Energy optimization of biological wastewater treatment using rotating belt filters upstream BNR

by

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Summary

Treatment of wastewater to acceptable permit standards requires energy, mostly as electricity. The typical energy demand for various wastewater treatment technologies vary from 0.30 to 1.50 kWh/m^3 of treated water. For conventional activated sludge, the energy demand is 0.30 – 0.70 kWh/m^3, in which 50 - 60 % is used for the aeration of the aerobic reactors. As a high fraction of wastewater, COD appear as total suspended solids (TSS), primary treatment has an impact on the performance and energy demand of the downstream processes. Consequently, efficient TSS removal during primary treatment will result in reduced organic load and a reduction in oxygen demand in the downstream biological treatment and resulting in significant energy cost savings. In addition, enhanced primary treatment generates more sludge suitable for anaerobic digestion and corresponding biogas and energy production. The goal of this research was to define the particle size cut-off for TSS and particulate COD removal prior to biological nitrogen removal. The main question would be how much TSS and associated COD removal is acceptable in order to maintain sufficient nitrogen removal and to maximize biogas production. Laboratory and pilot experiments performed at the laboratory of Aquateam in Oslo and at Nordre Follo WWTP (NFR) near Oslo, using wastewater and sludge from NFR and from Bekkelaget WWTP (BRA) in Oslo.

Anoxic batch tests were with both activated sludge and biofilm processes in laboratory scale sequencing batch reactors (SBRs) in order to determine the impact of TSS on denitrification rates. Filtration of wastewater upstream SBR was by several fine mesh sieves, from 150 µm to 1.2 µm pores. The TSS and COD removal were inversely proportional to the filter pores. COD removal was from 43 % with 18 µm and 21 % with 150 µm sieves for NFR wastewater. For BRA wastewater the removal was from 42 % with 18 µm to 32 % with 90 µm sieves. By analyzing the slope of the curve for nitrate reduction in the batch tests, identification of the denitrification rates according to readily biodegradable COD (RBCOD), K1, slowly biodegradable COD (SBCOD), K2, and endogenous denitrification, K3 came about. The tests with wastewater from BRA had higher K1, between 0.18 and 0.26 gNO_x-N/gVSS-d (Test 2) compared to wastewater from NFR with K1 between 0.05 and 0.09 gNO_x-N/gVSS-d

(Test 1). One reason for the difference could be that the activated sludge was collected from BRA and was adapted to that wastewater compared to the wastewater from NFR. However, the K2 and K3 rates were similar for the two wastewaters. In the tests with MBBR, K1 varied between 0.80 – 2.43 gNO_x-N/m^2-d for wastewater from NFR (Test 3) and between 1.22 – 2.69 gNO_x-N/m^2-d for wastewater from BRA (Test 4). The K2 rate was slightly higher for NFR wastewater compared to BRA, probably caused by the biofilm media from NFR, while the K3 rates were quite similar during Test 3 and Test 4. Regarding the effect of TSS removal on the specific denitrification rates, it appeared to be of minor importance, while the main effect was on the overall denitrification potential.

Three laboratory scale SBRs at three liter each were operated during three periods, investigating the effect of TSS removal with different sieves on biological nitrogen removal. In period 1 (P1) the wastewater was filtered with 1.2 and 18 µm sieve, in period 2 (P2) filtered with 33 and 90 µm sieves and in period 3 (P3) filtered with 55 and 150 µm sieves. In addition, one SBR had raw wastewater in all periods as control. The comparison of the performances showed that the SBRs fed filtered wastewater removed between 65 and 75 % COD while the SBRs fed raw wastewater removed between 70 and 91 % COD. This indicates that reducing the COD load on the SBR will affect the performance of the process. However, when including the removal of COD in the primary treatment, similar or slightly higher TSS and COD removals were observed in the SBRs fed with filtered wastewater compared to the control reactors. The nitrogen removal was about 60 % for the SBRs fed raw and wastewater filtered at 33 µm and larger pore sizes. The SBRs fed wastewater filtered with smaller pore sizes had reduced nitrogen removal efficiency. SBRs fed with filtered wastewater produced more sludge compared to the control reactor, about 70 – 184 % more in P1, up to 139 % in P2 and 41 – 64 % more in P3. The calculations show that the SBRs fed filtered wastewater required less oxygen compared to the control SBR. The oxygen requirement decreased by 37 % in the SBR fed wastewater filtered at 18 µm and by 59 % in the reactor fed wastewater filtered at 1.2 µm. The difference in the total oxygen demand during the biological process was mainly due to the oxygen consumed for degradation of COD.

Based on the laboratory tests conclusions a Salsnes Filter model SF 1000 with 33 µm filter cloth was used as a primary treatment for the pilot scale experiments with moving bed biofilm reactor (MBBR), membrane bioreactor (MBR) and activated sludge sequencing batch reactor (SBR).

The experiments used two parallel trains of MBBRs during the experiments, one fed raw wastewater (control reactor) and one fed filtered wastewater. Each MBBR train was composed of two 4 L anoxic reactors and two 6 L aerobic reactors in series, corresponding to 40 % anoxic and 60 % aerobic volume. Nitrified effluent from Reactor 4 was recycled to Reactor 1 at approximately twice the influent flow rate. The MBBR fed raw wastewater removed about 41 % of TCOD, 50 % of TN and 18 % of TP and the MBBR fed filtered wastewater removed 41 % of COD, 41 % of TN and 3 % of TP (secondary separation not included). When filtration was applied as secondary separation, the MBBR train fed filtered wastewater had COD, TN and TP removal efficiencies of 74 %, 61 % and 65 %, respectively. The removal efficiencies in the MBBR train fed raw wastewater were 91 % COD, 68 % TN and 73 % TP. However, accounting for removals in the filter, the overall removal efficiencies were similar for the two MBBR trains. The average sludge production in the control MBBR Train was 13.7 gTS/d compared to 17.3 gTS/d in the MBBR Train with filtered wastewater. The MBBR train fed raw wastewater consumed 10.8 gO_2/d while the MBBR train fed filtered wastewater consumed 8.3 gO_2/d. The difference in oxygen demand is because removal of about a third of the influent COD is by the filter. The reduction in oxygen demand was about 30 % compared to the control reactor. Consequently, the removal of particulate COD with a 33 µm filter did not have any significant impact on nitrogen removal but resulted in higher sludge production and potential savings in aeration.

Two pilot-scale MBRs operated in parallel with the MBBR trains. One MBR train (train A) was fed raw wastewater as control and one train (train B) was fed wastewater filtered with at 33 µm filter. Each MBR train was composed of two anoxic reactors of 10 L each, equipped with mechanical mixers and one aerobic reactor of 25 L with a submerged hollow fiber membrane ZeeWeed-10 at 40 nm nominal pore size. Nitrified activated sludge was recycled from Reactor 3 to Reactor 1 at twice the influent wastewater flow rate. The assessment of the MBRs' performance showed a removal efficiency at nearly the same level for both trains. For

Train B, if the filter performance was not considered, the MBR removed about 90 % COD, 69 % TN and 78 % TP. However, when the removal by the filter was accounted, the overall removal efficiencies were 94 % COD, 80 % TP and 73 % TN, as for train A. The results showed that at the present level of TSS and COD removal in primary treatment, there was still enough COD to achieve a high level of TN removal in both trains. The high phosphorus removal indicated some enhanced biological phosphorus removal caused by anaerobic conditions in the second anoxic reactor when denitrification was completed. TMP was higher for the membrane receiving raw wastewater with an average of 46 ± 9 mbar compared to 26 ± 7 mbar for the train treating filtered wastewater. The sludge production in Train A was only composed of biological sludge and was about 21.3 gTS/d. In Train B the total sludge production was 31.2 gTS/d and consisted of the filter (primary) sludge and biological sludge. The oxygen requirement was about 35 % higher for the MBR treating raw wastewater compared to the MBR treating filtered wastewater. Thus, removal of TSS and COD from the influent wastewater was beneficial in terms of cost savings in reduced energy demand for aeration. Overall, the removal of particulate COD with a 33 µm filter did not have a negative effect on biological nitrogen removal in MBR.

In the last experiments five pilot-scale SBRs evaluated the effect of filtration as primary treatment on biological nitrogen removal and investigated the influence of temperature and SRT on the process. Transparent PVC tanks each of 10 L were used as SBRs and the temperatures were 17 and 8 °C while the SBRs was operated at SRT of 12 and 6 days. At SRT of 12 days, the SBR fed filtered wastewater had removal efficiencies of 79 % TSS, 69 % COD and 51 % TN while the control SBR fed raw wastewater had removal efficiencies of about 83 % TSS, 72 % COD and 56 % TN. Including the TN removal of the primary filter, the overall TN removal was 54 %. At 6 days SRT the SBR treating filtered wastewater had removal efficiencies of 70 % TSS, 61 % COD and 72 % TN while the control SBR had removal efficiencies of 57 % TSS, 56 % COD and 56 % TN. At 17 °C nitrification was nearly complete at SRT of 12 days, while it was between 80 and 90 % complete at SRT of 6 days. The most significant effect of lower temperature was on nitrification, which was complete in the SBRs at 17 °C, while at 8 °C and 12 days SRT, only 50 % of the ammonium converted to nitrate. The TN removal efficiency was only 39 % at 8 °C while it was 56 % at 17 °C. The sludge production

increased by 39 % in the system with primary treatment at 12 days SRT; whereas it increased by 11 % at 6 days SRT. The oxygen demand for the control SBR was 22 % higher at SRT of 12 days compared to the SBR fed filtered wastewater. At SRT of 6 days, the oxygen demand was 66 % higher in the SBR fed raw wastewater compared to the SBR fed filtered wastewater. Consequently, the removal of particulate COD with 33 µm filter did not have any negative impact on the nitrogen removal in the SBR for the tested wastewater, mainly because of sufficient COD.

Thus, an overall assessment of the experimental data indicates that a filter of 33 µm is the optimum choice prior to biological nitrogen removal as it provided both satisfactory COD and TN removal.

The steady state version of the activated sludge model 1 (ASM1) were applied to simulate the experimental results of the MBR experiments. The simulation predicted the observed effluent concentration of the soluble nitrogen compounds and the nitrogen removal efficiency quite well. Predictions of the MLSS and MLVSS were close to the measured values for the test with filtered wastewater while for the test with raw wastewater there was about 15 % difference. The good predictions of nitrogen compounds could be due to that the system had near complete nitrification and denitrification, while the difference for the MLSS and MLVSS could be due to inaccurate analyses and the fact that many parameters are involved in those predictions and could affect the final results.

There is a big potential for savings in energy by applying primary treatment as filtration in wastewater treatment without affecting the downstream biological processes significantly. The results also indicate the importance of laboratory or pilot testing to determine the optimum level of applied primary treatment. The results also show the potential of expanding the use of experimental data by applying mathematical modelling of the system.

***Keywords**: Biological nitrogen removal, primary treatment, Salsnes Filter, organic matter, anoxic batch test, SBR, MBBR, MBR, oxygen demand, sludge production, methane production, energy, nitrification and denitrification*

Table of Contents

List of Figures

List of Tables

Abbreviations and Nomenclature

AD	Anaerobic digestion
Al	Aluminum
AOB	Ammonia Oxidizing Bacteria
APHA	American Public Health Association
AS	Activated Sludge
AUR	Ammonium Utilization Rate
BFT	Back Flushing Tank
BMP	Biomethane Potential
BNR	Biological Nitrogen Removal or Biological Nutrient Removal
BOD	Biochemical Oxygen Demand
BRA	Bekkelaget wastewater treatment plant
C/N	Carbon to Nitrogen ratio
$CaCO_3$	Calcium bicarbonate
CAS	Conventional Activated Sludge
CH_4	Methane
CHP	Combined heat and power
CO_2	Carbon dioxide
COD	Chemical Oxygen Demand
COD/VSS	COD to VSS ratio
COD_{bx}	Biodegradable particulate COD
d	Day
DAF	Dissolved Air Flotation
DO	Dissolved Oxygen
DS	Dry Solids
ENR	Enhanced nutrient removal
EPA	Environmental Protection Agency
F/M	Food to Microorganism loading (gCOD/gMLVSS-d) or Food to Microorganism ratio in case of lab-scale batch tests (gCOD/gMLVSS)
ff	Flocculated then Filtered
FP	Feed Pump
g	Gram
h	Hour
HRT	Hydraulic Retention Time

Km	Kilometer
KNO_3	Potassium Nitrate
kWh	Kilowatt per hour
L	Liter
LI	Level Indicator
LMH	$L/m^2.h$
m^2	Square Meter
m^3	Cubic Meter
MA	Membrane Aerator
mbar	Millibar
MBBR	Moving Bed Biofilm Reactor
MBR	Membrane Bioreactor
mg	Milligram
min	Minute
MLSS	Mixed Liquor Suspended Solids
MLVSS	Mixed Liquor Volatile Suspended Solids
mm	Millimeter
NaOH	Sodium Hydroxide
NEIWPCC	New England Interstate Water Pollution Control Commission
NFR	Nordre Follo wastewater treatment plant
NH_4-N	Ammonium-Nitrogen
NO_2-N	Nitrite-Nitrogen
NO_3-N	Nitrate-Nitrogen
NOB	Nitrite Oxidizing Bacteria
NO_x-N	Nitrate-Nitrogen + Nitrite-Nitrogen
NS	Norwegian Standard
NSFC	National Small Flows Clearinghouse
NUR	Nitrate Utilization Rate
O_2	Dioxygen
°C	Degree Celsius
OD	oxygen demand
OD_{COD}	Oxygen required for the degradation of organic matter
OD_{NH4-N}	Oxygen required for nitrification
OD_T	Total oxygen demand
OR_{NOx-N}	Oxygen recovered during denitrification
p.e.	Person equivalent

Pa.DEP	Pennsylvania Department of Environmental Protection
PBP	Permeate and Back-flushing Pump
pCOD	Particulate Chemical Oxygen Demand
PI	Pressure Indicator
PO_4-P	Orthophosphate-phosphorus
PSD	Particle size distribution
PVC	Polyvinylchloride
Q_i	Influent Flow Rate
Q_r	Recycle Flow Rate
R	Reactor
RA	Reactor Aeration
RAS	Return Activated Sludge
RBCOD	Readily Biodegradable Chemical Oxygen Demand
RBF	Rotating Belt Filter
RP	Recirculation Pump
S	Biofilm Surface Area
s	Second
SBCOD	Slowly Biodegradable Chemical Oxygen Demand
SBR	Sequencing Batch Reactor
sCOD	Soluble or filtered Chemical Oxygen Demand
SCVFA	Short chain volatile fatty acid
SDNR	Specific Denitrification Rate
SF	Salsnes Filter
SNR	Specific Nitrification Rate
SRT	Solids Retention Time
sTN	Soluble of filtered Total Nitrogen
sTP	Soluble of filtered Total Phosphorus
SVI	Sludge Volume Index
TCOD	Total Chemical Oxygen Demand
TMP	Transmembrane Pressure
TN	Total Nitrogen
TOC	Total organic carbon
TP	Total Phosphorus
TS	Total solids
TSS	Total Suspended Solids

V	Volume
VS	Volatile solids
VSS	Volatile Suspended Solids
WAS	Waste Activated Sludge
WEF	Water Environment Federation
Wh	Watt per hour
WP	Waste Pump
WW	Wastewater
WWTP	Wastewater Treatment Plant
Y_H	Heterotrophic yield
ZW	ZeeWeed Membrane

Thesis outline

This Thesis is subdivided into 7 chapters.

Chapter 1. Introduction and objectives focus on new challenges faced by wastewater treatment plants. A brief overview shows different primary treatment methods and their effects on downstream biological processes. Objectives listed at the end of Chapter 1.

Chapter 2. Literature review provides background knowledge for better understanding of this research. It focuses on the basic principles of biological nitrogen removal, mathematical modeling and filtration technology.

Chapter 3. Materials and methods present methodology used to gather all data presented in this thesis.

Chapter 4. Denitrification rates as a function of TSS focus on two screening tests based on anoxic batch tests and laboratory scale sequencing batch reactors fed filtered wastewater, filter openings between 1.2 µm and 150 µm.

Results obtained during the anoxic batch tests was the basis of the paper published in Water Science and Technology Journal, while the results from the lab-scale SBRs was published in the IWA Proceedings on Pretreatment of water and wastewater.

- **Razafimanantsoa, V. A**., Ydstebø, L., Bilstad, T., Sahu, A. K. & Rusten, B. 2013. Effect of selective organic fractions on denitrification rates using Salsnes Filter as a primary treatment. *Water Science and Technology* 69 (9), 1942-1948.
- **Razafimanantsoa, V. A**., Vargas Charry, P. A.; Ydstebø, L., Bilstad, T., Sahu, A. K. & Rusten, B. 2014. Impact of selective size distribution of influent suspended solids on downstream biological processes. *Proceedings*, *A069*. IWA Conference on Pretreatment of Water and Wastewater "The status and progress on Water Pretreatment Technology", Shanghai, China 18-21 May 2014.

Chapter 5. Verification of the defined particle size cut-off on pilot-scale BNR shows the application of the defined filter size from screening tests upstream of pilot scale **moving bed biofilm reactors**, **membrane bioreactors** and **sequencing batch reactors.** Two papers emerged from these experiments:

- **Razafimanantsoa, V. A.**, Adyasari, D., Ydstebø, L., Bilstad, T., Sahu, A. K. & Rusten, B. 2019. Pilot-scale study to investigate the impact of rotating belt filter upstream of MBR for nitrogen removal. *Water Science and Technology*, **79** (3), 458-465.
- Rusten, B., **Razafimanantsoa, V. A**., Andriamiarinjaka, M. A., Otis, C. L. & Sahu, A. K. 2016. Impact of fine mesh sieve primary treatment on nitrogen removal in moving bed biofilm reactors. *Water Science and Technology*, **73** (2), 337-344.

Chapter 6. Steady state simulation shows the usefulness of a steady state model to describe biological nutrient removal.

Chapter 7. Conclusions summarize findings from the different tests.

Figure 1 summarizes the different chapters of the thesis.

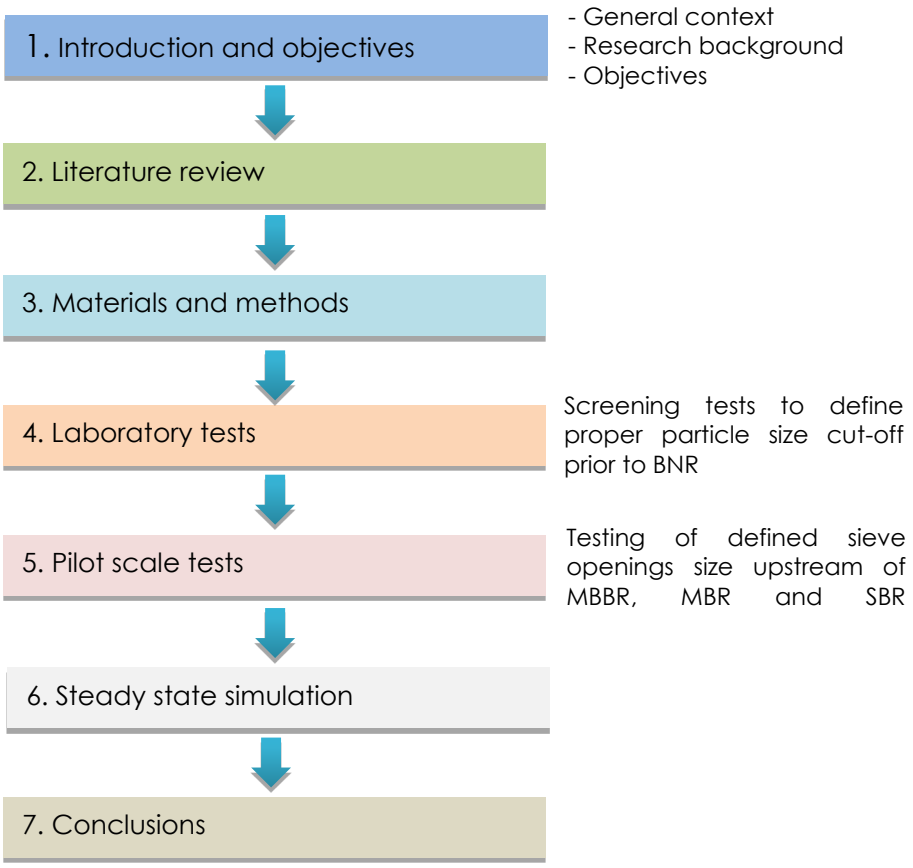


Figure 1. Schematic representation of the Ph.D. Thesis

Chapter 1

Introduction and objectives

1.1. New challenge in wastewater treatment

Prior to 1970 wastewater treatment focused on removing suspended solids and floatable materials, treatment of biodegradable organic matter and elimination of pathogenic microorganisms. Between 1970 and 1990 it became necessary to remove nutrients from wastewater due to eutrophication; i.e., fertilizing of nitrogen and phosphorus in rivers and lakes (Topare *et al*., 2011; Henze *et al*., 2008). From 1990, increasing scientific knowledge resulted in focus on health, related to toxic and potentially toxic chemicals released into the environment.

Increasing energy costs and concerns of global climate-change caused a new challenge to treatment plant operators. Thus, over the last few decades the concept of net-zero energy usage and net energy production in wastewater treatment facilities were gaining momentum (Gikas & Tsoutsos, 2014). Currently, there is increasing focus on reducing costs and increasing sustainability by minimizing net energy usage and recovery of materials (Hofman *et al.*, 2011; Sutton *et al*., 2011; Vestraete *et al*., 2009). Wastewater treatment facilities with nutrient removal are achieving a reduction of 20 – 30 % in energy for wastewater treatment by installing efficient and modern pumps, mixers and blowers of increased efficiencies and using fine bubble diffusers coupled to dissolved oxygen control systems. A net energy reduction of 40 – 60 % results by combining anaerobic sludge digestion and co-generation of power from biogas. However, this is still far from energy self-sufficient wastewater treatment (Roehl & Davey, 2011).

A typical wastewater treatment plant normally includes a preliminary, a primary and secondary treatment, in addition to sludge handling and sometimes disinfection. Preliminary treatment is the first stage of wastewater treatment. It comprises screening and grit removal. The purpose of coarse screens is to remove large particles through a bar rack with openings from 6 mm to 6 cm depending on the characteristics of the influent wastewater (Metcalf & Eddy, 2003). Grit chambers are removing sand, gravel, cinders or other heavy solid materials that have subsiding velocities or specific gravities substantially greater than the organic materials in wastewater. The role of preliminary treatment is to protect downstream equipment from abrasion and abnormal wear. A purpose is also to reduce formation of heavy deposits in pipelines, channels and conduits as well as reduce frequency of digester cleaning from excessive accumulation of grit (Metcalf & Eddy, 2003).

Primary treatment is usually a physical treatment method and in combination with chemical treatment removes part of suspended solids in the influent wastewater. Secondary treatment is the core process in wastewater treatment and removes the remaining organic matter and nutrients from primary treatment through biological process either under aerobic and/or anoxic conditions (Topare *et al.*, 2011). Some of the most common treatment processes employed are aerated lagoons (von Sperling, 2007; Metcalf & Eddy, 2003).

- activated sludge (Doherty, 2017; von Sperling & Chernicharo, 2005; Metcalf & Eddy, 2003; Grady *et al.*, 1999);
- sequencing batch reactor (Mahvi, 2008; Zhou *et al.*, 2006; NEIWPCC, 2005; Obaja *et al*., 2005; Li & Zhang, 2002; Lin & Chang, 2000; White & Schnabel 1998; Rim *et al*., 1997; Keller *et al.*, 1997);
- membrane bioreactor (Bracklaw *et al.*, 2007; Meng *et al*., 2007; Meng & Yang, 2007; Leiknes & Ødegaard, 2006; Jenkins *et al*., 2004; Gander et al., 2000; Dijk & Roncken, 1997; Gunder & Krauth, 1998; Casey *et al*., 1995; Yamamoto *et al*., 1989);
- moving bed biofilm reactor (Brinkley *et al*., 2013; Chu & Wang, 2011; Storhaug, 2009; Kermani *et al.*, 2009; Ødegaard, 2006; Ødegaard, 1999; Ødegaard *et al*., 1994; Rusten *et al*., 1995);
- anaerobic wastewater treatment (Metcalf & Eddy, 2003).

Sludge treatment consists of handling sludge produced during primary and secondary treatments to prevent sludge from harming the environment (Andreoli & von Sperling, 2007). Such treatment may include chemical conditioning by adding chemicals to improve sludge dewaterability, physical conditioning by heating, sludge dewatering to reduce the moisture content of the sludge either by belt filtration, sludge drying beds, vacuum filtration, centrifugation or filter presses, and finally sludge stabilization by aerobic or anaerobic digestion and storage (Guyer, 2011). Finally, the purpose of disinfection of effluent wastewater is to impede the growth of pathogenic microorganisms that may cause serious health effects, especially where treated wastewater is reused (Daw *et al.*, 2012).

Collecting, treating and discharging municipal wastewater to acceptable effluent standards require energy. WERF reported typical energy demands for various wastewater treatment technologies from 0.30 to 1.50 kWh/m^3 (Tarallo, 2014). For instance, conventional activated sludge process demands about 0.30 – 0.70 kWh/m^3, with 50 % - 60 % consumed for aeration of the aerobic reactors (Metcalf & Eddy, 2003). The energy consumption for wastewater treatment depends on water flow, influent characteristics, plant size, effluent quality and technologies adopted (Fitzsimons *et al.*, 2016).

The potential energy available in the raw wastewater exceeds significantly the energy requirement for the treatment process (Wett *et al.*, 2007). Energy stored in organics is a COD load.

Due to the strong correlation between wastewater COD and total suspended solids (TSS), primary treatment could have an impact on the performance of downstream processes (Bixio *et al.*, 2000; Levine *et al.*, 1985). Consequently, efficient particle removal during primary treatment results in reduced organic load and a reduced oxygen demand in downstream biological treatment, which results in significant energy savings in secondary treatment (Rusten, 2005a; van Nieuvenhuijzen, 2000).

Because of increased focus on energy consumption, design and operation of wastewater treatment plants have also focused more on energy efficiency. In addition, energy recovery as biogas from anaerobic digestion has gained increased focus as a part of the overall

energy budget. Combination of treatment methods of low energy consumption along with maximizing sludge production and subsequent energy recovery is now the dominating concept in treatment plant design.

1.2. Objectives

The goal of this research is to define the particle size cut-off for particulate organic matter removal prior to biological nitrogen removal. The main question to answer is how much COD removal is too much for negating proper BNR? Several fine mesh sieves prepared the different feeds for the biological process and their effects on denitrification were determined.

The specific objectives of this research study were to characterize the influent wastewater before and after primary treatment.

- Determine the optimum particle size cut-off prior to BNR by conducting anoxic batch tests using both activated sludge and biofilm processes and using laboratory scale activated sludge sequencing batch reactors (SBRs)
- Test the defined mesh size in front of a pilot-scale moving bed biofilm reactor (MBBR), a membrane bioreactor (MBR) and SBR
- Evaluate the impact of temperature and sludge retention time (SRT) on the biological process
- Apply the steady state activated sludge model to describe the relation between SS and COD removal in primary treatment and subsequent nutrient removal

Chapter 2

Literature review

2.1. Biological nitrogen removal

Nutrient removal became a requirement for wastewater treatment plants (WWTPs) from the second half of the 20th century owing to the fertilizing effect of nitrogen and phosphorus causing eutrophication. This research is concerned with nitrogen removal. Phosphorus removal in a conventional biological wastewater treatment is generally less than 20 % (Park *et al.*, 1997), however, phosphorus is commonly removed through chemical precipitation or by enhanced biological phosphorus removal. Nitrification and denitrification are biological processes removing nitrogen, but assimilation of N due to growth and removal of N in sludge are also mechanisms that contribute to total N removal.

2.1.1. Nitrification

a. Stoichiometry

Nitrification is a prerequisite for denitrification and describes oxidation of ammonium-nitrogen to nitrite and nitrate. This is a two-step aerobic process mediated by autotrophic bacteria in which ammonium-nitrogen is first converted to nitrite by ammonium oxidizing bacteria (AOB), and then converted to nitrate by nitrate oxidizing bacteria or NOB (Li *et al.*, 2005; Harms *et al.*, 2003; Aoi *et al.*, 2000).

The oxidation of ammonium to nitrite by nitrification with AOBs is given by

$$NH_4^+ + 1.5O_2 \rightarrow 2H^+ + H_2O + NO_2^-$$

From the above chemical reaction, 1.5 moles of oxygen are required to oxidize one mole of ammonium and two moles of hydrogen result. Thus,

the oxygen requirement to produce nitrite is 3.43 gO_2/gNH_4-N (=1.5*32/14).

The conversion of nitrite to nitrate (nitratation) by NOBs is

$$NO_2^- + 0.5O_2 \rightarrow NO_3^-$$

1.14 g of oxygen is required to oxidize 1 g of nitrite to nitrate. (=0.5*32/14).

The overall reaction for the two-step oxidation process also accounts for alkalinity consumption by the hydrogen produced

$$NH_4^+ + 2O_2 + 2HCO_3^- \rightarrow NO_3^- + 2CO_2 + 3H_2O$$

Theoretically, the conversion of ammonium-nitrogen to nitrate, both expressed as N, requires 4.57 $mgO_2/mgNH_4$-N and 7.14 mg as $CaCO_3/mgNH_4$-N. However, not all ammonium becomes nitrate, as some is for the cell synthesis of nitrifiers. The oxygen consumption becomes 4.33 $mgO_2/mgNH_4$-N, with 3.22 mgO_2 used for the conversion of ammonium to nitrite and 1.11 mgO_2 for the oxidation of nitrite to nitrate; and the alkalinity consumption is 6.8 mg $CaCO_3/mgNH_4$-N (Werzernack & Gannon, 1967).

b. Environmental conditions

Nitrifiers are highly susceptible to a wide range of environmental factors such as SRT, dissolved oxygen, pH, temperature, organic load, NH4-N concentration and presence of inhibitors (Gerardi, 2002).

SRT is the principal factor, which determines whether a biological process will support nitrification as nitrifiers have lower growth rate compared to heterotrophs. The maximum growth rate of nitrifiers at 20 °C is between 0.62 d-1 and 0.92 d-1. In contrast, that of heterotrophs is typically in the range of 4 d-1 to 13.2 d-1 (Rittman & McCarty, 2001). Therefore, minimum SRT for nitrification considers nitrifiers, not heterotrophs.

Nitrification is slow at low DO concentrations due to a relatively high half-saturation constant for oxygen (Metcalf & Eddy, 2003). The second step of nitrification is oxidation of nitrite to nitrate and is even more sensitive

to DO levels than the first step of oxidation of ammonia to nitrite. As a result, nitrite may accumulate with insufficient DO (Blackburne *et al.*, 2007). Minimum dissolved oxygen of 2 mg/L is common and recommended for efficient nitrification.

Nitrification consumes significant alkalinity and without adequate pH control, system failure may occur. AOB is responsible for loss of alkalinity due production of nitrous acid. The optimal pH for maintaining nitrification is in the range of 7.2 – 8 (Okabe *et al.*, 2011).

Temperature impose limitations for nitrification to proceed at an acceptable rate. Obaja *et al.* (2005) carried out an experiment using identical operational conditions with nine (9) different temperatures between 8 °C and 25 °C. The results showed a substantial reduction of the ammonia removal efficiencies at temperature below 16 °C. According to Metcalf & Eddy (2003), the optimum temperature for nitrification is between 28 °C and 32 °C.

The organic load to the biological process also affects nitrification efficiency as organics control the growth of heterotrophs. High organic load increases the growth of heterotrophs and sludge production, thus reducing the fraction of nitrifiers in the system (Okabe *et al.*, 2011). Nitrification has dual effect on COD removal. On one hand, COD removal decreases with high nitrification rates because of the higher activity of nitrifiers. On the other hand, when nitrification rate increases, nitrate forms and subsequently more COD is removed due to denitrification (Jonoud *et al.*, 2003).

2.1.2. Denitrification

a. Stoichiometry

Denitrification is biological reduction of nitrate to nitrogen gas under anoxic conditions. The conversion of nitrate to nitrogen gas involves a multiple step process (Payne, 1981). Nitrate reduces to nitrite, and then produces nitric oxide, nitrous oxide and finally nitrogen gas (Tisdale *et al.*, 1993). Denitrification is by heterotrophs, which can utilize nitrate (or nitrite) instead of oxygen as the final electron acceptor.

Each denitrification step is by a half-reaction where e^- denotes electron equivalents transferred from the organic substrate as described in the following equations

$$e^- + 0.5NO_3^- + H^+ \rightarrow 0.5NO_2^- + 0.5H_2O$$

$$e^- + NO_2^- + 2H^+ \rightarrow NO + H_2O$$

$$e^- + NO + H^+ \rightarrow 0.5N_2O + 0.5H_2O$$

$$e^- + 0.5N_2O + H^+ \rightarrow 0.5N_2 + 0.5H_2O$$

The net reaction for complete denitrification is by combining the four equations

$$e^- + 0.20NO_3^- + 1.2H^+ \rightarrow 0.1N_2 + 0.6H_2O$$

Denitrification offers several benefits in comparison to nitrification (Henze *et al.*, 2008) such as reduction in nitrate concentration, recovery of alkalinity (3.57 mg as $CaCO_3$/mg NO_3-N), and oxygen (2.86 mgO_2/$mgNO_3$-N).

b. Environmental conditions

Several factors affect denitrification; i.e., presence of dissolved oxygen (DO), pH, temperature, heterotroph concentration and presence of suitable electron donor.

Presence of DO inhibits denitrification reaction because oxygen acts as electron acceptor over nitrate, and aerobic conditions repress enzymes involved in denitrification (Zumft, 1997). The inhibition is reversible once DO levels decrease. Studies have shown a decrease in denitrification rates from 10 % to 50 % with an increase in DO from 0.2 mg/L to 2 mg/L (Naidoo, 1999).

Temperature influences growth of denitrifying bacteria, and thus on denitrification rates. Rates increase rapidly with increasing temperature until a maximum and decreases with further temperature increase. Conversely, denitrification decreases with declining temperature, with an optimal ranging from 35 °C to 50 °C (Barnes & Bliss 1983). A decrease

in temperature from 20 °C to 10 °C resulted in a significant reduction in denitrification rates, approximately 60 – 70 % (Mokhayeri *et al.*2006; Dold *et al.,* 2005; Nyberg *et al.*, 1996; Christensson *et al.*, 1994).

Arceivala (1981) indicated pH values in the range of 7.5 to 9.2, while Barnes & Bliss (1983) reported a range from 6.5 to 7.5. Several studies have shown that maximum denitrification occurs at pH between 7 and 7.5 and decreases approximately linearly with reduction and increase in pH from the optimum (Urbain *et al*., 1995, Wang *et al*., 1995).

The composition and concentration of COD also affect the rates of denitrification (Grabinska-Loniewszka, 1991). Denitrification needs enough organic COD to provide the required energy for conversion. The origin of the organic COD might be from the influent wastewater, self-generated by microorganisms through lysis, or provided externally.

2.1.3. Importance of COD on nitrogen removal

COD serves two purposes during denitrification. One fraction ($1-Y_H$) oxidized by denitrifying bacteria to CO_2 and H_2O, from which energy is released and this energy is utilized for assimilation of a second portion of COD (Y_H).

COD from influent wastewater subdivided into distinct biodegradable fractions are biodegraded by denitrifying bacteria at different rates. The readily biodegradable (RBCOD) fraction comprised mainly of soluble organic materials and the slowly biodegradable (SBCOD) fraction consist of large molecules, colloids and particulates (Ekama & Marais, 1979). Municipal wastewater, after primary treatment, usually contains approximately 10 % to 30 % readily biodegradable COD and 40 % – 60 % slowly biodegradable COD (Czerwionka *et al.*, 2008; Lagarde *et al.*, 2005; Orhon *et al.*, 1997).

When using influent wastewater as a carbon source for denitrification, three different rates are generally recognized. The first and highest rate reflects the simultaneous utilization of the RBCOD and SBCOD (K1 + K2) by denitrifying bacteria. Then, a second slower phase where the specific denitrification rate (K2) defined by the utilization of only SBCOD from the influent wastewater and self-generated through organism decay and

lysis. SBCOD must be broken into smaller compounds prior to uptake by microorganisms (Henze *et al*., 2008). Finally, the third and lowest rate (K3) is the endogenous decay and consumption of nitrate for cell maintenance (Gu & Onnis-Hayden, 2010).

Besides the type of carbon source, denitrification is strongly susceptible to the concentration of the carbon source and the carbon to nitrogen (C/N) ratio. C/N ratios exert some control over how heterotrophs and nitrifiers consume nitrate and COD (Chu *et al*., 2006). The C/N ratio required for complete nitrate reduction to nitrogen gas depends on the nature of the carbon source. Meng et *al.* (2008) investigated the impact of C/N ratio on nitrogen removal. Three C/N values were tested, 4.8, 10 and 15, and associated TN removal efficiencies were 49 %, 73 % and 68 %, respectively. At low C/N ratio, denitrifying organisms did not get enough carbon for proper denitrification, resulting in low nitrogen removal. The optimum nitrogen removal was with a C/N ratio of 10. Nitrogen removal was not continuously increasing along with the increase in C/N ratio due to nitrate limitation.

2.1.4. Process configurations

One aspect of system design is to make proper use of available resources. In this research, optimal utilization of available COD in the wastewater was important. Wuhrman process, Ludzack-Ettinger process and Bardenpho process configurations were developed.

In Wuhrman's post-denitrification process, Figure 2, the first reactor is aerobic, allowing nitrification to take place. The second reactor is anoxic. The energy source for denitrification is from endogenous degradation of biomass. The denitrification rate is low due to slow release of energy from biomass decay. To obtain a meaningful efficiency of the denitrification, the anoxic fraction of the system needs to be large, and depending on the sludge age, this may disturb nitrification (Henze *et al.*, 2008). Endogenous decay can potentially drive post-anoxic denitrification but is unreliable (Vocks *et al.*, 2005). Therefore, adding an external carbon source assures high efficiency nitrogen removal (Grady *et al*., 1999).

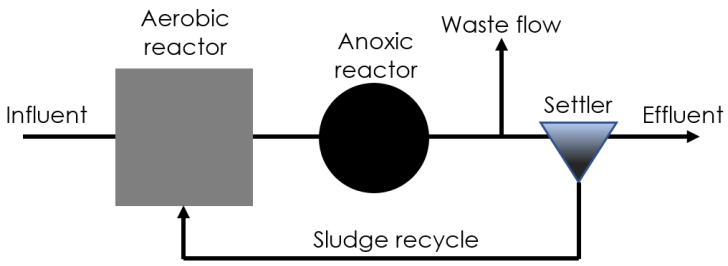


Figure 2. Post-denitrification system

COD from the influent wastewater in the Ludzack-Ettinger process, Figure 3, is the energy source for denitrification. It consists of two reactors in series, partially separated from each other. The denitrification efficiencies are variable due to the lack of control of the interchange of the content between the two zones.

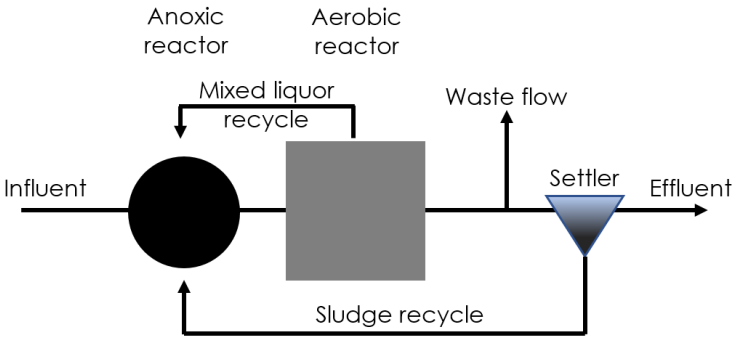


Figure 3. Modified Ludzack-Ettinger system

A significant improvement of the pre-denitrification process is the complete separation of the anoxic and aerobic reactors. Barnard (1973) introduced mixed liquor recycling from the secondary aerobic reactor to the pre-anoxic reactor. Pre-denitrification can potentially yield high denitrification rates given the RBCOD supply and can further reduce the energy demand due to the elimination of COD without aeration (Metcalf & Eddy, 2003). One drawback of the process is the operating costs associated with mixed liquor recycling. In addition, presence of oxygen in the recycled mixed liquor is upsetting the process. High mixed

liquor recycling dilutes the influent wastewater, reducing denitrification rates and therefore nitrogen removal.

To overcome such drawbacks, Figure 4 shows a combined Wuhrman and modified Ludzack-Ettinger process known as Bardenpho. It consists of adding secondary anoxic and aerobic reactors in the system.

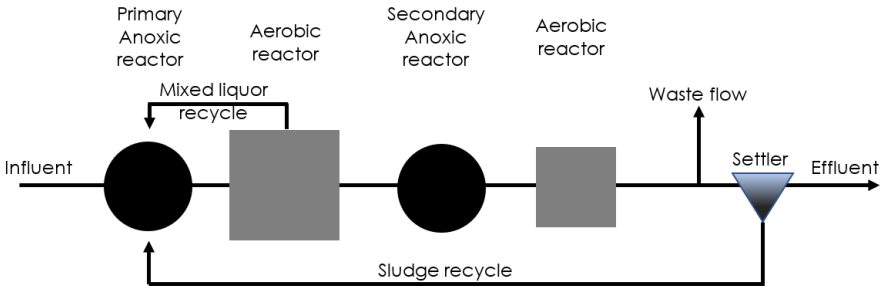


Figure 4. 4-stage Bardenpho system

2.2. Filtration technology

2.2.1. Principles of filtration

Filtration is the process of passing wastewater through a filter media to remove particulates and other impurities. In contrast to physical straining filtration can remove particles considerably smaller than the filter nominal pore size (EPA, 1995). Filtration relies on numerous mechanisms to achieve high removal such as sieve effect or cake filtration, adsorption, absorption and straining.

Depending on separation mechanisms, two categories of filtration are cross-flow and dead-end filtration (Figure 5).

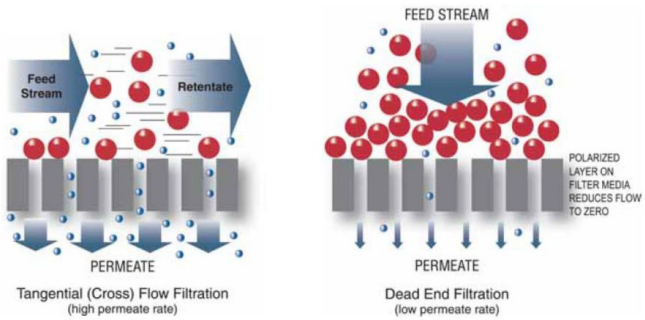


Figure 5. Tangential versus dead end filtration
(adapted from Ballew *et al.* 2002)

Filtration materials are depth filters, screen filters and membranes. Depth filters and screen filters are for dead-end filtration, while membranes are for tangential flow filtration (Lee *et al.*, 2011).

Deep bed filtration uses packed beds of particles between 300 and 5000 µm in diameter with bed height between 0.5 to 3 m. Various media may be used as bed, including sand, gravel, anthracite, activated carbon and garnet. Deposition of suspended solids takes place within the bed by a variety of particle adhesion and collection mechanisms, such as sedimentation, inertia, van der Waals, diffusion, electrostatic attraction and repulsion (Ballew *et al.,* 2002). Depth filters do not have a precise pore size or structure and thus are not absolute. This means that particles with wider range in size may permeate through the filter. Components

that are larger than the apertures of the filter remain on the filter surface. In the case of smaller particles, random entrapment and adsorption of matter occur within the structure of the media (Keir *et al.*, 2009). Depth filters are manufactured from fibrous materials, woven or nonwoven polymeric or inorganic materials.

Particles remain directly on the surface of the screen filters and mesh filters. Pores are precise, and only particles with diameter below the pore size will permeate the filter (Ballew *et al.,* 2002). However, when solids start to bridge over the entrance of the filter medium pores, a cake layer forms, allowing cake filtration or dynamic filtration (Figure 6). Cake filtration allows high removal efficiencies because the thick layer (Perry & Green, 2007) traps smaller particles. The filter cake forms cleaned and reforms throughout the filtration process (Seo *et al.*, 2007).

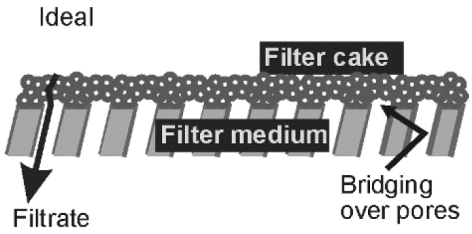


Figure 6. Cake filtration
(Adapted from Holdich, 2014)

Membrane filters are thin sheets often manufactured from polymeric materials but also from other materials. Due to manufacturing techniques, they sustain a well-defined flow rate and have a defined pore size, pore structure, pore density, bubble point and tensile strength. Membrane filters mainly function by trapping particles on its surface, with some entrapment into the membrane pores [Ballew *et al.*, 2002] Typical membrane configurations are tubular, flat sheet, hollow fiber, spiral wound and vibrating membrane systems (Lee *et al.*, 2011).

Common filtration technologies found in wastewater treatment are Salsnes Filter (Trojan Technologies), Eco-Mat RBF (Blue Water Technologies) and Hydrotech Belt filter (Veolia Technologies) (Figure 7).

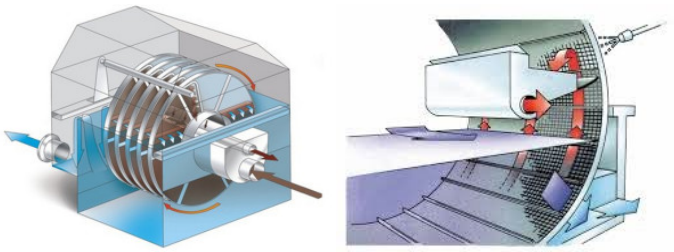


Figure 7. Disc and Drum filter (Hydrotech)

2.2.2 Parameters affecting filtration

Temperature is an important parameter affecting filtration. Increasing temperature reduces viscosity of the feed and leads to reduced filtration time (Lee *et al.,* 2011).

Another important parameter is the specific cake resistance of deposited solids. Cake resistance reduces by filter aids. These filter aids increase the porosity of the cake and thus increase the flow rate through the cake (Stanbury *et al.*, 2016).

Transmembrane pressure is important in dead-end filtration as it the driving force forcing filtrate to pass the filter. The inlet, outlet and filtrate pressure valves control the driving force. Cross-flow velocity needs to be sufficiently high to provide enough shear stress on the membrane surface to prevent settling (Lee *et al.*, 2011).

2.3. Salsnes Filter technology

2.3.1 Type of filters

Salsnes Filter technology, a rotating belt filter (RBF), is a compact unit for the mechanical separation of particulate materials, thickening and dewatering (optional) of sludge from wastewater. Two series of filtration units are commercially available. The SF models (SF1000, SF2000, SF4000, and SF6000) are suitable for wastewater flowing and arriving in pipes, and the SFK models (SFK200, SFK400, and SFK600) for wastewater arriving by channels (Figure 8). Each model has a submerged sieve cloth area varying from 0.25 to 2.2 m^2 and mesh sizes up to 1 mm. The hydraulic loadings of the different units vary from 31 to 394 m^3/h.

Figure 8. Models of Salsnes rotating belt sieves

A bench-scale unit is also available for laboratory purposes to simulate performance of a commercial unit. It is composed of the four components filter cloth, a filter cloth holder, a bench table and a transparent PVC tube for water reservoir. A screw coupling to keep the top and the bottom together during testing and no O-ring is required to seal the connection between the pieces as each filter cloth has a silicone seal around the circumference. Figure 9 shows a sketch diagram of the bench-scale filter unit.

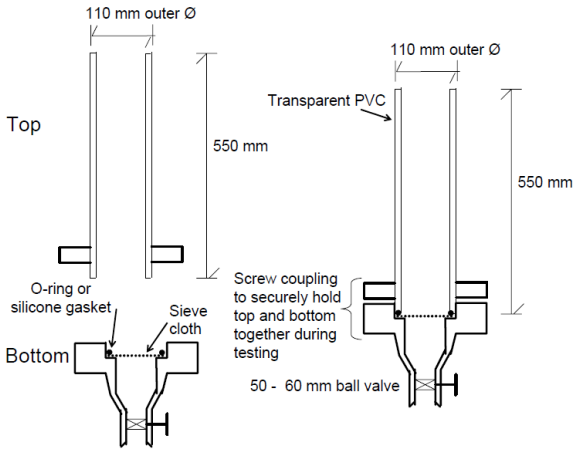


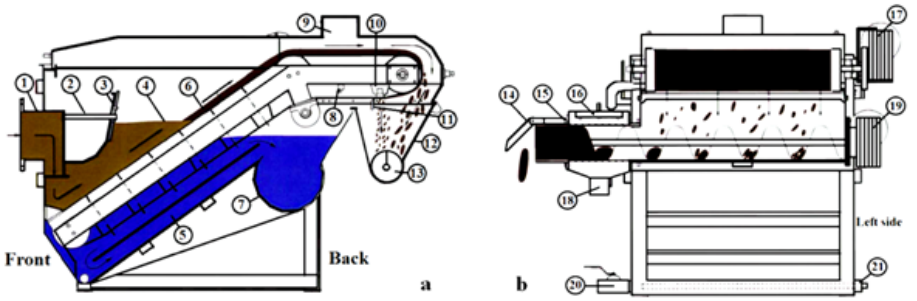
Figure 9. Sketch of a bench-scale Salsnes Filter

2.3.2 Operation principle

During filtration as observed in Figure 10, wastewater enters the inlet tube and filters through a continuously looped wire cloth removing suspended solids. PLC receives the level of the incoming wastewater. This enables setting the correct speed of the wire cloth to achieve the optimum performance at variable flow rates and variable influent suspended solid concentrations. RBF operates with a fixed belt speed and a variable water level, or a fixed water level and a variable belt speed (Rusten *et al.*, 2016). The latter is most common, where the belt speed depends on the water flow and the TSS concentration in the wastewater. The filter remains immobile if the water level in the inlet chamber is low. Particles start to accumulate on the surface of the filter forming a filter cake, and consequently, the water level increases and the pressure transmitter automatically engages the motor that move the wire cloth. If the water level keeps increasing while the wire cloth is moving, the speed automatically increases. If the water level drops below a preset limit, the motor stops until the water level rises again (Salsnes Filter, 2013; Nussbaum, 2006; Rusten, 2005b).

The filtered water flows from the back of the wire cloth and exits the system from the outlet tube. The rotating belt transports the sludge to the air-knife cleaning device where compressed air blows the sludge into

the sludge compartment. An auger press further dewaters the sludge to 40 % dry matter (Love & Lowe, 2015; Nussbaum, 2006). The dry solid content can be adjusted by regulating the tension of the spring-loaded lid. The wire cloth is flushed with hot water two to six times a day to remove fat, oil and grease.



1 Inlet	9 Ventilation	17 Gear/motor for wire cloth
2 Overflow	10 Air-knife cleaning device	18 Reject from press cylinder
3 Level Indicator	11 Rubber scraper	19 Gear/motor for screw press
4 Wastewater	12 Sludge compartment	20 Drain valve for settled waste
5 Filtered water	13 Screw press	21 Cold water pipe for settled waste removal
6 Wire cloth	14 Spring-loaded lid	
7 Outlet	15 Press cylinder	
8 Hot water nozzles	16 Hot water nozzles for cleaning press cylinder	

Figure 10. Fine mesh rotating belt filter

2.3.3 Applications

Since its introduction in 1992, Salsnes RBFs served as primary treatment or the only treatment of municipal and industrial wastewater prior to discharge (aquaculture, tanneries, food processing, and slaughterhouses) (Rusten, 2002). More than 800 systems are in service worldwide with hydraulic loading from 15 m^3/h to 56000 m^3/h. The system operates on the principle of thin cake filtration, allowing the removal of particulates up to three times smaller than the nominal pore size of the filter cloth (Chakraborty, 2015). The most common filter size used in wastewater treatment is 350 µm, operated with filter mat. There is practically no performance difference with different mesh sizes of the filter cloths about % TSS removal and filtration rate once a filter mat was formed on the filters (Rusten & Lundar, 2006). At least 20 % of the influent

suspended solids should be larger than 350 µm and have a ratio above 0.4 filtered COD to total COD (Rusten & Ødegaard, 2006). If these conditions are not met, a filter with lower openings or chemical dosing could be applied upstream the RBF. The system is automated.

The modular design of the Salsnes Filter unit allows installation configurations to serve any capacity. Each module can have up to 12 filters (six per side) and each side performs jointly, sharing components such as the blower for the air-knife cleaning system. The Agua Prieta WWTP in Guadalajara (Mexico) has the largest Salsnes Filter installation, treating 55 200 m^3/h of wastewater using 980 m^2 of land compared to 20 000 m^2 with conventional primary sedimentation. Tomasjord WWTP (Norway) uses only 150 m^2 ground off land compared with 2,000 m^2 for clarifiers.

Besides the reduced footprints, RBFs offer very high removal efficiencies for TSS and COD. Average TSS removal of 60 % to 90 % and COD removal of 30 % - 80 % resulted at Tiendeholmen and Breivika WWTPs (Norway) (Rusten, 2002; Rusten, 2000). A study conducted at the City of Enderby WWTP (British Columbia, Canada) showed a reduction of sludge production by 87 % and with higher VS content indicating higher methane potential for the anaerobic digesters (Paulsrud *et al.*, 2014; Salsnes Filter, 2013).

2.4. Overview of the primary treatment technologies

Several technologies are in use as primary treatment

- septic tank (Fayza & Basem, 2013; Meuler *et al.*, 2008; EPA, 2000),
- sedimentation tank (Lema & Suarez, 2017; Metcalf & Eddy, 2003; Ratnaweera *et al.*, 1994),
- dissolved air flotation (Bickerton, 2012, Edzwald, 2010; Al-Shamrani *et al.*, 2002; Lundh, 2000),
- drum and disc filters (Lema & Suarez, 2017; Libhaber & Jaramillo, 2012), and
- rotating belt filters (Rusten *et al.*, 2016; Franchi & Santoro, 2015; Nusbaum *et al.*, 2006, Rusten & Lundar 2006, Rusten & Ødegaard, 2006).

Properly designed and operated primary sedimentation tanks typically remove 40 – 70 % TSS, 25 – 35 % COD, 25 – 40 % BOD_5, and up to 10 % TN (Lema & Suarez, 2017; Warren, 2009; WEF, 2005; Metcalf & Eddy, 2003; Wahlberg et al., 1997). The efficiency of primary sedimentation tank depends on several factors including characteristics of the suspended solids, surface loading[1] of the sedimentation tank, hydraulic conditions and sludge withdrawal (Srinivas, 2008; von Sperling, 2007; Gray, 2004; EPA, 1993; Ødegaard, 1990).

DAF operated without chemicals removes about 10 – 60 % COD, 25 – 70 % BOD_5 and 50 – 85 % TSS. Performance depends on several factors such as the overflow rate of wastewater, the nature of pollutants, the extent of the treatment required, the particle size, the bubble size and the ratio between air and particles (Metcalf & Eddy, 2003, Bickerton, 2012; Telang, 1996). Drum and disc filters remove 40 – 60 % TSS, up to 60 % COD, 15 – 30 % BOD_5, 5 – 10 % TN and 0 – 40 % TP (Lema & Suarez, 2017; Libhaber & Jaramillo, 2012). The average removal efficiencies of RBF are 25 – 60 % TSS, 15 – 40 % COD, 15 – 30 % BOD_5, and less than 10 % TN (Franchi & Santoro, 2015; Nussbaum *et al.*, 2006; Rusten & Lundar, 2006; Rusten &

[1] $A = Q/V_S$ (A: surface area; Q: flow and V_s: overflow rate or hydraulic surface loading rate)

Ødegaard, 2006). The removal efficiencies of filters depend highly on the mesh size, the filtration rate and the characteristics of the influent wastewater.

The removal performance of primary treatment improves without consuming more space by adding chemicals upstream of the primary treatment unit. The chemicals precipitate colloidal and dissolved matter and increase the size/density of the particles. Higher removals results with chemically enhanced primary treatment, compared to the process operated without chemicals reported in Table 1.

Table 1. Removal performance of different primary treatment technologies, without and with chemical dosing.

Technology	Chemical dosing	TSS (%)	COD (%)	BOD_5 (%)	TN (%)	TP (%)	Sludge solids (TS) (kg/m3)	References
Primary sedimentation tank	-	40-70	25-35	25-40	< 10	15-20	10-40	Lema & Suarez (2017), DWA (2008), Metcalf & Eddy (2003)
	+	80-90	55-75	40-80	10-20	60-80	20-70	
Drum and disc filter	-	40-60	< 60	15-30	5-10	< 40	5-15	Lema & Suarez (2017), Väänänen *et al.* (2016), Libhaber & Jaramillo (2012),
	+	80-90	< 80	50-60	5-10	50-90	< 25	
Rotating belt filters	-	40-60	15-40	15-30	< 10	15-20	30-200	Lema & Suarez (2017), Franchi & Santoro (2015), Rusten & Ødegaard (2006)
	+	65-90	45	40	-	15-20	30-200	
Dissolved air flotation	-	30-80	70	50	45	50	-	Kim (2015), Johnson (2014), Ødegaard (1995)
	+	50-95	30-85	-	-	90-95	< 5	

2.5. The impact of primary treatment on the overall treatment process

Primary treatment plays an important role for the energy balance at wastewater treatment plants because it affects the amount of organic matter to the downstream biological process, especially BNR (Bixio *et al.*, 2000). Nitrogen compounds from wastewater are removed by a combination of nitrification and denitrification (Wang & Yang 2004) and these processes need enough organic matter (Tas *et al.* 2009). A high degree of COD removal in primary treatment will limit nitrogen removal. On the other hand, primary sludge is energy-rich and a source of substrate for biogas generation.

WERF has determined the typical energy demand for various levels of wastewater treatment necessary to meet effluent requirements and further model the net energy recovery potential available in different process configurations (Tarallo, 2014) (Tables 2 and 3).

Table 2. Typical energy demand for various wastewater treatments

Technology	Energy demand (kWh/m^3)
BOD removal only	0.35
Nitrification	0.45
Biological nutrient removal (BNR)	0.50
Enhanced nutrient removal (ENR)	0.52
Membrane bioreactor (MBR)	1.50

Adapted from Tarallo (2014)

Table 3. Modeled net energy recovery potential

Modeled facility	Net energy recovery
BOD removal + Enhanced primary + AD + CHP	139 %
Nitrification + Enhanced primary + AD + CHP	110 %
BNR + Enhanced primary + AD + CHP	61 %
ENR + Enhanced primary + AD + CHP	49 %

AD: Anaerobic digestion; CHP: Coupled heat power plant

The effect of primary treatment on sludge production, aeration demand and biogas production were evaluated using data from SNJ wastewater

treatment plant (IVAR IKS, Norway). The characteristics of the influent and some operational data are in Table 4.

Table 4. Input data

Parameter	Values
Input flow	100 000 m^3/d
TSS	200 mg/L
VSS	170 mg/L
Total COD	300 mg/L
Biodegradable COD (COD_b)	230 mg/L
Biodegradable particulate COD (COD_{bx})	190 mg/L
COD_{bx}/ COD_b	1.12 mg/L
SRT	5 d
kgO_2/kgCOD	0.556
COD/VSS	1.40 gCOD/gVSS
Degradation-Anaerobic digestion	55 %
Energy in methane	0.35 m^3/kgCOD; 10 kWh/m^3
Energy for aeration	1.5 kgO_2/kWh
Energy consumption **Filter plant** **Primary settling tank**	 1200 kWh/d 600 kWh/d

- ***COD removal as a function of TSS removal***

Primary treatment technologies typically remove up to 50 % of SS without added chemicals. For higher removal rates coagulants must be supplied upstream of the primary treatment unit. TSS removal from 0 % to 90 % is considered and evaluated.

As observed in Figure 11, the amount of COD removed during the primary treatment is proportional to the TSS removed. A major part of the influent COD consists as particulates embedded with the suspended solids. Influent COD usually consists of 40 % - 60 % particulate, 10 % - 30 % soluble biodegradable COD, and about 30 % of inert material (Drewnowski & Makinia, 2013; Orhon *et al.*, 1997). In this example, the particulate COD accounts for about 75 % and soluble COD 25 % of total COD.

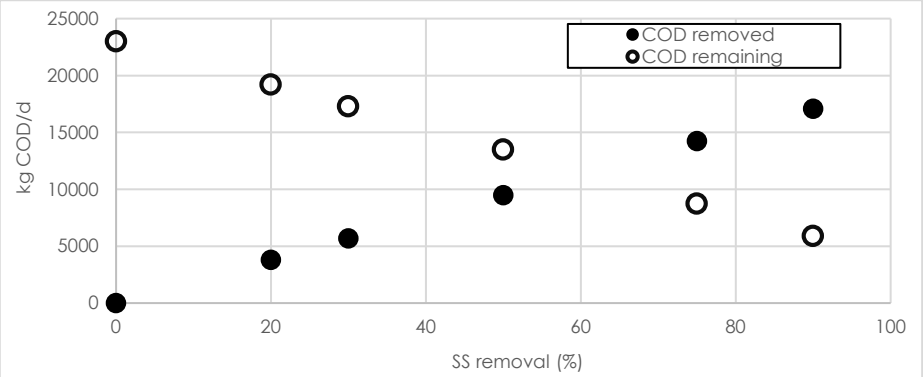


Figure 11. COD removal as a function of TSS removal

Sludge production is also proportional to TSS removal (Figure 12). The use of chemicals does not affect the sludge production since they do not form any extra sludge, just the SS attached. SS removal increases production of primary sludge but on the contrary decreases the production of biological sludge. The overall sludge production will increase when the proportion of primary sludge increases. There is no degradation of the primary sludge, while COD entering the biological treatment oxidizes and the remaining sludge production is lower. This effect is probably more significant when primary treatment is by filtration with sludge generated and pumped to the anaerobic digester within a short time. For settling tanks, this takes much longer time and some COD loss is expected.

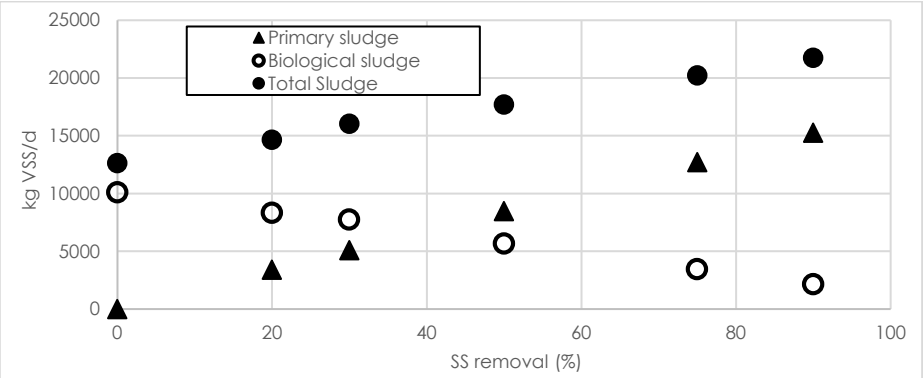


Figure 12. Production of primary and biological sludge as a function of TSS removal

Production of methane is proportional to sludge production (Figure 13). The higher the fraction of primary sludge the more methane produced from sludge. This is because primary sludge is more biodegradable than secondary sludge in an anaerobic digestion process. Some of the biodegradable COD oxidizes in the biological treatment, thus leaving a lower part of the COD available for methane production.

Energy demand related to the aeration in the biological process is inversely proportional to SS removed. The reduced organic load to the biological process results in a reduction of energy requirements for aeration. Gori *et al.* (2013) observed a reduction in energy demand for biological oxidation of 4 to 11 % by reducing the organic load to the biological process.

The difference in energy consumption and energy generation as a function of SS removal represents the net energy yield of the plant. Figure 13 illustrates the significant benefits of SS, and thus COD removal, in primary treatment on the energy balance of the plant. The greater the primary sludge entering the anaerobic digester the greater the energy recovered from the anaerobic digestion.

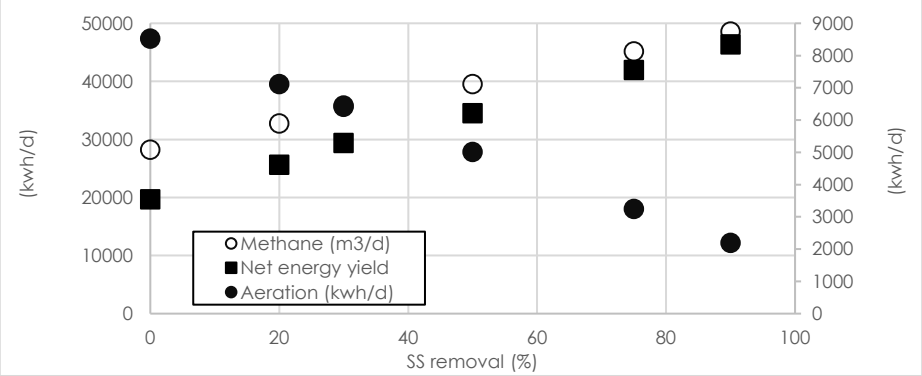


Figure 13. Energy production from methane, energy demand for aeration and net energy yield as a function of SS removal

In sum, solids separation prior to BNR, benefits both the biological process and the sludge treatment units up to a certain level. The amounts of organic material entering the biological process decreases, resulting in a lower oxygen demand. On the other hand, at a certain level available COD will limit the biological removal of N and P.

Therefore, for a specific wastewater and plant configuration there will be an optimum level for SS removal in primary treatment that will support enough nutrient removal and biogas production.

Chapter 3

Materials and methods

3.1. Sampling and tests sites

The experiments at Aquateam's laboratory and Nordre Follo WWTP used wastewater and sludge from Bekkelaget (BRA) or Nordre Follo (NFR) WWTPs.

Grab samples of influent wastewater were collected after the grit removal section (screens) at both WWTPs. Return activated sludge (RAS) was sampled from the recycle line at BRA, while biofilm carriers were sampled from the anoxic reactor at NFR for the anoxic batch tests and from the aerobic reactor for the aerobic batch tests. All samples were collected the day before the experiments and stored in a cooler at 4 °C.

BRA WWTP is the second largest sewage treatment plant in Norway. The plant is located in a cave at Ekeberg hill about 6.7 km Southeast of Oslo. BRA WWTP treats wastewater from the eastern part of Oslo and the neighboring municipalities Oppegård and Nittedal. The plant treats about 50 million m^3 of wastewater per year, 35 to 40 % of wastewater production in Oslo, or 290,000 person equivalents (p.e.). The plant consists of primary settling and activated sludge for COD and N removal and simultaneous precipitation for P removal. The tunnel system to BRA equals a volume of 35000 m3 and the plant can handle a maximum flow of 4000 L/s. Wastewater over 6000 L/s overflows directly to the bay via Alna river, but it rarely occurs. The sludge treatment consists of thickening, anaerobic digestion and dewatering. After upgrading, the biogas becomes fuel for busses in the Oslo region.

NFR WWTP is located about 30 km south of Oslo. The plant treats wastewater from the municipalities Ski, Ås and part of Oppegård using moving bed biofilm reactor (MBBR) technology. The plant has a capacity of 70,000 p.e. The first plant built in 1972, provided only primary

treatment prior to discharge at Bunnefjord, which is a narrow part of the Oslofjord, at 50 m depth and 350 m from shore. The plant was upgraded twice; in 1982 by adding chemical precipitation and flotation, and the second upgrade in 1997 added secondary biological treatment to meet stringent effluent requirements for COD, N and P removal. The wastewater is flowing through a 3 mm bar-screen to remove large particles followed by sand and fat removal. Thereafter wastewater flows through primary sedimentation units where the thickened sludge is further treated. Pumped wastewater enters the MBBRs systems. Two trains of MBBR at NFR combine pre- and post- denitrification processes for nitrogen and organic removal. Phosphorus removal is by chemical precipitation. The two trains of MBBR are composed of seven (7) reactors each. An added external carbon source to the sixth reactor is for nitrogen removal. Pumped MBBR effluent enters a flotation unit, where added polymers increase particle size, and further pumped for biogas production. (Telkamp, 2006).

3.2. Feed water preparation

3.2.1. Batch laboratory tests

The tests performed in the laboratory were for determining the specific denitrification rates of the wastewater as a function of TSS removal. Wastewater for the tests were prepared with filter openings from 1.2 µm to 150 µm (Figures 14 and 15). Separation in the bench-scale filter without mat formation (Rusten & Lundar, 2006) allowed removal of particulates above a certain size.

Figure 14. Vacuum filters with a Büchner funnel (Left) and a 47-mm funnel (Right)

Figure 15. Bench-scale Salsnes Filter unit with microscopic observation[2] of different filters at 10X magnification (From top-left to bottom-right: 150 µm, 90 µm, 55 µm, 33 µm, 18 µm)

[2]The filter cloths were photographed with a Nikon Eclipse 50i optical stage microscope with attached Nikon DS-Vi1 digital camera (Nikon, Germany)

Aluminum sulfate (alum) pre-flocculated some samples before separation with GF/C filter to remove both particulates and colloids from the influent wastewater. Doses of alum administrated at 12 mg Al/L, rapid mix at 150 rpm for 1 min flocculated at 20 rpm for 15 min (IKA RH basic 2 magnetic stirrer, Staufen, Germany). Since alum has a tendency to decrease pH, added aliquots of 1 M of NaOH secured the set-point of pH 7.

3.2.2. *Pilot scale testing*

A commercial Salsnes model SF1000 with 33 mm filter pores secured primary treatment during the pilot scale testing. The control unit without primary treatment had a 2 mm-screen to protect downstream equipment from clogging, since the tubes used in the peristaltic pumps for feed and RAS recycling had an internal diameter of only 4 mm. Similar to the previous tests, SF 1000 operated without filter mat. This exposed the filter at a very low influent flow (5 – 6 m^3/h) and with a relatively high belt speed (5 – 7 cm/s).

Two large tanks at 500 L each are equipped with mechanical mixers to prevent settling of particulate matter, served to store the feed for the two trains of BNR at ambient temperature (~ 17 °C). Preparation of new batches of wastewater three times a week in the afternoon secured influent wastewater composition at the daily average.

Ideally, two commercial SF 1000 ought to prepare the feed of the two trains at the same time so that the composition of influent wastewater would be identical before filtration. Unfortunately, only one unit was available at the site. Consequently, the preparation of feed during the pilot-tests with MBBR and MBR were separate at intervals of 19 to 40 minutes.

3.3. Analytical procedures

3.3.1. Temperature, pH and dissolved oxygen

Recorded environmental parameters such as temperature, dissolved oxygen (DO) and pH employed a multi-parameter model 3420 (WTW, Weilheim, Germany). It had a galvanic oxygen sensor (CellOx 325) with a range of 0 to 50 mg/L DO, and a pH-calibrated electrode (SenTix). CellOx recorded temperatures.

3.3.2. Total suspended solids analysis

Total suspended solids (TSS) was determined with GF/C filters at 1.2 µm, and volatile suspended solids (VSS) after burning at 550 °C, according to Standard Methods 2540 D and 2540 E, respectively (APHA, 2005).

3.3.3. Determination of biomass on biofilm carriers

Biomass on biofilm carriers were analyzed according to Aquateam AS standard procedure. The method consists of collecting fifteen biofilm carriers, dry at 105 °C overnight until constant weight. The dried carriers were cooled, and their mass recorded. Dried carriers soaked in concentrated domestic sodium hypochlorite solution for about 30 minutes. Again, the carriers washed and scrubbed in warm water for removal of traces of biomass dried at 105 °C overnight until constant weight. Recorded weight of the clean carriers and average biomass per carrier were calculated.

3.3.4. COD and Nutrient analysis

COD, nitrogen and phosphorus concentrations of wastewater were determined with Dr. Lange cuvette test kits (Hach-Lange, 2008), a Thermostat LT 200 to incubate samples when necessary and a DR 5000 UV-Vis Spectrophotometer (Hach Lange, Germany) shown in Figure 16.

T-18 Ultra-Turrax (IKA, Germany) homogenized the samples for one minute at 16,000 rpm before analyzing for soluble COD (sCOD), ammonium (NH_4-N), nitrate (NO_3-N), nitrite (NO_2-N) and orthophosphate (PO_4-P). The samples were filtered with Whatman GF/C fiber glass filter at 1.2 µm pore size (GE Healthcare, Buckinghamshire, UK) mounted on a Pall easy pressure syringe filter holder (Hach, Loveland, USA).

Figure 16. a) Dr. Lange cuvette test kits, b) Pall syringe filter holder, c) Dr. Lange thermostat LT 200, d) Dr. Lange spectrophotometer DR 5000

3.3.5. Sludge volume index (SVI)

The volume of sludge in milliliters occupied by 1 g of activated sludge defines the sludge volume index (WEF, 1994). SVI assesses the settling quality of activated sludge and evaluates the performance of settling tanks (NS, 2006). The test consists of filling a one-liter graduated cylinder with mixed liquor from the reactor and settling for 30 minutes. The SVI is the ratio of settled volume and the corresponding sample MLSS concentration.

$$\text{SVI (mL/g)} = \frac{1000 \times \text{Settled volume (mL)}}{\text{MLSS (mg/L)}}$$

3.3.6. Nitrate utilization rate test (NUR)

The anoxic batch tests for determining the nitrate utilization rate (NUR) in this study was adopted from the protocol elaborated by Gu and Onnis-Hayden (2010). Prior to any analysis and testing, the samples were stored in the cooler and brought to room temperature of 20 °C for 2 hours. Aeration of mixed liquor for a couple of hours removed remaining COD. Mixed liquor added a sample of the prepared wastewater with 20 mL of potassium nitrate solution ensured nitrate in excess. Purged with nitrogen gas ensured anaerobic conditions of the mixture. Analysis over time determined nitrate, nitrite, ammonium and soluble COD.

Calculating NUR by

$$NUR=\frac{\Delta NO_3}{time}\ (mgN/l\cdot h)$$

$$NUR=\frac{\Delta NO_3}{(time\cdot X_H)}\ (mgN/mgX_H\cdot d)$$

3.3.7. Nitrification rate test

Measuring nitrification rates were after aeration of the bioreactor. The drop of ammonium over time was determined by sampling at 30-min intervals for at least 3 hours, and analyzed for ammonia nitrogen, nitrate nitrogen, nitrite nitrogen and soluble COD. Calculations of specific nitrification rates were from the slope of the ammonium profile, the ammonia utilization rate, AUR (Melcer et al., 2003).

$$AUR=\frac{\Delta NH_4}{time}\ (mgN/l\cdot h)$$

$$AUR=\frac{\Delta NH_4}{(time\cdot X_A)}\ (mgN/mgX_A\cdot d)$$

3.3.8. Methane production and energy yield from methane

$$\text{Methane production} = P_X \times \frac{VSS}{TSS} \times \frac{COD}{VSS} \times \frac{CH_4}{COD} \times C_{AD}$$

With
Px: sludge production (gTSS/d)
VSS/TSS= 0.8 gVSS/gTSS
COD/VSS = 1.45 gCOD/gVSS
CH_4/COD= 0.35 L CH_4/gCOD
Assuming 50 % conversion rate in the anaerobic digester C_{AD} = 0.5

$$\text{Energy production} = \text{Methane production} \times E_{CH4}$$

E_{CH4}: energy yield from methane (9.95 Wh/L CH_4)

3.4. Steady state modelling of activated sludge biological N removal

Mathematical models represent complex processes of biomass activity, leading to complete descriptions of carbon and nitrogen removal in the activated sludge process (Henze et al., 1987). Models are to optimize and upgrade existing plants, meet effluent criteria, reduce cost of operation, to design reuse of effluent wastewater, to develop operation strategies, and to design new treatment plants (Vanrolleghem et al., 2003).

The steady state model of activated sludge biological N removal is based on the Activated Sludge Model no. 1 (ASM1) developed by IWA and the research on biological N removal at University of Cape Town (WRC manual 1983).

The basis of the model is the specification of the COD and N into different fractions and specification of the sludge into biomass and inert fractions. The most important division is between biodegradable and un-biodegradable fractions and between soluble and particulate fractions. The biodegradable fractions are the basis for growth of biomass, which is the basis for all the biological rates, and thus form a unified approach for determining the biological rates. The un-biodegradable fractions do not participate in the biological reactions but the particulate fractions accumulate as part of the sludge (VSS) while the soluble fractions pass through the plant and ends up in the effluent.

3.4.1. Wastewater characterization

a. COD fractionation

The advantage of selecting COD as the parameter for quantifying the strength of organic material in the influent, as opposed to BOD or TOC, is that it provides a consistent basis for the description of the activated sludge process. The total influent COD subdivides into biodegradable COD and un-biodegradable COD (Figure 17). The biodegradable material divides into a readily biodegradable portion (RBCOD, SS) and

a slowly biodegradable portion (SBCOD, XS). The RBCOD consists of fermentable rbCOD and short chain volatile fatty acids (SCFA) that can be absorbed directly by microorganisms. The slowly biodegradable portion consists of particulate/colloidal material and complex organic molecules that goes through hydrolysis prior to utilization.

The un-biodegradable material divides into soluble un-biodegradable portion (SI) and a particulate un-biodegradable portion (XI). Both components are unaffected by the biological activity in the system. Therefore, the soluble un-biodegradable COD leaves the system at a concentration similar to that in the influent, and the particulate un-biodegradable enmeshes in the sludge mass and accumulates in the system.

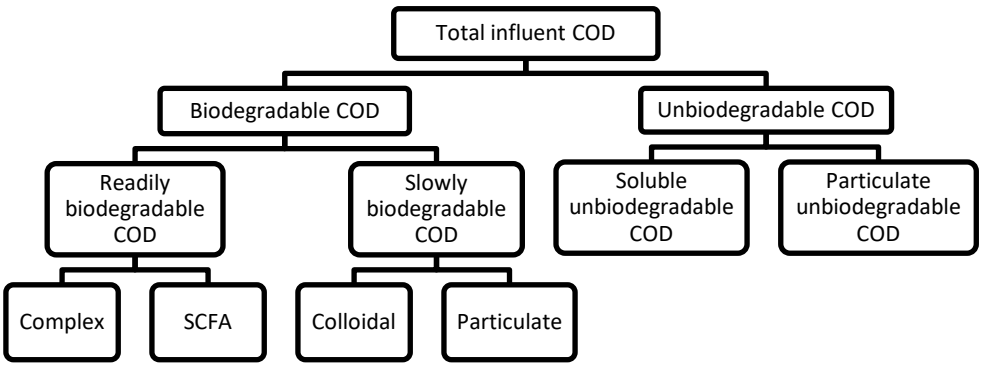


Figure 17. Fractionation of total influent COD

b. Nitrogen fractionation

The influent nitrogen divides into free and saline ammonia (S_{NH}) and organically bound nitrogen (Figure 18). The latter further divides into biodegradable and un-biodegradable fractions, and respective soluble and particulate sub-fractions. The un-biodegradable fractions are unaffected by the biological process. The soluble un-biodegradable organic nitrogen leaves the system at a concentration equal to that in the influent, while particulate un-biodegradable portion is associated with the un-biodegradable particulate COD and therefore leaves the system via waste sludge. The biodegradable particulate organic

nitrogen assumes hydrolyzed to soluble organic nitrogen, as a parallel to hydrolysis of particulate COD. The biodegradable soluble organic nitrogen converts to ammonia in ammonification by the heterotrophic biomass.

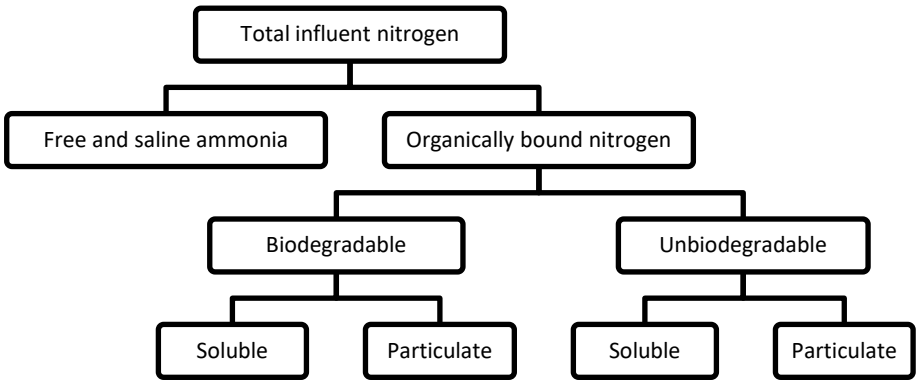


Figure 18. Fractionation of total N

3.4.2. Sludge concentration

The growth of biomass is by utilization of the biodegradable COD (COD_b). After SRT exceeds 3 days at 20 °C, utilization is complete (5 days at 10 °C). Thus, knowing the influent COD_b biomass, (X_H) results from the mass balance of COD_b and accounting for biomass loss in decay.

$$X_H = \frac{Q \times COD_b \times Y_H \times SRT}{(1 + k_{dH} \times SRT) \times V} \qquad (mgVSS/l)$$

Where:

X_H: biomass (mgVSS/l)
Q: influent flow (m^3/d)
COD_b: biodegradable COD (mgCOD/l)
Y_H: Heterotrophic yield coefficient (0.45 mgVSS/mgCOD)
SRT: sludge retention time (d)
k_{dH}: decay rate (d^{-1})
V: reactor volume (m^3)

Decay of biomass release substrate, and utilized by the remaining biomass. A fraction of the biomass is not biodegradable (f_d) and accumulates in the sludge and becomes a part of the inactive sludge fraction (endogenous residue; X_E). The biodegradable fraction (1-f_d) of the dead biomass oxidizes and appear as oxygen utilization rate (OUR_E).

$$X_E = k_{dH} \times f_d \times X_H \times SRT \qquad (mgVSS/l)$$

Where:

X_E: endogenous residue (mgVSS/l)
f_d: unbiodegradable fraction

The unbiodegradable COD does not undergo any biological reactions. The soluble unbiodegradable COD (COD_{us}) assumes not to interact with the sludge and ends up in the effluent. So soluble COD in the effluent mainly consists of COD_{us} when SRT exceeds 3 days with all COD_b utilized. The particulate un-biodegradable COD (COD_{up}) in the influent enmeshes and accumulates in the sludge and form a part of the VSS, (X_I). The concentration of X_I in the sludge is a function of the influent COD_{up} concentration, the SRT and HRT.

$$X_I = \frac{COD_{up} \times Q \times SRT}{V \times f_{cv}} = \frac{COD_{up} \times SRT}{HRT \times f_{cv}} \qquad (mgVSS/l)$$

Where:

X_I: inert (mgVSS/l)
Q: influent flow (m^3/d)
COD_b: biodegradable COD (mgCOD/l)

The organic fraction of sludge in the activated sludge consist of biomass, endogenous residue and un-biodegradable VSS from the influent:

$$MLVSS = X_H + X_E + X_I \qquad (mgVSS/l)$$

Where:

X_H: biomass (mgVSS/l)
X_E: endogenous residue (mgVSS/L)
X_I: Inert (mgVSS/l)
MLVSS: Mixed liquor volatile suspended solids (mgVSS/l)

Total sludge concentration is by the VSS/TSS

$$MLSS=\frac{MLVSS}{VSS/TSS} \quad (mgTSS/l)$$

Where:

MLSS: mixed liquor suspended solids (mgTSS/l)

3.4.3. Oxygen consumption

The corresponding oxygen consumption appears as the fraction of the substrate not incorporated in the biomass (1 - Y_H) and oxidation of the biodegradable fraction of the dead biomass (1 - f_d).

$$MO_G=Q\times(1-Y_H.f_{cv})\times COD_b \quad (gO/d)$$

$$MO_E=(1-f_d)\times k_d\times X_H\times f_{cv}\times V \quad (gO/d)$$

MO_G: Oxygen consumption in degradation of COD (gO/d)
MO_E: Oxygen consumption in degradation of dead biomass (endogenous respiration) (gO/d)

3.4.4. Biological N removal

The denitrification rate is a function of the COD source and the active biomass (X_H).

$$DN=K\times X_H\times HRT \quad (mgN/l)$$

DN: Nitrate reduction (mgN/L)
K: Specific denitrification rate (mgN/mgX_H*d)

There are thus three denitrification rates based on readily biodegradable COD (K_1), slowly biodegradable COD (K_2) and endogenous or the decay process (K_3). Typical values for the denitrification rates are:

RBCOD:	K_1 = 0.72 mgN/mgX_H*d	θ = 1.2
SBCOD:	K_2 = 0.101 mgN/mgX_H*d	θ = 1.08
Endogenous:	K_3 = 0.072 mgN/mgX_H*d	θ = 1.03

In systems where the wastewater provides COD for denitrification, rates will be according to K_1 and K_2, in effect a primary anoxic reactor. In systems where denitrification relies on endogenous substrate, the denitrification rate will be according to K_3, which refers to as a secondary anoxic reactor.

Nitrification occurs only under aerobic conditions, and the fraction of the reactor that is anoxic and not aerated is termed f_x, a value between 0 and 1. For practical purposes, f_x has a maximum of 0.50. The unaerated fraction will reduce the nitrification rate since no nitrification. The aerobic fraction is thus $(1 - f_x)$ and is incorporated in the equation for effluent ammonia:

$$N_A = \frac{K_N \cdot (1/SRT + k_{dA})}{\mu_{max\,A} \cdot (1-f_x) - (1/SRT + k_{dA})}$$

N_A: Effluent ammonia (mgN/L)
k_{dA}: Decay rate nitrifiers (d^{-1})
μ_{maxA}: Maximum specific growth rate nitrifiers (d^{-1})
f_x: Anoxic fraction (-)

The amount ammonia nitrified will thus be the difference between influent and effluent ammonia, ammonia generated in degradation of organic nitrogen, minus the amount assimilated in the biomass.

The removal of nitrogen is via succession of aerobic and anoxic conditions and depending on the combination, two main configurations a. and b. result.

a. Wuhrman system: Aerobic – anoxic sequence

COD and ammonia are oxidised in the aerobic reactor. Denitrification takes place with COD from decaying cells only, according to K3, since more or less all influent biodegradable COD oxidizes in the aerobic

reactor. Thus, the rate of decay determines rate of COD supply and finally the rate of denitrification.

Denitrification potential of the anoxic reactor of the Wuhrman system (secondary anoxic reactor) is

$$D_{P3} = K_3 \cdot X_H \cdot R_{hn}$$

$$D_{P1} = \frac{K_3 \cdot f_x \cdot \Delta COD_b \cdot Y_H \cdot SRT}{1 + k_{dH} \cdot SRT} \qquad (mgN/l)$$

b. Modified Lutzack-Ettinger (MLE) system: Anoxic – aerobic sequence

Anoxic – aerobic sequence as MLSS and nitrate recycle from aerobic to anoxic reactor. Wastewater (Q), return sludge (Q_r) and recycled mixed liquor (Q_a) from aerobic reactor containing nitrate are mixed in an anoxic reactor providing conditions for denitrification. COD in wastewater is utilised for denitrification. Recycle rate of mixed liquor and wastewater COD/TKN-ratio determines nitrate and total nitrogen concentrations in the effluent.

K_1 and K_2 describe denitrification in the MLE system, where COD from influent wastewater is utilised for denitrification. $K_1 >> K_2$ so K_2 will determine design while K_1 goes to completion as all RBCOD is consumed, replacing K_1 by a stoichiometric expression. The denitrification potential of the anoxic reactor of the MLE system (primary anoxic reactor) expressed as D_{P1}

$$D_{P1} = C_{RBCOD} \cdot (1 - Y_H) \cdot / 2.86 + K_2 \cdot X_H \cdot R_{hn}$$

$$= 0.117 \cdot C_{RBCOD} + K_2 \cdot X_H \cdot R_{hn}$$

$$D_{P1} = \frac{C_{bs} \cdot (1 - Y_H)}{2.86} + \frac{K_2 \cdot f_x \cdot \Delta COD_b \cdot Y_H \cdot SRT}{1 + k_{dH} \cdot SRT} \qquad (mgN/l)$$

Implementation of the steady state biological N removal model is by calculating the actual biomass based on the system characterization in order to present the denitrification rates relative to the active biomass. The main variables are the biodegradable COD fractions in the wastewater and the SRT. In addition, a known decay rate is necessary to determine the K_3 rate. Growth yield (Y_H), un-biodegradable fraction of dead biomass (f_d) and the COD/VSS ratio (f_{cv}) are considered constant according to literature values. The decay rate (k_d) can be estimated based on the observed denitrification rates in the K_3 region (Table 5).

Table 5. Default values

	Parameter	Units	Values
Heterotrophic organisms	Y_H	gCOD.gVSS^{-1}	0.45
	μ_{max}	d-1	2
	K_s	mgCOD/l	10
	k_d	d-1	0.2
	K_r		0.015
Autotrophic organisms	Y_A	gCOD.gN^{-1}	0.17
	μ_{maxA}	d-1	0.8
	k_{dA}	d-1	0.05
	K_N	mgNA/l	1.0
	f_N	gN/gVSS	0.1

3.4.5. Determination of decay rate based on NUR in the endogenous phase

The decay of biomass applied in the steady state model is based on loss of biomass proportional to the biomass concentration and the specific decay rate k_d. The k_d is usually determined in batch tests where the reduction in biomass (VSS) is as a reduction in VSS directly or as reduction in OUR. When there is no external COD, K_3 is the slowest rate of denitrification and according to the decay rate. Therefore, it should be possible to estimate the k_d based on the measurements of denitrification when all COD has been consumed.

$$\frac{dX_H}{dt} = -k_d \times X_H$$

$$\frac{dO}{dt} = -(1-f_d) k_d \times X_H \times f_{cv}$$

$$\frac{dX_E}{dt} = -(1-f_d) k_d \times X_H$$

$$1COD_{XH} = f_d . COD_{XH} + (1-f_d) . COD_{XH}$$

$$NUR = COD_{XH} \times \frac{1}{2.86}$$

$$NUR = \frac{dN}{dt} = (1-f_d) k_d \times X_H \times f_{cv} \times \frac{1}{2.86}$$

$$k_d = \frac{NUR}{(1-f_d) \times X_H \times f_{cv} \times \frac{1}{2.86}}$$

Estimation of k_d was done on several of the batch tests from the first experiments and indicated values within the ranges reported in other studies. Based on the batch test done 29.08 2012 (Figure 19) the NUR in the endogenous phase was 56.45 mgN/l/d, the active biomass was about 50 % of the MLVSS, corresponding to 895 mgVSS/l.

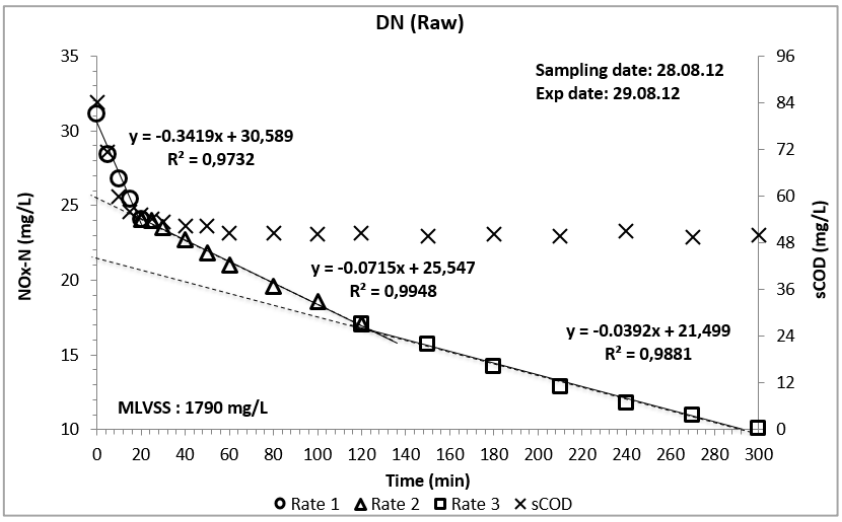


Figure 19. Batch test for determination of NUR, 29.08 2012

Calculation of the decay rate was based on the following data:

NUR = 56.45 mgN/l/d;

X_H = 895 mgVSS/l,

f_{cv} = 1.42 mgCOD/mgVSS,

f_d = 0.2 (-)

$$k_d = \frac{56.45}{(1\text{-}0.2)\times 895\times 1.42\times \frac{1}{2.86}} = 0.159\ d^{-1}$$

A decay rate of 0.159 d^{-1} is within reported values such as in Metcalf & Eddy (2003) and Henze *et al.* (2008), and employed for further use.

Chapter 4

Denitrification rates as a function of TSS

The objective of the experiment was to evaluate the effect of TSS removal on denitrification rates. The optimum level of TSS and corresponding COD removal is where acceptable nitrogen removal exists. As the COD requirements for nitrogen removal may vary depending on plant configuration, the tests performed in these experiments refer to a pre-denitrification system (MLE). The tests conducted at Aquateam lab in Oslo consisted of several anoxic batch tests and sequencing batch reactors (SBRs) fed with filtered wastewater, resulting in variable COD concentrations.

4.1. Anoxic batch tests

Filtered wastewater used in the anoxic batch tests were by different filters according to the procedure described in Chapter 3 (Feed water preparation). Two sets of batches tested activated sludge (AS) from BRA WWTP and fixed film (MBBR) from NFR WWTP. Wastewater from both BRA and NFR WWTPs investigated effects of TSS and corresponding COD on denitrification rates.

4.1.1. Operation and control

Figure 20 depicts two batch reactors each of 3 liters used for the tests. For activated sludge equal volumes of RAS and wastewater were mixed (1.5 L each), while the MBBR was filled at 50 % volume with biofilm carriers (Kaldnes K1). The first reactor served as control and filled with degritted raw wastewater, while the second reactor was with filtered wastewater. For each anoxic batch test the influent wastewater was characterized by suspended solids (TSS and VSS), COD, ammonium-nitrogen (NH_4-N), nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N) and orthophosphate (PO_4-P) before and after filtration. During the tests, temperature, pH and DO were recorded.

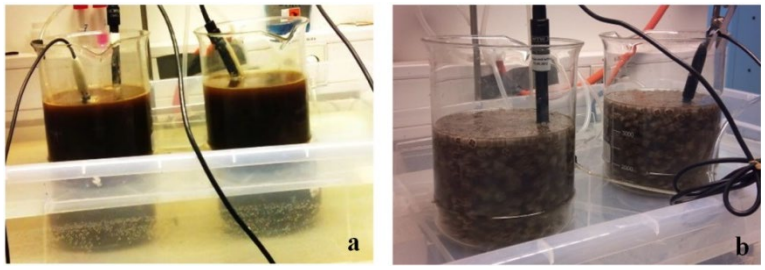


Figure 20. Anoxic batch tests
(a. Activated Sludge, b. Moving Bed Biofilm Reactor)

Table 6 summarizes conditions and characteristics of the different tests.

Table 6. Experimental plan

Process	Test	Reactor composition	Filters tested (µm)
AS	Test 1	ww from NFR + RAS from BRA	1.2 (ff)[a], 1.2, 33, 150
	Test 2	ww from BRA + RAS from BRA	1.2 (ff), 1.2, 33, 90
MBBR	Test 3	ww from NFR + Kaldnes K1 from NFR	1.2 (ff), 1.2, 18, 33, 90
	Test 4	ww from BRA + Kaldnes K1 from NFR	1.2 (ff), 1.2, 18, 33, 90

[a] ff: flocculated with alum prior to filtration

4.1.2. Results and Discussion

a. Wastewater characterization

Table 7 shows compositions of influent wastewater from the two WWTPs. As the experimental period for these tests were nearly two months, the composition of the wastewater varied.

Table 7. Characteristics of influent wastewater from BRA and NFR WWTPs during the experimental period (14 August – 9 October 2012)

Parameter	BRA WWTP		NFR WWTP	
	Range	Mean	Range	Mean
TSS (mg/L)	94 – 528	280 ± 110	132 – 262	220 ± 40
VSS (mg/L)	72 – 457	250 ± 110	105 – 228	190 ± 40
VSS/TSS	0.76 – 0.99	0.86 ± 0.07	0.79 – 0.90	0.86 ± 0.03
TCOD (mg/L)	143 – 801	500 ± 180	231 – 577	460 ± 100
pCOD (mg/L)	93 – 547	320 ± 130	156 – 375	300 ± 70
sCOD[3] (mg/L)	50 – 271	180 ± 70	75 – 205	160 ± 50
pCOD/VSS	1.02 – 1.57	1.35 ± 0.17	1.35 – 2.36	1.50 ± 0.16
RBCOD[4] (mg/L)	73 - 168	117 ± 34	42 - 133	76 ± 32
SBCOD (mg/L)	166 - 440	283 ± 95	109 - 314	234 ± 61
upCOD (mg/L)	34 - 107	68 ± 23	47 - 74	57 ± 9
usCOD (mg/L)	29 - 124	68 ± 36	17 - 163	89 ± 42
NO_3-N (mg/L)	0.26 – 1.16	0.57 ± 0.27	0.27 – 1.81	0.74 ± 0.57
NO_2-N (mg/L)	0.01 – 0.10	0.07 ± 0.04	0.02 – 0.17	0.08 ± 0.05
NH_4-N (mg/L)	9 – 24	18 ± 6	8 – 27	21 ± 6
PO_4-P (mg/L)	1.04 – 4.64	2.11 ± 1.40	1.21 – 3.86	2.82 ± 1.03

Average results given with the standard deviation. Number of samples per WWTP = 9

[3] After filtration through 1.2 µm Whatman GF/C filter
[4] RBCOD and SBCOD determined according to Mamais *et al.*, 1993

The lowest values of COD and TSS were due to dilution from rainfall. These variations may have affected the results. The soluble COD containing the RBCOD made up one-third of the total COD. The wastewater from BRA WWTP had concentrations of 280 mg/L TSS and 500 mg/L COD. This compared to NFR WWTP with 220 mg TSS/L and 460 mg COD/L.

The ratio of particulate COD to VSS at BRA and NFR WWTPs were 1.35 ± 0.16 and 1.50 ± 0.17 gCOD/gVSS, respectively. Higher fraction of particulate COD in the influent indicates higher fraction of COD removal during primary treatment, which result in higher organic load to the anaerobic digester and thus higher biogas production. On the other hand, increased COD removal in primary treatment may reduce the efficiency of the biological nitrogen removal. (Gori *et al*., 2011).

The nitrogen compounds were mainly present in the form of ammonium and were less than 30 mg NH_4-N/L for both WWTPs. Nitrate and nitrite concentrations were negligible. Orthophosphate was less than 5 mg PO_4-P/L. The average concentrations for both WWTPs were similar to typical values for diluted municipal wastewater (Tas *et al*., 2009; Henze & Comeau, 2008).

b. Separation performance

Figures 21 and 22 show COD and TSS removal efficiencies associated with the different filters with wastewater from NFR and BRA WWTPs, respectively.

TSS and COD removal were inversely proportional to the filter openings. At NFR, TSS removal varied from 54 % with 18 µm to 38 % with 150 µm. At BRA, removal of TSS varied from 57 % with 18 µm to 46 % with 90 µm. The corresponding COD removal was 43 % with 18 µm and 21 % with 150 µm at NFR and 42 % with 18 µm and 32 % with 90 µm at BRA.

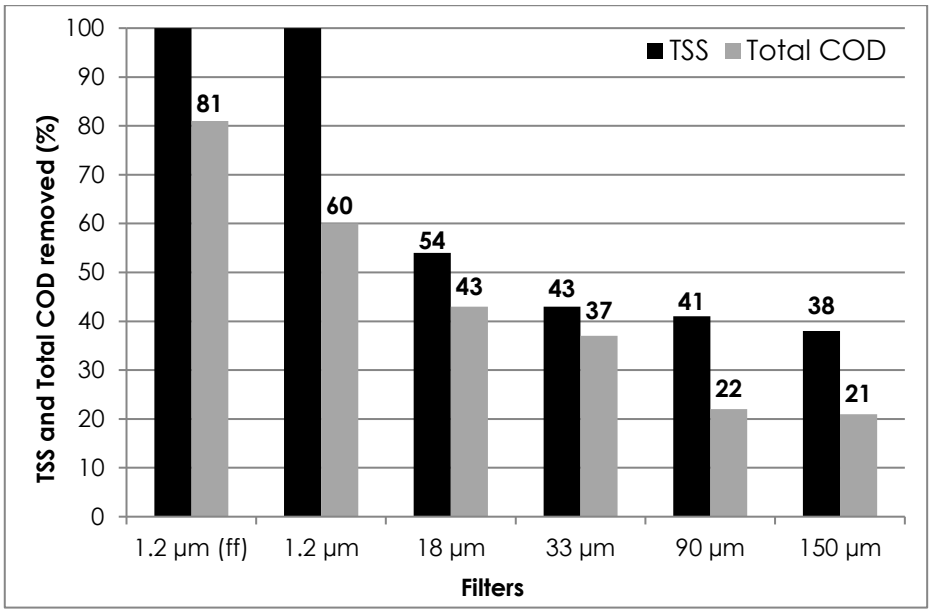


Figure 21. TSS and TCOD removal efficiencies after separation with different filters for NFR influent wastewater

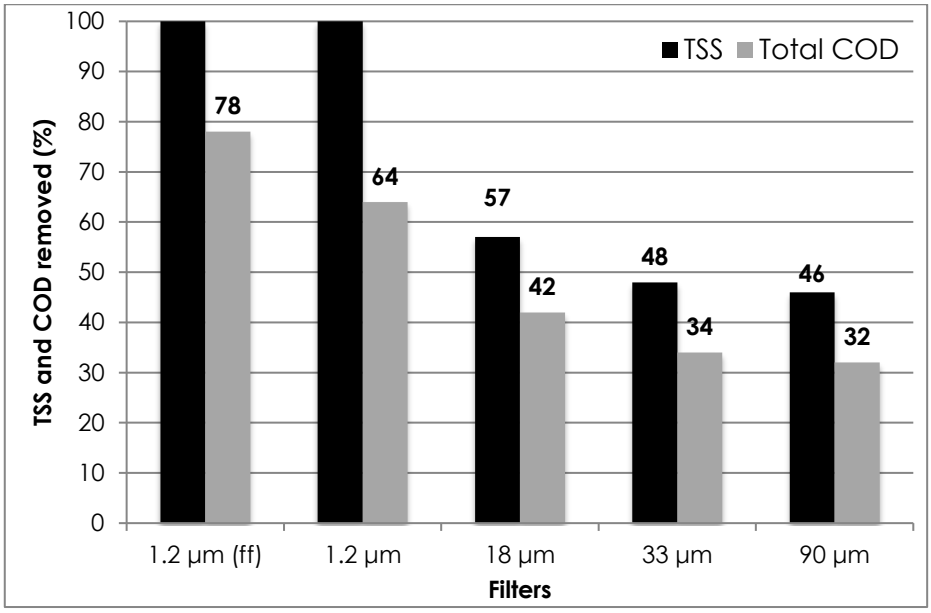


Figure 22. TSS and TCOD removal efficiencies after separation with different filters for BRA influent wastewater

With 1.2 µm filters, TSS removal was 100 % while the COD removal was 81 % and 60 % with and without Alum-addition, respectively, at NFR. The COD removal at BRA was 78 and 64 % with and without Alum-addition, respectively.

Based on the size distribution, COD in municipal wastewater are classified as soluble (< 0.001 µm), colloids (0.001 – 1 µm), supracolloidal (1 – 100 µm) and settleable (> 100 µm) fractions (Levine *et al.*, 1991). For the wastewater at NFR, 46 % of the TSS were between 1.2 and 18 µm, 13 % were between 18 and 90 µm and 41 % were above 90 µm. At BRA, 43 % of the suspended solids were between 1.2 and 18 µm, 16 % were between 16 and 90 µm and 43 % were above 90 µm. For COD at NFR the composition was 57 % below 18 µm, 21 % between 18 and 90 µm and 22 % above 90 µm. At BRA, the COD composition was 58 % below 18 µm, 10 % between 18 and 90 µm and 32 % above 90 µm. Figure 23 shows fractionation of COD according to the definitions by Levine et al. (1991). The difference in filtered COD between flocculated and non-flocculated sample indicates the fraction of colloids in the wastewater. At NFR, the fraction was 21 % and at BRA 14 % of total COD. The filtered or soluble COD in the flocculated samples corresponds to 19 % at NFR and 22 % at BRA of total COD, respectively. The fractions of TSS removed at the different filter openings show clearly the particle size distribution (PSD) in the influent wastewater (Newcombe *et al.*, 2011).

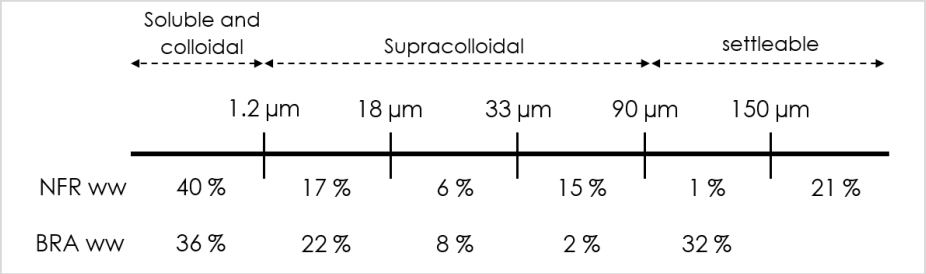


Figure 23. COD fractionations with NFR and BRA wastewater

Ravndal (2017) observed COD fractionations of 40 – 52 % soluble, 2 – 8 % colloids, 20 – 27 % supracolloidal and 20 – 31 % settleable when analyzing wastewater from Vik and Mekjarvik (Stavanger, Norway). The definition of colloids in that study was in the range of 0.001- 0.65 µm, while

Levine et al. (1991) used the definition of 0.001-1 µm. This explains some of the different values reported for characterization of COD in different wastewaters.

These overall results were similar to the reported removal efficiencies of commercial Salsnes Filter units using a filter of 300 µm or 350 µm operated with mat formation (Salsnes Filter, 2013; Rusten, 2002). According to Rusten & Ødegaard (2006) once a filter mat is on the surface of the filter, there is no difference in the performance of different filter openings. The performance of Salsnes Filter improved due a dynamic layer or fouling, that prevent further penetration of particles through the filters (Søraunet, 2012).

c. Specific denitrification rates

Table 8 presents specific denitrification rates (SDNR) obtained from the anoxic batch tests with activated sludge and MBBR. The results summarize the different rates and the corresponding influent total COD to nitrate (NO_3-N) ratio (C/N) for the different tests. All rates were adjusted to 20 °C using temperature coefficients of 1.20 for the first rate (Rate 1), 1.08 for the second rate (Rate 2), and 1.029 for the endogenous decay (Rate 3) (Henze *et al.*, 2008).

Due to the large variations in wastewater composition and other operating parameters, it was difficult to compare results between tests. For this reason, one reactor (R1) operated with raw wastewater in all tests, for direct comparison with reactors (R2) receiving filtered wastewater.

Table 8. SDNRs with the corresponding influent C/N and pCOD/TSS ratios from NUR tests with AS (Tests 1 & 2) and MBBR (Tests 3 & 4) processes

Test 1 (NFR WW)		Raw wastewater			Filtered wastewater		
Filters	**Rate**	**SDNR**	**pCOD/TSS**	**C/N**	**SDNR**	**pCOD/TSS**	**C/N**
1.2 µm (ff)	1	-	1.207	7.19	-	0	1.29
	2	0.06			0.05		
	3	0.03			0.02		
1.2 µm	1	0.07	1.250	8.40	0.07	0	3.41
	2	0.05			0.04		
	3	0.03			0.02		
33 µm	1	0.09	1.477	7.75	0.09	0.920	5.41
	2	0.05			0.04		
	3	0.03			0.03		
150 µm	1	0.05	1.227	7.67	0.06	1.473	6.10
	2	0.04			0.04		
	3	0.03			0.03		

Test 2 (BRA WW)		Raw wastewater			Filtered wastewater		
Filters	**Rate**	**SDNR**	**pCOD/TSS**	**C/N**	**SDNR**	**pCOD/TSS**	**C/N**
1.2 µm (ff)	1	0.21	1.266	9.66	0.19	0	2.38
	2	0.05			0.05		
	3	0.03			0.02		
1.2 µm	1	0.18	1.037	11.99	0.21	0	4.33
	2	0.05			0.07		
	3	0.03			0.03		
33 µm	1	0.26	0.936	8.56	0.24	0.812	6.38
	2	0.06			0.06		
	3	0.04			0.03		
90 µm	1	0.26	1.284	9.71	0.22	1.091	5.86
	2	0.05			0.05		
	3	0.03			0.03		

Test 3 (NFR WW)		Raw wastewater			Filtered wastewater		
Filters	**Rate**	**SDNR**	**pCOD/TSS**	**C/N**	**SDNR**	**pCOD/TSS**	**C/N**
1.2 µm (ff)	1	0.81	1.180	8.02	0.80	0	1.45
	2	0.61			0.60		
	3	0.34			0.30		
1.2 µm	1	1.23	1.320	19.18	1.19	0	7.52
	2	0.90			0.79		
	3	-			-		
18 µm	1	2.40	1.443	16.93	1.82	1.000	9.87
	2	1.83			1.67		
	3	1.39			1.28		
33 µm	1	2.43	1.625	18.03	2.09	1.150	11.06
	2	1.63			1.56		
	3	1.11			1.14		
90 µm	1	0.97	1.178	16.01	0.94	1.320	12.79
	2	0.59			0.57		
	3	0.35			0.35		

Test 4 (BRA WW)		Raw wastewater			Filtered wastewater		
Filters	**Rate**	**SDNR**	**pCOD/TSS**	**C/N**	**SDNR**	**pCOD/TSS**	**C/N**
1.2 µm (ff)	1	2.69	1.421	18.18	2.43	0	5.43
	2	2.36			1.94		
	3	-			-		
1.2 µm	1	2.23	1.077	12.13	1.97	0	4.23
	2	1.63			1.56		
	3	1.15			1.13		
18 µm	1	1.22	0.983	4.78	1.23	0.925	2.91
	2	1.04			0.99		
	3	0.78			0.83		
33 µm	1	2.33	0.912	15.08	2.10	1.215	10.79
	2	1.60			1.49		
	3	-			-		
90 µm	1	2.17	1.200	13.45	2.20	1.049	8.65
	2	1.88			1.76		
	3	1.24			1.17		

SDNR, expressed as gNO_3-N/gVSS.d
SDNR, expressed as gNO_3-N/m^2.d
pCOD/TSS expressed as gCOD/gTSS
C/N, expressed as gCOD/gNO_3-N

The results show that tests done with wastewater from BRA WWTP had higher first rate (K1) at 0.18 and 0.26 gNO_x-N/gVSS-d (Test 2) compared to NFR WWTP at between 0.05 and 0.09 gNO_x-N/gVSS-d (Test 1). One reason for the difference can be that collected activated sludge was from BRA, and adapted to that wastewater compared to the wastewater from NFR. However, one should then also have expected to have larger differences on K2 and K3. However, the K2 and K3 rates were similar during the two tests. The values of K2 rates were between 0.04 – 0.06 gNO_x-N/gVSS-d, and between 0.02 – 0.03 gNO_x-N/gVSS-d for the K3 rates.

The difference observed on K1 may be due to the composition of the soluble COD present in the wastewaters. It could also be due to experimental errors, as the first rate is more difficult to measure correctly due to the uptake and storage of soluble COD. Uptake and storage of soluble COD without direct utilization will affect the denitrification rate and will depend on the composition of the soluble COD. For the observed municipal wastewater up to 38 % carbohydrates, 30 % proteinaceous matter, and 38 % lipids were observed (Ravndal, 2017; Gorini *et al*., 2011; Raunkjaer *et al.*, 1994).

The SDNRs observed during this study were similar to the results from literature. The first SDNR ranges from 0.08 – 0.46 gNO_x-N/gMLVSS-d (Kapagiannidis *et al*, 2006; Barnard & Meiring, 1977), 0.04 – 0.12 gNO_x-N/gMLVSS-d for the second rate (Naidoo, 1999) and between 0.02 – 0.07 gNO_x-N/gMLVSS-d for the endogenous rate (Naidoo, 1999; Randall *et al*., 1992).

In the MBBR process tests, the first rates (K1) varied between 0.80 – 2.43 gNO_x-N/m^2-d in the reactor fed with wastewater from NFR WWTP (Test 3) compared to 1.22 – 2.69 gNO_x-N/m^2-d in the reactor fed with wastewater from BRA WWTP (Test 4). Unlike the tests done with the activated sludge process, K2 and K3 rates were different during Test 3 and Test 4. K2 rates were between 0.57 – 1.83 gNO_x-N/m^2-d in Test 3 and between 0.99 – 2.36 gNO_x-N/m^2-d in Test 4. The third rates (K3) values ranged between 0.30 – 1.39 gNO_x-N/m^2-d in Test 3 and between 0.78 – 1.24 gNO_x-N/m^2-d in Test 4.

Calculations of specific denitrification rates shown in Tables 9 and 10 are from active biomass (50 % of total MLVSS in the reactor) and tests with activated sludge.

Table 9. Specific denitrification rates based on calculated biomass-VSS (using wastewater from NFR WWTP)

Filters	Rate	Raw wastewater	Filtered wastewater
1.2 µm (ff)	1	---	---
	2	0.13	0.11
	3	0.06	0.05
1.2 µm	1	0.25	0.23
	2	0.15	0.10
	3	0.07	0.05
33 µm	1	0.27	0.26
	2	0.12	0.10
	3	0.07	0.07
150 µm	1	0.17	0.22
	2	0.12	0.12
	3	0.06	0.06

Table 10. Specific denitrification rates based on calculated biomass-VSS (using wastewater from BRA WWTP)

Filters	Rate	Raw wastewater	Filtered wastewater
1.2 µm (ff)	1	0.41	0.35
	2	0.10	0.09
	3	0.06	0.04
1.2 µm	1	0.44	0.46
	2	0.13	0.14
	3	0.06	0.09
33 µm	1	0.60	0.55
	2	0.14	0.13
	3	0.08	0.03
90 µm	1	0.50	0.41
	2	0.10	0.10
	3	0.05	0.05

The table shows that the second (K2) and third (K3) rates were close to the reported values in literature (Henze *et al.*, 2008), which were 0.1 and

0.07 gNO_x-N/gMLVSS.d, respectively. However, the first rates (K1) are lower than literature data, especially for the test with wastewater from NFR WWTP. The values were between 0.17 – 0.26 gNO_x-N/gMLVSS.d with NFR wastewater (Figure 24) and between 0.35 – 0.60 gNO_x-N/gMLVSS.d with BRA wastewater (Figure 25) compared to 0.72 gNO_x-N/gMLVSS.d from literature.

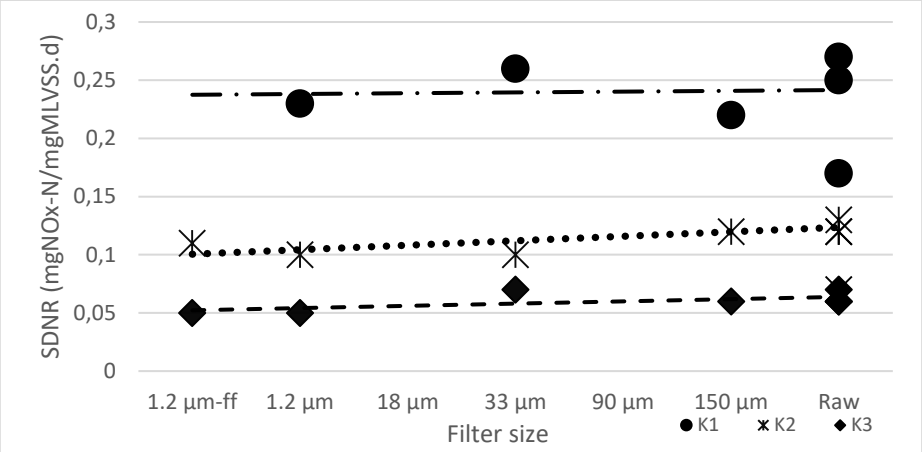


Figure 24. SDNR using wastewater from NFR WWTP

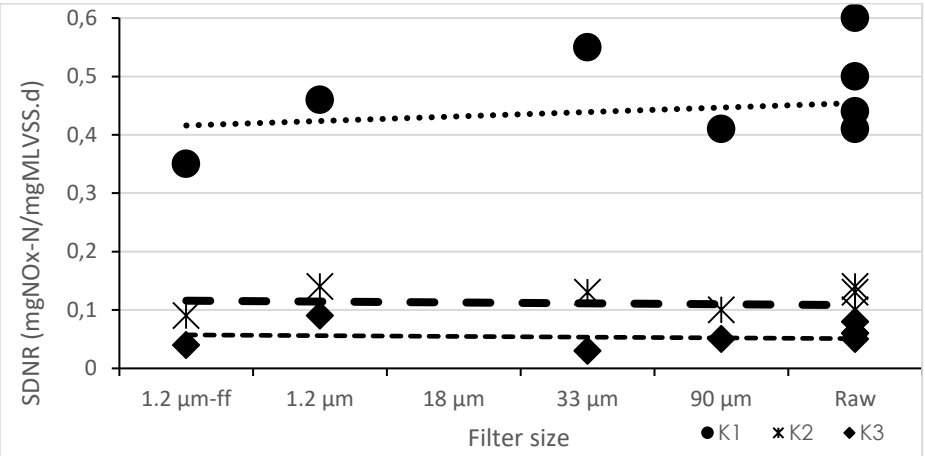


Figure 25. SDNR using wastewater from BRA WWTP

The comparison of the two reactors fed with raw and filtered wastewater showed that the SDNRs results were similar for tests with activated sludge

and with biofilm processes, despite the different organic loads to the reactors.

Soluble COD represented a small fraction of the total COD, with denitrification mostly driven by hydrolyzed particulate COD (Abufayed & Schroeder, 1986). In the case of tests fed filtered wastewater with only soluble COD, additional carbon source may be from the hydrolysis of the biomass itself. Therefore, conclusion is that endogenous carbon added to drive the denitrification reaction, leads to the similarity in denitrification rates.

4.1.3. Conclusions

Analysis showed that wastewater from BRA WWTP had COD concentrations of about 500 ± 180 mg TCOD/L and slightly lower at NFR WWTP, which was 460 ± 100 mg TCOD/L. The soluble COD made up about one-third of the total COD. Separation with the different filters revealed that about 75 – 80 % of the organic matter was removed when the sample was pre-flocculated prior to filtration with 1.2 µm filter. About 60 – 65 % of the COD separated by filtration with 1.2 µm filter. Salsnes Filters removed about 40 % of COD with 18 µm filter and about 28 % with 150 µm filter.

The comparative study showed that the tests performed with BRA wastewater had higher first rate (SDNR 1) compared to NFR wastewater. For the second (SDNR 2) and third (SDNR 3) rates the results were similar for both wastewaters operated in the activated sludge system.

Conclusion after these tests are that removal of particulate organic matter had little effect on the denitrification rates. The observed denitrification rates were also within reported values in literature, indicating that the rates show small differences between different wastewaters. Consequently, lab-scale sequencing batch reactors in the next experiment investigates the impact on denitrification of removal of certain fractions of the particulate COD.

4.2. Lab-scale sequencing bath reactors

SBR is a fill-and-draw activated sludge treatment system. The processes involved in the SBR are identical to the conventional activated sludge. SBR is compact, time oriented, and all processes are carried out sequentially in the same reactor (Mahvi, 2008; Zhou *et al.*, 2006; NEIWPCC, 2005; Obaja *et al.*, 2005; Li & Zhang, 2002; Lim *et al*., 2002; Lin & Chang, 2000; White & Schnabel 1998; Keller *et al.*, 1997; Rim *et al.*, 1997).

4.2.1. Experimental setup and operating conditions

Experiments performed in three transparent polyvinylchloride (PVC) plastic cylinder bioreactors with a working volume of 3 L show in Figure 26, with tests conducted over three periods. During the first period (P1) the three SBRs were fed with (R1) degritted raw wastewater, (R2) filtered wastewater from 18 µm filter, and (R3) wastewater filtered through GF/B Whatman filter of 1.2 µm. In the second period (P2) degritted raw wastewater was fed the control reactor, while filters of 90 µm and 33 µm were used to produce the feed for the two remaining SBRs. In the third period (P3) the control reactor fed with degritted raw wastewater was compared with two SBRs fed with filtered wastewater from 150 µm and 55 µm, respectively.

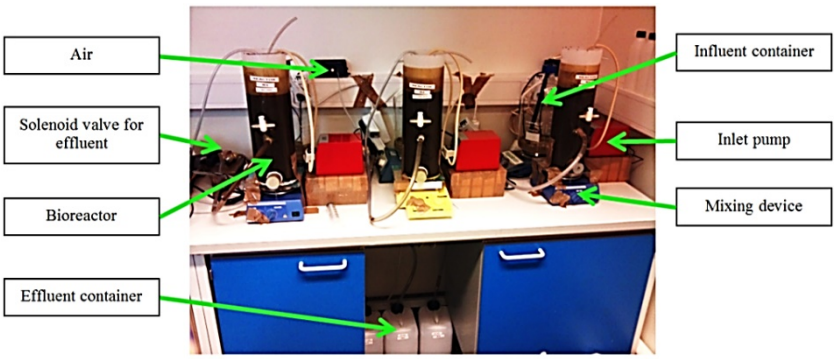


Figure 26. Lab-scale SBRs for nitrogen removal. From left to right: Reactor 1 (R1) fed raw wastewater, Reactors 2 (R2) and 3 (R3) fed filtered wastewater

Collected mixed liquor was from Bekkelaget (BRA) WWTP, whereas wastewater was from Nordre Follo (NFR) WWTP. Grab samples of degritted wastewater were in a 20-L plastic container, collected three times per week and stored at 4 °C. Equilibrated to 20 °C and sparged with nitrogen gas striped dissolved oxygen.

Analyzed three times a week for suspended solids, COD and nutrients (N & P) for the different wastewaters in the SBRs, and five times a week for SBRs effluents. Daily measurements performed for DO, pH and temperature.

After achieving steady-state conditions, the sludge in each SBR was characterized by determining the sludge volume index SVI (NS, 2006; WEF, 1994), specific utilization rates of ammonia (Melcer *et al.*, 2003; Kristensen *et al*., 1992) and nitrate (Gu & Onnis-Hayden, 2010). During each anoxic trial, added nitrate to the reactor achieved an initial nitrate concentration of 30 mgNO_3-N/L.

Table 11 summarizes operation parameters of the laboratory scale SBRs used during this experiment. In Period 1 the SBRs were operated with 2 minutes of static fill, 15h 55min in anoxic reactor with mixing, 7 hours in an aerobic reactor and finally 1 hour settling followed by 3 minutes decanting. While in Period 2 and 3, the operational cycle for the SBRs included 13 minutes of static fill, 4 hours of an anoxic period with mix, 3 hours of aeration with mix and 46 minutes of settle followed by 1 minute of decanting.

Table 11. Operating parameters for each SBR and each test period

	Period 1			Period 2			Period 3		
Parameter	**R1 Raw[a]**	**R2 18 µm**	**R3 1.2 µm**	**R1 Raw**	**R2 90 µm**	**R3 33 µm**	**R1 Raw**	**R2 150 µm**	**R3 55 µm**
Reactor volume (L)	3	3	3	3	3	3	3	3	3
Number of cycles per day	1	1	1	3	3	3	3	3	3
Cycle length (h)	24	24	24	8	8	8	8	8	8
Influent flow rate (L/d)	1.5	1.5	1.5	4.5	4.5	4.5	4.5	4.5	4.5
HRT[b] (h)	48	48	48	16	16	16	16	16	16
pH	7 - 8	7 - 8	7 - 8	7 - 8	7 - 8	7 – 8	7 - 8	7 - 8	7 - 8
Temperature (˚C)	20.8	20.9	20.7	22.8	21.8	23.1	22.6	21.3	22.6
MLSS[c] (mg/L)	1476	820	494	2560	2832	1954	5348	4788	3499
MLVSS[d] (mg/L)	1169	624	399	1932	1897	1367	4027	3376	2516
MLSS wasted per day (L)	0.2	0.2	0.2	0.15	0.15	0.15	0.15	0.15	0.15
SVI[e] (mL/g MLSS)	104	83	125	93	71	91	84	89	100
Total SRT[f] (d)	13.4	12	11	10.6	13.1	13.1	15.9	16.3	16
Aerobic SRT[g] (d)	4.1	3.7	3.3	4.6	5	5.6	6.8	7	6.8
Total F/M [h] (g TCOD/g MLVSS-d)	0.14	0.16	0.15	0.27	0.20	0.19	0.15	0.14	0.17
Aerobic F/M (g TCOD/g MLVSS-d)	0.48	0.53	0.52	0.71	0.54	0.50	0.41	0.37	0.45
Applied C/N (g TCOD/g TN)	10.2	5.8	3.8	14.4	13.1	9.4	10.3	8	7.5
Applied C/N (g sCOD/g TN)	3.6	3.3	3.8	4.7	5.5	6.3	3.5	3.4	3.6

Results are given as averages for the different test periods

[a] Raw: raw wastewater
[b] HRT: Hydraulic retention time (which is the volume of the reactor divided by the influent flow rate)
[c] MLSS: Mixed liquor suspended solids
[d] MLVSS: Mixed liquor volatile suspended solids
[e] SVI: Sludge volume index
[f] SRT: Sludge retention time (= mass of sludge in reactor/mass of sludge wasted per day)
[g] Aerobic SRT [= Total SRT x (aerobic cycle duration/total cycle duration)]
[h] F/M: Food to biomass ratio

4.2.2. Results and Discussion

a. Separation performance

Figure 27 shows filter performance with regard to TSS, COD and TN removals during 3 periods. The characteristics of the influent wastewater from NFR WWTP were medium strength wastewater (von Sperling, 2007c).

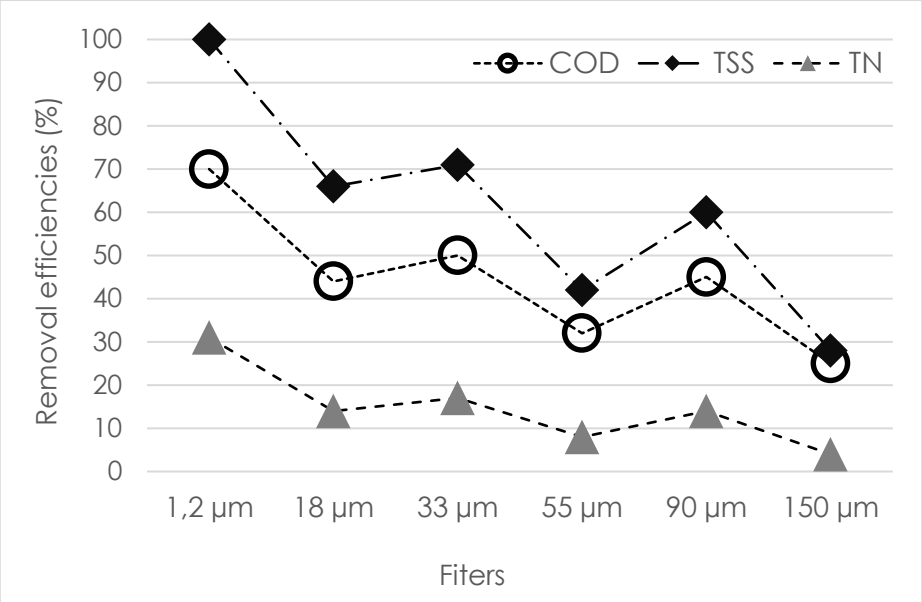


Figure 27. Separation performance

In Period P1 about 70 % of the total COD and 29 % of TN were removed by filtration with a 1.2 µm filter. Filtration with 18 µm removed nearly 70 % TSS, 44 % COD and 16 % nitrogen. These values correspond with results during the anoxic batch tests. Similar results were achieved at Tiendeholmen WWTP (Norway) using a filter cloth of 350 µm operated with filter mat (Rusten & Lundar, 2006; Rusten, 2000).

In Period P2, it was observed that a 90 µm filter removed 45 % of the COD and 14 % of the TN. 50 % of the COD and 17 % of the TN were removed with a 33 µm filter. Compared with results during the preliminary study

more COD was removed during this experiment. The difference is because 60 % of the TSS were larger than 90 µm compared to 42 % in the previous study. Therefore, also more COD associated with the TSS was removed. According to Newcombe *et al.* (2011), in addition to the filter cloth sizes, the performance of a filter depends also on the size distribution of particulates in the influent wastewater.

In Period P3 only 27 % of the TSS were removed with a 150 µm filter, which corresponded to about 25 % of the COD and 4 % of the TN shown in Figure 12. Filtration with 55 µm filter removed about 33 % of the COD and less than 10 % of the TN. This is similar to the removal efficiencies observed during the preliminary study. Tests at Blaricum WWTP (Netherlands) with a 350 µm filter and operated with filter mat, achieved a COD removal of 35 % and 50 % TSS removal (Ruiken *et al*., 2013), which is comparable to the performance of the Salsnes Filter 55 µm filter.

It was difficult to compare these results with available literature data as most of the reported studies with Salsnes Filter performed with filters of 300 µm or 350 µm and operated with a filter mat (Rusten, 2005a, 2002 and 2000). However, the analysis of particle size distribution (PSD) of the influent wastewater estimate the removal efficiencies of a specific filter. Examples of previous studies have shown the size distribution of COD in municipal wastewater of around 40 – 60 % below 0.001 µm, 7 – 16 % between 0.001 to 1 µm, 12 to 28 % between 1 to 100 µm and 15 to 29 % above 100 µm (Rickert & Hunter, 1971; Balmat, 1957). Table 12 (Ravndal, 2017) characterize wastewater from two WWTPs in Stavanger.

Table 12. Particle size distribution of the organic COD at Vik and Mekjarvik WWTP

Size class	< 1 kDa	1 kDa – 0.65 µm	0.65 µm – 25 µm	25 µm – 100 µm	>100 µm
Mekjarvik wastewater	52 %	8 %	15 %	5 %	20 %
Vik wastewater	40 %	2 %	25 %	2 %	31 %

It can be expected with wastewater from Vik that a filter of 25 µm will remove about 33 % of the particulate COD and a filter of 1 µm will

remove nearly 60 % of the organic COD. The analysis of the PSD of the influent wastewater shows most COD dissolved, while colloidal COD represents only a small fraction of the total. Another study by Dulekgurgen *et al*. (2006) observed that 65 % of the COD were particulates[5].

Table 13, Figures 28 and 29 present composition of different wastewater fractions. The values of N/VSS ratios varied between 0.075 and 0.122 gN/gVSS with average of 0.1 gN/gVSS for the raw wastewater. For filtered wastewaters varied between 0.134 and 0.316 gN/gVSS, indicating a higher removal of TSS and VSS relative to removal of N. The plot of the different values against the different filters shows that N/VSS ratio increases with increasing removal of suspended solids. The main reason for that is a higher removal of cellulose with low N content, so the N content relative to VSS increases as well as the N/VSS ratios.

Table 13. Influent wastewater characteristics

		1,2 µm	18 µm	33 µm	55 µm	90 µm	150 µm	Raw
N/TSS	P1	0.000	0.092					0.087
	P2			0.123		0.106		0.059
	P3				0.173		0.129	0.098
N/VSS	P1	0.000	0.316					0.100
	P2			0.149		0.134		0.075
	P3				0.231		0.198	0.122
N/pCOD	P1	0.000	0.222					0.070
	P2			0.111		0.053		0.049
	P3				0.128		0.113	0.072
pCOD/TSS	P1	0.000	1.430					1.280
	P2			1.160		2.000		1.210
	P3				1.280		1.180	1.370

[5] a filter of 1.6 µm was used to differentiate soluble and particulate fractions

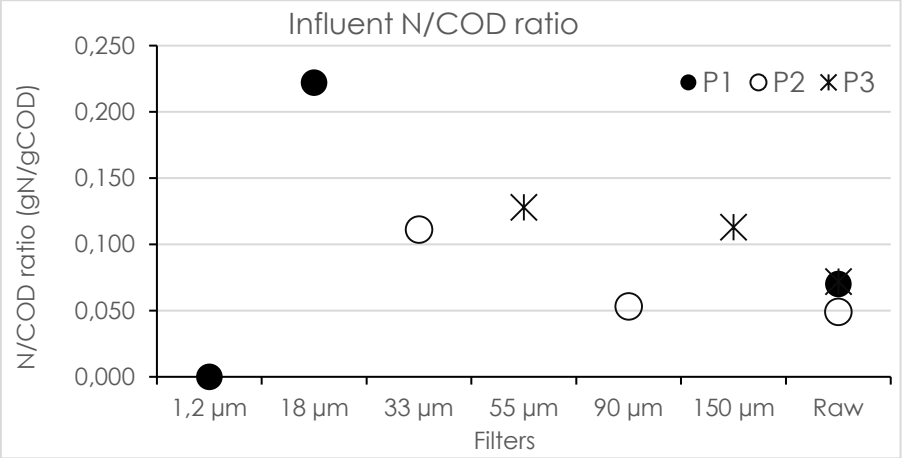


Figure 28. Influent N/COD ratio

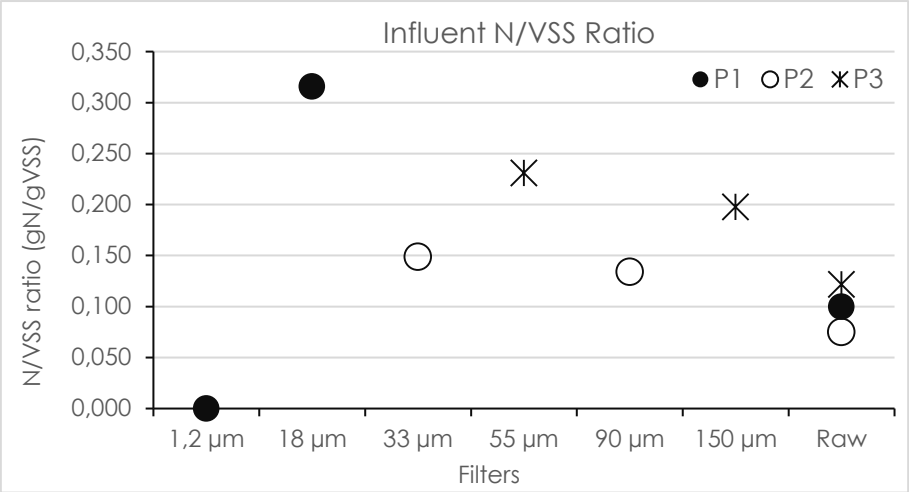


Figure 29. Influent N/VSS ratio

b. SBR performance

Table 14 summarizes the reactor performances during the three study periods. The performance of each SBR by examining the influent and effluent parameters determined the removal efficiencies.

Table 14. SBRs performances during the three study periods

	Filter	1.2 µm (P1)	18 µm (P1)	33 µm (P2)	55 µm (P3)	90 µm (P2)	150 µm (P3)	Raw
Removal efficiencies (%)	**Total COD**	90	90	90	86	89	89	91 (P1) 70 (P2) 75 (P3)
	SBR-COD	67	67	65	71	74	75	
	sCOD	81	65	63	50	60	80	80 (P1) 54 (P2) 45 (P3)
	TN	34	45	63	58	60	58	72 (P1) 59 (P2) 57 (P3)
	SBR-TN	12	21	26	33	25	34	
	TSS	93	93	93	97	90	97	90 (P1) 67 (P2) 90 (P3)
	SBR-TSS	86	87	58	90	75	91	
C/N ratio		3.8	5.8	9.4	7.5	13.0	8.0	10.2 (P1) 14.6 (P2) 10.3 (P3)

No post-treatment contributed after the settling stage of the SBRs. Therefore, the registered values for the effluent components were by analyzing directly the treated wastewater. The removal efficiencies observed in the control reactor correspond only to the biological removal. However, in the SBRs fed with filtered wastewater, the amount of removal of the different compounds with the filter during the primary treatment is accounted in the calculations.

The comparison of the SBRs' performances revealed that the removal efficiencies were lower in the reactors fed with filtered wastewater. The biological treatment removed between 65 to 75 % of the COD compared to 70 – 91 % in the control reactors. This indicates that reducing the COD load entering the reactor will affect the performance of the process. The main reason is the reduction in influent COD concentration, while effluent COD is more similar resulting in decreasing removal efficiency with decreasing influent COD.

However, when including the removal efficiency of the primary treatment, similar or relatively higher TSS and COD removals resulted in the SBRs fed with filtered wastewater compared to the control reactor. During P2 and P3 the removal efficiencies were slightly better in the reactors fed with filtered wastewater from 33 µm to 150 µm compared to the control reactor. For instance, removal of 90 % of the COD and TSS resulted in the reactors fed with 33 µm and 90 µm filtered wastewater. Conversely, in the control reactor, the values were 70 % (COD) and 67 % (TSS) for the same filters, respectively. TN removal efficiencies were between 12 % for wastewater filtered with 1.2 µm filters and 34 % for wastewater with 150 µm filter compared to between 59 and 72 % in the control SBRs (Figure 30). It was obvious that the TN removal was by the partial separation of the COD during primary treatment. However, the overall removal efficiencies (Primary treatment + Biological removal) were similar to the control reactor, except for the SBRs fed with filtered wastewater from filters below 33 µm where a net reduction in the TN removal resulted. The remaining COD in the filtered wastewater was not sufficient to achieve similar TN removal as in the control reactor.

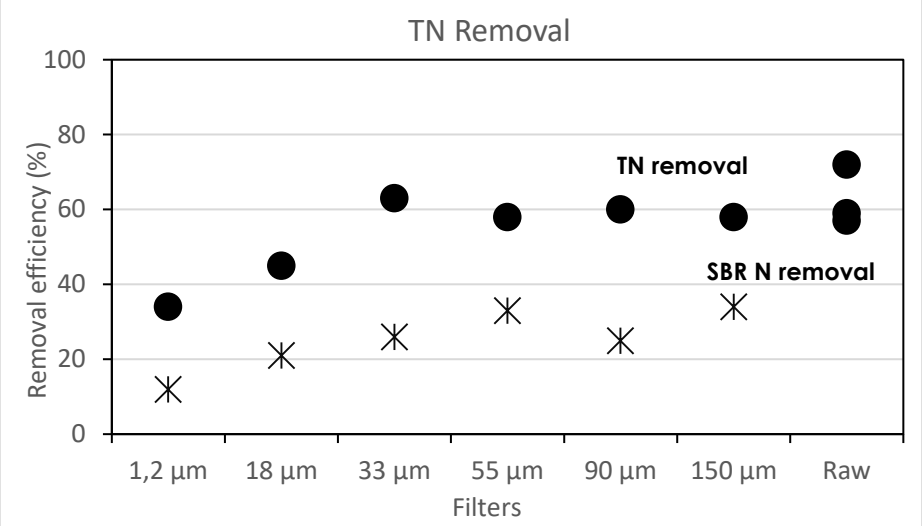


Figure 30. TN Removal efficiencies

The Denitrification process depends on the type and concentration of the carbon source and the C/N ratio (Breisha, 2010; Galvez *et al*., 2003). Theoretically, 2.86 g COD is required to denitrify 1 g NO_3-N (Kujawa & Klapwijk, 1999; Oostrom, 1995). But, as a part of the COD is used for biomass growth, the COD requirement increases correspondingly and the C/N ratio becomes 2.86/(1 – Y), and with a true yield of 0.66 gCOD/gCOD the C/N ratio becomes 8.6 gCOD/gN (Rusten *et al*., 1996). However, the observed yield is lower than the true yield and depends on the temperature, the nature of the carbon source and the loading rate (SRT) (Ramalingam et al. 2007; Bilanovic *et al*., 1999; Nyberg *et al.*, 1996; Rusten *et al*., 1996 and Christensson *et al*., 1994). A lower observed yield leads to a lower COD requirement for denitrification, mainly due to utilization of COD from dead biomass (endogenous COD).

A low C/N ratio (< 8 gCOD/gN) indicates a low denitrification potential due to lack of sufficient biodegradable COD (Henze & Comeau, 2008). The C/N ratios for the SBR fed with filtered wastewater from 1.2 µm and 18 µm filters were respectively 3.8 gCOD/gN and 5.8 gCOD/gN compared to 10.2 gCOD/gN in the control reactor. The lower C/N ratios explain the reduction in the TN removal efficiencies observed in Period 1. For comparison, observed TN removal efficiency of 20 % reported by

Chevakidagarn *et al.* (2012) at a C/N ratio of 4.5 gCOD/gN, while 98 % nitrogen removal at a C/N ratio of 12 gCOD/gN.

All reactors had low SVIs, with average values from 84 to 104 mL/g for the reactor fed with raw wastewater and between 71 and 125 mL/g for the reactors receiving filtered wastewater. These values are all above the minimum SVI of 50 mL/g recommended by Mesquita *et al*. (2008) and situated within the range of SVI leading to good sludge settling (von Sperling, 2007a). SVI values above 150 mL/g indicate filamentous bacteria and poor settling (Pa.DEP, 2014).

The results obtained during this study were to some extent in contradiction with the observation made during the anoxic batch tests. In the batch, the removal of TSS and COD with different filters did not affect the performance of the reactors significantly, perhaps attributed to the characteristic of the mixed liquor. Each SBR developed an activated sludge, which was a result of the specific feed for each reactor. The cycle times would also affect the performance. Increased frequencies between aerobic and anoxic periods would probably enhance TN removal. Several studies have showed that acclimation of the biomass is necessary in order to completely degrade different types of COD (Jung *et al.*, 2006; Hallin & Pell, 1998).

c. Sludge, biogas and energy productions

Table 15 show sludge production during the three study periods. For the control SBR the sludge production was only biological sludge, while in the SBRs fed with filtered wastewater, the sludge production is the combination of filter (primary) sludge and biological sludge.

With combined filtration and biological treatment more sludge resulted, as shown in Table 15. In Period 1 (P1), the overall sludge production varied from 0.32 gTS/d for the SBR fed raw wastewater to 0.59 gTS/d for the SBR fed filtered wastewater from 1.2 µm filtration. R2 fed with filtered wastewater from 18 µm filtration, produced about 70 % more sludge compared to the control reactor. In Period (P2), the total sludge production was 0.63 gTS/d for the control reactor and 1.49 gTS/d for the SBR fed filtered wastewater from 33 µm.

Table 15. Sludge, biogas and energy productions during the three study periods

Filter	1.2 µm (P1)	18 µm (P1)	33 µm (P2)	55 µm (P3)	90 µm (P2)	150 µm (P3)	Raw
Primary sludge (gTS/d)	0.46	0.34	1.10	0.73	0.94	0.47	
Biosludge (gTS/d)	0.13	0.20	0.391	0.575	0.568	0.769	0.32 (P1) 0.63 (P2) 0.88 (P3)
Total sludge production (gTS/d)	0.59	0.54	1.49	1.31	1.51	1.24	0.32 (P1) 0.63 (P2) 0.88 (P3)
biosludge yield (gTS/gCOD)	0.58	0.44	0.66	0.59	0.60	0.69	0.38 (P1) 0.56 (P2) 0.60 (P3)
Methane production (L CH_4/d)	0.13	0.11	0.29	0.27	0.29	0.25	0.06 (P1) 0.13 (P2) 0.15 (P3)
Energy production (Wh/d)	1.25	1.06	2.92	2.65	2.94	2.51	1.30 (P1) 1.50 (P2) 1.50 (P3)

Daily flow in P1: Q = 1.5 L/d,
Daily flow P2 & P3: Q = 4.5 L/d

The sludge production was about 136 % higher using filter of 90 µm and about 139 % higher in the using filter of 33 µm, compared to the control reactor. In Period 3, the average sludge productions were 0.88 gTS/d for the control reactor and 1.31 gTS/d for the reactor fed filtered wastewater at 55 µm. The lowest sludge production was in the reactor fed with raw wastewater. The overall sludge production in the reactor treating filtered wastewater at 150 µm and 55 µm increased by 41 %, and 64 %, respectively. The lower sludge production observed in Period 1 could be explained with the lower influent flow, which was only 1.5 L/d compared to 4.5 L/d in Period 2 and 3.

Most of the sludge produced was during primary treatment. As observed in Figure 31, the fraction of biological sludge reduced as the removal of TSS increased from filtration with smaller pores. In the control reactor, the sludge was only biological. The fraction of biological sludge varied from 20 % with the 1.2 µm filter to 62 % with the 150 µm pore size. The production of biosludge was proportional to the COD load into the reactor.

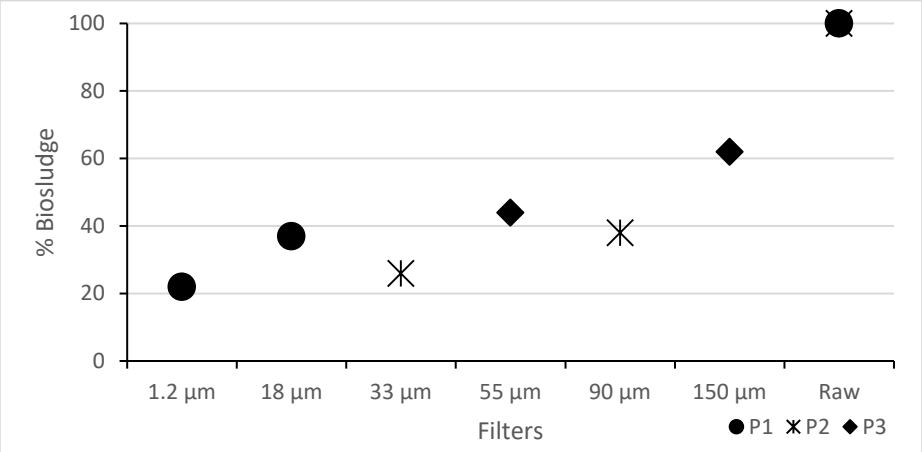


Figure 31. Percentage of biosludge as a function of the filter cloth

The benefit of high TSS removal in primary treatment is the potential of production of biogas. During period P1 the methane production was about 0.06 L CH_4/d in the control reactor and up to 0.13 L CH_4/d in the reactor fed with filtered wastewater. In Period 2 and 3, the methane

productions were between 0.13 – 0.15 L CH_4/d in the control reactor and up to 0.29 L CH_4/d in the reactor fed filtered wastewater.

The biogas production is higher for primary sludge compared to biological sludge, so the higher TSS removal significantly enhances the biogas potential of the sludge. By using filters as primary treatment, one could expect higher biogas production per kg VSS compared to settling tanks. A biomethane potential (BMP) of 318 NmL-CH_4/gDS[6] was measured by Paulsrud *et al.* (2014) with sludge from a filtration plant, while for primary sludge from a conventional sedimentation tank measured a biomethane potential (BMP) of 228 NmL-CH_4/gDS. The difference was due to sludge storage and partly degradation in the sedimentation tank. In addition, a part of the floating sludge, which mainly consist of fat and not captured during gravity settling, while it will be removed in a filtration process. Therefore, the sludge in a filtration process will have a higher energy content compared to settled sludge.

The observed biological yield during the three study periods were between 0.38 – 0.58 gTSS/gCOD (Period 1), 0.56 – 0.66 gTSS/gCOD (Period 2), and between 0.60 – 0.69 g TSS/gCOD (Period 3). Thus, the methane potential of the tests is more or less proportional to the sludge production.

d. Oxygen demand

In biological wastewater treatment, oxygen is for oxidation of organic and nitrogenous compounds and for endogenous respiration (von Sperling, 2007b). Table 16 presents calculated oxygen requirements for the removal of COD (OD_{COD}) and nitrification (OD_{NH4-N}), along with the oxygen recovered in denitrification (OR_{NOx-N}).

The calculations show that the SBRs fed filtered wastewater required less oxygen compared to the control SBR. This is a result of less COD oxidized. In period P1, the oxygen requirement was about 0.38 gO_2/d in the SBR fed with raw wastewater. The oxygen requirement decreased by 37 % in the SBR fed wastewater filtered at 18 µm and by 59 % in the reactor fed wastewater filtered at 1.2 µm. The oxygen required for nitrification was

[6] DS: Dry solids

quite similar in all SBRs so the difference in the total oxygen demand during the biological process was mainly due to the oxygen consumed for degradation of COD.

Table 16. Oxygen requirement for the biological process during the three periods

Filter	1.2 µm (P1)	18 µm (P1)	33 µm (P2)	55 µm (P3)	90 µm (P2)	150 µm (P3)	Raw
OD_{COD} (gO_2/d)	0.08	0.16	0.34	0.66	0.59	0.75	0.33 (P1) 0.69 (P2) 1.07(P3)
OD_{NH4-N} (gO_2/d)	0.13	0.14	0.26	0.53	0.27	0.542	0.14 (P1) 0.30 (P2) 0.56 (P3)
OR_{NO3-N} (gO_2/d)	0.054	0.063	0.052	0.18	0.053	0.19	0.08 (P1) 0.10 (P2) 0.18 (P3)
OD_T (gO_2/d)	0.156	0.242	0.552	0.96	0.806	1.07	0.38 (P1) 0.89 (P2) 1.45 (P3)

OD_{COD}: Oxygen required for the degradation of organic matter;
OD_{NH4-N}: Oxygen required for nitrification
OR_{NOx-N}: Oxygen recovery during denitrification
OD_T: Total oxygen demand

A high level of primary treatment is beneficial for the downstream biological process as it reduces the organic loading and therefore the cost related to oxygen supply. Gori et al. (2013) observed a reduction in energy demand for biological oxidation of 4 to 11 %, while partially removing particulate COD (~ 50 % of TSS) in primary treatment. Rusten (2005b) showed a reduction of about one third of the aeration power consumption in a conventional activated sludge plant upgraded with a Salsnes Filter as primary treatment.

e. Ammonium and nitrate utilization rates

The SBR cycle was a succession of anoxic and aerobic phases, which initiated denitrification and nitrification, respectively. Figure 32 presents specific nitrification rates (SNRs).

SNRs increased with decreasing C/N ratios. The SBR fed filtered wastewater at 1.2 µm (no particulate COD) had the highest SNR. The same observations resulted in Period 2 and 3. The SNRs values were 0.28 gNH4-N/gMLVSS-d in the SBR fed filtered wastewater at 1.2 µm and 0.16 gNH4-N/gMLVSS-d at 150 µm. This indicates that removal of COD in primary treatment improves the conditions for the nitrification process. Mainly availability of oxygen improves nitrification with reduced COD loading. Similar results were in an SBR plant treating shrimp aquaculture wastewater where C/N ratio of 10 gCOD/gN produced best nitrogen removal compared to C/N ratios of 20 gCOD/gN and 30 gCOD/gN (Fontenot *et al.*, 2007). This is due to an increased growth of heterotrophic bacteria.

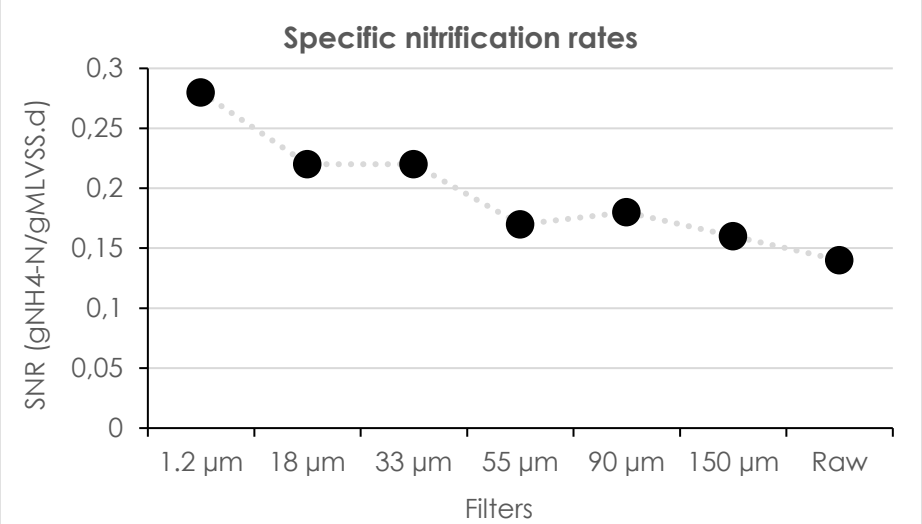


Figure 32. Specific nitrification rate as a function of the filter size

In a system with high C/N ratios heterotrophs compete for the available oxygen, as nitrifiers do not consume COD. Consequently, the fraction of

nitrifiers in the biomass are reduced and causing a significant reduction in nitrification (Ebeling *et al*., 2006; USEPA, 1993).

The denitrification rates (SDNRs) observed during this study were similar to results from the anoxic batch tests. Values ranged from 0.16 to 0.28 gNO_x-N/gMLVSS-d for the highest rate K1, and between 0.06 to 0.1 gNO_x-N/gMLVSS-d for the second-rate K2 (Figure 33). The first rate was lower than the reported value of 0.72 gNOxN/gMLVSS.d (Henze et al., 2008). Unlike the K2 and K3 rates, K1 experiences large variations because measurements of RBCOD utilization are difficult and inaccurate and can change depending on the mixing regime in the reactor. K2 usually remains constant, related to the slower use of SBCOD from the influent and from cell lysis.

Based on the findings there are no significant reduction of the specific denitrification rates with reactors fed filtrates. High COD concentrations are still available in the wastewater after filtration, resulting in denitrification rates independent of COD.

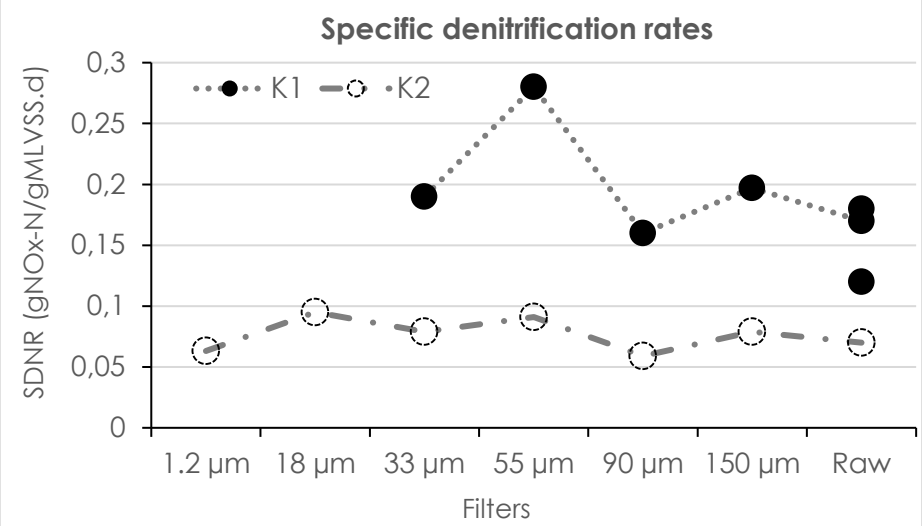


Figure 33. Specific denitrification rates as a function of the filter size

4.2.3. Conclusions

Tests on biological nitrogen removal in SBR-fed wastewaters with various levels of COD were for evaluation of the energy budget.

The filter removed 25 % of the COD at 150 µm and up to 70 % at 1.2 µm. Comparison of SBR performance showed removal efficiencies lower in the reactors fed with filtered wastewater. The biological treatment removed between 65 to 75 % of the COD compared to 70 – 91 % in the control reactors. However, when including the removal efficiency of the primary treatment, similar or relatively higher TSS and COD removal were observed in the SBRs fed with filtered wastewater compared to the control reactor.

The TN removal efficiencies were between 12 % for wastewater filtered with 1.2 µm filters and 34 % for wastewater filtered with 150 µm filter compared to between 59 and 72 % in the control SBRs. However, the overall removal efficiencies (Primary treatment + Biological removal) were similar to the control reactor, except for the SBRs fed with filtered wastewater from filters below 33 µm. A net reduction of TN removal resulted in this case.

The higher removal of TSS before biological treatment resulted in a higher total sludge production and thus an increased biogas production potential. In period P1 the overall sludge production varied from 0.32 gTS/d for the SBR fed raw wastewater to 0.59 gTS/d for the SBR fed 1.2 µm filtered wastewater. In period P2, the total sludge production was 0.63 gTS/d for the control reactor and 1.49 gTS/d for the SBR fed filtered wastewater from 33 µm. In period P3 the average sludge production were 0.88 gTS/d for the control reactor and 1.31 gTS/d for the reactor fed filtered wastewater at 55 µm. The methane production was about 0.06 L CH_4/d in the control reactor and up to 0.13 L CH_4/d in the reactor fed filtered wastewater (P1). In periods 2 and 3 the methane productions were between 0.13 – 0.15 L CH_4/d in the control reactor and up to 0.29 L CH_4/d in the reactor fed filtered wastewater.

The reduction in COD load on the biological process resulted in less energy consumption for aeration.

The specific nitrification rates (SNRs) increased when the C/N ratios decreased. The SBR-fed filtered wastewater at 1.2 µm (no particulate COD) had the highest SNR at values of 0.28 gNH4-N/gMLVSS-d in the SBR fed filtered wastewater at 1.2 µm to 0.16 gNH4-N/gMLVSS-d at 150 µm. The denitrification rates (SDNRs) ranged from 0.16 to 0.28 gNO_x-N/gMLVSS-d for the highest rate K1, and between 0.06 to 0.1 gNO_x-N/gMLVSS-d for the second-rate K2.

An overall assessment of the data indicates that a filter of 33 µm is the optimum choice prior to biological nitrogen removal for the NFR wastewater as it provided both satisfactory organic and TN removal.

A general recommendation is not possible for establishing optimum levels of TSS and COD removal in a system with biological nitrogen removal.

Chapter 5

Verification of the defined particle size cut-off on pilot scale biological nitrogen removal

From laboratory tests conclusions were that 33 µm seemed to be the optimum particle cut-off prior to BNR for NFR wastewater. The 33 µm filter was placed in front of pilot scale commonly used biological processes such as moving bed biofilm reactor (MBBR), membrane bioreactor (MBR) and activated sludge sequencing batch reactor (SBR). A Salsnes Filter model SF 1000 with 33 µm filter cloth was used as a primary treatment to the pilot scale BNR processes (Brinkley *et al.*, 2013; Ødegaard, 2006; Ødegaard, 1999; Ødegaard *et al.*, 1994).

5.1. Moving Bed Biofilm Reactor

5.1.1 Experimental setup and operating conditions

Two parallel trains of MBBRs were used during this experiment, one fed raw wastewater and one fed filtered wastewater. Each MBBR train was composed of two 4 L anoxic reactors (R1, R2) and two 6 L aerobic reactors (R3, R4) in series, resulting in 40 % of anoxic volume and 60 % of aerobic volume. The reactors were made of transparent plastic PVC. The anoxic reactors were equipped with mechanical mixers and the aerobic reactors had diffusers at the bottom of the tank, illustrated in Figures 34 and 35. Nitrified effluent from Reactor 4 was recycled to Reactor 1 at approximately twice the influent flow rate. The biofilm carriers (Kaldnes K1) used to fill the anoxic and aerobic reactors were collected from the pre-denitrification reactor (Reactor 1, Line 2) and the nitrification reactor (Reactor 4, Line 2) at NFR WWTP, respectively.

Temperature, DO and pH were measured daily in all MBBRs. Influent and effluent from the two MBBR trains were analyzed four to six times per week for suspended solids, COD, nitrogen and phosphorus concentrations. The same analyses were for samples from each MBBR compartment two days per week.

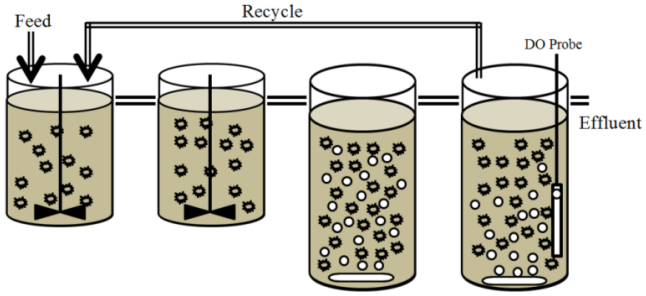


Figure 34. Simplified flowsheet of one MBBR train

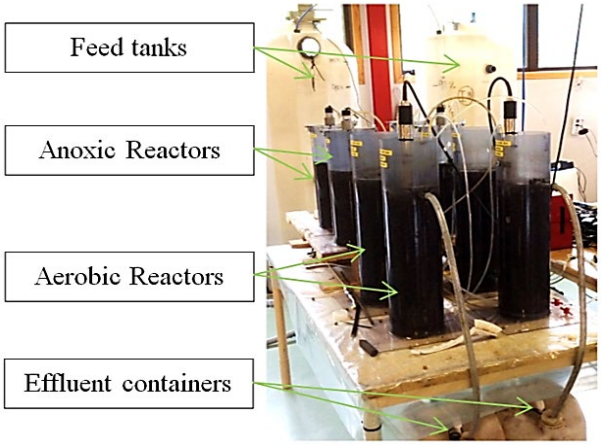


Figure 35. MBBRs pilot scale setup

All reactors were filled 60 % with Kaldnes K1 biofilm carriers, as presented in Table 17, corresponding to a biofilm surface area of 300 m^2/m^3.

Table 17. MBBRs characteristics

Reactor	Operating Mode	Wet Volume (L)	Carrier Fill (%)	Biofilm Surface Area (m^2)
Anoxic 1	Anoxic	4.0	60	1.2
Anoxic 2	Anoxic	4.0	60	1.2
Aerobic 1	Aerobic	6.0	60	1.8
Aerobic 2	Aerobic	6.0	60	1.8

Table 18 summarizes average operating parameters for 11 weeks of operation for the pilot scale MBBRs. The system had an average overall empty bed HRT of 7.1 h and a daily influent flow of 67.2 L.

Table 18. Average operating parameters for the pilot scale MBBRs during the 11 weeks of testing

Parameters	Train A (2 mm screen)	Train B (33 µm sieve)
Feed flow, L/h	2.8 ± 0.6	2.8 ± 0.5
Recirculation flow, L/h	5.5 ± 1.1	5.5 ± 1.2
DO Reactor 1, mg/L	0.13 ± 0.13	0.17 ± 0.15
DO Reactor 2, mg/L	0.03 ± 0.03	0.03 ± 0.03
DO Reactor 3, mg/L	7.0 ± 1.0	6.9 ± 1.0
DO Reactor 4, mg/L	5.9 ± 1.6	6.1 ± 1.4
pH Reactor 1	7.5 ± 0.1	7.4 ± 0.2
pH Reactor 2	7.5 ± 0.2	7.4 ± 0.2
pH Reactor 3	7.4 ± 0.2	7.3 ± 0.3
pH Reactor 4	7.3 ± 0.2	7.1 ± 0.4
Temperature, °C	19.0 ± 2.4	19.2 ± 2.4

5.1.2. Results and Discussion

a. Separation performance

As observed in Figure 36, a filter of 33 µm removed 41 % TSS, 32 % COD, 12 % of TN and 14 % of TP. The performance of the SF 1000 operated without filter mat was quite similar to the bench scale SF 1000 Salsnes Filter, even though the samples were not collected[7] the same way during the two experiments. The MBBR train with filtered wastewater had a COD load of 32 % less compared to the train fed raw wastewater. Any difference in performance between the two trains relates to this difference in COD loading.

Other tests with wastewater from NFR WWTP and 33 µm filter showed TSS removal efficiencies from 43 % to 72 % and from 37 % to 50 % for COD. These results showed clearly that the performance of the filter is strongly

[7] The sample collected with a bucket during the lab-scale experiment while using a grinder pump during this pilot scale study.

dependent on the size distribution of particles in the influent wastewater (Rusten & Lundar, 2006).

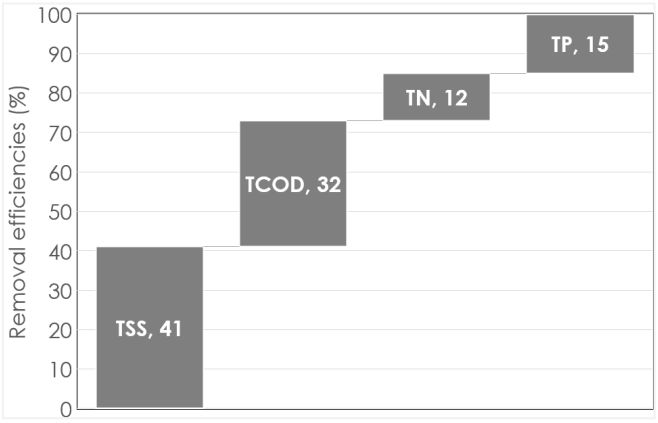


Figure 36. Filter removal efficiencies with 33 µm belt and no filter mat

b. MBBR performance

Table 19 presents wastewater characteristics and average removal efficiencies of the different wastewater parameters for the two MBBR trains. The first effluent samples are pollutant parameter concentrations after MBBR whereas the second effluent is concentrations after 1.2 µm filtration. This was because there was no sludge separation at the effluent.

The MBBR process removed about 41 % of TCOD, 50 % of TN and 18 % of TP (Control reactor) and 41 % of TCOD, 41 % of TN and 3 % of TP in the reactor fed filtered wastewater. When filtration was applied as secondary separation, the MBBR train fed filtered wastewater had COD, TN and TP removal efficiencies of 74 %, 61 % and 65 %, respectively. Higher removal efficiencies were in the MBBR train fed raw wastewater, at 91 % COD, 68 % TN and 73 % TP removals. However, accounting for removals in the filter for Train B, the overall removal efficiencies were similar for the two trains of MBBR for TN, COD and TP. Nitrogen and phosphorus removed were by both particle removal and assimilation in the biomass. The majority of the nitrogen removal, however, were by denitrification.

Table 19. Performance of the pilot scale MBBRs

Parameter	Train A (2mm screen)					Train B (33 µm)					
	Influent (mg/L)	Effluent 1 (mg/L)	Effluent 2 (mg/L)	Rem. eff. 1 (%)	Rem. eff. 2 (%)	Influent (mg/L)	Effluent 1 (mg/L)	Effluent 2 (mg/L)	Rem. eff. 1 (%)	Rem. eff. 2 (%)	Total Rem. eff. (%)
TSS	279 ± 97	204 ± 84	0.0	27	100	167 ± 72	137 ± 75	0.0	18	100	100
TCOD	515 ± 135	302 ± 105	45 ± 10	41	91	355 ± 93	209 ± 88	43 ± 10	41	87	92
Filtered COD	168 ± 53	45 ± 10	45 ± 10	73	73	143 ± 42	43 ± 10	43 ± 10	74	74	74
BSCOD	118 ± 45	--	--	--	--	98 ± 37	--	--	--		--
TN	44 ± 12	22 ± 4	15 ± 6	50	68	39 ± 11	23 ± 6	17 ± 6	41	61	66
Filtered TN	34 ± 10	15 ± 6	15 ± 6	56	56	32 ± 10	17 ± 6	17 ± 6	47	47	47
Organic TN	11.5 ± 4.1	--	14.0 ± 5.1	--	--	7.9 ± 3.9	--	11.1 ± 5.1	--	--	--
Filtered organic TN	1.8 ± 1.8	2.4 ± 1.7	2.4 ± 1.7	--	--	1.9 ± 2.4	2.7 ± 2.0	2.7 ± 2.0	--	--	--
NH_4-N	32 ± 11	1.3 ± 1.3	1.3 ± 1.3	96	96	30 ± 10	1.1 ± 1.1	1.1 ± 1.1	97	97	97
NO_2-N	0.03 ± 0.02	0.21 ± 0.19	0.21 ± 0.19	--	--	0.04 ± 0.05	0.19 ± 0.26	0.19 ± 0.26	--	--	--
NO_3-N	0.41 ± 0.2	10.9 ± 4.2	10.9 ± 4.2	--	--	0.41 ± 0.2	12.5 ± 4.1	12.5 ± 4.1	--	--	--
TP	4.1 ± 1.5	3.34 ± 1.28	1.1 ± 0.6	18	73	3.5 ± 1.1	3.4 ± 0.85	1.1 ± 0.6	3	65	70
Filtered TP	1.7 ± 0.6	1.1 ± 0.6	1.1 ± 0.6	35	35	1.3 ± 0.6	1.1 ± 0.6	1.1 ± 0.6	35	35	35
PO_4-P	1.5 ± 0.6	1.1 ± 0.6	1.1 ± 0.6	25	25	1.0 ± 0.6	0.9 ± 0.6	0.9 ± 0.6	38	38	38

* Results given as averages ± standard deviation, Number of data points varies from 13 to 44

Effluent 1: pollutant concentration after the MBBR treatment

Effluent 2: pollutant concentration after MBBR treatment + filtration at 1.2 µm (secondary separation)

Rem. eff. 1: removal efficiency based on the effluent 1

Rem. eff. 2: removal efficiency based on the effluent 2

Even though a third of the COD was removed from the influent wastewater with the 33 µm filter, the feed for train B contained sufficient COD to achieve nearly the same TN removal as the MBBR train fed raw wastewater.

Both trains had excess of COD for TN removal so higher recycling rates from the aerobic to the anoxic reactors would have increased denitrification and nitrogen removal efficiencies. This indicates that the optimum level of TSS removal in primary treatment with respect to nitrogen removal will vary depending on the influent C/N ratio and the fraction of COD in particulate and dissolved forms. A high C/N ratio will allow for a higher TSS removal in primary treatment, and a higher fraction of dissolved COD will allow for a higher TSS removal.

A Dutch study by Ruiken *et al.* (2013) showed that the majority of the solids removed by 350 µm mesh were paper fibers, which indicated that the particulate COD removed with the filter may be less biodegradable than the particulate COD that passed through to the biological reactors. The nature of cellulose on the filter sludge revealed absence of proteins, shown by removal of only 1 % of the TN over the filter.

In this study, the filter removed 12 % of TN, 31 % of total organic N and 38 % of particulate organic N. The removal of particulate organic N was about 5 % of the VSS removal, indicating that a significant fraction of the solids separated by the filter in the present study were proteins or other nitrogen compounds and not dominated by cellulose as reported by Ruiken *et al.* (2013).

c. Sludge production

The sludge production (primary sludge) in the filter unit was by TSS removal, while the sludge production the MBBR trains was calculated based on effluent TSS. Table 20 shows production of primary and secondary sludge where the average effluent TSS in train A was 204 mg/L and 141 mg/L in train B.

Table 20. Sludge production, methane production and energy gain in the pilot scale MBBRs

Parameter	Train A (2 mm)	Train B (33 µm)
Primary sludge production (g TSS/d)	--	7.8 ± 4.3
biosludge production (g TSS/d)	13.7 ± 5.4	9.5 ± 3.6
Total sludge production (g TSS/d)	**13.7 ± 5.4**	**17.3 ± 5.0**
Biosludge yield (g TSS/g COD)	**0.43 ± 0.21**	**0.45 ± 0.27**
Methane production (L CH_4/d)	2.78 ± 1.13	3.53 ± 1.05
Energy gain (Wh/d)	27.8 ± 11.3	35.3 ± 10.5

As observed in Table 20, more sludge came from Train B combining filter and biological treatment. The average sludge production in Train A was 13.7 gTS/d compared to 17.3 gTS/d in Train B, corresponding to 26 % higher sludge production in Train B compared to train A. The advantage of using filter was removal of about 32 % COD during primary separation, which is beneficial in terms of energy production and savings for aeration. During this study, the methane production was about 2.78 L CH_4/d in the control reactor and 3.53 L CH_4/d in the MBBR fed with filtered wastewater. Halim (2012) estimated the energy content of primary sludge to be between 4.2 and 4.5 kWh/kg DS and from 3.5 to 3.8 kWh/kg DS for the secondary sludge.

The average sludge yield was 0.43 g TSS/g COD removed in the control reactor (Train A), and it was slightly higher in the reactor fed with filtered wastewater at 0.45 gTSS/gCOD. The observed yield in Train B is probably due to the lower influent TSS compared to the control reactor, where more accumulation of TSS occurred. The organic load was 35.3 gCOD/d in the control reactor and 24.2 gCOD/d in the reactor fed with filtered wastewater. The higher organic load resulted in higher sludge production. On the other hand, longer retention time of the sludge in the

system (SRT) resulted in more endogenous respiration and decrease in sludge yield.

d. Biomass on biofilm carriers

Table 21 shows the amount of biomass on the carriers in each reacto. In Train A the biomass concentrations were 6.2 and 6.9 gTS/L in the first and second anoxic reactors, respectively, and 4.4 gTS/L in the aerobic reactors. The biomass concentrations were slightly lower in Train B at 4.8 and 6.5 gTS/L in the first and second anoxic reactors and 3.6 – 4.2 gTS/L in the 2 aerobic reactors. The lower biomass concentrations in Train B are because of lower organic loadings of about 31 % compared to the control reactor.

The biomass concentrations were lower in the aerobic reactors in both MBBR trains, because a higher COD removal in the anoxic reactors and thus more growth on the carriers. The remaining reduced COD entering the aerobic reactors thus resulted in less growth on the carriers. The aeration in the aerobic reactors may also result in more detachment of biomass compared to the anoxic reactors.

SRT is a parameter that is widely used for activated sludge systems, however, not for biofilm processes. In biofilm systems, different parts of the biofilm have different age. However, for the two MBBRs the average sludge age, based on the daily biomass production and the total amount of biomass attached on the biofilm carriers, were 7.7 days (Train A) and 9.6 days (Train B). The longer SRT in train B reduces the sludge and provides better conditions for nitrification especially at low temperatures during the winter.

Table 21. Biomass on biofilm carriers

	Train A (2 mm)				Train B (33 µm)			
Reactor	**mgTS/carrier**	**gTS/m²**	**gTS/L**	**gTS/reactor**	**mgTS/carrier**	**gTS/m²**	**gTS/L**	**gTS/reactor**
A1/B1 (anoxic)	10.6	20.7	6.21	24.8	8.2	16.0	4.79	19.2
A2/B2 (anoxic)	11.8	23.0	6.91	27.7	11.0	21.5	6.45	25.8
A3/B3 (aerobic)	7.6	14.7	4.42	26.5	7.1	13.9	4.17	25.0
A4/B4 (aerobic)	7.5	14.7	4.40	26.4	6.1	11.9	3.56	21.4

e. Oxygen demand

Table 22 shows the average daily consumption of oxygen calculated in Train A and B, based on removed COD and nitrified ammonia.

Table 22. Oxygen demand for COD removal and nitrification

Parameter	Train A (2 mm)	Train B (33 µm)
OD_{COD} (gO_2/d)	7.9 ± 4.3	6.2 ± 2.9
OD_{NH4-N} (gO_2/d)	9.8± 4.2	9.1 ± 3.3
OR_{NO3-N} (gO_2/d)	6.3 ± 2.8	5.9 ± 2.5
Total OD (gO_2/d)	**10.8 ± 5.0**	**8.3 ± 4.1**

OD_{COD}: Oxygen required for the degradation of organic matter;
OD_{NH4-N}: Oxygen required for nitrification
OR_{NOx-N}: Oxygen recovery during denitrification
OD_T: Total oxygen demand

The MBBR train fed raw wastewater consumed 10.8 gO_2/d while Train B fed filtered wastewater consumed 8.3 gO_2/d. The two MBBR trains consumed approximately the same amount of oxygen for nitrification, and recovered similar amount of oxygen during denitrification. The difference in oxygen demand is because the lower oxygen demand for Train B due to the removal of a third of the influent COD by the filter. The reduction in oxygen demand was about 30 % compared to the control reactor. As aeration represents the largest energy consumption in biological wastewater treatment, the reduction in oxygen demand and aeration will thus result in a significant reduction in energy demand of the plant.

f. Nitrification and denitrification rates

Table 23 shows the nitrification rates observed in the different aerobic reactors. Observations during this study showed that nitrification rates varied greatly in the first aerobic reactors, especially noted in Train A where the reactor received significantly more COD than Train B. The scattered rates obtained in the first aerobic reactors were probably due to the variations of COD loads in the influent. Hem *et al.* (1994) investigated the impact of organic load for nitrification.

Table 23. Nitrification in the aerobic reactors 3 and 4 of the two MBBRs

Reactor and parameter	Train A (2 mm) Avg. ± St.D	Train B (33µm) Avg. ± St.D
Reactor 3:		
Ammonium load (gNH_4-N/m^2-d)	1.18 ± 0.45	1.13 ± 0.31
Reactor DO (mg/L)	6.95 ± 1.09	6.66 ± 0.80
Reactor NH_4-N (mg/L)	2.48 ± 2.64	1.34 ± 0.87
Removal rate (gNH_4-N/m^2-d)	0.87 ± 0.25	0.97 ± 0.25
Reactor 4:		
Ammonium load (gNH_4-N/m^2-d)	0.31 ± 0.36	0.28 ± 0.41
Reactor DO (mg/L)	5.94 ± 1.54	6.07 ± 1.40
Reactor NH_4-N (mg/L)	0.77 ± 1.29	0.20 ± 0.23
Removal rate (gNH_4-N/m^2-d)	0.22 ± 0.22	0.13 ± 0.09

All loads and rates were adjusted to 20°C (Θ: 1.103), Number of data points: 21

The results showed that a COD load of 2 – 3 g bCOD[8]/m^2-d resulted in a nitrification rate of 0.3 – 0.8 g NH_4-N/m^2-d while a COD load of 1 – 2 g bCOD/m^2-d resulted in a nitrification rate in the range of 0.7 – 1.2 g NH_4-N/m^2-d. Figure 37 presents the lowest nitrification rates with high ammonium loads as observed in Train A. The latter observation was in accordance with the results obtained during the lab-scale SBRs where removal of particulates from the influent enhanced nitrification. Overall, nitrification rates obtained during this study were close to the values of Hem *et al.* (1994) reported as 0.1 - 1.3 g NH_4-N/m^2-d (with a DO concentration of 6 mg/L).

[8] Biodegradable COD

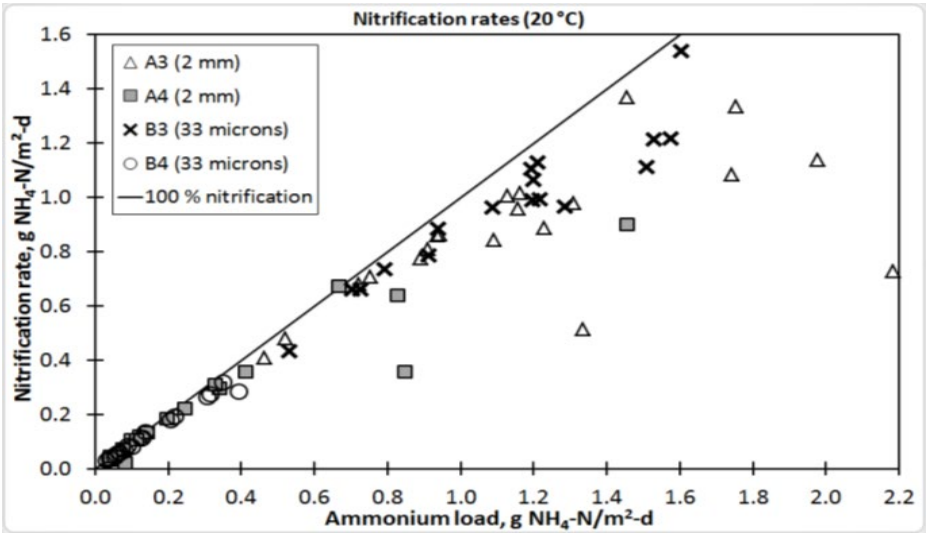


Figure 37. Nitrification rates versus ammonium loads in the two MBBRs. A3 and A4 are Reactors 3 and 4 in Train A, while B3 and B4 are reactors 3 and 4 in Train B. All loads and rates adjusted to 20 °C.

Table 24 shows the measured denitrification rates in the anoxic reactors. All loads and rates were temperature adjusted to 20 °C by a correction factor of 1.07 (Rusten *et al.*, 1995).

Table 24. Denitrification in the anoxic reactors for both Train A and Train B

	Train A (2 mm)	**Train B (33 µm)**
Reactor and parameter	**Avg. ± St.D**	**Avg. ± St.D**
Influent COD (gTCOD/d)	34.9 ± 12.9	24.2 ± 8.7
Influent bCOD (g bCOD/d)	28.4 ± 12.0	19.2 ± 8.0
Influent bsCOD (g bsCOD/d)	8.7 ± 4.4	7.0 ± 3.5
DO recycled	5.9 ± 1.1	6.1 ± 1.2
NOx-N load (gNO_x-N/m^2-d)	1.17 ± 0.51	1.36 ± 0.58
C/N removed (g sCOD/gNO_x-N)	4.7	3.35
Denitrification rate (gNO_x-N/m^2-d)	0.77 ± 0.51	0.85 ± 0.52

b: biodegradable
bs: biodegradable soluble
All loads and rates adjusted to 20 °C (Θ: 1.07)
Concentration after feed and recycle flows have been mixed
Number of data points: 6 to 21

The influent wastewater to Train B had low soluble COD. The influent concentration of biodegradable soluble COD in Train B ranged from 0.65 to 13.8 g bsCOD/d, whereas for Train A the values were between 0.63 and 18.4 g bsCOD/d. Henze & Harremoes (1990) stipulated that the biodegradability of substrates strongly influences the denitrification rates. Overall analysis of denitrification rates, however, showed rates 10 % higher in Train B than in Train A. The average value of the SDNR is 0.77 gNOx-N/m^2.d in Train A and 0.85 gNOx-N/m^2.d in Train B. Consequently, the removal of particulate organic matter with a 33 µm filter did not have any negative impact on denitrification.

5.1.3. Conclusions

The objective of this study performed at the NFR WWTP was to evaluate the effect of removal of particulate organic matter with a 33 µm filter on the performance of pilot scale MBBRs with nitrogen removal.

A 33 µm filter operated without filter mat, resulted in the removal of 41 % TSS. This corresponded to removal of 31 % COD, 12 % TN and 14 % TP.

The MBBR process removed about 41 % of TCOD, 50 % of TN and 18 % of TN (Control reactor) and 41 % of TCOD, 41 % of TN and 3 % of TP in the reactor fed filtered wastewater. By applying filtration as secondary separation, the MBBR train fed with filtered wastewater had COD, TN and TP removal efficiencies of 74 %, 61 % and 65 %, respectively. Higher removal efficiencies resulted in the MBBR train fed raw wastewater, at 91 % COD, 68 % TN and 73 % TP. However, when the removals in the filter accounted for in Train B, the overall removal efficiencies were similar for the two trains of MBBR for TN, COD and TP.

Train B produced more sludge by combining filter and biological treatment. The average sludge production in Train A was 13.7 gTS/d compared to 17.3 gTS/d in Train B, corresponding to 26 % higher sludge production in Train B compared to train A. The advantage of using filter was removal of about 32 % of the COD during primary separation, which is beneficial in terms of energy production and savings in aeration. During this study, the methane production was about 2.78 L CH_4/d in the control reactor and 3.53 L CH_4/d in the MBBR fed with filtered wastewater.

The MBBR train fed raw wastewater consumed 10.8 gO_2/d while Train B, fed filtered wastewater, consumed 8.3 gO_2/d. The reduction in oxygen demand was about 30 % compared to the control reactor.

The overall nitrification rates during this study were between 0.1 and 1.3 g NH_4-N/m^2-d. The analysis of the denitrification rates, however, showed that rates were 10 % higher in Train B than in Train A. The average value of the SDNR is 0.77 gNOx-N/m^2.d in Train A and 0.85 gNOx-N/m^2.d in Train B. Consequently, the removal of particulate COD with a 33 µm filter did not have any negative impact on denitrification.

5.2. Membrane bioreactors

MBR is the combination of biological treatment, such as activated sludge, and membrane filtration where the membrane primarily serves to replace the clarifier in the wastewater treatment plant (Gunder & Krauth, 1998). Membranes overcome several limitations of the conventional activated sludge such as settling problems and low mixed liquor concentrations (Jenkins *et al.*, 2004; Casey *et al.*, 1995). However, poor characteristics of sludge in a reactor, such as sludge bulking due to excessive development of filamentous bacteria, can have a huge impact on membrane fouling which affects membrane performance (Meng *et* al., 2007; Meng & Yang, 2007). The MBR effluent is completely free of SS (Leiknes & Ødegaard, 2006).

5.2.1. Experimental setup and operating parameters

Figures 38 and 39 depict the two pilot-scale MBRs with nitrogen removal, which operated in parallel during this experiment. MBR train A was fed raw wastewater and train B wastewater filtered with a 33 µm filter cloth (SF 1000).

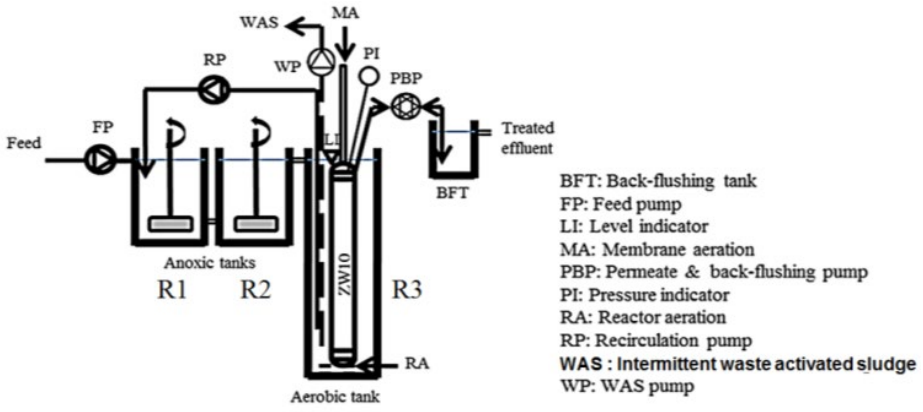


Figure 38. flowsheet of the pilot scale MBR

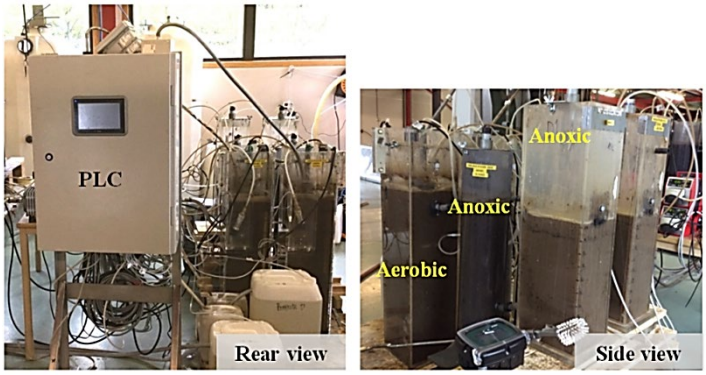


Figure 39. Pilot scale MBR setup

Each MBR train was composed of two anoxic reactors of 10 L each (R1 and R2), equipped with a mechanical mixer and one aerobic reactor of 25 L (R3) with a submerged hollow fiber membrane ZeeWeed-10 at 40 nm nominal pore size (ZW-10, Zenon Environmental Systems Inc., Canada). Nitrified activated sludge was recycled from Reactor 3 to Reactor 1 at twice the influent wastewater flowrate. The membrane ZW-10 was operated at normal flow for about 9.5 minutes and backwashed at twice the normal flow for about half a minute. A PLC controlled feed-, recycle- and permeate- pumps and continuously recorded the transmembrane pressure (TMP) and DO in the aerobic reactor.

The MBRs operated at an influent flow rate of 5 L/h and a recycle flow rate of 10 L/h. The hydraulic retention time (HRT) was 9 hours for both trains. The MLSS in the anoxic reactors was about 5 g/L, while the MLSS in the aerobic reactors was 7 g/L. The operating temperature varied between 16 °C and 21 °C during the experimental period, pH was maintained around 7 and the oxygen in the aerobic reactors was around 4 mg/L. Table 25 summarizes the operating parameters of the MBR systems.

Table 25. MBRs operating parameters

Parameter	Train A (2 mm)	Train B (33 µm)
Feed flow (L/h)	5.2 ± 0.2	5.2 ± 0.2
Recirculation flow (L/h)	10.3 ± 0.3	10.1 ± 0.4
Permeate flow (L/h)	5.9 ± 0.2	5.9 ± 0.2
Backflush flow (L/h)	~ 12	~ 12
MLSS, Reactor 1, 2 (mg/L)	5165 ± 1021	4724 ± 935
MLSS, Reactor 3 (mg/L)	7196 ± 1263	6862 ± 1188
SRT (d)	13.7 ± 2.7	16.8 ± 3.4
Temperature, Reactor 1 (°C)	18.7 ± 2.3	18.6 ± 2.3
Temperature, Reactor 2 (°C)	18.7 ± 2.4	18.6 ± 2.3
Temperature, Reactor 3 (°C)	18.8 ± 2.4	18.7 ± 2.3
pH, Reactor 1	7.3 ± 0.1	7.3 ± 0.2
pH, Reactor 2	7.3 ± 0.1	7.3 ± 0.1
pH, Reactor 3	7.1 ± 0.2	7.0 ± 0.3
DO, Reactor 1, 2 (mg/L)	< 0.02	< 0.02
DO, Reactor 3 (mg/L)	3.8 ± 1.4	4.1 ±1.2
Mixer speed, Reactor 1 (rpm)	216 ± 33	216 ± 46
Mixer speed, Reactor 2 (rpm)	226 ± 44	219 ± 40

5.2.2. Results and Discussion

a. Separation performance

The MBR tests were in parallel with the MBBR tests and with the same wastewater in the two MBR trains. As shown in Figure 40, the 33 µm-filter had a removal efficiency of 42 % TSS, 33 % COD, 12 % TN and 14 % TP. The performance of the filter was consistent during the experimental period.

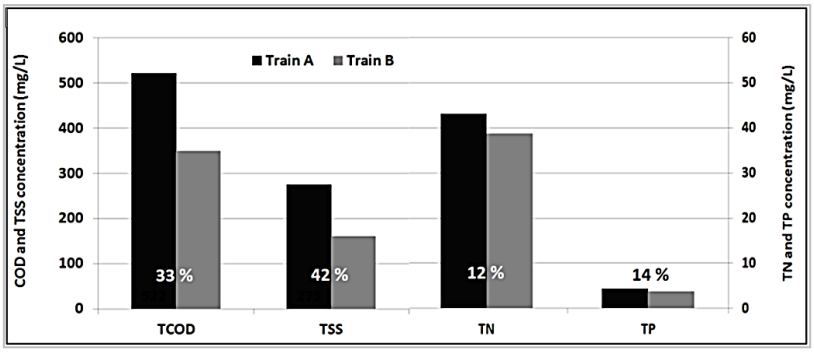


Figure 40. Feed characteristics and filter removal efficiencies

b. MBR performance

Table 26 summarizes the wastewater characteristics and removal efficiencies. The two MBR systems produced effluents free of particles so the measured values of phosphate and nitrogen are all soluble. The assessment of the MBRs' performance showed a removal efficiency at nearly the same level for both trains. For Train B, if the filter performance is not considered, the MBR removed about 90 % COD, 69 % TN and 78 % TP. However, when the filter removal efficiency is accounted for, the overall removal efficiencies were 94 % COD, 80 % TP and 73 % TN, similar to train A.

The results show that at the present level of TSS and COD removal in primary treatment, there was still sufficient COD to achieve a high level of TN removal in both trains. In order to see any difference in performance between the trains, an increase in recycle of nitrified mixed liquor or additional nitrogen could be added to the wastewater. The high TP removal indicates enhanced biological phosphorus removal caused by anaerobic conditions in the second anoxic reactor when denitrification was completed.

Other studies have shown that performance of MBR varies. However, there are many reported examples of excellent performance, such as 95 – 98 % for COD and 80 – 84 % for TN (Galil *et al.*, 2009; Bracklaw *et al*., 2007; Lobos *et al*., 2006; Jiang *et al*., 2004).

Table 26. Concentrations and removal efficiencies of the two MBRs

Parameter	Train A (2 mm)			Train B (33 µm)			
	Influent (mg/L)	Effluent (mg/L)	Total Rem. (%)	Influent (mg/L)	Effluent (mg/L)	MBR Rem. (%)	Total Rem. (%)
Total COD	522 ± 134	32.4 ± 7.2	**94 ± 1**	349 ± 88	32.1 ± 7.5	**90 ± 2**	**94 ± 1**
Soluble COD	168 ± 47	29.7 ± 7.3	**81 ± 5**	145 ± 35	29.1 ± 7.7	**82 ± 5**	**82 ± 6**
TSS	275 ± 99	0.27 ± 1.46	**100**	161 ± 69	0.04 ± 0.21	**100 ± 0**	**100**
VSS	216 ± 67	0.00 ± 0.00	**100**	128 ± 53	0.00 ± 0.00	**100 ± 0**	**100**
TN	43.2 ± 12.0	11.3 ± 3.5	**74 ± 6**	38.7 ± 11.8	11.3 ± 2.5	**69 ± 11**	**73 ± 9**
NH_4-N	31.5 ± 10.5	0.24 ± 0.74	**99 ± 2**	30.1 ± 9.9	0.32 ± 1.37	**99 ± 4**	**99 ± 4**
NO_3-N	0.41 ± 0.15	8.63 ± 3.0	**---**	0.40 ± 0.15	10.0 ± 3.2	**---**	**---**
NO_2-N	0.03 ± 0.01	0.14 ± 0.22	**---**	0.03 ± 0.05	0.13 ± 0.19	**---**	**---**
TP	4.26 ± 1.51	0.82 ± 0.62	**80 ± 14**	3.70 ± 1.25	0.74 ± 0.55	**79 ± 17**	**81 ± 15**
PO_4-P	1.51 ± 0.60	0.57 ± 0.44	**61 ± 28**	1.09 ± 0.65	0.55 ± 0.52	**65 ± 26**	**65 ± 26**

Number of samples: 16 - 42

c. Membrane performance

The performance of membranes was by monitoring of TMP over time at a constant flux, which correlates with the fouling rate (Leiknes *et al*., 2006). The ZeeWeed-10 membrane used in this study intended to operate at constant flux. In practice, however, the permeate flux ranged from 5.8 to 7.0 LMH (~ 6.4 LMH) in Train A and between 5.7 – 6.6 LMH (~ 6.3 LMH) in Train B. Figure 41 shows fluctuations of the TMP during the study period.

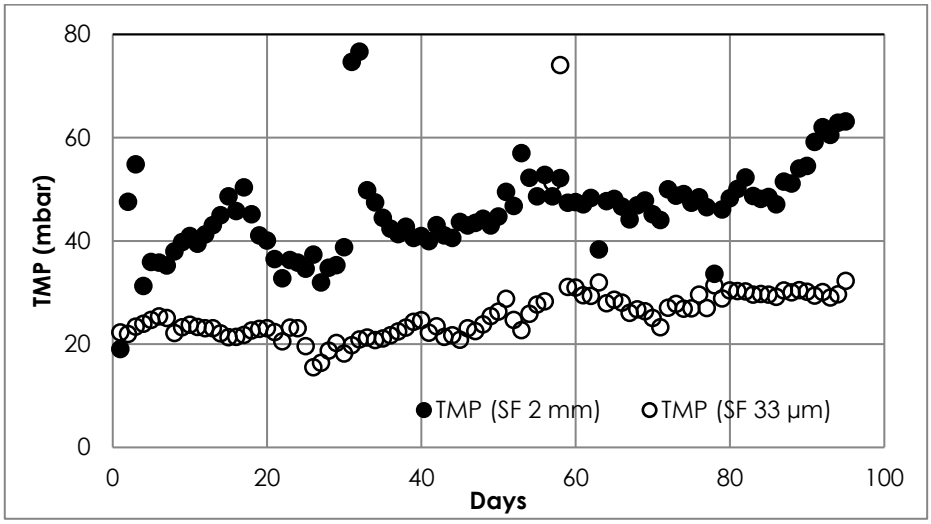


Figure 41. Fluctuation of the transmembrane pressure during the test

TMP was higher for the membrane receiving raw wastewater with an average TMP of 46 ± 9 mbar. For the train treating filtered wastewater, the average TMP value was 26 ± 7 mbar. The fouling rate was higher in the treatment of raw wastewater compared to the filtered feed. The MLSS in the aerobic reactors were nearly the same, supporting the explanation of wastewater characteristics causing the difference in TMP. The TMPs were relatively low for both raw and filtered wastewater, however, and no chemical cleaning of the membrane was necessary throughout the experiments. The TMPs were consistently below 300 mbar, the maximum recommended by the membrane supplier.

d. Sludge production

Table 27 summarizes the sludge production from both MBR trains, in addition to the calculated biological sludge yield and estimated methane production for all types of sludge.

Table 27. Sludge production during the pilot scale MBRs study

Parameter	Train A (2 mm)	Train B (33 µm)
Filter sludge production (gTSS/d)	--	14.5 ± 8.1
Biological sludge production (gTSS/d)	21.3 ± 4.6	16.4 ± 3.6
Total sludge production (gTSS/d)	**21.3 ± 4.6**	**31.2 ± 8.4**
Biological sludge yield (gTSS/gCOD)	**0.33**	**0.40**
Methane production (L CH_4/d)	**4.32 ± 0.93**	**6.35 ± 1.71**
Energy gain (Wh/d)	**43.2 ± 9.34**	**63.5 ± 17.1**

The sludge production in Train A was only composed of biological sludge and was about 21.3 gTS/d. In Train B the total sludge production was 31.2 gTSS/d and consisted of the filter (primary) sludge and biological sludge. The production of biological sludge in Train B was 16.4 gTSS/d, while the filtered sludge was 14.5 gTSS/d. The total amount of sludge produced in Train B was about 46 % higher compared to that of Train A because of the additional sieve sludge. The biological sludge yields were 0.33 gTSS/gCOD and 0.40 gTSS/gCOD for Train A and Train B, respectively.

Both primary sludge and biological sludge produced methane in anaerobic digestion (Appels *et al.*, 2008). Several studies have shown that primary sludge has a higher biogas production potential (BMP) compared to biological sludge. The estimated energy production for the MBR train without primary filtration was about 4.32 L CH_4/d /d, while the energy production was 6.35 L CH_4/d in train B with primary filtration. The energy gain in this study was 47 % higher when filtration was applied as primary treatment upstream the biological processes.

e. Oxygen demand

The oxygen demand for COD removal and nitrification was calculated and presented in Table 28. Oxygen recovered in denitrification was also calculated.

Table 28. Oxygen demand for the two MBR Trains

Parameter	Train A (2 mm)	Train B (33 µm)
OD_{COD}	19.5 ± 6.5	12.2 ± 3.9
OD_{NH4-N}	19.6 ± 7.0	18.7 ± 6.8
OR_{NO3-N}	10.2 ± 4.4	9.5 ± 3.9
OD_T	**28.9 ± 8.3**	**21.4 ± 6.0**

OD_{COD}: Oxygen required for the degradation of organic matter (gO_2/d)
OD_{NH4-N}: Oxygen required for nitrification (gO_2/d)
OR_{NOx-N}: Oxygen recovery from denitrification (gO_2/d)
OD_T: total oxygen demand (gO_2/d)

The oxygen requirement was about 35 % higher for the MBR treating raw wastewater in Train A compared to MBR treating filtered wastewater in Train B. Thus, removal of TSS and COD from the influent wastewater is beneficial in terms of cost savings for reduced energy demand for aeration. van Nieuwenhuijzen (2000) observed that removal of particulate COD resulted in a reduction of the overall energy demand and consequently operational cost.

f. Nitrification and denitrification

Full nitrification was for both MBR trains during this study. Table 29 show the ammonium load and the specific nitrification rate for the two MBRs.

Table 29. Nitrification in the aerobic reactor of the two MBRs

	Train A (2 mm)	Train B (33 µm)
Nitrification capacity ($mgNH_4$-N/L)	14.0 ± 5.4	12.4 ± 4.9
Ammonium load (gNH_4-N/gMLVSS.d)	1.45 ± 0.38	2.03 ± 0.47
Effluent ammonium ($mgNH_4$-N/L)	0.24 ± 0.75	0.32 ± 1.37
Specific nitrification rate (gNH_4-N/gMLVSS-d)	0.14 ± 0.04	0.16 ± 0.06

Number of samples: 10 - 42

The nitrification rates ranged between 0.088 and 0.204 $gNH_4N/gMLVSS$-d in the MBR-control and 0.072 – 0.265 $gNH_4N/gMLVSS$-d in the MBR fed 33 µm filtered wastewater. The reduced nitrification rates observed in Train A was most likely due to increased competition for COD between nitrifying and heterotrophic bacteria (Strauss, 2000). Therefore, filtration was beneficial for nitrification. The SNR in the MBR-control was similar to the value observed in the lab-scale SBR. However, the rate was slightly lower in the MBR fed 33 µm filtered wastewater at 0.20 $gNH_4N/gMLVSS$-d.

Table 30 shows the specific denitrification rates (SDNRs) in the two MBR trains. The rates varied from 0.013 to 0.060 gNO_x-N/g MLVSS-d in Train A, whereas the rates were 0.039 – 0.415 gNO_x-N/g MLVSS-d in Train B. The SDNR was higher in the MBR treating filtered wastewater. Compared with previous data, the rates were low. Recycled DO from the aerobic reactor caused part of the COD to be unused for denitrification. In the control reactor 2.8 gCOD/d was oxidized compared to 3 gCOD/d in the MBR fed filtered wastewater. Despite the oxidation of some COD, the system still had enough organics to conduct denitrification. The C/N ratios were 28.1 gCOD/gN and 13.8 gCOD/gN in the MBR control and MBR fed filtered wastewater, respectively. Therefore, the limited nitrate load might explain the lowest rates.

Table 30. Denitrification in the two MBR trains

	Train A (2 mm)	**Train B (33 µm)**
Nitrate load (gNO_x-N/d)	2.23 ± 0.74	2.83 ± 1.03
TCOD load (gCOD/d)	65.7 ± 16.6	42.3 ± 10.6
$TCOD_b$ load (gCOD/d)	55.1 ± 16.1	34.5 ± 10.2
$sCOD_b$ load (gCOD/d)	16.9 ± 5.4	14.3 ± 4.9
DO recycled (mg/L)	3.8 ± 1.4	4.1 ± 1.2
COD_{DO} (gCOD/d)	2.81 ± 1.02	2.47 ± 1.14
C/N ratio (gCOD/gN)	28.1 ± 11.6	13.8 ± 5.8
C/N ratio ($gCOD_b$/gN)	23.4 ± 10.3	11.5 ± 5.0
C/N ratio (g sCOD/gN)	6.3 ± 2.7	4.0 ± 2.4
SDNR (gNO_x-N/gMLVSS-d)	0.043 ± 0.013	0.095 ± 0.085

b: biodegradable

DO: dissolved oxygen

COD_{DO}: COD oxidized due to recycled DO

5.2.3. Conclusions

The goal of this study was to evaluate the influence of particle removal with 33 µm filter on BNR in a pilot scale MBR with two MBR trains. The first reactor Train A treated raw wastewater (2 mm) and the second MBR Train B treated filtered wastewater.

The characteristic of the influent wastewater used during this study indicated a medium strength wastewater. Separation with 33 µm filter resulted in removal of 42 % TSS, 33 % COD, 12 % TN and 14 % TP. These values are consistent with previous results with a 33 µm filter.

The assessment of the MBRs' performance showed a similar removal efficiency. If the filter performance is not considered for Train B, the MBR removed about 90 % COD, 69 % TN and 78 % TP. However, when the filter removal efficiency is taken into account, the overall removal efficiencies were 94 % COD, 80 % TP and 73 % TN, and similar to train A.

The TMP was higher for the membrane receiving raw wastewater, with an average TMP of 46 ± 9 mbar. For the train treating filtered wastewater the average TMP was 26 ± 7 mbar. The fouling rate was higher for treatment of raw wastewater compared to the fine filtered feed.

In Train A the sludge production was about 21.3 gTS/d compared to 31.2 gTSS/d in Train B. The estimated energy production as methane gas for the MBR train without primary filtration was about 4.32 L CH_4/d /d, while the energy production was 6.35 L CH_4/d in train B with primary filtration. The energy gain in this study was 47 % higher when filtration was applied as primary treatment upstream the biological processes. In addition, the oxygen requirement was about 35 % higher for the MBR treating raw wastewater compared to the MBR treating filtered wastewater.

Nitrification rates ranged between 0.088 and 0.204 gNH_4N/gMLVSS-d in the MBR control and between 0.072 – 0.265 gNH_4N/gMLVSS-d in the MBR fed filtered wastewater. The specific denitrification rates (SDNRs) observed in the two MBR trains varied from 0.013 to 0.060 gNO_x-N/g MLVSS-d in Train A, whereas the rates were 0.039 – 0.415 gNO_x-N/g MLVSS-d in Train B. Overall, the removal of particulate COD with a 33 µm filter did not have a negative effect on the MBR process with nitrogen removal.

5.3. Sequencing batch reactors

The objectives of this study were to evaluate the impact of filtration as primary treatment on pilot-scale SBRs with nitrogen removal and to investigate the influence of temperature and SRT on biological processes.

The pilot scale studies were over two periods. During the first period (P1) three SBRs operated in parallel. The first SBR (R1) received raw wastewater, the second (R2) and third SBR (R3) were fed 33 µm filtered wastewater. The first and second SBRs operated at a temperature of 17 °C, while the third SBR (R3) at 8 °C. During a second period two SBRs, R4 and R5, operated at a reduced SRT compared to the first period. R4 received raw wastewater while R5 received 33 µm filtered wastewater.

5.3.1. Experimental setup and operating conditions

Five transparent PVC tanks each of 10 L acted as reactors during this study. Three SBRs were rectangular (R1, R2, R3) whereas R4 and R5 were cylindrical (Figure 42). The pilot scale SBRs were fully automated by magnetic valves, model 120 Watson-Marlow peristaltic pumps (Cornwall, UK) for the circulation of wastewater and sludge and a digital timer switch (Cotech, UK) to automatically determine the different cycles of the SBRs. Daily calibrations of all pumps and tubing ensured correct volumes in and out of the reactors.

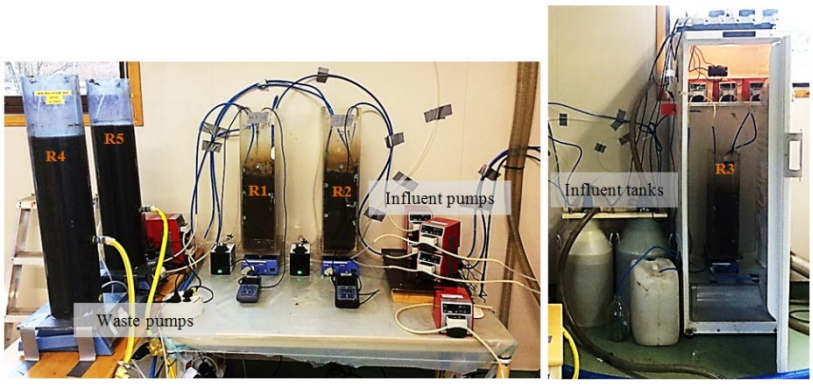


Figure 42. Pilot scale SBRs setup

R1 and R2 started with static fill during Period 1 for 15 minutes, an anoxic phase with mechanical mixing of 345 minutes, 240 minutes aerobic phase, 70 minutes settling phase and 50 minutes of effluent withdrawal. R3 operated with 15 minutes static fill, 285 minutes of anoxic phase, 300 minutes aerobic phase 70 minutes settling phase and 50 minutes effluent withdrawal. The duration of the anoxic phase reduced in R3, while the aerobic phase increased, compared to R1 and R2. In Period 2, R4 and R5 operated similar to R1 and R2 in the first period. Table 31 summarizes operating conditions of the five SBRs.

Table 31. Operating parameters of the pilot scale SBRs

Parameter	Period 1			Period 2	
	R1 - Raw[a]	R2 - 33 µm	R3 - 33 µm	R4 - Raw	R5 - 33 µm
Reactor volume (L)	10	10	10	10	10
Number of cycles per day	2	2	2	2	2
Cycle length (h)	12	12	12	12	12
Influent flow rate (L/d)	12	12	12	12	12
HRT[b] (d)	0.83	0.83	0.83	0.83	0.83
pH	7 – 8	7 – 8	7 – 8	7 – 8	7 – 8
DO, aerobic tank (mg/L)	5.6	5.6	8.2	6	8
Temperature (˚C)	15 – 19 (17)	15 – 19 (17)	5 – 12 (8)	15 – 19 (17)	15 – 19 (17)
MLSS[c] (mg/L)	2620	2460	2230	1870	1560
MLVSS[d] (mg/L)	1920	1660	1550	1410	1060
MLSS wasted per day (g/d)	1.72	1.57	1.48	2.67	2.07
SVI[e] (mL/g MLSS)	81.2	81.7	94.8	83.2	89.9
Total SRT[f] (d)	12.5	12.8	12.4	5.7	6.0
Aerobic SRT[g] (d)	5.1	5.3	6.3	2.3	2.5
Total F/M [h] (g TCOD/g MLVSS-d)	0.142	0.125	0.143	0.215	0.322
Aerobic F/M (g TCOD/g MLVSS-d)	0.425	0.374	0.343	0.644	0.967
Applied C/N (g TCOD/g TN)	11.6	9.35	8.70	11.6	9.97
Applied C/N (g sCOD/g TN)	3.48	3.97	3.17	0.30	3.27

[a] Raw: raw wastewater
[b] HRT: Hydraulic retention time (which is the volume of the reactor divided by the influent flow rate)
[c] MLSS: Mixed liquor suspended solids
[d] MLVSS: Mixed liquor volatile suspended solids
[e] SVI: Sludge volume index
[f] SRT: Sludge retention time (= mass of sludge in reactor/mass of sludge wasted per day)
[g] Aerobic SRT [= Total SRT x (aerobic cycle duration/total cycle duration)]
[h] F/M: Food to biomass ratio

5.3.2. Results and Discussion

a. Separation performance

The 33 µm-filter removed 35 – 37 % TSS, 24 % COD and 7 – 8 % TN as shown in Figures 43 and 44 for Periods 1 and 2, respectively. The performance of the filter was relatively stable during the experimental periods, but 3 – 5 % lower compared to the previous tests. This could be due to a generally less concentrated wastewater because of dilution by rainwater and some snow melting.

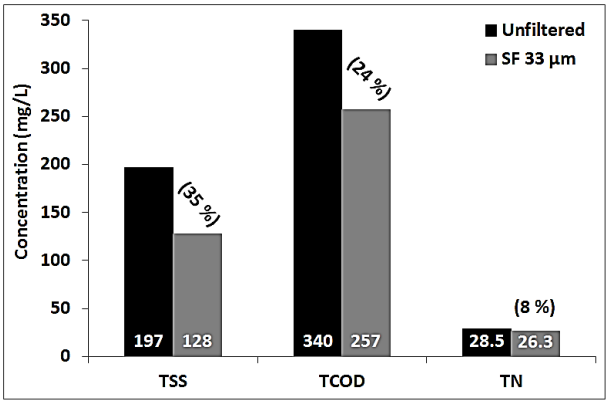


Figure 43. Feed characteristics and Salsnes Filter performance during Period 1 (20.10.14 – 28.11.14)

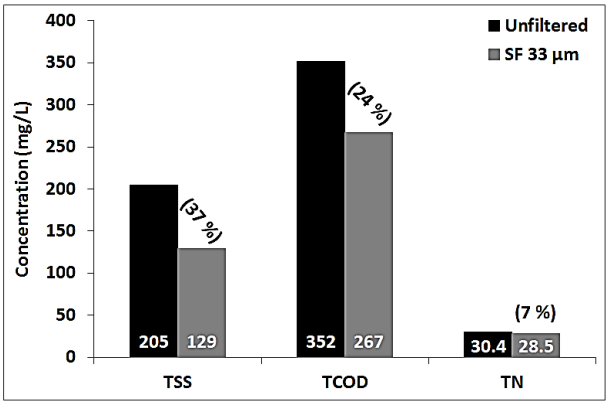


Figure 44. Feed characteristics and Salsnes Filter performance during Period 2 (03.11.14 – 28.11.14)

b. SBR performance

Table 32 and 33 summarize the performance of the SBRs during Period 1 and Period 2, respectively. In Period 1 the control SBR (R1) and the SBR fed filtered wastewater (R2) had relatively similar removal efficiencies of TSS, COD and TN. R1 had a removal of about 83 % TSS, 72 % COD and 56 % TN, while R2 removed 79 % TSS, 69 % COD and 51 % TN. Higher nitrate concentration in the effluent of R2 was the main cause for the difference. Less denitrification due to less COD seems to affect the TN removal results. Including the TN removal of the primary filter, the overall TN removal in R2 was almost the same as R1, 54 % vs. 56 %. The advantage of the system combining primary filtration and biological treatment was that more pollutants were removed with the filter, and therefore the overall removal efficiencies were similar or slightly higher compared to the control SBR. From these results, the removal of a portion of the particulate COD did not affect the performance of the SBR. Thus, the C/N ratio was still sufficient to achieve similar TN removal as in the control SBR.

In Period 2 at reduced SRT the SBR treating filtered wastewater (R5) had better TSS, COD and TN removal compared to the control SBR (R4). R5 had a removal efficiency of 70 % TSS, 61 % COD and 72 % TN removal, compared to 57 % TSS, 56 % COD and 56 % TN removal for R4. The observations made during the lab-scale SBR were similar and showed for this wastewater the partial removal of COD had only a small effect on the biological nitrogen removal downstream.

Table 32. Feed characteristics and SBR removal efficiencies during Period 1 (SRT: 12 d)

	R1 (Raw)			R2 (33 µm – Room temperature)				R3 (33 µm – 8 °C)			
	Influent	**Effluent**	**Removal efficiency (%)**	**Influent**	**Effluent**	**SBR Removal efficiency (%)**	**Total Removal efficiency (%)**	**Influent**	**Effluent**	**SBR Removal efficiency (%)**	**Total Removal efficiency (%)**
TSS (mg TSS/L)	124 ± 27	20 ± 13	**83**	82 ± 23	14 ± 13	**79**	**92**	86 ± 33	30 ± 8	**61**	**84**
VSS (mg VSS/L)	107 ± 41	18 ± 10	**83**	65 ± 18	12 ± 10	**77**	**77**	66 ± 18	25 ± 6	**85**	**85**
TCOD (mg COD/L)	225 ± 46	62 ± 19	**72**	164 ± 41	52 ± 21	**69**	**84**	179 ± 35	76 ± 10	**56**	**77**
sCOD (mg COD/L)	67 ± 18	42 ± 11	**39**	50 ± 29	39 ± 17	**40**	**40**	66 ± 18	44 ± 10	**56**	**56**
TN (mg TN/L)	29 ± 8	8.6 ± 4.8	**56**	18 ± 6	9.1 ± 3.4	**51**	**54**	22 ± 6	17.4 ± 6.4	**29**	**39**
NH_4-N (mg NH_4-N/L)	11 ± 3	0.27 ± 0.40	**98**	10 ± 4	0.11 ± 0.13	**99**	**99**	14 ± 5	9.0 ± 7.4	**54**	**54**
NO_3-N (mg NO_3-N/L)	0.36 ± 0.27	4.41 ± 3.67	**--**	1.98 ± 1.8	7.0 ± 3.5	**--**	--	2.0 ± 1.8	5.6 ± 5.0	**--**	**--**
NO_2-N (mg NO_2-N/L)	0.06 ± 0.05	0.13 ± 0.13	**--**	0.08 ± 0.06	0.15 ± 0.12	**--**	--	0.24 ± 0.20	0.65 ± 0.65	**--**	**--**
N_{org}	11 ± 4	3.81 ± 3.42	**65**	6.1 ± 3.2	2.88 ± 2.58	**53**		6.3 ± 2.4	2.2 ± 1.2	**65**	
N/TSS	0.08			0.11				0.11			
N/VSS	0.06			0.09				0.09			
C/N	11.6			9.3				9.3			
COD/VSS	1.51			1.76				1.77			
VSS/TSS	0.86			0.80				0.80			

Table 33. Feed characteristics and SBRs removal efficiencies during Period 2 (SRT: 6 d)

	R4 (Raw)			R5 (33 µm – Room temperature)			
	Influent	**Effluent**	**Removal efficiency (%)**	**Influent**	**Effluent**	**SBR Removal efficiency (%)**	**Removal efficiency (%)**
TSS (mg TSS/L)	150 ± 23	64 ± 38	**57**	191 ± 23	33 ± 23	**70**	**83**
VSS (mg VSS/L)	127 ± 21	51 ± 30	**55**	71 ± 17	26 ± 16	**85**	**85**
TCOD (mg COD/L)	243 ± 38	108 ± 40	**56**	181 ± 27	70 ± 23	**61**	**80**
sCOD (mg COD/L)	68 ± 18	47 ± 16	**55**	64 ± 15	37 ± 7	**59**	**59**
TN (mg TN/L)	21 ± 3	9.5 ± 1.3	**56**	20 ± 4	8.9 ± 2.5	**58**	**72**
NH_4-N (mg NH_4-N/L)	12 ± 3	2.0 ± 1.8	**83**	11.6 ± 3.3	1.4 ± 2.0	**91**	**91**
NO_3-N (mg NO_3-N/L)	1.03 ± 1.02	2.15 ± 2.15	**--**	2.5 ± 2.0	6.9 ± 4.2	**--**	**--**
NO_2-N (mg NO_2-N/L)	0.06 ± 0.06	0.07 ± 0.07	**--**	0.04 ± 0.03	0.14 ± 0.10	**--**	**--**
N_{org}	8.1 ± 1.6	4.2 ± 2.4	**48**	6.5 ± 1.1	2.6 ± 1.5	**60**	
N/TSS	0.06			0.09			
N/VSS	0.08			0.12			
C/N	11.6			9.9			
VSS/TSS	0.86			0.80			
COD/VSS	1.42			1.70			

c. Influence of SRT on the SBR performance

Figure 45 shows the impact of SRT on the effluent TSS. It was observed that the SBR operated at a lower SRT of 6 days had higher effluent TSS both in the SBR treating raw wastewater and in the SBR treating filtered wastewater. Higher effluent TSS indicates poor settling of sludge by bulking or floc breakup. The SVI, however, was similar in all SBRs so the cause of higher TSS in the effluent is likely weaker flocs and floc breakup resulting in dispersed particles in the effluent. Higher TSS in the effluent reduced treatment efficiency of the main parameters in the system such as COD, TN and TP. Similar observations were seen by Hajiabadi *et al.* (2009) operating SBRs at SRTs from 5 to 20 days. The average TSS concentrations in the effluent were below 31 mg TSS/L at SRT of 12 days and above 34 mg TSS/L at SRT of 6 days. High effluent TSS is a sign of weak flocs, which results in the formation of less dense particles that are harder to settle and are washed to the effluent (Novak *et al.*, 1998).

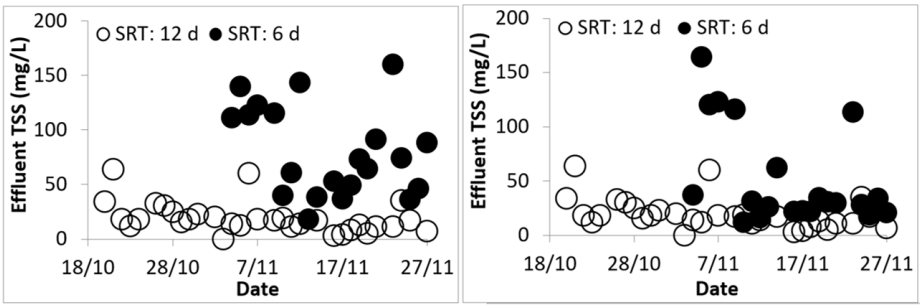


Figure 45. Influence of SRT on the effluent TSS (Left: Raw Feed; Right: Filtered Feed)

COD removal was slightly affected by the variation of SRT. For the SBR fed filtered wastewater and operated at SRT of 12 days COD was 69 %, and it was about 61 % in the SBR operated at 6 days. The main difference in COD removal efficiency is due to different effluent TSS, since the effluent filtered COD was similar for all SBRs. This is explained by the fact that heterotrophs have such a high growth rate and SRT of 6 days and above is sufficient for the biological removal of dissolved COD. The effluent TCOD decreased as the SRT increased (Figure 46). The result corresponds with the effluent TSS and if the amount of TSS in the effluent

increases, the COD also increases. During these experiments the COD/VSS ratios were between 1.5 and 1.8 gCOD/gVSS during Period 1 and 1.4 – 1.7 gCOD/gVSS during Period 2. These values are within the range found in literature, which were between 1.07 – 2.87 g COD/g VSS (Ekama, 2009; Henze *et al.*, 2008; Takacs & Vanrolleghem, 2006).

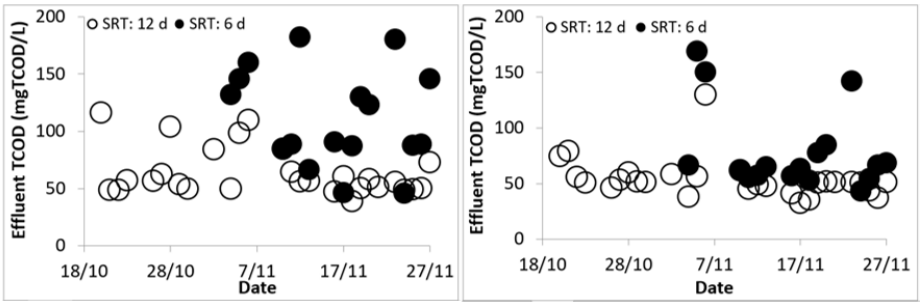


Figure 46. Influence of SRT on the effluent COD (Left: Raw Feed; Right: Filtered Feed)

The ammonium removal efficiencies decrease with reduced SRT. Nearly complete nitrification resulted at SRT of 12 days, while it was between 80 and 90 % at SRT of 6 days. Henze *et al.* (2008) recommended SRT of no less than 8 days for nitrification. However, the total SRT was as low as 6 days in this test, and nitrification still occurred, resulting in a TN removal efficiency nearly similar in all SBRs. Andreadakis (1993) while operating SBRs at SRTs between 4 and 9 days made similar observations.

d. Influence of temperature on the SBR performance

In Figure 47 shows the influence of temperature on the effluent concentrations of TSS, TCOD, NH_4-N and TN. In general, the SBR operated at low temperatures had higher concentrations in the effluent compared to the SBR operated under higher temperatures. The average effluent TSS was 30 mg TSS/L at 8 °C and 14 mg TSS/L at 17 °C, while for COD, the average effluent total COD was 80 mg COD/L at 8 °C and 50 mg COD/L at 17 °C.

The most significant effect of temperature was on nitrification, which was complete in the SBRs operated at 17 °C, while at 8 °C, only 50 % of the

ammonium was converted to nitrate. The TN removal efficiency was only 39 % at 8 °C compared to 56 % in SBRs at 17 °C.

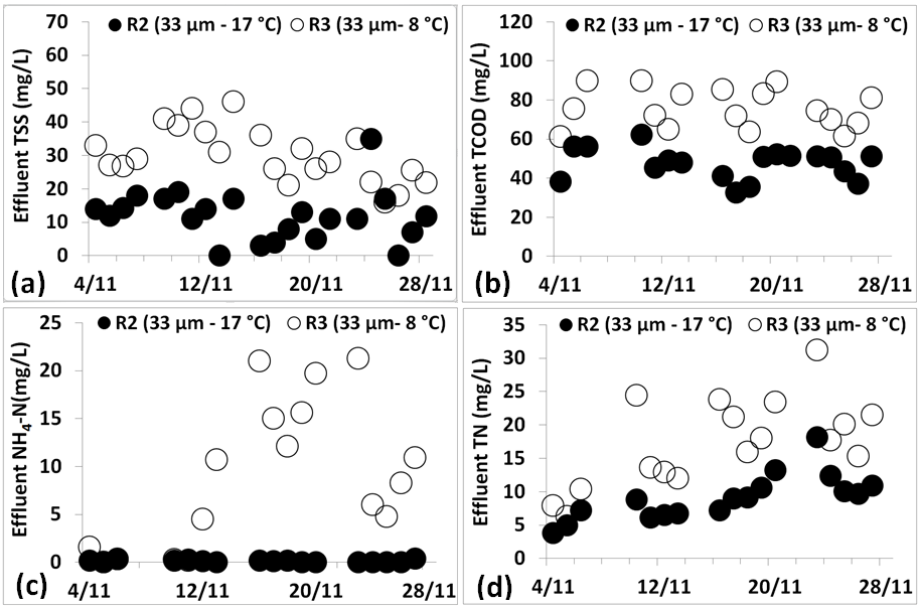


Figure 47. Influence of temperature on the SBR performance (a. effluent TSS, b. effluent TCOD, c. effluent NH_4-N, d. effluent TN)

It has been reported that high TN removal efficiency could be achieved in an activated sludge process operated at 10 °C by applying an F/M ratio lower than 0.10 mgCOD/mgMLVSS-d (Choubert *et al*., 2005). In this study, the F/M ratio was 0.143 mgCOD/mgMLVSS-d in the SBR operated at 8 °C with a relatively low nitrogen removal efficiency. At reduced temperature, longer HRT and SRT are required to achieve full nitrification. Rusten & Eliassen (1993) suggested a minimum aerobic SRT of 12 to 13 days for full nitrification at 10 °C. Thus, increased SRT at 8 °C was necessary for achieving nitrogen removal similar to SBR at 17 °C.

e. Sludge production

Table 34 shows the measured sludge production in each SBR system. The corresponding biogas production for each SBR was calculated and presented in Table 34.

Table 34. Sludge production during the two study periods

	R1	R2	R3	R4	R5
Filter	Unfiltered (P1)	33 µm (P1)	33 µm-LT (P1)	Unfiltered (P2)	33 µm (P2)
Primary sludge (gTS/d)	---	0.82 ± 0.31	0.82 ± 0.31	---	0.91 ± 0.29
Biosludge (gTS/d)	1.72 ± 0.22	1.57 ± 0.16	1.48 ± 0.28	2.61 ± 0.62	2.01 ± 0.42
Total sludge production (gTS/d)	**1.72 ± 0.22**	**2.39 ± 0.32**	**2.31 ± 0.28**	**2.61 ± 0.62**	**2.90 ± 0.43**
biosludge yield (gTS/gCOD)	0.51	0.63	0.65	0.93	0.87
Methane production (L CH_4/d)	0.35 ± 0.05	0.49 ± 0.07	0.46 ± 0.06	0.54 ± 0.14	0.60 ± 0.10
Energy production (Wh/d)	3.5 ± 0.47	4.93 ± 0.65	4.63 ± 0.58	5.42 ± 1.40	6.03 ± 1.04

*LT: Low temperature

The SBRs combining primary filtration and biological treatment (R2, R3 and R5) produced more sludge compared to the SBRs with only biological treatment (R1 and R4). This is corresponding with observations in the previous studies.

In Period 1, the SBR fed raw wastewater (R1) produced about 1.72 gTS/d of sludge, composed of biosludge only. The SBR fed with filtered wastewater (R2) produced less biosludge compared to the SBR fed raw wastewater due to the partial removal of COD. The values were about 1.57 gTS/d for R2 and 1.48 gTSS/d for R3. The filter removed about 0.82 g TS/d; which brought the overall sludge production in R2 and R3 to 2.39 gTS/d and 2.31 gTS/d, respectively. Consequently, the sludge production increased by 39 % in the system with primary treatment. In Period 2, the sludge production was 11 % higher in the SBR fed filtered wastewater (R5) compared to the SBR fed raw wastewater (R4).

The main advantage of using a filter was that 35 % of the sludge reduced with primary treatment. Numerous studies have shown that primary sludge has higher energy content than biological sludge (Paulsrud *et al.*, 2014; Halim, 2012). During Period 1, the biogas potential in the control SBR (R1) was 3.5 Wh/d compared to 4.9 Wh/d in R2 with primary treatment. The energy gains were higher at lower SRT as the sludge production was higher. The biogas potential was 5.4 Wh/d in the control SBR and 6.0 Wh/d in the SBR with primary treatment. The effects of SRT and the temperature on sludge production are that reduced SRT and lower temperature increase sludge production, as long as SRT is above the minimum SRT. In terms of treatment costs and energy yield as biogas a low SRT is preferred, as it requires less energy for aeration and will generate more biogas from higher sludge production. All values in this study adjusted to a temperature of 20 °C except for R3, in order to evaluate the effect of temperature. The average sludge yield in R3 was equal to 0.452 g TSS/gCOD at 8 °C compared to 0.352 g TSS/gCOD and 0.400 g TSS/gCOD in the SBRs R1 and R2 operated under room temperature, respectively.

f. Oxygen demand

Oxygen supply by aeration consume large amount of energy in biological wastewater treatment. It accounts for almost 55 % of the

energy usage of a typical municipal WWTP (Metcalf & Eddy, 2003). Table 35 summarizes calculated oxygen demand for each SBR system.

Table 35. Oxygen demand for each SBR system

Filter	R1	R2	R3	R4	R5
	Raw WW (P1)	33 µm (P1)	33 µm-LT (P1)	Raw WW (P2)	33 µm (P2)
OD_{COD}	1.74	1.32	1.28	1.35	0.71
OD_{NH4-N}	0.77	0.71	0.86	0.62	0.53
OR_{NO3-N}	0.42	0.33	0.25	0.28	0.24
OD_T	**2.09**	**1.71**	**1.93**	**1.69**	**1.02**

LT: Low temperature
OD_{COD}: Oxygen required for the degradation of organic COD (gO_2/d)
OD_{NH4-N}: Oxygen required for nitrification (gO_2/d)
OR_{NOx-N}: Oxygen recovered during denitrification (gO_2/d)
OD_T: Total oxygen demand (gO_2/d)

According to Table 35, the SBRs treating raw wastewater (R1 and R4) required more oxygen compared to the SBRs fed filtered wastewater (R2, R3 and R5). In Period 1, the oxygen demand for the control SBR (R1) increased by 22 % compared to the SBR fed filtered wastewater (R2). In Period 2, the oxygen demand increased by 66 % for the control SBR (R4) compared to R5 fed with filtered wastewater.

The difference in oxygen demand observed related to the degradation of COD, caused by different COD loads due to primary treatment. The exception was for the reactor operated under low temperature where the oxygen demand for nitrification was significantly lower.

Thus, the separation of a portion of the COD with Salsnes Filter had a positive effect on the overall process. By reducing the organic load to the bioreactor, the energy related to oxygen supply reduced, while the energy gain from biogas increased. Therefore, the overall treatment cost reduced.

g. Nitrification

Influence of ammonium load

Complete nitrification is necessary for efficient nitrogen removal. Figures 48 and 49 show nitrification rates as a function of the ammonium load during the two study periods.

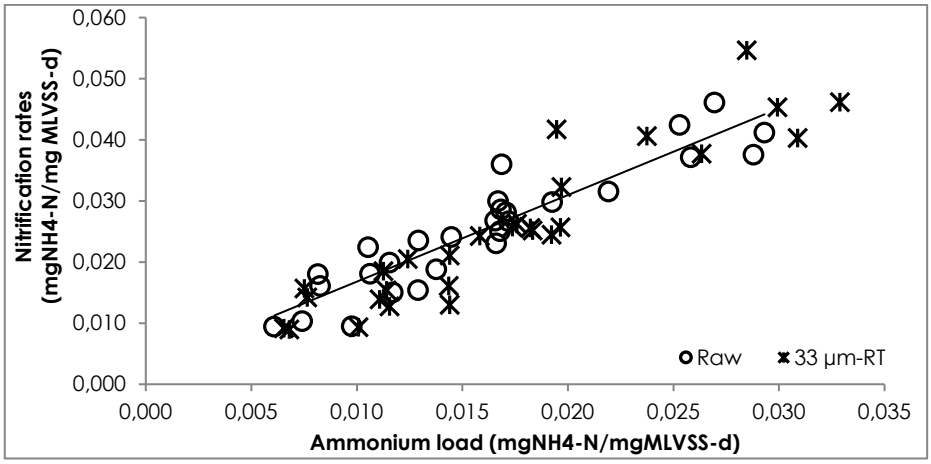


Figure 48. Ammonium load versus nitrification rates in Period 1

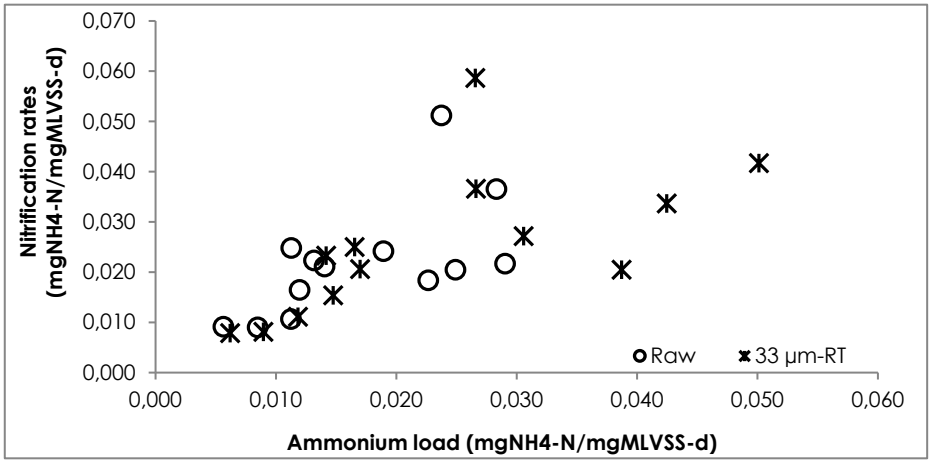


Figure 49. Ammonium load versus nitrification rates during Period 2

In Period 1, the nitrification rates were between 0.009 and 0.046 mgNH_4-N/mg MLVSS-d in the control SBR (R1) and 0.009 – 0.055 mgNH_4-N/mg MLVSS-d in the SBR fed filtered wastewater (R2) operated under room temperature (Figure 48). Observations from previous results was that partial removal of COD is beneficial for nitrification, resulting in higher nitrification rates compared to the SBR fed raw wastewater. However, during this study, the average nitrification rates were similar in the two SBRs. The reason for that is probably due to limiting ammonium load to the SBR fed filtered wastewater.

In Period 2, the nitrification rates in the reactor control (R4) varied from 0.016 to 0.051 mgNH_4-N/mg MLVSS-d, with an average value of 0.026 mgNH_4-N/mg MLVSS-d. The nitrification rates in the reactor fed filtered wastewater ranged between 0.015 and 0.059 mgNH_4-N/mg MLVSS-d, with an average of 0.030 mgNH_4-N/mg MLVSS-d. Similar to the result in Period 1, the partial removal of COD lead to a higher nitrification rates compared to the reactor control.

According to these results, the rates were slightly higher in the SBRs fed with filtered wastewater. Regulated nitrification rates are by several factors, such as ammonium availability, oxygen concentration in the reactor and organic availability (Triska *et al.*, 1990). In general, the nitrification rates increase with increasing organic and ammonium loading rates. However, according to Strauss *et al.* (2002), the positive effect of organic loading on nitrification rates was stronger at low C/N ratio because heterotrophic bacteria were not favored. High C/N ratio generally decreased nitrification rates as heterotrophic bacteria out-compete nitrifying bacteria for the available NH_4-N when its concentration is low and the organic matter is not rate limiting.

Influence of temperature

Nitrifying bacteria are sensitive to temperature (Barnes & Bliss, 1983). The temperature affects the kinetics of nitrifiers' growth such as maximum specific growth rate, substrate utilization rate, the yield coefficient and decay rate. Figure 50 shows the influence of temperature on nitrification rates.

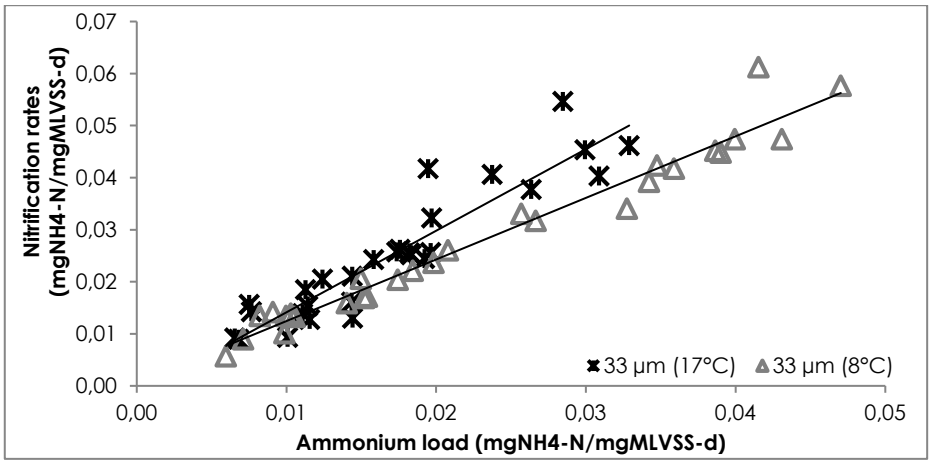


Figure 50. Impact of ammonium load on Nitrification rates at different temperature

The nitrification rates were between 0.006 and 0.061 mgNH_4-N/mgMLVSS-d, with an average value of 0.035 mgNH_4-N/mgMLVSS-d for the SBR operated at low temperature. The result is quite similar to the nitrification rate observed in the SBR fed filtered wastewater operated at room temperature. However, as observed in Figure 49, the SBR operated under room temperature had slightly higher rate if operated at the same ammonium load as the SBR operated at low temperature. The maximum nitrification rates reported in the literature at 10 °C were in a very wide range: 0.024 to 0.108 mgNH4-N/mgMLVSS-d (Burica *et al*., 1996; McCartney & Oleszkiwicz, 1990; Oleszkiwicz & Berquist, 1988, Palis & Irvine, 1985). Generally, the nitrification rates increased with temperature. However, temperature is not the only parameter affecting nitrification (Kemp & Dodds, 2001; Strauss & Lamberti, 2000; Strauss & Dodds, 1997; Triska *et al*., 1990).

Full nitrification was in the SBR operated at room temperature (R2). The nitrification was, however, incomplete in the SBR operated at low temperature (R3). The aerobic SRTs in R2 and R3 were 5.2 days and 6.3 days, respectively. By applying an ammonium load of 0.006 to 0.047 mgNH_4-N/mgMLVSS-d, the effluent ammonium concentrations varied from 5 mgNH_4-N/L to 21 mgNH_4-N/L in the reactor operated at low temperature; the higher the applied ammonium load, the higher would

be the effluent ammonium. In order to achieve full nitrification, R3 operated with a much longer SRT.

h. Denitrification

Nitrogen removal is by alternating anoxic and aerobic conditions for denitrification and nitrification, respectively. Table 36 show the specific denitrification rates (SDNRs) determined for each SBR system.

In Period 1, the first SDNRs (K1) were between 0.10 and 0.34 mgNO_x-N/mgMLVSS-d in the control SBR (R1) and 0.33 – 0.48 mgNO_x-N/mgMLVSS-d in the SBR fed filtered wastewater operated at room temperature (R2). The second SDNRs (K2), on the other hand, ranged from 0.04 to 0.16 mgNO_x-N/mgMLVSS-d in R1 and 0.09 – 0.16 mgNO_x-N/mgMLVSS-d in R2.

In Period 2, the first SDNRs varied from 0.24 to 0.49 mgNO_x-N/mgMLVSS-d in the control reactor (R4) and 0.25 – 0.58 mgNO_x-N/mgMLVSS-d in the reactor fed filtered wastewater (R5). The second SDNRs were between 0.12 and 0.31 mgNO_x-N/mgMLVSS-d in R4 and 0.05 – 0.42 mgNO_x-N/mgMLVSS-d in R5.

According to the results, the SBR fed filtered wastewater had higher SDNRs compared to the control SBR during the two study periods. It seemed that a C/N ratio of about 10 mgCOD/mgN was beneficial for denitrification. Similar results resulted from the lab-scale SBRs. In addition, SDNRs decreased as the SRT increased. The same observation came from Shahzad *et al.* (2015) when evaluating the performance of full-scale activated sludge at varying SRTs.

Temperature often imposes some limitation for denitrification to proceed at an acceptable rate. At lower temperature (R3), there was a substantial reduction in both the removal efficiency and the SDNRs. The first SDNRs (K1) were between 0.08 and 0.17 mgNO_x-N/mgMLVSS-d and the second SDNR (K2) was 0.05 mgNO_x-N/mgMLVSS-d. The first SDNR decreased by 65 % compared to the SBR operated at room temperature (R2). Obaja *et al*. (2005) also observed a reduction of the denitrification rate by 80 % when investigating the impact of temperature from 20 °C to 8 °C on SBR treating piggery wastewater.

Table 36. Specific denitrification rates in the different SBRs, including influent F/M and C/N ratios

	Period 1			Period 2	
Parameter	**R1 (Raw)**	**R2 (33 µm)**	**R3 (33 µm-LT)**	**R4 (Raw)**	**R5 (33 µm)**
SDNR ($mgNO_x$-N/mgMLVSS-d)					
Rate 1 (K1)	0.21 ± 0.09	0.40 ± 0.04	0.14 ± 0.03	0.34 ± 0.09	0.40 ± 0.10
Rate 2 (K2)	0.09 ± 0.05	0.12 ± 0.02	0.05 ± 0.00	0.21 ± 0.06	0.19 ± 0.12
C/N (mgCOD/mgN)	11.6 ± 2.67	9.8 ± 2.7	8.7 ± 2.2	12.0 ± 1.7	9.6 ± 1.4
F/M (mgCOD/mgMLVSS-d)	0.22	0.19	0.21	0.31	0.32

Temperature can exert an effect on biological reactions in two ways: by influencing the rates of enzymatically catalyzed reactions and by affecting the rate of diffusion of substrate into the cells (Krzeminski *et al.*, 2012; Leslie Grady *et al*., 1999).

5.3.3. Conclusions

The objectives of the study were to investigate the performance of SBRs with and without primary filtration operated at different SRT and temperatures.

The 33 µm filter removed about 33-35 % of the influent TSS, 25 % of the total COD, and 8 % of the TN, slightly lower compared to the previous tests.

The performances of the 5 SBRs were evaluated based on TSS, TCOD and TN removal efficiencies. In Period 1, the control SBR and the SBR fed filtered wastewater had relatively similar removal efficiencies of TSS, COD and TN. The values were 83%, 72 % and 56 %, respectively. In Period 2 at reduced SRT the SBR treating filtered wastewater (R5) had better TSS, COD and TN removal compared to the control SBR (R4). R5 had a removal efficiency of 70 % TSS, 61 % COD and 72 % TN removal, compared to 57 % TSS, 56 % COD and 56 % TN removal for R4.

In general, the SBR operated at low temperature had higher pollutants concentrations in the effluent compared to the SBR operated under higher temperature. The average effluent TSS was 30 mgTSS/L at 8 °C and 14 mgTSS/L at 17 °C, while for COD, the average effluent total COD was 80 mgCOD/L at 8 °C and 50 mgCOD/L at 17 °C. The most significant effect of temperature was on nitrification, which was complete in the SBRs operated at 17 °C, while at 8 °C, only 50 % of the ammonium was converted to nitrate. The TN removal efficiency was then only 39 % at 8 °C compared to 56 % in SBRs operated at 17 °C.

Primary filtration resulted in increased sludge production and potential more biogas and energy yield. In Period 1, the SBR fed raw wastewater (R1) produced about 1.72 gTS/d of sludge compared to 2.39 gTS/d and 2.31 gTS/d, in R2 and R3, respectively. The energy production in the control SBR (R1) was 3.5 Wh/d compared to 4.9 Wh/d in R2 with primary

treatment. The energy gains were higher at lower SRT as the sludge production was higher. The energy generation was 5.4 Wh/d in the control SBR and 6.0 Wh/d in the SBR with primary treatment.

The SBRs treating raw wastewater (R1 and R4) required more oxygen compared to the SBRs fed filtered wastewater (R2, R3 and R5). In Period 1, the oxygen demand for the control SBR (R1) increased by 22 % compared to the SBR fed filtered wastewater (R2). In Period 2, the oxygen demand increased by 66 % for the control SBR (R4) compared to R5 fed filtered wastewater.

Consequently, the removal of particulate COD with 33 µm filter did not have any negative impact on the TN removal in the SBR for the tested wastewater, mainly because of sufficient COD.

Chapter 6

Steady state simulation of activated sludge biological N removal in MBR

The steady state model of activated sludge biological N removal as described in chapter 3.4 was applied to the data from the experiments with the MBR system (chapter 5.2). The MBR system is like MLE-system but simpler as it includes only one recycle flow, from the aerobic to the anoxic reactor, the a-recycle (Figure 51).

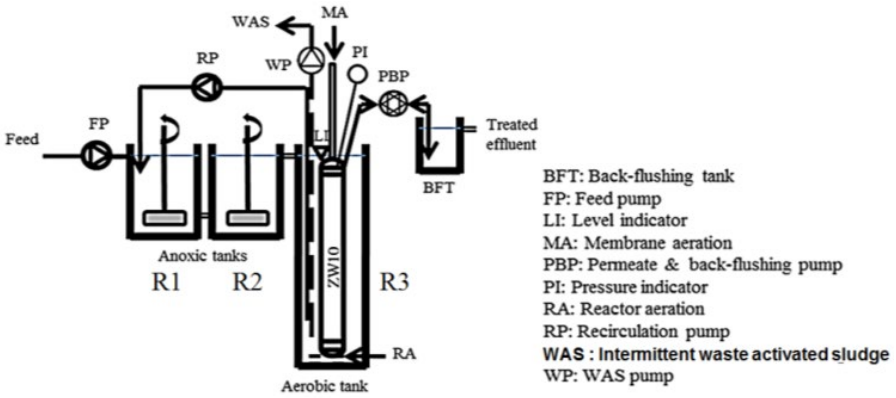


Figure 51. Schematics of MBR system.

6.1. Calculation of nitrogen removal

The effluent concentration of ammonia ($N_{a,e}$) is based on the solution of the Monod equation with respect to substrate (Chapter 3.4.4):

$$N_{a,e}=\frac{K_N\cdot(k_{dA}+1/SRT)}{\mu_{maxA}\cdot f_{ax}-(k_{dA}+1/SRT)} \quad (mgNH_4\text{-}N/l)$$

The organic nitrogen is converted to ammonia in ammonification according to a first order ammonification rate (K_r) and heterotrophic biomass (X_H), resulting in effluent organic N ($N_{org,e}$):

$$N_{org,e}=\frac{N_{org,i}}{1+\frac{K_r\cdot X_H\cdot V}{Q}} \quad (mgN/l)$$

$N_{org,i}$: Influent organic nitrogen (mg/l)
$N_{org,e}$: Effluent organic nitrogen (mg/l)
K_r: Ammonification rate (mgVSS·d/l)

The amount ammonia nitrified is defined as the nitrification capacity (N_C) and is the sum of ammonia removed, ammonia generated in ammonification and subtracted for ammonia utilized in heterotrophic growth.

$$N_C=(N_{a,i}-N_{a,e})+(N_{org,i}-N_{org,e})-\left(\frac{f_n\cdot X_{VSS}\cdot V}{SRT\cdot Q}\right) \quad (mgNH_4\text{-}N/l)$$

$N_{a,i}$: Influent ammonia (mg/l)
f_n: N content of biomass (mgN/mgVSS)

Based the assumption of full denitrification in the anoxic reactor the effluent nitrate was calculated as the nitrification capacity divided by the total flow through the reactor, recycle ratio (a) and influent flow.

$$NO_x\text{-}N = \frac{N_C}{1+a}$$

The simulation was performed at sludge ages (SRT) between 6 and 20 days and the simulation results were compared to the experimental results. The SRT of the test with raw wastewater was 13.7 days and with

filtered wastewater 16.8 days. The effluent nitrogen was calculated as the sum of ammonia, nitrate and soluble organic nitrogen in the effluent. Since separation of sludge was by membranes the effluent TSS was zero and thus also particulate nitrogen.

6.2. Application of steady state model

The steady state model was applied based on default values provided in the original model. Not all model parameters of the wastewater characteristics were determined experimentally, some estimated based on previous studies (Table 37). Not all biological parameters were determined experimentally but the significance of the most critical parameters were tested against the experimental results (Table 38).

Table 37. Wastewater characteristics

Parameter		
Total COD	COD	Measured
Soluble unbiodegradable COD	COD_{us}	Measured
Readily biodegradable COD	RBCOD	Estimated as filtered COD minus filtered effluent COD
Slowly biodegradable COD	SBCOD	Estimated based on total COD minus other fractions
Unbiodegradable particulate COD	COD_{up}	Assumed 10 % of total COD
Total Nitrogen	TN	Measured
Ammonia nitrogen	N_a	Measured
Nitrate+Nitrite nitrogen	NO_x-N	Measured
Soluble organic nitrogen	N_{org}	Estimated zero based on effluent analyses
Total organic nitrogen	N_{org}	Estimated as total nitrogen minus other fractions
Unbiodegradable particulate N	N_{up}	Assumed proportional to corresponding COD (COD_{up})

Table 38. Biological characteristics

Heterotrophs			
Maximum specific growth rate	μ_{maxH}	2	d^{-1}
Half saturation constant	K_S	10	mg/l COD
Decay rate	k_{dH}	0.16	d^{-1}
Growth yield	Y_H	0.45	gVSS/gCOD
Endogenous residue fraction	f_d	0.2	gVSS/gVSS
COD/VSS-ratio	f_{cv}	1.42	gCOD/gVSS
Ammonification rate	K_r	0.015	l/(gVSS*d)
Denitrification rate on RBCOD	K_1	0.70	gN/gX_H*d
Denitrification rate on SBCOD	K_2	0.115	gN/gX_H*d
Endogenous denitrification rate	K_3	0.07	gN/gX_H*d
N content of VSS	f_n	0.06-0.10	gN/gVSS
Autotrophs			
Maximum specific growth rate	μ_{maxA}	0.8	d^{-1}
Half saturation constant	K_N	0.5	mg/l NH_4-N
Decay rate	k_{dA}	0.05	d^{-1}
Growth yield	Y_A	0.17	$gVSS/gNH_4$-N

The system specific parameters applied were the anoxic fraction of the bioreactor, f_{ax}, sludge age (SRT), measured VSS/TSS-ratio and oxygen concentration in recycle from aerobic to anoxic zone (O_a).

Table 39. The state variables calculated

Variables		
Heterotrophic organisms	X_H	mg/l VSS
Endogenous residue from dead organisms	X_E	mg/l VSS
Accumulated particulate unbiodegradable COD	X_I	mg/l VSS
Mixed liquor volatile suspended solids	MLVSS	mg/l VSS
Mixed liquor total suspended solids	MLSS	mg/l TSS
Effluent ammonia	$N_{a,e}$	mg/l NH_4-N
Effluent organic N	$N_{org,e}$	mg/l N
Effluent nitrate N	NO_x-N	mg/l NO_3-N
Effluent total N	TN_{eff}	mg/l N
N removal efficiency	N-%	

The denitrification rates determined experimentally were applied in the model. However, it was only K_2 that was used since RBCOD was assumed to be consumed completely and therefore the denitrification potential based on stoichiometry. The K_3 rate was not applied since it was an MLE system where denitrification is based on COD in the influent wastewater. The decay rate for heterotrophic organisms was estimated based on K_3 that was determined in the batch tests (Chapter 3.4.5). The maximum specific growth rates were not determined experimentally, and for the heterotrophic organisms, the SRT is much longer than minimum SRT so it had no relevance. For the autotrophic organisms it is of more significance and will have effect on the system at low SRT, so the default model value was selected. The half saturation constant for ammonia had the biggest influence on the effluent ammonia concentration and was reduced from a default value of 1 to 0.5 mg/l. Based on research and thermodynamic considerations one has seen that the heterotrophic growth yield under anoxic conditions are lower than under aerobic conditions, typically 20 % lower. Testing of the model with reduced yield did not affect the results of any significance since the

6.3. Simulation of MBR experiments

Figures 52 – 57 present results from simulations including the calculated and measured values of MLSS, MLVSS, effluent nitrogen fractions and nitrogen removal efficiency. Table 40 summarizes calculated and measured parameters.

Table 40. Calculated and measured parameters of the MBR experiment

	Raw wastewater		**Filtered wastewater**	
	Calculated	Measured	Calculated	Measured
MLSS (g/l)	7.42	6.18	5.90	5.79
MLVSS (g/l)	5.49	4.57	4.04	3.97
Ammonia (mg/l)	0.31	0.24	0.26	0.32
Nitrate (mg/l)	9.59	8.60	9.39	10.0
Total N effluent (mg/l)	10.4	11.3	10.1	11.3
N-removal efficiency (%)	75.9	73.3	72.9	73.2

The largest differences between calculated and measured values were for the MLSS and MLVSS, explained by the many parameters involved in the calculations. Not all of these were by experiments and thus bigger chance for errors. For the other parameters, the differences between calculated and measured were smaller and seems to be a result of nearly full nitrification in the aerobic reactor and nearly complete denitrification in the anoxic reactor. The relatively low recycle ratio of 2 indicate that the nitrate load on the anoxic reactors were lower than the denitrification potential. That is also the cause of the observed high nitrogen removal efficiency, and by increasing the a-recycle the removal efficiency would be even higher according to Figure 57.

The simulations show that the experimental results are in accordance with the activated sludge model, and thus form a basis for further applications. More experiments at variable conditions and different wastewaters would contribute to verifying the results.

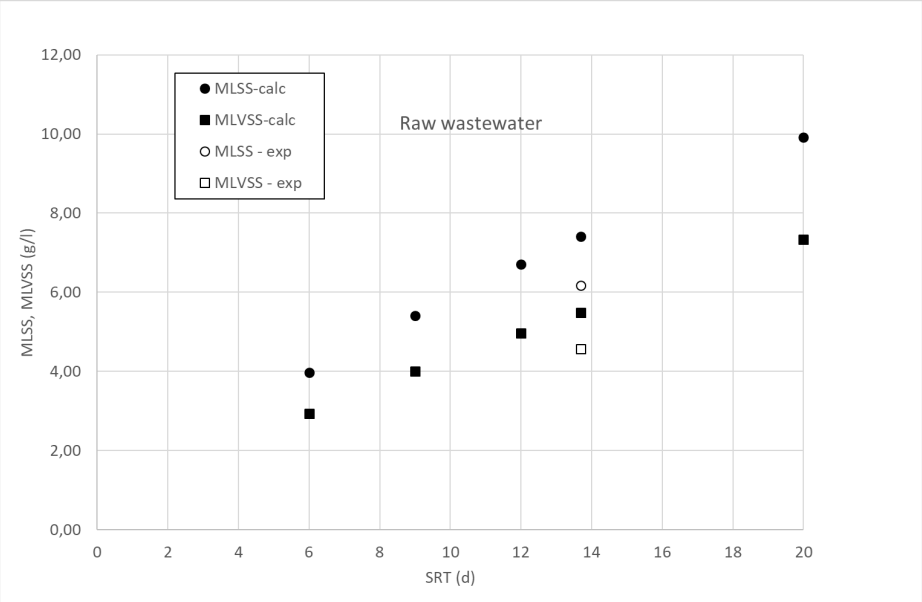


Figure 52. Calculated (calc) and measured (exp) MLSS and MLVSS for raw wastewater

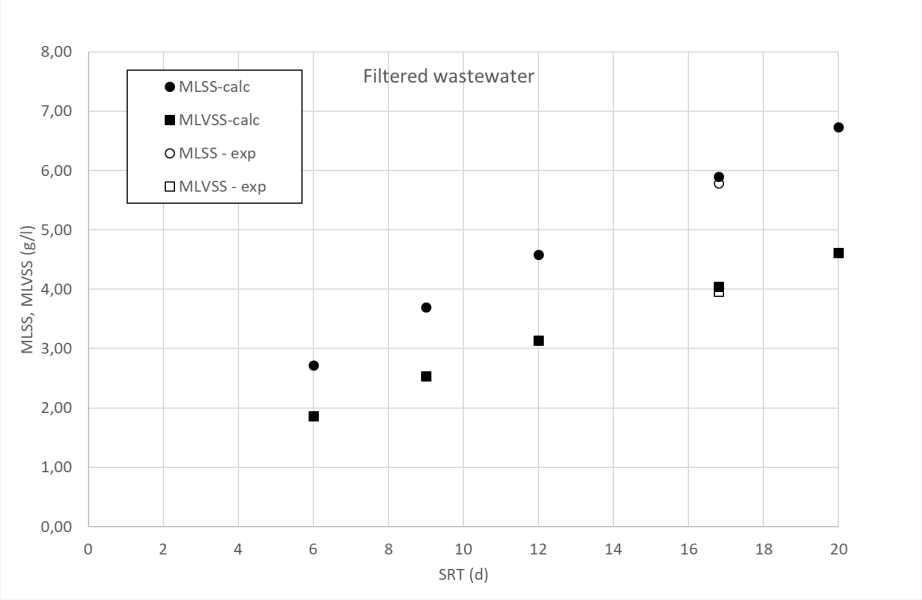


Figure 53. Calculated (calc) and measured (exp) MLSS and MLVSS for filtered wastewater

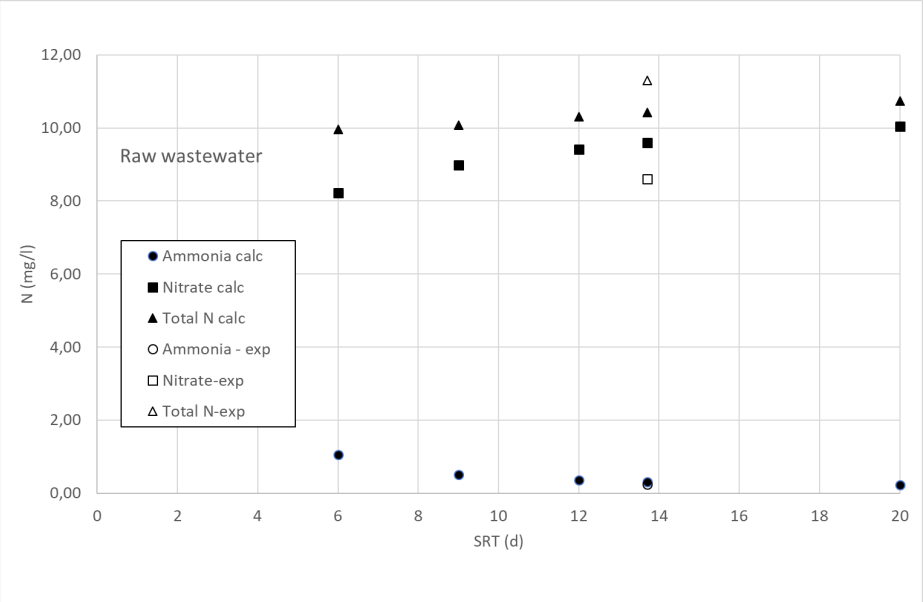


Figure 54. Calculated (calc) and measured (exp) effluent ammonia, nitrate and total N with raw wastewater

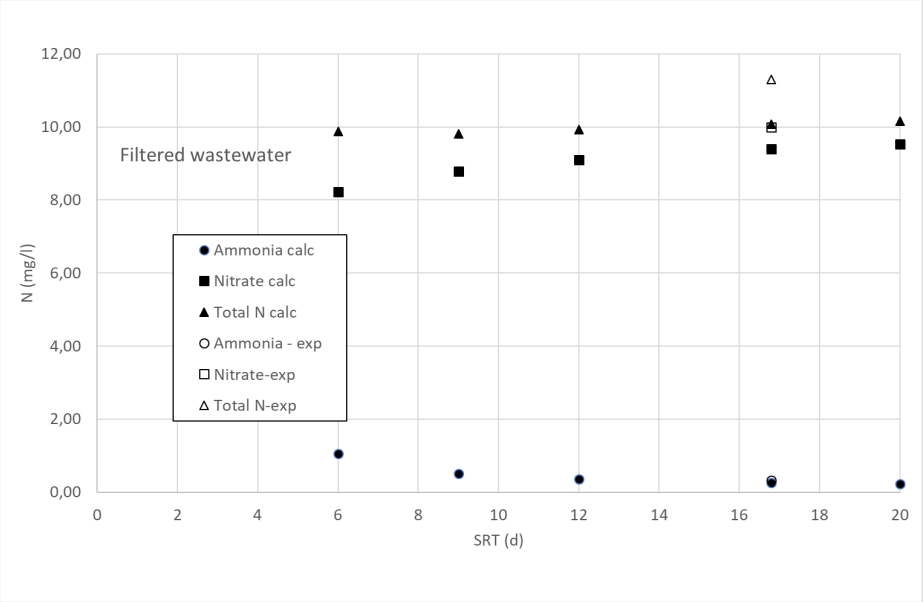


Figure 55. Calculated (calc) and measured (exp) effluent ammonia, nitrate and total N with filtered wastewater

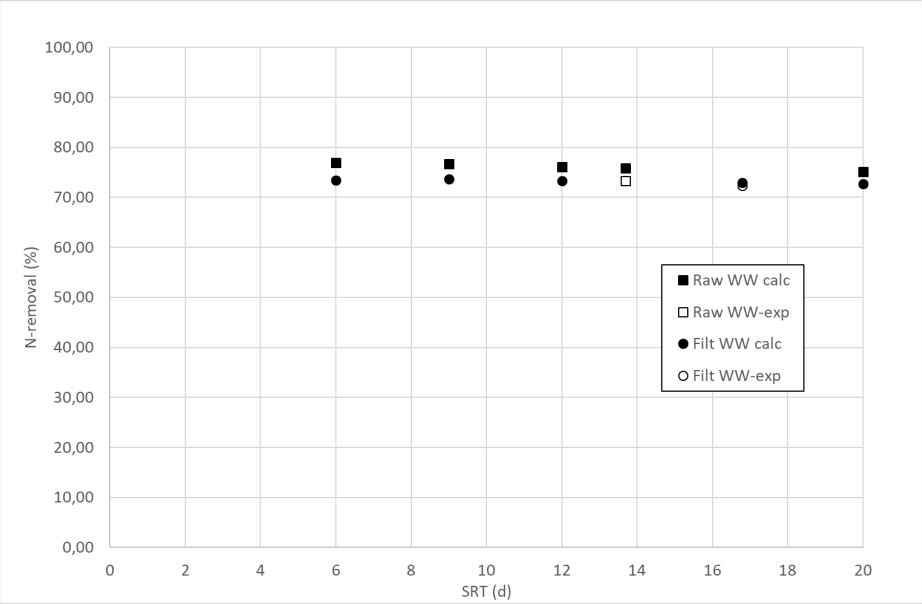


Figure 56. Calculated (calc) and measured (exp) nitrogen removal efficiency

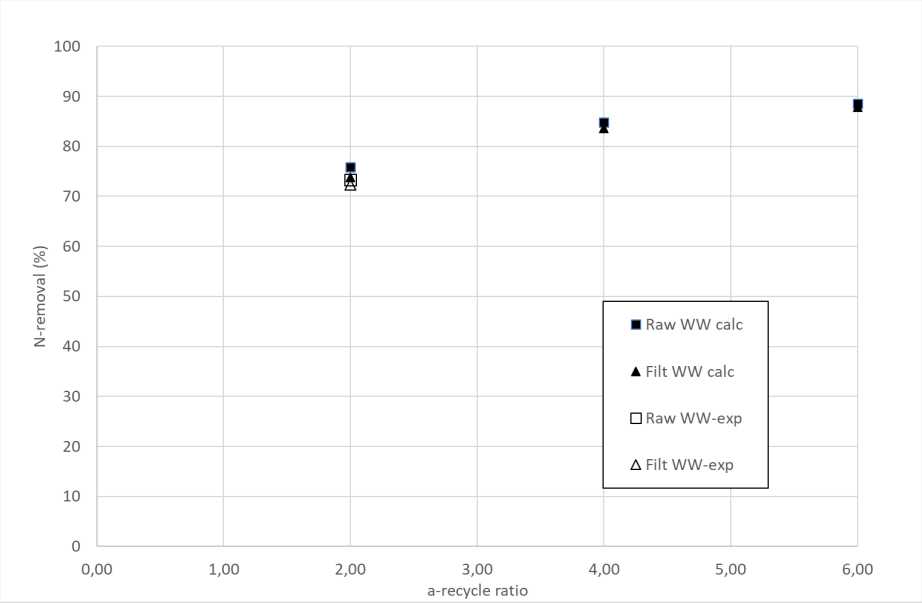


Figure 57. Calculated (calc) and measured (exp) nitrogen removal efficiency as function of a-recycle ratio

Chapter 7

Conclusions

7.1. General conclusions

The goal of this research was to define the optimum particle size for particulate COD removal prior to biological nitrogen removal. Therefore, a succession of experiments investigated the impact of particulate COD removal on biological nitrogen removal with two sets of laboratory tests in SBRs and three pilot-scale studies with MBBR, MBR and SBRs. Fine mesh sieves ranging from 150 µm to 1.2 µm, prepared the different feeds for the biological process. All tests compared to parallel reactors fed raw unfiltered wastewater as control.

7.1.1. Wastewater characterization

TSS and COD removal efficiencies were inversely proportional to the filter openings. At NFR, TSS removal varied from 54 % with 18 µm to 38 % removal with 150 µm filter. At BRA, removal of TSS varied from 57 % with 18 µm to 46 % with 90 µm filter. The corresponding COD removal was 43 % with 18 µm and 21 % with 150 µm at NFR and 42 % with 18 µm and 32 % with 90 µm at BRA. With 1.2 µm filters the COD removal was 81 and 60 % with and without Alum-addition, respectively, at NFR. The COD removal at BRA was 78 and 64 % with and without Alum-addition, respectively.

For the wastewater at NFR, 46 % of the TSS were between 1.2 and 18 µm, 13 % were between 18 and 90 µm and 41 % were above 90 µm. At BRA 43 % of the TSS were between 1.2 and 18 µm, 16 % were between 18 and 90 µm and 43 % were above 90 µm. For COD at NFR the composition was 57 % below 18 µm, 21 % between 18 and 90 µm and 22 % above 90 µm. At BRA the COD composition was 58 % below 18 µm, 10 % between

18 and 90 µm and 32 % above 90 µm. At NFR, the fraction of colloids was 21 % and at BRA 14 % of total COD. The filtered COD in the flocculated samples are soluble, corresponding to 19 % at NFR and 22 % at BRA of total COD, respectively.

7.1.2. Anoxic batch tests

The objective of the first experiment was to evaluate the effect of TSS removal on the denitrification rate. The tests consisted of several anoxic batch tests and lab scale sequencing batch reactors (SBRs).

The results with the activated sludge process showed that tests done with wastewater from BRA WWTP had higher first rate (K1) of 0.18 and 0.26 gNO_x-N/gVSS-d (Test 2) compared to NFR WWTP at between 0.05 and 0.09 gNO_x-N/gVSS-d (Test 1). However, the K2 and K3 rates were similar for the two wastewaters. The values of K2 rates were between 0.04 and 0.06 gNO_x-N/gVSS-d, and between 0.02 and 0.03 gNO_x-N/gVSS-d for the K3 rates. Based on the active biomass, the results showed that the second (K2) and third (K3) rates were close to the reported values from literature, which were 0.1 and 0.07 gNO_x-N/gMLVSS.d, respectively. However, the first rates (K1) were comparatively lower, especially for the test done with wastewater from NFR. The values were between 0.17 and 0.26 gNO_x-N/gMLVSS.d with NFR wastewater and between 0.35 and 0.60 gNO_x-N/gMLVSS.d with BRA wastewater compared to 0.72 gNO_x-N/gMLVSS.d in literature.

In the MBBR process tests, the first rates (K1) varied between 0.80 and 2.43 gNO_x-N/m^2-d in the reactor fed with wastewater from NFR (Test 3) compared to 1.22 – 2.69 gNO_x-N/m^2-d in the reactor fed with wastewater from BRA (Test 4). Compared to the tests with activated sludge the K2 and K3 rates were different for the two wastewaters in the MBBRs. The K2 rates were between 0.57 and 1.83 gNO_x-N/m^2-d in Test 3 (NFR) and between 0.99 and 2.36 gNO_x-N/m^2-d in Test 4 (BRA). The K3 rates were between 0.30 and 1.39 gNO_x-N/m^2-d in Test 3 (NFR) and between 0.78 and 1.24 gNO_x-N/m^2-d in Test 4 (BRA).

These tests showed that removal of particulate COD had little effect on the denitrification rates. The observed denitrification rates were also

within reported values from literature, thus indicating that the rates have small differences between different wastewaters.

7.1.3. Lab scale SBRs

The lab-scale SBR experiments in three transparent polyvinylchloride (PVC) plastic cylinder bioreactors had a working volume of 3 L. The tests were over three periods. During the first period (P1), the three SBRs were with (R1) raw wastewater, (R2) filtered wastewater through 18 µm, and (R3) wastewater filtered at 1.2 µm. In the second period (P2), raw wastewater fed directly to the control reactor, while filters of 90 µm and 33 µm produced the feed for the two remaining SBRs. In the third period (P3), the control reactor fed raw wastewater compared with two SBRs fed with filtered wastewater from 150 µm and 55 µm filters, respectively.

The comparison of the SBRs' performance indicated that the removal efficiencies were lower in the reactors fed with filtered wastewater compared to the control reactors. The SBRs fed filtered wastewater removed between 65 and 75 % of the COD compared to between 70 and 91 % in the control reactors. However, when including the removal efficiency of the primary treatment, similar or higher TSS and COD removal resulted from SBRs fed filtered wastewater, compared to the control reactors. The nitrogen removal efficiencies were between 12 % for wastewater filtered with 1.2 µm filters and 34 % for wastewater filtered through 150 µm, compared to between 59 and 72 % in the control SBRs. However, the overall removal efficiencies including primary treatment and SBR were like the control reactor, except for the SBRs fed with filtered wastewater from filters below 33 µm, where a net reduction in the nitrogen removal resulted.

The higher removal of TSS before biological treatment resulted in a higher total sludge production and thus an increased biogas potential. In Period 1, the overall sludge production varied from 0.32 gTS/d for the SBR fed raw wastewater to 0.59 gTS/d for the SBR fed filtered wastewater from 1.2 µm filtration. In Period 2, the total sludge production was 0.63 gTS/d for the control reactor and 1.49 gTS/d for the SBR fed filtered wastewater from 33 µm. In Period 3, the average sludge productions were 0.88 gTS/d for the control reactor and 1.31 gTS/d for the reactor

fed filtered wastewater at 55 µm. The methane potential was about 0.06 L CH_4/d in the control reactor and up to 0.13 L CH_4/d in the reactor with filtered wastewater (Period 1). In Period 2 and 3, the methane potentials were between 0.13 and 0.15 L CH_4/d in the control reactor and up to 0.29 L CH_4/d in the reactor receiving filtered wastewater.

The reduction in COD load for the biological process resulted also in less energy consumption for aeration and oxygen supply.

The specific nitrification rates (SNRs) increased when the C/N ratios decreased. The SBR fed filtered wastewater at 1.2 µm (no particulate COD) had the highest SNR. The SNRs values were 0.28 gNH4-N/gMLVSS-d in the SBR fed filtered wastewater at 1.2 µm and 0.16 gNH4-N/gMLVSS-d at 150 µm. The denitrification rates (SDNRs) ranged from 0.16 to 0.28 gNO_x-N/gMLVSS-d for the highest rate K1, and between 0.06 to 0.1 gNO_x-N/gMLVSS-d for the second-rate K2.

A filter mesh size of 33 µm is optimum cut-off prior to biological nitrogen removal for the NFR wastewater. It provided both satisfactory COD and nitrogen removals. Therefore, the 33 µm filter size was tested in front of pilot-scale MBBR, MBR and SBR processes.

7.1.4. Pilot scale MBBR process

Two parallel trains of MBBRs were set-up during this third experiment, one fed raw wastewater and one fed filtered wastewater. Each MBBR train was composed of two 4 L anoxic reactors (R1, R2) and two 6 L aerobic reactors (R3, R4) in series, resulting in 40 % of anoxic volume and 60 % of aerobic volume. The reactors were made of transparent plastic PVC. The anoxic reactors were equipped with mechanical mixers and the aerobic reactors had diffusers at the bottom of the tank. Nitrified effluent from Reactor 4 was recycled to Reactor 1 at approximately twice the influent flow rate.

The MBBR process removed about 41 % of TCOD, 50 % of TN and 18 % of TP (Control reactor) and 41 % of TCOD, 41 % of TN and 3 % of TP in the reactor fed filtered wastewater. By applying filtration as secondary separation, the MBBR train fed filtered wastewater had COD, TN and TP removal efficiencies of 74 %, 61 % and 65 %, respectively. Higher removal

efficiencies were observed in the MBBR train fed raw wastewater, with removal of 91 % COD, 68 % TN and 73 % TP. However, with removal in the filter for Train B, the overall removal efficiencies were similar for the two trains of MBBR.

More sludge resulted from Train B, with combined filter and biological treatment. The average sludge production in Train A was 13.7 gTS/d compared to 17.3 gTS/d in Train B, corresponding to 26 % higher sludge production in Train B. The advantage of using filter was that about 32 % of the COD ended as sludge (primary treatment), which is beneficial in terms of biogas-production and savings in aeration. The calculated methane potential was about 2.78 L CH_4/d in the control reactor and 3.53 L CH_4/d in the MBBR fed with filtered wastewater.

The MBBR train fed raw wastewater consumed 10.8 gO_2/d, while Train B, fed filtered wastewater, consumed 8.3 gO_2/d. The reduction in oxygen demand was about 30 % compared to the control reactor.

Overall, the nitrification rates obtained during this study were between 0.1 and 1.3 g NH_4-N/m^2-d. Analysis of the denitrification rates, however, showed that denitrification rates were 10 % higher in Train B than in Train A. The average value of the SDNR is 0.77 gNOx-N/m^2-d in Train A and 0.85 gNOx-N/m^2-d in Train B. Consequently, the removal of particulate organic matter with a 33 µm filter did not have any negative impact on denitrification.

7.1.5. Pilot scale MBR process

During the fourth experiment, two pilot scale MBRs with nitrogen removal operated in parallel. One MBR train (A) was fed raw wastewater and one train (B) was fed wastewater filtered with a 33 µm filter cloth. Each MBR train was composed of two anoxic reactors of 10 L each (R1 and R2), equipped with a mechanical mixer and one aerobic reactor of 25 L (R3) with a submerged hollow fiber membrane ZeeWeed-10 at 40 nm nominal pore size. Nitrified activated sludge was recycled from Reactor 3 to Reactor 1 at twice the influent wastewater flowrate. The membrane ZW-10 was operated at normal flow for about 9.5 minutes and backwashed at twice the normal flow for about 0.5 minute.

The assessment of the MBRs' performance showed a removal efficiency at nearly the same level. For Train B, if the filter performance is not considered, the MBR removed about 90 % COD, 69 % TN and 78 % TP. However, when the filter removal efficiency is considered, the overall removal efficiencies were 94 % COD, 80 % TP and 73 % TN, which was similar to train A.

The TMP was higher for the membrane receiving raw wastewater with an average TMP of 46 ± 9 mbar. For the train treating filtered wastewater, the average TMP value was 26 ± 7 mbar. The fouling rate was higher in the reactor with raw wastewater compared to the system fed filtered wastewater.

In Train A, the sludge production was about 21.3 gTS/d compared to 31.2 gTS/d in Train B. The estimated energy production as methane gas for the MBR train without primary filtration was about 4.32 L CH_4/d /d, while the energy production was 6.35 L CH_4/d in train B with primary filtration. The calculated energy gain in this study was 47 % higher with filtration applied as primary treatment. In addition, the oxygen requirement was 35 % higher for the MBR treating raw wastewater compared to the MBR treating filtered wastewater.

The nitrification rates ranged between 0.088 and 0.204 gNH_4N/gMLVSS-d in the control MBR and between 0.072 and 0.265 gNH_4N/gMLVSS-d in the MBR fed filtered wastewater. The specific denitrification rates (SDNRs) observed in the two MBR trains varied from 0.013 to 0.060 gNO_x-N/g MLVSS-d in Train A, whereas the rates were between 0.039 and 0.415 gNO_x-N/g MLVSS-d in Train B. Overall, the removal of particulate COD with a 33 µm filter did not have a negative effect on the nitrogen removal in MBR.

7.1.6. Pilot scale SBR

The objectives of the fifth study were to evaluate the impact of filtration as primary treatment on pilot-scale SBRs with nitrogen removal and to investigate the influence of temperature and SRT on the biological processes.

The pilot scale studies were over two periods. In the first period (P1), three SBRs operated in parallel. The first SBR (R1) received raw wastewater, while the second (R2) and third SBR (R3) were fed wastewater filtered with several fine mesh sieves, from 150 µm to 1.2 µm pores through 33 µm filters. The first and second SBRs were at room temperature of about 17 °C, while the third SBR (R3) was at a lower temperature of about 8 °C. In the second period, two SBRs were R4 and R5, but at a reduced SRT compared to the first period. The first SBR (R4), was fed raw wastewater while the second SBR (R5) was fed filtered wastewater at 33 µm filters.

The performances of the 5 SBRs were evaluated based on TSS, TCOD and TN removal efficiencies. In Period 1, the control SBR and the SBR fed filtered wastewater had relatively similar removal efficiencies of TSS, COD and TN. The values were 83 %, 72 % and 56 %, respectively. In Period 2 at reduced SRT, the SBR treating filtered wastewater (R5) had better TSS, COD and TN removal compared to the control SBR (R4). R5 had a removal efficiency of 70 % TSS, 61 % COD and 72 % TN removal, compared to 57 % TSS, 56 % COD and 56 % TN removal for R4.

In general, the SBR operated at low temperature had higher effluent concentrations, compared to the SBR operated under higher temperatures. The average effluent TSS was 30 mgTSS/L at 8 °C and 14 mgTSS/L at 17 °C, while the average effluent total COD was 80 mgCOD/L at 8 °C and 50 mgCOD/L at 17 °C. The most significant effect of temperature was on nitrification, which was complete in the SBRs operated at 17 °C, while at 8 °C, only 50 % of the ammonium was nitrified. The nitrogen removal efficiency was then only 39 % at 8 °C compared to 56 % in SBRs operated at 17 °C.

Primary filtration resulted in increased sludge production and potential more biogas and energy yield. In Period 1, the SBR fed raw wastewater (R1) produced about 1.72 gTS/d of sludge compared to 2.39 gTS/d and 2.31 gTS/d, in R2 and R3, respectively. The energy production in the control SBR (R1) was 3.5 Wh/d compared to 4.9 Wh/d in R2 with primary treatment. The energy gains were higher at lower SRT as the sludge production was higher. The calculated energy generation was 5.4 Wh/d in the control SBR and 6.0 Wh/d in the SBR with primary treatment.

The SBRs treating raw wastewater (R1 and R4) required more oxygen compared to the SBRs fed filtered wastewater (R2, R3 and R5). In Period

1, the oxygen demand for the control SBR (R1) increased by 22 % compared to the SBR fed filtered wastewater (R2). In Period 2, the oxygen demand increased by 66 % for the control SBR (R4) compared to R5 fed filtered wastewater.

Consequently, the removal of particulate COD with a 33 µm filter did not have any negative impact on the TN removal in the SBR for the tested wastewater, mainly because of enough COD.

7.1.7. Steady state simulation

The steady state version of the activated sludge model 1 (ASM1) were applied to simulate the experimental results of the MBR experiments. The simulation predicted the observed effluent concentration of the soluble nitrogen compounds and the nitrogen removal efficiency quite well. Predictions of the MLSS and MLVSS were close to the measured values for the test with filtered wastewater while for the test with raw wastewater there was about 15 % difference. The good predictions of nitrogen compounds could be due to that the system had near complete nitrification and denitrification, while the difference for the MLSS and MLVSS could be due to inaccurate analyses and the fact that many parameters are involved in those predictions and could affect the result.

There is a big potential for savings in energy by applying primary treatment with filtration in wastewater treatment without affecting the downstream biological processes significantly. The results also indicate that recommendation of performing laboratory or pilot testing to determine the optimum level of primary treatment. The results also show the potential of expanding the use of experimental data by applying mathematical modelling of the system.

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Appendix

Posters and Papers

Effect of particulate organic fractions on denitrification rates using Salsnes Filter fine mesh sieves for primary treatment

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Introduction

Biological wastewater treatment for nutrient removal can be improved by taking into consideration organic materials (COD) entering the biological system. About 70 % of influent COD are associated with suspended solids (SS); therefore, selective removal of solids at primary stage with Salsnes Filter fine mesh sieves (SF) without affecting biological nutrient removal could be advantageous for wastewater operators. Less organic solids entering the biological process, means less energy consumption in the aerobic systems; consequently, a reduced operating cost and also more organic matter for energy recovery in anaerobic digestion of primary sludge. But, the optimum size of particles to be removed at the primary stage is still unknown, which form the basis of this study. Untreated wastewater was passed through different SF (Rusten and Lundar, 2006) where particles above a given size were removed and then tested for denitrification (DN).

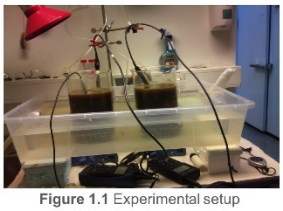

Figure 1.1 Experimental setup (Activated sludge process)

Materials and methods

In this study, denitrification rates (DNR) in activated sludge (AS) and MBBR batch tests were investigated based on the procedure outlined by Gu and Onnis-Hayden (2010). Degritted wastewater from two full-scale municipal wastewater treatment plants (wwtps) were used for the tests, Nordre Follo (NFR) and Bekkelaget (BRA) located around the Oslo region, Norway. Fractionation of wastewater were done with filter and sieves ranging from 1.2 µm to 150 µm and it was performed without mat formation, allowing only the removal of particulates larger than the filter pore size (Rusten and Lundar, 2006). Some samples were flocculated (ff) using aluminum sulfate prior to filtration to remove both colloidal and particulate COD (pCOD). Experiments were conducted using two 5-liter glass beakers (Figure 1.1), one was used for the test with filtered wastewater and the other was used for the unfiltered influent. The contents of each beaker during the experiments are shown in Table 1.1.

	Microorganism source	
Wastewater source	MLSS from BRA	Kaldnes K1 from NFR
BRA	Test 1*	Test 3**
NFR	Test 2*	Test 4**

Table 1.1 Contents of batch denitrification tests

*Mixed liquor suspended solids (MLSS)/wastewater ratio 1:1
**Filling ratio with Kaldnes K1: 50%

Results & Discussion

Influent from BRA and NFR wwtps were composed of soluble and particulate COD with a proportion of one-third and two-third respectively; in which 20-25 % of the total organic contents were present as readily biodegradable COD and about 50% as slowly biodegradable COD. Separation with bench-scale SF (Figure 1.2) allowed the removal of pCOD up to 20 to 50 % depending on the sieves and the characteristics of the influent. The effects of COD removal on the denitrification rates are shown in Table 1.2 (AS test) and Table 1.3 (MBBR Test). Overall, DN rates are higher for the batch test performed with BRA wastewater (Test 1 and 3) compared to test 2 and 4 (NFR wastewater). The difference is basically due to the wastewater composition. Regarding the effect of pCOD removal, for instance, separation of influent SS with SF (18 µm) reduced the first DN rate by 24% and the second rate 9 % for (Test 4). In general, however, the effect of removing pCOD was minor. On average 33 µm filtration reduced the first DN rate by only 5% for the AS test and the second DN rate by 6 %, while for the MBBR test the first DN rate was reduced by 12 % and by 6 % for the second DN rate.

Test 1		SDNR (gN/gVSS*.d)		
Sieves	Rate	Unfiltered ww	Filtered ww	Difference (%)
1.2 µm (ff)	SDNR 1	0.24	0.22	8.3
	SDNR 2	0.07	0.07	0
	SDNR 3	0.04	0.03	25
1.2 µm	SDNR 1	0.29	0.29	0
	SDNR 2	0.09	0.08	11
	SDNR 3	0.05	0.05	0
33 µm	SDNR 1	0.37	0.36	2.7
	SDNR 2	0.09	0.08	11
	SDNR 3	0.05	0.05	0
90 µm	SDNR 1	0.18	0.17	5.6
	SDNR 2	0.07	0.07	0
	SDNR 3	0.04	0.04	0

Test 2		SDNR (gN/gVSS*.d)		
Sieves	Rate	Unfiltered ww	Filtered ww	Difference (%)
1.2 µm (ff)	SDNR 1			
	SDNR 2	0.08	0.06	25
	SDNR 3	0.04	0.03	25
1.2 µm	SDNR 1	0.10	0.10	0
	SDNR 2	0.07	0.05	29
	SDNR 3	0.05	0.03	40
33 µm	SDNR 1	0.14	0.13	7.1
	SDNR 2	0.07	0.07	0
	SDNR 3	0.05	0.05	0
150 µm	SDNR 1	0.08	0.10	-25
	SDNR 2	0.06	0.06	0
	SDNR 3	0.04	0.04	0

Table 1.2 Specific denitrification rates (SDNR) from activated sludge process

Test 3		SDNR (gN/m2.d)		
Sieves	Rate	Unfiltered ww	Filtered ww	Difference (%)
1.2 µm (ff)	SDNR 1	2.69	2.43	9.7
	SDNR 2	2.36	1.94	18
	SDNR 3			
1.2 µm	SDNR 1	2.23	1.97	12
	SDNR 2	1.63	1.56	4.3
	SDNR 3	1.15	1.13	1.7
18 µm	SDNR 1	1.22	1.23	-0.82
	SDNR 2	1.04	0.99	4.8
	SDNR 3	0.78	0.83	-6.4
33 µm	SDNR 1	2.33	2.10	10
	SDNR 2	1.60	1.49	6.9
	SDNR 3			
90 µm	SDNR 1	2.17	2.20	-1.4
	SDNR 2	1.88	1.76	6.4
	SDNR 3	1.24	1.17	5.7

Test 4		SDNR (gN/m2.d)		
Sieves	Rate	Unfiltered ww	Filtered ww	Difference (%)
1.2 µm (ff)	SDNR 1	0.81	0.80	1.2
	SDNR 2	0.61	0.60	1.6
	SDNR 3	0.34	0.30	12
1.2 µm	SDNR 1	1.23	1.19	3.3
	SDNR 2	0.90	0.79	12
	SDNR 3			
18 µm	SDNR 1	2.40	1.82	24
	SDNR 2	1.83	1.67	8.7
	SDNR 3	1.39	1.28	7.9
33 µm	SDNR 1	2.43	2.09	14
	SDNR 2	1.63	1.56	4.3
	SDNR 3	1.11	1.14	-2.7
90 µm	SDNR 1	0.97	0.94	3.1
	SDNR 2	0.59	0.57	3.4
	SDNR 3	0.35	0.35	0

Table 1.3 Specific denitrification rates (SDNR) from MBBR process

Conclusions

In sum, the effects of pCOD removal on DN rates were not very substantial with the AS and MBBR batch tests. But more laboratory tests are in progress in order to see the change in DN rates with a continuous feeding of unfiltered and filtered wastewater, respectively.

Figure 1.2 Bench-scale SF

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Impact of Salsnes Filter Fine Mesh Sieves as Primary Treatment on Nutrient Removal in Membrane Bioreactors

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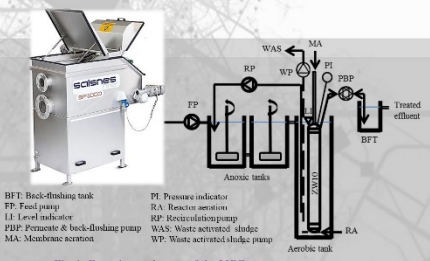
1. OBJECTIVES

To define the particle size cut-off of organic matter prior to biological nutrient removal (BNR).

In order to

- Reduce the energy consumption in the aerobic process,
- Increase the production of renewable energy from the removed solids.



BFT: Back-flushing tank
FP: Feed pump
LI: Level indicator
PBP: Permeate & back-flushing pump
MA: Membrane aeration
PI: Pressure indicator
RA: Reactor aeration
RP: Recirculation pump
WAS: Waste activated sludge
WP: Waste activated sludge pump

Fig. 1: Experimental setup of the MBR process

2. PRELIMINARY STUDIES

Earlier studies have shown using 33 µm Salsnes Filter fine mesh sieve (SF) to be optimum for particle removal prior to BNR (Razafimanantsoa et al., 2014 a,b).

3. OPERATING PARAMETERS

Table 1. MBRs characteristics (Nordre Follo wwtp, Norway)

	Train A	Train B
Feed	SF 2 mm	SF 33 µm
Feed Flow	5 L/h	
Recycle Flow	10 L/h	
Temperature	16 – 21 °C	
pH	7 - 8	
DO (aerobic)	~ 4 mg/L	
Reactor volume	10 L (anoxic), 25 L (aerobic)	
MLSS	~ 5 g/L (anoxic), ~ 7 g/L (aerobic)	

Membrane ZeeWeed-10 ™ (GE Technology)
Pore size: 40 nm
Permeate flow: 6 L/h
Back-flush flow: 12 L/h
Cycle: 9.5 min (normal flow), 0.5 min (back-flush)
Operating TMP: < 300 mbar

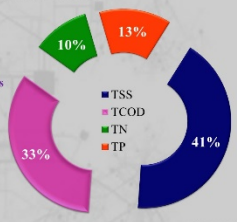


Fig. 2: Removal efficiencies of influent wastewater with SF 33 µm

4. RESULTS AND DISCUSSIONS

- ❑ Similar removal with regard to TSS, TCOD and TP in both MBRs
- ❑ Reduction of 5 % in term of TN removal for MBR fed with SF 33 µm (sCOD was 17% lower in Train B)

Table 2. Concentrations and removal efficiencies of the two treatment trains using MBRs. All numbers are given as averages

Parameters	Train A (2 mm)			Train B (33 µm)		
	Feed (mg/L)	Eff. (mg/L)	Rem. (%)	Feed (mg/L)	Eff. (mg/L)	Rem. (%)
Total COD	522	32.4	93.6	349	32.1	93.6
Soluble COD*	168	29.7	81.4	145	29.1	81.7
TSS	275	0.27	100	161	0.04	100
Total N	43.2	11.1	73.7	38.7	13.4	68.5
NH_4-N*	31.5	0.24	99.4	30.1	0.32	99.2
Total P	4.26	0.82	79.8	3.70	0.74	81.1

* After filtration through 1.2 micron Whatman GF/C glass fiber filter

- ❑ Higher trans-membrane pressure (TMP) was observed for the membrane in Train A (46 mbar vs 26 mbar in Train B)
- ❑ Higher sludge production in Train B (30 gTSS/d vs 24 gTSS/d), nearly half of it was produced during the primary treatment.

5. KEY TAKEAWAYS

- Removal of particles with SF 33 µm did not impend the downstream biological process (ability to reduce energy requirement in the aerobic reactor)
- Higher sludge production (ability to produce more biogas)
- Reduced fouling rate of the membrane due to the small amount of organics entering the biological process, which in the long run will reduce the membrane maintenance cost.

6. REFERENCES

Razafimanantsoa V. A., Ydstebø L., Bilstad T., Sahu A. K., Rusten B. (2014 a) Effect of Selective Organic Fractions on Denitrification Rates Using Salsnes Filter as Primary Treatment. Wat. Sci. Tech., 69 (9), 1942-1948.

Razafimanantsoa V. A., Ydstebø L., Bilstad T., Sahu A. K., Rusten B. (2014 b) Impact of selective size distribution of influent suspended solids on downstream biological processes. *Proceedings, A0969*. IWA Conference on Pretreatment of Water and Wastewater, Shanghai, China, 18-21 May.

Paper 1:
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Water Science and Technology, 69 (9), 1942-1948.

Effect of Selective Organic Fractions on Denitrification Rates Using Salsnes Filter as Primary Treatment

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ABSTRACT

The purpose of this project was to investigate the effect of selective particle removal during primary treatment on downstream biological nutrient removal processes. Bench-scale Salsnes Filter (SF) fine mesh sieves were used as a primary treatment to obtain different organic fractions to test the effect on denitrification. Activated sludge (AS) and moving bed biofilm reactor (MBBR) anoxic tests were performed on municipal wastewater collected from two full-scale wastewater treatment plants (wwtps) located around the Oslo region (Norway). About 43% of the suspended solids in ww were less than 18 μm, and 14 % were between the size 18 μm to 150 μm. The effect of particulate COD (pCOD) removal on denitrification rates was very minor.

Key words: Denitrification, organic matter, primary treatment, wastewater treatment

INTRODUCTION

Biological nitrogen removal (BNR) is the most reliable and cost-effective process for nitrogen removal from wastewater (Abufayed & Schroeder, 1986). Nitrogen compounds from wastewater are removed by a combination of two processes of nitrification and denitrification (Wang & Yang, 2004). These processes need sufficient organic matter (COD) to provide the energy required for the removal of nitrogen from wastewater (Tas et al., 2009). The origin of organic matter might be from the influent wastewater or self-generated by microorganisms through lysis or provided externally. Since a major part of the influent COD consists of particulate forms (about 70%), which are linearly related to the total suspended solids (TSS), advanced particle removal during pretreatment will result in a lower organic load on the biological treatment process (Abufayed & Schroeder, 1986; Henze et al., 2008, van Nieuwenhuijzen, 2000).

The evaluation of different solids separation technologies for primary treatment, such as, dissolved air flotation, large septic tanks, primary clarifiers, deep bed filtration and several types of sieve based technologies, including Salsnes Filter (SF), revealed that the SF fine mesh sieve is the most suitable technology to achieve a removal of at least 50 % of suspended solids and 20 % of the organic material measured as BOD as required by the European Union for primary treatment (Bixio et al., 2000; Rusten & Ødegaard, 2006). A saving of about 50 % of the costs for primary treatment was achieved with SF fine mesh sieve compared to the conventional primary clarifier, and a reduction of about one third of the aeration power consumption was observed at an activated sludge plant upgraded with SF fine mesh sieve primary treatment (Rusten, 2005). Moreover, the anaerobic digestion of the primary sludge with the excess sludge from the biological processes, to produce biogas for power generation, contributed to the overall operational cost savings (Bixio et al., 2000).

The resultant effective pore size of the SF screen can reach as low as 10-20 μm when operated properly, allowing a removal of about 40 % of the total COD (Newcombe et al., 2011). However, according to Tas et al. (2009), removal of such amount of COD can have a significant effect on denitrification potential of the treatment plant. In that aspect, the

objective of this study was to investigate the effect on denitrification of the removal of a certain particle fraction by means of SF fine mesh sieves, in order to define the optimum particle size cut-off for particulate COD removal prior to BNR. Therefore, the following activities were performed: (1) characterization of influent wastewater before and after physical separation with GF/C filters and SF fine mesh sieves; (2) nitrate utilization rate (NUR) tests with activated sludge (AS) and moving bed biofilm reactor (MBBR) processes; (3) and finally, the computation of the denitrification rates.

METHODOLOGY

Sampling location

Wastewater from two municipal wastewater treatment plants (wwtps) around the Oslo region (Norway) were investigated during this study, which are Bekkelaget (BRA) and Nordre Follo (NFR) wwtps. These two wwtps served also as source of nitrifying mixed liquor and Kaldnes K1 biofilm carriers, which were used during the laboratory tests. Grab samples of influent wastewater were collected just after the grit removal section at both wwtps. Recycled activated sludge (RAS) was taken from the recycle line from the secondary clarifiers at BRA, while the biofilm carriers, Kaldnes K1, were taken from the anoxic Reactor 1 in Line 2 at NFR.

Separation process

A standard vacuum filter apparatus, typically used for suspended solids analysis, and a bench scale SF (Rusten & Lundar, 2006) were used for the physical separation of wastewater in order to get samples with different concentration of particulates organic matter. Separation process with SF was carried out by filtering an influent grab sample through different sieves, from 150 μm to 18 μm pore size. It was necessary to perform all tests without mat formation, allowing only the removal of particulates above a given size to provide a good picture of the size distribution of particles (Rusten & Lundar 2006).

Whatman GF/C filters (1.2 μm pore size) were used on the vacuum filter apparatus to provide samples free of suspended solids. Some of the samples were pre-flocculated (ff) with aluminum sulfate (alum) before separation with GF/C filter to remove both particulates and colloids from the influent wastewater. Flocculation was carried out with an alum dose of 12 mg-Al/L, a rapid mix at 150 rpm for 1 min and a flocculation at 20 rpm for 15 min. Since alum has a tendency to decrease the pH, small aliquots of 1M NaOH was added until the set-point pH was attained (pH=7).

During this study the SF fine mesh sieve with 150 μm pore size was only used once (Test 1), as this was the first experiment performed. Since there was not much difference in the results with untreated wastewater and 150 μm filtered wastewater, sieves with lower openings were used during the rest of the tests.

Wastewater characterization

Good characterization of wastewater is very important in order to predict the removal efficiencies that can be expected for a given sieve (De Lucas et al., 2005; Rusten & Lundar, 2006). During each anoxic test, influent wastewater was characterized before and after the separation process. Suspended solids were analyzed according to *Standard Methods* (2005),

while chemical parameters such as COD, ammonium (NH_4-N), nitrate (NO_3-N), nitrite (NO_2-N) and orthophosphate (PO_4-P) were analyzed using the Dr Lange cuvette test kits and a DR 5000 UV-Vis Spectrophotometer (Hach Lange, Germany).

Experimental setup

Two bioreactors were set up during each test, where degritted influent wastewater was used in Reactor 1 and filtered wastewater in Reactor 2. The experiments were carried out from August 14 through October 9, 2012. The setup of the NUR experiment and the computation of the specific denitrification rate (SDNR) were adopted from the protocol outlined by Gu and Onnis-Hayden (2010). Activated sludge and MBBR anoxic tests were performed during this study. Table 1 summarizes the characteristics of the different tests.

Table 1 Experimental plan

Process	Test	Reactor [4] composition	Sieves tested
AS[1]	Test 1	ww from NFR + RAS from BRA	1.2 µm (ff)[3], 1.2 µm, 33 µm, 150 µm
	Test 2	ww from BRA + RAS from BRA	1.2 µm (ff)[3], 1.2 µm, 33 µm, 90 µm
MBBR[2]	Test 3	ww from NFR + Kaldnes K1 from NFR	1.2 µm (ff)[3], 1.2 µm, 18 µm, 33 µm, 90 µm
	Test 4	ww from BRA + Kaldnes K1 from NFR	1.2 µm (ff)[3], 1.2 µm, 18 µm, 33 µm, 90 µm

[1] volume ratio of wastewater and RAS of 1:1
[2] volumetric filling of about 50 % of Kaldnes K1 biofilm carriers
[3] ff: flocculated with alum prior to filtration
[4] Reactor volume : 3 L

Adjustement of SDNR for temperature effect and active biomass

Three rates were observed during the NUR test. The first and highest rate reflects the denitrification when utilizing rbCOD (Rate 1). Then, pCOD will be used by denitrifiers to continue the nitrogen removal process, resulting in a second lower rate (Rate 2). Finally, the third and lowest rate is the endogenous denitrification rate (Rate 3) (Kujawa & Klapwijk, 1999).

Studies have shown that different temperature coefficients Θ apply to specific denitrification rates associated with the different COD fractions: 1.20, 1.08 and 1.03 for the first, second and third rates, respectively (Clayton et al., 1991; Henze et al., 2008). Fraction of the active biomass was estimated from the measured test mixed liquor volatile suspended solids (MLVSS) concentration and plant data related to BOD or biodegradable COD removed (Metcalf & Eddy 1991).

RESULTS AND DISCUSSIONS

Wastewater characterization

Characterization results of the influent wastewater during the experimental period (August 14 – October 9, 2012) are outlined in Table 2. The soluble COD (sCOD) fractions, which contain the readily biodegradable substrate, made up only about one third of the total substrate. GF/C filter of 1.2 µm pore size was used to differentiate the soluble and particulate COD. The ratio of particulate COD/VSS found at the NFR and BRA wwtps were 1.50 ± 0.16 and 1.35 ± 0.17, respectively. The average results obtained in both wwtps compare well with the typical values suggested for municipal wastewater composition (Tas et. al, 2009).

Table 2 Characteristics of influent wastewater from BRA and NFR wwtps during the experimental period (August 14 – October 9, 2012), average results given with the standard deviation, Number of samples per wwtp = 9

Parameter	BRA wwtp		NFR wwtp	
	Range	Mean	Range	Mean
TSS (mg/l)	94 - 528	281 ± 110	132 - 262	218 ± 38
VSS (mg/l)	72 - 457	245 ± 105	105 - 228	190 ± 38
VSS / TSS		0.86 ± 0.07		0.86 ± 0.03
Total COD (mg/l)	143 - 801	494 ± 183	231 - 577	460 ± 100
pCOD (mg/l)	93 - 547	315 ± 127	156 - 375	299 ± 66
sCOD (mg/l)[1]	50 - 271	179 ± 69	75 - 205	161 ± 46
rbCOD (%)[2]		25 ± 4		20 ± 4
sbCOD (%)[2]		52 ± 5		51 ± 3
NO_3-N (mg/l)	0.26 – 1.16	0.57 ± 0.27	0.27 – 1.81	0.74 ± 0.57
NO_2-N (mg/l)	0.01 – 0.10	0.07 ± 0.04	0.02 – 0.17	0.08 ± 0.05
NH_4-N (mg/l)	9 – 24	18 ± 6	8 – 27	21 ± 6
PO_4-P (mg/l)	1 – 5	2 ± 1	1 – 5	3 ± 1

[1] after filtration through 1.2µm GF/C Whatman glass fiber filter

[2] rbCOD: readily biodegradable COD; sbCOD: slowly biodegradable COD (Melcer et al., 2003)

Separation performance

The removal efficiencies, in term of COD and TSS, associated with separation by means of GF/C filters and SF fine mesh sieves are illustrated in Figure 1. The graph combined both results from the separation of wastewater collected at the BRA and NFR wwtps, given as averages. The graph showcased that about 43 % of the suspended solids (corresponding to 58 % of total COD) in the influent wastewater was smaller than 18 µm and about 43 % (corresponding to 28 % of total COD) was larger than 150 µm. From 18 µm to 150 µm, the amount of particles in the tested wastewaters was only about 14 % (the same for total COD). However, the amount of solids removed depends not only upon the sieve openings, but also upon the particle size distribution in the influent wastewater (Newcombe et al., 2011).

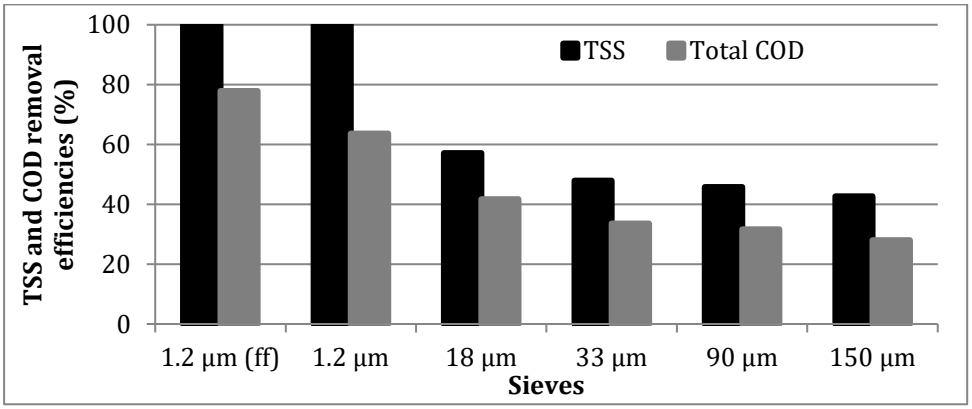

Figure 1 TSS and COD removal efficiencies after separation with different sieves, given as averages

Characterization of the influent wastewater based on COD/TSS ratio is an important parameter for understanding and interpretaing the fate of different COD fractions before biological treatment. It was observed during the separation process that SF fine mesh sieves decreased the pCOD/TSS ratio (Table 3), indicating the removal of a TSS with high COD/TSS ratio.

Denitrification rate

The specific denitrification rates (SDNR) obtained with activated sludge and MBBR processes anoxic tests are outlined in Table 3. The SDNRs were calculated based on the biomass (VSS) concentration in the reactor for the activated sludge process (Kapagiannidis et al., 2006), while biofilm surface area was used in the MBBR process (Rusten et al., 2000).

The results summarized the different rates and the corresponding influent total COD to nitrate ratio (C/N) for the different tests. In general, the SDNRs were higher during the tests performed with the wastewater from BRA wwtp in comparison to the NUR tests using NFR wastewater in both processes. For instance, with activated sludge process, the first rates were all above 0.17 gNO_3-N/gVSS.d in Test 2 compared to a maximum of 0.14 gNO_3-N/gVSS.d in Test 1. However, the second and third rates were similar in both tests, about 0.08 and 0.05 gNO_3-N/gVSS.d, respectively. The difference observed on the first SDNR might be attributed to the amount of rbCOD available in each influent wastewater. About 25 % of the total COD was readily biodegradable in the wastewater from BRA, while it is only about 20% in the wastewater from NFR.

The SDNRs results observed in the AS process fed with unfiltered wastewater were similar to the results found in literature, the first SDNR ranges from 0.07 – 0.32 gNO_3-N/gVSS.d (Kapagiannidis et al,2006; Barnard and Meiring, 1977), 0.08 gNO_3-N/gVSS.d for the second rate (Barnard & Meiring, 1977) and between 0.04 - 0.05 gNO_3-N/gVSS.d for the endogenous rate (Randall et al., 1992). In the MBBR process, the SDNRs for unfiltered wastewater were in accordance with the literature data, overall rates of 0.8 - 1 gNO_3-N/m^2.d (at 20 °C, Θ: 1.05) were observed in Rusten et al. (1995, 2000).

Due to the large variations in wastewater composition and other operating parameters it was difficult to compare results from one test to another. For this reason one reactor (R1) has been operated with unfiltered wastewater in all tests, for direct comparison with a reactor (R2) receiving filtered wastewater. The comparison of the two reactors showed that the SDNRs results were quite similar in both treatment processes, despite the large difference on the influent total COD/N. It was observed earlier that soluble COD was a small fraction of the whole, then the denitrification was mostly driven by hydrolyzed particulate COD (Abufayed & Schroeder, 1986). In the case of wastewater with soluble COD fraction only, the required carbon source must be from the hydrolysis of the biomass itself as no external carbon was added during the experiment.

In all these experiments, it can be concluded that, endogenous carbon was utilized to drive the denitrification reaction, leading to the similarity in denitrification rates. Based on the results, no significant effect on denitrification rates were observed while removing part of the influent organic material. Newcombe et al. (2011) reached the same conclusion while removing 45% of the TSS with SF fine mesh sieve from the influent and still achieved an effluent TN less than 10 mg/l.

CONCLUSIONS

Activated sludge and MBBR anoxic batch tests were performed to investigate the impact on denitrification of removal certain fractions of particulate COD by means of SF fine mesh sieves. Separation with SF allowed the removal of about 57 % of the suspended solids

(corresponding to 42 % of total COD) with 18 µm openings and about 43 % (corresponding to 28 % of total COD) with 150 µm pore size. From 18 µm to 150 µm, the amount of particles in the tested wastewaters was only about 14 %. It was observed that, solids removal during primary treatment had no significant effect on the denitrification process downstream. But, further studies should be carried out as many factors could affect the denitrification, making it difficult to document the effects of a single parameter (Rusten et al., 2000).

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Table 3 SDNRs with the corresponding influent C/N and pCOD/TSS ratios from NUR tests with AS (Tests 1 & 2) and MBBR (Tests 3 & 4) processes. Rates calculated for a temperature of 20˚C, using the temperature correction procedure described under "Methodology". BRA ww and NFR ww indicates wastewater collected at the BRA and NFR wwtps, respectively.

Test 1 (NFR ww)		Unfiltered ww			Filtered ww		
Sieves	**Rate**	**SDNR[1]**	**pCOD/TSS[3]**	**C/N[4]**	**SDNR[1]**	**pCOD/TSS[3]**	**C/N[4]**
1.2 µm (ff)	1	---	1.207	7.19	---	---	1.29
	2	0.08			0.06		
	3	0.04			0.03		
1.2 µm	1	0.10	1.250	8.40	0.10	---	3.41
	2	0.07			0.05		
	3	0.05			0.03		
33 µm	1	0.14	1.477	7.75	0.13	0.920	5.41
	2	0.07			0.07		
	3	0.05			0.05		
150 µm	1	0.08	1.227	7.67	0.10	1.473	6.10
	2	0.06			0.06		
	3	0.04			0.04		

Test 2 (BRA ww)		Unfiltered ww			Filtered ww		
Sieves	**Rate**	**SDNR[1]**	**pCOD/TSS[3]**	**C/N[4]**	**SDNR[1]**	**pCOD/TSS[3]**	**C/N[4]**
1.2 µm (ff)	1	0.24	1.266	9.66	0.22	---	2.38
	2	0.07			0.07		
	3	0.04			0.03		
1.2 µm	1	0.29	1.037	11.99	0.29	---	4.33
	2	0.09			0.08		
	3	0.05			0.05		
33 µm	1	0.37	0.936	8.56	0.36	0.812	6.38
	2	0.09			0.08		
	3	0.05			0.05		
90 µm	1	0.18	1.284	9.71	0.17	1.091	5.86
	2	0.07			0.07		
	3	0.04			0.04		

Test 3 (NFR ww)		Unfiltered ww			Filtered ww		
Sieves	**Rate**	**SDNR[2]**	**pCOD/TSS[3]**	**C/N[4]**	**SDNR[2]**	**pCOD/TSS[3]**	**C/N[4]**
1.2 µm (ff)	1	0.81	1.180	8.02	0.80	---	1.45
	2	0.61			0.60		
	3	0.34			0.30		
1.2 µm	1	1.23	1.320	19.18	1.19	---	7.52
	2	0.90			0.79		
	3	----			----		
18 µm	1	2.40	1.443	16.93	1.82	1.000	9.87
	2	1.83			1.67		
	3	1.39			1.28		
33 µm	1	2.43	1.625	18.03	2.09	1.150	11.06
	2	1.63			1.56		
	3	1.11			1.14		
90 µm	1	0.97	1.178	16.01	0.94	1.320	12.79
	2	0.59			0.57		
	3	0.35			0.35		

Test 4 (BRA ww)		Unfiltered ww			Filtered ww		
Sieves	**Rate**	**SDNR[2]**	**pCOD/TSS[3]**	**C/N[4]**	**SDNR[2]**	**pCOD/TSS[3]**	**C/N[4]**
1.2 µm (ff)	1	2.69	1.421	18.18	2.43	---	5.43
	2	2.36			1.94		
	3	---			---		
1.2 µm	1	2.23	1.077	12.13	1.97	---	4.23
	2	1.63			1.56		
	3	1.15			1.13		
18 µm	1	1.22	0.983	4.78	1.23	0.925	2.91
	2	1.04			0.99		
	3	0.78			0.83		
33 µm	1	2.33	0.912	15.08	2.10	1.215	10.79
	2	1.60			1.49		
	3	---			---		
90 µm	1	2.17	1.200	13.45	2.20	1.049	8.65
	2	1.88			1.76		
	3	1.24			1.17		

[1] SDNR, expressed as g NO3-N/g VSS.d

[2] SDNR, expressed as g NO3-N/m^2.d

[3] pCOD/TSS expressed as g pCOD/g TSS

[4] C/N, expressed as g total COD/g NO3-N

Paper 2:
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Pilot-scale scale study to investigate the impact of rotating belt filter upstream of a MBR for nitrogen removal

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ABSTRACT

The goal of this study was to investigate what kind of impact the removal of particulate organic matter with 33 µm rotating belt filter (RBF) (as a primary treatment) will have on the membrane bioreactor (MBR) performance. Two small MBR pilot plants were operated in parallel, where one train treated 2 mm screened municipal wastewater (Train A) and the other train treated wastewater that had passed through a RBF with a 33 µm filter cloth (Train B). The RBF was operated without a filter mat on the belt. About one third of the organic matter was removed by the fine mesh filter. The assessment of the overall performance showed that the two pilot plants achieved approximately the same removal efficiencies with regard to TSS, COD, total phosphorus and total nitrogen. It was also observed that the system with 33 µm RBF as a primary treatment produced more sludge, which could be used for biogas production, and required about 30 % less aeration downstream. Transmembrane pressure (TMP) was significantly lower for the train receiving 33 µm primary treated wastewater compared to the control receiving 2 mm screened wastewater.

Keywords: *Primary treatment, organic matter, membrane bioreactor, nitrogen removal, rotating belt filter*

INTRODUCTION

The use of membrane technology, combining conventional activated sludge with low pressure membrane filtration, has been proven to be a feasible and efficient method to achieve high effluent quality in biological wastewater treatment (wwt). However, membrane fouling remains a major drawback of MBR, as it significantly reduces the membrane performances and membrane lifespan, leading to an increase in maintenance and operating costs (Iorhemen *et al.* 2016). It is well accepted that removal of particulate and colloidal fractions will lead to considerable operational savings in the downstream aerobic biological processes while allowing the recovery of energy in the form of methane via anaerobic sludge treatment processes. Moreover, carbon management plays a very important role for biological nutrient removal processes where certain carbon fractions are preferred for optimal performance without the addition of external carbon source (Ho et al. 2017).

Newcombe *et al.* (2011) found through literature that 15 – 20 µm was the possible new particle size delineation for effective biological treatment, after evaluating in bench-scale studies the impact of primary treatment on the size distribution of particles prior to biological treatment. Based on the study by Razafimanantsoa *et al.* (2014a), no huge impact on the denitrification rates was observed with the anoxic batch tests fed with wastewater passed through filters from 150 µm to 1.2 µm openings. However, a clear particle size cut-off was determined with the laboratory scale SBRs (Razafimanantsoa *et al.* 2014b). The comparative

studies showed that all SBRs approximately had the same pollutants removal efficiencies, except for the reactors fed with filtrate passed through filters below 33 µm, where a reduction of the nitrogen removal efficiencies were noticed. Thereby, the objective of this present study was to determine if the removal of particulate organic matter with a 33 µm filter would affect the nitrogen removal performance of the MBR. The specific objectives were to evaluate the impact of fine mesh filters for particle removal upstream of the MBR (i.e. removal efficiencies of TSS, COD, nitrogen and phosphorus in the MBR); and further to determine the total sludge production and the oxygen demand for the aerobic process.

MATERIALS AND METHODS

Sampling and study location

The experiment was performed at Nordre Follo wastewater treatment plant (wwtp) (Akerhus, Norway). Influent wastewater was collected just after the coarse screens using a grinder pump, then passed through a 2 mm screen (Train A) or a fine mesh filter cloth of 33 µm (Train B) mounted on a commercial rotating belt filter (SF 1000 machine) (Salsnes Filter AS, Norway). Screened and filtered wastewater, respectively, were stored in tanks that were filled three times per week. The mixed liquor suspended solids (MLSS) used as seed activated sludge for the MBRs was collected from the return activated sludge line at Bekkelaget wwtp (Oslo, Norway).

Experimental setup

Two pilot scale MBRs were operated in parallel during the experiment. As illustrated in Figure 1, each MBR Train was composed of two anoxic reactors of 10 L (R1 and R2), equipped with a mechanical mixer rotating at about 220 rpm, one aerobic reactor of 25 L (R3), and a submerged hollow fiber membrane ZeeWeed-10 (ZW10, Zenon Environmental Systems Inc., Oakville, Ontario, Canada) with a 40 nm nominal pore size. Nitrified activated sludge was recycled from R3 to R1 at twice the flow of the influent wastewater (Figure 1). The membrane ZW10 was operated at normal flow for about 9.5 minutes and backwashed for about 0.5 minute. All pumps (feed, recycle and permeate) were controlled by a programmable logic controller (PLC). A pressure transmitter and a dissolved oxygen probe were also connected to the PLC to record continuously the transmembrane pressure (TMP) and dissolved oxygen (DO) in the aerobic reactor.

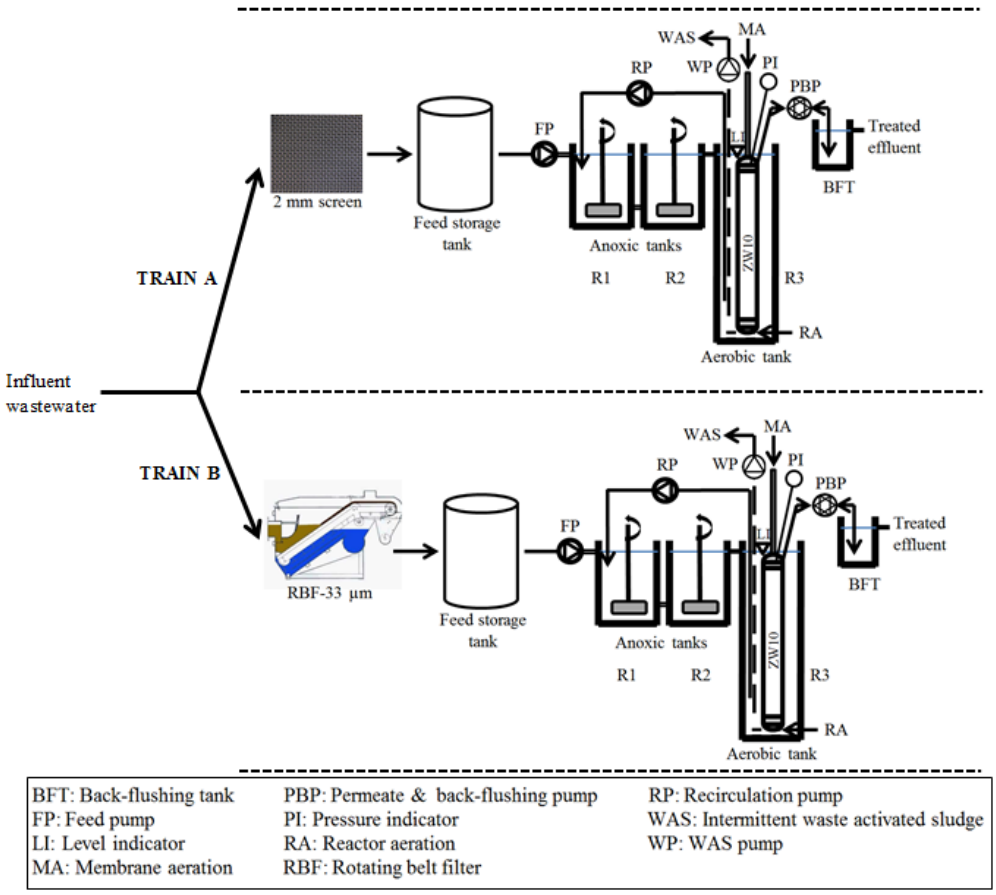

Figure 1. Simplified flowsheet of the pilot scale MBRs

Operating parameters

The MBRs were operated at an influent flow rate of about 5 L/h and a recycle flow rate of 10 L/h. The hydraulic retention time was 9 h for both trains. The MLSS in the anoxic reactors was around 5 g/L, while the MLSS in the aerobic reactors were 7 g/L. The temperature of the reactors varied from 16 to 21°C due to the seasonal variation of wastewater temperature and the room temperature at the wwtp. The pH in the three biological reactors was neutral throughout the study period for both MBR Trains and the DO in the aerobic reactors were about 4 mg/L. The operating parameters of the MBR systems are summarized in Table 1.

Table 1. MBRs operating parameters

Parameter	Train A (2 mm)	Train B (33 µm)
Feed flow (L/h)	5.2 ± 0.2	5.2 ± 0.2
Recirculation flow (L/h)	10.3 ± 0.3	10.1 ± 0.4
Permeate flow (L/h)	5.9 ± 0.2	5.9 ± 0.2
Backflush flow (L/h)	~ 12	~ 12
MLSS - R1, 2 (mg MLSS/L)	5165 ± 1021	4724 ± 935
MLVSS - R1, 2 (mg MLVSS/L)	4063 ± 729	3176 ± 521
MLSS - R3 (mg MLSS/L)	7196 ± 1263	6862 ± 1188
MLVSS – R3 (mg MLVSS/L)	5487 ± 945	4877 ± 837
SRT (d)	13.7 ± 2.7	16.8 ± 3.3
Aerobic SRT (d)	8.7 ± 1.7	10.8 ± 2.0
C/N ratio (g TCOD/g TN)	12.4 ± 2.8	9.4 ± 2.3
C/N ratio (g sCOD/g TN) *	4.0 ± 1.1	3.9 ± 1.2
Temperature R1 (°C)	18.7 ± 2.3	18.6 ± 2.3
Temperature R2 (°C)	18.7 ± 2.4	18.6 ± 2.3
Temperature R3 (°C)	18.8 ± 2.4	18.7 ± 2.3
pH - R1	7.3 ± 0.1	7.3 ± 0.2
pH - R2	7.3 ± 0.1	7.3 ± 0.1
pH - R3	7.1 ± 0.2	7.0 ± 0.3
DO - R1, 2 (mg/L)	< 0.02	< 0.02
DO - R3 (mg/L)	3.8 ± 1.4	4.1 ±1.2
Mixer speed R1 (rpm)	216 ± 33	216 ± 46
Mixer speed R2 (rpm)	226 ± 44	219 ± 40

*sCOD = soluble COD, measured after filtration through 1.2 µm filter

Analytical methods

Influent wastewater, filtrate from 33 µm belt filter and effluent from the pilot scale MBRs were analyzed four to six times per week for total suspended solids (TSS) according to Standard Methods (APHA, 2005). COD, nitrogen and phosphorus compounds were analyzed using the Dr Lange cuvette test kits and a DR 2500 UV-Vis Spectrophotometer (Hach Lange, Germany). The same analyzes were done for the samples taken twice per week from each biological reactor. Whatman GF/C glass fibre filters, with an average pore size of 1.2 µm, were used for filtration of samples and measurement of TSS. Temperature, DO and pH were measured daily in all biological reactors using a calibrated WTW multi-parameter meter, model 3420 (Weilheim, Germany).

RESULTS AND DISCUSSIONS

RBF performance

The 33 µm RBF filter removed about 42 % of TSS, 33 % of COD, 12 % of the total nitrogen (TN) and 14 % of the total phosphorus (TP). Ruiken *et al.* (2013) has determined that most of the solids removed by RBF (350 µm) were paper fibers. The main factors that could affect the filter performance were the influent wastewater characteristics, especially the particle size distribution, the filter mesh size and the hydraulic flow through the filter cloth, referred to as filter rate (Rusten *et al.* 2017). When comparing the present performance to literature data, it must be noted that the RBF in the present study was operated without a filter mat on the belt. Belt filters of 350 µm were typically used in various wwtps and with the right operating

conditions the belt filters could achieve very good pollutants removals (Rusten *et al.* 2017; Franchi & Santoro 2015). At the beginning of the operation the filter act as a sieve removing only solids smaller than the belt size, but as the operation progresses solids start to buildup on the surface of the RBF forming a so-called "mat" reducing the filter nominal pore size. Wet sludge retained on the filter cloth will be blown off the belt by an airknife mounted on the back of the filter as the belt rotates. In addition, scrappers and intermittent water spray could also be used to clean the filter (Rusten *et al.*, 2017).

Biological process

The effect of the biological process can be evaluated through the degree of pollutants removals (%). Table 2 shows the concentration of pollutants in the influent and effluent wastewater, as well as the removal efficiencies in the two MBR Trains. The removal efficiencies were determined by mass balance, first by calculating the average daily value for a given pollutant and then calculate the pollutants removal efficiencies based on the average values. As expected, the two MBR systems produced high effluent quality, free of particles and without contribution of particulate phosphate and nitrogen from the suspended solids due to the use of membrane as final solid-liquid separation.

The assessment of the performance of the MBR Trains revealed that the removal efficiencies for different pollutants were the same. The two trains removed 94 % of the COD, 80 % of the TP and more than 70 % of the TN. The performance of MBR varies from plant to plant but excellent pollutants removals were observed in several papers related to the treatment of municipal wastewater: 95 – 98 % for COD and 80 – 84 % for TN (Galil *et al*. 2009; Bracklaw *et al.* 2007; Lobos *et al*. 2006; Jiang *et al.* 2004). The effectiveness of the MBR systems can be affected by several factors such as MLSS concentrations to which the membrane modules were exposed, the operating temperature, DO levels, pH and organic loading rates (Johir *et al.* 2012). During this study, even though the organic load was reduced with a 33 µm RBF filter (Train B) the overall performance of the MBR system was not affected as observed in Table 2.

Table 2. Concentrations and removal efficiencies of the two MBRs

Parameter	Train A (2 mm)			Train B (33 µm)		
	Feed (mg/L)	Effluent (mg/L)	Rem. (%)	Out RBF (mg/L)	Effluent (mg/L)	Rem.* (%)
Total COD	522 ± 134	32.4 ± 7.2	**93.8**	349 ± 88	32.1 ± 7.5	**93.8**
sCOD	168 ± 47	29.7 ± 7.3	**82.3**	145 ± 35	29.1 ± 7.7	**82.7**
TSS	275 ± 99	0.27 ± 1.46	**99.9**	161 ± 69	0.04 ± 0.21	**99.9**
VSS	216 ± 67	0.00 ± 0.00	**100**	128 ± 53	0.00 ± 0.00	**100**
Total N	43.2 ± 12.0	11.3 ± 3.5	**74.2**	38.7 ± 11.8	11.3 ± 2.5	**74.2**
NH_4-N	31.5 ± 10.5	0.24 ± 0.74	**99.3**	30.1 ± 9.9	0.32 ± 1.37	**99.0**
NO_3-N	0.41 ± 0.15	8.63 ± 3.0	**---**	0.40 ± 0.15	10.0 ± 3.2	**---**
NO_2-N	0.03 ± 0.01	0.14 ± 0.22	**---**	0.03 ± 0.05	0.13 ± 0.19	**---**
Total P	4.26 ± 1.51	0.82 ± 0.62	**80.8**	3.70 ± 1.25	0.74 ± 0.55	**82.7**
PO_4-P	1.51 ± 0.60	0.57 ± 0.44	**62.3**	1.09 ± 0.65	0.55 ± 0.52	**63.6**

*the removal efficiencies in Train B were calculated based on the screened influent concentrations as the results take into account the overall removal efficiencies (RBF + biological process).

Nitrogen removal

Conventional biological nitrogen removal is accomplished via autotrophic nitrification and heterotrophic denitrification. During aerobic nitrification, ammonium is first oxidized to nitrite by ammonia-oxidizing bacteria (AOB), and then oxidized into nitrate by nitrite-oxidizing bacteria (NOB). In anoxic denitrification, nitrates are reduced into nitrogen gas by denitrifying bacteria. During this study, all rates were adjusted to a temperature of 20 °C using a temperature coefficient of $\theta = 1.103$ for nitrification (Urbini *et al*. 2015) and $\theta = 1.07$ for denitrification (Rusten *et al*. 1995).

Full nitrification was observed in both MBR Trains. The nitrification rates were in the range of 0.65 – 1.75 (average~1.12) mg NH_4-N/g MLVSS-h in Train A and between 0.67 – 1.82 (average~1.21) mg NH_4-N/g MLVSS-h in Train B. According to this result, the nitrification rates were slightly higher in the MBR Train B receiving filtered wastewater. The difference might be due to the lower MLVSS concentration in Train B which was 4.88 g MLVSS/L compared to 5.48 g MLVSS/L in Train A. The removal of particulate COD during the primary treatment reduces the organic loading, thus reducing the concentration of the MLVSS in Train B. Furthermore, the higher SRT (16.8 d) in Train B might contribute to the higher rates as well. In biological process, a longer SRT can result in a higher nitrifying bacteria concentration, thus improving the nitrification. Long SRT applied in the MBR prevent nitrifying bacteria from being washed out of the system and nitrifiers are less endangered by fast-growing heterotrophs, which are better competitors for the ammonia nitrogen (Rittman & McCarty 2001).

The specific denitrification rates varied from 0.63 - 1.82 (average~1.27) mg NO_x-N/g MLVSS-h in Train A, whereas the rates were between 0.75 – 2.41 (average~1.50) mg NO_x-N/g MLVSS-h in Train B. The values were within the range of the SDNRs with internal carbon source found in literature (at 20°C, $\theta = 1.07$), which were between 1.26 and 6.5 NO_x-N/g MLVSS-h (Kim 2004; Zhao & Ma 2002). The denitrification rate was slightly higher in the MBR treating filtered wastewater. The difference might be explained by the lower MLVSS concentration in the anoxic reactors in Train B (3.1 g MLVSS/L) compared to Train A (4.1 g MLVSS/L). Moreover, the highest denitrification rate was observed in Train B with a C/N ratio of 9.6 g TCOD/g TN (data not shown). This result confirmed the finding during the lab-scale SBRs, where the optimum C/N ratio upfront a biological process was around 9 g TCOD/g TN (Razafimanantsoa *et al.* 2014b). The C/N ratio required for complete nitrate reduction to nitrogen gas by denitrifying bacteria depends on the nature of the carbon source. Carbon limitation may result in incomplete denitrification and a concomitant accumulation of intermediate products, such as NO_2 and N_2O. Conversely, an excess of carbon constitutes an extra cost and will promote dissimilatory nitrate reduction to ammonia and the presence of carbon in the denitrified effluent (Carrera et al 2004). Therefore, the C/N ratio should be properly controlled to achieve good nitrogen removal efficiency as it directly effects on functional microorganism populations.

Membrane performance

The performance of a membrane can be evaluated by monitoring the development of the TMP over time for a constant flux, which is in correlation with the fouling rate (Leiknes *et al.* 2006). The membrane ZeeWeed-10 was operated at constant flux. The permeate flux ranged from 5.8 to 7.0 LMH (~ 6.4 LMH) in Train A and between 5.7 and 6.6 LMH (~ 6.3 LMH) in Train B. Figure 2 shows the plot of the TMP against the influent TSS of both feeds. The average TMPs registered with the two membranes were 46 ± 9 mbar in Train A and 26 ± 7 mbar in Train B. It can be concluded that the fouling rate is higher during the treatment of coarse screened influent wastewater compared to the filtrate from 33 µm RBF. The latter removed about 40% of the influent TSS. Thus, reducing the organic load to the biological process will reduce the amount of new biomass produced, and consequently the amount of biological sludge produced. However, no chemical cleaning of the membrane was performed throughout the experiment as the TMPs were still well below 300 mbar, the maximum limit recommended by the membrane supplier.

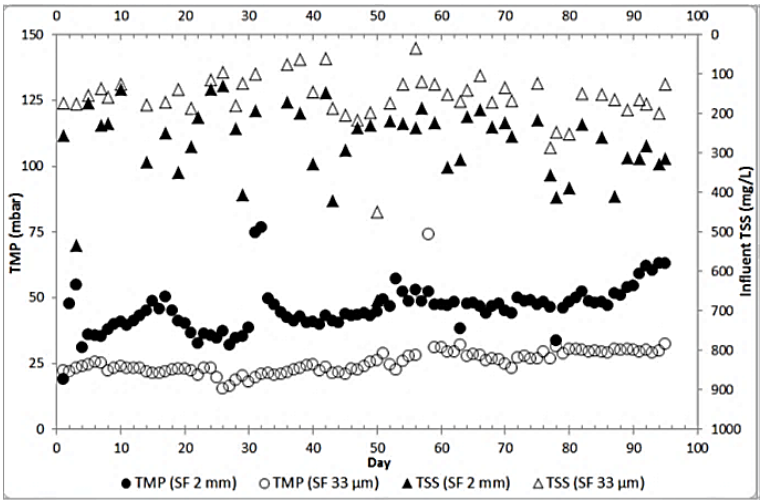

Figure 2. Impact of suspended solids on TMP

Sludge production

In Train A, the overall sludge produced from the system was only composed of biosludge. It was about 21.3 g TSS/d. On the other hand, in Train B, the total sludge production is the combination of the sieve sludge and the biosludge. The biosludge production in Train B was only 16.4 g TSS/d. But the total amount of sludge produced in Train B was about 46 % higher compared to that of Train A because of the sludge removed with the RBF. The sludge yields were 0.36 g TSS/g COD and 0.43 g TSS/g COD in Train A and Train B, respectively. Both primary sludge and biological sludge could be used to produce methane (Appels *et al.* 2008). However, several studies showed that primary sludge contains higher biogas production potential (BMP) because its energy content has not yet been consumed. The biogas production could be increased up to three times depending on the primary separation methods (Ucisik & Henze 2008). Paulsrud *et al.* (2014) investigated the BMP of sludge from Salsnes Filter (sieve sludge) and sludge from conventional primary clarifier. The results showed that

sieve sludge had higher volatile solids content and higher methane potential than primary sludge (345 NML CH_4/g VS versus 287 NML CH_4/g VS). It has been observed in combined conventional activated sludge and Anaerobic digestion (AD) processes that the nitrogen load increases when recycling the AD effluent back to the system. In such case, one may consider to remove less organic matter with the RBF during the primary treatment to have sufficient carbon source for the nitrogen removal. Tests done with laboratory scale SBRs showed that similar nitrogen removal was observed in the reactors treating filtrates from sieve cloth openings of 33 µm and above (Razafimanantsoa *et al.,* 2014b). Consequently, the right sieve cloth could be chosen depending on the AD effluent characteristics and the overall nitrogen load to achieve the optimal C/N ratio for nitrogen removal.

Oxygen demand

Energy demand, related to sludge transfer, permeate production and most significantly aeration, is a key cost factor when considering MBR technology (Henkel *et al.* 2011). The oxygen required for biological treatment depends on the influent biodegradable organic matter and biologically oxidizable nitrogen. During this pilot study, the oxygen demand in each MBR system was calculated according to the German design guidelines (ATV 2000). The average values of SRTs were 13.5 days for Train A (2 mm) and 16.8 days for Train B (RBF 33 µm). Using a ratio of $COD/BOD_5 = 2$ (ATV 2000) and a temperature of 19 °C, the specific oxygen demand for the removal of organic matter was 0.610 g O_2/g COD for Train A and 0.625 g O_2/g COD for Train B. The standard values of 4.3 g O_2/g N nitrified and 2.9 g O_2/g N removed were used to determine the oxygen demand for nitrification and the oxygen credit for denitrification, respectively. Overall, the aerobic reactor in Train A required about 0.41 kg O_2 per m^3 treated wastewater although that of Train B required only 0.29 kg O_2/m^3. More oxygen was required in the MBR fed with coarse screened wastewater when compared to the MBR treating filtrate from 33 µm RBF. The MBR in Train A, operated without fine mesh sieve as primary treatment required 40 % more air than Train B (with 33 µm RBF); and it was mainly due to the partial removal of the influent organic matter with RBF. Ruiken et al. (2013) evaluated the net energy demand of wastewater treatment (including sludge treatment and incineration) and found that the system using RBF as a primary treatment required 40 % less energy compared to the system without primary treatment. Therefore, efficient particle removal could lead to a reduction of the overall energy consumption.

CONCLUSIONS

During this experiment, an RBF with 33 µm filter cloth was used as a primary treatment to biological wastewater treatment and its impact on the downstream MBR system was evaluated. The 33 µm filter allowed the removal of 42 % of the TSS, which corresponded to 33 % of the organic matter (expressed as COD), 12 % of the total nitrogen (TN) and 14 % of the total phosphorus (TP).

The assessment of the performance of the MBRs revealed that the MBR fed with filtered wastewater had similar removal efficiencies as the MBR operated without primary treatment.

The two trains removed 94 % of the COD, 80 % of the TP and 74 % of the TN. The nitrification rates were slightly higher in the MBR receiving filtered wastewater (Train B), with an average value of 1.21 mg NH_4-N/g MLVSS-h compared to 1.12 mg NH_4-N/g MLVSS-h in MBR control (Train A). The difference might be due to the lower MLVSS concentration in Train B. Furthermore, the higher SRT in Train B might contribute to the higher rates as well. The specific denitrification rates varied from 0.63 - 1.82 mg NO_x-N/g MLVSS-h in Train A, whereas the rates were between 0.75 – 2.41 mg NO_x-N/g MLVSS-h in Train B. The average denitrification rate was slightly higher in the MBR treating filtered wastewater. The difference might be explained by the lower MLVSS concentration in the anoxic reactors in Train B (3.1 g MLVSS/L) compared to Train A (4.1 g MLVSS/L).

The result also showed that the fouling rate of the membrane, evaluated through the change of TMPs, was reduced to nearly half in the MBR fed with RBF 33 µm (Train B) compared to the system without fine mesh filter (Train A). The average TMPs registered with the two membranes were 46 ± 9 mbar in Train A and 26 ± 7 mbar in Train B. No cleaning of the membranes was required during the test.

The total amount of sludge produced in Train B was about 46 % higher compared to that of Train A because of the sludge removed with the RBF Filter. The biological sludge yields were 0.36 g TSS/g COD and 0.43 g TSS/g COD in Train A and Train B, respectively. Both primary sludge and biological sludge could be used to produce methane.

The aerobic reactor in Train A required about 0.41 kg O_2 per m^3 treated wastewater although that of Train B required only 0.29 kg O_2 per m^3. More oxygen was required in the MBR fed with coarse screened wastewater when compared to the MBR treating filtrate from 33 µm RBF. The reduction was about 30 % and it was mainly due to the partial removal of the influent organic matter with RBF. Overall, the use of RBF 33 µm as a primary treatment was beneficial for the downstream biological process.

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The following papers also form part of the thesis, but cannot be included in the repository for copyright reasons.

Paper 3:
Rusten, B., Razafimanantsoa, V. A., Andriamiarinjaka, M. A., Otis, C. L. & Sahu, A. K. (2016). Impact of fine mesh sieve primary treatment on nitrogen removal in moving bed biofilm reactors. *Water Science and Technology*, 73 (2), 337-344.

Paper 4:
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