Effects of Varying Sludge Quality on the Permeability of a Membrane Bioreactor

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EFFECTS OF VARYING SLUDGE QUALITY ON THE PERMEABILITY OF A MEMBRANE BIOREACTOR

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Abstract

This master thesis firstly includes a theory part describing, the conventional municipal wastewater treatment plant (WWTP) and especially the conventional activated sludge (CAS) process. As Stockholm municipality want to retrofit the current activated sludge system at Henriksdal into a membrane bioreactor (MBR), an extensive description of the MBR and its advantages and disadvantages are included.

Fouling is considered a really important issue for the operation of an MBR since it reduces an MBR’s productivity over time. Therefore, description of the fouling mechanisms and the potential foulants is included as well as a description of the membrane cleaning procedures. Sludge composition is considered a very important parameter which contributes to membrane fouling and thus this master thesis aims to identify the effects of varying sludge quality on the membranes operation. Precipitation chemicals used for phosphorus chemical precipitation and especially ferrous sulphate which is examined in this master thesis are also affecting the sludge quality and the membranes operation.

The report includes description of Henriksdal reningsverk and line 1 of the pilot MBR at Hammarby Sjöstadsværk where the experimental work was performed. The following chapter describes the experimental work performed in the laboratory including the determination of total suspended solids (TSS), volatile suspended solids (VSS), sludge volume index (SVI) and sludge’s filterability. The filterability was determined by performing the time to filter (TTF) method and the sludge filtration index (SFI) method. Furthermore, the samples were also examined in the optical microscope to determine their bulkiness and their filaments content. The iron content in the sludge was also measured from Eurofins Environment Testing Sweden AB.

In the results section, the different parameters measured are illustrated in charts and they are compared to each other in order to define which factors contribute positively or negatively to the sludge’s filterability and thus affect the sludge quality and the membranes operation. The results indicate that SFI is a more reliable method for measuring filterability compared to TTF. Furthermore, the iron content in the sludge is proportional to the permeability as well as the filaments content observed during microscopy is proportional to the SFI or TTF. Finally, this master thesis includes recommendations for future research which basically include more analyses to identify the sludge biology and more samples taken for longer time periods.
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<th>Description</th>
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<tbody>
<tr>
<td>AC</td>
<td>activated carbon</td>
</tr>
<tr>
<td>Al</td>
<td>aluminium</td>
</tr>
<tr>
<td>AlCl₃</td>
<td>aluminium chloride</td>
</tr>
<tr>
<td>Al₂(SO₄)₃</td>
<td>aluminium sulphate</td>
</tr>
<tr>
<td>AOX</td>
<td>adsorbable organic halogens</td>
</tr>
<tr>
<td>BAP</td>
<td>biomass associated products</td>
</tr>
<tr>
<td>BOD</td>
<td>biological oxygen demand</td>
</tr>
<tr>
<td>Ca</td>
<td>calcium</td>
</tr>
<tr>
<td>CaCO₃</td>
<td>calcium carbonate</td>
</tr>
<tr>
<td>Ca(OH)₂</td>
<td>calcium hydroxide</td>
</tr>
<tr>
<td>CAS</td>
<td>conventional activated sludge</td>
</tr>
<tr>
<td>CaO</td>
<td>calcium oxide</td>
</tr>
<tr>
<td>C₆H₈O₇</td>
<td>citric acid</td>
</tr>
<tr>
<td>CIP</td>
<td>cleaning in place</td>
</tr>
<tr>
<td>COD</td>
<td>chemical oxygen demand</td>
</tr>
<tr>
<td>COP</td>
<td>cleaning out of place</td>
</tr>
<tr>
<td>DOM</td>
<td>dissolved organic material</td>
</tr>
<tr>
<td>DOX</td>
<td>de-oxygenation</td>
</tr>
<tr>
<td>EPS</td>
<td>extracellular polymeric substances</td>
</tr>
<tr>
<td>Fe</td>
<td>iron</td>
</tr>
<tr>
<td>FeCl₂</td>
<td>ferrous chloride</td>
</tr>
<tr>
<td>FeCl₃</td>
<td>ferric chloride</td>
</tr>
<tr>
<td>FS</td>
<td>flat sheet</td>
</tr>
<tr>
<td>FeSO₄</td>
<td>ferrous sulphate</td>
</tr>
<tr>
<td>H₂C₂O₄</td>
<td>oxalic acid</td>
</tr>
<tr>
<td>HF</td>
<td>hollow fiber</td>
</tr>
<tr>
<td>HS</td>
<td>hollow sheet</td>
</tr>
<tr>
<td>L7Z6</td>
<td>line 7 zone 6</td>
</tr>
<tr>
<td>MBBR</td>
<td>moving bed biofilm reactor</td>
</tr>
<tr>
<td>MBR</td>
<td>membrane bioreactor</td>
</tr>
<tr>
<td>MF</td>
<td>microfiltration</td>
</tr>
<tr>
<td>MLSS</td>
<td>mixed liquor suspended solids</td>
</tr>
</tbody>
</table>
MT  multi tubular
NaClO  sodium hypochlorite
NF  nanofiltration
NOM  natural organic compound
PAC  powdered activated carbon
PACl  polyaluminium chloride
PO₄  phosphate
PP  polypropylene
PVDF  polyvinylidene fluoride
R²  coefficient of determination
RO  reverse osmosis
SMP  soluble microbial products
SFI  sludge filtration index
SOC  synthetic organic compound
SRT  solids retention time
SV  sludge volume
SVI  sludge volume index
TMP  transmembrane pressure
TOC  total organic carbon
TSS  total suspended solids
TTF  time to filter
UAP  substrate utilization products
UF  ultrafiltration
VSS  volatile suspended solid
WWTP  wastewater treatment plant
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1. Introduction

Nowadays the efficient treatment of the municipal wastewater is absolutely necessary as the intensive industrialization and urbanization are both leading to larger and more polluted water volumes which need to be treated before discharge (EPA, 2004).

Furthermore, the untreated wastewater can cause several environmental issues when it is directly discharged into the recipient. Eutrophication for instance is caused by nutrients’ discharge into water bodies (i.e. sea, rivers, and lakes). The high nutrients concentration contributes to uncontrolled algal and bacterial growth which eventually leads to oxygen depletion and living organisms’ death (Wright & Boorse, 2011). Additionally, high water quality in the water bodies is necessary for supplying humanity with drinking water and fish safe for consumption or even for recreational activities such as swimming (EPA, 2004).

The most common system for biological treatment of municipal wastewater is the conventional activated sludge (CAS) which is effective since the wastewater pollutants are significantly reduced (Persson, 2011). However, as higher efficiency is required in order to handle the increasing wastewater volumes the membrane bioreactor system (MBR) is increasing in popularity. The smaller tank volume due to increased solids retention time (SRT) and the avoidance of the sedimentation tanks due to membrane filtration are only some of the advantages of the MBR over the CAS which make it possible to achieve higher rates of pollutants reduction (Brepolsa, et al., 2008). However, the MBR usually requires intensive aeration which leads to extensive energy costs (Hai, et al., 2014).

Fouling of the membranes is also another parameter to consider when retrofitting from CAS to MBR since it can cause a lot of impacts on the system and therefore affect the removal efficiency of pollutants (Mutamim, et al., 2012). Therefore, the sludge quality should be determined in order to avoid problems with the membranes. In accordance with fouling, the membranes’ cleaning is also a procedure which should be taken into consideration in an MBR in order to maintain its productivity (Judd, 2008).

This master thesis was a joint thesis provided by Stockholm Vatten and IVL Swedish Environmental Research Institute and it was performed at Stockholm Vatten and Hammarby Sjöstadsverk laboratories. Stockholm Vatten AB is a company owned by City of Stockholm and is responsible for producing and providing drinking water to almost 1 million people. Furthermore, Stockholm Vatten AB is responsible for wastewater treatment and purification (Stockholm Vatten AB, 2011).

IVL Swedish Environmental Research Institute is a non-profit organization founded by the Swedish Government and industry responsible for research in the fields of environment and sustainability (IVL Svenska Miljöinstitutet, 2014). Hammarby Sjöstadsverk is an R & D and it was founded from Stockholm Vatten AB. Currently Hammarby Sjöstadsverk is owned by IVL Swedish Environmental Research Institute and Royal Institute of Technology (KTH) (IVL Svenska Miljöinstitutet, 2014).
Stockholm Municipality and Stockholm Vatten AB currently have two wastewater treatment plants (WWTP) for handling household wastewater, stormwater and a limited amount of industrial wastewater. The plants are located at Bromma and Henriksdal. Both plants have CAS for biological treatment of municipal sewage but Stockholm Municipality has decided to retrofit the activated sludge plant in Henriksdal into a membrane bioreactor plant (MBR) by 2020 (Ohlson & Winnfors AB, 2014).

The wastewater treatment plant (WWTP) at Henriksdal which is located close to Slussen, was first opened in 1941, with a capacity of about 150 000 m$^3$/day. In 1953, the plant was able to treat almost double amount of wastewater per day due to an extension of the facility. During the period of 1992-1997, the plant was again under modification in order to be able to remove more nutrients from the treated water. Currently, the WWTP at Henriksdal is the world’s largest underground WWTP since it covers 30 hectares and consists of 18 km of tunnels. The wastewater at Henriksdal is coming from Stockholm, Huddinge, Haninge, Nacka and Tyresö (Stockholm Vatten AB, 2011).

The WWPT plant at Bromma consists of two smaller facilities at Åkeshov and Nockeby. The facility at Åkeshov was the first WWTP facility in Stockholm and it first opened in 1934. The facility at Nockeby was opened during the 60’s and the last expansion for nitrogen removal finished on 2000. Currently, the two facilities treat about 126 000 m$^3$/day of wastewater and they are connected with a 600 m tunnel. This plant serves the north and west parts of Stockholm, some parts of Järfälla and Ekerö and Sunnyberg (Stockholm Vatten AB, 2011).

However, the plant at Bromma is planned to shut down and therefore the plant at Henriksdal must be able to have a larger capacity to serve the entire Stockholm. Therefore, Stockholm Vatten in cooperation with IVL Swedish Environmental Research Institute has retrofit the pilot MBR plant at Hammarby Sjöstadsverk next to Henriksdal to suit the need of the MBR project for further research prior to the implementation of the MBR at a municipality scale. Stockholm Vatten has employed IVL Swedish Environmental Research Institute to run all the research going on regarding the MBR. Practically, IVL Swedish Environmental Research Institute is responsible for the operation and all practical work of the pilot. The pilot plant which is built at Hammarby Sjöstadsverk is more or less a copy of the full-scale plant that is going to retrofit the current wastewater treatment facility at Henriksdal (Stockholm Vatten AB, 2013).
1.1. Aim

The aim of this master thesis is to examine the effects of varying sludge quality on the membranes permeability of the pilot MBR plant at Hammarby Sjöstadsverk, to examine which factors affect the sludge filterability and to compare sludge samples from the pilot MBR and Henriksdal reningsverk.

1.2. Objectives

- To determine the sludge quality in different sludge samples both from the pilot MBR at Hammarby Sjöstadsverk and Henriksdal reningsverk
- To examine different ways of determining the sludge filterability
- To examine the effects of different parameters on sludge’s quality and sludge filterability
- To determine the effects of different sludge’s qualities on the membranes’ permeability
- To compare the different sludge qualities from the pilot MBR at Hammarby Sjöstadsverk and Henriksdal reningsverk
2. Methods

This master thesis includes extended literature review in various books, scientific articles, publications and data collection from Stockholm Vatten and IVL Swedish Environmental Research Institute. The results of the thesis are based on experimental and laboratory work conducted in the facilities of Stockholm Vatten and IVL Swedish Environmental Research Institute at Henriksdal and Hammarby Sjöstadswerk respectively. Furthermore, this thesis is based on analyses conducted also at an external partner, Eurofins Environment Testing Sweden AB. Finally, the results analysis is based on calculations of the laboratory measured factors with several factors measured online.
3. Theoretical Background

The next sections include all the relevant information extracted during literature review, regarding conventional wastewater treatment plants, MBR technology, membrane technology, sludge characteristics, fouling, membrane cleaning and phosphorus chemical precipitation.

3.1. Conventional Municipal Wastewater Treatment Plant

Municipal wastewater mainly contains organic compounds, nitrogen and phosphorus compounds, suspended solids, and several species of pathogens. It can also contain heavy metals, several inorganic compounds, persistent organic compounds and other substances in small quantities. In order to handle the wastewater’s impurities, mechanical, biological and chemical treatment methods are used (Hai, et al., 2014).

During mechanical treatment coarse screens in combination with settling basins are mainly used in order to separate the large particulate matter from the wastewater. The second stage usually involves the removal of sand in order to avoid problem in the pumps afterwards. This is accomplished though sedimentation where the large particles settle. This stage is combined with aeration because the wastewater lacks of oxygen due to biodegradation and helps the inorganic material such as sand and gravel to settle. The organic material contained in the wastewater remains suspended and therefore pre-aeration prevents biological sludge from settling into the sand (Watercare, 2010). After that, a conventional plant usually requires a primary settling to force the small particles to settle before the biological treatment (Persson, 2011).

The biological treatment can be accomplished by using activated sludge, aerated lagoons, trickling filters or moving bed biofilm reactors (MBBR). This stage, as it is explained in the next sections, is used for oxidizing the oxygen consuming organic substances which are present in the municipal wastewater by using them as substrate for microorganisms. During the biological treatment, the nitrogen and phosphorus contents are also reduced to a limited extent. For nitrogen removal, the stages of nitrification and denitrification are required where different aerobic and anoxic stages are used for more efficient removal. The nitrification reaction involves the oxidation of ammonium (NH$_4^+$) into nitrate (NO$_3^-$) and the denitrification reaction involves the transformation of NO$_3^-$ into nitrogen gas (N$_2$). For more efficient removal of phosphorus, chemical treatment is then used by adding one precipitation chemical such as iron and aluminum salts (Persson, 2011).

The mechanical, biological and chemical treatments are not always used with this sequence and the different patterns of combining them are direct precipitation, pre-precipitation, pre-dosage, simultaneous precipitation and post-precipitation. The post-precipitation includes the initial sequence of mechanical, biological and chemical treatment and is the most common used all around Sweden. This method has 90-95 % BOD removal and 90 % phosphorus reduction. During the direct precipitation, there is no biological stage and the chemical treatment is accomplished just after the primary treatment. In this case the BOD reduction is approximately 60-70 % and the phosphorus reduction is around 90 %. The pre-precipitation include firstly mechanical treatment and chemicals addition just before the primary settling and finally
biological treatment and usually has 90 % BOD reduction and 90 % phosphorus reduction. The simultaneous precipitation includes the mechanical treatment as a first stage and it has a simultaneous biological and chemical stage. The last method has 80-90 % BOD removal and 75-95 % phosphorus removal. During pre-dosage which is similar to simultaneous precipitation since the phosphorus precipitation is partially accomplished during the biological treatment, usually a divalent ion chemical is added such as ferrous sulphate (FeSO₄) before biological treatment in order to be oxidized to trivalent ion in the aeration basin. However, the dosage point of the pre-dosage is similar to pre-precipitation (Persson, 2011).

In order to be more specific, good addition points of the precipitation chemicals are usually the tertiary stages where the most of the biological load is already removed. For instance, the chemicals can be added before sand filters but the accepted dosage for the filters operation shall be rather small to succeed very low phosphorus content. However, the chemicals can be also added in two stages during the secondary clarification after the aeration basin and before the sand filters. In any case, the most common addition point is before the secondary clarifier. Furthermore, the chemicals can be added in the primary clarifier in an effective way because the biological load is already reduces before entering the aeration basin. In that case, the chemicals addition can be accomplished at a two stage process, during the primary clarifier and during the secondary clarifier. Generally, the multiple additions have been proved to be more efficient in phosphorus removal but this also depends on the particular plant (Patoczka, et al., 2013).

3.2. Conventional Activated Sludge (CAS)

The most common technique used for the biological treatment in a municipal wastewater treatment plant is the conventional activated sludge. This treatment method is accomplished in an aeration basin where various types of microorganisms degrade the suspended and dissolved organic matter (see Figure 1). Apart from the carbon source which is used as a substrate for the microorganisms’ growth, several nutrients are also essential. As nutrients are already present at a municipal wastewater flow, no addition is usually necessary and therefore a small reduction in the nutrients content is also accomplished during this biological stage. The CAS technique is aerobic and for that reason powerful aeration is required. During the aerobic degradation of the organic material the microorganisms produce carbon dioxide (CO₂), water (H₂O) and cell mass (Persson, 2011).

![Figure 1: The CAS Process, modified from (Persson, 2011)](image-url)
After the sludge’s formation, the sludge is separated from the water in a settling basin. However, part of the sludge is recycled back to the aeration basin in order to enhance the microorganisms’ growth and therefore the degradation of the organic substances. The sludge’s recycling enhances as well the diversification of microorganisms’ populations which is really important for the efficient operation of the plant. This variety of different microorganisms is essential for an effectively operating biological treatment plant because each species has different abilities. For example, the free swimming bacteria have the affinity to easily degrade the dissolved organic matter but they are too small to settle. However, the flock forming bacteria are settleable while the micro-animals are feed with the particulate organic matter, the free swimming and flock forming bacteria. The nitrifying bacteria enhance the nitrification and therefore all species are necessary for balancing the entire biological system (Persson, 2011).

3.3. Membrane Processes

Membrane technology is based on a separation technique which separates contaminants from water. This separation technique is based on a semi-permeable membrane which is designed to allow some particles to pass through and retains others. The clean liquid that pass from the membrane is usually called permeate and the retained stream is called concentrate or retentate. So, the contaminants dissolved in the water can be separated out (Lee, et al., 2009).

Osmosis is a principal according to it if two liquids with different concentration are in contact across a semi-permeable membrane, the system will strive for the equalization of the difference by diffusion of the water through the membrane, from the low concentration solution to the higher. In reverse osmosis, the high concentration solution passes across the membrane with higher pressure than the osmotic one and forces the process to reverse. Then the water diffuses from the high concentration side through the membrane to the low concentration side (Lee, et al., 2009).

The materials used for manufacturing the membranes are usually plastic and ceramic materials, although there are some metallic membranes as well. The most common materials used are cellulosates, polyamides, polysulphone, poliacrylonitrile, polyvinylidene difluoride, polyethylysulphone, polyethylene and polypropylene (Radjenovic, et al., 2007).

The flow can be either perpendicular or parallel to the membranes and is usually called dead-end and cross-flow filtration respectively. The cross-flow as it is described below leads to less fouling while the dead-end is mainly used in batch procedures (Mutamim, et al., 2012). There are five different types of membranes configurations currently used widely, categorized according to their geometry the hollow fiber (HF), the spiral-wound, the plate and frame such as the flat sheet (FS), the pleated filter cartridge and the multi tubular (MT) (Radjenovic, et al., 2007), (Mutamim, et al., 2012). From them the HF, the FS and the MT are used for cross-flow filtration (Mutamim, et al., 2012).

There are four different membrane categories considering their separation ability: the microfiltration (MF), the ultra-filtration (UF), the nanofiltration (NF) and the
reverse osmosis (RO). Roughly, the range for particles separation for the different membranes corresponds to: 100-1000 nm, 5-100 nm, 1-5 nm, and 0.1-1 nm respectively (Radjenovic, et al., 2007).

During the first two processes, the separation takes place by screening molecules and particles through the pores of the membrane, where the smaller particles pass and the bigger than the pores are retained. On the other hand, in the RO, the substances that pass must be dissolved in the membrane and the substances which cannot be dissolved are retained. The NF has characteristics from both the ultra-filtration and the reverse osmosis. The membranes and especially the NF and RO are mainly polishing stage processes since they are more sensitive. Therefore the influent water should be pretreated in order to have high quality (Lee, et al., 2009).

There are many factors which contribute to the removal of the particles from the municipal waste water treatment through membrane processes and affect their efficiency. Firstly, the physical and chemical properties of the compound which need to be removed are very important in order to select the proper membrane. So the particles’ properties such as molecular weight, size and diameter, solubility in the water, diffusivity, polarity, hydrophobicity and electric charge should be analyzed. After the selection of the proper membrane, the operating conditions are also important to improve its removing efficiency (Lee, et al., 2009).

### 3.3.1. Removal Mechanisms

The contributing mechanisms in case of membrane separation processes are the size exclusion, charge repulsion, and adsorption totally linked with the functional mechanisms of the membranes. According to several studies, the ultra-filtration (UF) membranes are capable of capturing microconstituents which are usually really smaller than the pores of the membrane, even one hundred times smaller. Actually, the main mechanism in case of UF is sorption or adsorption (Lee, et al., 2009).

Several other factors can also interfere with adsorption capacity of the particles from the membrane, such as the size, charge and hydrophobicity of the chemicals, the charge, hydrophobicity and roughness of the membrane and the temperature, ionic strength and the substances variety of the wastewater (Lee, et al., 2009). The effects of cake formation onto the membrane surface is also contributing to the removal of contaminants since the cake can act as an additional filter and capture contaminants that would pass through the filter itself (Radjenovic, et al., 2007).

### 3.4. Membrane Bioreactor (MBR)

The membrane bioreactor process constitutes a combination of a conventional activated sludge with a membrane separation process. Therefore, the basic process is similar to an activated sludge plant. The two processes mainly differ in the part of sludge separation. In a conventional activated sludge, the separation is accomplished within a settling basin, as it is already mentioned, which is located just after the aeration basin (Hai, et al., 2014).

On the other hand, the removal of sludge in the membrane bioreactor is accomplished through a membrane located either inside (internal submerged MBR) or
outside (external submerged MBR) of the aeration basin. The internal submerged MBR, as it is illustrated in figure 2, is a more compact system where permeate is transported by using a vacuum pump while the remaining sludge is removed from the bottom of the aeration basin. This system is more energy and cost efficient due to the simple equipment (Hai, et al., 2014).

On the other hand, the external submerged MBR system, as it is illustrated in figure 3, requires a tank for the membrane and it is energy consuming and costly since it requires more pumps to transport the flows in and out of the membrane tank. In the external submerged MBR the cleaning can be accomplished in place while in the internal submerged MBR the membranes must be transferred to an external cleaning tank (Hai, et al., 2014). The external submerged MBRs are also usually more effective to prevent fouling as increased cross flow enhances the removal of cake from the membrane’s surface (Radjenovic, et al., 2007).

Parameters like trans-membrane pressure (TMP), flux, permeability and the membrane capacity are really important for the plant’s operation and they should always be under consideration. The flux of a membrane represents the amount of wastewater that passes through the membrane per unit of time and membrane area and can be calculated from equation 1:
\[ J = \frac{Q}{A} \]  

Where,

J is the flux in \( \text{m}^3/(\text{s} \cdot \text{m}^2) \)
Q is the flow in \( \text{m}^3/\text{s} \) and
A is the membrane area in \( \text{m}^2 \)

(Hai, et al., 2014)

The TMP represent the pressure difference between the average concentrate pressure and the permeate pressure and it can be calculated by using equation 2:

\[ TMP = \frac{(P_f + P_c)}{2} - P_p \]  

Where,

TMP is the trans-membrane pressure in bar
Pf is the feed pressure in bar
Pc is the effluent’s (concentrate) pressure in bar
Pp is the permeate pressure in bar

(Wilf, 2008)

The permeability of the plant is basically the amount of water passes through the membrane per unit of time, per unit of area and per unit of TMP. It basically expresses the ability of the liquid to pass through the semi permeable membrane and it can be calculated from equation 3:

\[ Perm = \frac{J}{TMP} \]  

Where,

Perm is the membrane permeability in \( \text{m}^3/(\text{s} \cdot \text{m}^2 \cdot \text{bar}) \)
J is the flux in \( \text{m}^3/(\text{s} \cdot \text{m}^2) \) and
TMP is the trans-membrane pressure in bar

(Hai, et al., 2014)

Meanwhile, the MBR plants usually operate at one specific and constant flux or at one specific and constant TMP. This is performed in order to realize if the membranes operate well or not. Finally, the feed water characteristics such as particles concentration, temperature, pH, ionic strength, hardness, and total organic matter should be taken into consideration. The clogging can be minimized by providing influent as clean from particles as possible. The MF, the UF and the NF may also have a backwash cycle which removes the larger particles building on the surface of the
membrane (Lee, et al., 2009). There are also other really important cleaning processes such as relaxation which are further analyzed in the next sections.

In case of MBR, the UF and MF membranes are used because the risk of fouling is smaller. The UF and MF are less sensitive to clogging, produce a good quality effluent and contribute to less energy consumption in the plant compared to RO and NF. The final choice between UF and MF depends on the plant’s characteristics and requirements. Additionally, by using the UF and MF, all the parasites, the most of the bacteria and several viruses are removed from the water, not as efficiently as in RO or NF systems but to very high extent (Hai, et al., 2014).

3.4.1. Removal Mechanisms

The removal mechanisms linked with the biological treatment are the sorption and the biodegradation or biotransformation. In both conventional activated sludge and membrane bioreactor the formed sludge has very good sorption capacity due to the high specific area of the suspended microorganisms. The biodegradation is a very important mechanism as well concerning the microconstituents and it is effectively accomplished in the membrane bioreactor. Specifically, due to the high retention time, the total organic carbon is sufficiently reduced and this enhances the biodegradation of microconstituents in the wastewater. The reduction of the total organic carbon enables the microorganisms to use the microconstituents as a substrate which are usually slowly degraded in order to maintain their rate of growth in the aeration basin. In case of sorption, the contaminants are being transferred to the sludge without being eliminated. On the other hand, during biodegradation the pollutants are being transformed into other substances which can be either harmless or toxic (Lee, et al., 2009).

3.4.2. Advantages and Disadvantages of the MBR

The membrane separation is usually more effective than sedimentation because the suspended solids are very low after the treatment and the effluent obtained is cleaner (Hai, et al., 2014). However, both techniques are sensitive to shock loads and thus the inlet sludge flow has to be stabilized (Persson, 2011).

One very important advantage of the membrane bioreactor is that the plant’s size is significantly smaller compared with the conventional activated sludge because the efficient removal of the sludge generated does not require large space as in sedimentation tanks (Lee, et al., 2009).

Furthermore, the solids retention time (SRT) in the membrane bioreactor can be higher compared with the conventional activated sludge which leads to better removal efficiency of the pollutants. Some colloidal particles which are hardly removed during the activated sludge are being attracted and absorbed by the suspended solids in an MBR and thus they can be biodegraded to some extent (Hai, et al., 2014). The solids retention time of an activated sludge process depends totally on the settling properties of the microorganisms and therefore it cannot exceed a particular value (approx. 15 days). However, the SRT in the MBR is not limited since the settleability of the particles is not an issue affecting the process and thus it can have higher values and improve the overall efficiency of the plant (Lee, et al., 2009).
The effluent generated from the membrane bioreactor is often better than the one generated from a conventional activated sludge plant including different parameters such as the biological oxygen demand (BOD), the chemical oxygen demand (COD), the total organic carbon (TOC) and the phosphorus and nitrogen content. Furthermore, due to the effective separation with the membrane process, post-disinfection processes are not usually necessary (Lee, et al., 2009).

Additionally, due to the longer SRT in the membrane bioreactor process, the sludge produced is lesser. As a result, the amount of sludge which needs further treatment is generally reduced and this leads to lower future costs for sludge handling (Lee, et al., 2009).

Another benefit of the membrane bioreactor which is related to the long solid retention time is the growth of nitrifying bacteria. The nitrifying bacteria benefit from long retention time due to the fact that they are slow growing. Therefore, the nitrification reaction can efficiently take place in the same stage as the biological degradation of organic material. (Lee, et al., 2009).

In addition, efficient mixing is always required and essential. However, the membranes are expensive to buy and the long SRT is linked with higher energy costs due to larger necessity for powerful aeration and with expenses for membranes’ cleaning since the system works with higher biosolids concentration (Hai, et al., 2014).

### 3.5. Membrane Fouling

The most important issue of the membranes is fouling. Fouling is happening due to sludge – membrane interactions over time and as a result the flux at a given TMP and the permeability are significantly reduced. In other words, fouling can be defined as a combination of physicochemical interactions between the potential foulants and the membrane (Guo, et al., 2012). In order to maintain the membrane’s flux, the trans-membrane pressure is increased. However, possible solutions are to run the system at low fluxes which is not very common as the capacity is lower or having frequent relaxation periods (Radjenovic, et al., 2007).

The composition of the inflow play a really important role as the chemistry of the constituents determines the fouling potential. The pH, the temperature and many other properties of the wastewater can of course affect the fouling potential of the entire system. The membranes properties can also contribute to the fouling phenomenon either positively or negatively. For example, the most vulnerable membranes to fouling phenomenon are the hydrophobic ones as the fouling agents interact with the membranes more from a hydrophobic perspective. Therefore, the membranes available in the market are being chemically treated in order to obtain a more hydrophilic character (Radjenovic, et al., 2007). The cross-flow membranes compared to the dead-end that are analyzed in previous sections are reducing the fouling potential at representative extent (Guo, et al., 2012).

The fouling can be categorized into physically reversible, chemically reversible and irreversible. In the physically reversible fouling the particles can be easily
removed through cleaning procedures as they are not strongly attached on the membrane. The chemically reversible fouling is more serious but still the membrane can be recovered though chemical treatment. However, the irreversible fouling, which usually happens after using the membrane for some time, harms some parts of membrane because no cleaning can be applied (Hai, et al., 2014).

3.5.1. Foulants and Fouling Mechanisms

The mechanisms contributing to fouling phenomenon are organic adsorption, cake formation and pore blocking, concentration polarization, inorganic precipitation and biological fouling (Guo, et al., 2012). However, the aging of the membrane can be also considered as part of membranes’ fouling (Radjenovic, et al., 2007). The types of fouling according to foulants physical properties are categorized into particles’ fouling, organic or inorganic fouling and biofouling (Guo, et al., 2012).

The particulate foulants can be categorized into settleable solids which are bigger than 100 μm, supra-colloidal solids which are between 1 μm to 100 μm, colloidal solids which are 0.001 μm to 1 μm and dissolved solids which are smaller than 0.001 μm. The particulates can be either organic or inorganic and they can either block the membranes pores or form a cake layer on the membrane surface (Guo, et al., 2012). The cake formation as it is shown in figure 4 is enhanced from rather big particles which are present in the wastewater flow and therefore they form a shallow layer on the membrane’s outer surface, the cake. The pore blocking is caused by the penetration of small particles (i.e. colloidal) into the membrane’s pores (Ferreira, 2011). The pore blocking can be categorized into standard pore blocking, complete pore blocking and intermediate pore blocking according to the extent of pores blockage as it is shown in figure 4 (Guo, et al., 2012).

The organic foulants are basically found in dissolved and colloidal form in the wastewater flow and they are attracted by the organic adsorption mechanism onto the membrane’s surface. Such foulants are the humic and fulvic acids or several hydrophilic and hydrophobic substances like proteins. The dissolved organic material (DOM) can be categorized into natural organic compounds (NOM), synthetic organic compounds (SOC) and soluble microbial products (SMP). The NOMs are present in water bodies while the SOCs come from different everyday consumers products. The SMP are products of microbial decomposition and metabolism and they are further described in the next section since they play an important role in fouling phenomenon (Guo, et al., 2012).

On the other hand the inorganic foulants are several dissolved constituents such as iron hydroxides or manganese oxides. These types of foulants may be present during chemical flocculation, coagulation or precipitation. Some inorganic compounds are
already present in the inflow water and can easily cause fouling. Such compounds are calcium carbonate or silicon dioxide (Guo, et al., 2012).

The biofoulants are different forms of microorganisms and their products which form a biofilm on the membrane’s surface. The mechanism of biofouling involves the deposition of different macromolecules or cells on the membranes’ surface in order to form a biofilm. According to several analyses the major component of the biofilm is the fraction of extracellular polymeric substances (EPS) and of course cell mass (Guo, et al., 2012).

The EPS constitute a broad category of polymers such as polysaccharides, proteins, nucleic acids, carbohydrates and lipids (Ferreira, 2011) which are coming from the microorganisms’ metabolic processes (Hai, et al., 2014). The EPS can be distinguished into bound EPS and soluble EPS or SMP (Ferreira, 2011). The SMP are categorized into biomass-associated products (BAP) which are generated from EPS hydrolysis and substrate utilization products (UAP) which are generated from the microorganisms’ substrate (Mishima & Nakajima, 2009).

### 3.5.2. Biomass Quality

The quality of the biomass contained in the wastewater is of great concern since it contributes to the membrane fouling and generally affects the plant’s efficiency. Many studies indicate that the most important parameters which are linked with biomass quality and actively contribute to fouling phenomenon are the mixed liquor suspended solids (MLSS) or total suspended solids (TSS), the extracellular polymeric substances (EPS), the soluble microbial products (SMP) and the particles’ size (Hai, et al., 2014).

Both EPS and SMP have effects on the oxygenation of the biological process and therefore they can lead to lower biodegradation rates. The EPS can be easily absorbed onto both hydrophobic and hydrophilic surfaces since they have the ability to have hydrophobic and hydrophilic parts. This happens because the EPS contain several charged and uncharged molecules. During the biological treatment, the bound EPS enhance the creation of flocks which adsorb smaller dissolved particles and some colloidal matter. Unlike the EPS, the SMP are not integrated in the biological flocks but they are dispersed in the liquid phase (Nguyen, et al., 2012). The formation of EPS is very important for the activated sludge but it can cause problems during the MBR operation. In the MBR, the EPS are usually responsible for the formation of cake layer onto the membrane surface. According to different studies, the EPS are affected by the SRT and in some cases the higher SRT resulted in lower EPS production while in other cases the results were opposite (Radjenovic, et al., 2007).

Furthermore, the EPS are also affected from temperature variations where their amount is lower with temperature reduction (Moreau, 2010). Other studies also indicated that in low temperatures the reversible fouling was more intense while the irreversible fouling was more profound in higher temperatures. This happens because the temperature differences usually affect the metabolic rates of microorganisms. Those findings were taken from an MBR plant with low SRT. However, in other plants with long SRT the temperature seem to be a very unimportant factor (Drews, 2010).
The particles’ size is also considered to be very important factor when the biomass quality is examined. By increasing the aeration inside the basin and through increased biomass recycling, the particles size can be reduced significantly and thus prevent reversible fouling caused by large particles. However, the results of different studies might be contradicting when investigations of the main foulants are taking place (Ferreira, 2011). A category called submicron particles is strongly considered to be major contributors of the irreversible fouling as they are slightly smaller than the membrane’s pores and they can easily block them. Their origin in the wastewater is basically the dismantling of large flocks or the metabolic processes of the microorganisms (Hai, et al., 2014). This phenomenon was identified from different researchers to be more intense during low temperatures (Moreau, 2010).

The MLSS concentration also constitutes an important parameter characterizing the biomass quality. They can be easily controlled though higher or lower SRT with respect to the efficient aeration of the process. However, there is no clear correlation between MLSS and membrane’s fouling (Ferreira, 2011). At very high MLSS concentrations (>12 g/l) membrane manufacturers estimate that fouling is increased.

3.5.3. Fouling Control and Membrane Cleaning

When the MBR operates at constant flux and the TMP is significantly increased, the membranes need to be cared of. The same applies if the MBR operates at constant TMP and the flux is significantly decreased. In order to avoid or reduce fouling, good sludge quality is required (Hai, et al., 2014). The most important measure for controlling fouling is the enhancement of the aeration within the aeration basin which of course is very expensive (Radjenovic, et al., 2007).

By adding activated carbon (AC) in the MBR, there is also a possibility for decreasing the fouling potential of the membranes. According to Mutamim et al powdered activated carbon (PAC) is always more effective as there is higher surface for better absorption of organic and inorganic microconstituents. Therefore, the biodegradation of pollutants is definitely enhanced simultaneously with reducing the fouling potential and the potential of shock loads due to varying inflows (Mutamim, et al., 2012).

Despite, the fouling control, the membranes will definitely need cleaning sooner or later. There are different types of cleaning which can be sorted into mechanical or physical and chemical methods as broader categories. However, combinations of physical and chemical cleaning are also possible (Mutamim, et al., 2012).

The mechanical cleaning processes involve continuous or periodic air scouring, continuous cross-flow, periodic backwashing and periodic relaxation. The air scouring includes the application of air bubbles onto the membrane in order to mitigate the formation of cake. Both the cross-flow and backwashing techniques involve water flow on the membrane’s surface from a different direction. In case of cross-flow the water comes perpendicular to the wastewater’s flow in order to prevent fouling. On the other hand, during backwashing which is a common technique, the flow of permeate is reversed for a limited amount of time to get rid of the sludge cake. The relaxation method requires a pause of the flow while the air bubbles are still present around the membrane and therefore the formed cake gradually comes off or it is
removed by air scouring (Hai, et al., 2014). For the FS membranes, membranes’ brushing in situ is also a possible physical cleaning procedure. Basically, all the mechanical ways of cleaning the membranes remove the cake layer which is formed onto the membrane’s surface (Mutamim, et al., 2012).

For more extensive removal of foulants chemical cleaning is usually necessary. The chemical cleaning of the membranes can be either accomplished in situ or ex situ. Both the cleaning in place (CIP) and the cleaning out of place (COP) involve the addition of various chemicals in order to remove the fouling agents from the membranes. The CIP is mainly used to recover the membranes from reversible and cake fouling when the flux is significantly reduced or the TMP is significantly increased. The COP is required after a long use of the membranes where the fouling agents need more extensive, out of place treatment in order to be recovered effectively (Hai, et al., 2014).

The most common chemicals used for chemical cleaning are sodium hypochlorite (NaClO) as a first cleaning step in order to remove organic material and usually citric (C\textsubscript{6}H\textsubscript{8}O\textsubscript{7}) or oxalic acid (C\textsubscript{2}H\textsubscript{2}O\textsubscript{4}) as a second step for removing the inorganic impurities. A low strength NaClO can be used couple of times within a month in order to keep the productivity of the membranes within the accepted limits. However, it is also needed to use a high strength NaClO one or two times per year in order to recover the lost permeability and maintain a high flux. Of course the frequency of chemical treatment always depends on the specific characteristics of the plant such as trans-membrane pressure, fouling and permeability. The chemical cleaning of the membranes is also dependent on the specific type of the membrane used in the particular plant. For example, the FS membranes require stronger aeration which subsequently reduces the necessity for chemical cleaning very often (Judd, 2008).

Cleaning is of course absolutely necessary for the plant’s performance but when it is applied in a regular basis the membrane becomes damaged and after some time it will need replacement (Mutamim, et al., 2012). Furthermore, extensive cleaning causes environmental impacts due to release of adsorbable organic halogens (AOX) for instance, extra costs for chemicals, extra work and chemicals handling (Drews, 2010).

3.6. Phosphorus Chemical Precipitation

The most common sources of phosphorus in the wastewater are detergents, excrement, fertilizers and manure. Phosphorus can be found either in particulate form or dissolved (Theobald, et al., 2013).

Phosphorus in the water is usually present in the form of orthophosphate, polyphosphate and organic phosphate. The group of orthophosphates consists of PO\textsubscript{4}\textsuperscript{3-}, HPO\textsubscript{4}\textsuperscript{2-}, H\textsubscript{2}PO\textsubscript{4}\textsuperscript{-} and H\textsubscript{3}PO\textsubscript{4} which are already in a biodegradable form. However, the polyphosphates (i.e. ATP, ADP etc.) which are rather complex constituents must be hydrolyzed in order to be transformed into orthophosphates and become easily biodegradable. The organic phosphate is bound to metabolic products and is converted to orthophosphates through decomposition (Tchobanoglous, et al., 2003).
The phosphorus which is present in the wastewater flow can be either removed biologically or chemically. In this section the chemical treatment is analyzed while the biological phosphorus removal is not taken into consideration (Tchobanoglous, et al., 2003). Phosphorus is chemically removed from the wastewater through the process of chemical precipitation. Chemical precipitation is a technique of transforming a compound into precipitant in order to remove it from the water body. After the chemical precipitation, the most common method to remove the precipitants is sedimentation (Patoczka, et al., 2013).

The chemicals used for phosphorus precipitation are aluminium (Al$^{3+}$), iron (Fe$^{3+}$, Fe$^{2+}$), calcium (Ca$^{2+}$) (Persson, 2011) and magnesium salts (Mg$^{2+}$). The chemicals added for phosphorus precipitation are usually required in larger quantities than the stoichiometric analogy due to the fact the secondary reactions (see Reactions 2,5 and 8) are taking also place in order to form Al(OH)$_3$, Fe(OH)$_3$ and Ca(CO)$_3$ and even more complex compounds. After the addition of chemicals the concentration of MLSS in the water is unavoidably increased and this can affect the plant’s operation (Patoczka, et al., 2013).

3.6.1. Phosphorus Chemical Precipitation with Aluminium and Iron Salts

The most common salts of aluminium added for chemical precipitation are aluminium sulphate (Al$_2$(SO$_4$)$_3$), polyaluminium chloride (PACl) and aluminium chloride (AlCl$_3$). In case of iron salts, the most common are the ferric chloride (FeCl$_3$), the ferrous chloride (FeCl$_2$) and the ferrous sulphate (FeSO$_4$) (The Water Planet Company, 2014).

The reactions taking place during chemical precipitation of orthophosphates with trivalent ions of Fe$^{3+}$ and Al$^{3+}$ are reactions 1 and 2.

\[
X^{3+} + PO_4^{3-} \rightarrow XPO_4 \text{(s)} \quad (1)
\]

\[
X^{3+} + 3HCO_3^- \rightarrow X(OH)_3 \text{(s)} + 3CO_2 \quad (2)
\]

Where X is either Fe$^{3+}$ or Al$^{3+}$

(Tchobanoglous, et al., 2003; The Water Planet Company, 2014)

During chemical precipitation with divalent ions, the reactions taking place are reactions 2,3 and 4.

\[
3X^{2+} + 2PO_4^{3-} \rightarrow X_3(PO_4)_2 \text{(s)} \quad (3)
\]

\[
X^{2+} \rightarrow X^{3+} \quad (4)
\]

\[
X^{3+} + 3HCO_3^- \rightarrow X(OH)_3 \text{(s)} + 3CO_2 \quad (2)
\]

Where X is Fe$^{2+}$
(Patoczka, et al., 2013; The Water Planet Company, 2014)

The first reaction is actually the precipitation reaction while the second is a parallel reaction involving some loss of the metal salts added. However, the metal hydroxides attract several other substances giving rise to flocculation of other precipitates which enhances the removal of phosphorus and suspended matter (Persson, 2011).

3.6.2. Phosphorus Chemical Precipitation with Calcium Salts

For the precipitation with Ca$^{2+}$, the most common chemicals used are the calcium hydroxide (Ca(OH)$_2$) and the calcium oxide (CaO) in order to form the precipitant hydroxylapatite (Persson, 2011). Mg$^{2+}$ reacts the same way as Ca$^{2+}$ ions but it is not commonly used. The hydroxylapatite formed during the reactions does not have a fixed composition (Patoczka, et al., 2013) and for that reason two possible reactions are being presented.

The reactions taking during chemical precipitation of orthophosphates with Ca$^{2+}$ ions are the reactions 5,6 and 7.

10Ca$^{2+}$ + 2OH$^{-}$ + 6PO$_4^{3-}$ → Ca$_{10}$(PO$_4$)$_6$(OH)$_$_2(s)  \tag{5}  

(Tchobanoglous, et al., 2003) Or,

5Ca$^{2+}$ + 4OH$^{-}$ + 3HPO$_4^{2-}$ → Ca$_5$(PO$_4$)$_3$(OH) + 3H$_2$O (s)  \tag{6}  

(Patoczka, et al., 2013), (Persson, 2011)

Ca$^{2+}$ + CO$_3^{2-}$ → CaCO$_3$ (s)  \tag{7}  

(Persson, 2011)

The Ca$^{2+}$ precipitation requires a lot of chemicals to be added during the process due the great loss of Ca$^{2+}$ during the reaction 7 for binding the free carbonates included in the water body (Persson, 2011).

3.6.3. Operational Parameters of Phosphorus Chemical Precipitation

In case of aluminium sulfate used as precipitation chemical, which is the most common situation, the pH of the wastewater might need to be adjusted, by adding sodium hydroxide for instance, as a lot of alkalinity is consumed. The same applies for ferrous or ferric salts. This is happening because the aluminium and iron salts are both acidic. Sodium aluminate as it is generated from sodium hydroxide reaction with aluminium hydroxide does not cause pH reduction during phosphorus precipitation. Furthermore, in order to begin the precipitation of phosphorus with Ca$^{2+}$ ions the pH should be above 10 which not in compliance with the conditions required during the biological treatment and therefore the usage of lime for phosphorus precipitation is rather limited compare with Fe$^{3+}$, Fe$^{2+}$ or Al$^{3+}$ ions usage (Patoczka, et al., 2013).
Apart from the pH adjustment in different cases, the efficient mixing is another factor contributing to the enhancement of phosphorus removal in order to more efficiently disperse the chemicals. Furthermore, the more the contact time that the wastewater meets the precipitants, the more the chances of soluble orthophosphates to be absorbed from the solids (Patoczka, et al., 2013).

### 3.6.4. Effects of the Precipitation Chemicals on the Sludge

The addition of chemicals results also in greater sludge loads. This sludge is basically chemical sludge containing the phosphorus precipitants, the metal hydroxides as excess of metal salts are added and some colloids and particles which are captured by chemicals as well (Patoczka, et al., 2013). This process of capturing colloidal particles in the wastewater is called coagulation and sometimes it can be applied only for particles separation instead of phosphorus removal. Therefore, the separation of solids is enhanced during the addition of chemicals. In case of precipitation with Ca$^{2+}$ ions, the amount of sludge produced is greater than during the precipitation with iron and aluminum salts due to carbonate formation (Persson, 2011).

When aluminium and mainly iron salts are used, the settling properties of the particles are also improved because the sludge specific density is significantly greater and bulking due to filamentous bacteria is less. The foaming phenomenon is also under better control when chemicals are added and improved biomass quality is expected. The sludge containing iron is also suitable for soil applications (Patoczka, et al., 2013).

The precipitation chemicals can be also efficiently used for controlling the fouling of the membranes by reducing the EPS and SMP (Nguyen, et al., 2012). According to Mishima & Nakajima (2009) the SMP were totally removed while the EPS were significantly reduced when any precipitation chemical was used. However, in order the chemical to be more effective high SRT is usually required which is already applied in a MBR system. Therefore, the reduction of SMP and EPS is more effective in MBRs than in activated sludge disregarding the fouling control which is simultaneously accomplished (Mishima & Nakajima, 2009).

Furthermore, the study of Mishima & Nakajima (2009) showed that the precipitation chemicals can both reduce the cake on the membrane’s surface but also the pore blocking in FS membranes (Mishima & Nakajima, 2009). The pore blocking is controlled because during the addition of chemicals the particles size is increased and they just form a layer on the membrane which is easily handled according to a study performed by Zheng, et al (2012) in FS membranes as well (Zheng, et al., 2012). By controlling the fouling phenomenon through the addition of precipitation chemicals, the membranes’ cleaning can be also significantly reduced (Mishima & Nakajima, 2009). Both Mishima & Nakajima (2009) and Zheng, et al (2012) report that Fe coagulants and specifically FeCl$_3$ were slightly more effective than Al coagulants in reducing the potential fouling (Mishima & Nakajima, 2009; Zheng, et al., 2012).
According to Zheng, et al (2012), the metal hydroxides formed during chemical precipitation as excess chemicals are added are contributing to the membrane’s irreversible fouling. Therefore, fouling is not eliminated as there are always foulants which are not removed but it is delayed and reduced (Zheng, et al., 2012). By comparing iron and aluminum salts, both the study of Mishima & Nakajima (2009) and Zheng, et al (2012) resulted that iron salts contribute in more efficient way in reducing the fouling phenomenon on the membranes (Mishima & Nakajima, 2009; Zheng, et al., 2012).
4. Examined WWTP Installations

This experimental work for this master thesis work was performed at the Line 1 of the pilot WWTP at Hammarby Sjöstadskverk and at Henriksdal reningsverk in Stockholm. Therefore, the next two sections contain a detailed description of both WWTP plants.

4.1. Henriksdal Reningsverk

The (WWTP) at Henriksdal consists of mechanical, chemical and biological treatment facilities (see Figure 5). After the conventional treatment procedure, it has several sand filters as polishing stages for increasing the purity of the effluent before discharge it in the recipient, the Baltic Sea (Stockholm Vatten AB, 2010).

The mechanical treatment consists of a grit chamber and two types of screens which are able to capture the rather big particles and the sand contained in the wastewater. The screens are conventional coarse screens with a mesh 20-30 mm and the second type are fine screens with a mesh of 3-10 mm. Afterwards, a primary sedimentation is used for separating out the primary sludge. The pre-treatment is always necessary in order to avoid clogging, disruptions and wear of the equipment due to very large particles. The larger objects separated during the mechanical treatment such as tampons and plastics are grounded down and added back to the inlet while the smaller particles such as sand are going for landfill. The total residence time of the mechanical treatment is around 2 to 3 hours (Stockholm Vatten AB, 2010).

During the mechanical treatment, chemical precipitation of phosphorus is accomplished at great extent by adding ferrous sulphate (FeSO$_4$) as pre-precipitation. The addition of FeSO$_4$ enhances the removal of organic material as well, due to the fact that they are attached to the precipitants. The Fe when it is hydrolyzed in water also forms hydroxides, which are able to capture colloidal matter and therefore contribute more in the removal of organic matter though the process of coagulation as it is already mentioned before (Stockholm Vatten AB, 2010).

Important parameters in this process are the pH adjustment and the efficient mixing though aeration. The pH adjustment is important since the coagulants are able to react at specific values of pH. For iron sulphate, the required pH range for the occurrence of the chemical reaction is between 5.5 and 7.5. Aeration is important for the pre-treatment in order to remove unpleasant odors and of course for enhancing the Fe$^{2+}$ oxidization into Fe$^{3+}$ for faster and more efficient chemical precipitation. However, the aeration should not be very strong during primary settling; otherwise the solids will need more time to settle (Stockholm Vatten AB, 2010).
After the mechanical treatment the wastewater is fed into an activated sludge process for biological treatment. For sludge separation sedimentation basins for sludge’s separation are following the aeration basins. There are 7 parallel lines and 7 different zones in series for each line where each zone runs in anoxic or aerobic conditions achieving both biological degradation and nitrogen reduction. Zones 1, 2, 3 are anoxic while 4, 5, and 6 are aerobic. Zone 3 and 6 can work in both anoxic and aerobic conditions according to the circumstances. Zone 7 is used for de-oxygenation of sludge before the sedimentation basins (Stockholm Vatten AB, 2010).

During the process strong aeration is required succeeding to a 90-95% of organic material reduction measured as BOD. The process is rather sensitive to toxic chocks, high sludge loads and fluctuations in the inflow wastewater. The sludge produced during the biological stage mainly includes cell mass and metal precipitants such as iron and calcium. A proportion of this sludge is returned back as returned sludge to zone 1 to enhance the growth of several microorganisms and biodegradation rates. During the activated sludge process nitrification and denitrification also occur (Stockholm Vatten AB, 2010).

The last step before discharging the treated wastewater into the recipient is to pass through a couple of sand filters which are mainly a polishing stage. The sand filters enhance the phosphorus removal and therefore is easier to achieve the stricter requirements regarding the quality of the effluent. Furthermore, before the sand filters, FeSO₄ is then added into the process for capturing the remaining phosphorus and improve its reduction. The filters are cleaned when this is required according to several indications of their productivity. The standard procedure involves flushing with air and water two or three times in a row and the last time is accomplished only with water rinsing (Stockholm Vatten AB, 2010).

**4.2. Line 1 of the Pilot MBR at Hammarby Sjöstadsverk**

The pilot MBR line at Hammarby Sjöstadsverk is similar to the current Henriksdal’s wastewater treatment plant with some differences which are described below (see Figure 6). However, it is a small copy of the WWTP plant at Henriksdal as it going to be after it is retrofit to MBR instead of activated sludge. This line consists as well of mechanical, biological and chemical treatment. The sand filters are not used as polishing stage because the MBR effluent is clean enough and no further treatment is required (Hammarby Sjöstadsverk, Stockholm Vatten VA AB, 2013).

![Figure 6: Simplified Flow Chart of Line 1 at Hammarby Sjöstadsverk](image)
The mechanical treatment of the pilot’s line 1 consists of fine screens and a pre-sedimentation basin just exactly as at Henriksdal’s WWTP. The fine screens used are though much finer than those used at Henriksdal today while they more similar to those that will be used in the future full scale plant. The difference from Henriksdal is that currently there is not grit separation since the wastewater flow is taken from the top of the inlet at Henriksdal, and that is probably enough to ensure that no sand enters the line and furthermore there is no space inside the pilot for a grit chamber. However, the full scale MBR plant will of course have a grit chamber prior to the screens. During this stage, several particles are separated from the process called primary sludge as it is already referred. After the pre-sedimentation, ferrous sulphate (FeSO₄) is added in the system for phosphorus chemical precipitation and the effluent from the primary settler is conveyed to the next step, the biological treatment (Grundestam, 2013).

The next step includes a sequence of 6 zones in series where anoxic and aerobic stages are following the one the other for more efficient separation. Zones 1, 2, 5 and 6 are anoxic zones while zones 3 and 4 are aerobic ones. Zones 2 and 5 can also work under aerobic conditions if it is necessary. However right now the aerators are broken in zone 2 and 5 therefore they cannot be aerated properly. Furthermore, in zone 5, there is a recirculation flow to zone 1 but the recirculation can also take place from zone 6 to zone 1. In this step the biological degradation of the organic material is achieved and of course through stages of nitrification and denitrification, nitrogen reduction is achieved as well. In the last zone, sodium acetate is also added as carbon source to enhance the denitrification process. All zones have also stirrers for mixing in order to avoid sedimentation of particles and enhance biodegradation (Hammarby Sjöstadsverk, Stockholm Vatten VA AB, 2013).

Unlike Henriksdal’s WWTP, the separation of the sludge is achieved though the aid of membranes by using an aerated external submerged MBR system instead of a settling basin just after the six zones. The returned sludge from the MBR tank is then conveyed to the first aerobic zone, zone 3 since no de-oxygenation is needed (Hammarby Sjöstadsverk, Stockholm Vatten VA AB, 2013).

The full-scale MBR plant that is going to be built at Henriksdal will also have grit chambers as it is already mentioned since there is no space limitations as there are in the pilot plant. Furthermore, the returned sludge from the MBR tank is designed to be conveyed at zone 1 instead or zone 3 in order to increase the solids content in the anoxic zone for more efficient denitrification. However, this cannot be performed at the pilot since a de-oxygenation zone is needed in order to remove the oxygen from the returned sludge before it enters the anoxic zone 1. Additionally, the FeSO₄ is planned to be added in the returned sludge as well for more efficient phosphorus removal. The addition of FeSO₄ before the membrane tank or directly in the membrane tank is under consideration since the iron ions might pass through the membranes before reacting and the iron oxides precipitating might cause problems on the membranes.

4.2.1. Membrane Specifications

The membranes used are constructed from Alfa Laval and they are called hollow sheet (HS) which is actually a combination of flat sheet and hollow fiber combing the
advantages of both membrane types (see Figure 7). When using hollow fiber membranes, it is possible to perform backflushing as cleaning method and furthermore the hollow fiber membranes have high packing density which means that they have higher filtering area and therefore higher membrane productivity. The flat sheet membranes have lower screening requirements, can run at low TMP values and they are affected less from fouling and clogging phenomena. Additionally, the flat sheet does not require pumping system as they operate gravitationally (Alfa Laval, 2013). However, when this membrane configuration is applied in large facilities such as future Henriksdal WWTP suction pumps are necessary. In practice, the hollow sheet membrane is a well-designed flat sheet configuration with improved internal (permeate side) structure to minimize the internal pressure drop. This means that it is actually a flat sheet membrane with improved permeability.

![Figure 7: HS Membrane as It is designed from Alfa Laval, (Alfa Laval, 2013)](image)

At the pilot MBR, there are two membrane filtration modules (MFM) 100-80 m², made of polyvinylidene fluoride (PVDF) on non-woven polypropylene (PP) support and each module consist of 44 elements which are placed inside a well-mixed tank and each element has a surface of 1.81 m². Therefore, the total surface of each module is 44 x 1.81 m² = 79.64 m². The membrane’s porosity is estimated between 0.17 and 0.26 μm and it is categorized as MF membrane. The wastewater flows upwards among the elements and the clean effluent flows through the membrane’s sheet. The MBR tank is aerated as it is already mentioned creating a cross flow filtration and simultaneously creating scouring which is really effective in minimizing the fouling potential. The permeate is poured from all over the surface of the membrane module, thus the TMP is very small decreasing for one more time the fouling potential (Alfa Laval, 2013).
5. Experimental Part

The results of this master thesis were based on data collected from several analyses described in this chapter. The aim was to determine the sludge’s quality in order to ensure the well and efficient function of the membranes. The quality of the sludge is affected from various factors as it is already discussed in previous sections.

The sludge’s quality can be indirectly estimated by measuring the ability of the sludge to filter or the sludge’s filterability. The sludge’s filterability is a poorly examined factor so far since there is no standard method to measure it. Several methods now exist to determine the filterability of the sludge but more research is definitely necessary in order to establish a universal method. The methods that were chosen in terms of this master thesis are the time to filter (TTF) and the sludge filtration index (SFI) found in literature which are both described in detail in the next sections. As for the sludge’s quality in general, the sludge’s filterability is affected from various factors linked such as the sludge’s initial composition, the aeration and the SRT.

The measurements of filterability are definitely a really good indicator of the sludge’s quality but also other measurements are necessary to be accomplished in order to obtain a greater overview of the sludge’s properties (Hai, et al., 2014). Such measurements include the measurements of total suspended solids (TSS), volatile suspended solids (VSS), extracellular polymeric compounds (EPS), soluble microbial products (SMP), and sludge volume index (SVI).

5.1. Sampling

The analyses were based on grab samples from the MBR tank at Hammarby Sjöstadsverk pilot MBR plant (see Figure 8). The samples were tested for TSS, VSS, SVI, TTF, and SFI. Some representative samples were also taken for conducting iron analysis in the sludge but the analysis performed by an external partner.

![Figure 8: Sampling Point, MBR Tank at Hammarby Sjöstadsverk](image)
Furthermore, a few samples were taken from Henriksdal WWTP and tested for the same factors. Of course, the basic interest was to examine the sludge quality of the MBR but as long as this technology is going to be applied at Henriksdal at large scale the coming years, it was rather interesting to perform a comparison between the two sludge qualities.

In total 26 samples were taken from the MBR tank, 7 samples from line 7 zone 6 from Henriksdal and 7 samples from line 7 returned sludge from Henriksdal during the time period 3/3/2014-8/5/2014.

5.2. Experimental Methods

The next sections include analytical description of the experimental procedure for each method performed in the laboratory in terms of this master thesis work.

5.2.1. Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS)

In order to measure the total suspended solids (TSS) in the sludge sample a standardized procedure was followed. A small amount of sludge (i.e. 15, 16 ml) was first measured with a measuring cylinder and then filtrated by using the standard glass fiber filter paper (0,45 μm) under suction in order to form a cake layer on the filter (see Figure 11). The filter has of course to be weighted before the filtration. The filtration is performed in standardized filter equipment showed in figure 10 (Svenska Vatten och Avloppsverksföreningen, 1984).
The filtering equipment is connected with a water pump for providing appropriate suction. After the filtration, the sample had to dry at 105 °C for one hour in the oven (Svenska Vatten och Avloppsverksföreningen, 1984).

Then the filter with the dry sludge was again weighted (see Figure 12) and the TSS amount was calculated by using equation 4:

$$TSS\left(\frac{mg}{l}\right) = \frac{(Dry\ Residue+\ Filter)(mg)-(Filter)(mg)}{Sample\ Filtered(ml)} \times 1000 \left(\frac{mg}{l}\right) \quad (4)$$

(Svenska Vatten och Avloppsverksföreningen, 1984)

The volatile suspended solids (VSS) content was then determined by putting the already dry sample in the oven at 550 °C for one hour where the organic compounds are ignited (Svenska Vatten och Avloppsverksföreningen, 1984).
After weighting the filter with the remaining sludge again (see Figure 13), the VSS content can be calculated by using equation 5:

\[
VSS \left( \frac{mg}{l} \right) = \frac{(Dry \ Residue)(mg)-(Ignited \ Residue)(mg)}{Sample \ Filtered(ml)} \times 1000\left( \frac{mg}{l} \right)
\]

(Svenska Vatten och Avloppsverksföreningen, 1984)

5.2.2. Sludge Volume Index (SVI)

In order to determine the SVI, the sludge volume (SV) at 30 minutes should be measured. The standard procedure to measure the SV includes 1000 ml of sludge to be poured on 1000 ml measuring cylinder and left to stand for 30 minutes, as it is illustrated in figure 14. After this period, the volume of sludge at the measuring cylinder indicates the SV (Svenska Vatten och Avloppsverksföreningen, 1984).

However, it is difficult to measure MBR sludge without dilution because this sludge usually does not settle at all within the period of 30 minutes as it is too thick.
Therefore, 1:5 or even 1:10 dilution with water is usually required. The best solvent to use is the effluent water of the MBR plant (see Figure 15) since the dilution error is very small. The amount of dilution depends on the SV in the first try. The limitation for performing higher dilution is chosen to be the 250 ml. For instance, if during 1:5 dilution the SV is greater than 250 ml, higher dilution is required; while if SV is equal or less than 250 ml the result is valid and no further dilution is necessary.

The SVI can be then calculated through equation 6:

\[ SVI = \frac{SV}{TSS} \times 1000 \]  

(6)

Where,
SVI is the sludge volume index in mg/l
SV is the volume of settled solids after 30 minutes in ml/l of sludge
TSS is the amount of total suspended solids in mg/l,
(Svenska Vatten och Avloppsverksföreningen, 1984)

5.2.3. Sludge Filtration Index (SFI)

The SFI method is a filtration method and it is used to determine sludge’s filterability. For the SFI measurement, it is necessary to have a 250 ml measuring cylinder, a 500 ml measuring cylinder an agitator, a Buchner funnel (1250 ml, 140 mm circular) and of course filter papers (0.6 μm) as it is illustrated in Figures 16&17. The Buchner funnel is located on a stand and a filter paper is placed in it. The 250 ml measuring cylinder is placed just under the funnel for receiving the filtrated water. Furthermore, the agitator should be placed at height of about 0.5 cm above the funnel in order to effectively mix the sample during filtration. (Thiemig, 2012).
The procedure starts by taking 500 ml of sludge in the 500 ml measuring cylinder. Then, the sample should be poured inside the funnel and the time between the 100 ml and 150 ml should be carefully counted as it indicates the SFI value and corresponds to the sludge’s filterability. If the sample is not at room temperature a water bath is also necessary (Thiemig, 2012).

After measuring the time period required for this method, the time value should be normalized against the effect of TSS which varies from day to day. The TSS value has linear effect on the SFI and this is also proved by several experiments performed along with the analyses in the laboratory. For the same sample, dilutions 1:2 and 1:4 with tap water were performed and the SFI results were the same which means that this linear connection is confirmed.

The normalization of SFI against TSS can be calculated through equation 7:
\[ SFI(\text{norm}) = \frac{SFI}{TSS/10^4} \]  
(7)

Where,
SFI is the time measured in sec
SFI (norm) is the normalized SFI against TSS in sec / TSS %w/v
TSS is the amount of total suspended solids in mg/l, (Thiemig, 2012)

5.2.4. Time to Filter (TTF)

The TTF method is also a filtration method similar to the SFI but the procedure and the required equipment are slightly different. For this method, it is required to have a Buchner funnel (100 mm OD), a 100 ml graduated cylinder, rubber stoppers, an Erlenmeyer suction flask (1000 ml) and a 200 ml measuring cylinder and standard filter paper as it shown in Figure 18. The 100 ml graduated cylinder should be little modified in order to fit inside the flask (see Figure 19). Therefore, the funnel is placed on the top of the flask by using rubber stops for sealing the empty space. The flask is then connected to a vacuum pump which is adjusted at 15 mm Hg in order to ensure good suction for better filtration (GE Water & Process Technologies, 2009).

![Figure 18: TTF Instrumentation](image)

Then the 200 ml measuring cylinder should be filled with the sludge sample and the process is about to start. The sample from the cylinder should be poured in the funnel and as soon as the filtered water is starting to pour from the bottom of the funnel three different time periods should be carefully measured, the time between 0-25 ml, 0-50 ml, and 0-100 ml. These time periods indicate the sludge filterability according to this method (GE Water & Process Technologies, 2009).

The measurement between 0 and 100 ml is the most important since it is the one changing a lot with varying sludge qualities. When this value for a sludge having TSS between 9000 - 11000 mg/l is below 100 seconds, the filterability is considered to be very good and no serious problems are occurring. Sludge with TTF measurements between 100 - 200 seconds are considered as medium quality while sludge with TTF measurements greater than 300 seconds are considered as low or bad quality. The
time between 0 and 25 ml is usually below 5 seconds for a sludge that is of a good quality and usually has good filterability (GE Water & Process Technologies, 2009).

The normalization of TTF against TSS can be calculated the same way as in the SFI through equation 8:

$$TTF_{(norm)} = \frac{TTF}{TSS/10^4}$$

Where,
TTF is the time measured in sec
TTF (norm) is the normalized TTF against TSS in sec / TSS %w/v
TSS is the amount of total suspended solids in mg/l, (Thiemig, 2012)

5.2.5. Microscopic Observation

The samples were also examined using an optical microscope (see Figures 20&21) in order to examine the differences in sludge’s structures among the samples. For examining the sludge sample in the microscope, a drop is enough. Carefully, a small drop of sludge should be placed on a standard glass and afterwards a second glass should be placed on it in order to seal the content and remove all the air bubbles.
Each sample was graded between 0 and 6 according to template photos given in the laboratory where 0 corresponds to a sample with almost no filaments and not bulky at all while 6 corresponds to a sample with a lot of filaments and very bulky. The grading system also contains fractions of 0.5 when the sample cannot be graded with any grade and it is located in the middle. The photos illustrate typical sludge samples corresponding to the grading system from 0-6. Basically, the differences between the various sludge photos are based on differences on sludge bulkiness and also differences on the filaments which are present in the sample.

Figure 21: Optical Microscope
6. Results

The next sections describe the results derived from the experimental work and also from the calculations and analysis in terms of this master thesis. Basically, the next sections include comparisons among different factors in order to identify the correlations between different sludge properties. The main factors analyzed are the total suspended solids (TSS), the volatile suspended solids (VSS), the sludge volume index (SVI), the membrane’s permeability, the time to filter (TTF), the sludge filtration index (SFI), the phosphates content in the MBR effluent, the iron in the sludge and microscopy.

6.1. Factors Affecting the System

The permeability, the filterability and all the other parameters are affected by a number of different factors; therefore it is rather difficult sometimes to extract safe conclusions. During the sampling period several changes and disruptions took place in the MBR system therefore the results might be affected from any of them in some cases. As an example, on 5th of March several control problems caused some disturbances on the system and on the 12th of March the system had several disturbances due to service and the recirculation to zone 1 was stopped. On 14th of March, no effluent was pouring and a lot of floating material was observed. On the 20th of March a CIP with sodium hypochlorite (NaClO) and oxalic acid (H₂C₂O₄) was accomplished. On the 25th of March, the control system was reprogrammed and the MBR and the aeration was shut down while the TMP values where extremely high even the next day because they needed calibration. Due to low permeability values after reprogramming, a new CIP was planned.

Furthermore, the MBR flow changed from fixed (2,5 m³/h) to dynamic flow (max 5 m³/h, min 1,8 m³/h) after the 31st of March. The dynamic flow was 2,5 m³/h on average. On the 8th of April, another CIP with only NaClO was performed and on 15th of April a third CIP was performed with both NaClO and H₂C₂O₄. The 29th the system shut down unexpectedly and therefore, the permeability value might be also affected. During week 19, the flow was increased for about 50% reaching a mean value of 3,7 m³/h increasing the incoming load. Furthermore, the same week, the recirculation from the MBR to zone 3 changed from 10 m³/h to 8,8 m³/h an the recirculation from zone 5 to zone 1 changed from 7,5 m³/h to 12,5 m³/h. As a result, small or big disturbances can occur in the system and thus affect the various factors.

6.2. TSS – VSS for MBR

Figure 22 shows the correlation between TSS and VSS during the sampling period. As it observed the correlation between them is almost stable (70,82-77,52 %) and specifically the VSS are 73,34 % in average of the TSS. The TSS are varying between 5 700 mg/l and 10 688 mg/l.

The fact that the VSS content is an almost stable percentage of the TSS is rather good and important because it shows that the content of organics in the sludge is not varying and therefore the treatment conditions do not need any particular modification from time to time. When the organics content is too high, the sludge will be more bulky and other phenomena like foaming might occur (Radjenovic, et al., 2007).
6.3. TSS – SVI for MBR

Figure 23 shows the correlation between TSS and SVI during the sampling period. The SVI and the TSS seem to be almost inversely proportional which means that when the TSS get low values the SVI is increasing and vice versa. The samples taken at 14th and 25th of March are not included in this graph because the SV couldn’t be measured since the sludge was floating.

According to literature, the SVI is not only a good indicator of the sludge settling ability but also it is a good indicator of sludge quality since it is affected by the flocks size and the amount of filamentous bacteria which are present in the wastewater. As
the amount of filaments is increasing, the SVI is also increasing and the sludge quality is worse (Mesquita, et al., 2008).

For this sampling period the SVI was varying between 151.12 ml/g on 4th of April and 232.16 ml/g if the really extreme value 333.33 ml/g on 5th of March is neglected. From the microscopic observation, the filaments content was also varying and definitely the days with very high SVI more filaments were observed. However, the microscopic observation is very subjective and no severe scientific results can be extracted.

6.4. Sludge Filterability MBR

Figure 24, shows the filterability measurements for the MBR period during the sampling period where SFI and TTF have almost the same trend. Specifically, the TTF at 25 and 50 ml have the same trend as the TTF at 100 ml while it is not very obvious as the time measured is always very short. Therefore, it is better to compare the permeability and all the other factors analyzed below only with the TTF at 100 ml and of course SFI.

![SFI - TTF](#)

Figure 24: SFI and TTF Measurements for the MBR Samples

Generally, the sludge filterability of all the samples analyzed can be characterized as very good according to the values found in literature. For example for the TTF method, if the measured time at 25 ml is less than 5 sec and less than 100 sec at 100 ml the filterability of the sample is very good (GE Water & Process Technologies, 2009).
6.5. Permeability Normalization

The most important membrane property which is affected from the varying sludge quality is the permeability because poor quality sludge will of course create problems in the filtering ability of the membrane.

The permeability was firstly calculated from the flux and TMP. The TMP is measured automatically online while the flux is calculated by dividing the flow with 79.64 m² which is the area of the particular membranes. The measurements were given in a daily basis taken every 5 minutes. Therefore, the average permeability of the day was further calculated for the days of sampling. However, the value of permeability calculated for each day is affected from many factors and not only from the sludge quality and thus this makes the comparison with the SFI or TTF rather difficult. In other words, the value of permeability needs normalization in order to be comparable with the SFI or TTF.

Firstly, one factor affecting the permeability is the wastewater’s temperature and thus this should be taken into consideration. The samples when they were analyzed for SFI or TTF are considered to be at room temperature (20 °C) therefore the reference temperature for permeability’s normalization is the 20 °C. The normalization against the effect of temperature can be done by using the equation for viscosity:

\[ (9.802 \times 10^{-6} \times T) + (1.13 \times 10^{-3} \times T^2) - (5.793 \times 10^{-2} \times T) + 1.785 \]  

(9)

Where,
T is the temperature in °C

(Yoon, 2011)

After calculating the viscosity for both the measured wastewater temperature and the reference temperature (20 °C), the normalized permeability against the temperature can calculated from equation 10:

\[ Perm_T = Perm \frac{\nu_T}{\nu_{20}} \]  

(10)

Where
Perm is the measured permeability
\( \nu_T \) is the viscosity at the measured wastewater’s temperature
\( \nu_{20} \) is the viscosity at the reference temperature of 20 °C
T is the wastewater’s temperature in °C

The permeability is mainly affected from the membranes’ fouling and some other factors. In the term other factors the sludge quality is the main contributor. However, other unpredictable factors are also included such as system’s failures, aeration disruptions etc. Therefore, the aim is to normalize the permeability against the fouling which is cumulative and thus causes a slight reduction in permeability from day to day. The idea for excluding the fouling effect from the calculated permeability was to firstly calculate the theoretical permeability which would only be affected from the
fouling. The theoretical permeability could be estimated by finding the equation representing the rate of permeability’s reduction between two CIPs and then calculate the theoretical permeability for each sampling day.

Figures 25&26 represent the permeability for the two membranes respectively between two successional CIPs from the week 48 of 2013 until the week 12 of 2014. The permeability values in this chart are weekly averages and they are normalized against temperature, the way it is described above. The two equations indicated on the graphs represent the rate of permeability’s reduction if the fouling is the only factor affecting the membranes efficiency. These equations are chosen to indicate the permeability reduction over time as they were had the highest coefficient of determination ($R^2$) according to the calculations. They are of course approximate equations since no other way to normalize permeability against fouling was found in literature. The fouling is considered the only factor affecting the permeability since the graph represents a long time period where other factors are negligible.

**Figure 25: Permeability of Membrane A between two CIPs**

\[
y = 1931.7x^{0.135} \\
R^2 = 0.8271
\]

**Figure 26: Permeability of Membrane B between two CIPs**

\[
y = -178.2\ln(x) + 1868.1 \\
R^2 = 0.7121
\]
From the two equations on the charts, the y represents the permeability while the x represents the time after the CIP in weeks. However, the sampling in terms of this master thesis was performed on a daily basis. Therefore, the x should be somehow converted into days in order to calculate the theoretical permeability for each sample analyzed. As each week has 7 days and considering Monday as the first day, each day after the CIP can be calculated by using equation 10:

\[
Day \ After \ the \ CIP = Week \ After \ the \ CIP + \left( \frac{D}{7} \right) \tag{11}
\]

Where,
D are the days of each week, with Monday indicated as 1 and Sunday as 7

In order to normalize the measured permeability against the fouling effect, it is considered in a simple way that the equation of measured permeability in connection with the fouling is:

\[
Perm_T = -(Fouling + Other \ Factors) \tag{12}
\]

The minus symbol is used as the fouling and other factors have negative contribution to the permeability. And the theoretical permeability as it is affected only from the fouling it is considered to be:

\[
Perm_{Th} = -(Fouling) \tag{13}
\]

Therefore, the normalized permeability can be calculated by equation 14:

\[
Perm_{T,F} = Perm_T - Perm_{Th} = Other \ Factors \tag{14}
\]

Where,
Perm_{Th} is the theoretical permeability
Perm_{T} is the normalized permeability against temperature
Perm_{T,F} is the normalized permeability against temperature and fouling

### 6.6. SFI, TTF – Permeability

Theoretically, the correlation between the sludge’s filterability and the membrane’s permeability is proportional as when the filterability is good the permeability should be high as well. The filterability though is measured in time units which mean that the filterability is good when the measuring time is short and the other way around. In other words, the filterability is good if TTF / SFI are low. Thus, the connection between the SFI and TTF with the permeability, it is expected to be inversely proportional. However, in reality the connection between factors is not always exactly the same as in theory as other factors interact with the system.

In Figures 27&28, it is shown the connection between the normalized SFI against the normalized permeability for membranes A and B respectively.

The connection between the SFI and normalized permeability seems to be inversely proportional as it was expected from the beginning. When the permeability
is high the SFI is low and vice versa. These results indicate that the SFI method is a good method for measuring the sludge filterability in order to estimate the permeability variations.

The normalized permeability in the membrane B seems to match more with the SFI measurements as for example it shown for the sample taken in 24th of March. That day the SFI measurement is higher than before and thus the permeability in membrane B is obviously decreased while in membrane A is not obvious that reduction in normalized permeability.

At Figures 29&30, it is shown the connection between the normalized TTF at 100 ml against the normalized permeability for membranes A and B respectively. The measured TTF at 25 and 50 ml are not included in these charts because they don’t vary too much from sample to sample as it is already mentioned in order to show a trendline and the chart could be very crowded.
The connection between the TTF and normalized permeability seems to be also inversely proportional like the SFI. When the permeability is high the TTF is low and vice versa. This is a good indication of the TTF that it could possibly be used to predict changes in membranes’ normalized permeability. However, some peaks for the TTF on the 19th of March and 4th of April are completely different from both the measured SFI that days and the permeability variation. Therefore, either the TTF is a very sensitive method for small sludge compositional variations which are not even affect the permeability or SFI is more reliable method in measuring the sludge filterability. From these graphs, the SFI is chosen as a better method to measure the filterability but of course in the next sections more factors will be analyzed in order to conclude in more safe results.
6.7. SFI, TTF – Percentage of VSS for MBR

At Figures 31&32, it is shown the connection between the percentage of volatiles in the sludge with SFI and TTF at 100 ml respectively.

From the above graphs, it is observed that the filterability is better when the percentage of volatiles is lower while when the volatiles content of the sludge is increasing the filterability is poorer. There is no clearer concentration but it is very important to have an indication if the volatiles content affects positively or negatively the sludge quality and further on the membranes’ operation.
6.8. Fe Content in Sludge – Permeability for MBR

Figures 33 & 34 show the correlation between the permeability and the iron contained in the sludge. The iron in the sludge was measured from an external partner, Eurofins Environment Testing Sweden AB and one sample per week was analyzed.

The connection between these two factors is proportional and really clear. The permeability is better when there is more iron in the sludge while it seems that the permeability is lower when the iron is also less. The connection seems to be better for membrane A since the value of permeability on the 19th of March is lower than the expected for membrane B. According to literature, it is observed that the addition of metal salts and specifically iron salts improve the permeability of the membranes as it is already mentioned (Radjenovic, et al., 2007)

![Fe - Permeability (Membrane A)](image1)

Figure 33: Fe content in the Sludge vs. Permeability of Membrane A

![Fe - Permeability (Membrane B)](image2)

Figure 34: Fe content in the Sludge vs. Permeability of Membrane B
6.9. Fe Content in Sludge – SFI, TTF for MBR

As it is expected from the results given in the previous section, the iron content in the sludge should be inversely proportional with the SFI and TTF since it is proportional with the permeability. Figures 35&36 show the correlation between SFI and TTF and the iron in the sludge respectively.

The trend between the two factors is as it is expected inversely proportional in both charts. However, the SFI represents a clearer connection than the TTF in that case. The measurement on the 19th of March should be lower for the TTF as the iron content is increasing compared to the measurement taken on the 10th of March. However, this measurement is really good for the SFI since it is decreasing according to what is expected. For both methods, the measurement on 7th of April is an outlier and other factors might interfere as well.
6.10. SFI, TTF – Phosphates for MBR

Figures 37&38 show the connection between phosphates content measured in the effluent of the MBR and SFI and TTF respectively. The trend between the phosphates content and SFI or TTF is not very clear. However, the only thing that can be extracted from this graph is the fact that in higher SFI and TTF values the phosphate content is higher while in lower SFI and TTF values the phosphates are lower as well. Therefore, a proportional rate between the two factors is observed from the graphs. Of course it is not expected to have a very clear connection between these factors since the low phosphorus content is not the only parameter affecting the sludge filterability.

The weak correlation that seems to exist here can quite possible be explained by both \( \text{PO}_4 \) and SFI being affected by iron content. The SFI trendline shows a better connection with phosphate trendline despite the value on the 19\(^{th}\) of March while the TTF trendline has some peaks on the on the 25\(^{th}\) of March and on the 4\(^{th}\) of April.
which do not corresponds to similar peaks at the phosphates trendline. The last two measurements on 7th and 8th of May are not that connected with the phosphates content in both the SFI and TTF case.

6.11. SFI, TTF – Microscopy for MBR

The differences among the samples regarding the bulkiness and the filaments content was not very large and therefore the grades give for each sample are 4, 4.5 and 5 according to the grading system explained in the experimental part. Furthermore, it would be very interesting to compare the measured SFI or TTF with the grades given for each sample as the filaments content is own of the major factors affecting the sludge quality. Therefore, Figures 39&40 represent the connection between SFI, TTF with microscopy grades respectively.

![Microscopy - SFI](image1.png)

Figure 39: Microscopy Grades vs. SFI

![Microscopy - TTF](image2.png)

Figure 40: Microscopy Grades vs. TTF
The connection between SFI and microscopy observation is very clear since there is obviously a proportional rate between the two factors analyzed. When the filamentous content was observed to be higher compared to other samples, the SFI measurement seems to follow the same pattern. This is something that it can be expected as a result according to literature (Radjenovic, et al., 2007).

The TTF measurements do not show exactly the same trend as it was observed for the SFI. They seem to have a more general connection with the microscope grades while in some particular days such as the 19th of March, the 4th of April or the 29th of April, the connection seems inversely proportional and totally different from the SFI. Therefore, the SFI is seems to be a much better factor of examining the filterability in connection with the filaments content and bulkiness of the sludge.

During the microscopy, several photos were taken from each sample and some of them are included in Appendix I.

6.12. Comparing TSS and SVI of Pilot MBR and Henriksdal

Figure 41 illustrates the TSS and VSS variation during the sampling period for the MBR, the biological line 7 - zone 6 (L7Z6) of the activated sludge process from Henriksdal and the returned sludge from line 7 at Henriksdal.

As it was expected and it is illustrated in the graph the WWTP at Henriksdal is running in lower TSS values compared to the MBR since the SRT is much lower in an activated sludge plant. However, the returned sludge samples are thicker in biosolids content since they represent a more concentrated version of the same wastewater sample as the biological line.

The percentage of volatiles is a little higher in the samples from the activated sludge compared to those collected from the MBR tank. As an example, the average VSS percentage over the total TSS for the biological line at Henriksdal is 78.51 %, for the returned sludge it is 76.21 % and for the MBR it is 73.34 %.
As Henriksdal and MBR receive approximately the same input every day, the difference between the VSS content of the above samples could be explained because the two plants are running in different TSS values. In other words, the sludge in the MBR is slightly richer in inorganic substances since the organics are degraded more efficiently than in the activated sludge. Therefore, the solids return as returned sludge in the MBR have less organics content and therefore the percentage of VSS is lower in the MBR tank.

Figure 42 illustrates the SVI variation during the sampling period for the MBR, the Henriksdal L7Z6 and the L7 returned.

![SVI Henriksdal vs MBR](image)

As it is observed, the SVI measured in both L7Z6 and L7 returned sludge is better than the SVI measured in the MBR. For the L7Z6 the SVI varied between 130 ml/g and 250 ml/g, for the L7 returned sludge between 133.33 ml/g and 181.23 ml/g and for the MBR between 151.12 ml/g and 333.33 ml/g. It is logical that the SVI for the activated sludge plant is better compared to MBR since the sludge separation in the activated sludge is based on solids settleability. Furthermore, the higher SRT in the MBR results in poorer settleability which is not required though.

For the MBR not all the samples are included in the charts since samples from Henriksdal were taken in a weekly basis and therefore only the samples of the MBR taken the same days are being included.

### 6.13. Comparing Filterability of Pilot MBR and Henriksdal

Figures 43&43 show the measured TTF at 25, 50, 100 ml and the measured SFI for Henriksdal’s L7Z6 and L7 returned sludge.

As it is observed in these filterability measurements for Henriksdal, the TTF at 25 and 50 ml is almost stable all across the sampling period; therefore it is not a good indicator to compare with the MBR. As a result, the next charts do not include the TTF measurements at 25 and 50 ml.
Figures 43 & 44 illustrate the TTF and SFI variation during the sampling period for the MBR, the biological line 7 - zone 6 of the activated sludge process from Henriksdal and the returned sludge from line 7 at Henriksdal.

The TTF and SFI are meant to be performed for MBR sludge and not for CAS sludge. However, in terms of this master thesis, few samples were taken from Henriksdal which is a CAS system in order to compare them. The comparison is not very easy though. The TTF measurements show that the MBR sludge has the worst filterability at the beginning while at the end both Henriksdal’s samples seems to be worse than MBR. On the other hand according to the SFI method the biological line from Henriksdal seems to have the worse filterability while the MBR and returned sludge follow the same behavior as before.

Figures 47-50 show the measured TTF and SFI in comparison with the microscopy grades given for each sample for Henriksdal L7Z6 and L7 returned sludge.

As it is observed, the microscopy grades have a proportional tendency with both the SFI and TTF. For the L7Z6, both SFI and TTF seem good methods to determine the sludge’s bulkiness and filaments. While for the L7 returned sludge, the SFI measurement seems to be a better indicator since it follows the microscopy rates.
Of course, all these are based on subjectivity but on the other hand these charts are an extra indication to support that the methods examined are suitable to measure filterability and furthermore to support that possibly SFI match better with all the factors analyzed above.

However, since the grading is based on comparisons between the prototype picture and the sample, the comparison between MBR and Henriksdal is easy.
The samples from MBR were always more bulky than Henriksdal and this is expected due to higher SRT. The filamentous content of the samples was varying from sample to sample and therefore slightly different grades were given mainly according to the amount of filaments. Henriksdal samples are rich in filaments even though the SRT is very short. One explanation might be that the aeration at Henriksdal is not as intense as in MBR. The MBR samples have also high content of filaments which of course are varying significantly. When the TTF and SFI values were high for a sample, the filaments content was usually greater for both MBR and Henriksdal samples. That means as it is expected than very high filaments content might affect the sludge quality and further on the membranes operation.
7. Discussions

This master thesis mainly aimed to identify a suitable and reliable method for measuring the sludge filterability and further define the sludge quality in order to prevent severe disturbances on the membranes operation. The two methods examined for measuring the filterability were the SFI and TTF. However, it is difficult to definitely conclude which of them is the best since many factors interfere with them and generally affect the system as it is already mentioned. Therefore, all the parameters examined and compared in the results section should be mentioned and discussed in order to extract some conclusions.

The way of comparing the factors in the results section by using double y-axis was chosen because it was easier to identify the correlation between the factors both from a quantitative and a qualitative perspective. The results seemed difficult to be examined when a normal x-y plot was used probably because more measurements are needed in order to calculate a correlation factor and thus find the mathematical connection among the factors. Furthermore, when the term proportional or inversely proportional was used was not exactly referring to the mathematical terms but it was used in order to show that when a factor is increasing the other is increasing or decreasing respectively.

The error of the measurements performed in the laboratory is relatively low since usually at least three measurements were done for determining each parameter. The average of the three different measurements was used in the results section. In all cases the three or more measurements for each sample and parameter were very close therefore the standard deviation is very low in all cases. In case of permeability, the values used represent the daily average in order to minimize the error. However, if the system was not properly working for some time, the values of permeability corresponding to the problematic time are excluded from the average since they are not typical and the error can be rather increased in that case.

In order to facilitate the decision of which method is better for determining the sludge filterability, Table 1 summarizes the factors compared in the results section and the connections identified among them.

<table>
<thead>
<tr>
<th>FACTORS</th>
<th>Permeability</th>
<th>TSS</th>
<th>SFI</th>
<th>TTF</th>
</tr>
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<tr>
<td>SFI</td>
<td>IP</td>
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<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>TTF</td>
<td>IP</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>VSS</td>
<td>N/A</td>
<td>P</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>%VSS</td>
<td>N/A</td>
<td>N/A</td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td>SVI</td>
<td>N/A</td>
<td>IP</td>
<td>N/A</td>
<td>N/A</td>
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<td>N/A</td>
<td>P</td>
<td>P</td>
</tr>
<tr>
<td>Fe</td>
<td>P</td>
<td>N/A</td>
<td>IP</td>
<td>IP</td>
</tr>
<tr>
<td>Microscopy</td>
<td>N/A</td>
<td>N/A</td>
<td>P</td>
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</tr>
</tbody>
</table>

*Where P corresponds to proportional and IP to inversely proportional

In general, the first days of experimentation, basically between the 10th and 28th of March, the sludge showed pourer filterability compared with later days. During the
first days, the normalized permeability was also low, the iron in the sludge was low, the phosphates content in the effluent was high and the system was functioning in a fixed flow. Therefore, a combination of these factors affected negatively the sludge’s filterability resulting in higher SFI and TTF values. After the 1st of April, the filterability was better as the SFI and TTF values were significantly reduced. This filterability improvement was followed by improved normalized permeability while the iron content in the sludge was higher, the phosphates in the effluent were less and the flow was changed from fixed to varying or dynamic flow.

In order to be more specific, on the 5th of March the control problems caused a couple of disturbances in the system and probably for that reason the calculated SVI was too high, actually the highest during the entire period of experiments reaching the value of 333.33 ml/g. Furthermore, the VSS:TSS ratio was also affected resulting in the highest also value of 77.52 % during the entire period. The filaments content was also high according to the microscopic observation but unfortunately no filterability measurements were performed since the equipment was not yet available. The normalized permeability for both membranes and the iron content in the sludge was very low that day which means that the system was seriously affected from the bad sludge quality and therefore the filterability should be estimated considering all the other parameters mentioned as being very poor resulting in high SFI and TTF values.

On the 12th of March, a significant increase in both SFI and TTF measurements was observed without following significant permeability reduction. The phosphates were also increased that day but there is no clue for the iron since no sample was collected for iron analysis. The system was under service and the recirculation from the MBR to zone 1 was shut down resulting in higher TSS content in the MBR tank compared to the previous days. Possibly, the permeability was not affected since the service performed resulted anyway in keeping the permeability at a reasonable level.

On the 14th of March, the sludge quality was very bad and the SV could not be measured since a lot a floating material was present. The SFI, TTF and phosphates content were lower compared to the previous day while the permeability in membrane A was higher and in membrane B lower. During the sampling, the MBR effluent was not pouring and the tank was full of floating substances. Therefore, the results from that day are confusing since membrane A is affected from the higher filterability measured while membrane B seem to be affected from the floating material observed during SV and thus shows a worse permeability value. In that case the SFI and TTF predicted only the change in membrane A, but combining with SV measurement and optical observation during sampling the behavior of membrane B is reasonable.

On the 19th of March a totally opposite measurement is observed for the two methods measuring filterability, the SFI and TTF. The SFI decreased while the TTF increased. The normalized permeability of both membranes and the iron content in the sludge that day are all increased following the behavior of SFI method as inversely proportional factors. The following statement enhances the opinion that SFI predicted reliably and precisely the permeability increase. However, that day the SV could not be measured in the first try since a lot of floating material was present which means that the TTF method was probably affected from the floating substances resulting in wrong results. Another explanation can be that the collected samples from MBR tanks are collected in bottles of 0.5 l and usually 4 bottles needed in order to perform the
entire series of all the experiments. Therefore, small variations from bottle to bottle even if they are collected the one after the other might affect the various measurements in a different way.

On the 20th of March the CIP performed did not had any particular effect on membranes’ permeability however, the values might be affected just after it. On the 21st and 24th of March the SFI value increased while the TTF value decreased compared with the previous day. In that case the permeability followed the TTF behavior and it was slightly increased that day and the phosphates as proportional with TTF or SFI were decreased enhancing the opinion that TTF predicted better the behavior of the other factors and of course the filterability. However, the SVI was rather high that day showing that the filterability is not as good as it was assumed by checking the TTF or the normalized permeability and in that case SFI was again better correlated with that factor. Of course the comparison with the previous or next sampling day might lead in misunderstandings or wrong results since the conditions and sludge composition differ from day to day. Therefore, the results should be analyzed in both day by day basis but they should also be linked with all the measurements performed for the same sample. In that case, the SFI might be again better since without comparing it with the previous day but only with the other measurements, the results favor it.

On the 25th all the results can be seriously affected from the shut down due to reprogramming. However, both SFI and TTF seem to have predicted in a good way the behavior of normalized permeability and they are in good correlation with the phosphates content. However, the SVI it was impossible to be measured due to floating as the aeration was closed for a couple of hours.

On 27th and 28th of March, the results were generally better than previous days. On the 27th specifically, the results were in absolute connection among all the factors being analyzed since the sludge had a really good quality and less filaments compared with the other days. On the 28th the results for normalized permeability for membrane A are better connected with the SFI since SFI is slightly increasing and permeability A is slightly decreasing while the results for normalized permeability B are better connected with TTF since TTF is slightly decreasing and permeability B is slightly increasing. The differences between permeability’s and filterability’s increase or decrease is very low and thus the results concerning only the permeability are concerned as good for both SFI and TTF. However, the phosphates content and the microscopy, where the filaments content was greater for the 28th, are following the proportionality of SFI for these two days even though the difference is small. As a result, the SFI is considered again better method.

On the 31st of March, the inlet flow in line 1 changed from fixed flow to dynamic as it is already mentioned. By observing the results just after this change, the filterability seem to be much better even if that correlation is random and not due to flow change and the filaments observed during microscopy are less compared to the previous days. The 1st of April, higher permeability values for both membranes can be explained with lower SFI and TTF values, thus better filterability. The iron in the sludge measured that day is also greater facilitating the idea that the iron improves both the filterability and permeability. The phosphates are lower that day having proportional trendline with both SFI and TTF.
On the 2\textsuperscript{nd} of April, only permeability of membrane A is following a further increase due to even better filterability while membrane B shows a slight decrease in permeability. During microscopy, fewer filaments were observed for this day compared with the previous one; therefore it is again reassured that the sample had a better image leading to higher permeability. The phosphates are not proportional that day and they are increasing but this can be explained since they were 0.09 mg/l on the 1\textsuperscript{st} of April which is one of the lowest values reached; thus it is not expected to further decrease the day after.

On the 4\textsuperscript{th} of April, TTF is considered as better than SFI since it shows a significant increase while the permeability of both membranes is decreasing. The SFI has totally the opposite behavior together with the microscopy. The phosphates are remaining stable and no further increase or decrease is observed. On the 7\textsuperscript{th} of April, the results are similar with the 4\textsuperscript{th} but neither SFI nor TTF are compatible with the permeability. Both SFI and TTF are decreasing and the permeability and the phosphates are decreasing as well which is rather paradox. However, the only true and reliable correlation of the day is the decrease of the iron in the sludge which is proportional to the permeability. The results of 7\textsuperscript{th} should not really affect the decision regarding TTF or SFI as a filterability method because none of them was found to match with the other factors and also the differences discussed are very minor and they can be considered even negligible from a more general perspective.

On the 8\textsuperscript{th} of April, a new CIP was accomplished as it is already mentioned and in somehow the results of the coming days might be affected. On the 10\textsuperscript{th} of April, TTF is again considered as a better method since the normalized permeability of both membranes is showing a small reduction and the TTF a small increase. The same happens for the phosphate content in the effluent while the SFI shows a slight decrease. On the 11\textsuperscript{th} of April, on the other hand, the SFI is now following the further reduction of permeability and the phosphate increase. However, all the details discussed from the 1\textsuperscript{st} of April until the 11\textsuperscript{th} of April are so minor since the permeability is almost stable showing small differences.

On the 14\textsuperscript{th} of April, the normalized permeability of both membranes is significantly increased and the SFI is reduced showing a clear trend. The TTF and the phosphates content are increasing though without predicting the permeability trend. The iron analysis that day shows an even larger iron content in the sludge which results in higher permeability showing one more time that the iron possibly affects positively the membranes’ permeability.

The 29\textsuperscript{th} of April, the system shut down due to electricity failure and therefore this might affect significantly the values analyzed despite the fact that the MBR and the aeration was still working. The normalized permeability showed a significant increase and the filterability was improved since both SFI and TTF reached lower values. However, this high increase of the normalized permeability might be misleading since the real permeability faced a high decrease that day.

The 30\textsuperscript{th} of April, the permeability shows a decreasing behavior in comparison to the previous day but as long as the results obtained during the 29\textsuperscript{th} are controversial it is better not to correlate these results with the 29\textsuperscript{th}. Comparing the results with previously days from the 29\textsuperscript{th}, the permeability shows a slight increase and this is
reassured by checking the SFI and TTF values which are even lower than before. The iron content in the sludge is slightly less than before, not really contributing to the better filterability that day and the phosphates are also slightly less showing a different trend than it is discussed before.

The last week of experiments the flow was increased for about 50% reaching a mean value of 3.7 m³/h increasing the incoming load, the recirculation from the MBR to zone 3 changed from 10 m³/h to 8.8 m³/h and the recirculation from zone 5 to zone 1 changed from 7.5 m³/h to 12.5 m³/h. These changes seem to have affected the filterability of the sludge since the SFI and TTF are slightly increased during the 5th, 7th and 8th of May. During the 5th the normalized permeability of membrane B is showing a small reduction but during the 7th and 8th the normalized permeability of both membranes seem to have a very small increase. This can be explained due to the fact that the iron measured on the 5th and 8th of May was really high by reaching its higher value of 580 mg/l and 720 mg/l respectively during the entire sampling period. The phosphates showed a continuing decrease thought these days which show an inversely proportional behavior compared with SFI and TTF.

In all cases analyzed in the results section, the SFI seems to be more reliable method and it definitely has a better connection with all factors it was compared with. Of course in some cases the TTF was found to be a better method but the most important thing is to identify a method that is reliable on a continuous basis. However, a good way of measuring the filterability is to perform both SFI and TTF in order to minimize the error of one method. Furthermore, since the biological system is a very complex system all the experiments performed in terms of this thesis should be performed in a daily basis in order to have a more complete idea of the sludge quality and in order to ensure the efficient operation of the membranes.

The best connection of the SFI was with the microscopy in both MBR and Henriksdal samples but it would be less scientific to support the results only according to values which are very subjective. The connection between SFI and permeability was obvious in a more general and overall basis and not for each and every measurement. This of course can be explained in many ways. The most profound reason is that the normalized permeability despite the normalization against temperature and fouling is a value affected from many parameters which cannot be easily identified. More stable conditions without changes in the system for larger time periods are probably necessary in order to have more accurate results. One limitation factor as well regarding the changes occurring was the time limitation since the biological system is not a fast responding system and thus some time is always required in order to observe significant results.

The connection of SFI and phosphates concentration in the effluent was rather good, not in an every sample basis but as an overall image and trendline. However, the phosphates content in the effluent are not considered as the most important measurement in order to correlate them with the filterability or normalized permeability. The iron in the sludge, on the other hand, had a rather obvious inversely proportional connection with the SFI but the best and clearer result was the connection between the iron content and the normalized permeability. Especially, the permeability of the membrane A was perfectly matching with the iron measurement in a proportional way supporting the relative literature. Following these results, the iron
content affects significantly the filterability and the permeability and it is a really good factor to estimate for having an idea for the sludge’s quality.

By choosing the SFI as the most suitable method to determine the sludge filterability, it is also easy to perform a comparison between MBR and Henriksdal filterability which was rather difficult before since different results are coming up from SFI and TTF measurements. As a result, the filterability of L7Z6 is worse than L7 returned sludge or MBR despite the fact of low solids content. The MBR and L7 returned sludge have similar filterability with the second being in a rather worse situation since it shows rather big variations among the samples both in filterability and solids content. The fact that MBR shows a better filterability according to the SFI compared to Henriksdal can be considered as logical since the MBR operates under more efficient aeration and higher SRT. Higher SRT according to literature has been linked with lower EPS and SMP production which are factors responsible for poor sludge filterability (Mishima & Nakajima, 2009).

Another factor connected with lower sludge quality in Henriksdal is the percentage of volatiles (Radjenovic, et al., 2007). In both L7Z6 and L7 returned sludge the percentage of VSS is higher than in the MBR reaching the value of 78,51 % for L7Z6 and 76,21 % for L7 returned sludge while the MBR has 73,34 % volatiles in average. This is also obvious in the MBR sludge which had a larger percentage of volatiles during the first days of the experiments when the filterability was poorer while after the 1st of April that the filterability is better the percentage of VSS in the sludge is decreasing.

Finally, the SVI is also a good indicator of the sludge filterability, despite the fact that measures the solids settleability which is not a matter of concern for the MBR. With the SVI you can estimate the degree of floating material and the bulkiness due to filaments and therefore you can estimate if the sludge’s quality is better or worse compared with other days.
8. Conclusions

Concerning all the results, the limitations and the analysis above, sludge filtration index (SFI) is considered as a better method compared to time to filter (TTF) for measuring the sludge filterability. However, as both methods are similar, under different conditions or during a larger time period the results might be in favor of TTF.

Furthermore, the most significant and important result derived from this master thesis work were the proportionality between the iron content (Fe) in the sludge and normalized permeability. The iron content seems to be a very valuable indicator considering its connection with the permeability and it can be also controlled in order to conclude in higher permeability values.

The microscopy and specifically the filaments content can be directly connected with the SFI and TTF in a proportional way and this is extremely interesting since the worse sludge quality can be linked with filaments growth.

The shutdown, several disruptions due to service or several changes in the system are observed to affect as well the sludge quality to a high extent. During the shutdown, the sludge shows a poor filterability since the aeration is reduced or stopped. Furthermore, the change from fixed to dynamic flow affected positively the filterability as it was observed resulting in lower SFI and TTF values.

Finally, for determining the sludge quality, the filterability measurements are generally not enough since a biological system is very complex and many factors contribute for improving or not the mixed liquor quality. As a result, the SVI, the TSS, the percentage of VSS, the iron content and the filaments content are very important factors to be determined for an MBR sludge sample as well.
9. Further Research

In order to have a better overview of the sludge quality in the MBR, more samples are needed to be analyzed for a larger time period. Specifically, it would be really good to collect samples for an entire year in order to get as much information linked with varying sludge composition, climatic variations, system modifications, disruptions etc. as possible. In order to have more accurate results stable conditions are also necessary for larger time periods. In other words changes in the system should be performed after the determination of sludge quality and permeability is identified for specific conditions such as flow, recirculation rate, iron dosage etc.

Generally, a more specific identification of the sludge biology might be of interest since the sludge quality is totally linked with the biology. Furthermore, the extracellular polymeric compounds (EPS) and the soluble microbial products (SMP) should be measured in the grab samples since they seem to affect very much the sludge’s filterability according to the literature review. Therefore, by estimating the conditions which are more favorable for EPS and SMP production, it would be much easier to get a better overview of the sludge’s quality. It would also be very interesting to perform a particles size distribution in order to identify several particles categories such as the submicron and colloidal which are usually responsible for bad sludge quality according to literature. Other filterability measurements might also be done as part of extensive research in order to identify in practice if SFI and TTF are better or not.

As the iron connection with permeability was identified to some extent, the connection with aluminium might be interesting to examine as well in order to see which one has better effects on permeability and of course in filterability as well. Different types of chemicals might also have different results, therefore ferric chloride (FeCl₃) would also be good to examine as precipitation chemical since it contains trivalent ions of iron and thus the concentration of the iron in the sludge might be different and therefore affect differently the sludge quality and the permeability.
References


Appendix I – Microscopy

Figures 51-76, represent typical microscopy photos from the MBR, the Henriksdal L7Z6 and the Henriksdal L7 returned sludge, including the grade which given by examining the entire sample and not the specific photo given below.

Figure 51: Sample MBR, 5/3/2014. Grade: 4.0

Figure 52: Sample MBR, 19/3/2014. Grade: 4.5

Figure 53: Sample MBR, 24/3/2014. Grade: 5.0
Figure 54: Sample MBR, 28/3/2014. Grade: 4,5

Figure 55: Sample MBR, 1/4/2014. Grade: 4,5

Figure 56: Sample MBR, 2/4/2014. Grade: 4,5
Figure 57: Sample MBR, 4/4/2014. Grade: 4,0

Figure 58: Sample MBR, 7/4/2014. Grade: 4,0

Figure 59: Sample MBR, 10/4/2014. Grade: 4,0
Figure 60: Sample MBR, 11/4/2014. Grade: 4.0

Figure 61: Sample MBR, 14/4/2014. Grade: 4.0

Figure 62: Sample MBR, 29/4/2014. Grade: 4.0
Figure 63: Sample MBR, 7/5/2014. Grade: 5,0

Figure 64: Sample MBR, 8/5/2014. Grade: 5,0

Figure 65: Sample MBR, 5/5/2014. Grade: 4,5
Figure 66: Sample Henriksdal L7Z6, 5/3/2014. Grade: 2.5

Figure 67: Sample Henriksdal L7Z6, 19/3/2014. Grade: 3.0

Figure 68: Sample Henriksdal L7Z6, 24/3/2014. Grade: 3.0
Figure 69: Sample Henriksdal L7Z6, 1/4/2014. Grade: 3.0

Figure 70: Sample Henriksdal L7Z6, 14/4/2014. Grade: 3.5

Figure 71: Sample Henriksdal L7Z6, 14/4/2014. Grade: 3.5
Figure 72: Sample L7 Returned Sludge, 5/3/2014. Grade: 3,0

Figure 73: Sample L7 Returned Sludge, 19/3/2014. Grade: 3,5

Figure 74: Sample L7 Returned Sludge, 24/3/2014. Grade: 3,5
Figure 75: Sample L7 Returned Sludge, 8/5/2014. Grade: 3,5

Figure 76: Sample L7 Returned Sludge, 14/4/2014. Grade: 3,5