

How is the composition of wastewater affected by variations in the environment?

A characterisation of organic matter, nitrogen, and phosphorus in the influent wastewater at Rya wastewater treatment plant in Gothenburg

Master's thesis in Industrial Ecology

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DEPARTMENT OF ARCHITECTURE AND CIVIL ENGINEERING
CHALMERS UNIVERSITY OF TECHNOLOGY

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Stormwater drain causing algae growth on the rock surface. Leif Rehnberg. [Photograph].
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Abstract

More stringent future demands on effluent wastewater quality, together with changes in precipitation patterns due to climate change, make it increasingly important to understand how the composition of wastewater can be affected by environmental parameters. Organic matter, nitrogen, and phosphorus are the main pollutants wastewater treatment plants are built to remove. To ensure efficient removal, it is essential to know more about the speciation of these compounds. This study characterised the influent wastewater at Rya wastewater treatment plant in Gothenburg, mainly focusing on organic matter, but also including nitrogen and phosphorus in the analysis. The aim was to identify how the composition of wastewater is affected by external factors such as precipitation, water temperature, infiltration and inflow, and design of the sewer system (i.e. combined or separate sewers).

The total COD concentration in the influent wastewater ranged between 310 – 620 mg COD/L during the sampling period. Approximately 85% was particulate COD, where 45% was slowly degradable COD, 31% inert particulate COD and 9% heterotrophic biomass. The colloidal COD corresponded to around 6%. Approximately 15% was defined as soluble COD, with 8% inert soluble and 7% readily degradable COD. The readily degradable COD consisted of 2% acetate and 5% other readily degradable COD. The results were compared with previous characterisations and variations in the results were discussed.

Precipitation diluted the wastewater and caused a decrease in the concentrations of soluble fractions of organic matter, nitrogen, and phosphorus in the water. However, the particulate fractions increased with increased precipitation, probably since particles were transported with the infiltration and inflow. Gothenburg has a large share of combined sewers, which carry both stormwater and wastewater to the wastewater treatment plant. The results were, therefore, compared to a previous wastewater characterisation conducted in Linköping, which has a large share of separate sewers, to identify the impact of infiltration and inflow on the composition of the influent wastewater. Low volumes of infiltration and inflow found in separate sewers seem to result in lower concentrations of soluble inert organic matter and increasing concentrations of slowly degradable organic matter.

Results from this study show that there are soluble fractions of nitrogen and phosphorus that are not removed to the same extent as other fractions in the treatment process but could be important to look into further with the increasing demands on treatment in the future. Current

characterisations focus on fractioning organic matter, but future studies should include nitrogen and phosphorus to increase the knowledge on how specific fractions of these pollutants can affect treatment processes.

Key words: Wastewater characterisation, COD, organic matter, nitrogen, phosphorus, fractions, speciation, wastewater treatment, environmental impact

Hur påverkas sammansättningen av avloppsvatten av variationer i miljön?

En karaktärisering av organiska ämnen, kväve och fosfor i inkommande avloppsvatten till Ryaverket avloppsreningsverk i Göteborg

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Sammanfattning

Med framtida ökade krav av kvaliteten på utgående avloppsvatten, i samband med nederbördsskillnader orsakade av klimat- och miljöförändringar, är det viktigt att förstå hur sammansättningen av avloppsvattnet kan påverkas av miljöfaktorer. Organiska ämnen, kväve och fosfor är de främsta föroreningarna avloppsreningsverk är konstruerade för att avskilja. För att försäkra sig om en effektiv rening är det avgörande att ta reda på mer om fördelningen av fraktioner inom parametrarna. Den här studien har karaktäriserat inkommande avloppsvatten till Ryaverkets avloppsreningsverk i Göteborg med primärt fokus på organiska ämnen, men även inkluderat kväve och fosfor i analysen. Syftet var att identifiera hur sammansättningen av avloppsvattnet påverkas av yttre faktorer som nederbörd, vattentemperatur, tillskottsvatten och design på avloppsledningar; kombinerade eller separata system.

Den totala COD-koncentrationen i det inkommande avloppsvattnet sträckte sig mellan 310 – 620 mg COD/L under provtagningsperioden. Närmre 85% var partikulärt COD, där 45% var långsamt nedbrytbart COD, 31% inert partikulärt COD och 9% heterotrof biomassa. Kolloidalt COD motsvarade ca 6%. Ungefär 15% var lösligt COD, där 8% var inert lösligt COD och 7% lättnedbrytbart COD varav 2% av det lättnedbrytbara var acetat och 5% annat lättnedbrytbart COD. Resultaten jämfördes med tidigare karaktäriseringar och variationer i resultaten diskuterades.

Nederbörd spädde ut avloppsvattnet och orsakade en minskning i koncentrationerna av lösliga fraktioner organiska ämnen, kväve och fosfor. Det visade sig emellertid att partikulära fraktioner ökade med ökad mängd nederbörd, troligen orsakat av att partiklar fördes med tillskottsvattnet. Göteborg har en stor andel kombinerade system som transporterar både dagvatten och spillvatten till avloppsreningsverket. Resultaten jämfördes därför med en tidigare avloppsvattenkaraktärisering utförd i Linköping som har en hög andel separata system, för att identifiera påverkan av tillskottsvatten på sammansättningen av inkommande avloppsvatten. Låga volymer tillskottsvatten, som uppkommer i separata system, ser ut att leda till lägre koncentrationer av inert lösligt organiskt material och ökade koncentrationer av långsamt nedbrytbart organiskt material.

Resultat från denna studie visar att det finns lösliga fraktioner av kväve och fosfor som inte avskiljs till samma grad som övriga fraktioner i reningsprocessen, men som kan vara betydelsefulla att studera vidare i kombination med framtidens ökade reningskrav. Nuvarande

karaktäriseringar fokuserar på fraktionering av organiska ämnen men framtida studier bör även inkludera kväve och fosfor för att öka kunskapen om hur specifika fraktioner av dessa föroreningar kan påverka reningsprocesser.

Nyckelord: Avloppsvattenkaraktärisering, COD, organiska ämnen, kväve, fosfor, fraktionering, avloppsvattenrening, miljöpåverkan

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Preface

This Master's thesis is the result of five years of studying, a Bachelor's degree in Civil and Environmental engineering together with a Master's degree in industrial Ecology. It was carried out at the department of Architecture and Civil Engineering, at the Division of Water Environment Technology during the spring of 2021.

I have a lot of people to thank for sharing their knowledge and helping me throughout the writing process. Firstly, I would like to pay special thanks to my supervisor Maria Neth, industrial Ph.D. student at the Division of Water Environment Technology at Chalmers University of Technology and Gryaab AB. She has answered my never-ending stream of questions about wastewater characterisation and guided me and my work in the right directions. With the help of Maria, I have been able to access data from Gryaab AB regarding the wastewater treatment at Rya wastewater treatment plant. I would also like to thank Oskar Modin, my examiner, who has also been there for me whenever I have had questions about the chemistry or laboratory work at Chalmers University of Technology. When mentioning the laboratory, I am also indebted to Amir Saeid Mohammadi and the rest of the people at the laboratory at Chalmers University of Technology who taught me how to conduct laboratory work and also answered a lot of questions.

Of course, I would also like to thank my family and friends who are always there for me. I would not have been where I am today without your unconditional support.

Frida Rehnberg
Göteborg, May 2021

Abbreviations

BOD – Biochemical oxygen demand

C_{tot} – Total colloidal fraction of COD

COD – Chemical oxygen demand

COD_{tot} – Total concentration of COD

$COD_{\text{filt},0.45}$ – Concentration of filtered COD using a filter with pore size of $0.45\mu\text{m}$

$COD_{\text{filt},0.1.6}$ – Concentration of filtered COD using a filter with pore size of $1.6\mu\text{m}$

COD_{ff} – Concentration of flocculated and filtered COD

CSO – Combined sewer overflow

I/I – Infiltration and inflow

MBBR – Moving bed biofilm reactor

$MBBR_{\text{ff}}$ – Concentration of flocculated and filtered water sampled after the nitrifying MBBR step

N – Nitrogen

OUR – Oxygen uptake rate

P – Phosphorus

PO_4 – Orthophosphate

S_B – Readily degradable fraction of COD

S_N – Total soluble fraction of nitrogen

S_{NH_4} – Ammonium nitrogen fraction

S_{NO_2} – Nitrite nitrogen fraction

S_{NO_3} – Nitrate nitrogen fraction

$S_{N,\text{other}}$ – Other soluble fractions of nitrogen

S_{other} – Other soluble fractions of COD

$S_{P,\text{other}}$ – Other soluble fractions of phosphorus

S_P – Total soluble fraction of phosphorus

S_{tot} – Total soluble fraction of COD

S_U – Inert soluble fraction of COD

S_{VFA} – Soluble fraction of COD containing volatile fatty acids

TN – Total nitrogen fraction

TP – Total phosphorus fraction

TSS – Total suspended solids

VFA – Volatile fatty acids

VSS – Volatile suspended solids

WWTP – Wastewater treatment plant

X_B – Slowly degradable fraction of COD

X_N – Total particulate fraction of nitrogen

X_{OHO} – Heterotrophic biomass fraction of COD

X_P – Total particulate fraction of phosphorus

X_{tot} – Total particulate fraction of COD

X_U – Inert particulate fraction of COD

1. Introduction

The world is undergoing changes in many ways where the climate and wildlife is affected negatively by climate change caused by anthropogenic activities around the world. One example is the increase in precipitation which will have a great impact on the climate and its ecosystems (Konapala et al., 2020; Ploughe et al., 2021; Trenberth, 2011). The UN has developed 17 Sustainable development goals (SDGs) that are aiming at creating a more sustainable world with global partnership and cooperation between countries (United Nations, n.d.-b). One of the goals, SDG 6: Clean water and sanitation, focuses on ensuring sustainable water management and sanitation. SDG 14: Life below water, specifically focuses on marine life with a target of preventing all kinds of pollution, including pollution caused by nutrients (United Nations, n.d.-a). By analysing these pollutants and its sources, improvements can be made regarding design of treatment methods leading to a more efficient and sustainable water management.

Wastewater is rich in organic matter and nutrients such as nitrogen and phosphorus originating from human activities (Warwick et al., 2013). These elements are essential for living organisms; however, too high concentrations can be harmful for the environment. Even though a lot of the organic matter, nitrogen, and phosphorus is removed from the wastewater, there are still some left in the effluent water from the treatment plant that is released back into the environment (Davidsson, 2020). Organic matter can cause lack of oxygen in water bodies and harm the environment and wildlife as stated by the Swedish Environmental Protection Agency (Naturvårdsverket, n.d.-a). There is also a risk of eutrophication due to high concentrations of nitrogen and phosphorous released into the environment from wastewater and these consequences can lead to great harm on the environment and its inhabitants. Future changes in efficiency of water utilities will cause the influent wastewater to Rya wastewater treatment plant (WWTP) to decrease which in turn could lead to increases of Chemical oxygen demand (COD), total nitrogen (TN) and total phosphorus (TP) concentrations when grey water is being reused to a greater extent (Bailey et al., 2019). With the changes in climate, impact of anthropogenic activities and urbanisation, the design of wastewater treatment processes and methods have to be considered for optimising removal of compounds and fractions from the wastewater (Bergenstråle, 2019).

1.1 Aim

The aim of this master's thesis is to characterise the influent water to Rya WWTP situated in Gothenburg. The sources of the influent water entering Rya WWTP will be discussed and how they can affect the composition of the wastewater. The focus will be on analysing organic matter in the form of COD and Biochemical oxygen demand (BOD). There will also be analyses of nutrients: phosphorus (P) and nitrogen (N). The characterisation will facilitate more sustainable design choices of processes and treatment methods and contribute to an updated characterisation of Rya WWTP.

The impact of infiltration and inflow (I/I) on the influent water flows to Rya WWTP and how this compares to releasing untreated wastewater directly into nature will be discussed. There will also be comparisons between the current sewer water system, with a combination of separate and combined system, and how future changes in the design of the sewer system will affect the concentrations in the influent wastewater.

1.1.1 Research questions

The following research questions will be discussed:

1. What are the fractions of the organic matter, nitrogen, and phosphorus in the influent wastewater to Rya WWTP?
 - a. Are the fractions of the organic matter, nitrogen, and phosphorus dependent on the days of the week, retention time in the sewer system or other factors?
 - b. Are the fractions of organic matter, nitrogen, and phosphorus comparable with previously conducted characterisations? If not, what are possible reasons for the differences?
2. What is the environmental impact of the variations in wastewater content in the sewer system in Gothenburg?
 - a. How should the wastewater be treated to be able to efficiently remove pollutants?
 - b. Is it possible to identify if certain fractions of organic matter, nitrogen, and phosphorus mainly originates from I/I?
 - c. Would increasing the degree of separated sewer systems likely decrease the concentration of some pollutant fraction entering Rya WWTP?
 - d. Are there any benefits of treating stormwater in the treatment plant compared to other treatment methods?
3. Are there any correlations between certain fractions in the influent and effluent water from Rya WWTP? Are there any fractions that to a large extent pass through Rya WWTP untreated?

1.2 Delimitations

The water that is analysed will be gathered from Rya WWTP and is therefore specific for the Gothenburg region and the municipalities that are connected to the WWTP.

Due to Covid-19, it was not possible to access Rya WWTP and its facilities as usual which limited the possibilities of the sampling, setting duplicates and re-running analyses. There is also a delimitation in the number of days of sample collection. There were approximately three weeks of sample gathering and analysing at weekdays which means that the aim of sampling during different types of weather conditions with varying amount of rainfall was not achieved to a great extent. The sampling was conducted in February – March, meaning that seasonal changes were not included in this study but only winter/early spring conditions. All samples

were taken as flow proportional daily samples and therefore, changes in wastewater composition throughout the day could not be analysed.

Further on, there is a delimitation with the amount of water that was collected in the samplers at Rya WWTP. Since some samples are quite large, they were not all extracted at the same time. For example, the wastewater used in the long-term BOD-test was sampled at a different time than the other samples.

Stormwater contains a lot of pollutants in addition to the organic matter and the nutrients, according to Naturvårdsverket (n.d.-b), however, this study will not analyse further pollutants or metals. Further on, there were no measurements of filtered nitrogen using a 0.45 µm filter since it was considered too time consuming.

2. Background

In this section, the current wastewater treatment processes and water management in Gothenburg is presented together with the speciation of organic matter, nitrogen, and phosphorus used in this study.

2.1 The sewer system of Gothenburg

The composition of the wastewater from human activities will depend on several factors, including human behaviour and standard of living (Henze & Comeau, 2008). However, there are also other factors affecting the content of the water such as the design of the sewer system, (i.e. if there are combined or separate systems in use) as well as natural phenomena like precipitation and temperature. Combined sewer systems transport wastewater together with stormwater to WWTPs, in contrast to separated sewers which have separated pipes for the different types of water (Naturvårdsverket, 2016). With approximately 25% of combined sewer system in Gothenburg, heavy rainfall might cause overflows in these pipes (Brunsten, 2020). When there is too much water in the sewer system, the water has to be released using combined sewer overflows (CSOs), and the water is thereby not treated from pollutants or other environmentally degrading substances and material. The combined sewer system will transport larger volumes of water in the pipes and will therefore affect the concentration of pollutants, organic matter and nutrients ending up at Rya WWTP and according to Wilén et al. (2006) even the oxygen levels. By replacing the combined sewers with separated ones, the issues connected to overflows and large quantities of water transported to the WWTP will decrease (Hey et al., 2016). However, this comes to a large economic cost and could also increase risks connected to releasing stormwater to local water bodies instead of transporting it to the WWTP. An estimation by Nivert & Alenius (2017) and The city of Gothenburg (Göteborgs stad) shows that the cost of replacing combined sewers with separates sewers is approximately 10 billions and will take 50 to 100 years, reducing the I/I with approximately 28%.

2.1.1 Infiltration and inflow

Examples of I/I could be rainwater, groundwater, water leaking into the pipes and water from the sea as seen in Figure 2.1. The letters represent as follows: D – Drainage and stormwater pipe, V – drinking water pipe, S – Sanitary sewer pipe.

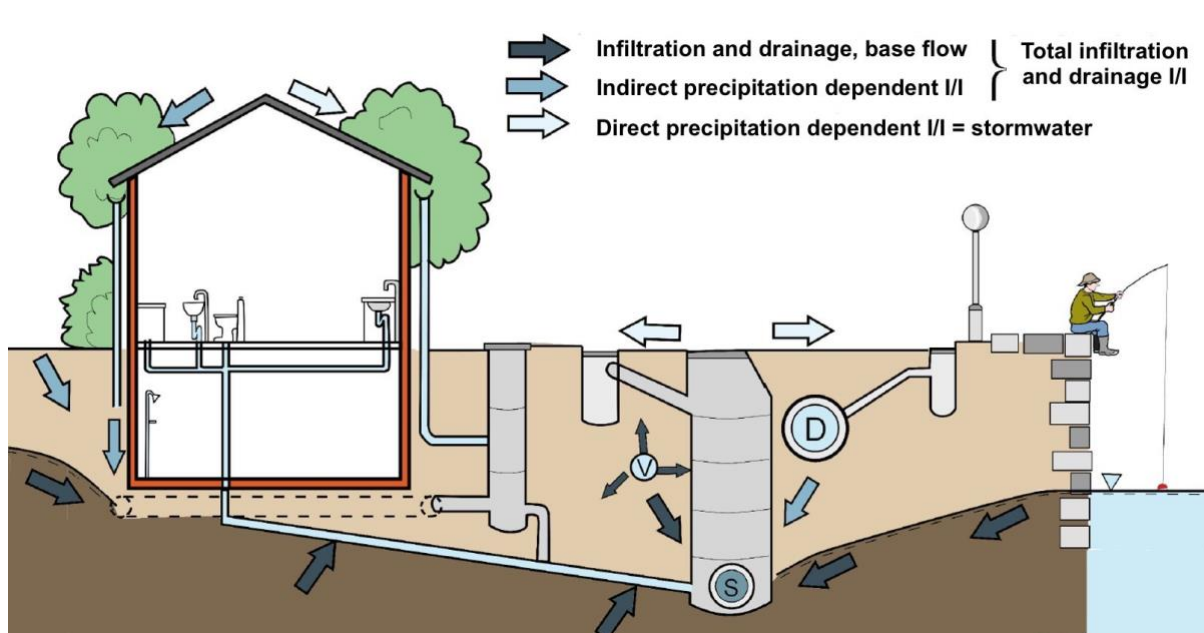


Figure 2.1. Different sources of I/I entering the pipes in a separate sewer system where the house foundation drainage is connected to the sewer system. Adapted and translated from Swedish from “P110 del 1: Avledning av dag-, drän- och spillvatten” by Svenskt Vatten AB, 2016, (p.17). Adapted with permission.

I/I is divided into three components, as follows (Clementson et al., 2020):

- Infiltration and drainage dependent I/I: Impact from leakage and drainage water referring to groundwater leaking into the wastewater sewer system. This will increase during wet periods and will not show any change during separate rainfall events.
- Indirect precipitation dependent I/I: There will be a flow increase during the first three days of a rain event.
- Direct precipitation dependent I/I: This will lead to an increase in water flows due to rain falling on hard surfaces, usually referred to as fast rainfall response and stormwater. It will decrease with the time that corresponds to the concentration time in the specific area.

By replacing the combined sewers with separated sewers, the direct precipitation dependent I/I is affected (Clementson et al., 2020). Since the I/I is entering the water system in different ways, there is a need of different measures mitigating the impact of the excess water. One cause for I/I entering the sewer system can be due to damages of the pipes both in the municipal part of the sewer system as well as in the privately owned (Lundblad & Backö, 2016). Measurements that could be implemented are e.g. either fixing the pipes directly or implementing Sustainable urban water drainage systems (SUDS), in Swedish called Lokalt omhändertagande av vatten (LOD). It might be difficult finding a standardised solution preventing I/I of entering the sewer system since the reasons for the I/I might differ between cases. There are also connections between the geological material surrounding the pipes and the infiltration rate, e.g if the material is permeable, the water will more easily be transported (Ohlin Saletti, 2021).

Throughout a year, approximately 60% of the incoming water to Rya WWTP is assumed to be I/I (Davidsson, 2020). Compared to other cities in Sweden, Rya WWTP receives a large volume of I/I each year which is probably due to precipitation and geographical differences (Nivert & Alenius, 2017). With the increase of precipitation due to climate change, volumes of water entering WWTPs will increase which in turn will increase the operational costs. In the future there will be more strict requirements of the removal efficiency of organic matter, nitrogen, and phosphorus and the larger influent water volumes have to be considered (Mattsson et al., 2016). There are costs related to the WWTP such as investment cost for expanding Rya WWTP and the operational cost for running the plant (Nivert & Alenius, 2017). However, there are also the costs outside of the WWTP caused by e.g. flooding and replacing combined systems with separate ones. All these parameters have to be considered when deciding which measurements towards the I/I that have to be made.

2.1.2 Stormwater management in Gothenburg

There are no guidelines on a national level in Sweden regarding I/I management (Hey et al., 2016). However, stormwater is managed mainly through SUDS (Göteborgs Stad, 2010). These solutions can be implemented to treat stormwater on site, as well as prolonging the time it takes for the water to reach the WWTP. Water can also be released directly to the recipients or into the storm drains which is possible if the sewer system is separated (Göteborgs stad, 2020). When releasing the water directly to recipients, it is important to ensure that concentrations of pollutants and harmful substances the water are not exceeding specific guidelines. The composition of the stormwater has seasonal variations and it is also affected by locations and precipitation (Viklander et al., 2019). Additionally, there is generally a larger concentration of pollutions in the beginning of a rain event, often referred to as the *first flush*. The impact of stormwater on the wastewater treatment is noticeable and is important to consider when discussing the design and efficiency of the treatment processes (Molander, 2015).

2.2 Wastewater treatment in Gothenburg

Rya WWTP is the wastewater treatment plant in Gothenburg, operated by Gryaab AB, treating wastewater with its main focus on removing organic matter, nitrogen, and phosphorus, as well as particles and sedimentary material (Davidsson, 2020). Rya WWTP is responsible for treating wastewater originating from the municipalities of Gothenburg; Ale, Härryda, Kungälv, Lerum, Mölndal and Partille (Göteborgs stad, n.d.; Gryaab AB, n.d.-a). In 2020, there were 797 485 people connected to Rya WWTP (Gryaab AB, 2020). The sewer system as well as pump stations and combined sewer overflow pipes are operated by the corresponding municipality. The tunnel system leading the water to Rya WWTP is approximately 130 kilometres long, and a simplified drawing of the tunnel system can be seen in Figure 2.2 (Davidsson, 2020). Since the tunnels are tilted with one millimetre per meter, the water can be transported to Rya WWTP without using additional pumps. The tunnel that is connected to Rya WWTP is located 20 meters underneath the surface and thereafter, the water is pumped to the WWTP.

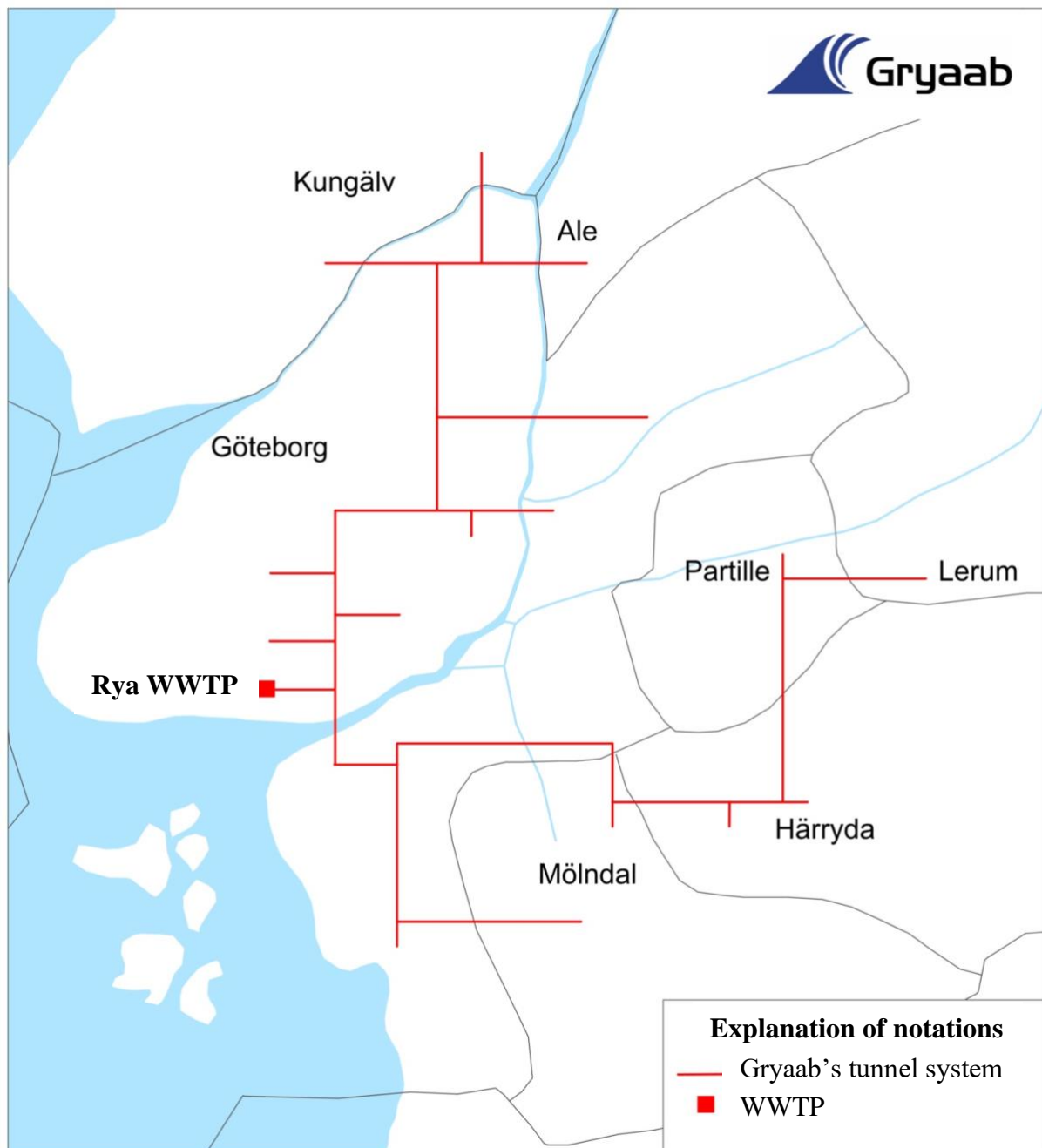


Figure 2.2. Map overview of the tunnel system in Gothenburg. Adapted and translated from Swedish from “Teknisk beskrivning – Tillståndsansökan för Ryaverket” by Nunes et al., 2017, (p. 9). Adapted with permission.

At Rya WWTP, there are mechanical, physical, chemical, and biological treatment of the wastewater. Figure 2.3 presents the wastewater treatment processes at Rya WWTP (Davidsson, 2020). Firstly, the water is pumped into the WWTP through coarse bars and is thereafter transported to an aerated sand trap where sand and other heavy particles are removed. By using aeration, the water and particles will rotate and therefore allow heavy sand particles to settle while other compounds and organic matter can flow through the water without settling at the bottom of the tank (Svenskt Vatten AB, 2013). The wastewater flows through fine bars with a

2 mm width where the sand is separated and washed to remove particles and organic matter (Davidsson, 2020).

Preliminary sedimentation removes particles from the wastewater and the particulate material that is removed consists of organic matter and nutrients (Svenskt Vatten AB, 2013). When there are large volumes of influent water to Rya WWTP, the treatment process of direct precipitation can be used for part of the flow (Davidsson, 2020). In this step, chemicals are added to the wastewater, usually polyaluminium chlorides (PAC) together with a polymer, flocculating the phosphate and adsorbing pollutants. This process is occurring right after the fine bar screen and replaces all the following treatment steps. The wastewater that is released through direct precipitation contains approximately the same concentration of phosphorus as the treated wastewater. When the water has gone through all treatment steps described below, it is released into Göta river at Rya Nabbe whereas separated particles and objects are cleansed, drained of water and incinerated (Nunes et al., 2017).

2.2.1 Removal of organic matter

According to Henze & Comeau (2008), the major pollutant of wastewater is organic matter meaning that the treatment methods removing this compound need to be efficient. Biological treatment using microorganisms in activated sludge is one process for removal of organic matter (Svenskt Vatten AB, 2013). The sludge that is removed from the secondary sedimentation is recycled to the activated sludge process to maintain a specific level of sludge concentration in the basin. The activated sludge basins are separated into aerated and anoxic zones to be able to use nitrate to remove organic matter (Gryaab AB, n.d.-d). The yearly average concentration of organic matter entering the environment from Rya WWTP was 5.7 mg BOD/L in 2020, meaning a 95% reduction compared to the concentrations in the influent wastewater (Gryaab AB, 2021). The limit value of BOD in effluent water was 10 mg/L in 2020 (Gryaab AB, 2020).

Measuring organic matter can be done using several methods. The COD analysis will identify all organic material in the wastewater. By using a chemical oxidising agent, the oxygen depletion in the water sample will be estimated (Niessen, 2010). The BOD analysis will identify the readily degradable organic material in the wastewater conducted within the course of a specific amount of days, where five and seven days usually are the standards, in that case denoted as BOD₅ and BOD₇ (Niessen, 2010). In the BOD analysis, microorganisms use the organic material as fuel which will partly be converted back into microorganisms but also carbon dioxide and water. This method will only identify the biodegradable organics and the need of time required can be considered a drawback.

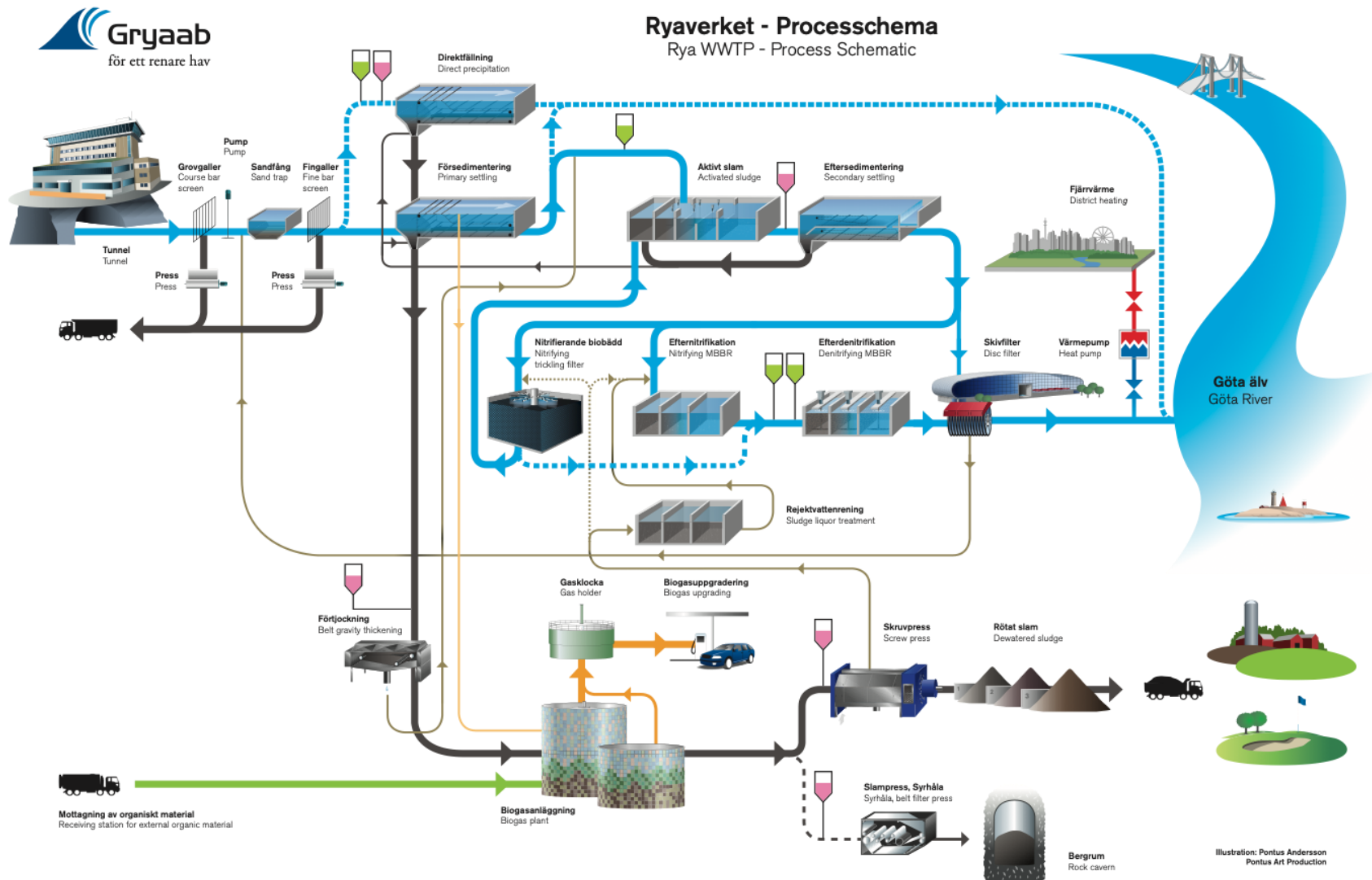


Figure 2.3. Process schedule for the wastewater treatment at Rya WWTP. Reprinted from “Miljörapport Ryaverket 2019” by Davidsson, 2020, (p. 5). Reprinted with permission.

2.2.2 Nitrogen removal

Nitrogen is present in several parts of the environment, for instance in the atmosphere which constitutes of 78% nitrogen (Naturvårdsverket, n.d.-c). One of the largest sources of total nitrogen ending up in nature is from WWTPs, causing harm on the environment since nitrogen can, similarly as phosphorus, lead to eutrophication.

Nitrification is one method to remove nitrogen from the wastewater by letting bacteria grow on a dark coloured plastic material (Gryaab AB, n.d.-c). The ammonium nitrogen (NH_4) will in this step be transformed to nitrate (NO_3). Denitrification, in the activated sludge step, will lead to reduction of NO_3 through an anoxic environment where there is no oxygen present. The Moving Bed Biofilm Reactor (MBBR) treatment step appears after the denitrification and contains mobile plastic carriers on which bacteria can grow (Gryaab AB, n.d.-c). In this step, NO_3 will be transformed to nitrogen gas (N_2) and is released into the air. The yearly average concentration of nitrogen entering the environment from Rya WWTP was 7.1 mg N/L in 2020, meaning a 75% reduction compared to the concentrations in the influent wastewater (Gryaab AB, 2021). The limit value of nitrogen in effluent water was 8 mg/L in 2020 (Gryaab AB, 2020).

2.2.3 Phosphorus removal

Phosphorus is a compound that occurs naturally in the environment, but there are also anthropogenic sources increasing the concentrations which could lead to potential issues regarding biodiversity (Naturvårdsverket, 2004). Excess of phosphorus in the environment and in water bodies can lead to eutrophication and by that the extinction of wildlife following the large decrease in oxygen levels and the production of toxins.

The main phosphorus removal at Rya WWTP is through adding ferrous sulphate before the activated sludge step (Gryaab AB, n.d.-c). When using direct precipitation in the case of large influent water flows, phosphorus is incorporated into the sludge and therefore removed from the water (Svenskt Vatten AB, 2013). A part of the phosphorus, in the form of orthophosphate (PO_4), is bound to the bio sludge in the biological treatment which will lead to phosphorus removal. In the stabilisation of primary sludge and bio sludge, degradation will lead to some part of the phosphorus being separated from the sludge treatment process. Phosphorus is also removed in the last sedimentation step as well as in the disc filter. The yearly average concentration of phosphorus entering the environment from Rya WWTP was 0.2 mg P/L in 2020, meaning a 93.6% reduction compared to the concentrations in the influent wastewater (Gryaab AB, 2021). The limit value of phosphorus in effluent water was 0.3 mg/L in 2020 (Gryaab AB, 2020).

2.3 Speciation of organic matter

The organic matter, analysed as COD, will be fractioned into speciation such as soluble, colloidal, and particulate form. A colloid is generally defined as a particle with a diameter of the size 0.45 - 5 μm (van Nieuwenhuijzen et al., 2004). This type of particle can provide

sorption to a great extent, meaning that they can have a great impact on the composition of the wastewater as well as the rate of transport of other compounds.

The speciation is a part of the characterisation of the wastewater, to be able to compare the results with previous studies and to discuss the treatment methods that will be used for the different fractions. The characterisation of the fractions will facilitate the design of the wastewater treatment plant, creating a model of efficient treatment methods and processes (Lysberg & Neth, 2012). The fractioning of COD used in this study can be seen in Figure 2.4.

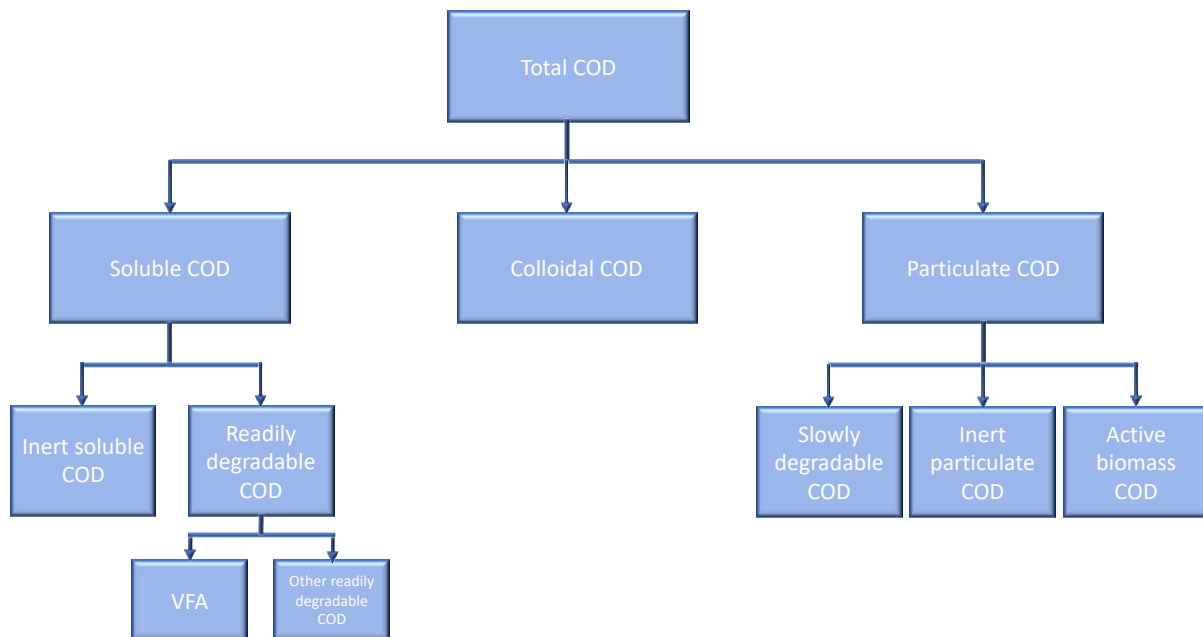


Figure 2.4. Fractioning of COD.

The notation of the parameters can be seen in Table 2.1, using notations from Corominas et al. (2010). Previously there have been several different ways of annotating fractions when characterising wastewater treatment modelling which can lead to misunderstanding and confusion when comparing different research articles. The subscripts of the parameters used in this study are referring to the degradability where U means undegradable and B means biodegradable. In the active biomass COD, the subscript OHO represents ordinary heterotrophic organisms.

Table 2.1. Notation of COD parameters.

COD	Notation
Total COD	COD_{tot}
Soluble COD	S_{tot}
Inert soluble	S _U
Readily degradable	S _B
VFA	S _{VFA}
Other readily degradable	S _{other}
Colloidal COD	C_{tot}
Particulate COD	X_{tot}
Slowly degradable	X _B
Inert particulate	X _U
Active biomass COD	¹
Heterotrophic biomass	X _{OHO}

¹Active biomass COD = heterotrophic biomass (X_{OHO}) + autotrophic biomass (neglected) + other (neglected). This equation is used with the argument that the neglected fractions are very small compared to the total COD (Roeleveld & Van Loosdrecht, 2002).

2.4 Speciation of nitrogen

The fractionation of nitrogen used in this study can be seen in Figure 2.5 where the focus is on the soluble and particulate organically bound nitrogen. Ammonium is the main part of nitrogen and therefore further characterisation of nitrogen is seldom conducted (Roeleveld & Van Loosdrecht, 2002).

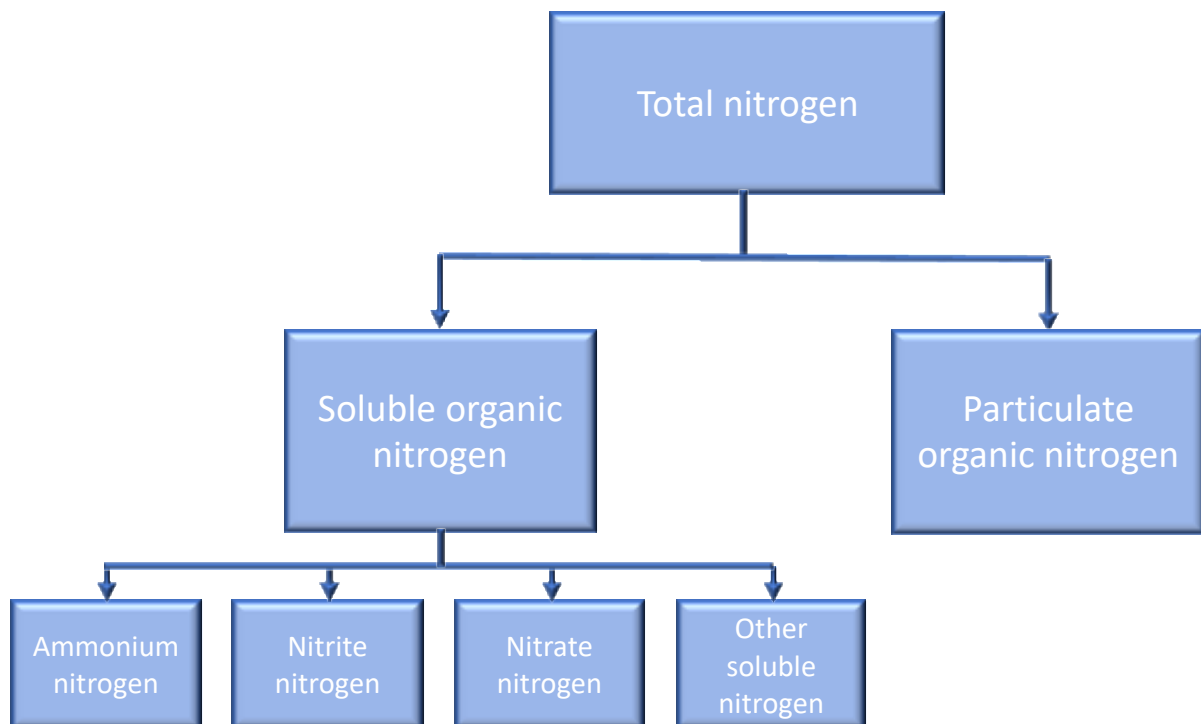


Figure 2.5. Fractioning of nitrogen.

Table 2.2 presents the notation of the nitrogen parameters.

Table 2.2. Notation of nitrogen parameters.

Nitrogen	Notation
Total nitrogen	TN
Soluble organic nitrogen	S_N
Ammonium nitrogen	S _{NH4}
Nitrite nitrogen	S _{NO2}
Nitrate nitrogen	S _{NO3}
Other soluble fractions	S _{N,other}
Particulate nitrogen	X_N

2.5 Speciation of phosphorus

The fractioning of phosphorus used in this study can be seen in Figure 2.6. The total phosphorus concentration is divided into soluble and particulate phosphorus. The reactive soluble fraction of phosphorus constitutes mainly of orthophosphate (Calvo-López et al., 2021).

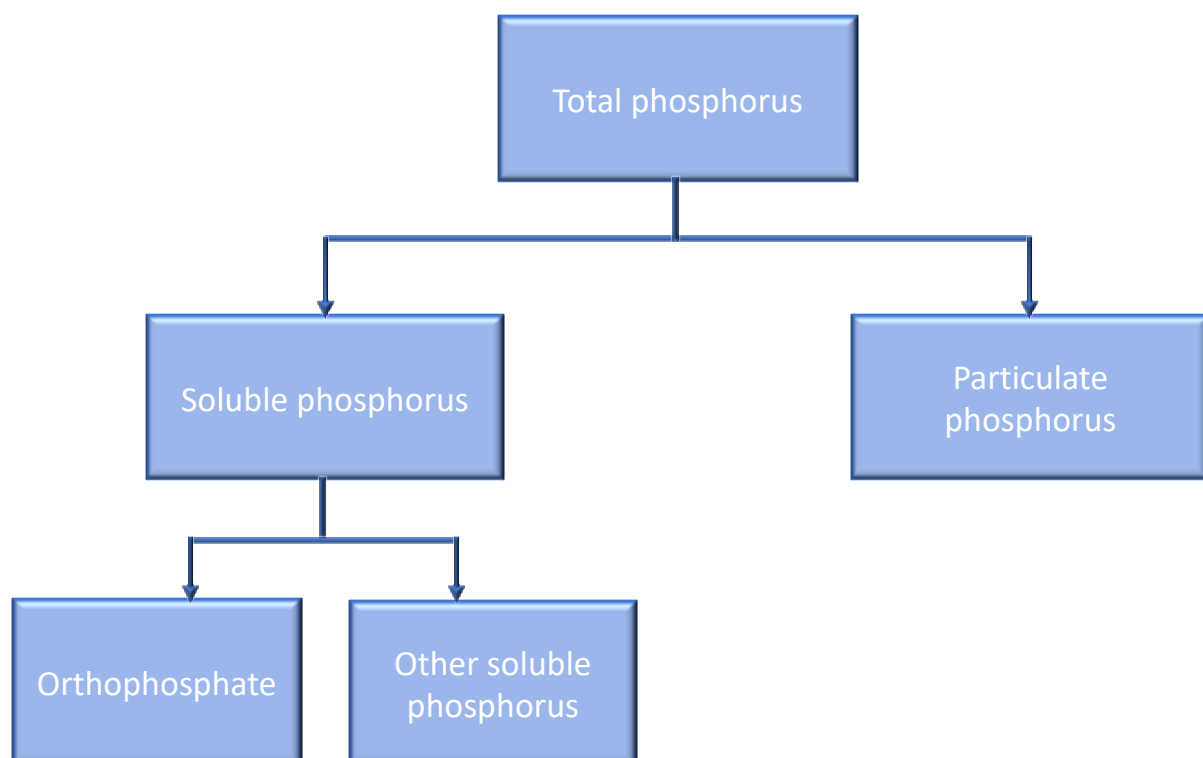


Figure 2.6. Fractioning of phosphorus.

Table 2.3 presents the notation of the phosphorus parameters.

Table 2.3. Notation of phosphorus parameters.

Phosphorus	Notation
Total phosphorus	TP
Soluble phosphorus	SP
Orthophosphate	PO ₄
Other soluble phosphorus	S _{P,other}
Particulate phosphorus	X_P

3. Materials and method

This section will describe the methodology used in this study with a literature study and laboratory work.

3.1 Method of literature review

Firstly, a literature research of previous analyses and characterisation of wastewater was conducted. Literature was found through Gryaab AB as well as at Scopus, Elsevier, Google Scholar, information from books and at company websites. The literature review focused on the following areas:

- Wastewater treatment in Gothenburg, specifically treating organic matter and nutrients
- Composition of wastewater and its fractions
- Characterisation of organic matter, nitrogen, and phosphorus
- I/I and its impact on treating water at WWTPs
- Precipitation, temperature, and other parameters with potential impact on wastewater composition
- Stormwater management in Gothenburg

3.2 Method of laboratory work

Further on, there was a laboratory work conducted within the course of five weeks from the 8th of February to the 15th of March, studying the samples gathered at Rya WWTP. The first two weeks of the laboratory work were mainly for trying out methods for the specific parameters and fractions. After that, there were three weeks of actual testing and analysing the samples from Rya WWTP. The laboratory work was conducted at Chalmers University of Technology and focused on organic matter by analysing COD, whereas nitrogen and phosphorus was analysed at Rya WWTP, and BOD analyses were conducted by Göteborgs Kemianalys. The following sections will describe the methods used in the sample gathering and laboratory work.

Samples of the influent and effluent water from Rya WWTP was gathered through flow proportional daily sampling throughout the course of approximately 21 days. Since variations in precipitation and temperature might affect the characterisation of the influent wastewater, it is important to ensure that these differences are captured and represented in the results (Martin & Vanrolleghem, 2014). With data from the Swedish meteorologic and hydrologic institute (SMHI), temperature and precipitation data were identified and is presented in the results and discussion section.

The samples on influent wastewater were collected in the morning, around 7.00 am, with sampling occurring from 7.00 am the day before. The sample after the nitrifying MBBR step was a grab sample collected in the morning of the sampling day. The samples arrived at Chalmers University of Technology at approximately 10.00 am each day and to acquire correct results, the samples needed to be analysed as soon as possible after they had been collected.

Table 3.1 presents the parameters and methods used when analysing the influent water to Rya WWTP as well as references and standards for the different procedures.

Table 3.1. Analyses of the influent water at Rya WWTP (continuing on the next page).

Parameter	Notation	Method	Filter size	Reference
Organic matter - COD				
Total COD	COD_{tot}	Cuvette method		Standard ISO 6060-1989 (International Organization for Standardization (ISO), 1989)
Filtered COD	$COD_{filt,1.6}$	Filtering + cuvette method	1.6 μm	
Filtered COD	$COD_{filt,0.45}$	Filtering + cuvette method	0.45 μm	
Flocculated and filtered COD	COD_{ff}	Flocculation of COD with $Zn(OH)_2$ + filtering + cuvette method	0.45 μm	(Roeleveld & Van Loosdrecht, 2002)
Organic matter - other				
Inert organic matter in COD (particulate and soluble)	S_U, X_U	Long-term degradation test for 10 days of unfiltered and flocculated and filtered wastewater. Flocculation with $Zn(OH)_2$.	0.45 μm	(Lesouef et al., 1992)
BOD for 1,2,5,7, and 9 days	BCOD	BOD – test		(Choi et al., 2005)
VFA	S_{VFA}	HPLC (High-performance liquid chromatography)		(De Sá et al., 2011)
Heterotrophic biomass	X_{OHO}	Respirometry/OUR analysis (oxygen uptake rate)		(Weijers, 1999; Wentzel et al., 1995)
Nitrogen				
Total nitrogen	TN	Spectrophotometer		Standard SS-EN 12260:2004 (Swedish Standards Institute (SIS), 2004)
Filtered total nitrogen	TN_{filt}	Filtering + spectrophotometer	1.6 μm	Standard ISO 15923-1:2013 (ISO, 2013)

Table 3.1 (cont.). Analyses of the influent water at Rya WWTP.

Parameter	Notation	Method	Filter size	Reference
Nitrogen				
Ammonium	S_{NH_4}	Filtering + spectrophotometer + AQ400 instrument	1.6 μm	Standard ISO 15923-1:2013 (ISO, 2013)
Nitrate and nitrite	S_{NO_2}, S_{NO_3}	Filtering + spectrophotometer + AQ400 instrument	1.6 μm	Standard ISO 15923-1:2013 (ISO, 2013)
Phosphorus				
Total phosphorus	TP	Spectrophotometer		(Nobel, 2015)
Filtered total phosphorus	TP_{filt}	Filtering	1.6 μm	
Orthophosphate	PO_4	Filtering + spectrophotometer	0.45 μm	(Warwick et al., 2013)
Other				
Total suspended solids	TSS	Oven method at 150°C		Standard SS-EN 872:2005 (SIS, 2005)
Volatile suspended solids	VSS	Filtering + burning at 550°C	1.6 μm	Standard SS-EN 872:2005 (SIS, 2005)

Table 3.2 presents the parameters and methods used for analysing the effluent water from Rya WWTP as well as references and standards for the different procedures.

Table 3.2. Analyses of the effluent water at Rya WWTP.

Parameter	Notation	Method	Filter size	Reference
COD				
Total COD	COD _{tot}	Cuvette method		Standard ISO 6060-1989 (International Organization for Standardization (ISO), 1989)
Filtered COD	COD _{filt,1.6}	Filtering + cuvette method	1.6 µm	
Nitrogen				
Total nitrogen	TN	Spectrophotometer		Standard SS-EN 12260:2004 (SIS, 2004)
Filtered total nitrogen	TN _{filt}	Filtering	1.6 µm	Standard ISO 15923-1:2013 (ISO, 2013)
Ammonium	S _{NH4}	Filtering + spectrophotometer + AQ400 instrument	1.6 µm	Standard ISO 15923-1:2013 (ISO, 2013)
Nitrate and nitrite	S _{NO2} , S _{NO3}	Filtering + spectrophotometer + AQ400 instrument	1.6 µm	Standard ISO 15923-1:2013 (ISO, 2013)
Phosphorus				
Total phosphorus	TP	Spectrophotometer		(Nobel, 2015)
Filtered total phosphorus	TP _{filt}	Filtering	1.6 µm	
Orthophosphate	PO ₄	Spectrophotometer	0.45 µm	(Warwick et al., 2013)

Table 3.3 presents the analyses, including reference, of the water extracted after the nitrifying MBBR step where most of the biodegradable organic matter is assumed to be removed. After the MBBR step, methanol is added to the water as a carbon source for the microorganisms which is easily overdosed and will lead to an increase of organic matter (la Cour Jansen et al., 2019). The reason for measuring the flocculated and filtered water after the nitrifying MBBR step (noted as MBBR_{ff} in this study) is to estimate the lowest amount of organic matter in the WWTP.

Table 3.3. Analyses of the water extracted after the nitrifying MBBR step.

Parameter	Notation	Method	Filter size	Reference
Organic matter - COD				
Flocculated and filtered COD	MBBR _{ff}	Flocculation with Zn(OH) ₂ and filtered COD	0.45 µm	(Roeleveld & Van Loosdrecht, 2002)

3.2.1 Measurement of organic matter

Three samples of influent wastewater were analysed for COD; unfiltered, filtered as well as flocculated and filtered, with two replicates of each sample. The water sampled from after the nitrifying MBBR step was flocculated and filtered and analysed using the same method as the other samples. Analyses of COD with filters of the pore size 1.6 µm was conducted at Rya WWTP as well as unfiltered COD analyses. Table 3.4 presents the relationships and equations for the fractions of COD, including Equation (1) showing the total COD relation to soluble, colloidal, and particulate COD.

Table 3.4. Equations and relationships used for the COD measurements.

Parameter	Relationship	Equation notation
Total COD	$COD_{tot} = S_{tot} + C_{tot} + X_{tot}$	(1)
VSS	$VSS = TSS - \text{non-volatile solids}$	(2)
Total soluble COD	$S_{tot} = S_U + S_{VFA} + S_{other}$	(3)
Inert soluble COD	$S_U = MBBR_{ff} - S_{VFA}$	(4)
Soluble readily degradable COD	$S_B = S_{tot} - S_U$	(5)
Total colloidal COD	$C_{tot} = COD_{filt,0.45} - COD_{ff}$	(6)
Total particulate COD	$X_{tot} = X_B + X_U + X_{OHO}$	(7)
Biodegradable COD	$BCOD = X_B + S_B$	(8)
	$BOD_{tot} = \frac{1}{1 - e^{-k_{BOD}t}} BOD_t$	(9)
	$BCOD = \frac{1}{1 - f_{BOD}} BOD_{tot}$	(10)
Particulate slowly degradable COD	$X_B = BCOD - S_B$	(11)
Inert particulate COD	$X_U = \frac{COD_{final} - S_i - Yield (COD_{initial,tot} - S_i)}{1 - Yield}$	(12)
Heterotrophic biomass	$X_{OHO} = \frac{e^{(y-intercept)}}{\frac{1 - Y_H}{Y_H} \cdot (slope \cdot 24 + b_h)}$	(13)

Filtering

The filtering at the laboratory at Chalmers University of Technology was conducted using sterile syringe filter with the pore size of $0.45\ \mu\text{m}$, from VWR International as seen in Figure 3.1. The wastewater was pushed through a $0.45\ \mu\text{m}$ filter using a syringe and when the filter was clogged, it was replaced with a new one. Part of the filtrate was put in the freezer to be used in the HPLC analysis.



Figure 3.1. Syringe filters and syringe used in the laboratory work.

Flocculation

To be able to identify soluble and colloidal COD in the influent wastewater and the water from the nitrifying MBBR step, the water was flocculated using zinc hydroxide $\text{Zn}(\text{OH})_2$ (Mamais et al., 1993). The flocculation captured the colloidal particles and by filtering the solution using a $0.45\ \mu\text{m}$ filter, soluble COD was identified. This was done by adding one ml of 100g/L zinc sulphate (ZnSO_4) to a wastewater sample and mixing the sample vigorously for about one minute. After that, 6M sodium hydroxide (NaOH) was added until reaching a pH of 10.5. The solution settled for a few minutes and some of the supernatant was removed through filtering using syringe filters. A part of the filtrate was put in the freezer to be used in the HPLC analysis.

Hach Lange cuvette method

The COD measurements were conducted using a cuvette method (ISO, 1989). Each cuvette was filled with 2 ml of a sample; unfiltered, filtered and flocculated and filtered water, using a pipette (Pütz, n.d.). After that, they were inverted a few times to mix the sample with the content of the cuvettes. The cuvettes contain potassium dichromate in a 50% sulfuric acid solution, specific for the concentration of COD which for the wastewater was assumed to be 15-150 mg

COD/L. The cuvettes were placed in a heating block thermostat for two hours at 150 °C as seen in Figure 3.2. After two hours, the samples were left to cool down for 20 minutes in the thermostat. When they had cooled down reaching a temperature of approximately 120°C, they were inverted a few times, removed from the thermostat, and put in a rack to cool down even further reaching room temperature. After this, they were wiped off with a paper towel and placed in the Hach Lange photometer providing the COD concentration in [mg/L]. One blank sample with MilliQ water was included in the cuvette method as well as a COD standard of 50 mg COD/L provided from Gryaab AB, to check the instrument and the method procedure.



Figure 3.2. Heat block used in the COD analysis.

Analysis of suspended solids

The definition of total suspended solids (TSS) in a water sample is varying depending on the standard filter used (la Cour Jansen et al., 2019). This type of compound consists of particulate matter that is transported as suspended particles, however, they will sediment if the water is still (Naturvårdsverket, n.d.-b). Other compounds and pollutants will adsorb to the surface of TSS, specifically after heavy rainfall or snow melt since this will increase the mobility of particles. The TSS was measured at Rya WWTP using a standard from the Swedish Standards Institute (SIS) (2005) with some modifications to decrease the time of analysis. The sample was filtered through glass fibre filters with a pore size of 1.6 μm and afterwards dried in a microwave oven at 150°C. After that, the filter was cooled down and weighted.

By burning the TSS at a high temperature and calculating the weight of the remaining ashes, the volatile suspended solids (VSS) was estimated (Queenan et al., 1996). The VSS was measured by placing the sample in an oven at 550°C according to a standard from SIS (1981).

Equation (2) in Table 3.4 shows the calculation of VSS by subtracting the non-volatile solids from the TSS.

3.2.2 Identification of soluble COD

S_U was determined by flocculation with $Zn(OH)_2$ and filtering using syringe filters with a pore size of $0.45\ \mu m$ (Roeleveld & Van Loosdrecht, 2002). The reason for using this pore size of the filter is that if smaller pore sizes are used, the filter might be clogged. According to Roeleveld & Van Loosdrecht (2002) when using the flocculation in addition to the filtering through a $0.45\ \mu m$ filter, this corresponds to filtering through a $0.1\ \mu m$ filter. In the end, a COD analysis using the cuvette method was made. S_{tot} consists of S_U , S_{VFA} and S_{other} as seen in Equation (3) in Table 3.4.

Long-term BOD test to determine S_U and X_U

To determine S_U and X_U , a long-term BOD test was conducted for 10 days (Lesouef et al., 1992). Two different samples of 50 ml were analysed, one unfiltered and one flocculated and filtered using a $0.45\ \mu m$ syringe filter. Since the pH value was high, approximately 10.5 due to the flocculation, 2 M hydrogen chloride (HCl) was added to reduce the pH to 7 – 8. This was done since microorganisms prefer an environment around this pH value (Svenskt Vatten AB, 2013). To ensure that there is degradation occurring, 50 μm inoculate in the form of untreated wastewater, was added to this sample. The samples were poured into two separate 0.5 L glass bottles adding a magnet and putting on a cap. They were covered in aluminium foil to reduce the risk of light impact as can be seen in Figure 3.3. After that they were placed on a magnet stirrer and kept there for ten days.



Figure 3.3. Long-term BOD test.

The concentration of oxygen present in the bottle and the expected oxygen demand was calculated with values from Lysberg & Neth (2012) as seen in Appendix I. It was decided that the cap should be opened every day during the experiment. The calculations showed that the oxygen needed for the degradation in the unfiltered sample was quite close to the oxygen present in the bottle and therefore, excess air should be added. However, since the experiment only occurred on weekdays, the bottles were not aerated during the weekends. To let in air, the cap was opened and then quickly put on again to mitigate the risk of evaporation during the aeration. When 10 days had passed, the flocculated and filtered sample was flocculated and filtered once more to remove the particulate or colloidal matter that had been formed. COD was then measured on both samples using the cuvette method.

Another method that was used to decide S_U was flocculation, filtering and COD cuvette measurement of the water extracted after the nitrifying MBBR step. If there would be any VFAs present, this value was subtracted from the measured COD concentration to obtain S_U , as seen in Equation (4) in Table 3.4. The calculations used to determine X_U obtained from the long-term BOD test are explained in section 3.2.4. To be able to determine S_B , S_U is subtracted from S_{tot} as seen in Equation (5) in Table 3.4, according to Roozeveld & Van Loosdrecht (2002).

High performance liquid chromatography to determine S_{VFA}

Volatile fatty acids (VFAs) are defined as the smallest forms of molecule organic material that can be found and are formed through fermentation with the main species being acetate (Ekama, 2011). VFAs are important and easily accessible carbon sources used in the denitrification step

of the wastewater treatment (Owusu-Agyeman et al., 2020; Pan, 2019). It can be used to create a closed-loop system of the wastewater treatment instead of adding external carbon sources. $SVFA$ was measured using a High performance liquid chromatography instrument (HPLC) (De Sá et al., 2011). The specific instrument used in this study was a Shimadzu UFLC (Ultra Fast Liquid Chromatograph). This measurement was conducted at the end of the weeks of the laboratory work, using frozen samples from each day. The analysis was studying acetate, propionate and butyrate using H_2SO_4 as a mobile phase. Approximately one ml of filtered wastewater samples and flocculated and filtered water after the nitrifying MBBR step from each day was added to small glass bottles using a pipette. They were placed in a plastic holder and put into the HPLC instrument as can be seen in Figure 3.4. The column was an Aminex HPX-87H (Bio-Rad) with temperature set to $40^{\circ}C$ and a UV detector at 210 nm. The injection volume of the needle volume was set to 100 μl . Each sample was analysed for 42 minutes and when the analysis was finished, the data was exported to Excel and results were presented in [mmol/L]. The concentration in the unit of [mg COD/L] was calculated using specific mol fractions for acetate (64 mg COD/mmol), propionate (112 mg COD/mmol) and butyrate (160 mg COD/mmol).

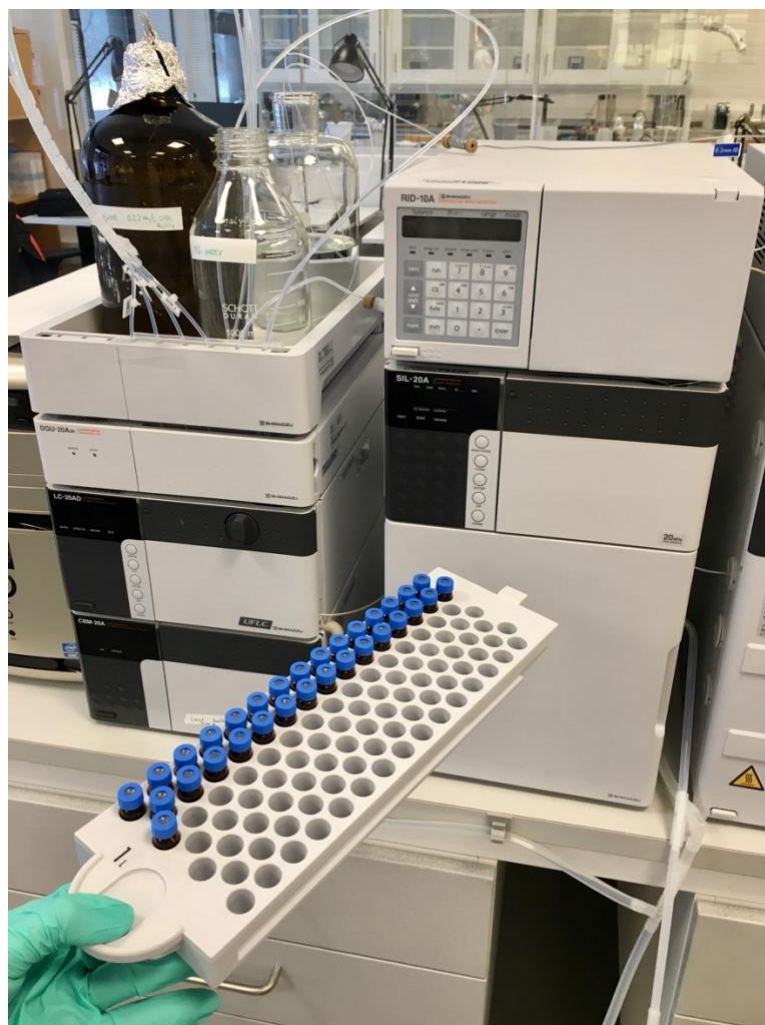


Figure 3.4. Glass bottles in a plastic tray placed in the HPLC instrument to be able to determine $SVFA$.

3.2.3 Identification of colloidal COD

The measurement of C_{tot} was conducted by first flocculating the sample and after that filtering it through 0.45 μm syringe filters as well as conducting a COD analysis on the filtrate using the cuvette method (Roeleveld & Van Loosdrecht, 2002). Due to the previously mentioned risk of clogging if using a smaller pore size, the 0.45 μm filter was used in combination with flocculation. C_{tot} is calculated by subtracting COD_{ff} from $\text{COD}_{\text{filt}, 0.45}$ as seen in Equation (6) in Table 3.4. The definition of the colloidal fraction is therefore 0.1 – 0.45 μm in this study.

3.2.4 Identification of particulate COD

X_P consists of X_B , X_U and X_{OHO} as seen in Equation (7) in Table 3.4. The analyses of particulate fractions are presented in this section.

BOD analysis to determine BCOD and X_B

To determine the biodegradable soluble COD (referred to as BCOD), BOD analyses were conducted for varying amount of days (Choi et al., 2005). In this study, analyses of BOD_1 , BOD_2 , BOD_4 , BOD_7 and BOD_9 were conducted, and the samples were analysed at an external lab, Göteborgs Kemanalyt. The method for a BOD test is according to la Cour Jansen et al. (2019) as follows; the wastewater is diluted using clean aerated water and put in a bottle. For this specific analysis, a dilution of 33 was used, with 7.5 ml wastewater sample and 250 ml dilution water. Inoculate from Rya WWTP was added to make sure that reactions are occurring. The bottle was sealed, preventing oxygen to enter it, and was thereafter placed in a dark incubator at 20°C for the specific number of days. When these days have passed, the oxygen level of the wastewater was measured and obtained in $[\text{g}/\text{m}^3]$. The composition of BCOD can be seen in Equation (8) in Table 3.4. To obtain the BCOD from the BOD analysis, Equation (9) and Equation (10) from Table 3.4 were used (Roeleveld & Van Loosdrecht, 2002). The constant k_{BOD} and the value of BOD_{tot} was identified through non-linear regression using Equation (9) and a graph of the BOD concentration together with time was created using MATLAB. Some of the biodegradable COD will be transformed into inert material and therefore the initial BCOD concentration will be higher than BOD_{tot} . The correlation factor f_{BOD} was used to obtain a correct result using the following set value:

$$f_{\text{BOD}} = 0.15 [-]$$

BCOD was calculated using Equation (10) and plotted in the same graph as BOD_{tot} . X_B was determined by subtracting S_B from the BCOD concentration as seen in Equation (11) in Table 3.4.

Long term BOD analysis to determine X_U

X_U was measured by an alternative BOD analysis conducted for approximately 10 days as described in section 3.2.2, identifying the inert particles in the wastewater. However, the particulate COD and biomass that was formed during the analyses had to be considered and the calculation can be seen in Equation (12) in Table 3.4. The yield is calculated using the

heterotrophic exchange coefficient Y_H multiplied with the endogenous residue factor f_e with the following set values:

$Y_H = 0.666 \text{ mg COD/mg COD}$ (Ekama et al., 1986; Mamais et al., 1993; Tebini, 2020; Yang et al., 2019)

$f_e = 0.08 \text{ gCOD/gCOD}$ (Dold & Marais v., 1986).

Oxygen uptake rate analysis to determine X_{OHO}

X_{OHO} was measured with a respirometry/Oxygen uptake rate (OUR) method measuring the oxygen utilisation rate (Choi et al., 2005; Wentzel et al., 1995). This measurement was conducted by using a YSI5100 Dissolved oxygen and a YSI5010 BOD Probe from YSI which can be seen in Figure 3.5. Firstly, the water was poured into a bottle and aerated for a few minutes to ensure a high concentration of oxygen in the sample. On the eighth day of analysis, the method was changed to adding ethanol to the wastewater after the aeration step. The reason for the change was due to the fact that the values obtained from the initial measurements showed a trend that was not corresponding to the references studied. The graphs indicated that there was not enough easily biodegradable organic matter in the sample for the heterotrophic bacteria to degrade and therefore, ethanol was added as a carbon source. In the aerated water with a volume of approximately 310 ml, 0.2 ml of 1% ethanol was added, and the water was mixed by shaking the bottle. The calculation deciding the amount of ethanol used can be found in Appendix II. The aerated water was poured into a glass bottle of 300 ml and the YSI5010 BOD Probe was placed in the bottle. Excess water was filled so that no air bubbles were present, and the top of the bottle and the probe was covered in laboratory film from Parafilm to mitigate the risk of oxygen exchange. The outside of the bottle was cleaned with chlorine to prevent contamination. Thereafter, the bottle was placed in a water bath to ensure a constant temperature of 20 °C since an increase in temperature generally leads to an increase in degradable activity (Hagman & Jansen, 2007).



Figure 3.5. Instruments and water bath used for the OUR measurement.

If the dissolved oxygen level in the water decreased to 4 mg/L, the water was transferred to a 0.5 L glass bottle which was shaken to add oxygen and thereafter the water was returned to the original bottle and the experiment continued. The oxygen level uptake was measured every 15 second and the data was transferred to a computer using a RS232 to USB cable with a USB-C adapter to a MacBook Pro computer. Since the YSI1000 instrument had a maximum measurement time of one hour, the instrument was restarted every hour. The data was processed using a software from Apple called Serial and later transferred to Excel. The experiment was conducted for 3.5 hours to make sure to capture the drop in oxygen uptake rate when all the easily biodegradable organic matter had been consumed. The reason for the time chosen was the delimitation in choice of measurement instrument and the time available. When the test was finished, 0.4 ml of 4 M sulphuric acid (H_2SO_4) was added to 40 ml of the wastewater and put in the fridge keeping a temperature of 4°C (Arnell et al., 2016). The next day, the sample was analysed using the cuvette method.

The result from the OUR – measurement was plotted with time [minutes] on the x-axis and OUR [mg O_2 /L/h] on the y-axis. The OUR was calculated using the Excel function SLOPE using 25 values, from the measured dissolved oxygen concentration [mg O_2 /L] and the time

[minutes]. The characteristic drop in the OUR was identified and an ln-graph of the OUR values up until the drop was created with a trend line that was adjusted to the values. The heterotrophic biomass was calculated using Equation (13) from Wentzel et al. (1995) found in Table 3.4, with the following set values for the heterotrophic exchange coefficient Y_H and the specific heterotrophic degradation rate b_H :

$$Y_H = 0.666 \text{ mg COD/mg COD}$$

$$b_H = 0.62/\text{day}$$

SDR SensorDish® Reader to determine biodegradable organic matter

Another type of OUR measurement was conducted using an optic sensor where samples were analysed through an SDR SensorDish® Reader. This measurement was conducted to obtain the biodegradable organic matter after three days, referred to as BOD_{SDR} in this study. The oxygen concentration was measured through integrated sensors spots in glass vials as can be seen in Figure 3.6 (Olafsdottir, 2021). This analysis resulted in graphs showing the oxygen degradation [mg O₂/L] over time [minutes] and is an alternative BOD analysis of the wastewater. There were two analyses made. The first one included unfiltered, filtered, flocculated and filtered influent water and flocculated and filtered water after the nitrifying MBBR step and was conducted for four days. The filtering was in all cases conducted using a 0.45 µm filter. The second analysis was focusing on the filtered wastewater with different dilution factors. The samples were diluted using Phosphate Buffered Saline (PBS) and two replicas of each dilution were included. The first two samples contained 50% PBS and 50% filtered wastewater. The second contained 50% PBS and 50% of the previous diluted sample and so on, with specific concentrations presented in Appendix III. In total there were nine different samples with two vials for each sample, as well as two vials containing a 100% PBS solution.

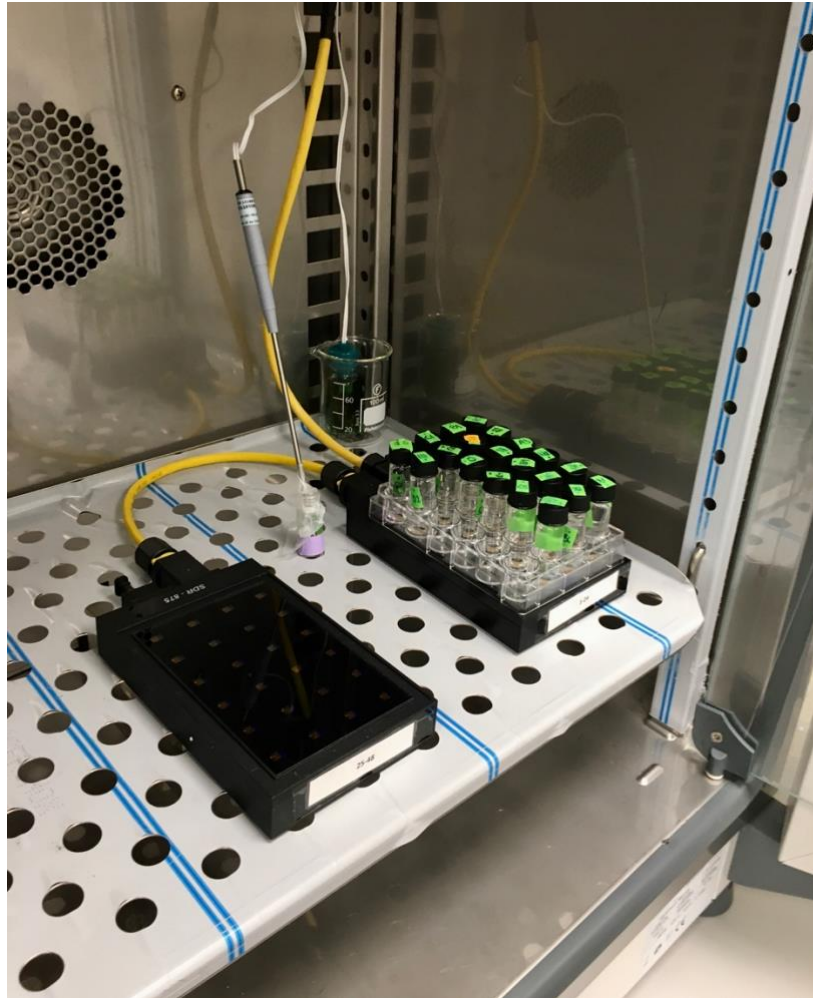


Figure 3.6. OUR measurement using an SDR SensorDish® Reader.

To be able to calculate the BOD_{SDR} concentration from the SensorDish® Reader analysis, the minimum and maximum DO concentration of each sample was identified, and the difference was obtained. This was multiplied with the dilution factor resulting in the concentration of BOD_{SDR} .

3.2.5 Measurement of phosphorus fractions

Table 3.5 presents the relationships and equations for the fractions of phosphorus used in this study.

Table 3.5. Equations and relationships used for the phosphorus measurements.

Parameter	Relationship	Equation notation
Total P	$TP = S_P + X_P$	(14)
Soluble P	$S_P = PO_4 + S_{P,other}$	(15)
Particulate P	$X_P = TP - S_P$	(16)

Analyses of phosphorus fractions were conducted at Rya WWTP on the influent and effluent wastewater. An autoclave, Hach DR6000, was used at 120°C for 30 minutes measuring the TP concentration. TP is divided into S_P and X_P , as seen in Equation (14) in Table 3.5.

To be able to identify S_P , TP was filtered using a 1.6 μm filter. PO_4 was measured by filtering the water through 0.45 μm filter and using a Hach DR6000 spectrophotometer at 890 nm. The reactive soluble fraction of phosphorus mainly consists of PO_4 , but also other soluble phosphorus as seen in Equation (15) in Table 3.5. To calculate the other soluble phosphorus, PO_4 was subtracted from the total soluble phosphorus. X_P was calculated by subtracting S_P from TP as seen in Equation (16) in Table 3.5.

3.2.6 Measurement of nitrogen fractions

Table 3.6. Presents the relationships and equations for the fractions of nitrogen used in this study.

Table 3.6. Equations and relationships used for the phosphorus measurements.

Parameter	Relationship	Equation notation
Total N	$\text{TN} = S_N + X_N$	(17)
Soluble N	$S_N = S_{\text{NH}_4} + S_{\text{NO}_2}/S_{\text{NO}_3} + S_{\text{N,other}}$	(18)
Particulate N	$X_N = \text{TN} - S_N$	(19)

Analyses of nitrogen fractions were conducted at Rya WWTP on the influent and effluent wastewater. TN was measured by oxidising to nitric oxide (NO) within the span of 0.5 – 60 mg/L using a standard from SIS (2004). Equation (17) in Table 3.6 shows the composition of the total nitrogen.

Measurements of S_{NH_4} , S_{NO_2} and S_{NO_3} were conducted using a standard from the International Organization for Standardization (ISO) (2013). An AQ400 instrument from SEAL was used, as well as measurements through a spectrophotometer. The water was measured after filtering through a 1.6 μm glass fibre filter. For $S_{\text{NO}_2}/S_{\text{NO}_3}$, the spectrophotometer was set to 520 nm and for S_{NH_4} , it was set to 600 nm.

The composition of the soluble nitrogen can be seen in Equation (18) in Table 3.6, with $S_N = \text{TN}_{\text{filt}}$. $S_{\text{N,other}}$ was calculated by subtracting S_{NH_4} , $S_{\text{NO}_2}/S_{\text{NO}_3}$ from S_N . X_P was calculated by subtracting S_P from the TN, meaning that this is including the colloidal fraction, as seen in Equation (19) in Table 3.6.

4. Results and discussion

This section will present and discuss the results from the laboratory work analysing organic matter, nitrogen, and phosphorus at the laboratory at Chalmers University of Technology and at Rya WWTP. There will also be a discussion of the literature review and background, connected to the findings from the laboratory work.

4.1 Results from COD analyses

The results from the COD measurements are divided into sections of different fractions with a compiled table of all results from the analyses of influent wastewater presented in Appendix IV. The results are presented with the date corresponding to the daily average sampling occurring at 7.00 am – 7.00 am.

4.1.1 Total COD and other parameters

A comparison between the different measurements of COD concentrations is shown in Figure 4.1 with [mg COD/L] on the y-axis. The concentrations of COD_{tot} show the overall highest COD concentrations. Second and third highest are $COD_{filt,1.6}$ and $COD_{filt,0.45}$ and thereafter comes COD_{ff} as well as $MBBR_{ff}$ with the lowest concentrations.

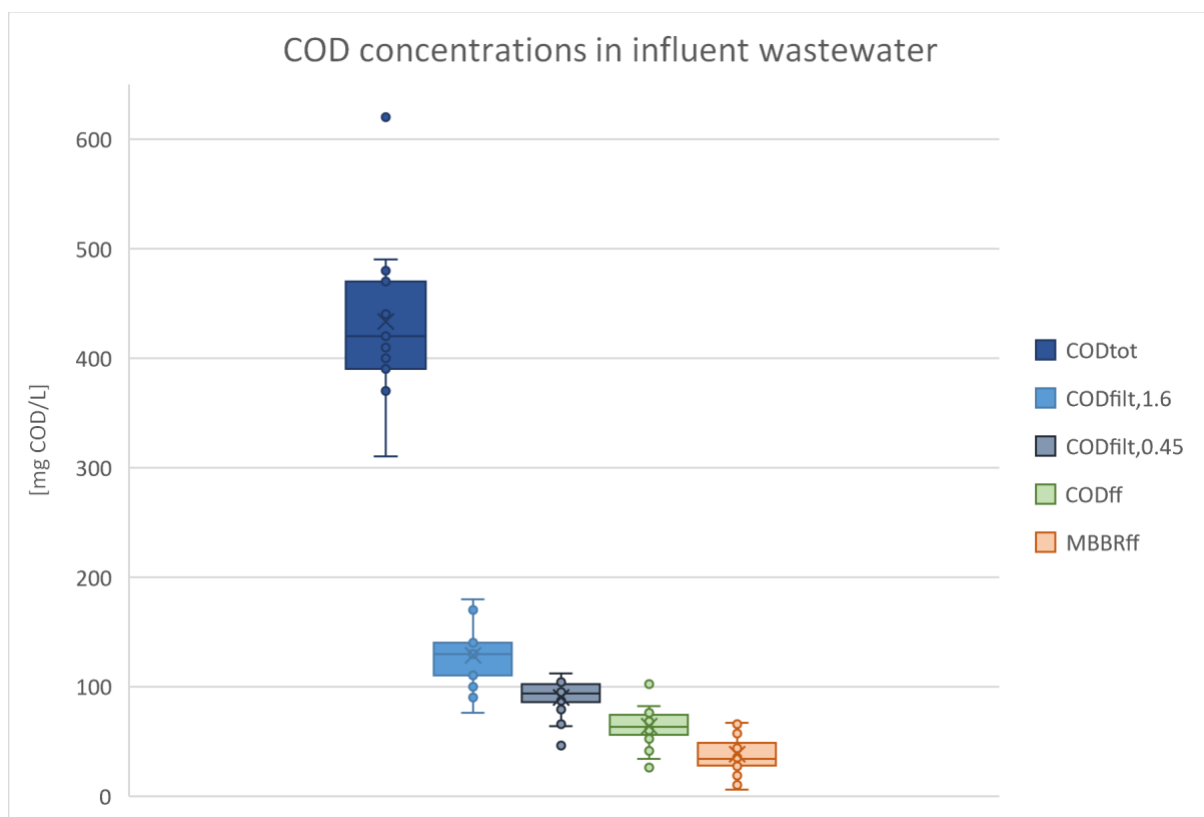


Figure 4.1. Comparison between different measurements of COD concentrations in influent wastewater.

Figure 4.2 presents the inflow of water to Rya WWTP during the sampling period with [m^3/s] on the y-axis and [date] on the x-axis. It shows a large peak in the end as well as a smaller increase in the beginning.

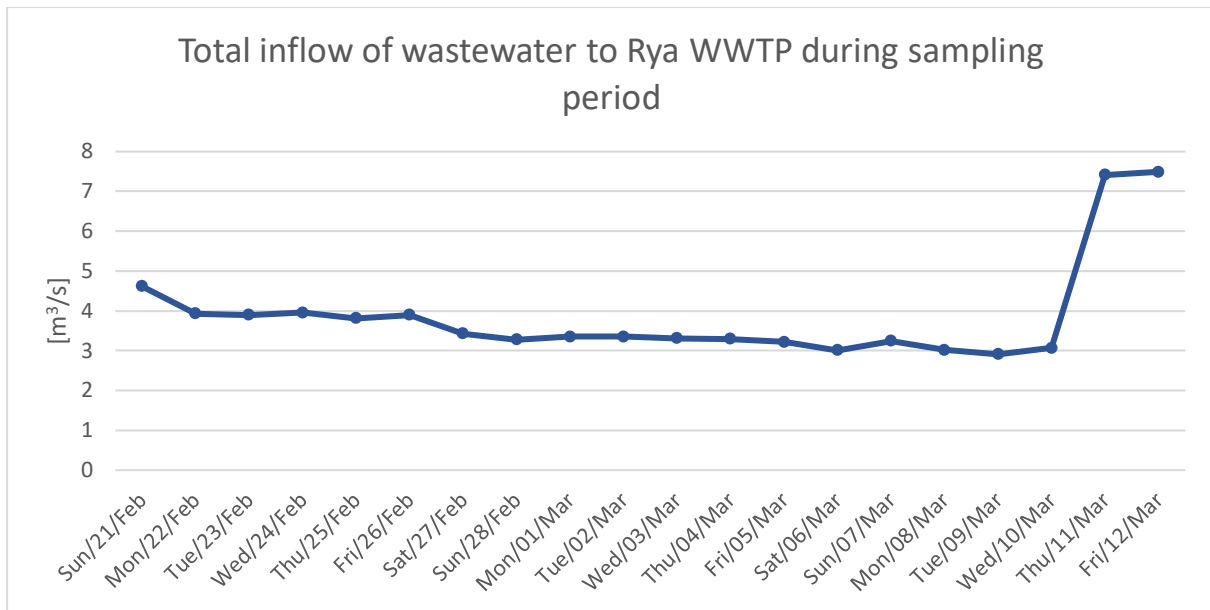


Figure 4.2. Daily average flow to Rya WWTP during sampling period with data from Gryaab AB.

Figure 4.3 presents the precipitation during the sampling period with data from Gryaab AB, with [mm] on the y-axis and [date] on the x-axis. The data can be compared to data from SMHI found in Appendix V.

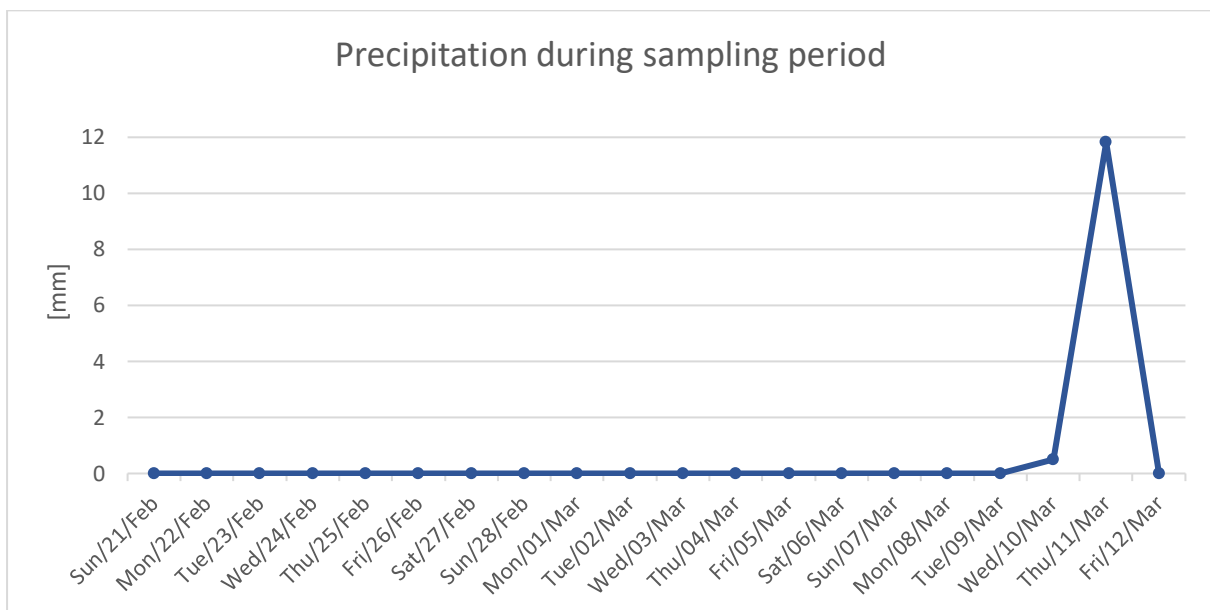


Figure 4.3. Precipitation during sampling period with data from Gryaab AB.

Figure 4.4 presents the water temperature of the effluent water from Rya WWTP with data provided by Gryaab AB with [°C] on the y-axis and [date] on the x-axis. As can be seen, the temperature is lower in the beginning and end of the sample period, in line with the precipitation and snow melt seen in Appendix V.

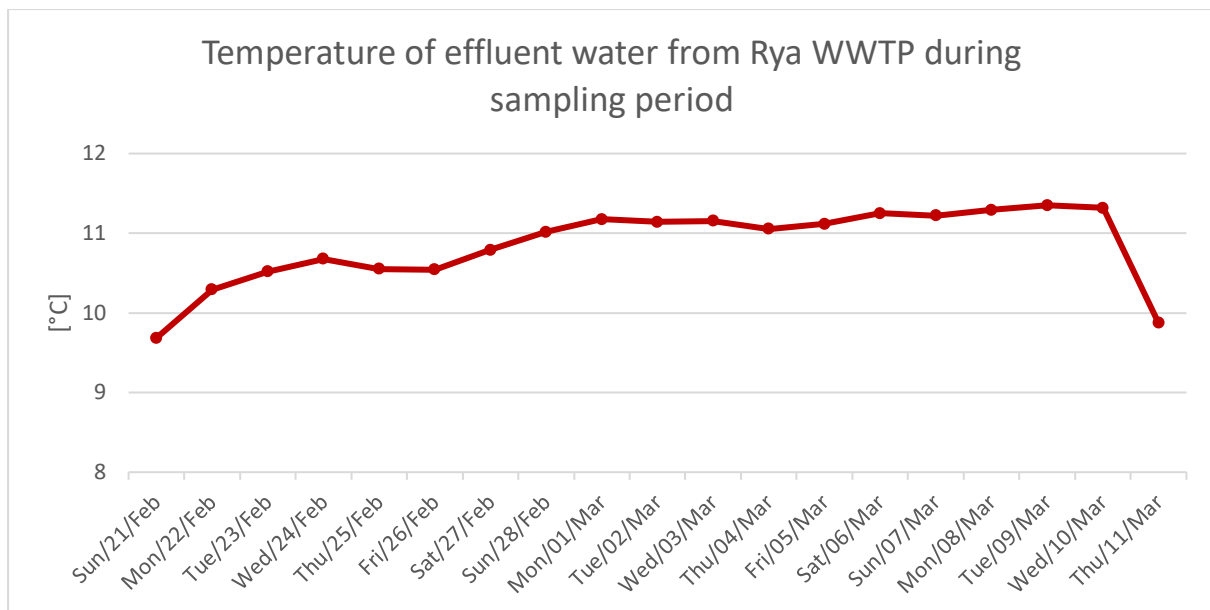


Figure 4.4. Temperature of the effluent water from Rya WWTP during sampling period with data from Gryaab AB.

Graphs of the different COD measurements are shown in Figure 4.5a-4.5d with [mg COD/L] on the y-axis and [date] on the x-axis. The different series in Figure 4.5b-4.5d represent the two duplicates. Specific data can be found in Appendix IV where there are two rows showing the COD concentration from each replica. The results confirm that precipitation and snow melting influence the concentrations of substances reaching the sewer systems, with a decrease of pollutants following an increase of precipitation. During the sampling period there is a peak in the precipitation and snow melting right before the sampling begins as well as in the end of the sampling period. According to SMHI, (n.d.), there was heavy rainfall at around 17th – 18th of February, as well as on the 11th – 12th of March with snow melting afterwards. This can be seen in Appendix V. At the same time, the results from the laboratory work shows a general decrease in concentrations of organic matter in the influent wastewater, as seen in Figure 4.5b-4.5d. The concentration of COD_{tot} in the influent wastewater shows no distinct decrease in the end of the sampling period seen in Figure 4.5a. This could be related to the *first flush* concept and the fact that a large part of COD_{tot} consists of particulate matter, discussed further in section 4.1.4. Another parameter that can have an impact on the result is the water temperature. As can be seen Figure 4.4, the temperature decreases with an increase of precipitation and snow melting.

Figure 4.5a shows a clear pattern of increases on Mondays of each week. This can be connected to the flushing of the pipes which is often conducted on Mondays at Rya WWTP. Water is saved up in the tunnels and through flushing, material that is stuck to the walls will be transported with the water. The increase of MBBR_{ff} concentration in the middle of the sample period seen in Figure 4.5d could be related to inaccuracies in the laboratory method since it is not connected to any external factors such as precipitation or temperature. Since the filtrate was extracted using a syringe, the flocculated organic matter could possibly have been extracted as well, leading to an overestimation in the concentration.

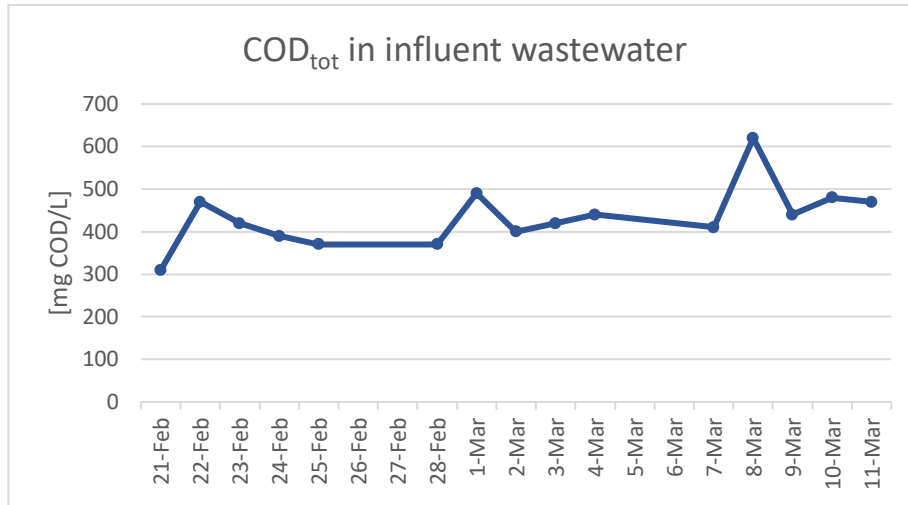


Figure 4.5a. COD concentrations of unfiltered influent wastewater.

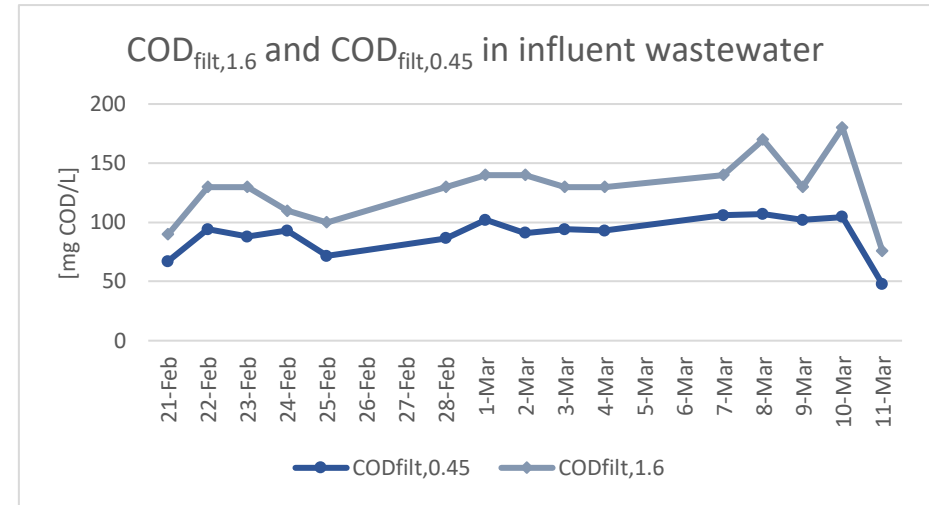


Figure 4.5b. COD concentrations of filtered influent wastewater.

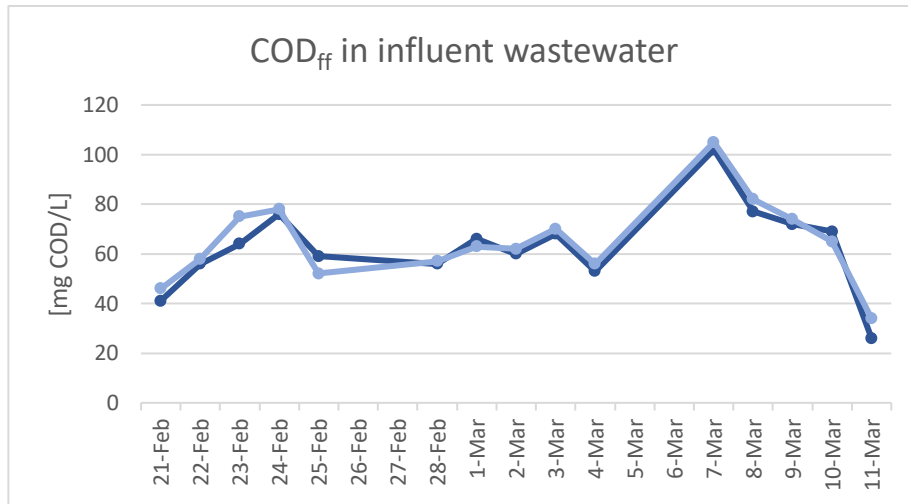


Figure 4.5c. COD concentrations of flocculated and filtered influent wastewater.

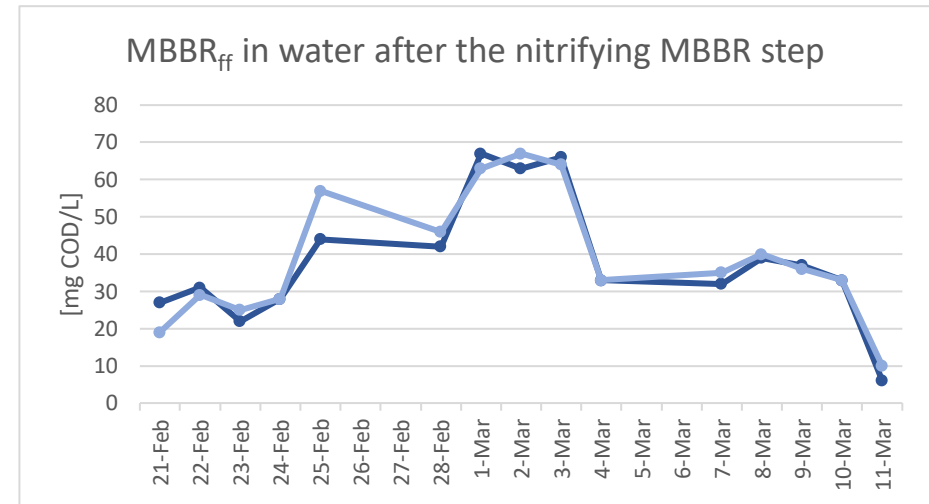


Figure 4.5d. COD concentrations of flocculated and filtered water after the nitrifying MBBR step.

Suspended solids (TSS and VSS)

The results from the analyses of TSS and VSS can be seen in Figure 4.6, where VSS was calculated by subtracting the non-volatile solids from the TSS. Specific values can be found in Appendix VI. The concentration on the 24th of February is deviating compared to the rest of the days. However, there is no connection between the high concentration of TSS and VSS and precipitation or temperature and there were no other disturbances causing the increase of concentration. TSS concentrations could have thought to be larger in the end of the sampling period with the argument of *first flush* further discusses in section 4.1.4. There is an increase of TSS, however, it is only noticeable larger than concentrations from previous days.

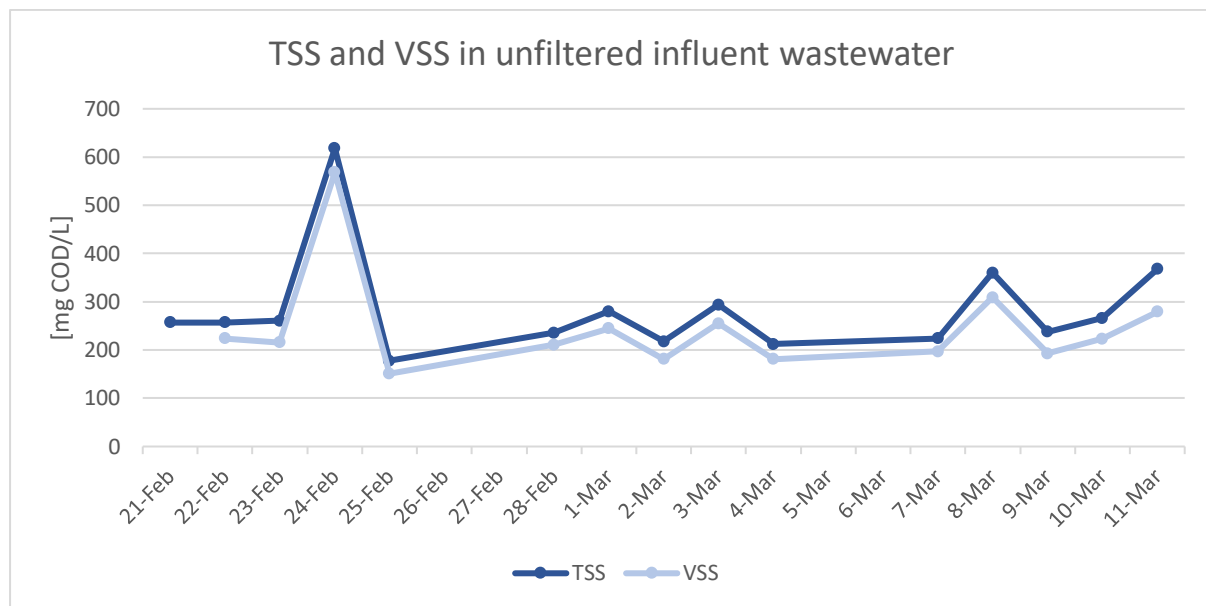


Figure 4.6. TSS and VSS concentrations in unfiltered influent wastewater.

4.1.2 Soluble COD (S_{tot})

The total soluble fraction can be found in Appendix IV where COD_{ff} represents S_{tot} . Figure 4.5c presents the concentration of S_{tot} during the sampling period with [mg COD/L] on the y-axis and [date] on the x-axis.

Inert soluble COD (S_U)

The results of the S_U fractions based on the long-term BOD test and the corresponding S_U fractions based on flocculated and filtered COD analysis of water after the nitrifying MBBR step are found in Table 4.1. Appendix VIIa shows the values used for calculating S_U by using average values from the long-term BOD test. Appendix VIIb show calculations of S_U using values from the MBBR_{ff} measurement and as can be seen, the values calculated from MBBR_{ff} has a large standard deviation meaning a greater uncertainty. Therefore, the values and calculations from the long-term BOD test are used for the main part of fraction S_U . Although, on the 11th of March there is a high concentration of S_U calculated from the long-term analysis, larger than to S_{tot} . For this date, the concentration of S_U calculated using the MBBR_{ff} is used.

Table 4.1. Results of S_U fractions based on the long-term BOD test and calculations using $MBBR_{ff}$.

Date	Influent wastewater		Water after the nitrifying MBBR step ¹	
	S_U [mg COD/L] (results from long-term BOD test)	Average S_U [mg COD/L]	S_U [mg COD/L] (results from calculations using $MBBR_{ff}$)	Average S_U [mg COD/L]
21-Feb	28	30	24.1	20.1
21-Feb	32		16.1	
04-Mar	35	34.5	32.5	32.5
04-Mar	34		32.5	
11-Mar		41.2		8.0 ²
Standard deviation including further values found in Appendix VIIb		6.05		15.6

¹These values are generally not used for S_U in this study since the standard deviation is large compared to the long-term BOD tests.

²On the last day of sampling, the concentration of S_U calculated from the long-term test is too large compared to S_{tot} and is therefore replaced with the calculation of S_U using $MBBR_{ff}$.

Readily degradable COD (S_B)

Concentrations of S_B are presented in Table 4.2 by subtracting S_U from S_{tot} . Concentrations of the remaining dates are found in Appendix VIIc. The value of S_U used in the calculation is an average value from the long-term test. Although, on the 11th of March, S_B is calculated using concentrations of S_U calculated from $MBBR_{ff}$ as seen in Table 4.1. Since there are only two values from the long-term BOD test available for calculations of S_B , there are as large uncertainties with the results as for S_U .

Table 4.2. Results from the calculation of readily degradable COD.

Date	Flocculated and filtered influent wastewater		Inert soluble	Readily degradable
	S_{tot} [mg COD/L]	Average S_{tot} [mg COD/L]	Average S_U [mg COD/L]	S_B [mg COD/L]
21-Feb	41	43.5	30	13.5
21-Feb	46			
04-Mar	53	54.5	34.5	20
04-Mar	56			
11-Mar	26	30	8.0 ¹	22.0
11-Mar	34			

¹The concentration calculated using $MBBR_{ff}$ is used at this date.

Volatile Fatty Acids (S_{VFA})

Figure 4.7 presents $S_{Acetate}$ for influent wastewater and water after the nitrifying MBBR step at Rya WWTP since there was mainly acetate identified in the HPLC analysis. Butyrate and propionate were included in the analysis but for butyrate there were deviating values and therefore, this parameter is not included in the result. Specific concentrations of COD for the different VFAs are presented in Appendix VIII. There is still some acetate in the water after the nitrifying MBBR step. This was not expected since after the nitrifying MBBR step almost all easily available organic matter should have been consumed and therefore no S_{VFA} should be present. As can be seen in Figure 4.7, the result is not altogether accurate where the concentration in the water after the nitrifying MBBR step at some dates exceeds the concentration in the influent water. There are decreases of $S_{Acetate}$ in the beginning and end of the sampling period and using the same arguments as for COD which could be explained by the larger precipitation rate and snow melting at this time. Overall, the results from this analysis should be used carefully since the method can be considered uncertain.

Calculations of S_{other} are shown in Appendix IX calculated using S_{tot} , S_U and $S_{Acetate}$. On the 22nd of February where the concentration of S_U is exceeding S_{tot} , the concentration of S_{other} is set to zero. On the 11th of March, S_{other} is calculated using concentrations of S_U calculated from $MBBR_{ff}$.

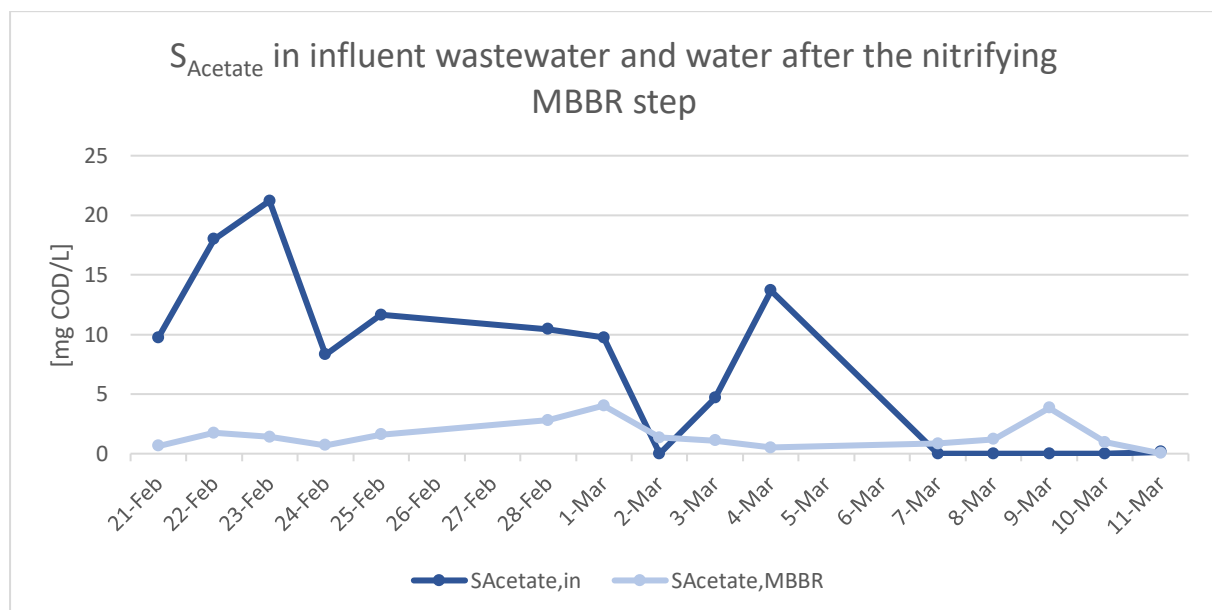


Figure 4.7. $S_{Acetate}$ in influent wastewater and water after the nitrifying MBBR step.

4.1.3 Colloidal COD (C_{tot})

The result from subtracting the COD of the flocculated and filtered water from the COD of the filtered water shows the colloidal fraction of the wastewater and is found in Figure 4.8 where an average of S_{tot} is subtracted from an average of $COD_{filt, 0.45}$. Specific concentrations can be found in Appendix X. The concentration 7th of March was low compared to the other dates and was therefore replaced by a calculation using the average percentage fraction of 6% from the other dates which can be seen by the dashed line in Figure 4.8. This can be connected to

potential errors made in the laboratory, e.g. when the supernatant was extracted using a syringe. By doing so, the flocculate might have been extracted by mistake.

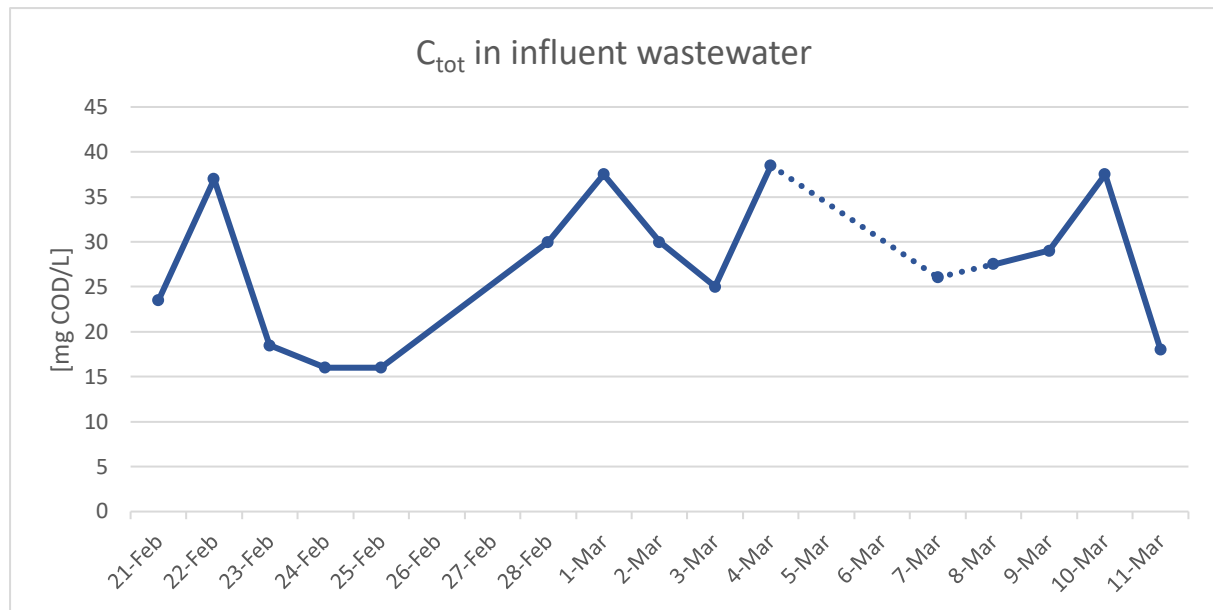


Figure 4.8. Colloidal COD concentration of the influent wastewater. The dashed line represents the date on which the concentration was calculated using an average percentage fraction of the remaining days. This was due to a deviating value.

4.1.4 Particulate COD (X_{tot})

There is an increase of X_{tot} in the end of the sampling period as seen in Figure 4.9 with specific values found in Appendix XI. This can be connected to the high precipitation rate during this time and the *first flush*, meaning a greater concentration of particulates being transported with the stormwater in the beginning of a rain event (Viklander et al., 2019). As can be seen, similar to COD_{tot} presented in Figure 4.5a, there is an increase of COD every Monday connected to the flushing of the tunnels.

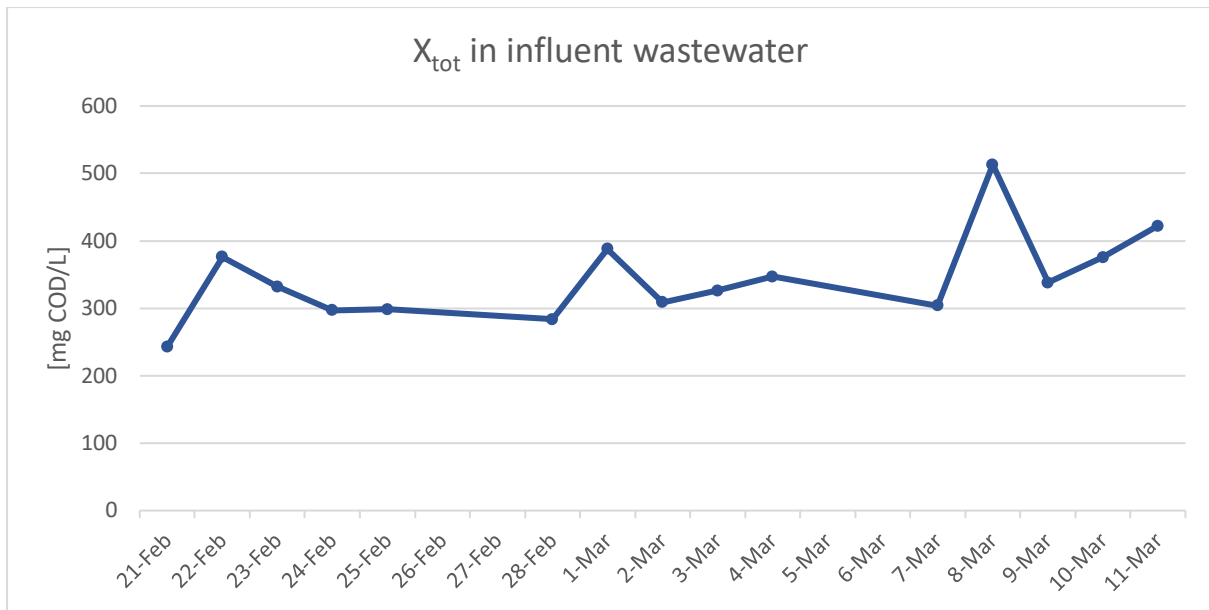


Figure 4.9. Particulate COD concentration of the influent wastewater.

Biodegradable soluble COD (BCOD)

Figure 4.10 presents the graph where k_{BOD} , BOD_{tot} and BCOD for Sample 1 from the 7th of March are obtained, with [mg O₂/L] on the y axis and [days] on the x axis. The horizontal magenta line represents the BCOD concentration for the specific date. Further graphs and specific concentrations from 8th – 11th of March can be found in Appendix XII. There were two BCOD – analyses made and in between the analyses the samples were frozen. The first analysis resulted in values that were not considered accurate. In the second analysis, a modification of the dilution was made and resulted in a dilution of 7.5 wastewater sample and 250 ml dilution water used. The uncertainty of this measurement of 10 – 20% has to be taken into account when analysing the results (Roeleveld & Van Loosdrecht, 2002). This together with the potential impact on freezing the samples in between analyses.

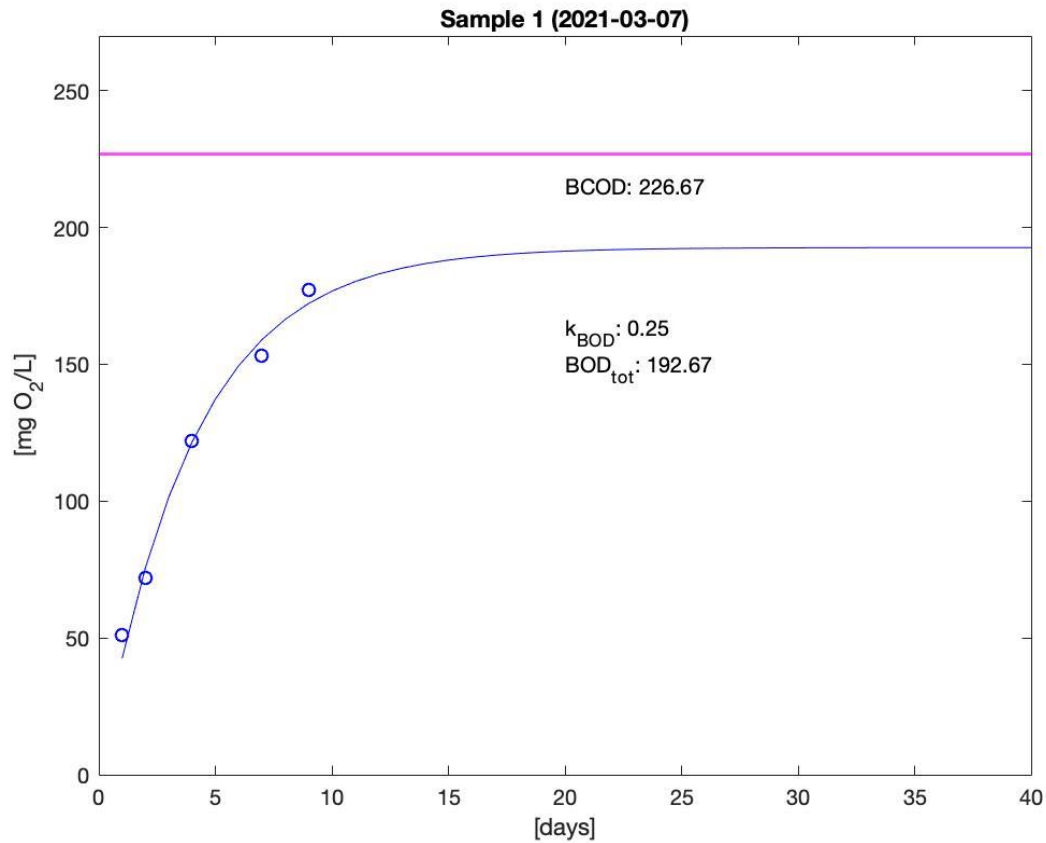


Figure 4.10. Identification of k_{BOD} , BOD_{tot} and $BCOD$ for the 7th of March using MATLAB.

The results from the $BCOD$ – analysis are shown in Table 4.3. There is a decrease of $BCOD$ at the end date compared to the beginning, which can be connected to the increase of precipitation in the end of the sampling period.

Table 4.3. Concentrations of k_{BOD} , BOD_{tot} and $BCOD$ from the BOD -analysis.

Date				
	Sample notation	BOD_{tot} [mg/L]	k_{BOD}	$BCOD$ [mg/L]
07-Mar	1	192.7	0.25	226.7
08-Mar	2	311.6	0.20	366.6
09-Mar	3	195.2	0.24	229.6
10-Mar	4	222.9	0.27	262.2
11-Mar	5	157.7	0.25	185.5

SDR SensorDish® Reader

Table 4.4 presents the concentration of BOD_{SDR} of four samples from the SDR SensorDish® Reader analysis. Samples 3 – 6 were assumed to be the most representative for the BOD concentrations, in this study referred to as BOD_{SDR} . Further BOD_{SDR} concentrations obtained from the measurement are presented in Appendix XIII.

Table 4.4. Degradable BOD in four samples from the SDR SensorDish® Reader analysis.

Date 12/3						
Sample	Dilution factor	PBS	Max DO [mg O ₂ /L]	Min DO [mg O ₂ /L]	Difference in DO [mg O ₂ /L]	Degradable BOD (BOD _{SDR}) [mg BOD/L]
3	1/4	3/4	8.9	3.4	5.4	21.7
4	1/4	3/4	8.6	3.7	4.9	19.6
5	1/8	7/8	8.7	5.2	3.5	28.2
6	1/8	7/8	8.9	5.6	3.3	26.2

The results from the SDR SensorDish® Reader are presented in Figure 4.11a-b with dissolved oxygen in [mg O₂/L] on the y-axis and time [hours] on the x-axis. The concentration of biodegradable organic matter obtained using the SDR SensorDish® Reader was not compared to results from previous studies since it is a relatively new instrument and not widely used. However, a possible future use is to compare the results of this analysis, BOD_{SDR}, with results from the BCOD – analysis, if samples from the same dates are used.

The increase of O₂ in the beginning of the analysis is probably due to an initial adjustment of the instrument. Thereafter, the degradation is faster during the start of the analysis, connected to the amount of readily degradable organic matter. The stabilisation of O₂ can be explained by the absence of organic matter that is available to the bacteria. The low concentration compared to the concentration of BCOD is due to the filtration of the samples. If unfiltered samples would have been analysed instead, there would have been more organic matter present, and the consumption of organic matter would have been faster. However, since the SDR SensorDish® Reader relies on optic measurements of dissolved oxygen concentrations, too turbid samples cannot be used.

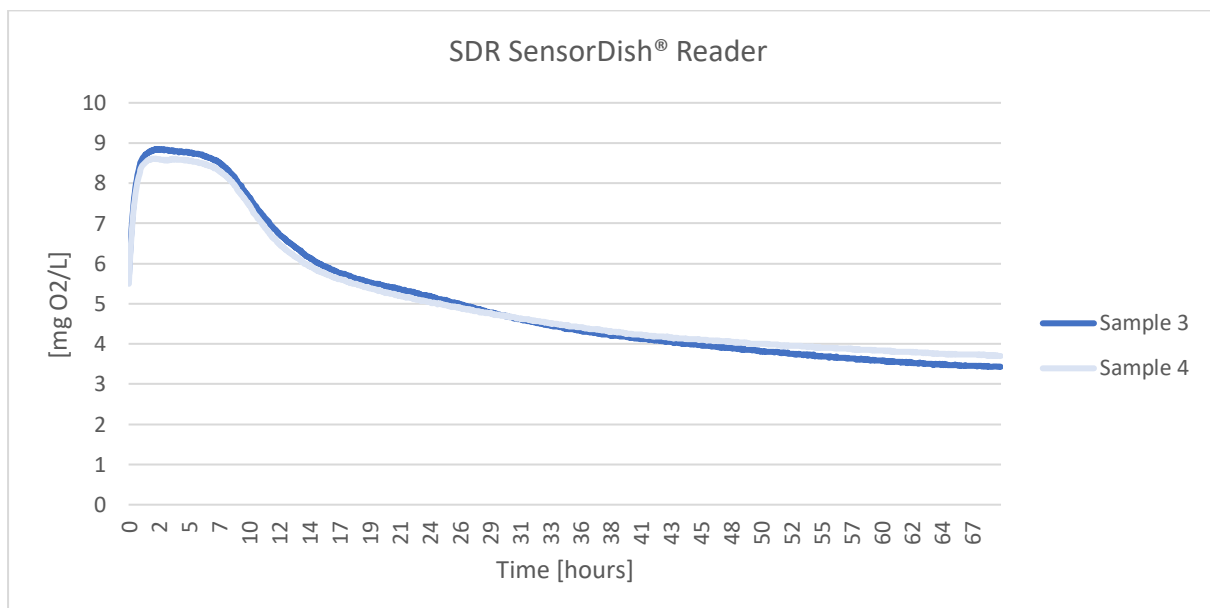


Figure 4.11a. Result of biodegradable organic matter from the SDR SensorDish® Reader analysis with dilution factor 1/4.

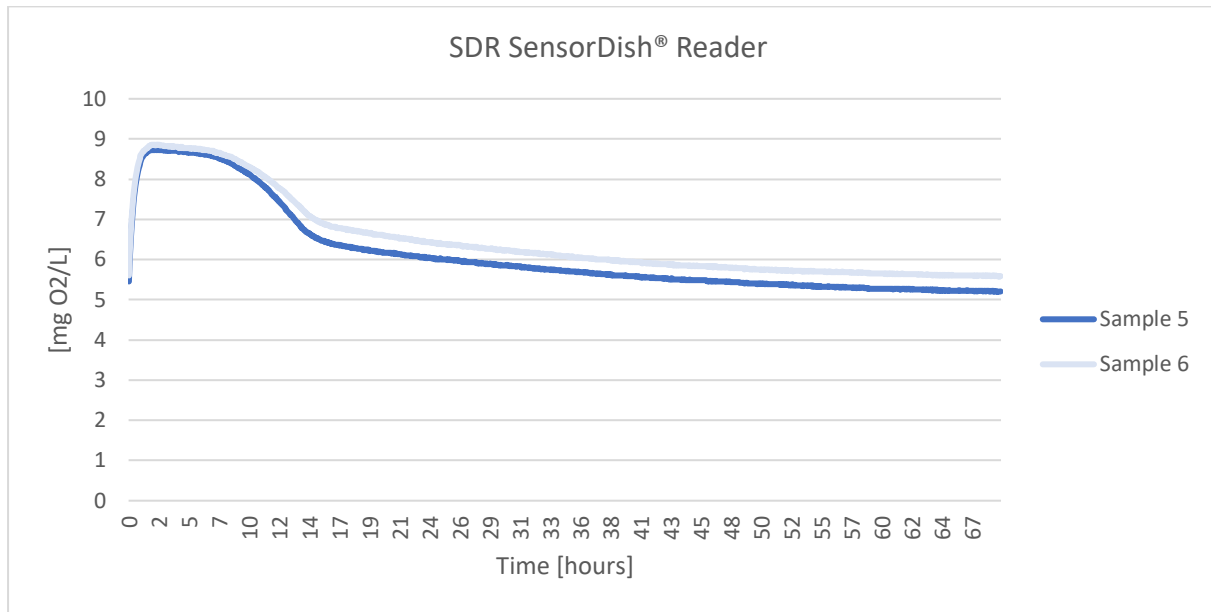


Figure 4.11b. Result of biodegradable organic matter from the SDR SensorDish® Reader analysis with dilution factor 1/8.

Inert particulate COD (X_U)

The concentrations of X_U obtained from calculations of the long-term BOD test can be found in Table 4.5 with specific values presented in Appendix XIV. The replicates of the unfiltered sample had a large variation in total COD concentrations which could be due to the possibilities of unevenly distributed particulate matter present in the test. This could lead to overestimations of COD concentrations when using the average concentration to estimate the remaining concentration.

Table 4.5. Calculated results of X_U fractions based on the long-term BOD test.

Date	Unfiltered wastewater		Inert particulate
	COD _{tot,10 days} [mg COD/L]	Average COD _{tot,10 days} [mg COD/L]	Calculated X_U [mg COD/L]
21-Feb	132	160	114.1
21-Feb	188		
04-Mar	172	164	106.5
04-Mar	156		
Standard deviation	23.9	2.8	20.4

Slowly degradable COD (X_B)

The concentration of X_B is presented in Figure 4.12 where the concentrations were obtained from the BCOD – analysis, minus S_B . Since there are only BOD results for 7th – 10th of March, the remaining concentrations of X_B are calculated based on an average percentage fraction of 52% from the analysed samples of BCOD. This is represented with the dashed line in the graph. The concentration for the 11th of March is uncertain due to uncertainties with S_B used for

calculating X_B . The COD_{tot} concentration is high on the 8th of March, 620 mg COD/L, leading to a high concentration of X_B . Specific concentrations of X_B and the calculations using concentrations of BCOD and S_B are found in Appendix XV.

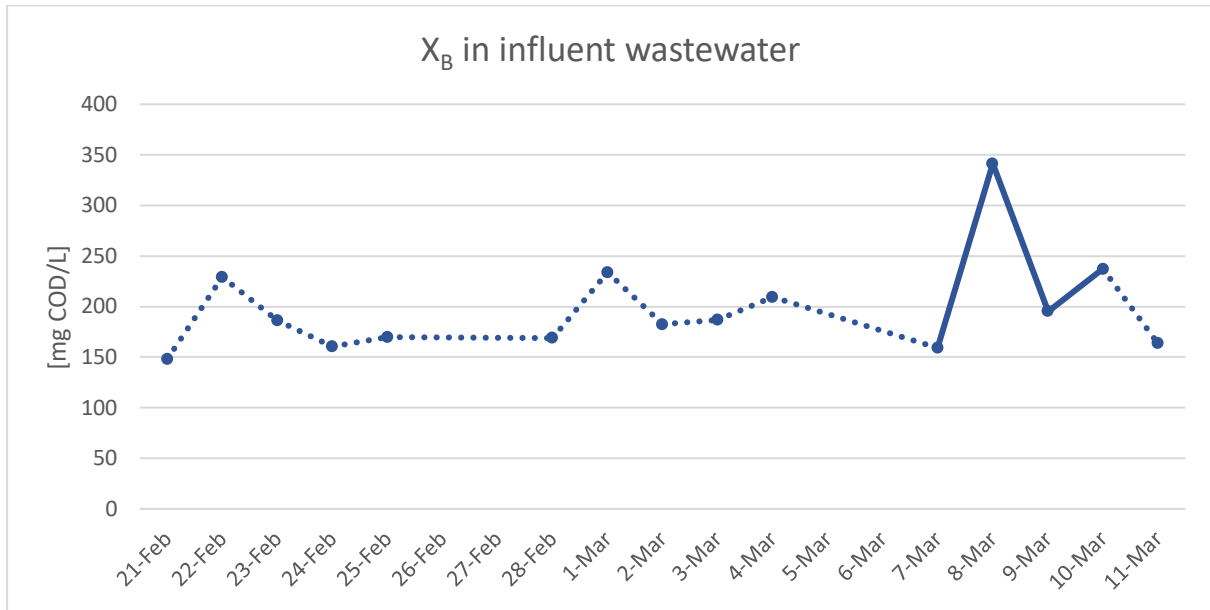


Figure 4.12. Concentrations of X_B in influent wastewater.

Heterotrophic biomass (X_{OHO})

Figure 4.13 shows the OUR results from a sample with added ethanol (blue) and without ethanol (orange) with OUR [mg O₂/L/h] on the y-axis and time [minutes] on the x-axis. The comparison between samples with and without adding ethanol leads to the conclusion that there was not enough easily degradable organic matter naturally occurring in the influent wastewater for this method of determining the fraction of heterotrophic biomass. The results from the samples with added ethanol show an increase of oxygen uptake rate in the beginning of the measurement indicating that the organic matter has already been degraded before starting the analysis. This shows the importance of conducting the analyses as quick as possible after sampling to mitigate the risk of missing reactions occurring in the sample. By adding ethanol as a food source, the oxygen uptake rate will increase.

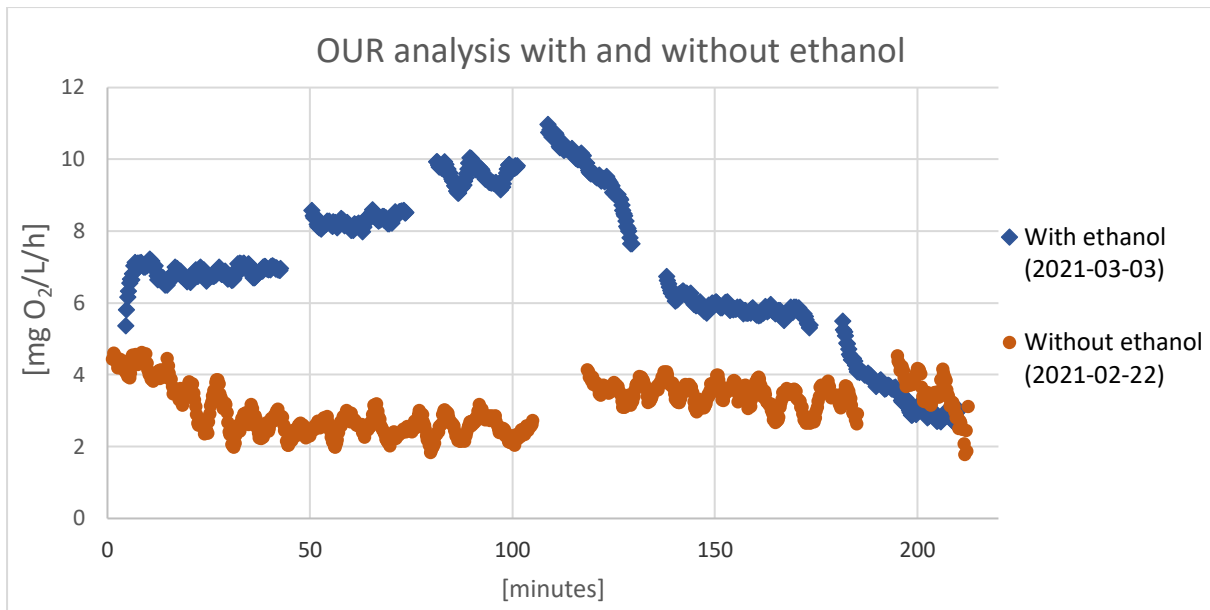


Figure 4.13. OUR analysis with and without ethanol.

Figure 4.14 presents the OUR concentrations during the OUR – measurement with [mg O₂/L/h] on the y-axis and time in [minutes] on the x-axis. The first smaller increase is assumed to be readily available organic matter that quickly gets consumed by the microorganisms whereas the second increase is assumed to appear due to the addition of ethanol to the wastewater which was a trend throughout all analyses. When all ethanol (the readily available organic matter) is consumed, there is only slowly degradable organic matter left in the sample. Figure 4.15 presents how X_{OHO} was calculated using a ln-curve obtained from the increase of OUR in Figure 4.14, between 21.25 and 47.75 minutes. The y-axis is in the unit [ln (OUR)] the x-axis is in the unit [hours].

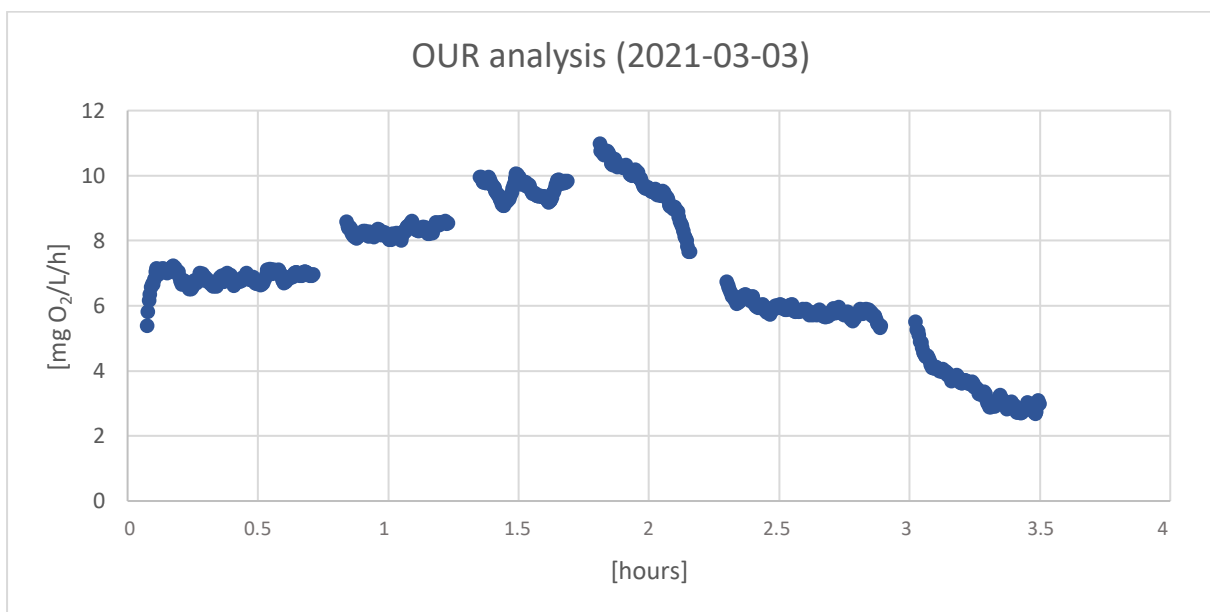


Figure 4.14. OUR analysis of wastewater with added ethanol.

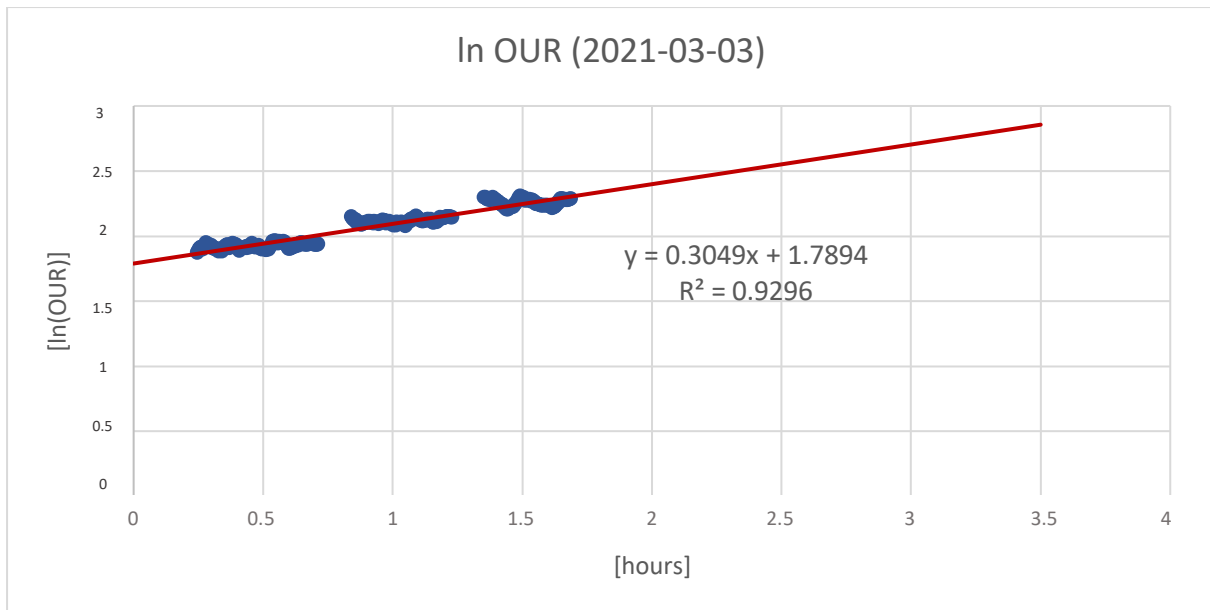


Figure 4.15. ln-curve of the OUR-curve excluding the first increase of OUR.

Figure 4.16 presents the concentrations of X_{OHO} in the unfiltered influent wastewater with specific values found in Appendix XVI. Since the first seven analyses were conducted without ethanol and are not considered useful, the concentrations are instead calculated from an average percentage value of 9% based on the OUR analyses made with ethanol addition. This is also the case for the concentration on the 7th of March where no ethanol was added. On this date, the time from sampling to analysis was only one hour, compared to the remaining dates where the time was approximately two hours. This analysis was conducted to get a further understanding of the impact of the time between sampling and start of the analysis. Due to time constraints, it was not possible to deliver the remaining samples as early as on the 7th of March.

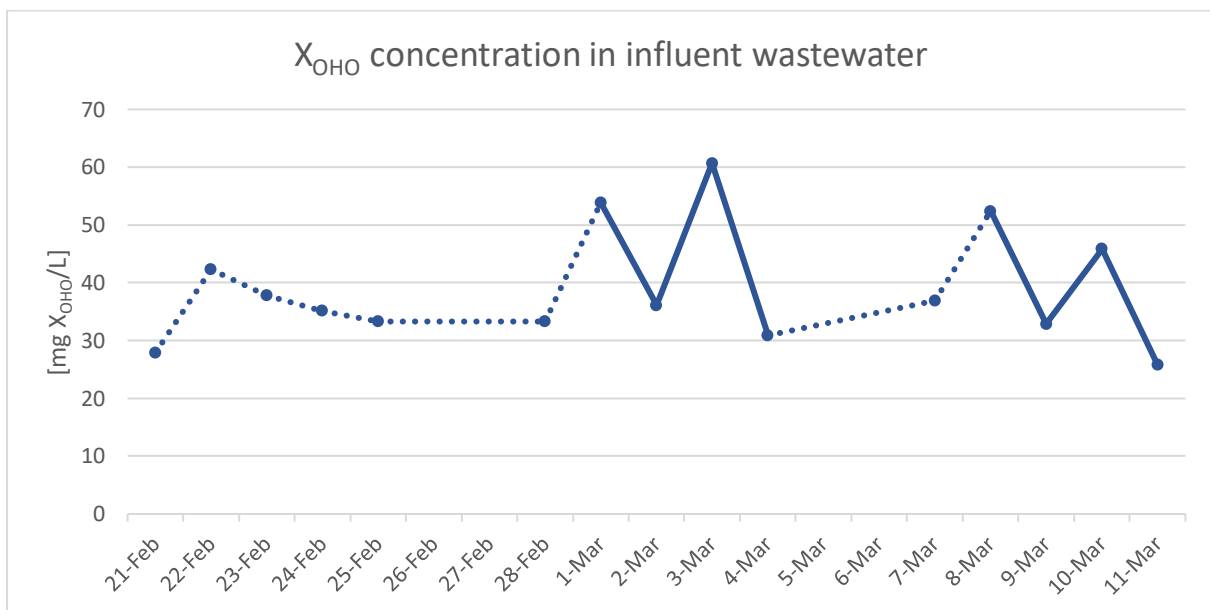


Figure 4.16. Concentrations of X_{OHO} in unfiltered influent wastewater.

4.2 COD fractions in influent wastewater

Figure 4.17 presents the percentage fractions of X_{tot} , C_{tot} and S_{tot} throughout the sampling period by subtracting X_{tot} from the total COD and the results of C_{tot} and S_{tot} . Specific values can be found in Appendix XI. The results show that the fractions are varying throughout the sample period. This further indicates that the composition of the organic matter in the wastewater is not constant but is affected by environmental parameters such as precipitation and water temperature. As can be seen, the particulate fraction is dominating and increases in the end of the sampling period, most likely due to the rain event.

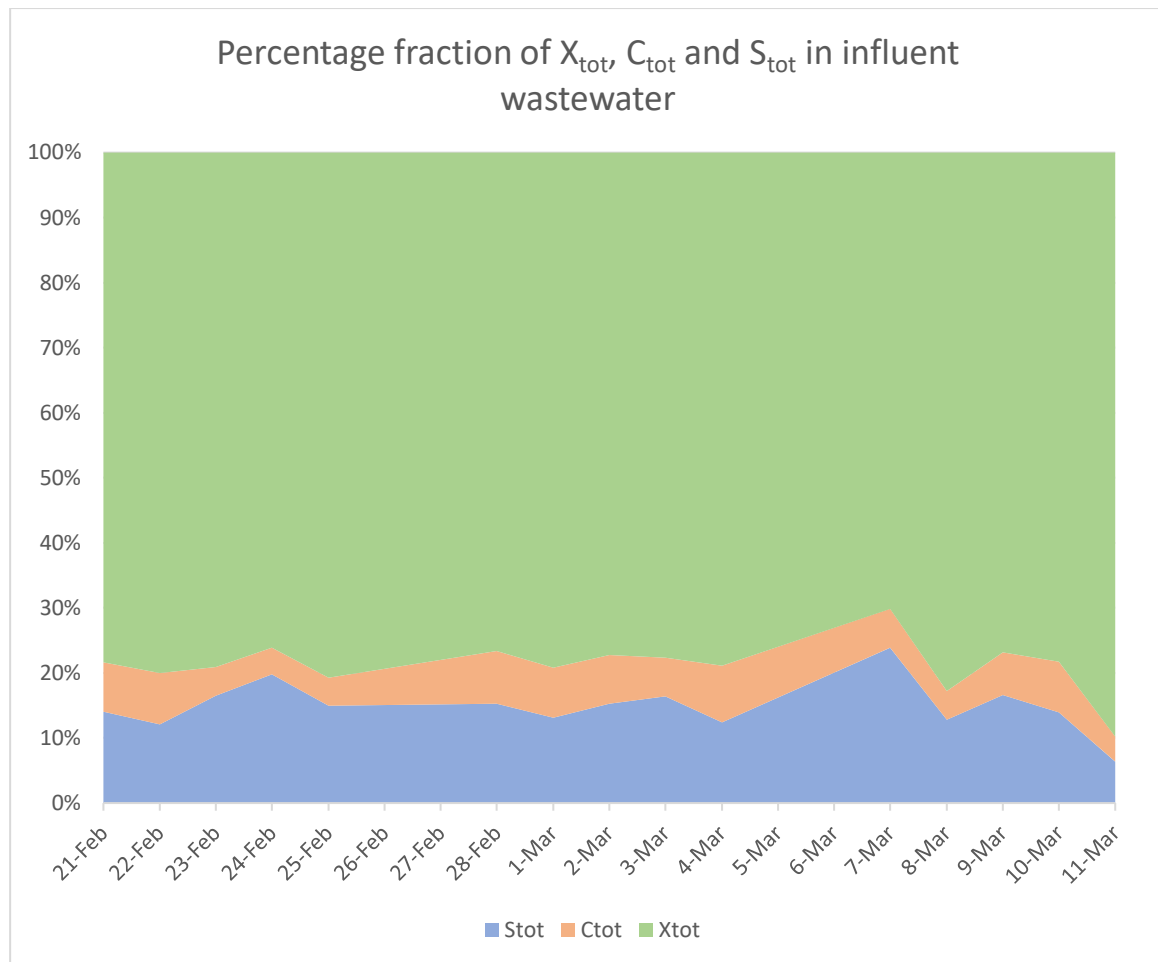


Figure 4.17. Variations in percentage fractions of X_{tot} , C_{tot} and S_{tot} .

Figure 4.18 presents the percentages of each fraction of organic matter in the influent wastewater with specific values found in Appendix XVII. As can be seen, the particulate fraction is dominating with the largest concentrations of X_U and X_B . Since average percentages of concentrations are used to calculate parts of fractions, the share of some fractions is relatively constant which can be seen for e.g. S_U . When interpreting the result, it is important to take this into consideration.

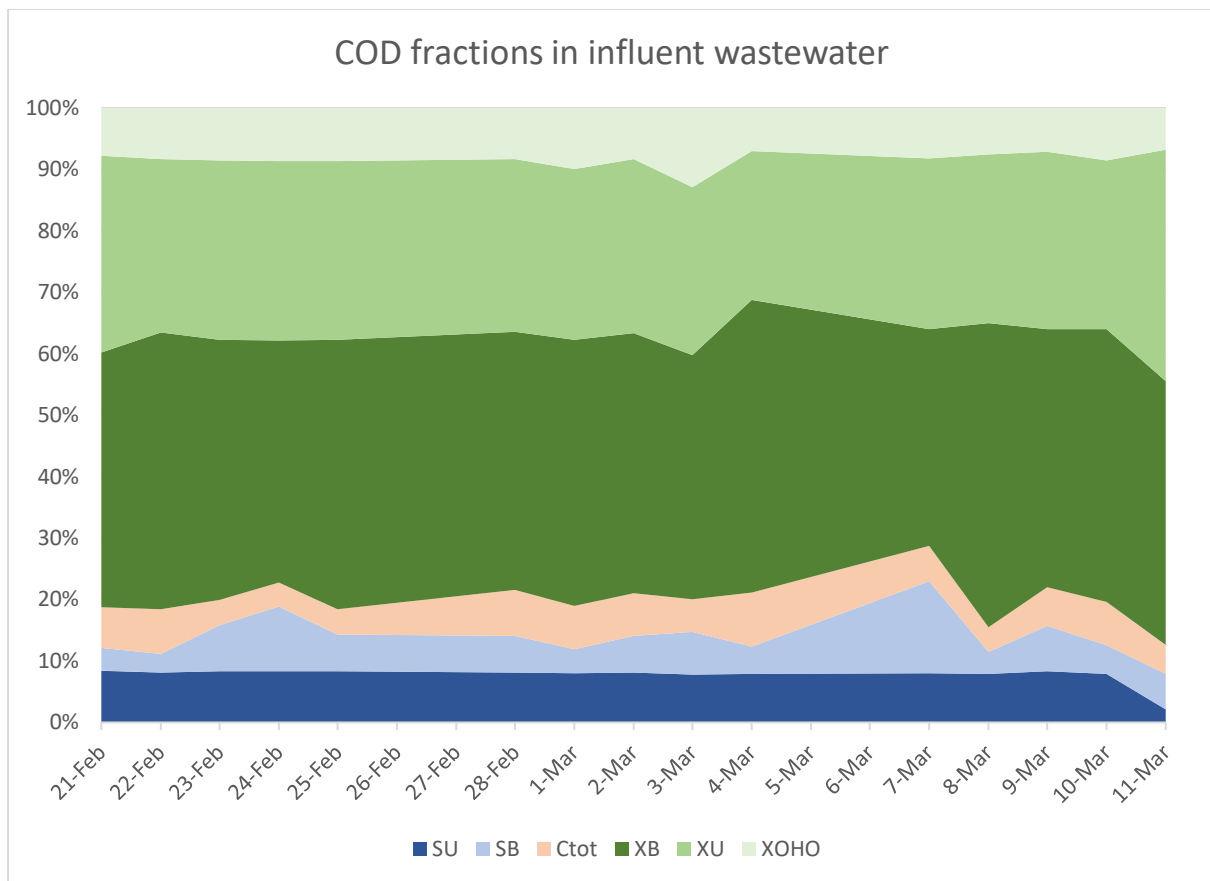


Figure 4.18. Percentages of the different fraction of organic matter in the influent wastewater.

Figure 4.19 presents the variations of the different fractions in the influent wastewater. Since several parameters and fractions are not measured daily and calculated using the average percentage fraction from the remaining days, there are uncertainties when interpreting the results. There are also potential inaccuracies in the laboratory methods, with possibilities that the deviating concentrations are connected to independent faulty measurements.

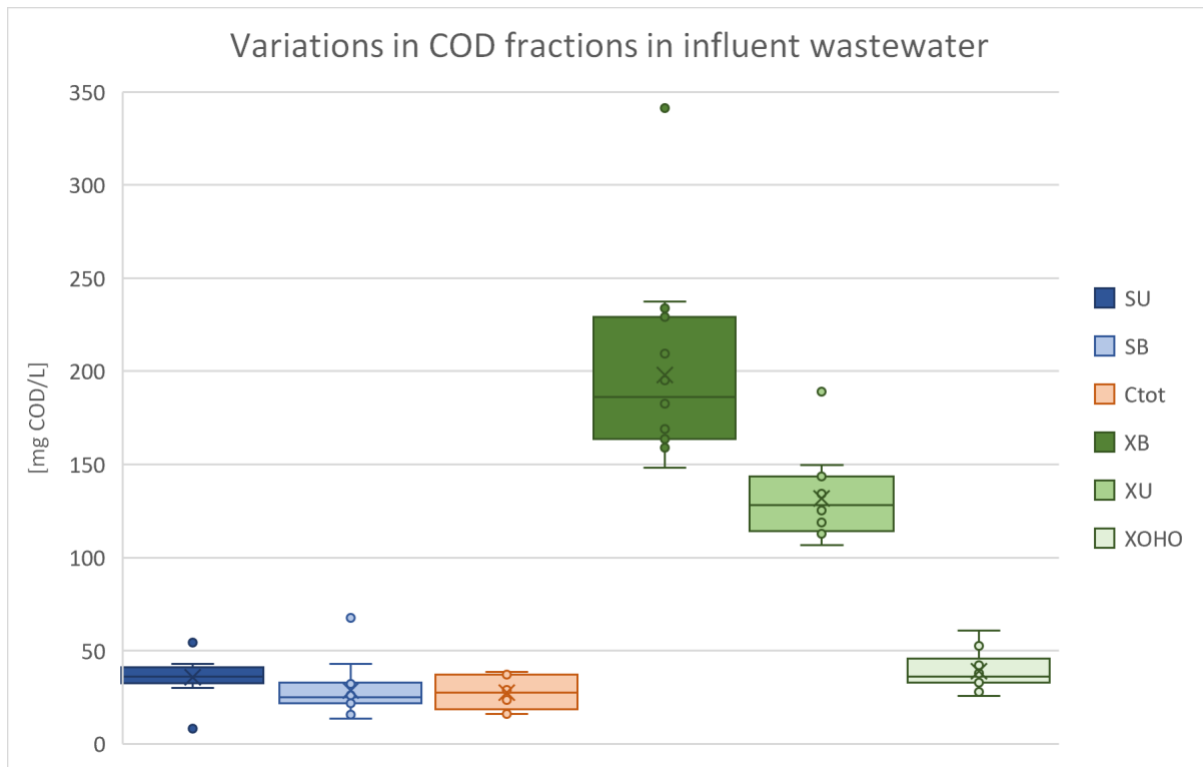


Figure 4.19. Variations in COD fractions in influent wastewater.

4.3 Results from COD analyses of the effluent water

Since the analyses on the effluent water is not as extensive as the influent analyses, they are not comparable considering all fractions. Figure 4.20 shows the COD concentrations of unfiltered and filtered effluent water with specific concentrations found in Appendix XVIII. The increase of COD on the 28th of February cannot be seen in the analysis of TOC for the same date and cannot be explained further. In section 4.6, the removal efficiency of organic matter is discussed.

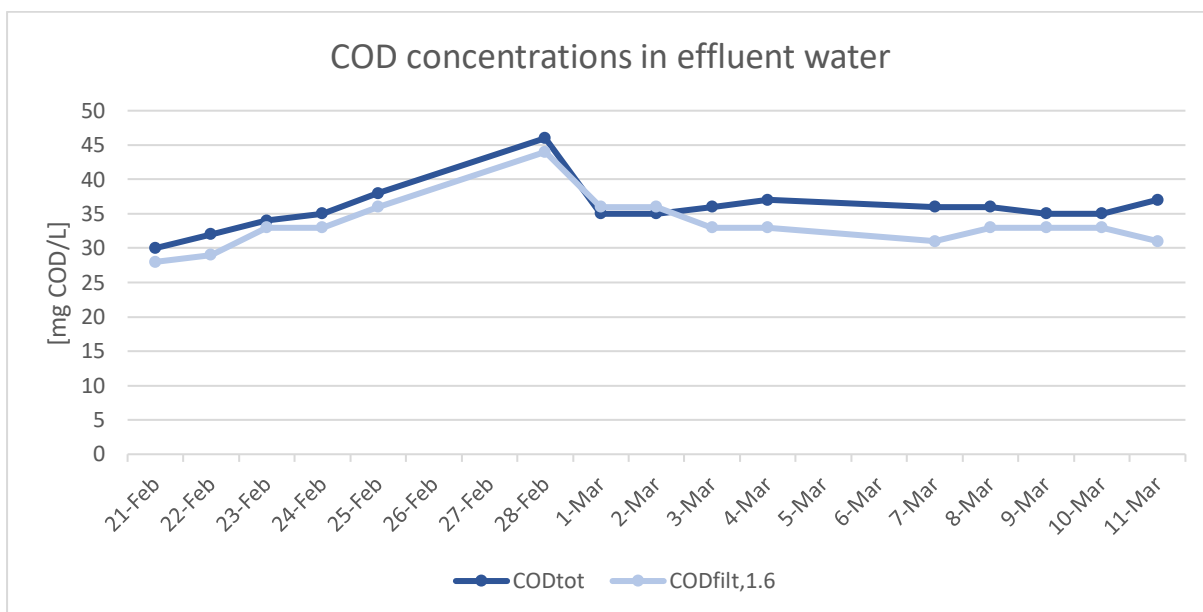


Figure 4.20. COD concentrations of unfiltered and filtered effluent water.

4.4 Results from nitrogen analyses

This section will present the results from the analyses of nitrogen conducted at Rya WWTP. The total nitrogen concentration is presented in Figure 4.21 with [mg N/L] on the y-axis and [date] on the x-axis. Specific concentrations of the fractions can be found in Appendix XIX. The concentrations of TN are not increasing on Mondays as was the case for COD_{tot} . This can be explained by the fact that TN consists mainly of soluble fractions which are not affected by the flushing of the pipes to the same extent as particulate fractions. As can be seen, the X_N concentrations have a similar pattern as TN, with a deviation in the end of the sampling period. This indicates that there is a difference in the composition of the water that is used for flushing the tunnels and the I/I entering the sewer system due to the rain event.

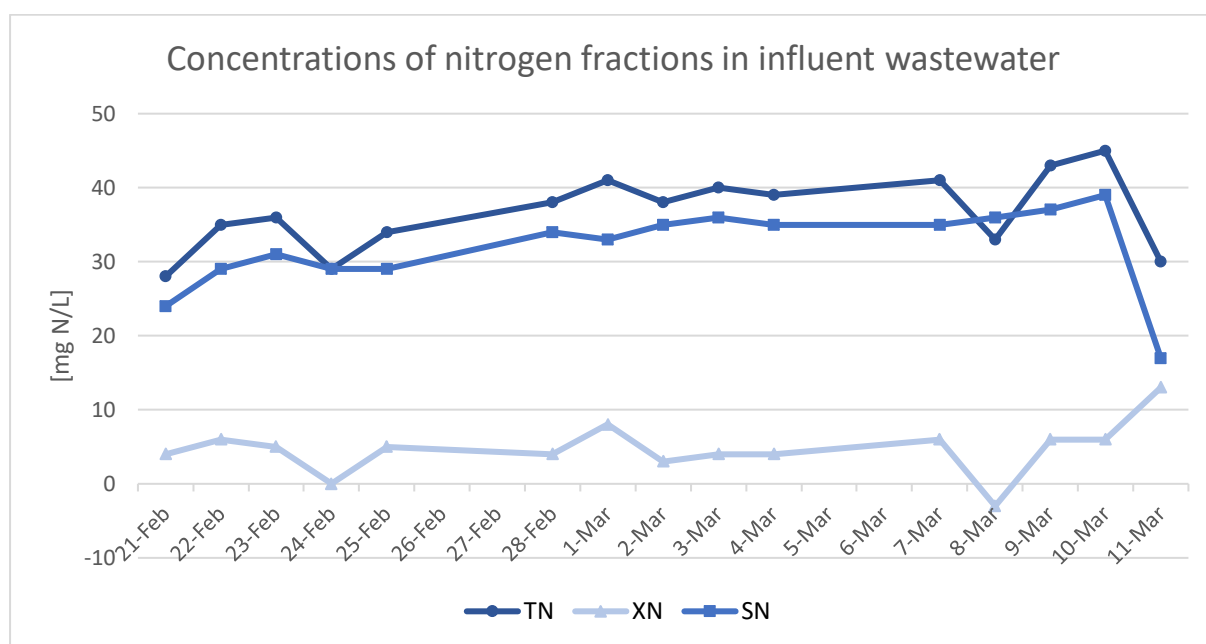


Figure 4.21. Nitrogen concentrations in influent wastewater.

The result of $\text{S}_{\text{NO}_2}/\text{S}_{\text{NO}_3}$ show a concentration that is mainly below 0.10 mg/L, however, on the 21st of February and 11th of March it is above 0.2 mg/L as can be seen in Figure 4.22 with [%] on the y-axis and [date] on the x-axis. Since NO_3 is soluble in water, the increase in concentration can be connected to the greater precipitation during and around those dates. Analyses show that TN_{filt} was larger than TN on the 8th of March leading to a negative concentration for X_N . This is probably a result of uncertainties of the analysis, which in this case is around 10 – 13%. There is also a possibility of mixing up the samples at the laboratory. This is noted with a circle in Figure 4.22. On the 24th of February, the concentration of X_N is zero. On the same date, the concentration of TSS and VSS is higher than the remaining days as seen in Figure 4.6. However, studies have shown that the concentration of TSS and nutrients have a positive relationship, meaning that if one parameter increases, the other parameter generally also increases (Li & Zuo, 2020; Paudel et al., 2019). The low concentration of X_N is possible due to inaccuracies in the laboratory. X_N is increasing in the end of the sampling period, in contrast to the other fractions. Since the *first flush* of stormwater often consists of particulate matter, greater precipitation in the end of the sampling period could be the reason

for this. Throughout the sample period, TN constitutes of approximately 17% $S_{N,other}$ and the importance of this fraction is discussed further in section 4.6.

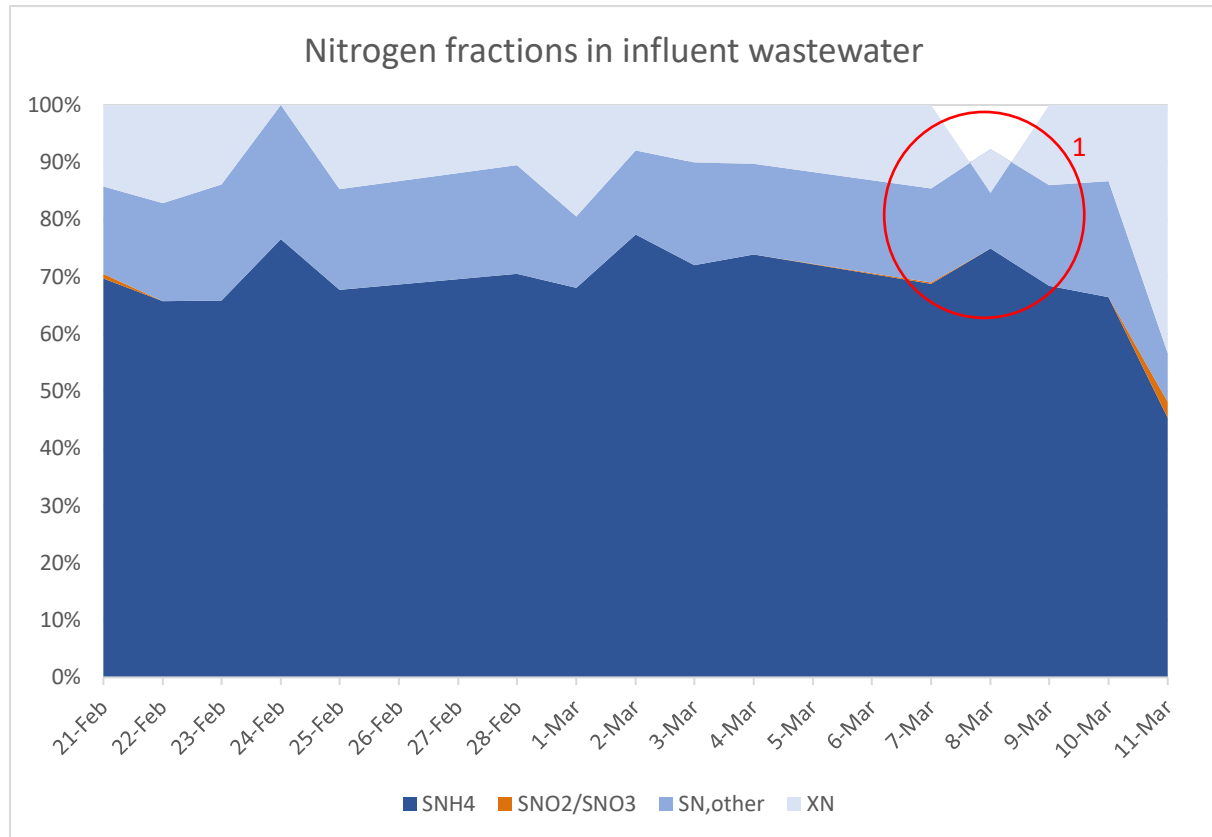


Figure 4.22. Fractions of nitrogen in the influent wastewater.

¹On the 8th of March, TN_{filt} was larger than TN resulting in a negative value for X_N . Since the analyses are connected, all results from this day might be faulty and therefore these values are not trustworthy.

Figure 4.23 presents the effluent percentage of nitrogen fractions with specific concentrations for the effluent concentrations found in Appendix XXI. All nitrogen fractions decrease both in concentration and in total load after the treatment process, with an exception for NO_3 which is increasing in concentration in the effluent water. This is due to the aerobic process in the biological treatment where nitrogen is oxidised and NO_3 is created (Svenskt Vatten AB, 2013). On the 11th of March, there was no data for the filtered nitrogen and therefore, this concentration is set to zero as seen in Figure 4.23. $S_{N,other}$ does not increase significantly, although the importance of this parameter is discussed further in section 4.6.

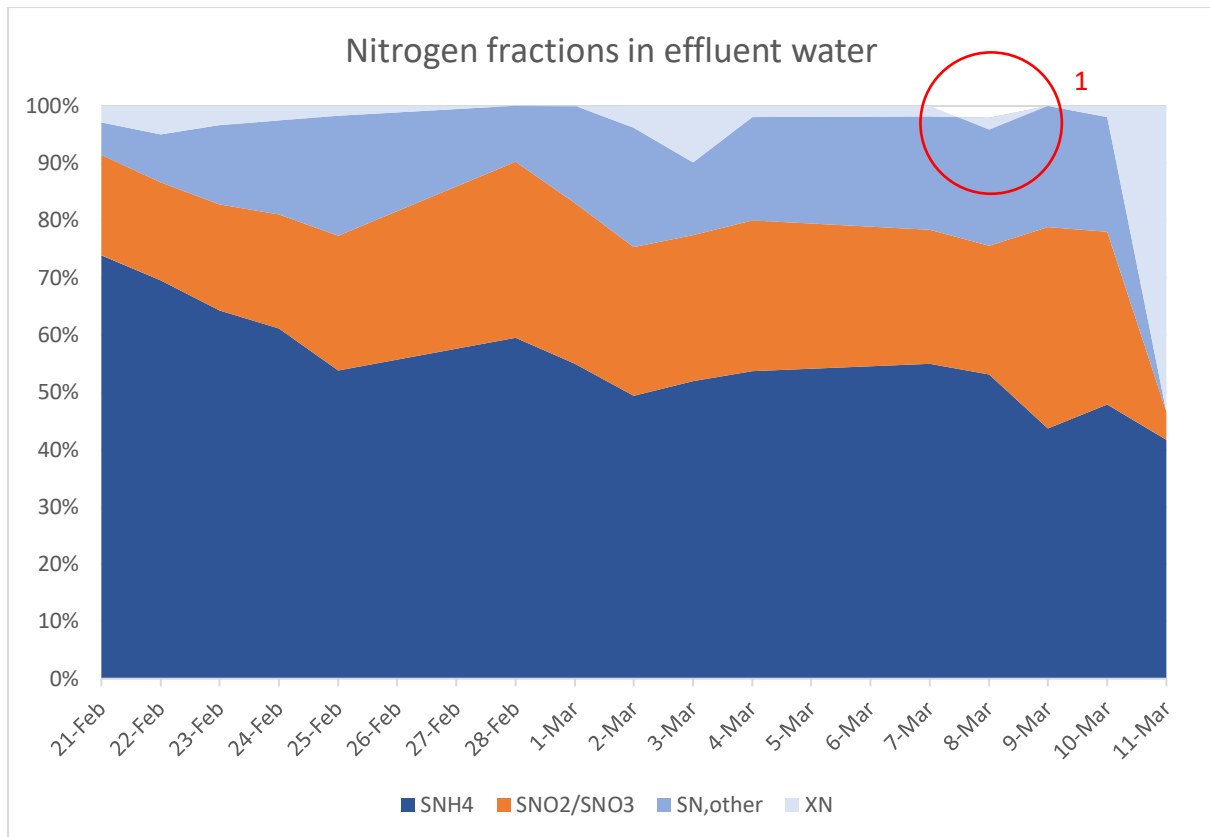


Figure 4.23. Percentage fraction of nitrogen in the effluent water. On the 11th of March, there was no data of the filtered nitrogen from the laboratory and therefore, TN_{fil} is set to zero which is affecting the concentration of the remaining soluble fractions.

¹Similar as for the influent water, on the 8th of March, the effluent water has a larger concentration of TN_{fil} compared to TN , leading to a negative concentration of X_N on this date.

4.5 Results from phosphorus analyses

This section will present the results from the analyses of the phosphorus concentration in the influent wastewater and effluent water. The concentrations of phosphorus fractions are presented in Figure 4.24 with [mg P/L] on the y-axis and [date] on the x-axis. Overall, the phosphorus concentrations are lower in the beginning and end of the sampling period, similar to the result from the COD analyses. This is in line with the arguments of more diluted water during rainfall and melting snow and therefore lower concentrations of compounds present in the wastewater. The concentration of TP has an increased value every Monday with the same arguments as for COD_{tot} and the flushing of the pipes in addition to the fact that the particulate phosphorus is the main fraction of the total phosphorus. However, with this argument, the concentration of TP should also increase on the 11th of March with the increase of precipitation. The reason for the variations in transportation of particulate matter needs further investigation.

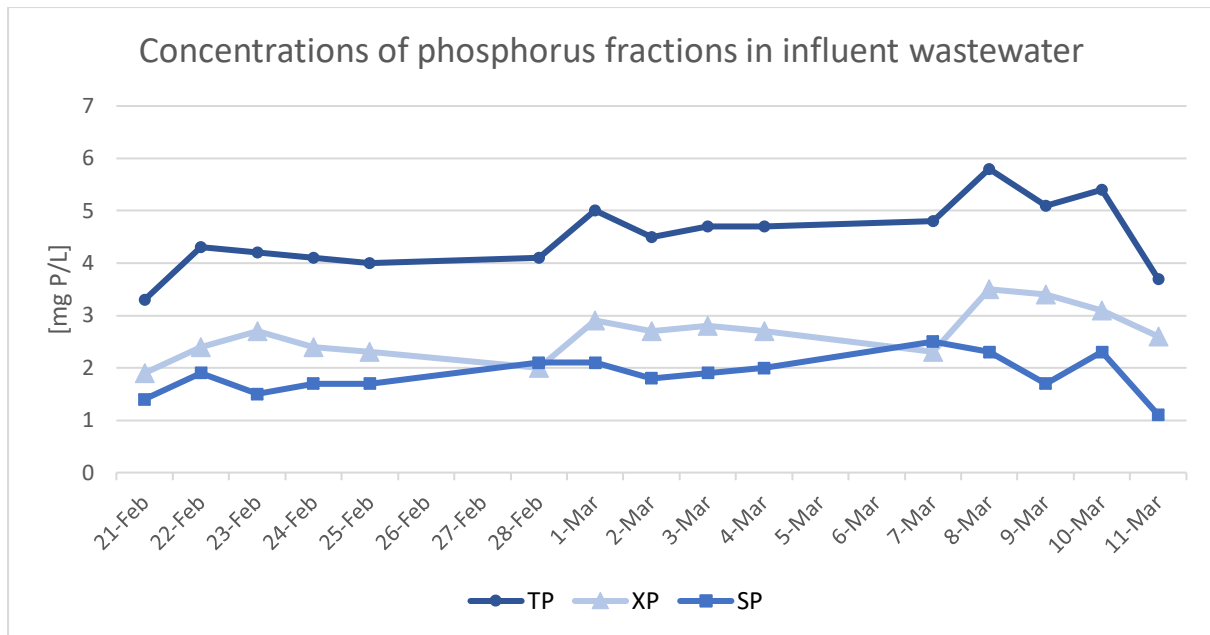


Figure 4.24. Phosphorus concentrations in influent wastewater.

The compiled fractioning of the phosphorus analyses in the influent wastewater is found in Figure 4.25 with [%] on the y-axis and [date] on the x-axis. Specific concentrations of the fractions can be found in Appendix XX. The concentration of PO_4 is on the 2nd of March greater than TP_{filt} . This leads to a negative concentration of $S_{P,other}$ and is probably a result of the uncertainties of the analysis, which in this case is around 16%. The concentration of $S_{P,other}$ is set to zero at this point. Throughout the sample period, TP constitutes of approximately 9% $S_{P,other}$ and the percentages relationships between the fractions are relatively constant with an exception at the end of the sampling period. The $S_{P,other}$ fraction is only 2% of TP on the 11th of March, related to the decrease of soluble matter with the increase in precipitation. The potential importance of this fraction is discussed further in section 4.6.

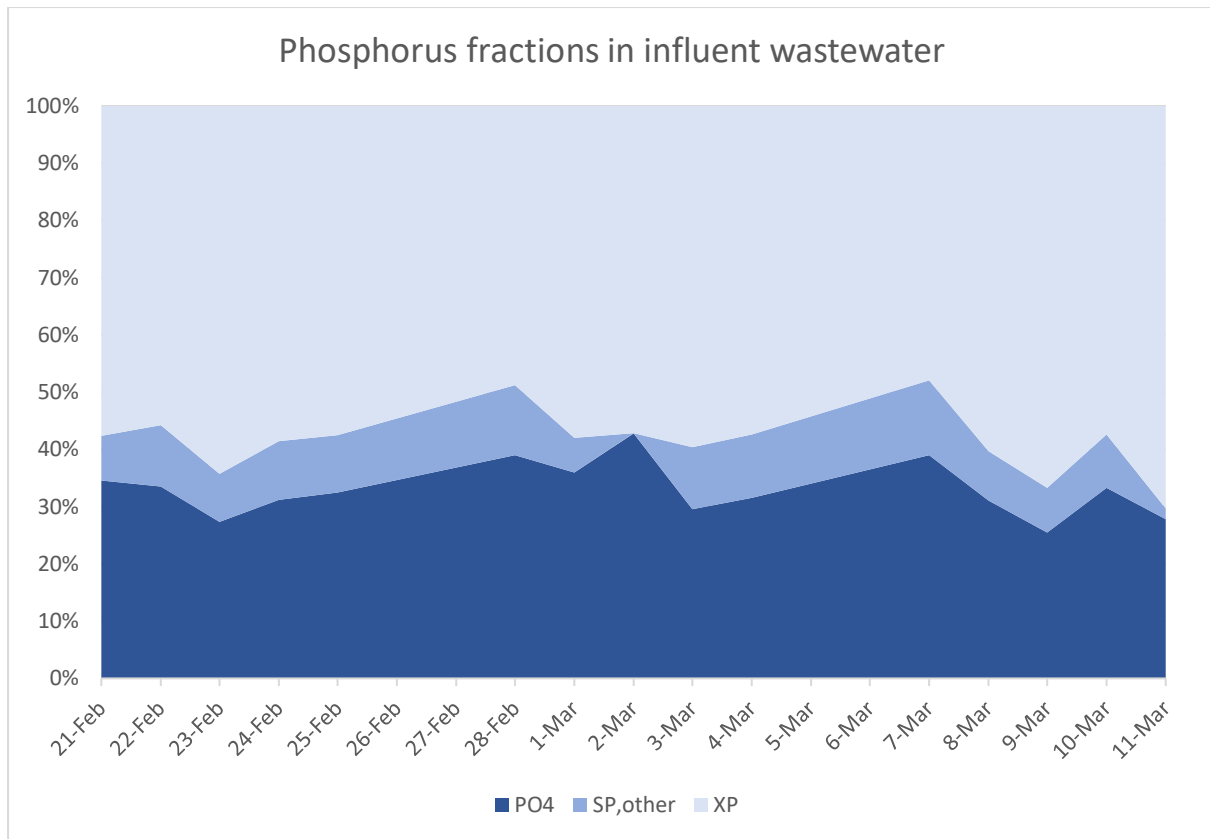


Figure 4.25. Percentages of phosphorus fractions in influent wastewater.

Figure 4.26 presents the effluent percentage of phosphorus fractions with specific concentrations found in Appendix XXI. All phosphorus fractions are decreasing both in concentration and in total load in the effluent water. Although, the fraction SP_{other} increases in percentage share after treatment compared to the influent concentrations found in Figure 4.25. The importance of this fraction is discussed further in section 4.6.

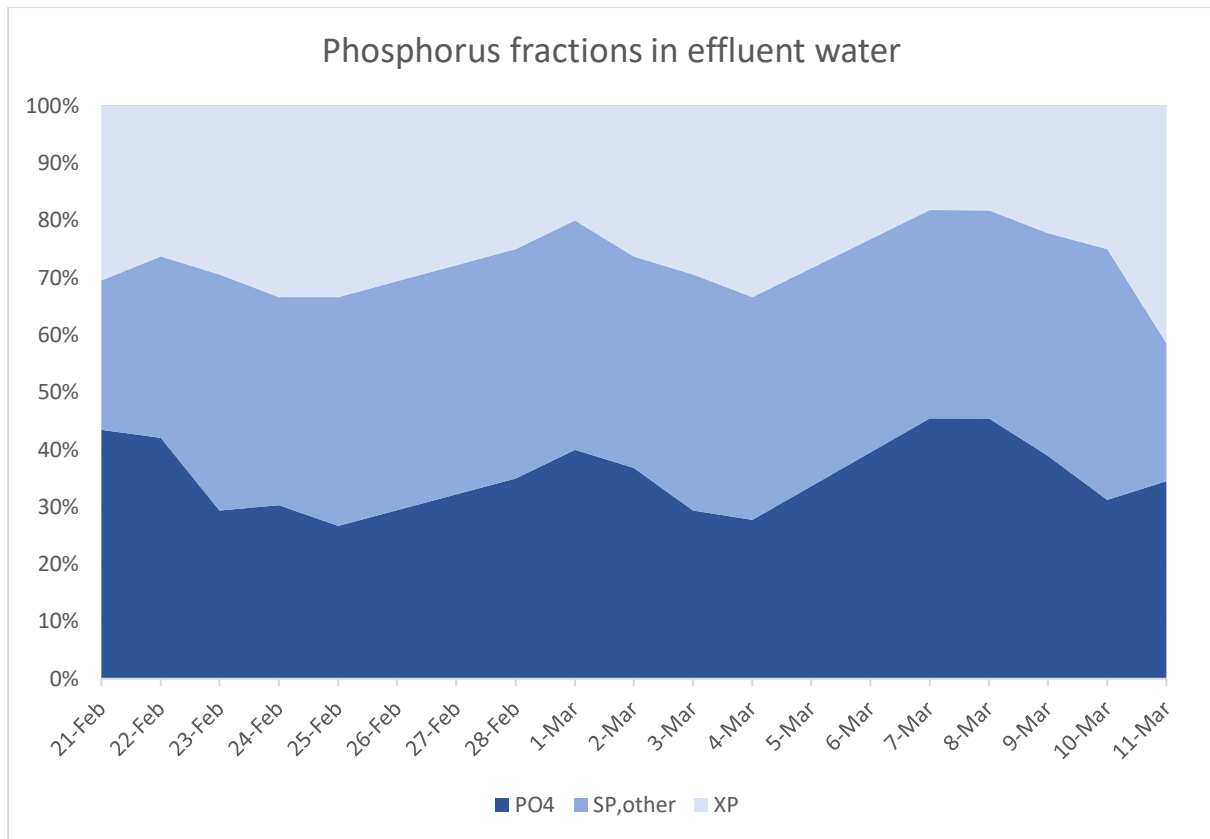


Figure 4.26. Percentage fraction of phosphorus in the effluent wastewater.

4.6 Removal efficiency

The average removal efficiency for organic matter, nitrogen, and phosphorus is presented in Table 4.6 as percentage of removal. These values are obtained by comparing the effluent concentrations to the influent concentrations of the different compounds. Specific values for the different parameters can be found in Appendix XXII. The limit values of organic matter, nitrogen, and phosphorus at Rya WWTP is with an estimated removal efficiency identified by I'Ons (2018) 95% or more for organic matter and phosphorus, and 60 – 80% removal of nitrogen. The average removal efficiency of the unfiltered water in this study is 92% for organic matter which is slightly below the estimated requirement of removal. Total phosphorus, with a removal efficiency of 96% is above the estimated requirement of 95% and nitrogen is also exceeding 60 – 80% with a removal efficiency of 84%.

Table 4.6. Average removal efficiency of COD, nitrogen, and phosphorus from Rya WWTP during the sampling period.

Removal efficiency between influent and effluent water at Rya WWTP								
	COD _{tot} [mg COD/L]	TN [mg N/L]	TP [mg P/L]	COD _{filt, 1.6} [mg COD/L]	TN _{filt} [mg N/L]	S _{N,other} [mg N/L]	TP _{filt} [mg P/L]	S _{P,other} [mg P/L]
Average	92%	84%	96%	73%	84%	87%	92%	77%
Standard deviation	2%	6%	1%	6%	4%	4%	3%	23%

As can be seen in Table 4.6, SP_{other} and SN_{other} are removed with a relatively high efficiency but with future increasing demands on removal efficiency, they should be looked further into. Current treatment processes are not focusing on these specific fractions and when requirements of effluent water changes and become stricter, the decrease of these fractions could be essential. In particular SP_{other} since this fraction has a lower removal efficiency together with a relatively high overall removal for TP of 96%. There is also a high standard deviation, further showing the importance of removing this fraction from the wastewater. The high standard deviation value is obtained since the last day of analysis, no SP_{other} is removed from the effluent water.

With increasing treatment requirements of concentrations of organic matter and nutrients in effluent water there is a need of improvements of treatment methods. For the COD fractions, $COD_{\text{filt},1.6}$ has a relatively low removal efficiency. Although, since the limit values for organic matter in the effluent water are in BOD_7 , the relationship between COD and BOD_7 must be known. By transforming organic phosphorus fractions to inorganic fractions, the treatment and recovery efficiency of phosphorus can be improved (Carrillo et al., 2020). The fraction SP_{other} has an average share of 19% in the effluent water, meaning a relatively large percentage fraction of TP. Together with SN_{other} with an average share of 15% in the effluent water, these fractions should become a main focus point for nitrogen and phosphorus removal. In addition, since the effluent water consists mainly of SNH_4 , further treatment methods in addition to already implemented ones could be ammonium separation (Svenskt Vatten AB, 2013). Ammonium sulphate and ammonium nitrate which is removed from the process can be used as fertilisation.

4.7 Comparison to previous studies

To be able to compare the results obtained in this study to previous characterisations, it is important to consider the differences in the WWTPs. Table 4.7 presents the yearly average concentrations of organic matter, nitrogen, and phosphorus in the influent water of Rya WWTP and five other WWTPs in Sweden. They receive varying amounts of influent water flows during a day with differences in the share of I/I, where Rya WWTP has the largest amount of 60% and at the same time lowest concentrations of pollutants. This can be explained by the large dilution of the water due to the excess influent water. Ängen WWTP also has a high share of I/I with 48.1% in the influent water, but higher overall concentrations of pollutants compared to Rya WWTP. There is a lot of industrial influent water to Ängen WWTP, originating from Lantmännen Reppe AB, an agricultural company (Arnell & Wärff, 2021). This affects the composition of the wastewater with higher concentrations of COD. At Nykvarn WWTP in Linköping, there is a high concentration of BOD_7 in the influent water with 339 mg BOD_7/L . The concentrations of nitrogen are also higher at Nykvarn WWTP compared to Rya WWTP, with 52 mg N/L relative to 27 mg N/L. Nykvarn WWTP has a low amount of I/I entering the pipes due to the usage of separate sewers and the water is thus less diluted and could therefore contain higher concentrations of pollutants.

Table 4.7. Yearly average concentrations of organic matter and nutrients in the incoming water to WWTPs in Sweden.

Influent wastewater	Average inflow of wastewater [m³/d]	Share of I/I [%]	Total phosphorous [mg P/L]	Total nitrogen [mg N/L]	BOD₇ [mg BOD₇/L]	COD [mg COD/L]
Rya WWTP (Davidsson, 2020)	387 470	60%	3.4	27	160	349 ¹
Nykvarn WWTP (Tekniska Verken, 2020) ²	40 900	16%	5.01	52	339	-
Ängen WWTP (Lidköping WWTP, 2019) ³	12 138	48.1% ⁴	4.9	40	269	469 ¹
Källby WWTP (VA SYD, 2020a) ³	28 715	17%	5.8	48	189	376 ¹
Klagshamn WWTP (VA SYD, 2020b) ⁵	23 009	25%	4.9	45	187	439
Sjölunda WWTP (VA SYD, 2020c) ⁶	105 900	27%	5	45	269	580

¹Measured using COD_{Cr}.

²Based on 52 samples throughout the year.

³Based on daily and weekly sampling.

⁴Referred to as extraneous water.

⁵Based on flow proportional sampling.

⁶Based on weekly and monthly sampling throughout the year.

Table 4.8 presents previous characterisations together with the results from this study. When percentage fractions are mentioned, they are related to the COD concentration of COD_{tot}. As can be seen, the total percentage fraction of COD is 106% in this study. This is since every fraction was measured and not calculated from the percentages for the remaining fractions. This further acknowledges the uncertainties with the methods used in this study.

Table 4.8. Data from characterisations of organic matter in influent wastewater.

	Characterisations of organic matter in influent wastewater							
	Results from this study	(Lysberg & Neth, 2012)	(Welch, 1994)	(Arnell & Wärff, 2019)	(Arnell & Wärff, 2021)	(Tebini, 2020)	(Tebini, 2020)	(Tebini, 2020)
	Rya WWTP (Gothenburg)	Rya WWTP (Gothenburg)	Rya WWTP (Gothenburg)	Nykvarn WWTP (Linköping)	Ängen WWTP (Lidköping)	Källby WWTP (Lund)	Sjölunda WWTP (Malmö)	Klagshamn WWTP (Malmö)
Filter size used	0.45 µm – 1.6 µm	1.6 µm	0.45 µm	0.45 µm – 1.6 µm	0.45 µm – 1.6 µm	0.1-100 µm	0.1 – 100 µm	0.1 –100 µm
Total COD (COD_{tot})	106%	99%	100%	100%	99.9%	100%	100%	99.9%
Soluble COD (S_{tot})	15%	21%	23%	19%	23.8%	22%	34%	16.5%
Inert soluble (S _I)	8%	13%	11%	3.3 %	4.7%	9%	10%	5.5%
Readily degradable (S _B) ¹	7%	8%	12%	15.7 %	19.1%	13%	24%	11%
VFAs (S _{VFA})	2% ²				7.6%	1%	9% ³	5.5% ⁴
Other readily degradable (S _{other})	5%				11.5% ⁵	12% ⁵	15% ⁵	5.5% ⁵
Colloidal COD (C_{tot})	6%				9.6%⁶			
Particulate COD (X_{tot})	85%	78%	77%	81%	66.5%	78%	66%	83.4%
Slowly degradable (X _B)	45%	36% ⁷	36.5% ⁷	46.5 % ⁷	32.5%	44% ⁷	33% ⁷	40.9% ⁷
Inert particulate (X _I)	31%	24%	22%	30.9%	29%	22% ⁸	22%	42.5% ⁸
Heterotrophic biomass (X _{OH})	9%	17%	18%	3.6 % ⁹	5% ⁹	12%	11%	0% ¹⁰
Autotroph biomass (X _{BA})		1%	0.5%					

¹Consists of S_{VFA} and S_{other}.

²Only acetate is included in S_{VFA} in this study.

³The wastewater had already gone through the preliminary sedimentation and therefore fermentation might have occurred.

⁴Assumed to be half of the concentration of S_B since the analysis resulted in too low concentrations.

⁵Referred to as fermentable easily degradable.

⁶The colloidal fraction was divided into biodegradable and undegradable fractions.

⁷The slowly degradable particulate COD contains colloidal fractions.

⁸Was calculated through subtraction.

⁹Was estimated based on previous characterisations and standard value.

¹¹Assumed that X_{OH} was included in the other particulate fractions.

4.7.1 Soluble COD

In this study, the soluble fraction for organic matter is defined as $<0.45\ \mu\text{m}$, however, there are differences when it comes to defining the fractions in earlier conducted characterisations with varying filter pore sizes and methods used to obtain concentrations. This will affect the comparability between studies. There are also seasonal differences and according to Yang et al. (2019), there is a need for a sufficient amount of days of a wastewater characterisation with stable and normal plant operation to obtain a robustness of the results. Characterisations from Rya WWTP are conducted mainly during the summer, meaning that the temperatures are probably higher than the average in combination with less precipitation and a longer retention time in the sewer system. The amount of water in the pipes will have an impact on the wastewater characterisation and with less water in sewer systems, anaerobic conditions constitute a major part of the processes (Wilén et al., 2006). An anaerobe environment could create hydrolysis processes, leading to degradation of particles which in turn will lead to higher concentrations of soluble fractions. (Tebini, 2020). This is in line with the result of S_{tot} of 15% from this study which is low compared to earlier studies. Previous characterisations from Rya WWTP, conducted during the summer presents S_{tot} concentrations of 21% and 23% (Lysberg & Neth, 2012; Welch, 1994).

The concentration of S_B is relatively low in the results from this study, with an average value of 7%. Previous characterisations generally show a concentration above 10%. This could be explained by the lower temperatures and the lack of hydrolysis during the time of year when this study was conducted, in February – March. The measurement at Rya WWTP by Lysberg & Neth (2012) is conducted during the summer, but has a low concentration of S_B with 8%. In the study, there are observations of the low concentration compared to references and could be caused by uncertainties in the analysis. The concentration of S_B at Sjölanda WWTP is large, 24%, compared to the other characterisations and the overall S_{tot} concentration is also larger than the other ones which can be connected to the sampling method occurring after the preliminary sedimentation (Tebini, 2020). However, the S_B concentration of 12% was not particularly large for Welch (1994) where the sampling also occurs after the preliminary sedimentation. At Ängen WWTP in Lidköping, the S_B concentration is high, with a share of 19.1%. This WWTP also receives large volumes of industrial wastewater containing a lot of S_B (Arnell & Wärff, 2021).

Since the results from the HPLC analysis presented in this study only consists of acetate, there are uncertainties with the comparison of S_{VFA} to previous studies and the fraction of 2% S_{VFA} is probably underestimated. The fraction of S_{VFA} of 5.5% at Klagshamn WWTP is based on an assumption and could therefore be considered relatively uncertain (Tebini, 2020). Analyses of VFAs are conducted at different steps of the wastewater processes in the studies, meaning that varying degradation rates have occurred and therefore it might be difficult to compare these measurements. The high concentration at Sjölanda WWTP with 9% can be explained by the fact that the water had passed through the preliminary sedimentation and therefore VFAs might have been created through fermentation (Tebini, 2020). Not all previous characterisations

include the VFA parameter, and therefore a comparison of concentrations of the fraction is not easy. This will also cause uncertainties with the concentration of the S_{other} fraction.

The characterisation from Nykvarn WWTP, Ängen WWTP together with Klagshamn WWTP show less S_U compared to the other WWTPs with 3.3%, 4.7% respectively 5.5%. In Linköping there are a lot of separated sewers, meaning less I/I transported to the WWTP (Tekniska Verken, n.d.). The low concentrations at Nykvarn WWTP could mean that I/I contains large concentrations of S_U . Since less I/I is transported to the WWTP, at the same time less S_U is transported. Klagshamn WWTP is considered a relatively small WWTP and according to Tebini (2020), low volumes of wastewater from industries are received. The industrial water is assumed to contain larger concentrations of S_U and therefore, in this study S_U should be assumed to be higher than at Klagshamn WWTP, which is the case with an average of 8%. Although, the low concentrations of S_U at Ängen WWTP are not in line with the argument of S_U content in industrial water. The analysis methods vary between the studies, where e.g. in this study, a long-term BOD test was used for the S_U measurement whereas Tebini (2020) analysed the soluble fraction in the effluent water. Thus there will be difficulties when comparing the results of the different characterisations. Further uncertainties in this study are the calculations of S_U using concentrations from only two long-term analyses. Calculations of concentrations during dry weather might not be applicable to rain events as can be seen on the last day of analysis where the S_U concentration from the long-term BOD test remained high despite lower concentrations of S_{tot} . This resulted in the use of S_U concentrations obtained from the analysis after the nitrifying MBBR step, and therefore further uncertainties.

4.7.2 Colloidal COD

The colloidal fraction was analysed in this study which was generally not done in previous studies. The only characterisation that included this fraction was Wärff & Arnell (2021) at Ängen WWTP. The average fraction C_{tot} obtained through analyses in this study significantly lower than the one obtained at Ängen WWTP with 6% compared to 9.6%. The filter pore sizes are the same, with 0.1 μm and 0.45 μm used to identify the fraction. A potential explanation is the fact that Lidköping receive more industrial water compared to Rya WWTP, however no further conclusions are drawn regarding this fraction due to lack of data.

4.7.3 Particulate COD

Since previous characterisations generally did not separate between colloidal and particulate fractions, X_{tot} in this study was assumed to be lower in comparison. However, it is larger than previous characterisations with 85% of COD_{tot} . Since there are uncertainties connected to the measurements, this fraction could have been calculated inaccurately. The sewer system connected to Källby WWTP is not managed through pumping, and when there are large rain events, the flushing water might bring particulate matter to the WWTP (Tebini, 2020). However, the overall particulate fraction at Källby WWTP is not larger than the other characterisations. The sampling of wastewater from Sjölanda WWTP and one of the analyses at Rya WWTP are sampled after the preliminary sedimentation and therefore, the concentration of X_{tot} is lower than other characterisations (Tebini, 2020; Welch, 1994).

Since the BCOD analysis conducted in this study was not measured using duplicates, the result of 45% X_B could be considered uncertain. There were only five tests analysed and the rest of the concentrations were decided through the average percentage fraction of the results. The concentrations used in the results are from the second analysis meaning a possibility of inaccuracies with the freezing of the samples. As Roeleveld & Van Loosdrecht (2002) states, there is an uncertainty of 10 – 20% when conducting the BCOD analysis, which was done in this study as well as for Ängen WWTP, Nykvarn WWTP and at Rya WWTP. A fraction of 45% X_B is larger than previous characterisations, but in line with the results from Nykvarn WWTP with 46.5%. Since Linköping has a large share of separate sewers, the wastewater that is transported to the WWTP will not be affected by the stormwater and will therefore have a constant retention time in the sewer system. This will have an impact on the wastewater composition and possibly the concentration of X_B . Overall, there are many uncertainties within all different studies and methods used to obtain X_B and this therefore complicates a conclusion regarding this fraction.

The percentage fraction of X_U found in this study is 31%, which is similar to the characterisation at Nykvarn WWTP and Ängen WWTP obtaining a value of 30.9% respectively 29%. Arnell & Wärff (2019) at Ängen WWTP had issues with the long-term BOD tests, which was also the case for Lysberg & Neth (2012) presenting a value of 24% X_U . At Rya WWTP, there were issues with evaporation leading to few samples being included in the result meaning a less robust analysis. The value of 22% from Rya WWTP, conducted by Welch (1994) indicates a low fraction which can be connected to the fact that the sample is extracted after the preliminary sedimentation and the particulate matter have had time to settle. The high concentration of 42.5 % X_U at Klagshamn WWTP was determined by subtraction of other particulate matter, meaning no individual analysis was conducted. The same was the case for Sjölanda WWTP and Källby WWTP, both with 22% X_U .

In accordance with the average X_{OHO} concentration from this study of 9%, previous characterisations conducted in Sweden show a general trend of an X_{OHO} concentration of approximately 10%. However, this is not the case for Nykvarn WWTP, Klagshamn WWTP and Ängen WWTP. The low value of 3.6% X_{OHO} at Nykvarn WWTP was assumed, based on previous characterisations (Tekniska Verken, n.d.). At Klagshamn WWTP, the fraction of 0% X_{OHO} was assumed to be included in the other particulate fractions and therefore no analysis was conducted (Tebini, 2020). At Ängen WWTP, the concentration of X_{OHO} was estimated using a standard value of 5% (Arnell & Wärff, 2021). These assumptions lead to difficulties when comparing the result from this study to previous studies. At Rya WWTP, X_{OHO} has been measured to 17 – 18% of the total COD concentration (Lysberg & Neth, 2012; Welch, 1994). Both characterisations are conducted during spring and summer which means a higher temperature in general and therefore possibly a greater generation of X_{OHO} . Since there are few results of X_{OHO} concentrations from this study it is difficult to come to a conclusion regarding this fraction, but the potential impact of time spent between extraction of samples to analyses is noticed and should be considered in future studies.

4.7.4 Nitrogen and phosphorus fractioning

This section will compare the results of nitrogen and phosphorus fractionation from this study to previous characterisations. When percentage fractions are mentioned, they are related to the concentration of N_{tot} and P_{tot} . Previous characterisations of nitrogen and phosphorus in influent wastewater from Nykvarn WWTP and Ängen WWTP are presented in Table 4.9 together with the results from this study.

Table 4.9. Data from characterisation of nitrogen and phosphorus in influent wastewater.

Characterisations of nitrogen and phosphorus in influent wastewater			
	Results from this study	(Arnell & Wärff, 2019)	(Arnell & Wärff, 2021)
	Rya WWTP (Gothenburg)	Nykvarn WWTP (Linköping)	Ängen WWTP (Lidköping)
Nitrogen			
Total nitrogen (TN)	100%	100%	99.9%
Soluble nitrogen (S_N)	87%¹	72.7%²	84.5%²
Ammonium nitrogen (S_{NH_4})	70%	60 % ⁴	75.5%
Nitrite/nitrate nitrogen (S_{NO_2/SNO_3})	0%	0.9 %	0%
Other soluble fractions ($S_{N,other}$)	17%	11.3 ⁵	9% ⁵
Particulate nitrogen (X_N)³	13%	27.8 %⁶	15.4%⁶
Phosphorus			
Total phosphorus (TP)	98%		99.9%
Soluble phosphorus (S_P)	39%		
Orthophosphate (PO_4)	30%		48.9%
Other soluble phosphorus ($S_{P,other}$)	9%		1.1% ⁷
Particulate phosphorus (X_P)	59%		49.9%⁸

¹Was obtained by filtering through a 1.6 μm filter.

²Was obtained by filtering and flocculation in addition to filtering through 0.1 μm filter.

³The particulate fraction of nitrogen and phosphorus contains colloidal fractions.

⁴Consists of ammonia and ammonium nitrogen.

⁵Consists of inert soluble nitrogen and biologically available soluble nitrogen.

⁶Consists of inert particulate nitrogen and biologically available nitrogen. Is calculated using concentrations of COD by using typical fitting content of nitrogen.

⁷Consists of soluble biodegradable and undegradable phosphorus.

⁸Consists of particulate biodegradable and undegradable phosphorus as well as colloidal fractions.

Nitrogen fractioning

Since previous characterisations mainly have focused on organic matter, there is a lack of comparable results of nitrogen and phosphorus analyses. However, two nitrogen

characterisations have been made and they show that, in line with the results from this study, the main fraction of nitrogen in the influent wastewater consists of S_{NH4} (Arnell & Wärff, 2019, 2021). The result from this study presents a concentration of S_{NH4} equal to 70% and previous studies show 60% and 75.5%. Another similar result is the low concentration of S_{NO2}/S_{NO3} which in this study was equal to 0%, as at Ängen WWTP, and 0.9% at Nykvarn WWTP. $S_{N,other}$ is larger in this study with a fraction of 17% compared to 11.3% and 9% respectively. The soluble fraction at Ängen WWTP was identified both using flocculation and filtering as in this study, but also through only filtering using a 0.1 μm filter. At Nykvarn WWTP, S_N is calculated both from the effluent and influent water to obtain degradable and undegradable fractions leading to results that might not be comparable. However, the result shows a lower overall fraction of S_N which can be related to the presence of separate sewer systems and therefore faster degradation of the soluble fraction. In this study, S_N is calculated using filters with a pore size of 1.6 μm and therefore results in a larger soluble fraction compared to previous characterisations.

The average concentration of X_N in this study is lower, with a concentration of 13% compared to Nykvarn WWTP with 27.8% and Ängen WWTP with 15.4%. At both Nykvarn WWTP and Ängen WWTP, X_N was identified by using concentrations of the particulate fraction of COD and through fitting of the nitrogen content of COD. Thus, this fraction is estimated and therefore not measured or calculated which could lead to inaccuracies.

Phosphorus fractioning

Only one previous phosphorus fractioning from Ängen WWTP could be found to compare the results from this study (Arnell & Wärff, 2021). This study has a larger concentration of PO_4 , with 30%, compared to 48.9% from the previous characterisation. The fraction of $S_{P,other}$ is larger in this study with a concentration of 9% compared to 1% at Ängen WWTP. Concentrations of X_P are also larger in this study, 59% compared to 49.9%. Since there are large volumes of industrial influent wastewater to Ängen WWTP, the phosphorus concentration is probably affected. However, since there is only one previous characterisation conducted studying phosphorus fractions it is difficult to draw any conclusions from this study.

4.8 Uncertainties with the laboratory work

With laboratory work there are potential uncertainties when handling the sampling and analyses which are described in this section.

- The instruments used in the laboratory work have to be calibrated and this could have been done in a faulty way.
- Degradation might occur in the wastewater before the tests start which will lead to an underestimation of the S_B and the X_{OHO} fraction present in the samples. For the determination of X_{OHO} , this was especially important to consider for the OUR test before starting with ethanol addition.
- Oxygen has varying solubility at different temperatures, meaning that if the temperature changes when measuring OUR, the result might be affected. A water bath was

implemented to minimise the risk of varying temperature during the experiment, however, the temperature was not stabilised until after a few minutes of analysis.

- In the OUR-measurements, the water was sometimes not aerated when it reached 4 mg/L and therefore the degradation was slower than it should be. There were sometimes seconds or minutes of no logging since the instrument did not notify when it had stopped sending data to the computer. The YSI5100 only measures for 60 minutes and it had to be restarted at least 3 times during the test.
- There is a possibility of cross contamination between samples.
- The vials were not shaken/homogenised before the Hach Lange cuvette test for the first four days of the sampling period meaning that the concentration of organic matter in the samples could be underestimated. By homogenising the tests, the organic matter will be evenly spread through the sample leading to a representative analysis.
- Chlorine might still be present after cleaning the instruments leading to a potential decrease in microorganisms in the samples.
- When extracting the supernatant using a syringe, there is a possibility that the flocculant might follow. This will lead to an overestimation of the COD concentrations for the filtered and flocculated samples.

4.9 Discussion of the literature review and background

This section will discuss the literature review and background, together with its connection to this study.

4.9.1 Impact of I/I and stormwater in the sewer system

I/I entering the sewer system could be caused by damages on the pipes in different parts of the sewer system, both on the municipal and privately owned system. The variations in ownership could lead to issues when considering suitable solutions to mitigate the amount of I/I (Lundblad & Backö, 2016). Gryaab AB (2020) investigated and gave proposals on how to mitigate I/I to Rya WWTP through a holistic vision for the municipalities and Gryaab AB. This investigation showed the importance of cooperation when deciding how to manage the sewer system as well as individual prioritisation of the local measures specific for the area of interest.

It is important to identify potential sources of I/I and measurements preventing or mitigating the risks connected to this issue. Examples of these issues are flooding due to the excess water, the need of CSOs and the increase of influent wastewater to Rya WWTP affecting the treatment performance. There might be a need of expanding Rya WWTP in the future to be able to meet the requirements of treatment and these investment costs have to be set in relation to the costs of mitigating the I/I at the source (Mattsson et al., 2016). It is a difficult task deciding which measurements are the most profitable ones and further studies regarding the subject of I/I have to be made. Results from a study by Nivert & Alenius (2017) and Göteborgs stad shows that a small mitigation of I/I on the sewer system might be economically justifiable, but the expansion of Rya WWTP could be considered a better solution taken into consideration the future increasing requirements of wastewater treatment. The composition of I/I varies with environmental factors, but could contain metals, organic matter and nutrients which could have

a negative impact on the efficiency of treatment processes at the WWTP. In addition to this, as can be seen in section 4.7.1, the concentration of S_U seems to increase with the increase of I/I. Section 4.7.3 also indicates increasing concentrations of X_B with an increase in I/I. One main rain event was captured within the sampling period of this study and therefore, the fast and slow precipitation dependent I/I is most probably included. However, the absence of analyses of groundwater infiltration and inflow dependent I/I indicates that further studies should be conducted throughout a longer period with variations in seasonal and yearly impact.

The performance of the WWTP will be reduced with increasing influent water since the plant is not designed to manage the excess water (Molander, 2015). If the inflow rate is exceeding the capacity of the WWTP, a fraction of the flow is bypassed and discharged without treatment. There are variations in quality of the stormwater which will affect the treatment processes and its efficiency (Viklander et al., 2019). By releasing large quantities of water through CSOs, pollutants from both stormwater and wastewater will end up in the environment (Molander, 2015). In this case, local treatment processes might be beneficial since the stormwater will be treated before ending up in the environment. In addition to the environmental impact of stormwater, the operating and maintenance costs of the WWTP will increase with the increase of influent water. The extraneous water will also have an impact on the quality of the sludge derived from the treatment processes since stormwater contains large concentrations of metals and slowly biodegradable organic matter (Bergenstråle, 2019). The limit values of metal concentration in the sludge could be exceeded by the additional stormwater.

According to Viklander et al. (2019), the implementation of SUDS' can mitigate the amount of polluted stormwater entering the pipes at the source, despite the increasing urbanisation and hard surfaces implemented in cities. However, there are also drawbacks with these types of solutions, such as the time for investigation and finding the most suitable solution for the area as well as the cost of the project (Göteborgs Stad, 2010). When releasing water into the environment instead of transporting it to the WWTP it is important to ensure that it is sufficiently clean. This will demand testing the water which could be time consuming and costly (Viklander et al., 2019). Since the source of stormwater, and therefore the solution needed, is specific for the area it is difficult to find a standard solution for preventing stormwater and I/I to enter the sewer system. This is a complex problem with several factors to consider to be able to conclude what is the best solution for managing and treating stormwater and I/I in Gothenburg.

4.9.2 Combined and separate sewers

Replacing combined sewer systems with separate ones would decrease the amount of CSOs, and generally the amount of water transported to Rya WWTP. Although, as mentioned by Göteborgs stad and Nivert & Alenius (2017), it would be an expensive measure and the costs related to releasing stormwater directly into the environment instead of transporting it to Rya WWTP might increase. There would be more untreated stormwater released to the environment and the large costs in relation to the relatively small reduction of I/I have to be evaluated to be able to find the most profitable solution. Calculations by Nivert & Alenius (2017) show that

the reduction of I/I due to the replacement of combined sewer systems with separate systems is only 28%.

Since the combined sewers transport a large volume of I/I to Rya WWTP there will be an impact on the concentrations of pollutants if these sewers were to be replaced with separate sewers. The main fraction of organic matter that will be affected by this change in sewer system is most probably S_U , as discussed in section 4.7.1. As mentioned, with the separate sewers, the water volumes will not vary with precipitation but will have a constant retention time in the sewer system. Whereas in combined sewers, the stormwater dilutes the water and lead to lower concentrations of pollutants, organic matter and nutrients (Henze & Comeau, 2008). Apart from this, the quality of the sludge will be affected by the variations in sewer systems. Rya WWTP receives wastewater from a large capture area meaning a widespread sewer system with long distances for the water to travel before it ends up at the WWTP. If Gothenburg were to implement separate sewer systems, the retention time for the water in the pipes would be long, leading to favourable environments for fermentation and degradation of organic matter which might cause further changes in the composition of the wastewater.

4.9.3 Future wastewater treatment

It is important to consider the costs connected to future demands of reducing the effluent concentrations of organic matter and nutrients in wastewater. As mentioned by Bailey et al. (2019), household utilities will be more efficient in the future, meaning that greywater will be reused and in turn less water will be used. The concentrations of organic matter and nutrients in the wastewater could increase when the water flows decrease, meaning that future treatment processes have to consider higher concentrations of pollutants in addition to the stricter demands. Chemicals used in treatment processes corresponds to approximately 10-20% of the operational costs of a WWTP according to Svenskt Vatten AB (2013). However, if the influent water flows decrease, e.g. by mitigating the amount of I/I, the overall costs will not increase since it is more expensive to treat large volumes of water than to treat higher concentrations of pollutants (Sun et al., 2015). As mentioned in section 4.6, there are some fractions that remains in the effluent wastewater to a relatively large extent and might need an increase in removal efficiency in the future. These are specifically other soluble nitrogen and phosphorus fractions.

5. Conclusion

This section will conclude the work of this study, with regard to the research questions stated in the introduction of the report.

The results from the literature review together with the laboratory work show that organic matter, nitrogen, and phosphorus in the wastewater are not constant in their fractioning but vary due to external factors. The characterisation identified the largest fraction of COD in the wastewater as particulate matter with 85% of the total COD concentration. Further average shares of particulate fractions are slowly degradable COD (X_B) with 45%, inert particulate COD (X_U) with 31% and heterotrophic biomass (X_{OHO}) with 9%. The average colloidal fraction of COD was determined as 6%. The total soluble fraction (S_{tot}) was found to be 15% of the total COD. Further average soluble fractions are inert soluble COD (S_U) with 8% and readily degradable COD (S_B) with 7%. The readily degradable fraction consists of 2% volatile fatty acids (S_{VFA}) and 5% other readily degradable COD (S_{other}). The sum of all fractions results in a total of 106%, concluding that analyses contain uncertainties.

For nitrogen, SNH_4 is the largest fraction of TN, with a share of 70%. Particulate phosphorus is the largest fraction of TP with a share of 59%. High concentrations of particulate matter in the wastewater are connected to larger precipitation rates. When water is flushed through the pipes, particulate substances are detached from the walls of the pipes and transported to the WWTP. This together with lower water temperatures during rain events and snow melting will lead to changes in the wastewater composition and its fractions.

Comparisons to previous characterisations of wastewater at WWTPs in Sweden showed that the methods and analyses used for obtaining the different fractions for organic matter, nitrogen, and phosphorus varies. Some previous studies used assumptions or calculations to obtain one or several parameters instead of analysing them in the wastewater. There were also differences in the sampling period as well as season of the year when the characterisation was conducted which is another explanation of the differences between the results from the studies. These variations complicated the comparison between studies. Results from previous characterisations of nitrogen and phosphorus fractions were lacking and the result is therefore uncertain. However, comparison between concentrations of influent and effluent water from Rya WWTP show that there are some fractions of nitrogen and phosphorus that to a larger extent remain in the effluent treated water. The removal efficiency of SN_{other} and SP_{other} is 87% respectively 77%, meaning that parts of the fractions still remain in effluent water. With increasing future demands of efficient wastewater treatment, they might be important to consider. For organic matter, the removal efficiency for $COD_{fil,1.6}$ is 73% which is relatively low and should be noticed. Although, the requirements of the effluent water at Rya WWTP is considering BOD.

The variations in amount of I/I and different sewer systems have an impact on the composition of wastewater. With larger volumes of I/I, the wastewater will be diluted due to excess water leading to lower overall concentrations of pollutants. Further on, the results from this study

indicated lower concentrations of S_U with a characterisation conducted in a city with large shares of separated sewers, together with higher concentrations of X_B . Analyses conducted during summer, with warmer temperatures and less precipitation resulted in higher concentrations of S_{tot} , probably due to an increase in hydrolysis where particulate matter is degraded to smaller particles. Since less water is flowing in the pipes, the retention time increases, and the degradation will occur for a longer time. This study, conducted during winter, resulted in low fractions of S_{tot} and S_B .

Instead of treating the wastewater at the WWTP, local measurements such as SUDS's could be implemented. However, when considering these solutions, the cost of the project together with the potential risks for the environment have to be taken into account.

5.1 Recommendations for future studies

Since there was a time constraint when conducting the literature review and laboratory work of this study, there is a need of a broader and more extensive characterisation and analysis of wastewater in Gothenburg. Below, some recommendations for future studies are listed.

- The measurements analysing the degradation of organic matter should be started as quickly as possible after sampling to ensure that the early degradation in the sample is caught.
- In the OUR analysis, either start the sample directly after sampling, or add ethanol as a readily degradable carbon source.
- Save samples to ensure that it is possible to run duplicates and to re-run tests if the results are inaccurate.
- Variations occurring due to time of the day or seasonal changes should be included in future research to be able to obtain robust results.
- If it is possible to add sampling during weekends, this will increase the accuracy of the results.
- The usage of the SDR SensorDish® Reader could be used at a larger scale to determine readily degradable organic matter. A further improvement would be to include several duplicates using the same dilution and making sure the samples are not diluted too much or too little. If the dilution factor is too large, small changes in the dissolved oxygen will cause an overestimation of degradable COD, as seen in Appendix XIII. This analysis can be used as a complement to other BOD-tests.
- Measurements of the effluent water could be conducted on all fractions of COD. If these analyses are made there is a possibility of improving treatment methods for specific fractions in the wastewater.
- The analyses in this study did not investigate different areas in Gothenburg and their possible variations in I/I due to variations in the environment. In future studies, it could be interesting to identify areas with greater I/I entering the sewer system and see if measurements could be taken to mitigate these water flows at a reasonable cost.

- The changes in wastewater composition could have an impact on the choice of processes in the WWTP and therefore, future studies could also include active sludge processes and the impact due to variations in the climate, temperature, and precipitation.
- Further parameters could be included in analyses in the future, e.g. conductivity and turbidity.

6. References

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Appendix I - Calculation of oxygen present in long-term experiment: bottle and wastewater

The calculations of the COD concentrations in the wastewater were conducted using values from Lysberg & Neth (2012) where the maximum concentration was used to estimate the worst case scenario.

Unfiltered wastewater

Maximum COD concentration: 410 mg/L

The bottle used in the experiment was a 0.5L bottle, with 50 ml wastewater, meaning that the COD concentration is multiplied by 0.45L to obtain the oxygen concentration in the bottle.

$$410 \left[\frac{mg}{L} \right] \cdot 0.45[L] = 184.5 [mg]$$

Filtered wastewater

Maximum COD concentration: 100 mg/L

$$100 \left[\frac{mg}{L} \right] \cdot 0.45[L] = 45 [mg]$$

By using the ideal gas law, the oxygen present in the air of the bottle is calculated as follows:

$$PV = nRT \rightarrow n = \frac{pV}{RT}$$

P: atmospheric pressure [Pa]

V: volume [m³] (multiplied by 21% oxygen in air)

N: amount of substance [mol]

R: ideal gas constant [J/mol·K]

T: temperature [K]

$$n = \frac{101.3 \cdot 10^3 \cdot 4.5 \cdot 10^{-4} \cdot 0.21}{8.3144621 \cdot 289.15} = 3.98 \cdot 10^{-3} \text{ mol}$$

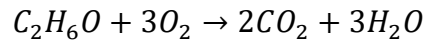
Calculating the amount of oxygen present in milligrams:

$$32 \left[\frac{g}{mol} \right] \cdot 3.98 \cdot 10^{-3} [mol] = 0.127 \text{ g} = 127 \text{ mg}$$

The oxygen concentration needed for a reaction to happen is relatively close to the oxygen present in the bottle. Due to this, the cap was opened every day except on weekends to ensure enough oxygen was present.

Appendix II – Calculation of amount of ethanol used in the OUR – measurement

The formula used for the ethanol reaction is the following:



All the ethanol is supposed to oxidise to be able to identify the drop in OUR, however it should not be a too small amount.

Molar mass:

$$M_{C_2H_6O} = 46.07 \text{ g/mol}$$

$$M_{O_2} = 32 \text{ g/mol}$$

Density:

$$\rho_{C_2H_6O} = 789\,000 \text{ g/m}^3$$

$$\rho_{O_2} = 1429 \text{ g/m}^3$$

O₂:

The bottle used for the analysis contains 300 ml with 100% oxygen meaning approximately 9 mg O₂/L.

$$m_{O_2} = 3 \cdot 10^{-4} \cdot 9 = 2.7 \text{ mg } O_2$$

$$n_{O_2} = \frac{m_{O_2}}{M_{O_2}} = \frac{2.7 \cdot 10^{-3}}{32} = 0.0844 \text{ mmol}$$

C₂H₆O:

The relation of 3O₂ = C₂H₆O gives the following equation:

$$n_{C_2H_6O} = \frac{m_{C_2H_6O}}{M_{C_2H_6O}} = \frac{0.0844}{3} = 0.0281 \text{ mmol}$$

$$m_{C_2H_6O} = 0.0281 \cdot 46.07 = 1.296 \text{ mg}$$

$$V_{C_2H_6O} = \frac{m_{C_2H_6O}}{\rho_{C_2H_6O}} = \frac{1.296 \cdot 10^{-3}}{789\,000} = 16.6 \cdot 10^{-9} \text{ m}^3 = 0.00164 \text{ ml ethanol, 100\%}$$

$$0.00164 \text{ ml 100\% ethanol} = 0.16 \text{ ethanol, 1\%}$$

To ensure that a reaction is happening, 0.2 ml 1% ethanol is used.

Appendix III – Dilution factors in OUR analysis using the SDR SensorDish® Reader

Table III.1. Concentration and dilution for the samples in the SensorDish® Reader.

Sample	Unfiltered wastewater	PBS
1, 2	1/2	1/2
3, 4	1/4	3/4
5, 6	1/8	7/8
7, 8	1/16	15/16
9, 10	1/32	31/32
11, 12	1/64	63/64
13, 14	1/128	127/128
15, 16	1/256	255/256
17, 18	1/512	511/512
19, 20	0	1

Appendix IV – Compiled data from COD analyses of influent wastewater to Rya WWTP

Table IV.1. Results from COD analyses of influent wastewater to Rya WWTP.

Date	Unfiltered influent wastewater							Filtered influent wastewater					Flocculated and filtered influent wastewater		Water after the nitrifying MBBR step			
	COD _{tot} [mg COD/L]	X _{HOH} [mg COD/L]	X _U [mg COD/L]	TSS [mg COD/L]	Non- volatile solids [mg COD/L]	VSS [mg COD/L]	BCOD [mg COD/L]	COD _{fil,0.45} [mg COD/L]	COD _{fil,1.6} [mg COD/L]	Acetate [mg COD/L]	Propionate [mg COD/L]	Butyrate [mg COD/L]	COD _{ff} (S _{HO}) [mg COD/L]	S _U [mg COD/L]	MBBR _{ff} [mg COD/L]	Acetate [mg COD/L]	Propionate [mg COD/L]	Butyrate [mg COD/L]
		Ethanol increase																
21-Feb	310	27.90 ¹	114.1	257	- ²	- ²	161.6 ³	66	90	9.73	0	4.32	41	28	27	0.64	0	2.24
								68					46	32	19			
22-Feb	470	42.30 ¹	143.4	257	33	224	245.1 ³	95	130	17.98	0	14.08	56	41.2	31	1.728	0	0
								93					58		29			
23-Feb	420	37.80 ¹	128.2	261	45	216	219.0 ³	86	130	21.21	0	16.64	64	36.8	22	1.408	0	2.4
								90					75		25			
24-Feb	390	35.10 ¹	119.0	618	50	568	203.4 ³	92	110	8.32	0	15.2	76	34.2	28	0.704	0	1.76
								94					78		28			
25-Feb	370	33.30 ¹	112.9	178	27	151	192.9 ³	64	100	11.65	0	14.24	59	32.4	44	1.6	1.456	0
								79					52		57			
28-Feb	370	33.30 ¹	112.9	236	25	211	192.9 ³	86	130	10.43	0	14.24	56	32.4	42	2.816	0	0
								87					57		46			
01-Mar	490	53.90 ¹	149.5	280	35	245	255.5 ³	100	140	9.73	0	14.56	66	42.9	67	4.032	1.456	0
								104					63		63			
02-Mar	400	36.09	122.1	218	37	181	208.6 ³	88	140	0	0	17.6	60	35.0	63	1.344	0	0
								94					62		67			
03-Mar	420	60.70	128.2	294	39	255	219.0 ³	96	130	4.67	0	16.96	68	36.8	66	1.088	0	0
								92					70		64			
04-Mar	440	30.89	106.5	212	31	181	229.4 ³	96	130	13.70	0	16.16	53	35	33	0.512	0	0
								90					56	34	33			
07-Mar	410	36.90 ¹	125.1	224	27	197	226.7	106	140	0	2.24	11.36	102	35.9	32	0.832	1.792	0
								106					105		35			
08-Mar	620	52.40	189.2	360	51	309	366.6	102	170	0	0	11.36	77	54.3	39	1.204	1.456	0
								112					82		40			
09-Mar	440	32.86	134.3	238	45	193	229.6	99	130	0	2.13	2.56	72	38.5	37	3.84	1.456	1.76
								105					74		36			
10-Mar	480	45.83	146.5	266	43	223	262.2	104	180	0	1.57	12.32	69	42.0	33	0.96	0	0
								105					65		33			
11-Mar	470	25.81	143.4	368	89	279	185.5	50	76	0.14	0	0.057	26	7.97 ⁴	6	0.028	0	0.011
								46					34		10			

¹On these dates, no ethanol was added to the sample and therefore, the value is calculated using the average percentage fraction of the remaining concentrations.

²Error at the laboratory, no data was reported at this date.

³The concentration of BCOD was calculated using the average percentage fraction of the analysed samples.

⁴The concentration of S_U was calculated using MBBR_{ff} on this date.

Appendix V – Precipitation data from SMHI

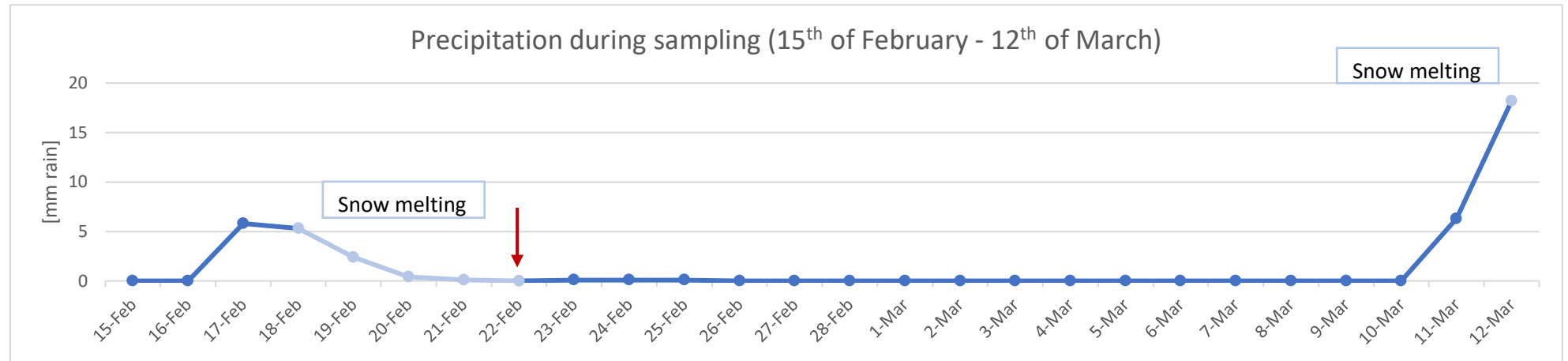


Figure V.1. Precipitation during the sampling period with an arrow indicating the start date of the sampling (SMHI, n.d.).

Appendix VI – Results from the analyses of TSS and VSS

Equation (2) was used to calculate VSS from the results of TSS.

Table VI.1. COD concentrations of TSS and VSS in influent wastewater to Rya WWTP.

Date	Unfiltered influent wastewater		
	TSS [mg COD/L]	Non-volatile solids [mg COD/L]	VSS [mg COD/L]
21-Feb	257	- ¹	-
22-Feb	257	33	224
23-Feb	261	45	216
24-Feb	618	50	568
25-Feb	178	27	151
28-Feb	236	25	211
01-Mar	280	35	245
02-Mar	218	37	181
03-Mar	294	39	255
04-Mar	212	31	181
07-Mar	224	27	197
08-Mar	360	51	309
09-Mar	238	45	193
10-Mar	266	43	223
11-Mar	368	89	279
Average	284.5	42	246
Standard deviation	105.4	16.8	101.6

¹No data was reported from the laboratory at this date.

Appendix VII – Calculation of S_U and S_B

a.

Table VII.1. Estimation of S_U by using the values obtained from the long-term BOD test. The average of 0.0876 was multiplied with COD_{tot} for the specific date.

Date	COD_{tot} [mg COD/ L]	S_U (from 10 days analysis)	Fraction of total COD (S_U/COD_{tot})	Average fraction	Average fraction multiplied with COD_{tot}	Calculated S_U [mg COD/L]	S_U from MBBR (average) ¹ [mg COD/L]
21-Feb	310	28	0.0903	0.0968		30 ²	29.36
		32	0.1032				
22-Feb	470					41.2	28.27
23-Feb	420					36.8	22.09
24-Feb	390					34.2	27.30
25-Feb	370				0.0876	32.4	47.44
28-Feb	370					32.4	41.18
01-Mar	490					42.9	59.51
02-Mar	400					35.0	63.66
03-Mar	420					36.8	63.91
04-Mar	440	35	0.0795	0.0784		34.5 ²	33.99
		34	0.0773				
07-Mar	410					35.9	36.88
08-Mar	620					54.3	36.84
09-Mar	440					38.5	31.20
10-Mar	480					42.0	32.04
11-Mar	470					41.2 ³	7.97
Average							37.4
Standard deviation						6.20	15.6

¹Due to high standard deviation, these values were not used for S_U except for the concentration on the 11th of March.

²The value from the long-term BOD test was used here instead of the calculated concentration.

³The concentration of S_U exceeds S_{tot} so on this date, the concentration of S_U calculated using MBBR_f is used.

b.

Equation (4) was used to obtain S_U from the $MBBR_{ff}$ fraction. The values were not used for S_U in the results except for the concentration on the 11th of March.

Table VII.2. Calculation of S_U using average values of $MBBR_{ff}$ and $S_{Acetate}$.

Date	Water after the nitrifying MBBR step		Inert soluble
	$MBBR_{ff}$ [mg COD/L]	$S_{Acetate}$ [mg COD/L]	S_U [mg COD/L]
21-Feb	30 ¹	0.64	29.36
22-Feb	30	1.73	28.27
23-Feb	23.5	1.41	22.09
24-Feb	28	0.70	27.30
25-Feb	50.5	1.60	48.90
28-Feb	44	2.82	41.18
01-Mar	65	4.03	60.97
02-Mar	65	1.34	63.66
03-Mar	65	1.09	63.91
04-Mar	34.5 ¹	0.51	33.99
07-Mar	39.5	0.83	38.67
08-Mar	39.5	1.20	36.30
09-Mar	36.5	3.84	32.66
10-Mar	33	0.96	32.04
11-Mar	8	0.03	7.97 ²
Average	39.5	1.5	38.0
Standard deviation	16.3	1.2	15.8

¹The values from the long-term BOD test was used on the 21st of February and the 4th of March.

²The concentration of S_U on the 11th of March was used to calculate S_B and $BCOD$.

c.

Equation (5) was used to obtain concentrations of S_B .

Table VII.3. Concentration of S_B calculated by subtracting S_U from S_{tot} .

Date	Influent wastewater		
	S_{tot} [mg COD/L]	S_U [mg COD/L]	S_B [mg COD/L]
21-Feb	43.5	30	13.5
22-Feb	57	41.17	15.83
23-Feb	69.5	36.79	32.71
24-Feb	77	34.16	42.84
25-Feb	55.5	32.41	23.09
28-Feb	56.5	32.41	24.09
01-Mar	64.5	42.92	21.58
02-Mar	61	35.04	25.96
03-Mar	69	36.79	32.21
04-Mar	54.5	34.50	20.00
07-Mar	103.5	35.91	67.59
08-Mar	79.5	54.31	25.19
09-Mar	73	38.54	34.46
10-Mar	67	42.04	24.96
11-Mar	30	7.97 ¹	0 ¹
Average	64.1	37.9	28.9
Standard deviation	16.8	6.0	13.5

¹The concentration of the calculated S_U from the long-term analysis was larger than S_{tot} at this date. The concentration of S_U calculated using $MBBR_{ff}$ is used instead, found in Appendix VIIb.

Appendix VIII – Result from the HPLC analysis showing S_{VFA} in influent wastewater and water after the nitrifying MBBR step at Rya WWTP

Table VIII.1. Concentrations of S_{VFA} in filtered influent wastewater and water after the nitrifying MBBR step.

Date	Filtered influent wastewater				Water after the nitrifying MBBR step			
	$S_{Acetate, in}$ [mg COD/L]	$S_{Propionate, in}$ [mg COD/L]	$S_{Butyrate, in}$ [mg COD/L]	$S_{VFA, tot, in}$ [mg COD/L]	$S_{Acetate, MBBR}$ [mg COD/L]	$S_{Propionate, MBBR}$ [mg COD/L]	$S_{Butyrate, MBBR}$ [mg COD/L]	$S_{VFA, tot, MBBR}$ [mg COD/L]
21-Feb	9.73	0	4.32	14.05	0.64	0	2.24	2.88
22-Feb	17.98	0	14.08	32.06	1.73	0	0	1.73
23-Feb	21.21	0	16.64	37.85	1.41	0	2.40	3.81
24-Feb	8.32	0	15.20	23.52	0.70	0	1.76	2.46
25-Feb	11.65	0	14.24	25.89	1.60	1.46	0	3.06
28-Feb	10.43	0	14.24	24.67	2.82	0	0	2.82
01-Mar	9.73	0	14.56	24.29	4.03	1.46	0	5.49
02-Mar	0	0	17.60	17.60	1.34	0	0	1.34
03-Mar	4.67	0	16.96	21.63	1.09	0	0	1.09
04-Mar	13.70	0	16.16	29.86	0.51	0	0	0.51
07-Mar	0	2.24	11.36	13.60	0.83	1.79	0	2.62
08-Mar	0	0	11.36	11.36	1.20	1.46	0	2.66
09-Mar	0	2.13	2.56	4.69	3.84	1.46	1.76	7.06
10-Mar	0	1.57	12.32	13.89	0.96	0	0	0.96
11-Mar	0.14	0	0.057	0.19	0.028	0	0.011	0.039
Average	7.2	0.4	12.1	7.6	1.5	0.5	0.5	2.6
Standard deviation	7.1	0.8	5.5	10.3	1.2	0.8	1.0	1.9

Appendix IX – Calculation of S_{other}

Equation (3) was used to obtain concentrations of S_{other} .

Table IX.1. Calculation of S_{other} using subtracting concentrations of $S_{Acetate}$ and S_U from S_{tot} .

Date	Influent wastewater			
	S_{tot} [mg COD/L]	$S_{Acetate, in}$ [mg COD/L]	S_U [mg COD/L]	S_{other} [mg COD/L]
21-Feb	43.5	9.73	30	3.77
22-Feb	57	17.98	41.17	0 ¹
23-Feb	69.5	21.21	36.79	11.50
24-Feb	77	8.32	34.16	34.52
25-Feb	55.5	11.65	32.41	11.44
28-Feb	56.5	10.43	32.41	13.66
01-Mar	64.5	9.73	42.92	11.85
02-Mar	61	0	35.04	25.96
03-Mar	69	4.67	36.79	27.54
04-Mar	54.5	13.70	34.50	6.30
07-Mar	103.5	0	35.91	67.59
08-Mar	79.5	0	54.31	25.19
09-Mar	73	0	38.54	34.46
10-Mar	67	0	42.04	24.96
11-Mar	30	0.14	7.97 ²	21.89
Average	64.1	7.2	35.7	21.4
Standard deviation	16.8	7.1	9.7	16.7

¹The concentration of $S_U + S_{Acetate}$ exceeded S_{tot} and therefore, S_{other} is set to zero.

²The concentration of the calculated S_U from the long-term analysis was larger than S_{tot} at this date. The concentration of S_U calculated using $MBBR_{ff}$ is used instead, found in Appendix VIIb.

Appendix X – Calculation of C_{tot}

Equation (1) was used to obtain concentrations of C_{tot} .

Table X.1. Colloidal fraction in the influent wastewater, calculated subtracting the average value of COD_{ff} from the average value of $COD_{filt,0.45}$.

Date	Filtered influent wastewater		Flocculated and filtered influent wastewater		Colloidal fraction of influent wastewater	
	$COD_{filt,0.45}$ [mg COD/L]	Average $COD_{filt,0.45}$ [mg COD/L]	COD_{ff} [mg COD/L]	Average COD_{ff} [mg COD/L]	C_{tot} [mg COD/L]	Percentage of COD_{tot} [%]
21-Feb	66	67	41	43.5	23.5	8%
	68		46			
22-Feb	95	94	56	57	37	8%
	93		58			
23-Feb	86	88	64	69.5	18.5	4%
	90		75			
24-Feb	92	93	76	77	16	4%
	94		78			
25-Feb	64	71.5	59	55.5	16	4%
	79		52			
28-Feb	86	86.5	56	56.5	30	8%
	87		57			
01-Mar	100	102	66	64.5	37.5	8%
	104		63			
02-Mar	88	91	60	61	30	8%
	94		62			
03-Mar	96	94	68	69	25	6%
	92		70			
04-Mar	96	93	53	54.5	38.5	9%
	90		56			
07-Mar	106	106	102	103.5	26.0 ¹	6% ¹
	106		105			
08-Mar	102	107	77	79.5	27.5	4%
	112		82			
09-Mar	99	102	72	73	29	7%
	105		74			
10-Mar	104	104.5	69	67	37.5	8%
	105		65			
11-Mar	50	48	26	30	18	4%
	46		34			
Standard deviation	16.3	16.7	10.8	6.1	10.2	
Average percentage [%]						6%

¹The concentration of C_{tot} was very low on this date and was replaced by a calculation using the average percentage fraction of the remaining dates.

Appendix XI – Calculation of X_{tot}

Equation (1) was used to obtain concentrations of X_{tot} .

Table XI.1. Concentration of X_{tot} in influent wastewater.

Date							
	COD_{tot} [mg COD/L]	C_{tot} [mg COD/L]	C_{tot}/COD_{tot} [%]	S_{tot} [mg COD/L]	S_{tot}/COD_{tot} [%]	X_{tot} [mg COD/L]	X_{tot}/COD_{tot} [%]
21-Feb	310	23.5	8%	43.5	14%	243	78%
22-Feb	470	37	8%	57	12%	376	80%
23-Feb	420	18.5	4%	69.5	17%	332	79%
24-Feb	390	16	4%	77	20%	297	76%
25-Feb	370	16	4%	55.5	15%	298.5	81%
28-Feb	370	30	8%	56.5	15%	283.5	77%
01-Mar	490	37.5	8%	64.5	13%	388	79%
02-Mar	400	30	8%	61	15%	309	77%
03-Mar	420	25	6%	69	16%	326	78%
04-Mar	440	38.5	9%	54.5	12%	347	79%
07-Mar	410	2,5	1%	103.5	25%	304	74%
08-Mar	620	27.5	4%	79.5	13%	513	83%
09-Mar	440	29	7%	73	17%	338	77%
10-Mar	480	37.5	8%	67	14%	375.5	78%
11-Mar	470	18	4%	30	6%	422	90%
Average	433.3	25.8	6%	64.1	15%	343.5	79%
Standard deviation	70.8	10.2	2%	16.8	6%	65.5	4%

Appendix XII – Calculation of BCOD

Equation (9) and (10) was used to obtain the concentration of BCOD. MATLAB was utilised to obtain the results with different values for the different days found in Table XII.1.

Table XII.1. Concentration of BOD-test conducted by Göteborgs Kemanalys.

Date	Sample notation	Days of sampling				
		1	2	4	7	9
07-Mar	1	51	72	122	153	177
08-Mar	2	61	99	173	244	257
09-Mar	3	45	72	125	151	178
10-Mar	4	56	88	157	177	211
11-Mar	5	40	59	109	109	155

The following MATLAB script was used to obtain the concentration of BOD_{tot} and k_{BOD} .

```
%% Linear regression function S1

function fErr1 = lsqFun(phi)

%Data from sample 1
t = [1 2 4 7 9]';
BODt_1 = [36 73 85 147 161]';

%Parameters
kBOD_1 = phi(1);
BODtot_1 = phi(2);

%Computing error
S1model = BODtot_1 * (1-exp(-kBOD_1*t));
fErr1 = BODt_1-S1model;
```

```
%% For solving using lsqnonlin

phi0 = [1, 36]; %initial values
phi = lsqnonlin(@ (p) lsqFun(p), phi0);
```

```
%% BODtot plot S1

t = [1 2 4 7 9]';
BODt_1 = [36 73 85 147 161]';

% BCOD
fBOD=0.15; %constant
BCOD1 = (1/(1-fBOD))*phi(2);
```



```

plot(t,S1, 'o b','LineWidth',1)
hold on
y=yline(BCOD1,'- m','LineWidth',1.5);
title ('Sample 1')
xlabel ('[days]')
ylabel ('[mg O2/L]')
axis([0 40 0 270])

t=1:40;
plot(t, 1.*(1-exp(-t*phi(1)))*phi(2), 'b')

text(20,220,sprintf('BCOD: %.2f\n', BCOD1));
text(20,150,sprintf('kBOD: %.2f\nBODt: %.2f\n', phi(1), phi(2)));

```

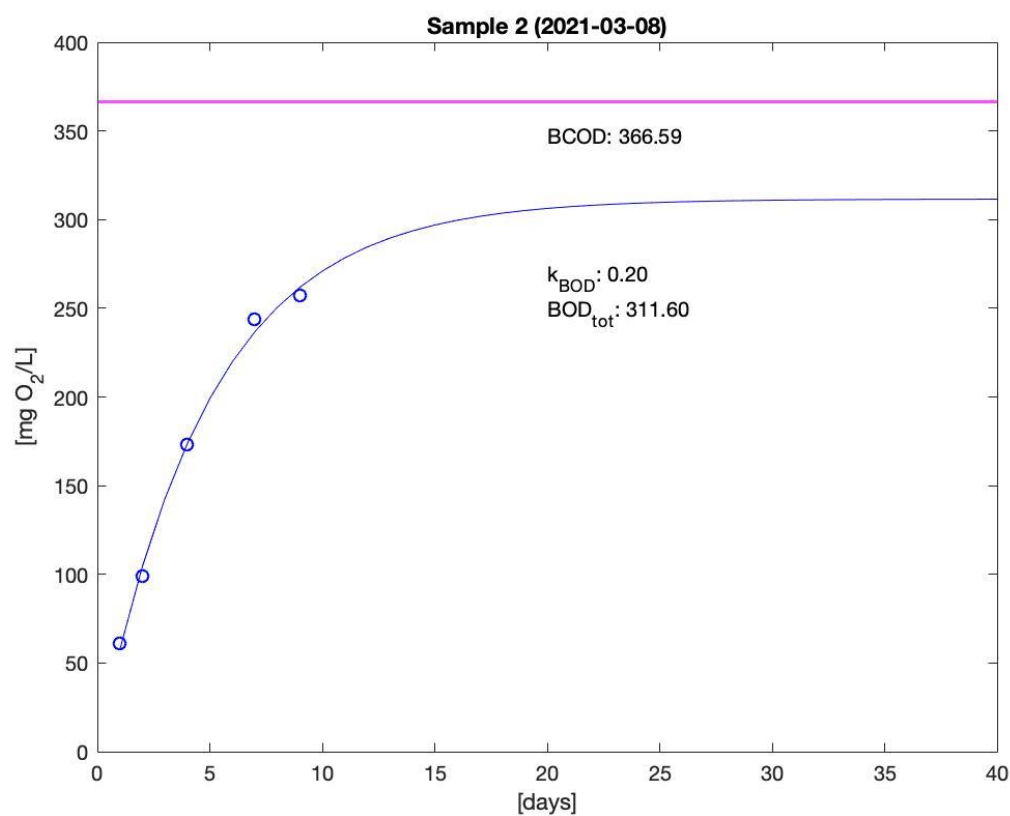


Figure XII.1. Identification of k_{BOD} , BOD_{tot} and BCOD on the 8th of March.

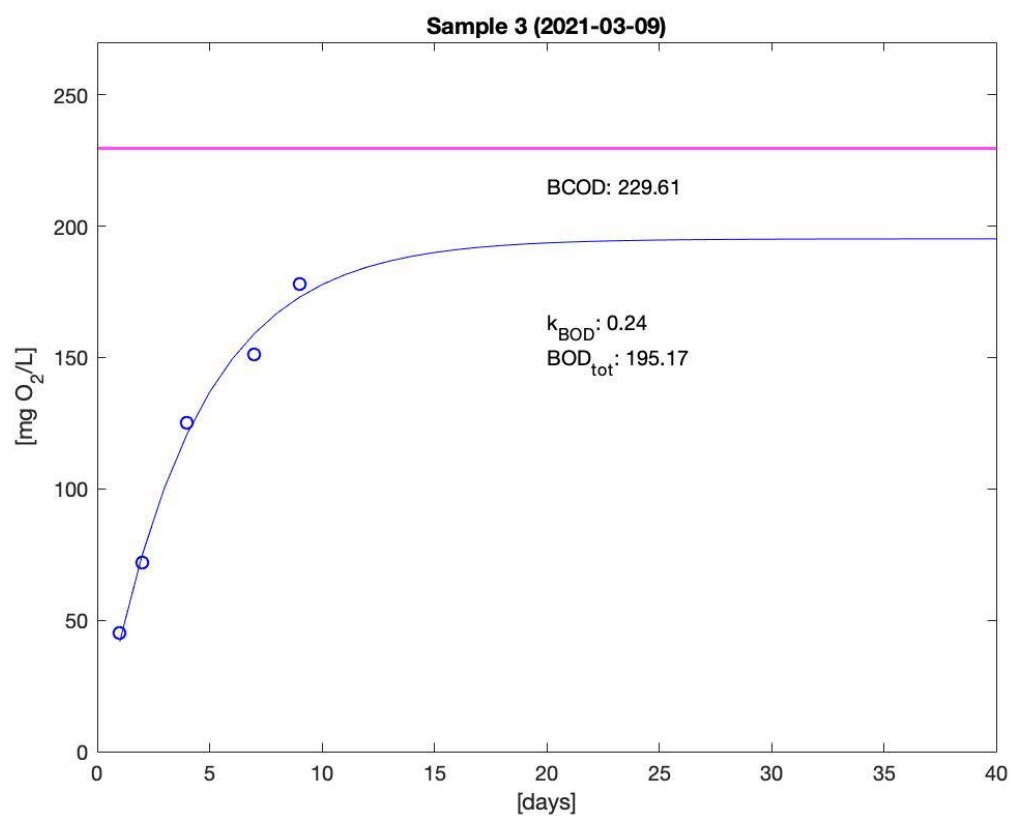


Figure XII.2. Identification of k_{BOD} , BOD_{tot} and BCOD on the 9th of March.

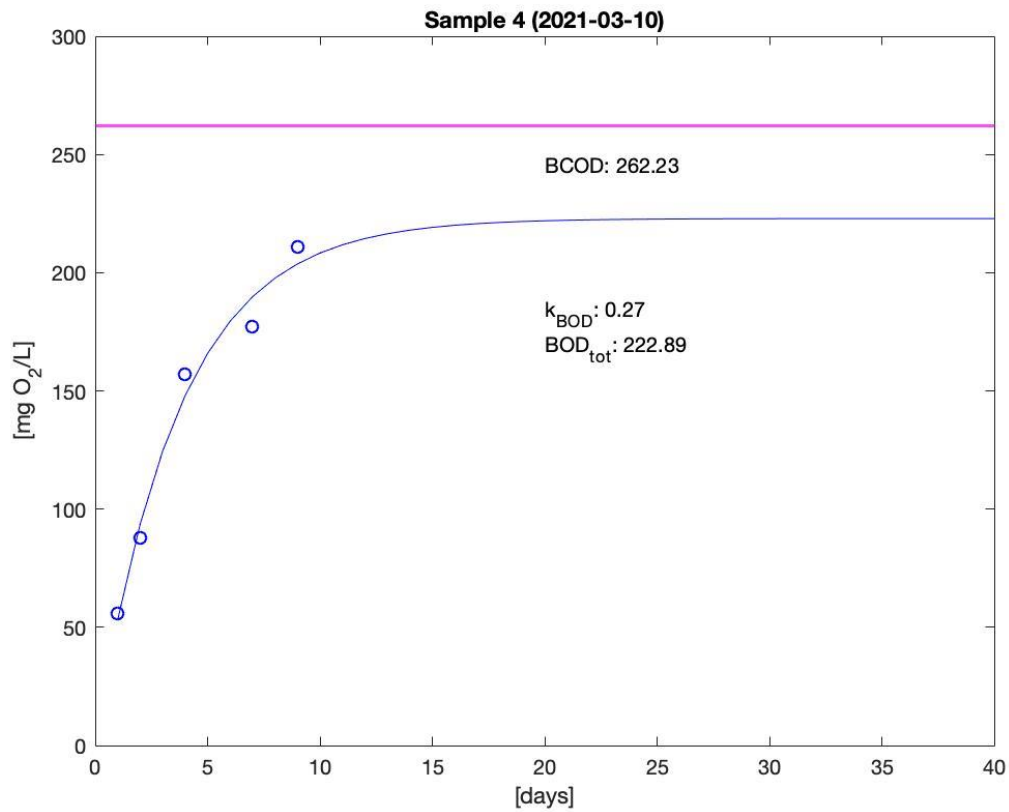


Figure XII.3. Identification of k_{BOD} , BOD_{tot} and $BCOD$ on the 10th of March.

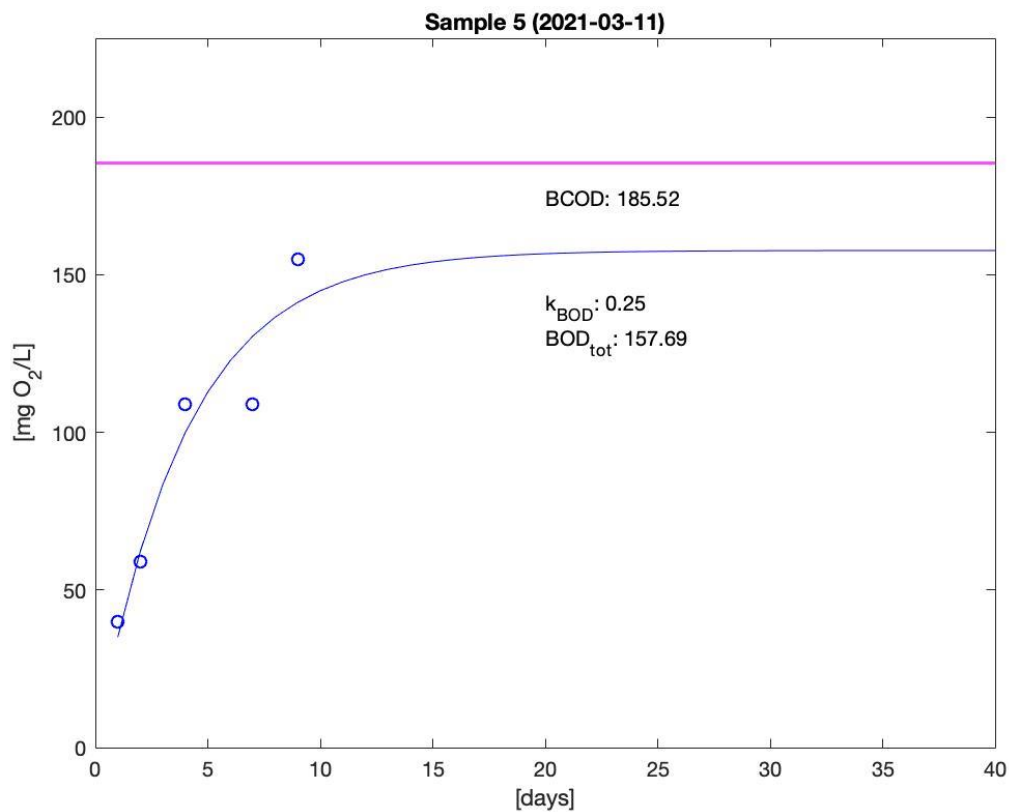


Figure XII.4. Identification of k_{BOD} , BOD_{tot} and $BCOD$ on the 11th of March.

Appendix XIII – Biodegradable COD concentrations from the SDR SensorDish® Reader analysis

The concentration of biodegradable COD, referred to as BOD_{SDR}, was calculated by dividing the difference in DO with the dilution factor. The samples marked in bold writing were further studied in the BCOD analysis since they were considered the most representative.

Table XIII.1. Concentrations of biodegradable COD from the SDR SensorDish® Reader analysis.

Sample	Dilution factor	PBS	Max DO	Min DO	Difference	BOD _{SDR} [mg COD/L]
1	1/2	1/2	8.82	0.21	8.61	17.22
2	1/2	1/2	8.65	0.23	8.42	16.84
3	1/4	3/4	8.85	3.43	5.42	21.68
4	1/4	3/4	8.61	3.7	4.91	19.64
5	1/8	7/8	8.74	5.21	3.53	28.24
6	1/8	7/8	8.86	5.59	3.27	26.16
7	1/16	15/16	8.91	7.04	1.87	29.92
8	1/16	15/16	8.88	6.4	2.48	39.68
9	1/32	31/32	8.92	7.04	1.88	60.16
10	1/32	31/32	8.74	7.34	1.4	44.80
11	1/64	63/64	9.02	7.7	1.32	84.48
12	1/64	63/64	9.1	8.06	1.04	66.56
13	1/128	127/128	8.66	7.26	1.4	179.2
14	1/128	127/128	8.97	8.02	0.95	121.60
15	1/256	255/256	9.21	8.27	0.94	240.64
16	1/256	255/256	9.03	8	1.03	263.68
17	1/512	511/512	9.17	7.68	1.49	762.88
18	1/512	511/512	9.17	7.82	1.35	691.20

Appendix XIV – Calculation of X_U

Equation (12) was used to obtain concentrations of X_U .

Table XIV.1. Estimation of X_U by using the values obtained from the long-term BOD test. The average of 0.44 was multiplied with COD_{tot} for the specific date.

Date	$COD_{tot, i}$	Calculated S_U	$COD_{tot, 10}$	Fraction ($COD_{tot, 10} / COD_{tot, i}$)	Average fraction	Average percentage [%]	Calculated X_U (44%) ($COD_{tot, 10}$)	Final X_U
21-Feb	310	30	132	0.43	0.52		160 ¹	114.14
			188	0.61				
22-Feb	470	41.17					208.88	143.43
23-Feb	420	36.79					186.66	128.17
24-Feb	390	34.16					173.33	119.01
25-Feb	370	32.41					164.44	112.91
28-Feb	370	32.41				0.44	164.44	112.91
01-Mar	490	42.92					217.77	149.53
02-Mar	400	35.04					177.77	122.06
03-Mar	420	36.79					186.66	128.17
04-Mar	440	34.5	172	0.39	0.37		164 ¹	106.53
			156	0.35				
07-Mar	410	35.91					182.22	125.12
08-Mar	620	54.31					275.55	189.20
09-Mar	440	38.54					195.55	134.27
10-Mar	480	42.04					213.33	146.48
11-Mar	470	41.17					208.88	143.43
Average								135.0
Standard deviation								20.4

¹The value from the long-term BOD test was used here instead of the calculated concentration.

Appendix XV – Calculation of X_B

Equation (11) was used to obtain the concentrations of X_B .

Table XV.1. Concentrations of X_B calculated from BCOD-concentrations.

Date					
	COD_{tot} [mg/L]	S_B [mg/L]	BCOD [mg/L]	BCOD/C_{tot} [%]	X_B [mg/L]
21-Feb	310	13,5	161.63 ¹	52%	148.13
22-Feb	470	15,83	245.06 ¹	52%	229.23
23-Feb	420	32,71	218.99 ¹	52%	186.28
24-Feb	390	42,84	203.35 ¹	52%	160.51
25-Feb	370	23,09	192.92 ¹	52%	169.83
28-Feb	370	24,09	192.92 ¹	52%	168.83
01-Mar	490	21,58	255.49 ¹	52%	233.91
02-Mar	400	25,96	208.56 ¹	52%	182.60
03-Mar	420	32,21	218.99 ¹	52%	186.78
04-Mar	440	20,00	229.42 ¹	52%	209.42
07-Mar	410	67,59	226.67	55%	159.08
08-Mar	620	25,19	366.59	59%	341.40
09-Mar	440	34,46	229.61	52%	195.15
10-Mar	480	24,96	262.23	55%	237.27
11-Mar	470	7.97	185.52	39%	163.49 ²
Average	433.3	28.9	159.2	52%	198.1
Standard deviation	70.8	13.5	40.2	4%	48.7

¹The concentration of BCOD was calculated using the average percentage fraction of 52% of the analysed samples.

²The concentration of S_B was calculated using concentrations on S_U obtained from MBBR_{ff}, found in Appendix VIIb. Since the concentration of S_B was negative at this date the value is set to zero. This will affect the accuracy of the concentration of X_B .

Appendix XVI – Calculation of X_{OHO} from the OUR analysis

Equation (13) was used to obtain the concentrations of X_{OHO} .

Table XVI.1. Concentration of X_{OHO} obtained from the OUR analysis.

Date		Fraction of X_{OHO}/COD_{tot}	Unfiltered influent wastewater
	COD_{tot} [mg COD/L]	[%]	X_{OHO} [mg COD/L]
21-Feb	310	9%	27.90 ¹
22-Feb	470	9%	42.30 ¹
23-Feb	420	9%	37.80 ¹
24-Feb	390	9%	35.10 ¹
25-Feb	370	9%	33.30 ¹
28-Feb	370	9%	33.30 ¹
01-Mar	490	9%	53.90 ¹
02-Mar	400	9%	36.09
03-Mar	420	14%	60.70
04-Mar	440	7%	30.89
07-Mar	410	9%	36.90 ¹
08-Mar	620	8%	52.40
09-Mar	440	7%	32.86
10-Mar	480	10%	45.83
11-Mar	470	5%	25.81
Average		9%	41.7
Standard deviation			10.1

¹These analyses were conducted without adding ethanol and are therefore calculated using the average percentage fraction of 9% from the remaining days.

Appendix XVII – Percentage fractions in influent wastewater

Table XVII.1. Percentage of each fraction in the influent wastewater in relation to COD_{tot} .

Date	S_U [mg COD/L]	$S_{Acetate}$ [mg COD/L]	S_{other} [mg COD/L]	C_{tot} [mg COD/L]	X_B [mg COD/L]	X_U [mg COD/L]	X_{OHO} [mg COD/L]	Tot percentage [%]
21-Feb	10%	3%	1%	8%	48% ⁴	37%	9% ⁶	115%
22-Feb	9% ¹	4%	0%	8%	49% ⁴	31% ⁵	9% ⁶	109%
23-Feb	9% ¹	5%	3%	4%	44% ⁴	31% ⁵	9% ⁶	105%
24-Feb	9% ¹	2%	9%	4%	41% ⁴	31% ⁵	9% ⁶	105%
25-Feb	9% ¹	3%	3%	4%	46% ⁴	31% ⁵	9% ⁶	105%
28-Feb	9% ¹	3%	4%	8%	46% ⁴	31% ⁵	9% ⁶	109%
01-Mar	9% ¹	2%	2%	8%	48% ⁴	31% ⁵	11%	110%
02-Mar	9% ¹	0%	6%	8%	46% ⁴	31% ⁵	9%	108%
03-Mar	9% ¹	1%	7%	6%	44% ⁴	31% ⁵	14%	112%
04-Mar	8%	3%	1%	9%	48% ⁴	24%	7%	100%
07-Mar	9% ¹	0%	16%	6% ³	39% ⁴	31% ⁵	9% ⁶	109%
08-Mar	9% ¹	0%	4%	4%	55%	31% ⁵	8%	111%
09-Mar	9% ¹	0%	7%	7%	44%	31% ⁵	7%	105%
10-Mar	9% ¹	0%	5%	8%	49%	31% ⁵	10%	111%
11-Mar	2% ²	0%	5%	4%	35%	31% ⁵	5%	81%
Average	8%	2%	5%	6%	45%	31%	9%	106%
Standard deviation	2%	2%	4%	2%	5%	2%	2%	
	Soluble ⁷			Colloidal	Particular			
	15%			6%	85%			106%

¹ S_U was on these dates calculated using the average percentage fraction of the concentrations from the analysed dates.

²The concentration of S_U was calculated using concentrations of $MBBR_{ff}$ and is therefore uncertain. This value was used to calculate S_B and thereafter $BCOD$.

³The colloidal fraction was on this date calculated using the average percentage fraction of the remaining dates.

⁴ X_B was on these dates calculated using the average percentage fraction of $BCOD$ concentrations.

⁵ X_U was on these dates calculated using an average percentage fraction of the concentrations from the analysed dates.

⁶Since no ethanol was added on these dates, X_{OHO} was calculated using an average percentage fraction of the concentrations from the analysed date.

⁷The soluble fraction consists of S_U and S_B , where $S_B = S_{Acetate} + S_{other}$.

Appendix XVIII – Concentrations of COD in effluent wastewater

Table XVIII.1. COD concentrations in effluent wastewater.

Date	Unfiltered effluent wastewater	Filtered effluent wastewater
	COD _{tot,eff} [mg COD/L]	COD _{filt,1.6,eff} [mg COD/L]
21-Feb	30	28
22-Feb	32	29
23-Feb	34	33
24-Feb	35	33
25-Feb	38	36
28-Feb	46	44
01-Mar	35	36
02-Mar	35	36
03-Mar	36	33
04-Mar	37	33
07-Mar	36	31
08-Mar	36	33
09-Mar	35	33
10-Mar	35	33
11-Mar	37	31
Average	36	33
Standard deviation	3.5	3.7

Appendix XIX – Results from nitrogen analyses in influent wastewater

Equation (17), (18) and (19) was used to obtain the concentrations of nitrogen fractions.

Table XIX.1. Concentrations of nitrogen fractions in influent wastewater.

Date	Unfiltered influent wastewater		Filtered influent wastewater			
	TN [mg N/L]	X _N [mg N/L]	TN _{filt} [mg N/L]	S _{NH4} [mg N/L]	S _{NO2} /S _{NO3} [mg N/L] ¹	S _{N,other} [mg N/L]
21-Feb	28	4	1.4	19.5	0.24	4.26
22-Feb	35	6	1.9	23.0	<0.10	6.00
23-Feb	36	5	1.5	23.7	<0.10	7.30
24-Feb	29	0	1.7	22.2	<0.10	6.80
25-Feb	34	5	1.7	23.0	<0.10	6.00
28-Feb	38	4	2.1	26.8	<0.10	7.20
01-Mar	41	8	2.1	27.9	<0.10	5.10
02-Mar	38	3	1.8	29.4	<0.10	5.60
03-Mar	40	4	1.9	28.8	<0.10	7.20
04-Mar	39	4	2.0	28.8	<0.10	6.20
07-Mar	41	6	2.5	28.2	0.10	6.70
08-Mar	33	-3 ²	2.3	29.2	<0.10	6.80
09-Mar	43	6	1.7	29.4	<0.10	7.60
10-Mar	45	6	2.3	29.9	<0.10	9.10
11-Mar	30	13	1.1	13.6	0.83	2.57
Average	37	5	32	26	0	6
Average percentage fraction of TN	100%	13%	87%	70%	0%	17%
Standard deviation	5.1	3.5	0.4	4.7	0.4	1.5

¹The uncertainty of the measurement is 16%.

²On the 7th of March, TN_{filt} was larger than TN resulting in a negative value for X_N. Since the analyses are connected, all results from this day might be faulty and therefore these values are not trustworthy.

Appendix XX – Results from phosphorus analyses in influent wastewater

Equation (14), (15) and (16) was used to obtain the concentrations of phosphorus fractions.

Table XX.1. Concentrations of phosphorus fractions in influent wastewater.

Date	Unfiltered influent wastewater		Filtered influent wastewater		
	TP [mg P/L]	X _P [mg P/L]	TP _{filt} [mg P/L]	PO ₄ [mg P/L]	S _{P,other} [mg P/L]
21-Feb	3.3	1.9	1.4	1.14	0.26
22-Feb	4.3	2.4	1.9	1.44	0.46
23-Feb	4.2	2.7	1.5	1.15	0.35
24-Feb	4.1	2.4	1.7	1.28	0.42
25-Feb	4.0	2.3	1.7	1.30	0.40
28-Feb	4.1	2.0	2.1	1.60	0.50
01-Mar	5.0	2.9	2.1	1.80	0.30
02-Mar	4.5	2.7	Non-detected ¹	Non-detected ¹	Non-detected ¹
03-Mar	4.7	2.8	1.9	1.39	0.51
04-Mar	4.7	2.7	2.0	1.48	0.52
07-Mar	4.8	2.3	2.5	1.87	0.63
08-Mar	5.8	3.5	2.3	1.80	0.50
09-Mar	5.1	3.4	1.7	1.30	0.40
10-Mar	5.4	3.1	2.3	1.80	0.50
11-Mar	3.7	2.6	1.1	1.03	0.07
Average	4.5	2.6	1.9	1.5	0.4
Average percentage fraction of TP	100%	59%	39%	30%	9%
Standard deviation	0.7	0.5	0.4	0.3	0.2

¹On the 2nd of March, the PO₄ fraction was larger than the TP_{filt} fraction and resulted in a negative concentration. Therefore, this value was set to non-detective and noted as zero.

Appendix XXI – Compiled data from nitrogen and phosphorus analyses of effluent wastewater

Table XXI.1. Concentrations of nitrogen and phosphorus fractions in effluent wastewater.

Date	Unfiltered effluent wastewater		Filtered effluent wastewater						
	TN [mg N/L]	TP [mg P/L]	TN _{filt} [mg N/L]	S _{NH4} [mg N/L]	S _{NO2/SNO3} [mg N/L]	S _{N,other} [mg N/L]	TP _{filt} [mg P/L]	S _{P,other} [mg P/L]	PO ₄ [mg P/L]
21-Feb	6.9	0.23	6.70	5.10	1.21	0.39	0.16	0.06	0.10
22-Feb	6.0	0.19	5.70	4.17	1.03	0.50	0.14	0.06	0.08
23-Feb	6.1	0.17	5.90	3.92	1.13	0.85	0.12	0.07	0.05
24-Feb	6.0	0.17	5.80	3.64	1.18	0.98	0.11	0.06	0.05
25-Feb	5.7	0.15	5.60	3.07	1.34	1.19	0.10	0.06	0.04
28-Feb	4.6	0.20	4.60	2.74	1.41	0.45	0.15	0.08	0.07
01-Mar	5.0	0.20	5.00	2.75	1.40	0.85	0.16	0.08	0.08
02-Mar	5.2	0.19	5.00	2.57	1.35	1.08	0.14	0.07	0.07
03-Mar	5.1	0.17	4.60	2.65	1.30	0.65	0.12	0.07	0.05
04-Mar	5.2	0.18	5.10	2.79	1.37	0.94	0.12	0.07	0.05
07-Mar	5.4	0.22	5.30	2.97	1.26	1.07	0.18	0.08	0.10
08-Mar	4.6	0.22	4.70	2.55	1.08	1.07	0.18	0.08	0.10
09-Mar	4.3	0.18	4.30	1.88	1.51	0.91	0.14	0.07	0.07
10-Mar	5.2	0.16	5.10	2.49	1.57	1.04	0.12	0.07	0.05
11-Mar	10.4	0.29	Non-detected ¹	8.12	0.94	- ²	0.17	0.07	0.10
Average	5.7	0.2	4.9	3.4	1.3	0.9	0.1	0.07	0.07
Standard deviation	1.5	0.04	1.5	1.5	0.2	0.3	0.03	0.008	0.02

¹ No data from the laboratory was obtained this day.

² Since no data was found of TN_{filt}, the concentration of S_{N,other} could not be obtained

Appendix XXII – Removal efficiency of organic matter, nitrogen, and phosphorus from Rya WWTP

The removal efficiency was obtained by subtracting the division of the effluent concentration and the influent concentration from one.

Table XXII.1. Removal efficiency in percentage for organic matter, nitrogen, and phosphorus, comparing influent and effluent concentrations.

Date	Unfiltered influent wastewater			Filtered influent wastewater						
	COD _{tot} [mg COD/L]	TN [mg N/L]	TP [mg P/L]	COD _{filt,1.6} [mg COD/L]	TN _{filt} [mg N/L]	SNH ₄ [mg N/L]	SN _{other} [mg N/L]	TP _{filt} [mg P/L]	PO ₄ [mg P/L]	SP _{other} [mg P/L]
21-Feb	90%	75%	93%	69%	72%	74%	91%	89%	91%	77%
22-Feb	93%	83%	96%	78%	80%	82%	92%	93%	94%	87%
23-Feb	92%	83%	96%	75%	81%	83%	88%	92%	96%	80%
24-Feb	91%	79%	96%	70%	80%	84%	86%	94%	96%	86%
25-Feb	90%	83%	96%	64%	81%	87%	80%	94%	97%	85%
28-Feb	88%	88%	95%	66%	86%	90%	94%	93%	96%	84%
01-Mar	93%	88%	96%	74%	85%	90%	83%	92%	96%	73%
02-Mar	91%	86%	96%	74%	86%	91%	81%	92%	97%	- ³
03-Mar	91%	87%	96%	75%	87%	91%	91%	94%	96%	86%
04-Mar	92%	87%	96%	75%	85%	90%	85%	94%	97%	87%
07-Mar	91%	87%	95%	78%	85%	89%	84%	93%	95%	87%
08-Mar	94%	86%	96%	81%	87%	91%	84%	92%	94%	84%
09-Mar	92%	90%	96%	75%	88%	94%	88%	92%	95%	83%
10-Mar	93%	88%	97%	82%	87%	92%	89%	95%	97%	86%
11-Mar	92%	65%	92%	59%	Non-detected ¹	40%	- ²	85%	90%	0%
Average	92%	84%	96%	73%	84%	85%	87%	92%	85%	95%
Standard deviation	2%	6%	1%	6%	4%	13%	4%	3%	2%	23%

¹No data was present for TN_{filt} in the effluent water.

²Since no data was present for TN_{filt}, the concentration of SN_{other} could not be obtained.

³The concentration of SP_{other} was zero in influent water and larger than zero in the effluent water.



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