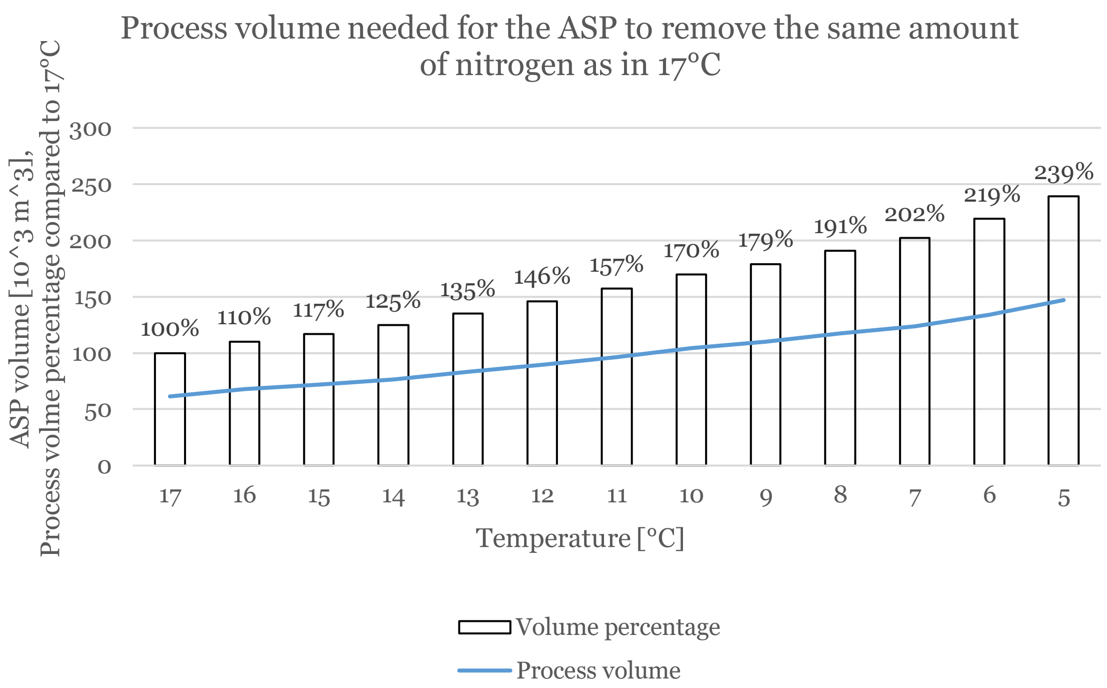

Figure 31. Simulated results of the needed process size for the nitrogen load into receiving waterbodies not to grow.

In *Figure 32*, the influence increasing the process volume has on MLSS and SRT is shown. MLSS stays around 4200 mg/l trough out the temperature decrease. SRT increases similarly to the process volume. SRT increases from 15 days in 16 °C to 33 days in 5 °C.

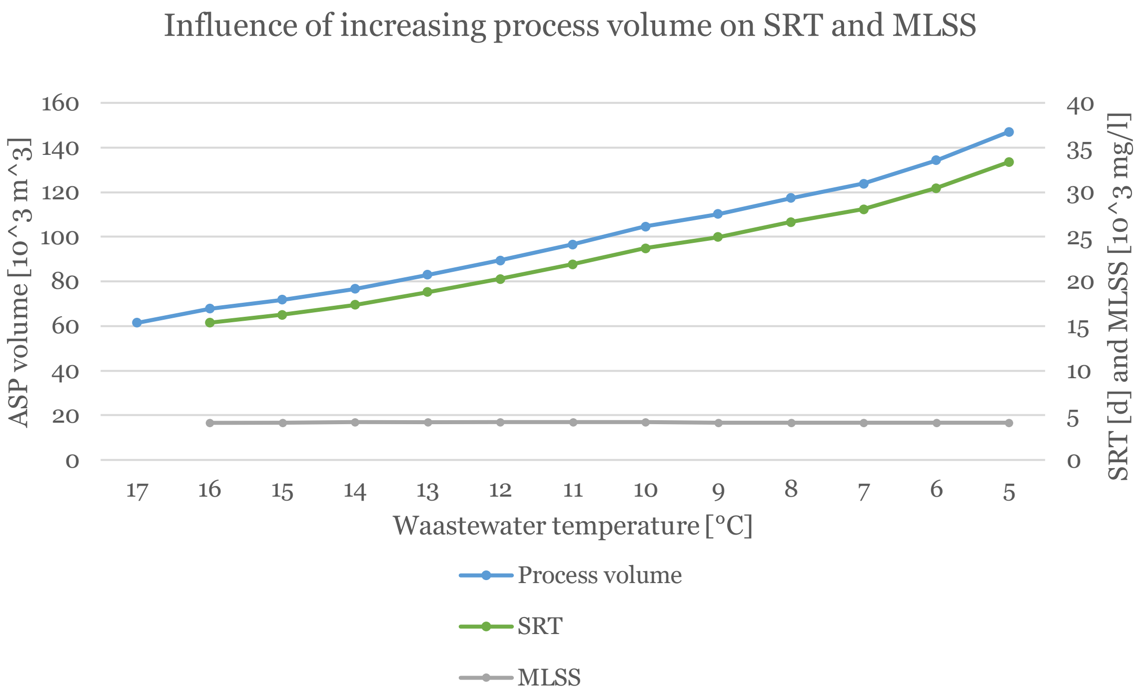

Figure 32. Influence of increasing process volume on SRT and MLSS.

6.2.4 Sensitivity analysis

To further assess the reliability of the model used in this thesis, a sensitivity analysis was done. The sensitivity analysis was done by testing various temperature coefficient values for autotrophic maximum growth rate. The value used in the process simulations was 1,111, and it had been calibrated for temperatures from 16 °C to 18 °C (Pöyry Finland Oy, 2016). A temperature coefficient example value found in the literature is 1,123 (Henze, et al., 2008). Simulations were done in different temperatures with different maximum nitrification rate temperature coefficients. Maximum nitrification rate temperature coefficient was the chosen attribute to analyze the sensitivity of, as it has the largest influence in nitrification efficiency. The chosen analyzed temperature coefficients are +/- 0,5% of the value used in the model (1,105 and 1,116), as well as +/- 1% of the value used in the model (1,1 and 1,122). The value 1,123 that was found in the literature was also used.

The results of the analysis are presented in Figure 33. As can be seen from Figure 33, the nitrogen removal percentage stays relatively coherent between when the wastewater temperature decreases form 17 °C to 10 °C. As the wastewater temperature decreases below 10 °C, the effect of the temperature coefficient starts to grow. In 7 °C, 1% variation in the temperature coefficient creates a difference of 3% to the result. In 5 °C the effect of 1% increase in the temperature coefficient increases the nitrogen removal efficiency by 4%, while the decrease of 1% reduces the result by 6%. In 17 °C, all of the results fit in a range of 1% and in 10 °C in the range of 2%.

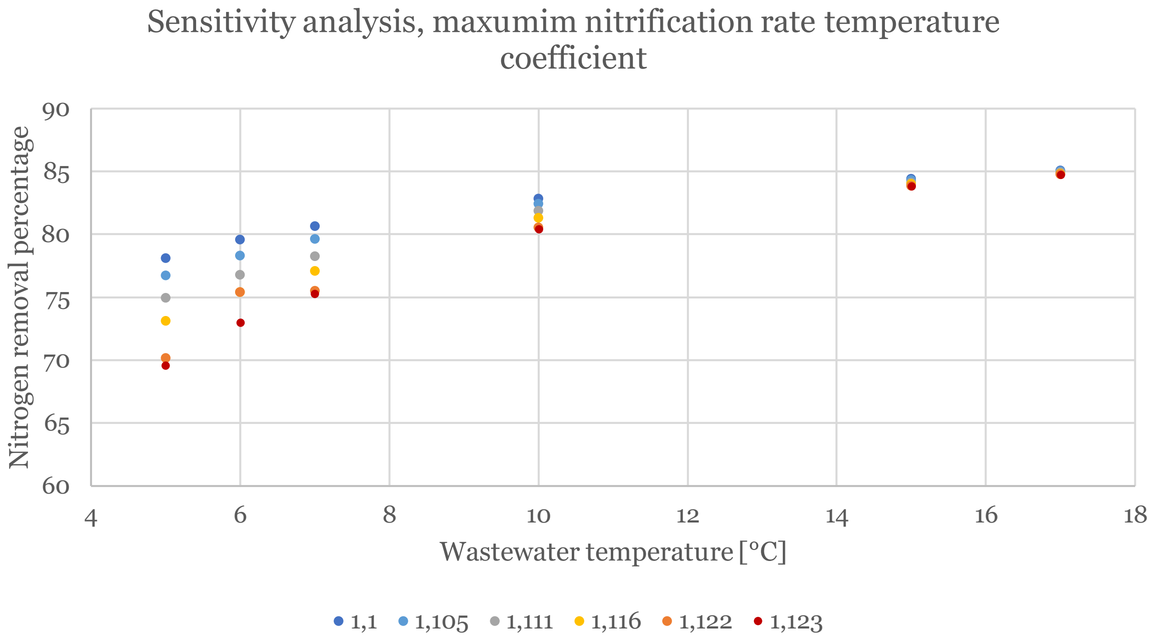

Figure 33. Sensitivity analysis on temperature coefficient for maximum nitrification rate.

The sensitivity analysis illustrates, that the influence of decreasing temperature on the simulated nitrogen removal efficiency might be partly due to the chosen maximum nitrification rate temperature coefficient. Therefore, there is a definite need for more research on temperature coefficients in cold temperatures.

7 Discussion

7.1 Data analysis

Data analysis illustrates how the actual way WWTPs work can differ from the presumed knowledge based on literature. The five-year data collection period was sufficient in showcasing some trends and correlations in a relatively confident manner. The rainfall and flow of incoming wastewater and the quality of it as well as the manner in which the WWTP is run will influence the functioning of the ASP as well as nitrogen removal capability of the WWTP.

Based on the literature reviewed, it was clear that high flows of cold water, that often occur during springtime, pose a risk for ASP process functions (Tchobanoglous, et al., 2003). This was also observed in the data analysis results. The data analysis also supported the common view in literature, that nitrogen removal is less efficient during cold conditions compared to warm conditions (Henze, et al., 1987). Some research has been able to provide a clear relationship between nitrogen removal and wastewater temperature (Wanner, et al., 2005), but such could not be observed in the data analysis, as the observed relationship was fairly scattered (see *Figure 19*, *Figure 20* and *Figure 21*). This is why the process simulations are needed in this thesis.

In literature, the relationship between temperature and used sludge age has been presented to be straightforward. A decrease of 5 °C is theorized to almost double the required sludge age (Laitinen, et al., 2014) as well as 1 °C decrease causing 10% longer aerobic sludge retention times (Wanner, et al., 2005). Data-analysis show no clear relationship between wastewater temperature and sludge age (see *Figure 13*). This contrast between literature values and data analysis results could be due to the values in literature being the minimum values for nitrification, but in real life the plants are run with higher sludge ages to ensure the process does not fail after slight decrease in the wastewater temperatures. The sludge age being stagnant through varying temperatures could also be explained by the use of the switch zone. As during winters, by turning the switch zone on, the aerobic sludge age increases even though the total sludge age might not change.

The literature reviewed here offered few varying theories, of the way temperature influences aeration air volume. Frijns et al. (2013) state that as less energy is used for gas transfer in low temperatures, this could mean that less aeration air is needed in cold conditions. On the contrary, the way WWTPs are operated often include an increased aerated process volume during cold periods, which is why an increase in aeration air volume could also be expected. In the end, aeration air volume is more depended on the incoming nitrogen load than the wastewater temperature, but the effect of wastewater temperature on this could not really be deduced neither based on the literature nor the data analysis. This is why the effect wastewater temperature has on operational finances of WWTPs trough aeration air could not be deduced bast on this data analysis.

As lime and sodium carbonate usage were theorized to have a relationship with wastewater temperature, as they can be used to recover alkalinity after nitrification process if denitrification process volume is decreased due prioritization of nitrification during cold periods (Tchobanoglous, et al., 2003). The data analysis results supported the literature. Especially sodium carbonate use seemed reflect the trends expected based on literature. Alkalinity drops could also be observed during winter times, which also supports the literature revied. The need for more alkalinity chemicals during cold time periods, would then also add to the operational costs of the WWTPs.

The literature reviewed suggested, that the release of N2O could be linked to the volume of used aeration. This is due to aeration air conveying N2O from the liquid phase to gas phase (Law, et al., 2012). On the other hand, as the wastewater temperature decreases, gas transfer requires less energy (Frijns, et al., 2013), which is why less air might be needed in low temperatures. Based on the data analysis, a clear relationship could not be observed between aeration air volume and wastewater temperature, but it seems that there are slightly more N2O emissions during high wastewater temperatures than during low temperatures. The share of N2O emissions of the total removed nitrogen also seems to rise as the wastewater temperature increases. This means that the growth of N2O emissions is more drastic than the increase in N2 emissions. It was also noted that as a larger share of the arriving nitrogen is being removed, a larger part of it leaves the plant as N2O instead of N2.

7.2 Process simulations

The process simulations were successful in illustrating, that any decrease in the wastewater temperature has negative effects on the functioning of nitrogen removal efficiency of the WWTP, even though this temperature decrease would occur in temperatures that are considered warm (>12 °C). The simulation results show that as the process temperature is 17 °C, 85% of total nitrogen is removed, and when the wastewater temperature is 5 °C is 75% of total nitrogen is removed. The data analysis seemed to support this result, as the temperatures below 12 °C resulted in about 10% lower nitrogen removal efficiency than during times when wastewater temperature is above 12°C, as is shown in table 5. Some research had proposed a 10% drop in nitrification efficiency when the process temperature decreases 1 °C (Wanner, et al., 2005), and some that for every 5 °C decrease, nitrogen removal efficiency would decrease almost to half (Laitinen, et al., 2014). The process simulations show that 1 °C decrease only leads to at most 2% decrease in the nitrogen removal efficiency, depending on the initial wastewater temperature. It should be remembered, that based on the conducted sensitivity analysis the influence of the used maximum nitrification rate temperature coefficients is great especially during cold conditions.

The influence of the decrease of 1 °C in the wastewater temperature intensifies as the temperature decreases in the process simulations. This effect can be seen especially under 10 °C. It seems that the literature reviewed supports this. For example, Laitinen et al. (2014) state a similar strong increase in nitrification capacity when the temperature decreases below 12 °C.

The simulation result of how much the ASP volume needs to increase in order for the nitrogen removal not to be aggravated illustrate that in relation to 17 °C, the simulated WWTP would need to increase the ASP size by 25% wen the process temperature has decreased by 2 °C. In this particular simulated WWTP, increasing the process size by 25% would mean one additional ASP line. When the process temperature has decreased down to 7 °C, the ASP volume should grow by 100%. These results and their applicability are very case depended, since often in real life there could be some capacity left in the ASP so that it could be utilized more efficiently rather than increasing the process size right away. Wanner et al. (2005) state that decrease of 1 °C would require 10% more aerobic process volume. In the process simulations made in this thesis, a growth between 5% and 10% was seen in the total process volume every time the wastewater temperature decreased 1 °C, which fit in quite well with the literature values. The difference in the results stem from the fact that literature values were given for aerobic volume, and the process simulations only observe the total process volume.

Based on the information received from the needed growth of ASP volume to counteract on the loss of nitrogen removal efficiency inflicted by decreasing temperature, the needed investments could be estimated for individual cases. In many cases, the extension of WWTP might not be possible, which leads to the need of a new WWTP in another location. The required financial investments depend on the feasibility of the extension of the existing WWTP. Building new WWTPs as well as extending current ones would create significant amount of GHGs, which also vary greatly between different cases.

7.3 Uncertainties in data analysis and process simulations

Connecting wastewater heat recovery into the ASP in WWTPs is quite complicated through data analysis. This is due to all the wastewater components, such as stormwater and industrial wastewater, that arrive to the sewer system on top of municipal wastewater. The share of actual municipal wastewater varies greatly according to the location and time of the year. If wastewater heat is recovered at the source close to municipal buildings, it does not have an unambiguous effect on the process temperature in WWTPs. The wastewater might also include varying amount of substances from industrial origins, that have effects on the ASP, e.g. glycol. The shares and dynamics of these substance flows might not be known to the WWTPs. The data received can be estimated to be reliable, but the sampling tactics and humane errors are always possible, when such long data sampling periods are considered.

It should be remembered, that even though this data analysis and literature review can be seen to represent somewhat confidently the dynamics of how wastewater temperature influences nitrogen removal, the results represent the Finnish conditions, or WWTPs in similar conditions. As the model was built to describe one WWTP, the comparability of it to the other WWTPs can be seen as uncertain. The used model reacts to the changes in ammonium content of the leaving wastewater with changes in the aeration, which optimizes the nitrogen removal. This replicates the way the ASP is driven in real life at the WWTPs, which makes the results of the simulations appear more reliable. The model seemed to cope with the decreasing temperature quite well, and one reason for this could be that the plant this model was built after has good nitrogen removal potential. This idea is also supported by the data analysis, which illustrated that all the WWTPs can continue to remove nitrogen relatively well even in cold conditions.

The uncertainties relating to the simulations include the used wide temperature range of the simulation. ASM2d is not recommended to be used under 10 °C (Henze, et al., 1999). Also, the model was calibrated for relatively narrow temperature range, the accuracy of if during very low temperatures can create inaccuracies. The calibration of the model was performed dynamically on a detailed set of data which included rain events. This makes the calibration of the model good quality. The model has not been validated for as cold temperatures which were used in these simulations. ASM3 model has been successfully validated for under 10 °C (Pöyry Environment Oy, 2008) which creates confidence in that ASM2d would also be able to be validated in temperatures under 10 °C.

The chosen values and mathematical descriptions also have an influence of the end result which can create differences between different approaches. The values used for the temperature dependency fit the literature values well, as the literature provides a large scale of different values (see tables 3 and 4). The influence of temperature coefficient of maximum nitrification rate on the nitrogen removal was analyzed in the sensitivity analysis section. The temperature coefficient had a large effect in the results especially in cold temperatures (see *Figure 33*). This means, that the magnitude of the influence of wastewater temperature on total nitrogen removal can vary from the results received in this thesis. However, the influence of the decreasing wastewater temperature on the nitrogen removal can be assumed to reduce the total nitrogen removal efficiency, as such trend was observed with all the temperature coefficients in *Figure 33*. The temperature coefficients are a significant uncertainty in this thesis. More research needs to be done on this matter.

8 Conclusions

In summary, this thesis studied the effects of decentralized wastewater heat recovery in Finnish conditions on the function of ASP in WWTPs. The efficiency of nitrogen removal in ASP decreases as the process temperature decreases which leads to rising nitrogen emissions to waterbodies.

This study was conducted through literature review, data-analysis on three Finnish WWTPs, and process simulations. The data analysis supported the process simulation results and also provided an additional point of view into the intricacy of observing the impacts of decreasing temperature in the real world. The process simulations were successful in illustrating that any decrease in the wastewater temperature has negative effects on the functioning of nitrogen removal efficiency of the WWTP, even though this temperature decrease would occur in temperatures that are considered warm (>12 °C). The simulation results show that as the process temperature is 17 °C, 85% of total nitrogen is removed, and when the wastewater temperature is 5 °C 75% of total nitrogen is removed.

The simulation result of how much the ASP volume needs to increase in order for the nitrogen removal not to be aggravated illustrate that in relation to 17 °C, the simulated WWTP would need to increase the ASP size by 25% when the process temperature has decreased by 2 °C. When the process temperature has decreased down to 7 °C, the ASP volume should grow by 100%. This level of increase in ASP volume can be assumed to be in many cases almost impossible to achieve due to WWTPs having financial and spatial limitations.

This thesis can be seen to increase the potential of finding a sustainable way to implement wastewater heat recovery in Finnish conditions, without compromising nitrogen removal in WWTPs. These results help to assess the potential of wastewater heat recovery in Finnish conditions and illustrate the needed considerations to implement wastewater heat recovery in an environmentally sustainable manner. Further research opportunities include the more detailed studies of possible management efforts of decreasing temperature and their financial and environmental aspects, innovative ways to utilize batteries in wastewater heat recovery to avoid great effects in WWTPs, as well as research on nitrification rate temperature coefficients in cold conditions. Finally, reliable, and easily comprehensible information should be produced to be utilized in city level decision making concerning the implementation of sustainable energy options.

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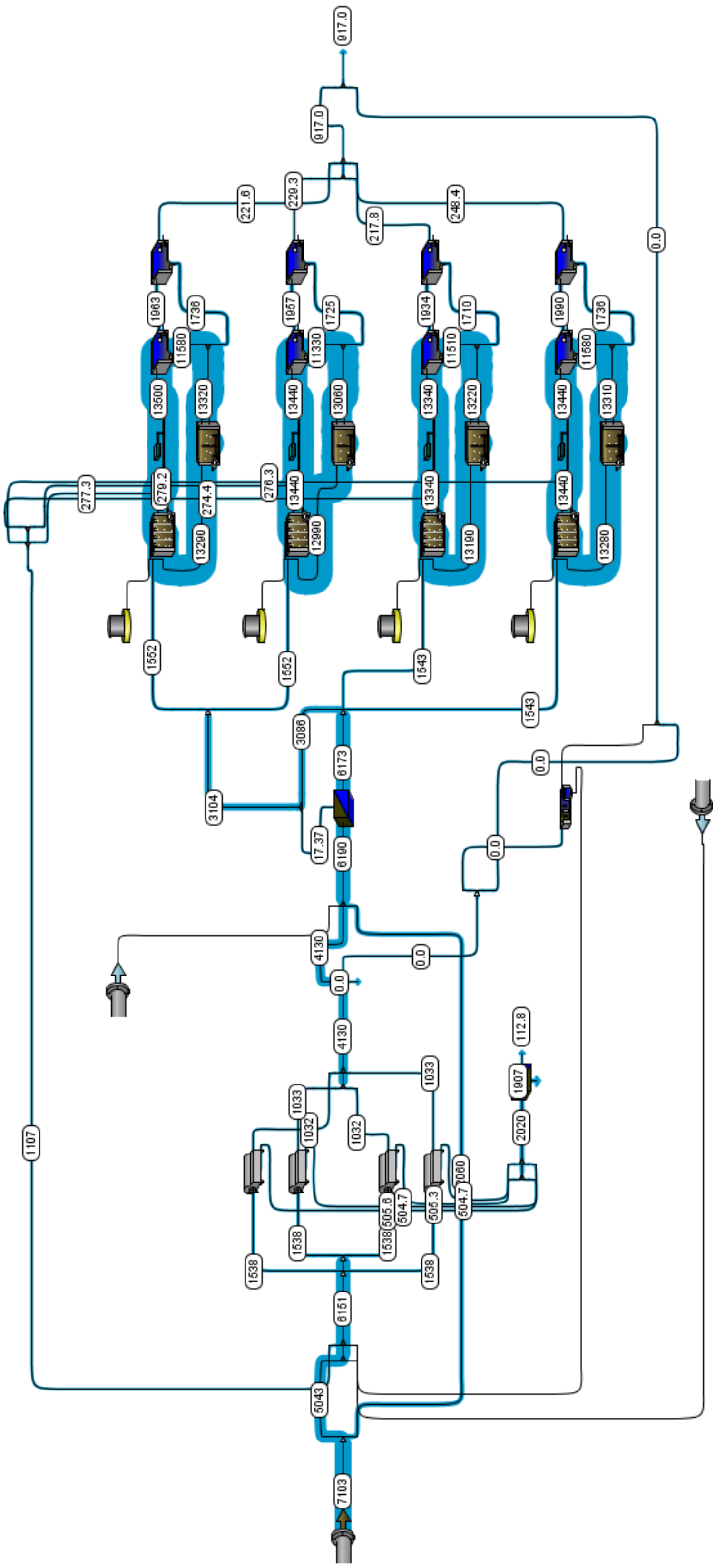
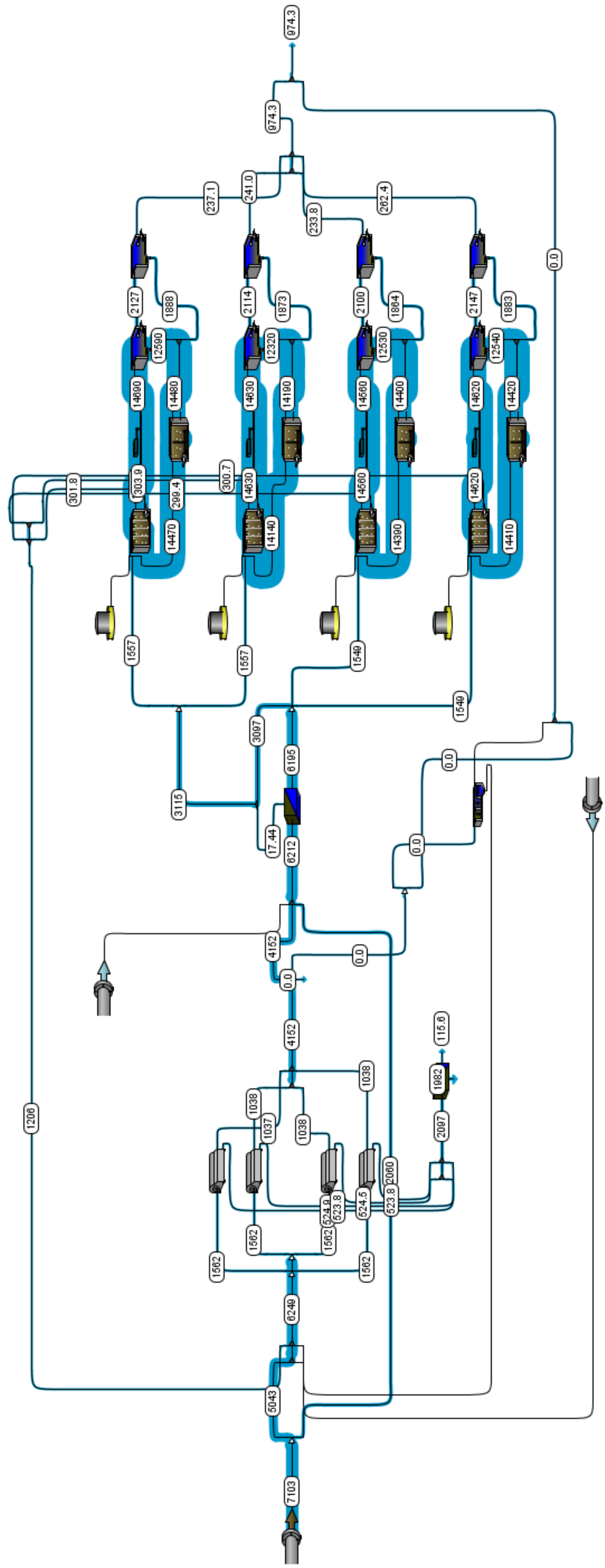
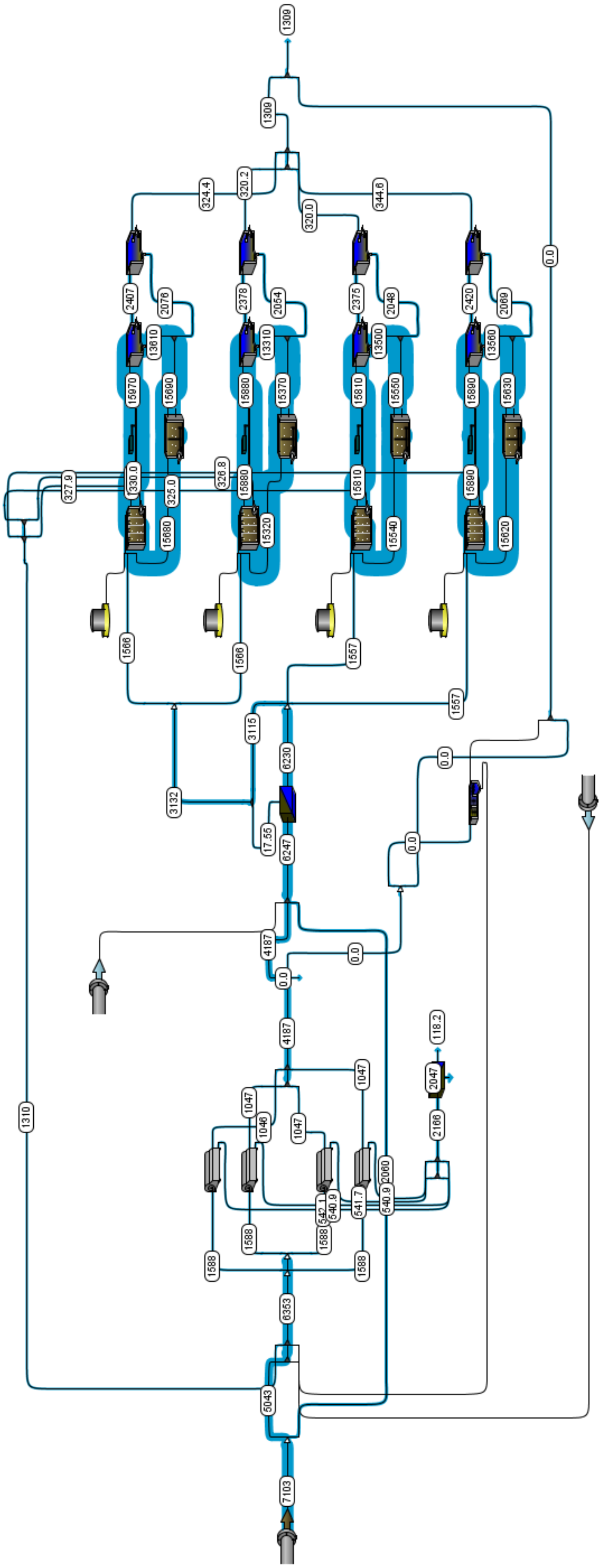
Appendices



Appendix 1. Sankey diagram of the simulated WWTP at 15 °C illustrating the nitrogen balance of the plant. The thickness of the lines indicates the magnitude of the nitrogen flow. The numbers illustrated are kg/d.



Appendix 2. Sankey diagram of the simulated WWTP at 10 °C illustrating the nitrogen balance of the plant. The thickness of the lines indicates the magnitude of the nitrogen flow. The numbers illustrated are kg/d.



Appendix 3. Sankey diagram of the simulated WWTP at 5 °C illustrating the nitrogen balance of the plant. The thickness of the lines indicates the magnitude of the nitrogen flow. The numbers illustrated are kg/d.