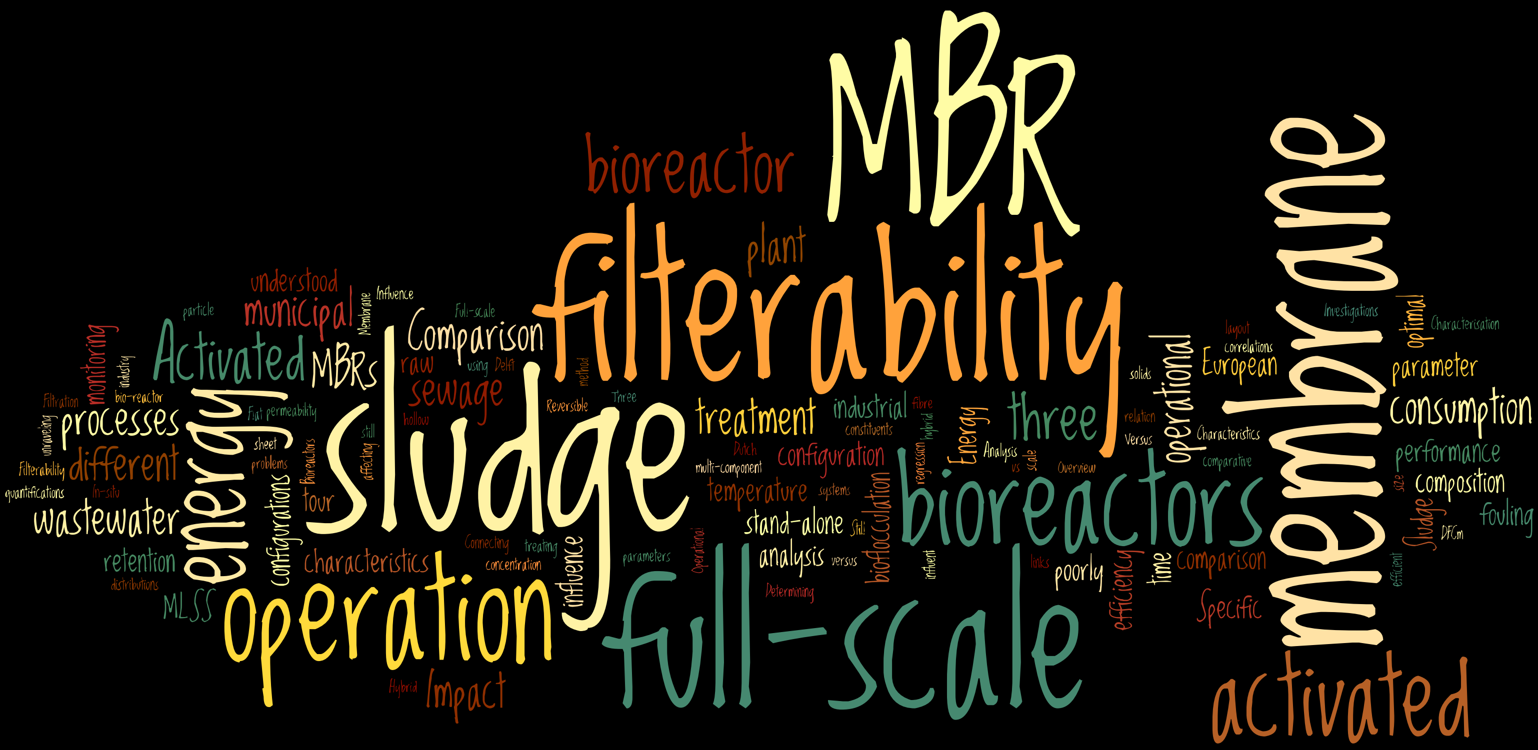
Activated sludge filterability and full-scale membrane bioreactor operation



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Proefschrift

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Summary

Membrane bioreactors (MBR) have become a mature technology and an alternative for conventional wastewater treatment processes. Among other prospects, MBRs provide high effluent quality, free of suspended solids and very low levels of bacteriological contamination, at a relatively small plant footprint. Despite continuous developments, membrane fouling mitigation and related high operational and maintenance (O&M) costs remain a major challenge and restrain wide MBR application. Furthermore, the efficiency of the filtration process in an MBR is governed by the activated sludge filterability, which is still limitedly understood and is determined by the interactions between the biomass, the wastewater and the applied process conditions.

The purpose of this thesis is to increase understanding of the factors impacting activated sludge filterability during full-scale MBR operation. Therefore, the research links activated sludge filterability assessment and full-scale MBR functioning, i.e., design options, operation, performance and energy efficiency. The overall research goal was to determine conditions for enhanced and efficient operation of the MBR technology.

The research work included both extended on-site measurements and operational data analysis. The Delft Filtration Characterization method (DFCm) was applied to experimentally determine the activated sludge filterability in full- and pilot-scale MBRs treating both municipal and industrial wastewater. During the studies activated sludge samples were

collected from 14 different MBRs and subjected to filtration tests and a set of physicochemical analyses. Subsequently, the most influential parameters influencing activated sludge filterability were determined. In addition, the design, operational and performance data were collected from the selected full-scale MBR plants and analysed in respect to plant functioning, i.e., operation, energy efficiency and operational costs.

Our results show that the temperature and wastewater composition are important influencing parameters with respect to filterability. Deterioration of filterability under low temperatures was linked to a slower biodegradation of the wastewater in the mixed liquor compared to high temperatures.

The results revealed that sludge parameters usually denoted in literature as membrane fouling indicators, e.g., BPC, SMP and TOC, are not clearly correlated with sludge filterability. Moreover, every parameter alone is a weak indicator of biomass fouling propensity. Combination of activated sludge parameters, i.e., the sludge morphology and relative hydrophobicity, better indicated sludge filterability than the parameters alone.

Results clearly show that applying a separate membrane tank and low return flows from the membrane tank to the bioreactor the MLSS concentration should be above a critical value of about 10 g/L to promote filterability improvement in the membrane compartment.

Furthermore, an undesired and refractory composition of incoming wastewater, hydraulic and/or organic load shocks, as well as abrupt temperature changes of the influent lead to operational problems and affect sludge filterability. Nevertheless, MBR is a robust and reliable technology as permeate quality mostly complies with the regulations and is independent of the activated sludge quality and encountered operational problems.

It was found that, both the MBR plant layout and membrane configurations do have some influence on overall plant functioning. Different membrane configurations require different mechanical pre-treatments and are associated with different filtration protocols, applied fluxes and membrane cleaning methods. In other words, membrane configuration selection influences mainly the plant operational strategies. The MBR plant layout has more distinct influence on overall plant functioning due to indirect impact on operational flexibility and reliability, performance and O&M costs. Moreover, the activated sludge filterability was found independent of membrane configuration but not of the MBR plant layout.

The investigated MBRs are operated below the design loading rates and consequently are operated under sub-optimal flow conditions which in turn results in reduced energy efficiency of the plant. Other factors like the system design and layout, the membrane hydraulic utilization and the strategy applied for the membrane air-scouring are influencing the energy consumption and energy efficiency of an MBR system. Aeration is still the major energy consumer, often exceeding 50% share of total energy consumption, therefore the coarse bubble aeration applied for continuous membrane cleaning remains the main target for energy saving actions, especially for installations with flat sheet membranes.

Overall, it can be concluded that good filterability of the activated sludge is indispensable for efficient and optimal operation of an MBR. Operation with poor sludge filterability will be

associated with a cost penalty due to sub-optimal filtration conditions. Wastewater composition and temperature were identified as main parameters influencing activated sludge filterability. MBR plant layout and membrane configuration influence overall MBR functioning and should be chosen carefully. The energy efficiency of an MBR is driven by the hydraulic utilization of the membranes and can be improved by implementation of flow equalization, new aeration strategies and adjusting operational settings to the incoming flow.

Samenvatting

Het membraanbioreactor (MBR) proces vormt een inmiddels bewezen technologie die een alternatief biedt voor conventionele afvalwaterzuiveringtechnieken. Zuivering met behulp van MBR-technologie vindt plaats in een relatief compacte installatie en levert een hoge effluentkwaliteit met lage concentraties aan zwevende stof en bacteriologische verontreinigingen. Ondanks voortdurende ontwikkelingen blijft membraanvervuiling en de hieraan gerelateerde hoge kosten voor bedrijfsvoering en onderhoud van het proces een struikelblok dat een brede toepassing van MBR in de weg staat. De efficiëntie van het filtratieproces is onder andere afhankelijk van de filtreerbaarheid van het actiefslib, welke wordt bepaald door de interacties tussen de biomassa, afvalwater en de procescondities.

Het doel van dit onderzoek is om het inzicht te verbeteren in de factoren die de actiefslib filtreerbaarheid beïnvloeden bij MBR-toepassing op praktijkschaal. Elementen hiervan zijn bijvoorbeeld: ontwerpopties, bedrijfsvoering, prestatie- en energie-efficiency. Het algehele doel is een verbeterde en efficiëntere werking van de MBR-techniek.

Het onderzoek omvat zowel uitvoerige metingen bij praktijkinstallaties als analyse van bedrijfsgegevens. De ‘Delft Filtration Characterization method’ (DFCm) is gebruikt om de actiefslib-filtreerbaarheid experimenteel vast te stellen. Actiefslib-monsters zijn verzameld van 14 verschillende proef- en praktijkschaal MBR-installaties die zowel huishoudelijk als industrieel afvalwater zuiveren.

Deze monsters zijn onderworpen aan filtratietests en een set fysisch-chemische analyses. Daaropvolgend zijn de meest invloedrijke parameters op de actiefslib-filtreerbaarheid bepaald. Daarnaast zijn van de geselecteerde MBR's data verzameld (ontwerp, bedrijfsvoering en prestaties) en geanalyseerd in relatie tot het functioneren van de installatie (werking, energie-efficiëntie en operationele kosten).

De resultaten tonen aan dat de temperatuur en de afvalwatersamenstelling belangrijke parameters zijn die de filtreerbaarheid van het actiefslib beïnvloeden. Verlaagde filtreerbaarheid bij lagere temperaturen wordt gedeeltelijk veroorzaakt door een lagere biodegradatie van de bestanddelen in het afvalwater vergeleken met hogere temperaturen.

Slibparameters die in de literatuur vaak genoemd worden als indicatoren voor membraanvervuiling, bijvoorbeeld BPC, SMP en TOC, kunnen niet duidelijk worden gecorreleerd aan de filtreerbaarheid. Iedere parameter afzonderlijk is zelfs een zwakke indicator voor membraanvervuiling. Het combineren van meerdere actiefslib parameters, zoals slibmorfologie en relatieve hydrofobiciteit, resulteert in een betere indicatie van slib-filtreerbaarheid dan de individuele parameters. Het onderzoek toont aan dat, bij omstandigheden van lage retourslibstroom in combinatie met een separate membraantank, de MLSS concentratie boven een kritieke MLSS concentratie moet blijven van ongeveer 10 g/L om de filtreerbaarheid in de membraantank te verbeteren.

Een ongewenste of problematische samenstelling van het inkomende afvalwater, schokken in hydraulische en/of organische belasting, evenals abrupte temperatuurfluctuaties van het influent beïnvloeden de slib filtreerbaarheid in negatieve zin en kunnen leiden tot operationele problemen. Desalniettemin is MBR robuust en betrouwbaar, omdat de permeaatkwaliteit bijna altijd voldoet aan de regelgeving en onafhankelijk is van de actiefslib-kwaliteit en operationele knelpunten.

Zowel indeling van een MBR-installatie als de membraanconfiguratie hebben invloed op het algehele functioneren van de installatie. Verschillende membraanconfiguraties vergen een andere mechanische voorbehandeling en worden geassocieerd met uiteenlopende filtratieprotocollen, toegepaste fluxen en reinigingsmethoden voor de membranen. Met andere woorden, de keuze voor de membraanconfiguratie beïnvloedt de algehele operationele bedrijfsvoering van de installatie. De indeling van een MBR-installatie heeft een duidelijk effect op het algeheel functioneren van de installatie, vanwege het indirecte effect op de operationele flexibiliteit en betrouwbaarheid, de prestaties en de kosten voor beheer en onderhoud. De actiefslib-filtreerbaarheid blijkt onafhankelijk te zijn van de membraanconfiguratie, maar niet van de indeling van de MBR-installatie.

De in dit onderzoek beschouwde MBR-installaties zijn onderbelast en functioneren daardoor onder suboptimale stromingscondities, wat leidt tot verminderde energie-efficiëntie. Andere factoren die de energieconsumptie en -efficiëntie beïnvloeden zijn: systeemontwerp en -

indeling, hydraulische condities en de strategie voor de luchtreiniging van de membranen. Beluchting blijft een belangrijke energiepost, welke vaak een aandeel heeft van meer dan 50% van het totale energieverbruik. Om deze reden blijft optimalisatie van de bellenbeluchting, toegepast voor continue membraanreiniging, het belangrijkste aandachtspunt voor energiebesparingsacties, vooral voor installaties met plaatmembranen.

De conclusie van dit onderzoek is dat goede filtreerbaarheid van het actiefslib noodzakelijk is voor efficiënte en optimale werking van een MBR. Procesvoering met matige slib-filtreerbaarheid leidt tot suboptimale filtratiecondities.
Afvalwatersamenstelling en –temperatuur zijn de meest invloedrijke parameters op de actiefslib-filtreerbaarheid. Het ontwerp van de MBR-installatie en membraanconfiguratie beïnvloeden de algehele werking van de installatie en zouden daarom zorgvuldig aandachtig gekozen moeten worden. De energie-efficiëntie van een MBR wordt bepaald door de bedrijfsvoering en hydraulische condities rond het membraan en zou verbeterd kunnen worden door een meer gelijkmatige verdeling van de stroming, nieuwe beluchtingsstrategieën en het aanpassen van de operationele instellingen van de inkomende stroming.

List of abbreviations and symbols

#1pixel	average number of sludge particles equal to 1pixel [1/image]
%1pixel	surface fraction of sludge particles equal to 1pixel [%]
α_R	specific cake resistance at reference filtration resistance [m/kg]
$\alpha_R \cdot c_i$	product of the specific cake layer resistance
ΔR_{20}	added resistance after filtration of 20 L/m² in the DFCm [m^{-1}]
μ_p	apparent viscosity of the permeate [Pa·s]
A	membrane area [m^2]
A_i	area of activated sludge particle i [μm^2]
A_{mean}	activated sludge mean particle size [μm^2/particle]
ACTIAS	activated sludge image analysis system
AE	aerobic
AN	anaerobic
AX	anoxic
BFM	Berlin filtration method
BOD	biological oxygen demand [mgO_2/L]
BPC	biopolymer cluster [mg/L]
BSA	Bovine Serum Albumin

c_i	solids concentration involved in the fouling process [kg/m^3]
CAPEX	capital expenses
CAS	conventional activated sludge
CFV	cross-flow velocity near the membrane surface [m/s]
COD	chemical oxygen demand [mgO_2/L]
CST	capillary suction time
DC	activated sludge dissociation constant [Abs_{650nm}/washing step]
DFC_i	Delft Filtration Characterization installation
DFC_m	Delft Filtration Characterization method
DO	dissolved oxygen [mgO_2/L]
DSVI	diluted sludge volume index [mL/gTSS]
DWF	dry weather flow
eEPS	extractable extracellular polymeric substances [mg/gMLVSS]
eEPS PN	protein fraction of eEPS [mg/gMLVSS]
eEPS PS	polysaccharide fraction of eEPS [mg/gMLVSS]
EPS	extracellular polymeric substance [mg/L]
F/M	food to microorganism ratio [kgBOD/kgMLSS.day]
FC	facultative tank
FID	flame ionisation detector
FS	flat sheet
GC	gas chromatography
HF	hollow fibre
HRT	hydraulic retention time [hours]
iMBR	immersed or submerged MBR
J	permeate flux [$L/m^2 \cdot h$]
K	permeability [$L/m^2 \cdot h \cdot bar$]
MATH	microbial adhesion to hydrocarbons
MBR	membrane bioreactor
MCP	mechanical cleaning process
MF	microfiltration
MLSS	mixed liquor suspended solids [g/L]
MLVSS	mixed liquor volatile suspended solids [g/L]
MT	multi tube
N.A.	not applicable
N.P.	not provided
NaCl	sodium chloride
NaOCl	sodium hypochlorite

NF	nanofiltration
NH_4-N	ammonia
NH_4/NO_3	ammonia/nitrate ratio
NO_3-N	nitrate
O&M	operational and maintenance
OM	organic matter
OPEX	operational expenses
OUR	oxygen uptake rate [mgO_2/L·h]
PAO	phosphate accumulating organisms
PB	polybrene
PBS	phosphate buffered saline solution
PD	process disturbance
PE	person equivalent
PE_{design}	plant design capacity equal to a pollution load of 54 g BOD/day
$PE_{removed}$	removed pollution load
PLS	partial least squares
PN	proteins
PO_4-P	orthophosphorus
PS	polysaccharides
PSU	process start-up
PVDF	polyvinylide difluoride
PVSK	polyvinyl sulphate potassium salt
Q	effluent flow rate [m^3/h]
R_a	adsorption resistance [1/m]
R_{add}	added filtration resistance [1/m]
R_{cl}	cake layer resistance [1/m]
R_f	total fouling resistance [1/m]
R_m	membrane resistance [1/m]
R_{pb}	pore blocking resistance [1/m]
R_{total}	total resistance to filtration composed of the membrane resistance and the additional resistance resulting from fouling mechanism [1/m]
r_p	Pearson's correlation coefficient
RH	sludge relative hydrophobicity [%]
RO	reverse osmosis
RWF	rain weather flow
s	compressibility coefficient [-]
SAD_m	specific aeration demand per membrane area [Nm^3/m^2·h]
SAD_p	specific aeration demand per permeate volume [Nm^3/m^3]
SC	surface charge [meq/gMLSS]
SF	sand filter
SFI	sludge filtration index

sMBR	sidestream MBR
SMP	soluble microbial products [mg/L]
SMP PN	protein fraction of SMP [mg/gMLVSS]
SMP PS	polysaccharide fraction of SMP [mg/gMLVSS]
sOUR	specific oxygen uptake rate [mgO_2/gMLVSS.h]
SRT	solids retention time [days]
STOWA	Foundation for Applied Water Research
SVI	sludge volume index [mL/gTSS]
T	temperature [°C]
TB	toluidine blue
TDS	total dissolved salts [g/L]
TKN	total kjeldahl nitrogen [mg/L]
TMP	transmembrane pressure [bar]
TN	total nitrogen [mgN/L]
TOC	total organic carbon [mgC/L]
TOC_{perm}	total organic carbon of permeate [mgC/L]
TOC_{sup}	total organic carbon in the supernatant [mgC/L]
TOD	total oxygen demand [mgO_2/L]
TP	total phosphorus [mgP/L]
TSS	total suspended solids [g/L]
TTF	time to filter
UF	ultrafiltration
UPAC	Union of Pure and Applied Chemistry
VFAs	volatile fatty acids [mg/L]
VFM	VITO fouling measurement
VSS	volatile suspended solids [g/L]
VT	pre-aeration tank
WRD	Regge and Dinkel Water Board
WRIJ	Rijn and IJssel Water Board
WSHD	Water Board Hollandse Delta
WWTP	wastewater treatment plant
XRD	X-Ray powder Diffraction
XRF	semi-quantitative X-ray Fluorescence

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CHAPTER 1

Introduction

1 Introduction

1.1 Background

Fresh water is indispensable for all life on earth. However, it was only in 2010, when the United Nations General Assembly (Resolution A/RES/64/292, July 2010) and the Human Rights Council (Resolution A/HRC/15/L.14, September 2010) recognised the right to access drinking water and sanitation as a human right. Despite this international recognition, there is still much to be done. Although, according to latest UN reports (United Nations 2010), the world is on track to meet one of Millennium Development Goals (MDG) to halve the population without drinking water access, the 2015 sanitation target to halve the population of developing regions without sanitation appears to be out of reach (Figure 1.1).

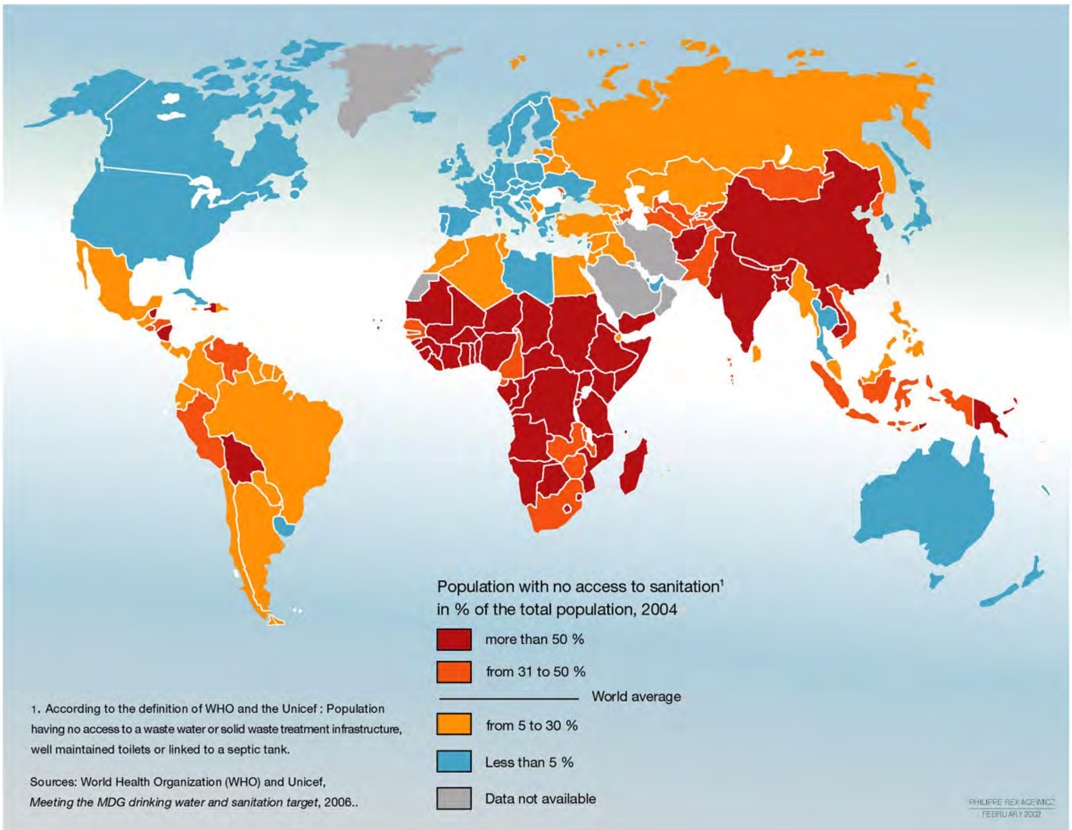

Figure 1.1: Populations access to sanitation (Diop et al. 2008)

The growth of societies and rapid urbanization results in increasing water demands due to human consumption, industrial activities and agriculture expansion. Subsequently, growth in water consumption results in a higher production of used and polluted water, i.e., wastewater. The wastewater contains a wide range of pollutants which may be harmful to the environment. Hence, when discharged without some kind of treatment it deteriorates the quality of the available water sources. In turn, when people use polluted sources as a drinking water it pose a serious risk and has negative consequences to human health. Therefore, collection and treatment of wastewater is of major importance for the public health and environment protection.

Apart from the issue of water quality, also the quantity of freshwater resources has become locally an issue and a drive towards effective wastewater treatment. Despite the fact that $^2/_3$ of the earth surface consists of water, only about 0.3 % is actually available for human use as rivers and lakes (Shiklomanov 1999). So, in many parts of the world, the water demand already exceeds water supply causing water stress or even water crisis (UNESCO 2006). At various places, the already limited amount of available clean and consumable fresh water is becoming a scarce resource (Figure 1.2). The water stress is aggravated by the occurring climate changes, subsequently leading to global water scarcity. By 2025, two-thirds of the world's population could be living under water stressed conditions and 1.8 billion under water scarcity (UN-Water 2007). Fortunately, water is a renewable resource and reclaiming wastewaters for subsequent usage is possible. Therefore, to deal with water stress and to tackle global issues of water shortage, efficient and cost effective purification and reuse methods are required.

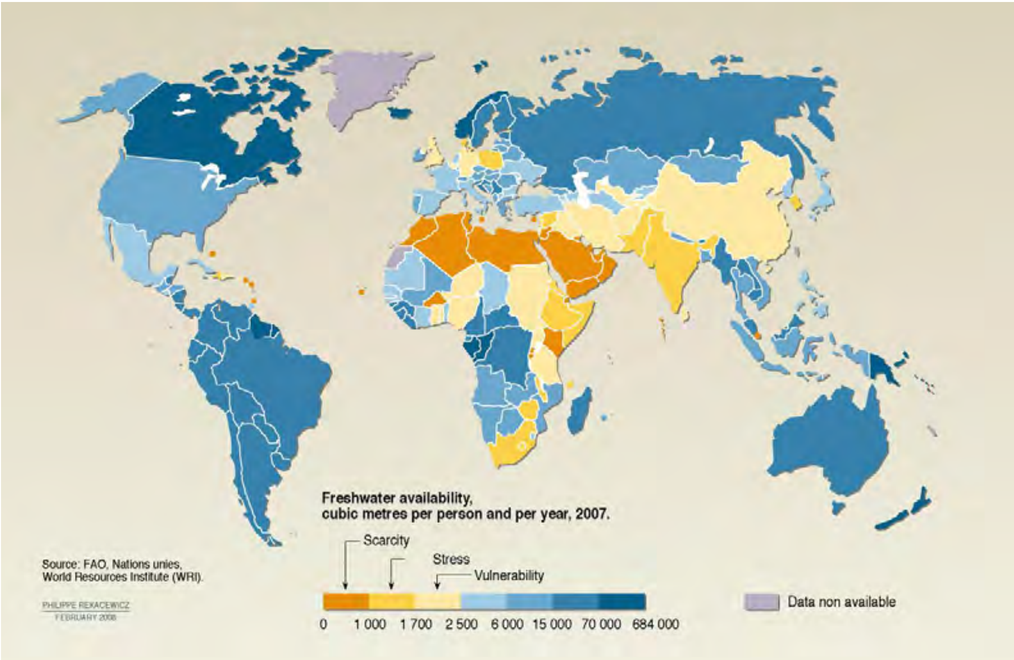

Figure 1.2: Global water stress and scarcity (Diop et al. 2008)

Membrane based treatment technologies represent an attractive tool in wastewater management and are widely used for various treatment and reuse applications. Over the last years membranes are receiving increasing interest for treating water and wastewater possibly becoming the preferred treatment technologies for both municipal and industrial water treatment sectors (Cummings and Frenkel 2008). An example of such technology is the membrane bioreactor (MBR) process. The MBR technology has attracted considerable attention as wastewater treatment process offering significant advantages in terms of effluent quality and required footprint (Lesjean and Huisjes 2008).

1.2 Problem statement

Despite continuous developments in the field of MBR technology, membrane fouling together with the associated energy demand and related costs issues remain major technological and research challenges. Due to the interactions between the membrane and constituents present in activated sludge and wastewater, the phenomenon of membrane fouling occurs during the filtration process. In result, the performance of membrane filtration decreases over time. Implemented strategies for prevention and removal of membrane fouling result in high operational and maintenance costs of the treatment system. In particular, the high energy requirements arisen from frequent membrane cleaning remains a challenge in terms of energy consumption and overall cost efficiency of full-scale MBRs.

1.3 Research objectives

The overall hypothesis of our research is that a better understanding of the factors impacting activated sludge filterability in relation to full-scale MBR functioning will lead to enhanced operation and implementation of MBR technology. The hypothesis is further visualized in Figure 1.3.

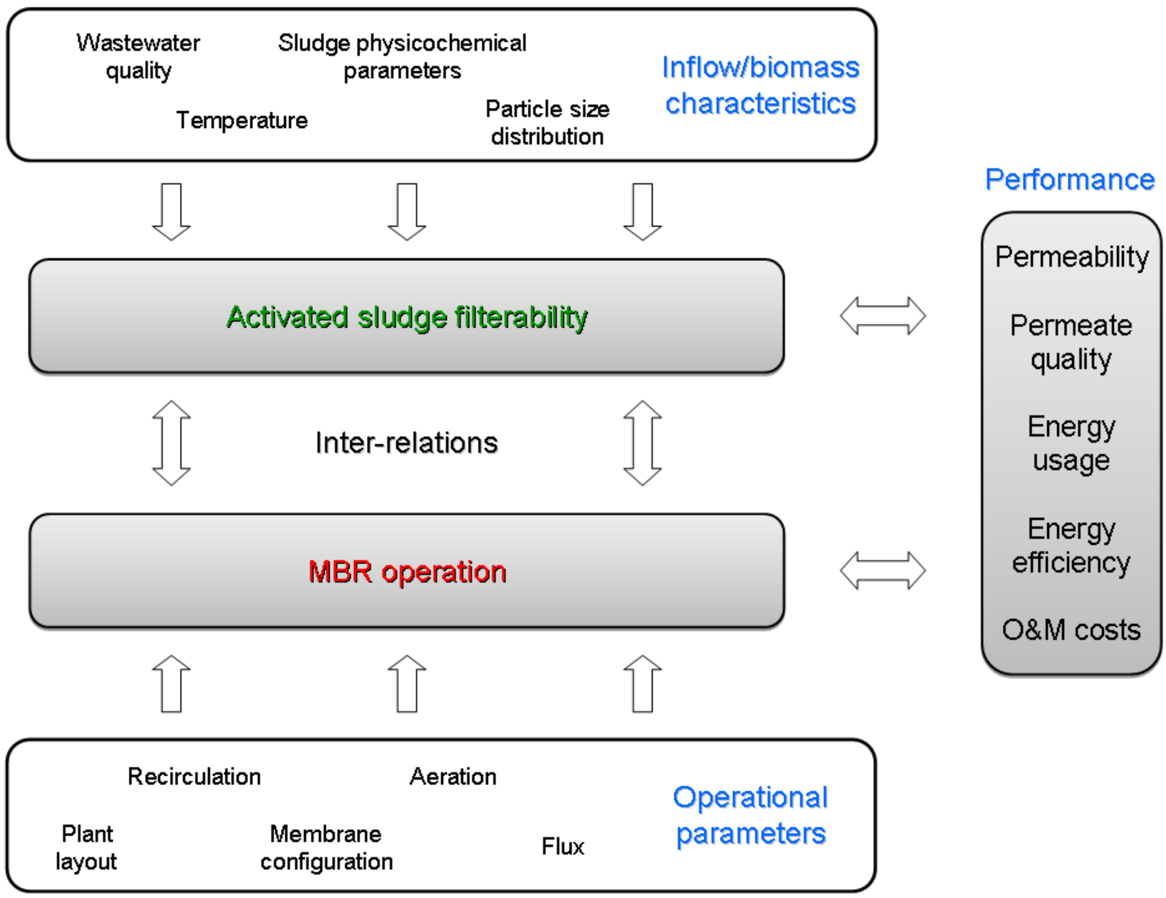

Figure 1.3: Visualization of the research hypothesis

In order to address this hypothesis the following research questions are formulated: what are the most important parameters influencing activated sludge filterability and how poor filterable sludge affects MBR operation in different full-scale MBR installations.

The research objectives of this thesis are twofold: firstly, to provide better understanding of membrane fouling propensity based on activated sludge filterability assessment. Secondly, to provide important insights on full-scale MBR overall functioning, i.e., design options, operation, performance and energy efficiency, in order to provide a step forward towards optimum performance conditions and efficient operation of the MBR technology.

The specific objectives of this thesis are following:

- to monitor the sludge filterability of full-scale municipal MBRs in order to quantify the impact of activated sludge filterability on plant operations and performances (Chapter 4 & 6)
- to inspect activated sludge filterability in different compartments of the full-scale MBR in order to assess variations along the process flow (Chapter 4);
- to assess the relations between temperature, raw municipal wastewater composition and filterability, and activated sludge filterability (Chapter 5).
- to unravel the impact of the activated sludge mixed liquor characteristics on activated sludge filterability (Chapter 6).
- to assess the impact of the influent characteristics on activated sludge filterability and operation of full-scale MBRs (Chapter 6).
- to assess the impact of using different membrane configurations on activated sludge filterability, operation and efficiencies of the full-scale MBR plants (Chapter 7).
- to study design plant layouts, and their impact on operation, performance, energy consumption and economy of the MBR plant (Chapter 7)
- to evaluate a stand-alone MBR in comparison to a hybrid concept of MBR design, and asses the impact on operation, performance, energy consumption and economy of the MBR plant (Chapter 7)
- to investigate the specific energy requirements of full-scale MBRs and elucidate where possible future energy consumption reduction can be achieved (Chapter 8).

1.4 Research approach

To facilitate further development and optimisation of membrane bioreactor technology, an extensive research programme was established and implemented at Delft University of Technology (TU Delft). The Sanitary Engineering section of TU Delft participated in the MBR research in the framework of two European research programmes: MBR-TRAIN and EUROMBRA, as well as two Dutch projects: MBR2 and MBR2+. Among other specific targets in the projects, TUD focused on the filterability – one of the parameters to characterise activated sludge (Metcalf&Eddy 2003) – and filterability influencing parameters aiming at optimisation of full-scale membrane bioreactors operation.

The experimental work in this thesis was performed in the framework of two projects. The MBR-TRAIN project aimed at process optimisation and fouling control in membrane bioreactors for wastewater and drinking water treatment. MBR-TRAIN (contract no. MEST-CT-2005-021050) was a Marie Curie Host Fellowship for Early Stage Research Training

supported by the European Commission under the 6th Framework Programme (Structuring the European Research Area - Marie Curie Actions). The consortium of MBR-TRAIN comprises 10 partners from the water-industry, research institutes and universities across Europe representing a cross-section of relevant disciplines, sectors and regions.
The research project 'MBR2+: towards an energy efficient MBR', was dedicated to identify and further develop the energy saving in design, operation and management of MBR plants. The project aim was to expand application opportunities of the MBR technology by increasing the efficiency of MBR systems through the design, implementation and management. The project consortium consists of Evides Industriewater, Witteveen+Bos, Water Board Hollandse Delta and Delft University of Technology, representing a balanced partnership of industrial, consultancy and engineering, government and research organizations.

The detailed approach exploited in each of the discussed sub-research topics is described below.

Assessment of activated sludge filterability in full-scale MBRs

For a period of two years an extensive measurement campaign was performed, during both summer and winter period. Representative measurements for the summer period were performed in months of June–August and for the winter period in the months of January–March. During those measurement periods, the filtration characterisation installation (DFCi), see section 3.2, was placed at the wastewater treatment plant (WWTP) for a period of 4 – 5 days. The investigated plants, namely MBR Heenvliet, MBR Varsseveld and MBR Ootmarsum, are described in detail in Table 3.1. Activated sludge filterability tests and physicochemical analyses were carried out. The filterability of the activated sludge was monitored in different compartments of the MBR, i.e., membrane tank, aerobic, anaerobic and anoxic tank. The results, described in *Chapter 4*, became a reference point for the further studies on MBR operation and performance.

Impact of temperature on activated sludge filterability

In this research, raw wastewater and activated sludge samples were taken from a full-scale membrane bioreactor treating municipal wastewater. The MBR is located at the Heenvliet WWTP. A detailed description of the Heenvliet plant is presented in Table 3.1. Sampling campaigns were conducted during different seasons of the year (November 2010–August 2011) to assess the influence of temperature and its seasonal fluctuations. During this period, both influent and activated sludge were analysed in terms of filterability, respirometry, particle size distribution and a set of physicochemical properties. This research work and its outcomes are discussed in *Chapter 5.*

Activated sludge characteristics affecting sludge filterability

Ten different MBRs in Belgium and the Netherlands, treating either municipal or industrial wastewaters, were sampled in both winter and summer (Table 3.2). The Delft Filtration Characterization method (DFCm), described in detailed in Chapter 3, was chosen to determine the activated sludge filterability, next to calculating the process permeability, i.e., flux and TMP, in each installation. Each sample was subjected to a set of activated sludge analyses, such as relative hydrophobicity, image analysis and EPS. The selection of parameters was based on state-of-the-art literature on membrane fouling in MBR. The obtained relationships may help in selecting the main parameters influencing fouling and can assist in implementing the correct remedial actions to improve process efficiency. This research is discussed in detailed in section 6.2 in *Chapter 6*.

Influence of influent characteristics on sludge filterability and MBR operation

Parallel to the filterability tests, described in *Chapter 4*, plant operations and performances were monitored, analysed and linked with influent characteristics and sludge filterability. For this purpose, the plant process and membrane performance data were collected for the respective periods. Moreover, several characteristics of influent and effluent were analysed. Together with the information on removal efficiencies, the performances of each MBR were determined. Operational data collected from the investigated WWTP allow to undertake necessary analysis and further comparison studies of the MBR operations. In turn, the impact of influent characteristics on sludge filterability and operation of full-scale MBRs was evaluated. The results of the operational studies are presented in section 6.3 and 6.4 in *Chapter 6*.

Impact of membrane configuration on MBR operation: flat sheet versus hollow fibre

A measurement campaign was performed for a period of nearly two years. During those measurements, activated sludge samples were collected from four investigated MBRs and subjected to filtration characterisation test. The selected plants, namely MBR Heenvliet, MBR Varsseveld (Table 3.1) and MBR Fujifilm and MBR Rendac (Table 3.2), include MBRs treating municipal and industrial wastewater. Samples were collected directly from the membrane tanks or as close as possible to the membranes. Furthermore, parallel to the filterability tests, plant operation and performance were monitored, analysed and compared with the other investigated plants. For this purpose, design, operational and membrane performance data were collected from each MBR for the respective periods. This provides information on the effects of different membrane configurations on global performance of the MBR plants. The results and discussion are presented in section 7.2 in *Chapter 7*.

Impact of MBR plant layout on its operation: hybrid versus stand-alone

This research evaluates two different hybrid MBR configurations, i.e., in series and in parallel, and a stand-alone MBR. The impact of these MBR configurations on operation, performance, energy consumption and economy was evaluated. Three full-scale MBR plants were monitored for a period of 2 years, both in summer and winter period. Two of the plants – MBR Heenvliet and MBR Ootmarsum – are hybrid installations and one – MBR Varsseveld – is a stand-alone MBR (Table 3.1). During the research period, filterability of activated sludge, as a potential quality indicator of the MBR filtration process, was quantified experimentally by the DFCm. The filterability results were compared with automated image analysis results, influent and effluent characteristics and collected process data of the plants. Together with the removal efficiency information, the performances of the MBR plants were evaluated in environmental and economical terms based on major performance indicators as proposed by Benedetti et al. (2008) and Yang et al. (2010):

- effluent concentration of pollutants (mg/L),
- removal efficiencies of pollutants expressed as % of incoming load,
- energy consumption per volume of treated wastewater (kWh/m^3), and
- operational costs per population equivalent load (€/PE).

During the energy studies, total and specific energy consumption data were analysed, emphasizing the relation to treated flow, design capacity, membrane area and effluent quality. Additionally, economic studies were performed analysing the cost efficiency in design and operation of the full-scale MBR plants. The results of this research work are presented in section 7.3 in *Chapter 7*.

Energy consumption and energy efficiency of the full-scale membrane bioreactors

To research the specific energy requirements of MBRs, determine realistic operational costs and elucidate where possible future energy consumption reduction can be achieved, extensive research on the energy consumption in full-scale MBR plants was performed. Four full-scale MBR installations treating mainly municipal wastewater in the Netherlands were investigated and assessed (Table 3.1). The selected MBRs include plants equipped with flat sheet and hollow fibre membranes submerged in the separate filtration tank along with a plant equipped with sidestream externally placed tubular membranes.
Energy requirements of analysed MBRs were linked to operational parameters, and reactor performance. Total and specific energy consumption data were analysed on a long term basis with a special attention given to treated flow, design capacity, membrane area and effluent quality. Moreover, operational processes associated with aspects of energy efficiency are investigated in this study. Finally, a number of potentially available energy saving options related to design, operation and equipment were identified. The energy consumption and efficiency of the full-scale MBR installations is discussed in *Chapter 8.*

1.5 Thesis outline

The structure of the thesis and the aspects discussed in each of the chapters are presented in Figure 1.4. The chapters can be summarized as follows.

Chapter 2 deals with the fundamentals of wastewater treatment and membrane technology. The basics of MBR technology together with main advantages and disadvantages are presented. In the following section, membrane fouling, its causes, remediation, quantification and implications are addressed. Parameters to assess activated sludge quality, with a focus on filterability, are presented. The chapter final section provides the current knowledge on energy consumption in full-scale MBRs.

Chapter 3 describes the material and methods used during this research work. The research locations, analytical tools and mathematical techniques will be presented.

Chapter 4 reports on the results of the activated sludge filterability assessment carried out in the full-scale municipal MBRs. The differences in filterability along the MBR process flow and seasonal fluctuations of activated sludge will be commented.

Chapter 5 describes the experiments on the influence of seasonal temperature fluctuations on raw domestic wastewater composition and MBR sludge filterability. This chapter discusses and evaluates the results of the filterability, particle size distribution, respirometry, fractionation and set of physicochemical measurements. In this way, a temperature effect on activated sludge filterability is assessed.

Chapter 6 focuses on the influence of activated sludge and influent characteristics on activated sludge filterability and operation of full-scale membrane bioreactors. First, the results of an extensive survey on activated sludge characteristics affecting sludge filterability are presented. The DFCm, image analysis and set of standardized measurements were used to unravel correlations between activated sludge characteristics and filterability. Then, the impact of influent characteristics on sludge filterability and MBR operation is addressed. The last section of this chapter is dedicated to operation and performance of full-scale MBRs.

Chapter 7 deals with practical knowledge concerning the impact of different plant layouts and membrane configurations on the overall functioning of the MBR plant. Based on performed DFCm experiments and full-scale MBR data, comparison on the use of flat sheet and hollow fibre membranes was carried out. The analysis of the consequences on operation, process performance, treatment efficiency and operational costs is presented. Afterwards, an evaluation of a stand-alone MBR in comparison with two hybrid MBR configurations, i.e., in series and in parallel, is presented. The effect of design plants layout is discussed in terms of operation, performance and operational costs of the full-scale MBRs.

Chapter 8 provides an overview of current electric energy consumption of full-scale municipal MBR installations and available energy reduction opportunities based on literature review and case studies analysis. Moreover, design and operational issues of full-scale installations and associated aspects of energy efficiency are investigated in this study. Apart from a comprehensive investigation of energy issues, some economic aspects of membrane bioreactors are also commented. The last section of the chapter provides an analysis on energy efficient operation and potential energy savings in the MBRs.

Chapter 9 gives the general evaluation of the research, formulates the perspectives for future research directions and provides recommendations for MBR end users.

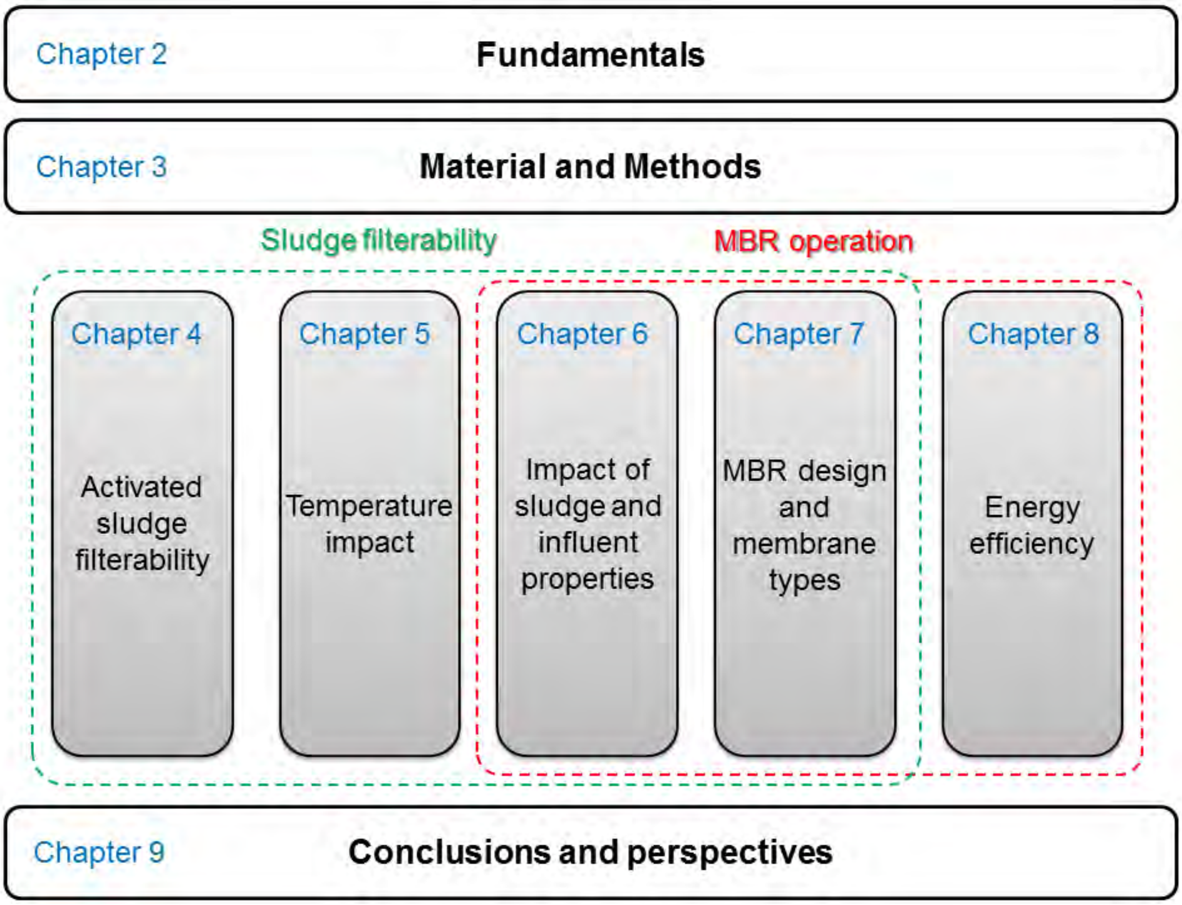

Figure 1.4: Schematic outline of the thesis

CHAPTER 2

MBR fundamentals

2 MBR fundamentals

2.1 Chapter outline

In this chapter fundamentals of municipal wastewater treatment and application of membrane technology to the MBR process are discussed (section 2.2 and 2.3). MBR fundamentals together with main prospects and constraints of the MBR technology are presented in section 2.4. Membrane fouling, its causes, remediation and implications are discussed in section 2.5. In addition, fouling quantification methods and parameters to assess activated sludge quality, with a focus on filterability, are presented in section 2.5. Energy consumption of the full-scale MBRs and energy related issues are presented and discussed in section 2.6. The description of the MBR energy consumption is restricted to information relevant to this research.

2.2 Activated sludge process

2.2.1 Process description

Since the development in early 1910s by Arden and Locket (1914a, b, 1915), the activated sludge process is widely applied for biological treatment of municipal and industrial wastewater around the world. The activated sludge process utilizes aeration, mixing and recirculation to activate biomass in order to remove organic constituents from wastewater. An activated sludge process consists of three processes in series:

- a bioreactor with suspended and aerated biomass responsible for biodegradation of wastewater,
- a liquid-solid separation step usually based on sedimentation,
- an activated sludge recycle transporting settled biological solids back to the bioreactor.

The activated sludge process is commonly a part of the complete treatment scheme carried out in a WWTP. The first step consists of pre-treatment to remove coarse material, sand and fat. Usually pre-treatment is followed by primary clarifiers in order to remove part of the suspended solids. The biological treatment, i.e., the activated sludge process, aims at the mineralisation and removal of organic matter. In order to provide biological nitrogen and phosphorus removal, the biological process can be modified by introduction of aerobic, anaerobic, anoxic conditions and internal recycle flows following a specified sequence. Finally, incorporation of additional treatment processes, e.g., sand filtration and/or disinfection, can provide tertiary treatment if improved effluent quality is required. The typical layout of a conventional activated sludge (CAS) process is presented schematically in Figure 2.1.

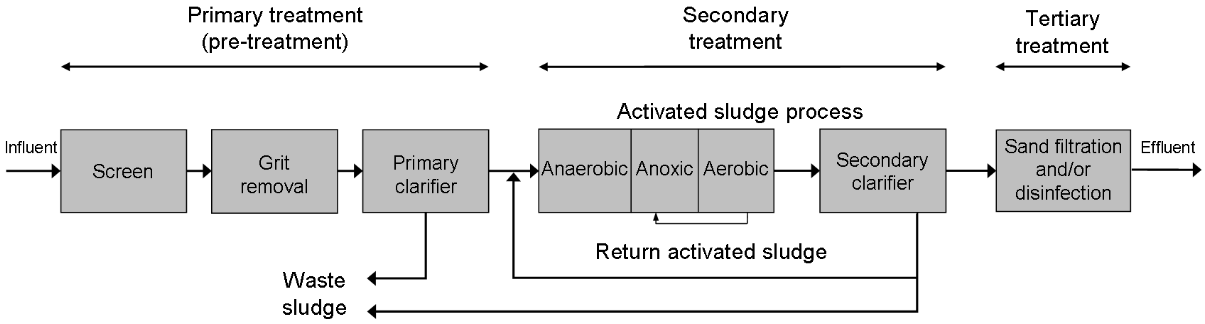

Figure 2.1: Scheme of a biological wastewater treatment plant (WWTP) with a conventional activated sludge (CAS) process (adopted from Metcalf&Eddy (2003)

The activated sludge process is capable of reaching effluent quality of <5 mgTN/L and <0.3 mgTP/L on regular basis, yet reaching effluent quality of <2.2 mgTN/L and <0.15 mgTP/L seems rather difficult (Van Nieuwenhuijzen et al. 2008). The worldwide popularity of the CAS process can be ascribed to good effluent quality at moderate cost, flexibility in design, ease of operation and high stability of the process. However, the process and, in particular, the effluent quality is strongly dependent on the performance of the final clarifier. Therefore, settling properties of the activated sludge are a limiting factor in the CAS process and main technology weakness. In response to this and among other disadvantages, such as effluent quality and space requirements, membrane technology and MBRs in particular were introduced to the water sector and applied for wastewater treatment.

2.3 Membrane technology

2.3.1 Membrane classification

Membrane filtration is a separation process in which a membrane acts as a physical and selective barrier between two phases. In the water treatment field, membrane is a finely porous medium allowing water to pass through the pores while retaining water constituents (Figure 2.2). The effectiveness of the separation process strongly depends on the membrane characteristics, e.g., pore size, porosity and material of the membrane.

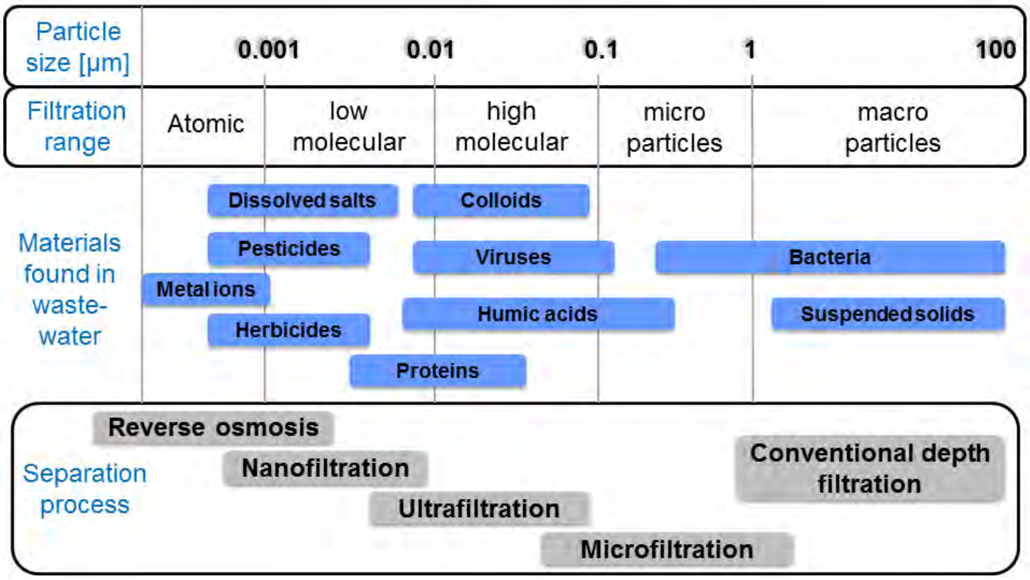

Figure 2.2: Filtration classification - overview of membrane separation processes and associated components removal (adopted from Metcalf&Eddy (2003) & Judd (2011))

The transport through the membrane can occur when a driving force is present, e.g., a gradient based on temperature, electric potential, concentration or hydraulic pressure. In water treatment, pressure driven membrane processes are usually applied and transmembrane pressure (TMP) is a driving force for permeation. Based on membrane selectivity, i.e., the pore sizes, the filtration processes used in (waste)water treatment can be classified as microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) and reverse osmosis (RO) (Table 2.1).

Table 2.1: Pressure driven membrane filtration processes used in water and wastewater treatment (Metcalf&Eddy 2003, Pinnekamp and Friedrich 2009, Judd 2011)

Membrane process	Pore size [nm][1]	Pressure [bar][1]	Dominant mechanism	Application
Microfiltration (MF)	100 – 1000	0.1 – 3	Size exclusion	Separation of solid matter from suspension
Ultrafiltration (UF)	10 – 100	0.5 – 10	Size exclusion	Separation of macromolecular or colloids, disinfection
Nanofiltration (NF)	1 – 10	2 – 40	Size exclusion + solution/diffusion + exclusion	Separation of dissolved organic molecules and polyvalent inorganic ions
Reverse osmosis (RO)	0.1 – 1	5 – 100	solution/diffusion + exclusion	Separation of organic molecules and of all ions

[1] Not strict thresholds values as different ranges are reported in the literature.

2.3.2 Membrane configurations

Depending on the manufacturing process, membranes can have a flat (planar) or a tubular (cylindrical) geometric form. These membranes are further available in different configurations, i.e., the geometry and the way it is mounted and oriented in relation to the flow of water. In general, six basic configurations are distinguished: flat sheet (FS), hollow fibre (HF), multi tubular (MT), spiral wound, pleated filter cartridge and capillary tube. The first three membrane configurations are predominant in the municipal MBR market. Examples of the flat sheet, hollow fibre and multi tube configurations are presented in Figure 2.3. Each of these configurations, namely flat sheet (Figure 2.3a), hollow fibre (Figure 2.3b) and multi tube (Figure 2.3c), has specific characteristics with its own advantages and disadvantages (Table 2.2) as discussed in the literature (Stephenson 2000, Judd 2006, van Bentem et al. 2008a, Brannock et al. 2010). According to Judd (2011), the ideal membrane configuration should be identified by:

- a high membrane area to module bulk volume ratio (or packing density),
- a high degree of turbulence for mass transfer promotion on the feed side,
- a low energy expenditure per unit product water volume,
- a low cost per unit membrane area,
- a design that facilitates cleaning, and
- a design that permits modularization.

Table 2.2: Characteristic of membrane modules used in MBR applications (Pearce 2008, Judd 2011)

Characteristic	Flat sheet (FS)	Hollow fibre (HF)	Multi tubular (MT)
Packing density [m^2/m^3]	40-150 low	200-500 high	150-300 low-moderate
Turbulance promotion	Fair	Very poor – fair	Very good
Operating mode	Cross-flow	Dead-end or cross-flow	Cross-flow
Flow through membrane	Outside-inside	Outside-inside	Inside-outside
Flux	Fair	High	Very high
Backflush	No[2]	Yes	Yes
Chemical cleaning frequency	Occasional	Frequent	Moderate
Cost	high	very low	very high
MBR configuration	submerged	submerged	sidestream
Main suppliers	Kubota, Toray, Weise, A3, Huber	GE Zenon, Mitsubishi, Koch, Siemens Memcor	Pentair (Norit), Berghof, Wehrle

[2] Some newer FS membranes are backflushable.

Separate membrane elements are joined together to form a module, which in turn are arranged into cassettes (FS, HF) or skids (MT). Multiple cassettes or skids can be put together to form a process train, i.e., membrane lines, as presented in Figure 2.3.

Figure 2.3: Images of (a) flat sheet, (b) hollow fibre and (c) multi tube membranes, modules and trains. Courtesy of Toray, GE-Zenon and Pentair (Norit), respectively

2.4 MBR technology

2.4.1 Process description

The technology of membrane separation of activated sludge, commonly referred to as MBR, is the combination of activated sludge treatment together with a separation of the biological sludge by micro- or ultra-filtration membranes to produce the particle-free effluent[3] (Figure 2.4). The membrane separation, a central process in MBR, replaces the sedimentation stage of

[3] www.mbr-network.eu

a conventional activated sludge process. Therefore, the settling properties of the activated sludge are not anymore a limiting factor in the process.

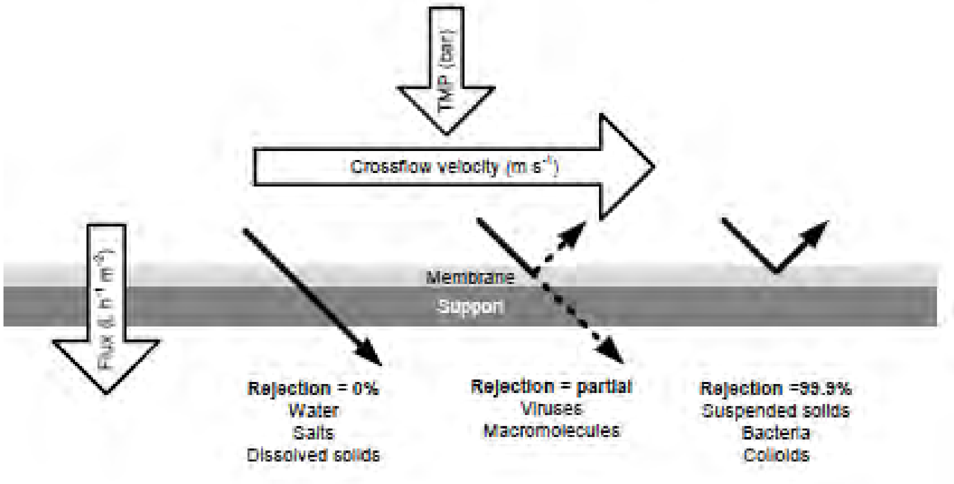

Figure 2.4: The fundamentals of the MBR membrane filtration process (Huyskens 2012)

The driving force in membrane filtration is the TMP, which is the pressure drop across the membrane, i.e., the difference between the feed side and the permeate side. The other main process parameters are flux and permeability. The permeate flux represents the quantity of produced water passing through a unit area of membrane per unit time. Since the flow through the membrane pores is considered laminar (Lojkine et al. 1992), the flux can be calculated based on the Darcy's law:

$$J=\frac{Q}{A}=\frac{TMP}{\mu_p \cdot R_{total}} \tag{2-1}$$

where,

J	–	flux, $[L/m^2 \cdot h]$
Q	–	effluent flow rate, $[m^3/h]$
A	–	membrane area, $[m^2]$
TMP	–	transmembrane pressure, [bar]
μ_p	–	apparent viscosity of the permeate, [Pa·s]
R_{total}	–	total resistance to filtration composed of the membrane resistance and the additional resistance resulting from fouling mechanism, [1/m]

The permeability (K), in $L/m^2 \cdot h \cdot bar$, is generally used as the parameter to express the performance of a membrane system under operation. The permeability represents the ease of flow through membrane, expressed by flux-pressure ratio:

$$K=\frac{J}{TMP}=\frac{1}{\mu_p \cdot R_{total}} \tag{2-2}$$

2.4.2 Technology and market development

The first MBRs were introduced to the market in the late 1960s, by Dorr-Oliver Inc. as an application for ship-board sewage treatment. The activated sludge process was combined with a cross-flow membrane separation, i.e., a sidestream MBR system. The combination of the two processes proved to be feasible and the MBR system entered the market with some success. However, due to energy intensive cross-flow pumping of the liquid, energy requirements of the first sidestream MBR installations were reported to be very high and about 6.0-8.0 kWh/m^3 (Van Dijk and Roncken 1997). Therefore, MBRs were only applied to the treatment of highly concentrated waste streams like ship-board sewage, landfill leachate and high strength industrial waste streams (Judd 2011). In 1989, an immersed MBR with membranes submerged in the bioreactor was developed by Yamamato et al. (1989). The submerged membranes concept reduces the pumping energy requirement, introduced air for fouling control and applied modest fluxes, subsequently, significantly reducing average power consumption. In combination with decreasing membrane cost, the MBR technology became competitive and the number of MBR applications has grown exponentially (Stephenson 2000). The increasing number of applications together with the growing interests of the research community resulted in further development of the MBR technology.

The number of MBR installations in operation increased from 154 in 2002 to 409 in 2005, and further on to about 800 in 2008 in Europe alone (Lesjean and Huisjes 2008, Huisjes et al. 2009). Another 258 full-scale MBR plants were in operation in North America by 2006 (Yang et al. 2006). By 2006, four MBR suppliers, namely GE Zenon, USFilter (Siemens), Kubota and Mitsubishi-Rayon, had more than 2250 MBR installations in operation or under construction worldwide (Yang et al. 2006). In addition about 3800 installations, albeit generally small, are reported to be installed in Japan (www.thembrsite.com). The total number of MBR installations provided by main market players, i.e., GE Zenon, Kubota and Mitsubishi-Rayon, holding 85-90% of the municipal MBR market was about 4400 (Figure 2.5) (Judd 2011).

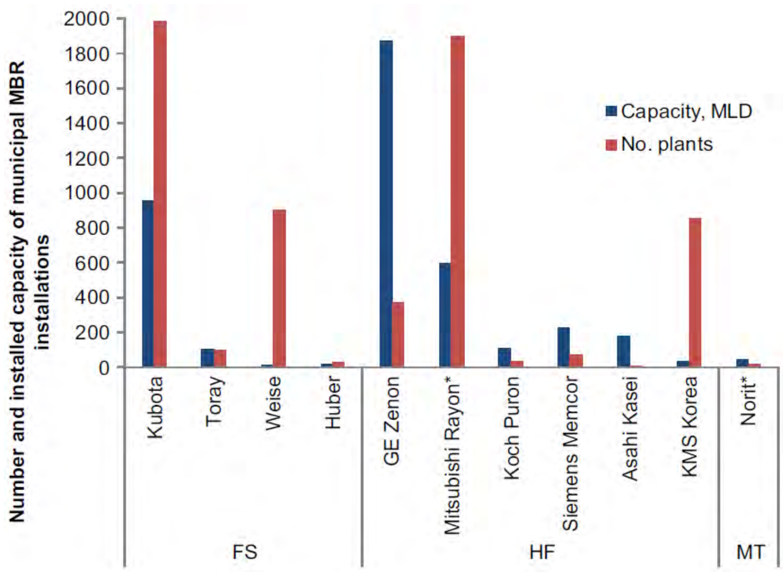

Figure 2.5: The MBR municipal market (Santos and Judd 2010). The capacity is expressed in megalitres per day (MLD)

The global market for MBR technology for wastewater treatment has steadily increased with an annual growth rates between 9.5 and 13.9% as stated in market analysis reports (BCC 2008, Frost&Sullivan 2008, BCC 2011). The value of the market was estimated at $337 million in 2010, is expected to approach $500 million in 2013 and to reach $627 million by 2015, BCC (2011). According to other market research report, the global MBR market will reach $1.3 billion by 2015 (GIA 2009). As such, the MBR market is growing faster than the market for other types of membrane systems (BCC 2011, Judd 2011).

2.4.3 MBR key drivers

The main drivers behind the growth of the MBR technology are: (i) the high quality of produced water, (ii) the increased water scarcity, (iii) the increasingly strict discharge quality legislation, (iv) the decreasing investment costs, (v) the acceptance of the technology and (vi) th epotential for upgrading existing WWTPs.

- **Quality of produced water**

  The effluent produced in the MBR, also called permeate, is of excellent and stable quality, free of suspended solids, with a low turbidity and partially disinfected. Thus, MBR permeate provides a positive hygienic and environmental impact.

- **Water scarcity**

  The increasing global water stress and local water scarcity highlighted the water resource problems and the importance of effective wastewater treatment and water reuse. The high quality effluent produced in the MBR permits to consider the use treated wastewater as an alternative water source. MBR effluents can be used, either directly or after additional treatment, for example with a reverse osmosis. In that way, freshwater consumption and demand can be reduced with a benefit for the water supplies.

- **Legislation**

  In order to challenge the global water scarcity and deteriorating freshwater quality, new regulations were introduced. The new regulations and associated more stringent environmental legislations are considered one of the key drivers behind MBR success, especially in the municipal sector. Introduction of more stringent discharge limits, e.g. EU Water Framework Directive (WFD), Urban Waste Water Treatment Directive (91/271/EEC), EC Bathing Water Directive in Europe or the Clean Water Protection Act (2009), the Pollution Prevention Act (1990), the Clean Water Act (1983) in the USA, were of most importance in relation to the MBR market growth.

- **Costs reduction**

  The capital costs of the MBRs and, in particular, membrane costs decreased significantly during the past 20 years (Churchouse and Wildgoose 1999, Kennedy and Churchouse 2005). Further decrease in the membrane costs is expected in the coming future (Judd 2011).

- **Acceptance of the technology**

  With the technological developments and membrane technology maturation, the confidence in and acceptance of MBR technology increased. Consequently, the decision-makers were more prone to select an MBR over other treatment technologies.

- **Potential for upgrading existing WWTPs**

  MBR technology can be a cost-effective option in case of upgrading and retrofitting of existing wastewater treatment plants, especially the ones based on conventional activated sludge process. Although, different options on how to modernize the WWTPs exists, two general solutions are distinguished, namely stand-alone and hybrid MBR systems. The small space requirements, leading to high capacity-to-footprint ratio, high quality effluent with a potentials of water reuse and design flexibility are reported as main reasons leading to suitability of MBR technology for the retrofitting market (Brepols et al. 2008a).

2.4.4 MBR configurations

Depending on the location of the membranes, MBRs may be configured as sidestream MBRs (Figure 2.6a) or submerged, also called immersed, MBRs (Figure 2.6b).

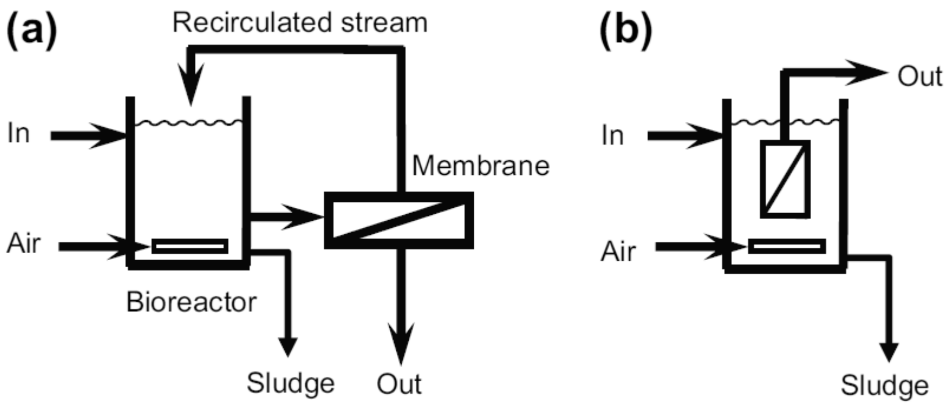

Figure 2.6: Process configurations of a membrane bioreactor: (a) sidestream and (b) submerged (Judd 2011)

The first MBRs were originally based on the cross-flow process and designed as sidestream MBRs. In this system, the activated sludge is recirculated through the externally located membranes in a loop. Tubular membranes are commonly installed in the sidestream MBRs. The sidestream concept is characterised by high cross flow velocities which provides good protection against membrane fouling. In result, high fluxes and high TMPs can be applied providing high performances, yet with the penalty of high energy requirements. Furthermore, sMBRs require more space compared to iMBRs for the externally placed membrane modules.

The MBRs with submerged membranes are typically equipped with hollow fibre or flat sheet membranes. The membranes may be submerged in a separate membrane tank or placed directly in the bioreactor. The membrane surface is typically scoured by coarse bubble aeration providing a high shear force and reducing membrane fouling. The submerged MBRs are characterized by reduced pumping energy requirements, but also smaller driving force, i.e., TMP, and subsequently lower fluxes.

The submerged MBRs are often applied to the treatment of high flows of a low strength wastewater, whereas sMBR are rather applied in case of low flows or high strength wastewaters. The comparison of the main characteristic of a sidestream and submerged MBRs is provided in Table 2.3.

Table 2.3: Summary of the main characteristic of sidestream and submerged MBRs (Judd 2004)

Characteristic	Sidestream	Submerged
Membrane location	external to bioreactor	inside bioreactor
Membranes	MT	FS, HF
Flow through membrane	Inside-out	Outside-in
TMP	2 – 6 bar	0.2 – 0.5 bar
Flux	40 – 100 $L/m^2 \cdot h$	10 – 40 $L/m^2 \cdot h$
Cross flow velocity	1 – 6 m/s	0.5 m/s
Turbulence promotion	Liquid cross-flow	Coarse bubble aeration
Cleaning strategy	In-situ	Ex-situ
Energy consumption	High	Low
Required footprint	High	Low
Flexibility	High	Low

Nevertheless, in recent years the differences between the membrane and MBR configurations tend to be less apparent, as the membrane manufacturers developed products, which combine features of different configurations. One of the examples is a concept of Pentair (Norit) called airlift system (Futselaar et al. 2007, Futselaar et al. 2012). The airlift system is a combination of traditional cross-flow sidestream MBRs and submerged MBRs. The sidestream filtration of activated sludge in externally located tubular membranes is combined with a bubble aeration known from submerged MBRs. The airlift system combines advantages of both configurations, e.g., high fluxes, accessibility and flexibility, with reduced power requirements. Another example of a combination of two technologies, developed by Microdyn-Nadir, is the Bio-Cel flat sheet membrane that can be backwashed (Krause and Dickerson 2011). This concept combines the advantages of submerged flat sheet (high flux) and hollow fibre membrane (backwash) modules.

2.4.5 MBR prospects and constraints

As already discussed in the previous section, MBR technology has many advantages, and some limitations, compared to conventional activated sludge process. The main prospects and constraints of the MBR technology are presented in Table 2.4.

Table 2.4: Prospects and constraints of the MBR technology (adopted from Metcalf&Eddy (2003) and Lousada-Ferreira (2011))

Prospects	Constraints
Excellent effluent quality with high-level disinfection	High capital costs (membranes)
Small footprint	Membrane fouling
Potential to operate at high biomass MLSS concentrations	High energy requirements for fouling control
High volumetric loading rates and thus short HRTs (compact design)	High O&M costs (fouling control, membrane cleaning chemicals)
Long SRTs and thus reduced sludge production	Eventual membrane replacement due to limited membrane lifetime
Effluent suitable for reuse applications	Regular chemical cleanings
Easy retrofit and upgrade of old wastewater treatment plants	Rigorous pre-treatment required due to clogging risk
No reliance upon achieving good sludge settleability	Qualified operators needed

2.5 Membrane fouling in MBRs

The single and uniform definition of fouling does not exist and vary from source to source. The most accurate, and thus popular, definitions are provided by the Union of Pure and Applied Chemistry (UPAC) and The MBR Book (Judd 2011). According to UPAC, membrane fouling is *the process resulting in loss of performance of a membrane due to the deposition of suspended or dissolved substances on its external surfaces, at its pore openings, or within its pores* (Koros et al. 1996). The MBR Book defines fouling slightly less comprehensively as *processes leading to deterioration of flux due to surface or internal blockage of the membrane* (Judd 2011). The fouling should be distinguished from the more serious problem of clogging. Clogging is the accumulation of solids within the membrane channels or between the sheets (Judd 2011).

2.5.1 Fouling mechanisms

Fouling, and thus foulants, can be classified in three different ways: (i) based on the mechanism, (ii) based on degree of permeability recovery and (iii) based on material nature of the foulants. Different fouling mechanisms may occur during membrane filtration, whereas with respect to MBRs, membrane fouling has been associated with the following three mechanisms:

- Adsorption (R_a): foulants with a size smaller than the membrane pore channels may be absorbed to the membrane pore wall thereby narrowing the pore channel.
- Pore blocking (R_{pb}): foulants with a size comparable to the membrane pore channels may stuck in the membrane channels causing pore blocking.

- Cake layer formation (R_{cl}): foulants with a size larger than the membrane pores may accumulate at the membrane surface forming a gel and/or cake layer.

Cake layer formation was reported to have major contribution to membrane fouling (Meng et al. 2009). The abovementioned mechanisms occur simultaneously during the filtration process as expressed by the resistance-in-series model. According to the model, the total resistance is the sum of the membrane resistance (R_m) and total fouling resistance (R_f):

$$R_{total} = R_m + R_f = R_m + R_a + R_{pb} + R_{cl} \qquad (2\text{-}3)$$

In practice, based on degree of permeability recovery, the fouling can be subdivided into reversible, residual, irreversible and irrecoverable fouling as graphically illustrated in Figure 2.7 and explained in Table 2.5.

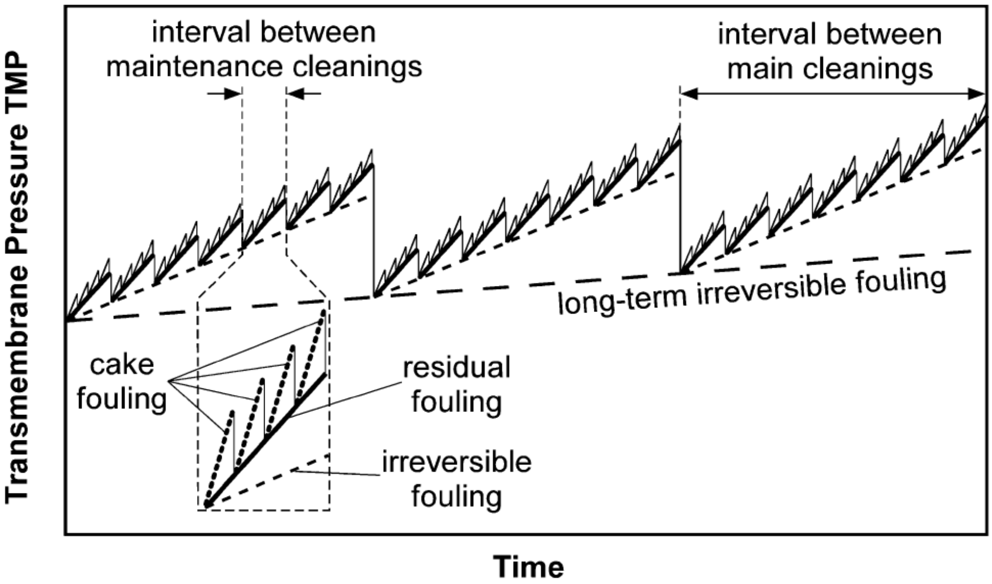

Figure 2.7: Fouling rates during long-term MBR operation; expressed as theoretical TMP over time under constant flux conditions (Kraume et al. 2009)

Table 2.5: Fouling classification and appropriate cleaning methods (adapted from (Kraume et al. 2009, Judd 2011)

Fouling definition	Fouling rate [mbar/min]	Time interval	Cleaning method
Reversible fouling	0.1 – 1	10 minutes	Physical cleaning (e.g. relaxation or backflush)
Residual fouling	0.01 – 0.1	1-2 weeks	Maintenance chemical cleaning (e.g. chemically enhanced backflush)
Irreversible fouling	0.001 – 0.01	6-12 months	Recovery only by chemical cleaning
Irrecoverable fouling	0.0001 – 0.001	Several years	Cannot be removed

Fouling also can be classified with respect to the nature of the foulants, based on their physical and chemical nature, e.g., size, surface charge, chemical type and origin. For a more complete overview of the foulants nature, the readers is referred to Judd (2011) or one of the membrane fouling review papers (Chang et al. 2002, Le-Clech et al. 2006, Meng et al. 2009, Drews 2010).

2.5.2 Factors influencing fouling

Despite some contradictions with respect to membrane fouling, it is generally accepted that the three major factors affecting membrane fouling are: feed and biomass characteristics, membrane properties and MBR operating conditions (Chang et al. 2002, Judd 2006, Le-Clech et al. 2006, Zhang et al. 2006, Drews 2010, Judd 2011). Thus, the efficiency of the filtration process in an MBR is governed by activated sludge filterability, which is determined by the interactions between the biomass, the wastewater and the applied process conditions. Due to interdependency of the aforementioned factors and dynamic nature of the feed and biomass nature, membrane fouling is a very complex phenomenon as schematically illustrated in Figure 2.8.

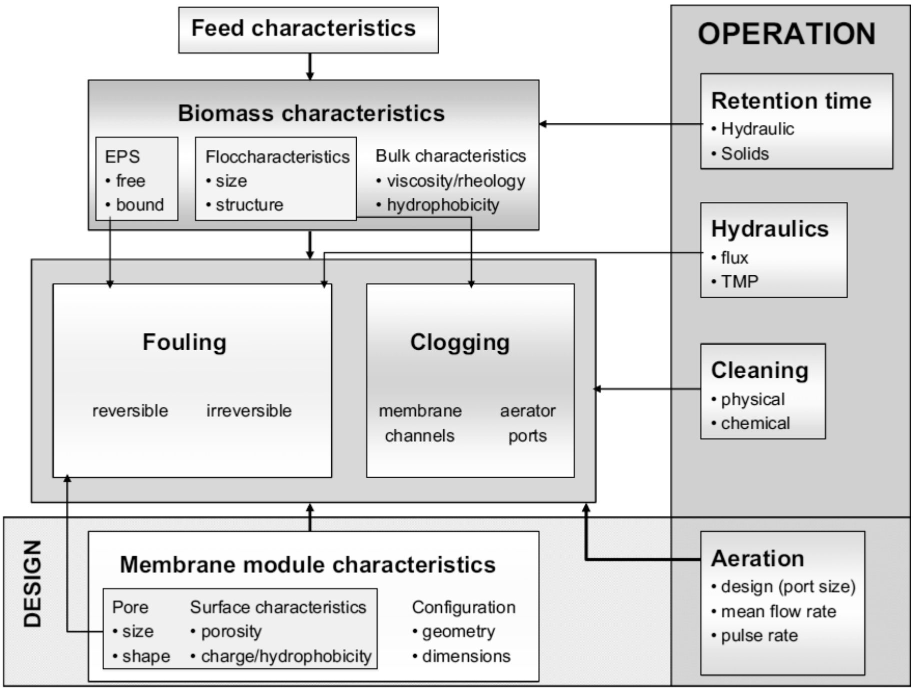

Figure 2.8: Inter-relationships between MBR parameters and fouling as proposed by Judd (2006, 2011)

This dissertation focuses on the impact of feed, i.e., influent, and biomass, i.e., activated sludge, characteristics affecting membrane fouling. Apart from feed and biomass, also influence of design and operation factors on membrane fouling will be investigated.

2.5.3 Remediation of membrane fouling

Despite extensive research efforts, the fouling process is still not fully understood and remains unavoidable. Fortunately, fouling may be reduced and its consequences to some extent can be remediated. Generally, membrane fouling prevention or mitigation actions are based on process control level, e.g., operation at critical or sustainable flux and improvement of hydrodynamic conditions by means of membrane air-scouring intensity. Membrane air-scouring is an essential part of all submerged MBRs as coarse aeration induces a turbulent current along the membrane and creates shear forces, reducing the cake layer on the membrane surface (Brepols 2010). During operation below or at the critical flux no or little fouling occurs, whereas at sustainable flux TMP increases at an acceptable rate and chemical cleanings are not necessary. The fouling consequences are traditionally cured by means of membrane cleaning, either by physical or chemical methods (Figure 2.9).

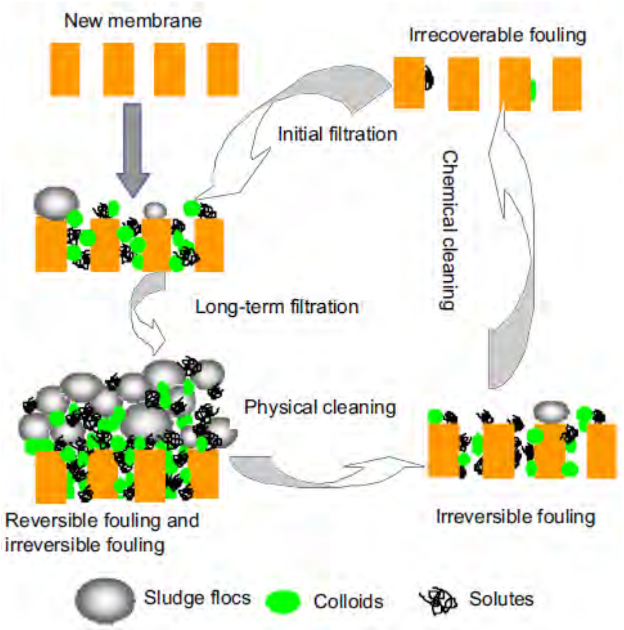

Figure 2.9: Illustration of fouling formation and fouling cleaning (Meng et al. 2009)

Physical cleaning techniques are based on turbulence promotion close to the membrane surface and include frequent relaxation and/or backwash steps. During relaxation, filtration and thus permeation is stopped, while the membrane is continued to be scoured with air. During the backwash step, permeate flows in the opposite direction, due to a reversed pressure gradient across the membrane.

Chemical cleanings are performed by either backwashing with cleaning agents as a maintenance cleaning in place or by intensive recovery cleaning out of place (Brepols 2010). The appropriate chemical cleaning protocol depends on the membrane supplier as each membrane manufacturer advises different cleaning protocol. Differences lie in the different criteria of when to start a chemical cleaning, different application methods, sequence of cleaning steps, cleaning frequencies, chemicals to be use and their concentrations. The choice of the cleaning agents mainly depends on the nature of the contaminants to be removed as different cleaning agents have different cleaning effects. The physical cleaning helps to maintain membrane performance over a short period, whereas chemical cleaning helps over a long period. Furthermore, both of the physical and chemical cleaning methods lead to discontinuous filtration process and thus lower permeate yield.

Alternatively, chemical reagents may be added to the mixed liquor in order to modify the activated sludge characteristics. The filtration enhancing additives, namely coagulants, flocculants, adsorbents and cationic polymers, all aim at fouling prevention and providing control over membrane fouling (Judd 2011).

2.5.4 Implication for MBR cost efficiency

The fouling of the membranes during MBR operation has negative consequences on the cost efficiency of the MBR process. Fouling affects both, capital (CAPEX) and operational (OPEX) expenditures. The CAPEX are influenced by installation of required equipment for fouling prevention or mitigation, e.g., pre-treatment, blowers, pumps, mixers, periodical membrane replacement due to limited lifespan of the membranes and, in some cases, installation of additional membrane area if operation at critical or sustainable fluxes is chosen. The OPEX are influenced by energy cost due to power required for aeration, pumping and mixing, chemical cleanings of the membrane and waste sludge treatment. The energy requirements account for the majority of the operational and maintenance costs (O&M) and often exceed 50% of the share. The main fouling prevention actions, i.e., air scouring of the membrane surface and liquid crossflow are the most energy intensive processes in submerged and sidestream MBR systems, respectively. The periodical physical cleanings are not an energy intensive processes, yet they still increase the total O&M costs. The chemical cleanings carried out to recover membrane performance and utilized cleaning agents also add to the total costs and environmental impact. Also the addition of any sort of filtration enhancing additives increases the operational costs. Finally, during membrane cleanings, filtration is not performed. Subsequently, permeate production is reduced. Thus, specific costs increase, leading to a less cost-efficient process.

2.5.5 Activated sludge characterisation: filterability

Membrane fouling strongly affects the performance of the filtration process, which in turn significantly influences operation and cost efficiency of an MBR. As previously mentioned in section 2.5.2, the biomass, i.e., activated sludge, properties are among the main factors influencing the membrane fouling process. In fact, membrane fouling is closely related with the activated sludge fouling propensity. Therefore, in order to relate fouling with influencing factors associated with biomass properties, activated sludge fouling propensities need to be quantified. Activated sludge filterability provides information about the sludge fouling propensity and as such, is often referred to as the quality of activated sludge (Lousada-Ferreira 2011). In order to determine the quality and to physically characterise the activated sludge, different parameters can be used (Clesceri et al. 1998, Metcalf&Eddy 2003). The most important ones are:

- Sludge Volume Index (SVI): determines the settleability of activated sludge. Represents the ratio between the volume and MLSS concentration of activated sludge sample after 30 minutes of settling. SVI provides information about flocculation state of activated sludge. However, as the undiluted MBR sludge usually does not settle, a dilution is necessary. The diluted SVI (DSVI) can be thus measured, yet the changes in the flocculation conditions should be taken into account.

- Capillary suction time (CST): determines the rate of water release from activated sludge, i.e., dewaterability of activated sludge.
- Time-to-filter (TTF): determines the time required to produce standard volume of filtrate. Likewise CST, TTF is a measure of sludge dewatering, thus appropriate for the sludge flow but not for the liquid flow.
- Sludge filtration index (SFI): determines the ratio of the filtrate production in a certain period and the MLSS concentration of the sample. SFI is a simple filtration test, based on time-to-filter method, proposed by (Raudies et al. 2007, Thiemig 2012) for the on-site filterability measurement by the plant operators to assess fouling potential of activated sludge.

Except SFI, the aforementioned parameters were developed for the CAS process, therefore they all present limitations when applied to MBR sludge. Therefore, in the past years, different research groups developed specific protocols often with advanced test cells, and associated parameters, to quantify the activated sludge filterability.

- Flux step method: determines the critical flux (Le Clech et al. 2003) and helps to assess the sludge fouling potential. This ex situ measurement has neither a standard set-up nor protocol defined, thus data comparison is difficult.
- Delft Filtration Characterisation method (DFCm): the small scale installation equipped with tubular membrane and standardised protocol was developed by TU Delft to measure activated sludge filterability (Evenblij et al. 2005). The DFCm is an ex situ filtration measurements described in detailed in section 3.3.
- VITO fouling measurement (VFM): the Flemish institute for technological research (VITO) has developed the VITO fouling measurement to characterise the reversible and irreversible fouling propensity of MBR sludge (Huyskens et al. 2008). VFM uses a tubular membrane, is operated based on a standardised protocol with filtration/relaxation periods and can be applied as in situ or ex situ filtration measurement.
- Berlin filtration method (BFM): developed by the TU Berlin and Berlin Centre of Competence for Water to provide information about reversible and irreversible fouling (De la Torre et al. 2009, De la Torre et al. 2010). The BFM uses a flat sheet membrane, is operated with filtration/relaxation intervals and determines the critical flux based on a modified flux step method. The BFM assesses the filterability of activated sludge in situ.

In this research, the DFCm was exploited because it was proven by previous researchers to be a good method, providing reliable and comparable results. In addition, a large database with significant amount of experimental data was available for the comparison.

2.6 MBR energy consumption

2.6.1 Background

Although energy is scientifically expressed in Joules (N·m), for the analysis into power consumption kilowatt-hour (kWh) is commonly used as energy unit due to practical reasons. The kilowatt-hour is a measure of energy typically used to express the power consumption of electric devices and as such is a billing unit for the consumed electrical energy. A kilowatt-hour is the amount of energy equivalent to 3.6 megajoules or to the energy demand of a device of 1 kilowatt running for 1 hour. The energy consumption may be also expressed specifically, e.g., per volume of treated wastewater (kWh/m^3), per membrane area installed (kWh/m^2), per design population equivalent capacity of the plant (kWh/PE_{design}), per pollution removed at the plant ($kWh/PE_{removed}$) or per various energy users/processes at the plant. This adds up to the complexity of the energy comparison between the various treatment plants.

At MBR plants part of the energy is provided to the biological treatment and part to the membrane operation. Therefore, it is of major importance to state if the published energy data are for the biological sub-system or membrane sub-system or for the combined system. Although it is worthy to indicate the energy consumption of the separate membrane filtration system, however, as the MBR term refers to the membrane and the bioreactor, they should be analysed and discussed accordingly. In addition, a membrane filtration system works jointly with the biological section and is thus (inter)dependent of it, e.g., the overall treatment performance achieved in an MBR strongly depends on membrane separation and biological treatment, provided membrane aeration can be utilized for the biological processes, etc. Hence, the energy efficiency obtained at one of the sub-systems might be related to the other sub-system. Therefore, both sub-systems contribute to overall energy consumption of an MBR system and should be mentioned. Moreover, unless detailed information is available about both of the sub-systems, the energy analysis should be based on the total energy consumption of the overall system.

Typically, power consumption of the following processes is considered to be related to the membrane operation: membrane aeration, membrane feed pumps (supply), sludge recirculation, permeate extraction, backwash and chemical cleaning pumps. The energy demand of the processes like aeration for biological purposes, propellers and/or mixers is considered to be related to the biological treatment part. Other energy usage required for plant operation, e.g., influent pumping, pre-treatment, sludge post-treatment, heating of the buildings, electricity for offices, control and operation, is generally considered as ‘the rest’.

In addition, the components of the treatment plant included in the analysis should be verified because different plants are designed differently and may include additional processes not included in other plants, e.g. post-treatment of activated sludge like thickening, dewatering or incineration may vary between the locations. Without knowing those details comparison of the different MBRs is most likely not complete. Nevertheless, such information is usually not provided making a fair comparison difficult.

Furthermore, because many parameters may have an important influence on the energy consumption the energy consumption, comparison should not be based only on the single average value of the specific energy consumption. It is due to the fact, that single value can be often misleading. For the sake of analysis comprehensiveness, additional information should be provided together with the average value: the length/period of the analysis, energy consumption at different flow conditions (hydraulic utilization), typical range of the specific energy consumption, concentration of organics (BOD, COD) and nutrients (NH_4), operational settings and processes included in the study. It is also of importance to study the general design of the system and understand potential factors that could influence the performed energy study. For example for the analysis, it is relevant whether the raw wastewater is treated directly or preceded by pre-clarification before the biological treatment. The presence of primary sedimentation is generally indicative for the presence of a sludge digester for energy recovery, which in turn reduces the overall energy consumption of the system.

In this research, attention was paid to the total, thus biological and membrane related, energy consumption of full-scale MBRs treating municipal wastewater. In order to compare similar systems, analysis boundaries were defined and whenever possible consisted of: the pre-treatment, the biological treatment, the membrane operation, the on-site sludge treatment and other energy users required for normal plant operation.

2.6.2 Literature review

In the past 50 years, developments in MBR technology resulted in an energy demand reduction from about 5.0 kWh/m^3, needed for the first cross-flow sidestream MBRs, to 1.0 kWh/m^3 in 2001-2005 and very recently to about 0.5 kWh/m^3 for the present Zenon submerged MBRs (Buer and Cumin 2010).

The energy requirement of the first tubular sidestream MBR installations, based on the cross-flow process, was reported to be typically 6.0-8.0 kWh/m^3 (Van Dijk and Roncken 1997), mainly due to energy intensive cross-flow pumping of the liquid. The introduction of the submerged membranes concept reduces the pumping energy requirement to 0.007 kWh/m^3 of permeate compared with values exceeding 3.0 kWh/m^3 required for the sidestream mode (Visvanathan et al. 2000). The submerged concept allows reducing average power consumption to 2.0 kWh/m^3 of treated water comparing to 3.0-4.0 kWh/m^3 for a sidestream MBR (Ueda et al. 1996). The sidestream systems became again competitive with the submerged system with introduction of the airlift system, where traditional cross-flow filtration is coupled with bubble aeration (Futselaar et al. 2007).

In 2003, Cornel et al. (2003) investigated the energy consumption of two full-scale municipal MBRs with and without a separate membrane tank. The one with membranes submerged in the aeration tank consumed about 1.0 kWh/m^3 and the one with a separate membrane tank about 2.5 kWh/m^3. In 2005, STOWA and Global Water Research Coalition published the State of the Science Report (STOWA 2005) on MBRs for municipal wastewater treatment, where energy consumption was reported to be in range of 1.5-2.5 kWh/m^3. Also Krause (2005) reported the specific energy consumption of MBR plants to be in the range from 0.8-

2.2 kWh/m^3. During the period of 2001-2006 the energy consumption of European MBRs was notably reduced from 2.0 to less than 1.0 kWh/m^3, mainly due to membrane module development and optimizations in process operation (Giesen et al. 2008). One of the key operational improvements was implementation of intermittent aeration, instead of continuous aeration, to reduce aeration energy demand (Buer and Cumin 2010). Other authors (Van der Roest et al. 2002, Lesjean and Luck 2006) also observed improvement in energy efficiency and reported the energy demand for full-scale municipal MBR installations to be about 0.9-1.0 kWh/m^3. Further improvement is possible, as the theoretical energy consumption for a full-scale municipal MBR with a separate membrane tank was estimated to be 0.8 kWh/m^3 (Krause and Cornel 2007). According to recent publications, even a better energy efficiency can be achieved and was estimated to be 0.5 kWh/m^3 for the present Zenon submerged MBRs (Tao et al. 2008, Tao et al. 2009, Buer and Cumin 2010, Tao et al. 2010) or when new mechanical cleaning process (MCP) and optimized PLC programming is used (Krause and Cornel 2007, Krause and Dickerson 2010, Krause et al. 2010).

The case of MBR Ulu Pandan and energy consumption at the level of 0.4 kWh/m^3 supports those findings (Tao et al. 2008, Tao et al. 2009, Buer and Cumin 2010, Tao et al. 2010). The energy optimization started already during the pilot-plant studies by increasing flux and lowering air supply by installing variable frequency drivers for the air blowers, subsequently leading to energy reduction from initial 1.3-1.7 kWh/m^3 to 0.8-1.1 kWh/m^3 (Tao et al. 2005). Further optimization was performed during the design of a full-scale plant (23,000 m^3/d); multiple treatment lines, gravity driven flow between bioreactor compartments and high throughput filtration protocol (9 min filtration, 1 min relaxation and backwash every 10 filtration cycles) lead to a potential consumption of 0.59 kWh/m^3 and guarantee power consumption lower than 0.7 kWh/m^3 (Tao et al. 2009). The baseline during normal operation was 0.5-0.6 kWh/m^3 mainly due to a stable net flux of 25.3 L/m^2·h and treatment of the settled sewage (Tao et al. 2008). The additional energy savings were made by reduction of MLSS to 6 g/L, leading to a further energy reduction down to 0.55 kWh/m^3. Recirculation optimisation reduced the energy consumption further to 0.54 kWh/m^3, implementation of ammonia-nitrogen and TOC meter for the biological aeration control to 0.48 kWh/m^3 and switch 10 sec on/30 sec off cyclic membrane aeration mode to 0.37 kWh/m^3, respectively (Tao et al. 2010).

Information on the energy demand of full-scale MBR plants published in peer-reviewed journals is limited. However, a considerable number of references can be found in other non-peer-reviewed publications. Typical energy demand values for MBR systems are reported to be in the range of 0.8-1.4 kWh/m^3 but a wide range of energy consumption figures are reported in the literature (Lazarova et al. 2010). For example, the energy usage of 7 German full-scale municipal MBRs was reported to be: 0.7, 0.8, 1.0, 1.0, 1.2, 1.6 and 1.8 kWh/m^3 (Palmowski et al. 2010). A summary of the energy requirements for various municipal MBRs is provided in Table 2.6 while Figure 2.10 presents histograms separated on basis of membrane configuration (Figure 2.10a) and flow rate (Figure 2.10b).

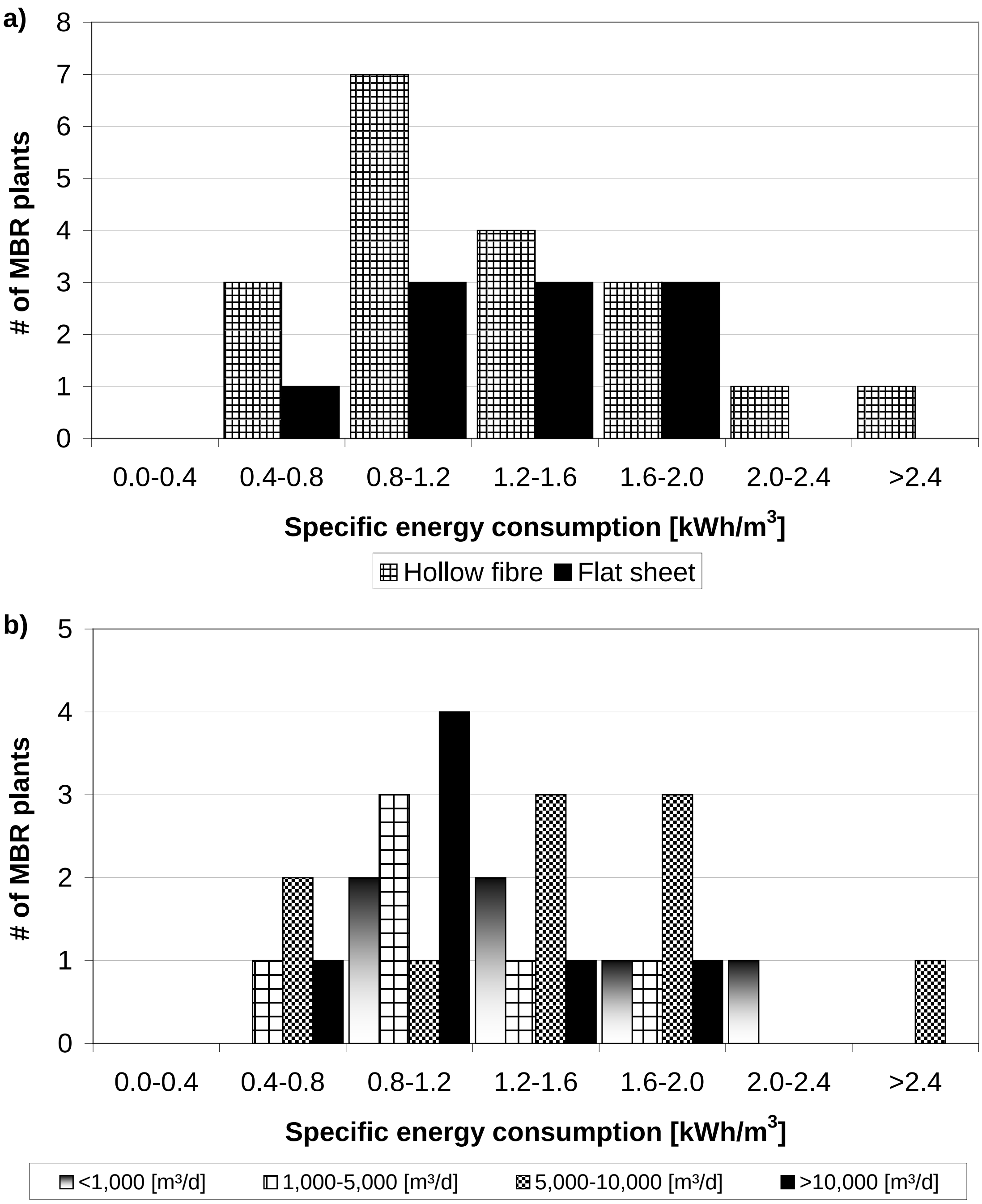

Figure 2.10: Energy consumption histograms on basis of: (a) membrane configuration and (b) flow capacity

Table 2.6: Energy consumption of various municipal MBR installations

Installation	Membrane type	Capacity	Dry weather flow	Rain weather flow	Start of operation	Period of analysis	Energy consumption	Reference
-	-	[PE]	[m^3/d]	[m^3/d]	-	-	[kWh/m^3]	-
Schwagalp (DE)	FS / Hubert	780	100	156	2003	N.A.	1.40	(Judd 2006)
Park Place (US)	HF / Memcor	N.A.	610	890	2003	N.A.	1.10	(Fatone et al. 2007)
METU Ankara (TR)	FS / Hubert	2,000	144	N.A.	2005	N.A.	1.0-2.0 (~1.4)	(Komesli and Gokcay 2010)
Grasse Roumiguières (FR)	HF / Zenon	24,000	6,250	N.A.	2007	N.A.	0.47-2.2	(Lazarova et al. 2010)
Glessen (DE)	HF / Zenon	9,000	2,000	6,500	2008	N.A.	0.90	(Brepols et al. 2009)
Rodingen (DE)	HF / Zenon	3,000	300	3,200	1999	2001	2.0-2.4	(Cornel et al. 2003, Brepols et al. 2009)
Markranstadt (DE)	HF / Zenon	12,000	2,700	4,320	2000	2001-2003	0.8-1.5 (~1.36)	(Cornel and Krause 2004, Giesen et al. 2008, Pinnekamp 2008)
Knautnaundorf (DE)	FS / Hubert	900	113	432	2002	2002-2003	1.3-2.0	(Judd 2006, Fatone et al. 2007, Giesen et al. 2008)
Cauley Creek (US)	HF / Zenon	N.A.	9,464	18,930	2002	2003	1.59	(Pellegrin and Kinnear 2010)
Brescia-Verziano (IT)	HF / Zenon	46,000	12,000	42,500	2002	2003-2005	0.85	(Fatone et al. 2007, Giesen et al. 2008, Wallis-Lage and Levesque 2009)
Monheim (DE)	HF / Zenon	9,700	1,820	6,900	2003	2003-2005	1.00	(Giesen et al. 2008)
Viareggio (IT)	HF / Zenon	24,000	5,250	6,000	2005	2006	<0,60	(Fatone et al. 2007)
Nordkanal-Kaarst (DE)	HF / Zenon	80,000	16,000	45,000	2004	2004-2005	0.4-0.9 (~0.9)	(Engelhard and Lindner 2006, Judd 2006, Fatone et al. 2007, Brepols et al. 2008b, Giesen et al. 2008, Wallis-Lage and Levesque 2009, Judd 2011)
Seelscheid (DE)	FS / Kubota	11,500	8,544	11,000	2004	2004-2005	0.9-1.7 (~1.5)	(Giesen et al. 2008, Pinnekamp 2008, Wallis-Lage and Levesque 2009)
Pooler (US)	HF / Zenon	N.A.	N.A.	11,400	2004	2005	1.74	(Pellegrin and Kinnear 2010)
Schilde (BE)	HF / Zenon	10,000	5,520	8,500	2004	2005-2006	0.62-0.64	(Garcés et al. 2007, Wallis-Lage and Levesque 2009, Fenu et al. 2010)
Fowler (US)	HF / Zenon	N.A.	N.A.	9,500	2004	2005-2007	4.23	(Pellegrin and Kinnear 2010)
Varsseveld (NL)	HF / Zenon	23,150	6,000	18,120	2005	2005-2009	0.75-1.0	(Giesen et al. 2008, Van Bentem et al. 2008b, Van Bentem et al. 2010)
Westbury (UK)	FS / Kubota	4,700	4,150	5,008	2002	2006-2007	1.98	(Ryan 2007)
Dundee (US)	FS / Kubota	N.A.	2,990	5,700	2005	2006-2007	0.66-1.23	(Stone and Livingston 2008)
Heenvliet (NL)	FS / Toray	3,300	912	2,400	2006	2006-2009	0.7-1.2	(Mulder et al. 2007, Mulder et al. 2008, Mulder 2009)
Ulu Pandan (SG)	HF / Zenon	N.A.	23,000	23,000	2006	2007	0.54-0.55	(Tao et al. 2008, Wallis-Lage and Levesque 2009)
Delphos (US)	FS / Kubota	50,000	5,700	45,500	2006	2007-2009	1.59-1.95	(Livingstone 2008)
Healdsbrug (US)	HF / Memcor	N.A.	6,057	15,142	2004	2008-2009	1.82	(Pellegrin and Kinnear 2010)
LOTT (US)	HF / Memcor	N.A.	N.A.	7,600	2006	2008-2009	1.61	(Pellegrin and Kinnear 2010)
Bonita Springs (US)	HF / Zenon	N.A.	15,250	N.A.	2007	2008-2010	1.43	(Pellegrin and Kinnear 2010)
Running Springs (US)	FS / Kubota	5,000	2,300	4,500	2003	2009-2010	1.3-3.0 (~0.7)	(Judd 2011)
Sabadell-Riu Sec (ES)	FS / Kubota	200,000	35,000	62,880	2008	2010	0.8-1.0	(Judd 2011)
Santa Paula (US)	HF / Koch	42,500	12,900	27,000	2010	2010	1.16	(Koch 2011)

Legend: HF – hollow fibre; FS – flat sheet; N.A. - not available

CHAPTER 3

Materials and Methods

3 Materials and Methods

In this chapter the materials and methods used during research work and described in this thesis are discussed.

3.1 Description of research locations

Four full-scale MBR plants treating mainly municipal wastewater were under investigation during this research work. Those MBR installations located in the Netherlands are: MBR Heenvliet, MBR Ootmarsum, MBR Varsseveld and MBR Terneuzen (Figure 3.1).

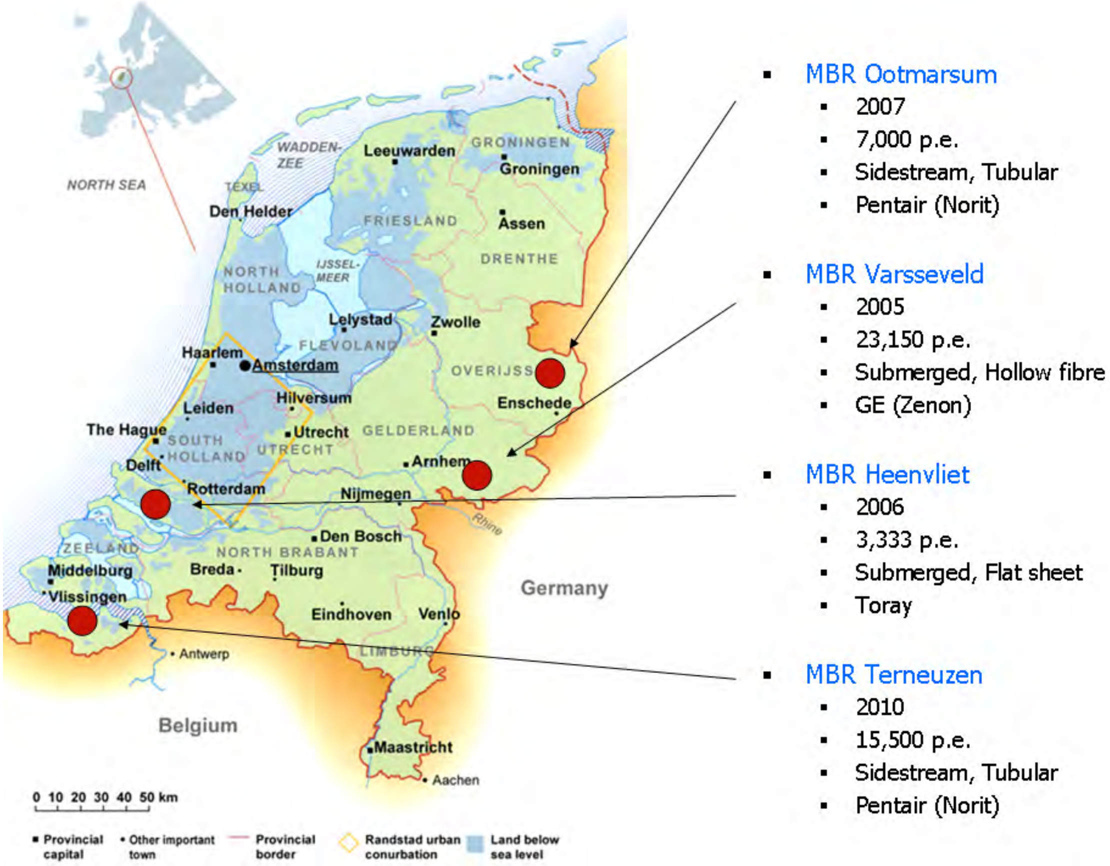

Figure 3.1: Overview of the investigated MBR plants in the Netherlands

The selected MBRs include plants equipped with FS and HF membranes submerged in the separate bio-aeration and filtration tank along with a plant equipped with sidestream externally placed MT membranes. Three of the plants are hybrid installations and one is a stand-alone MBR. A detailed description of the investigated plants is presented in Table 3.1. For additional technical information about the discussed wastewater treatment plants see elsewhere (Giesen et al. 2006, STOWA 2006, Futselaar et al. 2007, Giesen et al. 2007, Mulder et al. 2007, Geraats and de Vente 2008, Giesen et al. 2008, Mulder et al. 2008, Van Bentem et al. 2008, Futselaar et al. 2009a, Futselaar et al. 2009b, STOWA 2009a, b, Mulder et al. 2010, Van Bentem et al. 2010, Van 't Oever and Chia 2011).

Table 3.1: Characteristics and operational parameters of the MBR plants

Parameter	Unit	MBR Heenvliet	MBR Varsseveld	MBR Ootmarsum	MBR Terneuzen
Picture	–	 [1]	 [2]	 [3]	 [4]
Water Board	–	Hollandse Delta	Rijn&Ijssel	Regge&Dinkel	Scheldestromen
Start of operation	–	March 2006	January 2005	October 2007	June 2010
WWTP type	–	Hybrid	Stand-alone	Hybrid	Hybrid
WWTP configuration	–	MBR+CAS	MBR	MBR+CAS/SF	MBR+CAS
MBR configuration	–	Parallel or serial to CAS	Stand-alone	Parallel to CAS system	Parallel to CAS system
Membrane configuration	–	Submerged	Submerged	Sidestream	Sidestream
Membrane location	–	Separate filtration tank	Separate filtration tank	External	External
Membrane type	–	Flat sheet (FS)	Hollow fibre (HF)	Tubular (MT)	Tubular (MT)
Membrane supplier	–	Toray	Zenon–GE	Pentair (Norit X-Flow)	Pentair (Norit X-Flow)
Product name	–	Unibrane	ZeeWeed 500d	AirLift type 38 PRV/F4385	AirLift type 38 PRV/F4385
Flow through membrane	–	out-to-in	out-to-in	in-to-out	in-to-out
Membrane material	–	PVDF	PVDF	PVDF	PVDF
Number of lanes	m^2	2 parallel tanks	4 parallel tanks	6 skids	14 skids
Total membrane area	m^2	4,115	20,160	2,436	12,936
Packing density	m^2/m^3	115	304	308	308
Separation process	–	UF	UF	UF	UF
Membrane pore size	µm	0.08	0.035	0.038	0.038
MBR biological capacity	PE	3,333	23,150	7,000–9,250	15,500

[1] www.waterforum.net
[2] www.mbrvarsseveld.nl
[3] www.grontmij.nl
[4] www.scheldestromen.nl

Hydraulic capacity (DWF)	m^3/h	38 – 50 (serial) 20 – 30 (parallel)	250–300	75	250
Hydraulic capacity (RWF)	m^3/h	100	755	150	620
Hydraulic capacity (average)	m^3/d	1,480	5,000	2,600	9600
Design flux (net/gross) RWF	LMH	24/30	37.5/45	40-55/65	40/65
Average flux (net) DWF	LMH	12–24 (serial) 9–13 (parallel)	15–25	26–40	25–40
Maximum flux (net) RWF	LMH	45	50	65	65
MLSS	g/L	8–13	6–10	10–12	6-10
F/M ratio	gBOD/gMLSS*d	0.027-0.045	0.03–0.04	0.05	0.001-0.002
SRT	Days	31–40	24–26	40–42	20-25 (2010-2011) 15 (2012)
HRT	Hours	20 (serial) 28 (parallel)	4–14		10
Sludge production	m^3/d	45–50 (serial) 10 (parallel)	26–32	21–30	100-120
Pre-treatment	–	6 mm screens + 3 mm punch-hole	6 mm grid removal + sand trap + 0.7/1.0 mm microsieve	6 mm bar screens + 2 mm microsieve	6mm stepscreen + sand trap + 2 mm microsieve
Filtration period	min	9	6	7	7
Physical cleaning period	sec	60 (relaxation)	25 (backwash)	7 (backwash) + 30 (drainage; every 3-4 h)	7 (backwash) + 40 (drainage; every 1-3 h)
Cleaning types	–	Physical and chemical	Physical and chemical	Drainage stage & physical and chemical	Drainage stage & physical and chemical
Chemical cleaning frequency	–	1-4 per year	1 per week or 1 per 2 weeks	1 per month	1 per month
Cleaning agent #1	–	Citric acid	Sodium hypochlorite	Citric acid	Sodium hypochlorite
Cleaning agent #2	–	Sodium hypochlorite	Citric acid	Sodium hypochlorite	Citric acid
Aeration module – biology	fine bubble	disc diffusers	plate diffusers (2x1 m)	tubular diffusers	disc diffusers
Aeration module – membrane	coarse bubble	part of Toray module	part of Zenon module (1–2 cm holes)	part of Norit Airlift system	part of Norit Airlift system
Membrane aeration	Nm^3/h	1,200	4,105	360	280
Aeration intervals	Sec	Continuously	15 ON / 15 OFF	continuously	continuously
SADm	Nm^3 air/$m^2\cdot h$	0.3	0.2–0.6	0.3	0.3–0.6
SADp	Nm^3/m^3 permeate	12.9	12.3	6.0	13.3–15.0
Energy consumption	kWh/m^3	1.05±0.24	0.84±0.16	0.92±0.20 (WWTP)	0.97±0.15

Legend: DWF - dry weather flow; RWF - rain weather flow; UF – ultrafiltration; PE – person equivalent; SF- sand filter; CAS – conventional activated sludge; F/M - food to microorganism; *SADm –specific aeration demand per membrane area; SADp - specific aeration demand per permeate volume; N.A. - not applicable; N.P. – not provided;*

3.1.1 Other investigated MBR plants

Additionally, ten other plants of different scales located in Belgium and the Netherlands were selected for the particular sampling campaigns (described in sections 6.2 and 7.2). The selected set of MBRs includes plants treating municipal and a variety of industrial wastewaters, and equipped with different membrane types (Table 3.2).

Table 3.2: Type of wastewater, scale and membranes of the sampled MBRs

#	Location	Feed water	Scale	Type of membrane module (pore size, manufacturer)
1	Tervuren	Municipal	Full-scale	Flat sheet (0.4µm, Kubota)
2	Schilde	Municipal	Full-scale	Hollow fiber (0.04µm, Zenon)
3	KU Leuven	Municipal	Pilot-scale (15m³)	Flat sheet (0.08µm, Toray)
4	Boortmalt	Industrial – food	Full-scale	Hollow fiber (0.04µm, Koch – Puron)
5	Rendac	Industrial – rendering	Full-scale	Hollow fiber (0.04µm, Zenon)
6	Agristo	Industrial – food	Full-scale	Hollow fiber (0.04µm, Koch – Puron)
7	Undisclosed	Industrial – chemistry	Pilot-scale (9m³)	Flat sheet (0.04µm, Microdyn-Nadir)
8	Kloosterboer	Industrial – food	Full-scale	Hollow fiber (0.4µm, Mitsubishi – Sterapore)
9	Cargill	Industrial – food	Full-scale	Micro-tubular (0.03µm, Norit – X-flow)
10	FujiFilm	Industrial – chemistry	Full-scale	Flat sheet (0.08µm, Toray)

3.2 The Delft Filtration Characterisation method (DFCm)

The MBR research at Delft University of Technology was started with the work of Herman Evenblij (2006) and continued by Stefan Geilvoet (2010), Adrien Moreau (2010) and Maria Lousada-Ferreira (2011). To study the activated sludge filterability, a small-scale filtration characterization installation (DFCi) combined with a standardised measuring protocol, the Delft Filtration Characterization method (DFCm), has been developed by Evenblij et al. (2005a). Since then, the installation and the method were used by researchers of Delft University of Technology, but also other institutions, for many years (Evenblij et al. 2005a, Evenblij et al. 2005b, Evenblij 2006, Geilvoet et al. 2006, Geilvoet and Van der Graaf 2008, Moreau et al. 2009, van der Graaf et al. 2009, Geilvoet 2010, Lousada-Ferreira et al. 2010a, Lousada-Ferreira et al. 2010b, Moreau 2010, Gil 2011, Gil et al. 2011a, Gil et al. 2011b, Lousada-Ferreira 2011, Lousada-Ferreira et al. 2011, Van den Broeck et al. 2011, Gil et al. 2012b, a, Krzeminski et al. 2012a, Krzeminski et al. 2012c, Krzeminski et al. 2012d, Van den Broeck et al. 2012).

During those years, three versions of the installation with an identical design and characteristics have been used for the research. The only difference between the installations was the degree of its mobility. Evenblij and Geilvoet used the first, most stationary, generation of the DFCi. Moreau and Lousada-Ferreira applied the second, mobile, generation of the filtration characterisation unit. During this thesis the second and the third generation of DFCi were used. The design was identical for both except the third generation was further developed to improve the unit mobility and the ease of assembly and disassembly. Both, the second and the third generation, of the installation gave comparable results and with a strong correlation of R^2=0.88 and with a standard deviation between the test of ±0.067. Furthermore, results obtained with third generation of DFCi (DFCm III) were in average 7% higher compared to the results of second generation DFCi (DFCm II) (Figure 3.2). Therefore, the results from both installations can be considered as equal.

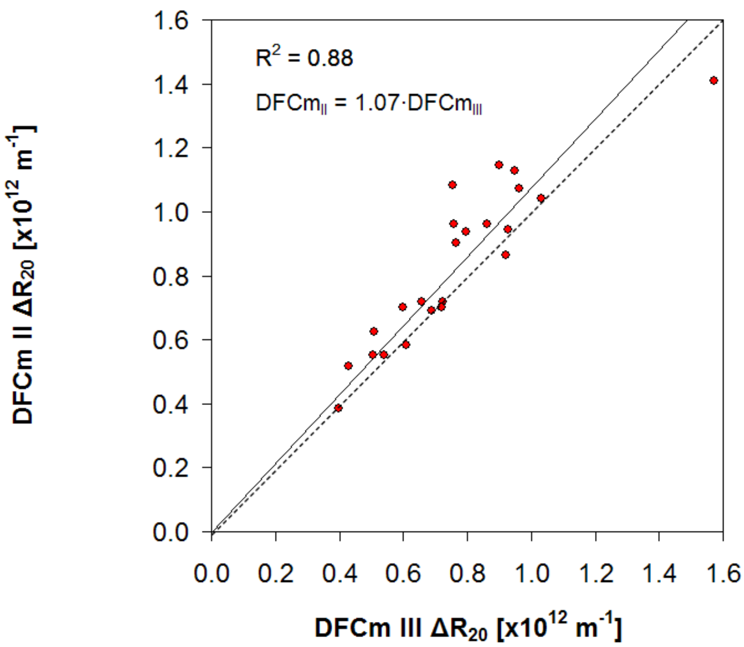

Figure 3.2: Comparison of the second and the third generation of the DFCi. Dotted diagonal is 1:1 line

The mobile construction of the installation permits measurements to be performed directly on the MBR site, minimising the time between sampling and sample analysing. Subsequently, the characteristics of the activated sludge can be assumed relatively unchanged.

The DFCm facilitates the measuring and characterisation of different samples of activated sludge under the same standardised conditions (Geilvoet and Van der Graaf 2008). By using the DFCm, activated sludge samples collected from different MBRs are filtered under identical operational (hydraulic) circumstances. Membrane characteristics and membrane operation remain constant and filtration behaviour can be related to the feed water characteristic, i.e., the activated sludge properties. Subsequently, differences in measured filterability of fed activated sludge are related, exclusively, to the properties of the MBR activated sludge. In this way, activated sludge from different MBRs can be compared.

The DFCm is a short-term experiment aiming at the filterability assessment of a sample at the certain time. Therefore, the DFCi is a measuring device and as emphasised before by Geilvoet (2010) and Moreau (2010) should not be considered a lab-scale MBR aiming at process simulation.

3.2.1 Delft Filtration Characterisation installation (DFCi)

The DFCi constitutes of a single tubular membrane, the equipment for sludge recirculation and permeate extraction, the membrane cleaning equipment and the data acquisition facilities. The filtration installation is presented schematically in Figure 3.3.

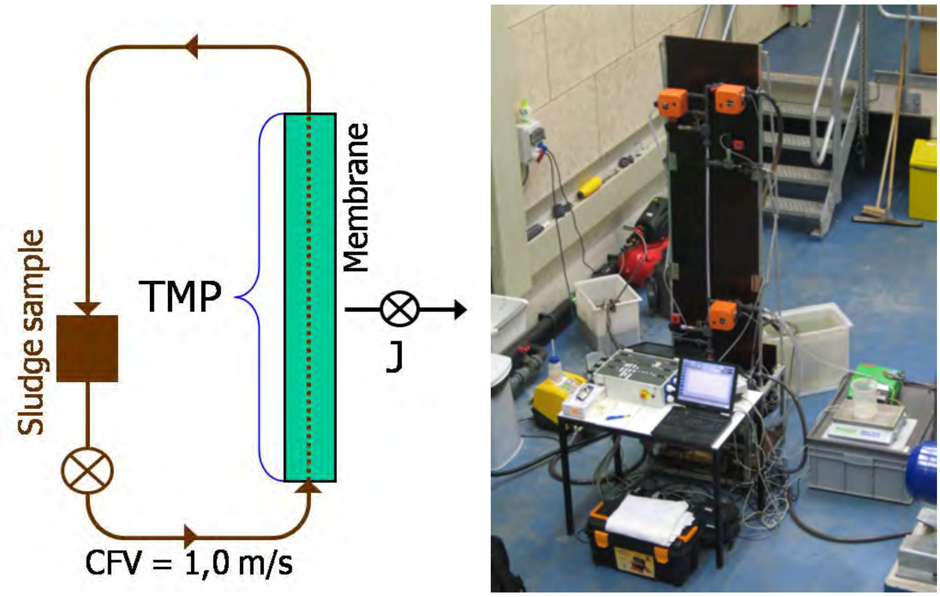

Figure 3.3: Schematic representation of the DFCm filtration characterization installation

The detailed characteristic of the membrane installed in the DFCi is presented in Table 3.3.

Table 3.3: DFCi membrane characteristic

Parameter	Unit	Value
Membrane type	–	tubular
Membrane configuration	–	sidestream
Membrane brand	–	Norit (Pentair)
Membrane type	–	X-flow F5385
Flow direction	–	inside-out
Membrane material	–	polyvinylide difluoride (PVDF)
Nominal pore size	μm	0.03
Membrane selectivity	–	ultrafiltration (UF)
Internal diameter	mm	8
Length	m	1
Membrane area	m^2	0.025

During each experiment activated sludge (or raw wastewater) sample is circulated through a membrane with a constant cross flow velocity of 1 m/s. Activated sludge circulation and permeate extraction is achieved by two peristaltic pumps. Extraction of permeate during a standard experiment is achieved at a constant flux of 80 L/m^2.h. At the end of each filtration test, the installation and the membrane itself are cleaned physically (mechanically) and/or chemically (see section 3.2.2). A detailed scheme of the DFCm filtration characterization installation is presented in Figure 3.4.

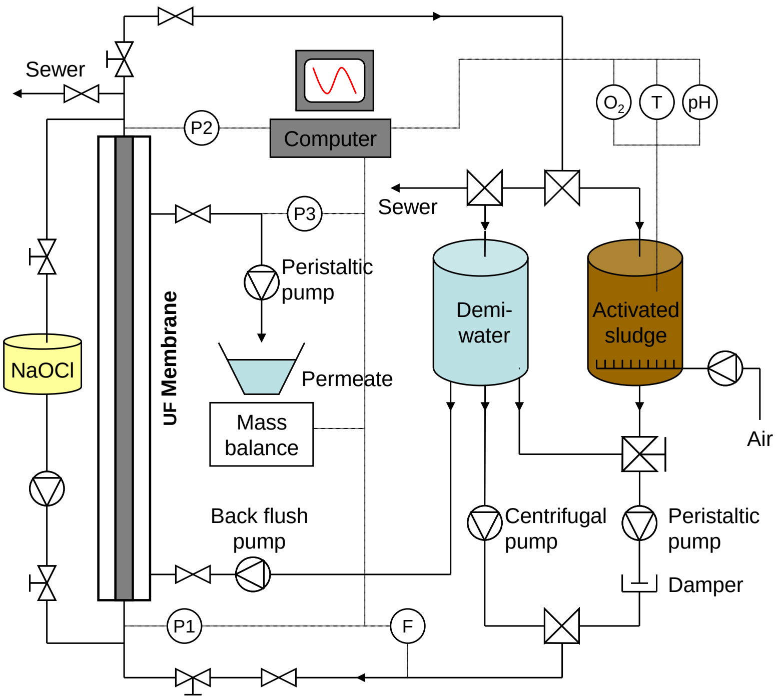

Figure 3.4: Detailed scheme of the DFCm filtration characterization installation (Geilvoet 2010)

During the filtration step, several parameters, such as, TMP, flux, cross-flow velocity, temperature, pH and dissolved oxygen concentration are monitored every 12 seconds and stored in a computer data file using the software application TestPoint (National Instruments). These parameters are plotted against the specific permeate production volume, allowing to control the filtration step online. The TMP is monitored during filtration with three pressure sensors installed at the feed, concentrate and permeate side of the membrane. Subsequently, filtration resistance is calculated according to the Darcy's law as presented in equation (2-1). For detailed description of the installation and method, the reader is referred to the work of the previous TU Delft researchers (Evenblij 2006, Geilvoet 2010, Moreau 2010, Lousada-Ferreira 2011).

3.2.2 DFCm measuring protocol

The DFCm measuring protocol is a standardised protocol for getting comparable results between various sets of filtration experiments. The DFCm protocol defined and explained by Evenblij et al. (2005a), Evenblij (2006) and Geilvoet (2010) comprises several steps, including the determination of membrane resistance, sample filtration, membrane cleaning. These steps, mentioned also by Moreau (2010) and Lousada-Ferreira (2011), are described as follows:

1. Membrane resistance determination:
    - Clean water recirculation at a cross-flow velocity of 1 m/s
    - Permeate extraction at a flux of 80 $L/m^2.h$
    - Filtration resistance determination with a threshold of $0.5 \cdot 10^{12}$ m^{-1}
    - Membrane cleaning when membrane resistance exceeds threshold value
2. Sample filtration:
    - Sample recirculation at a cross-flow velocity of 1 m/s
    - Permeate extraction, at a flux of 80 $L/m^2.h$, until a permeate production of 25 L/m^2 or a TMP value of 0.6 bar is reached
    - Flux adjustment in case of extremely bad or good filterability
3. Membrane cleaning:
    - Physical cleaning by forward flush with clean water at a cross-flow velocity higher than 5 m/s
    - Chemical cleaning by soaking the membrane with NaOCl (at 1500 mg/L) for at least 15 min.

After membrane cleaning, the clean water resistance is determined again to verify whether the membrane has been cleaned properly. If not, supplementary more intensive chemical cleaning can be applied, e.g. by extended cleaning time or by cleaning with citric acid. If the membrane resistance is still higher than the threshold value of $0.5 \cdot 10^{12}$ m^{-1}, the membrane needs to be replaced with a new membrane.

3.2.3 DFCm output and result processing

The main output of a DFCm experiment is the evolution of the resistance during the sludge filtration step. This resistance is calculated using Darcy's law and includes temperature correction of flux and permeate viscosity. The total resistance consists of membrane resistance and fouling resistance, also called added resistance (R_{add}). This added filtration resistance is plotted as a function of the permeate production per unit of membrane surface (Figure 3.5). As a result of membrane fouling during filtration, caused by cake layer filtration, filtration resistance will increase. The slope of the curve gives an indication of the activated sludge filterability, e.g., a steep curve corresponds to poor filterability. The calculated values of added resistance are used to express, with a power law equation, a filtration tendency as a mathematical equation. Afterwards, computation of a ΔR_{20} value is made.

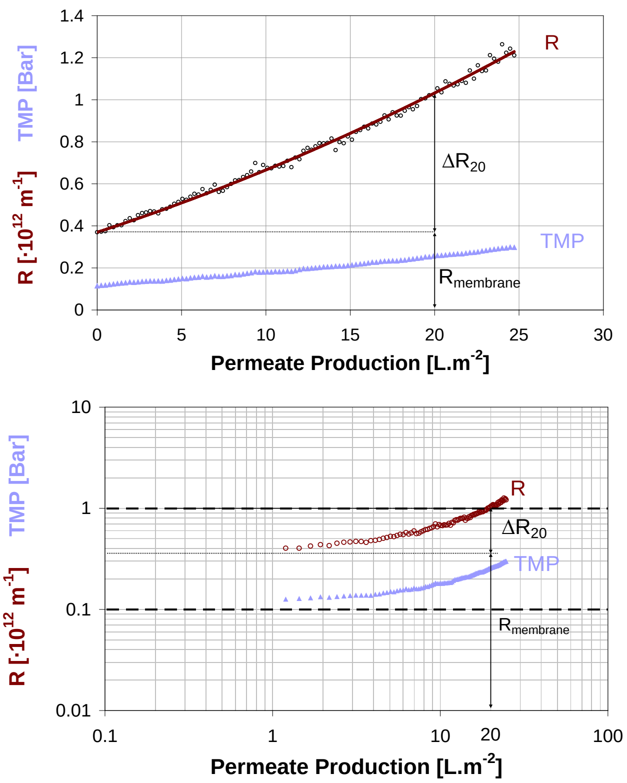

Figure 3.5: Typical DFCm output example: (a) linear scale (b) logarithmic scale

For easy comparison between different tests, the value ΔR_{20} is used (Table 3.4) based on the classification proposed by Geilvoet (2010). This value is defined as the increase in resistance after a specific permeate production of 20 L/m^2 (Figure 3.5).

Table 3.4: ΔR_{20} and corresponding filterability classification for municipal activated sludge samples

ΔR_{20} [10^{12} m^{-1}]	Classification
0 – 0.1	Good
0.1 – 1.0	Moderate
> 1.0	Poor

Using MBR sludge with bad filterability, i.e., > 3.0, too high cake resistance resulted in permeate production less than 20 L/m^2. In those cases ΔR_{20} values were computed by mathematically extrapolating resistance build-up until a predicted total volume of 20 L/m^2 permeate was reached. For the interpretation of ΔR_{20} values, an adaptation of the classification proposed by Geilvoet (2010) is used in this thesis. The original classification comprised only 3 classes and was only based on experiments with municipal activated sludge samples, which usually have better filterability compared to industrial activated sludge samples. In this thesis, both municipal and industrial samples were analysed. Hence, the scale was extended with an additional low class with 3.0 as cut-off level (Table 3.5).

Table 3.5: ΔR_{20} and corresponding filterability classification for industrial activated sludge samples

ΔR_{20} [10^{12} m^{-1}]	Classification
0 – 0.1	Good
0.1 – 1.0	Moderate
1.0 – 3.0	Poor
> 3.0	Bad

Besides the ΔR_{20} value, more detailed information about the membrane cake layer can be extracted from the DFCm outcome after detailed data analysis is performed. According to Geilvoet (2010), when the cake layer filtration theory is fitted to the DFCm output the coefficients a and b are derived to further characterize the cake layer.

$$\Delta R = a \cdot (V)^b \tag{3-1}$$

The compressibility coefficient (s) and the product of the specific cake layer resistance ($\alpha_R \cdot c_i$) can be calculated from the a and b coefficients:

$$s = \left(\frac{b-1}{b}\right) \tag{3-2}$$

$$(\alpha_R \cdot c_i) = a^{1/b} = a^{1-s} \tag{3-3}$$

Where:

α_R – specific cake resistance at reference filtration resistance [m/kg]

c_i – solids concentration involved in the fouling process [kg/m^3]

s – compressibility coefficient [-]

For a better understanding of the $\alpha_R \cdot c_i$ product and compressibility coefficient contribution to the total cake layer resistance increase, logarithmic scale representation (Figure 3.5b and Figure 3.6) was proposed (Geilvoet 2010). This representation of the DFCm results is possible when the power relationship between the resistance increase and specific permeate production is considered. An explanation of the DFCm data and results processing methodology was discussed in detail by Lousada-Ferreira (2011).

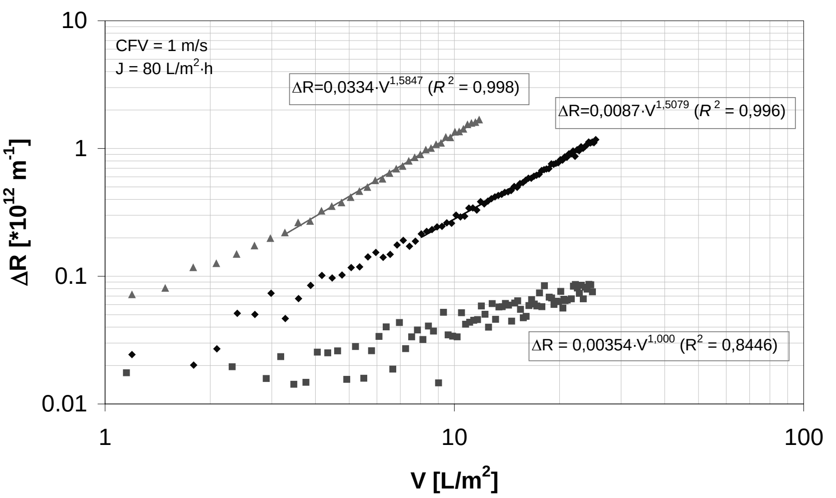

Figure 3.6: Example of DFCm output in double logarithmic scales (Geilvoet 2010)

The compressibility coefficient s expresses the compression potential of the cake layer. The coefficient value can vary between 0 and 1, where 0 indicate no compression and 1 indicate a complete compression. However, according to Geilvoet (2010), the compressibility coefficient can only be determined when the samples have moderate to poor filterability. It is due to the fact that the DFCm accuracy is approximately 0.05 and when a sample has a good filterability, i.e., $\Delta R_{20} < 0.1$, the correlation factor R^2 (Figure 3.6) is low. Subsequently, reliability of the compressibility coefficient is low in this case. Additionally, in case of activated sludge samples with good filterability the filtration resistance increases linearly with the specific permeate production (Geilvoet 2010).

3.3 Particle size distribution (PSD) analysis

The grab samples of MBR activated sludge, permeate and raw wastewater were submitted to particle counting.

3.3.1 Particle counting in the range of 2-100 μm

The particle size distribution of the sludge is determined with a particle size analyzer. The particle counting measurements in the range of 2 to 100 μm were performed using a Met One PCX particle counter. The instrument counts particles by the light blocking method with a LB 1020 sensor with a coincidence loss lower than 10% at 16,000 particles per mL. During analysis sample is directed into the sensor and funneled through an optical flow cell measuring 750 x 750 microns. Particle counting samples were diluted with permeate by a factor of 100 and sieved with a 100 μm sieve before the measurement to comply with maximal number of measured particles and to avoid instrument contamination (Lousada-Ferreira et al. 2010a, Lousada-Ferreira et al. 2011). For a detailed description of the equipment, method, sample preparation and data processing, the reader is referred to Lousada-Ferreira et al. (2011).

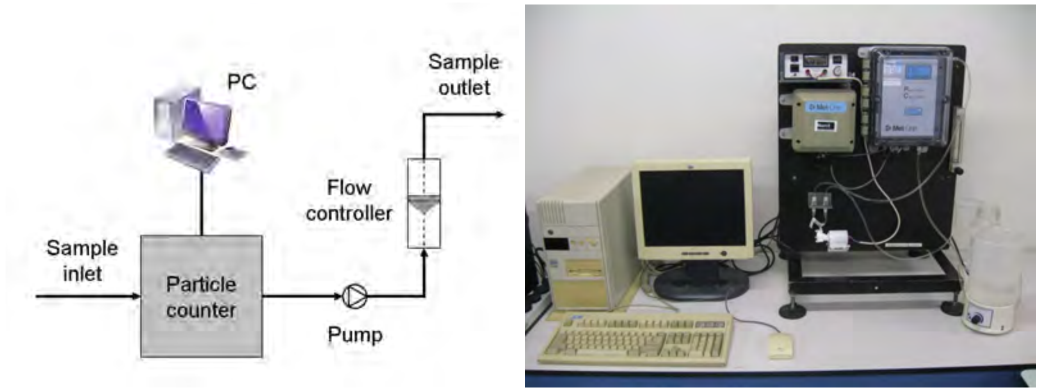

Figure 3.7: Scheme of the particle counting set up, in the range 2-100 µm

3.3.2 Particle counting in the range of 0.4-5.0 µm

The particle counting measurements in the range 0.4-5 µm were performed in a HIAC MicroCount 100 Series particle counter (Hach, USA) combined with "Particle Vision Online" software. The instrument counts particles through the light scattering method. The activated sludge samples were filtrated through a paper filter with a pore size of 7-12 µm prior to measurement, to avoid contamination of the particle counter. In addition, the samples were diluted with pre-filtered (0.1 µm) de-mineralized water by a factor of 100 to comply with the instrument upper detection limit of 100,000 particles/mL (Lousada-Ferreira et al. 2010b). The obtained results are corrected for the dilution and presence of particles in the de-mineralized water.

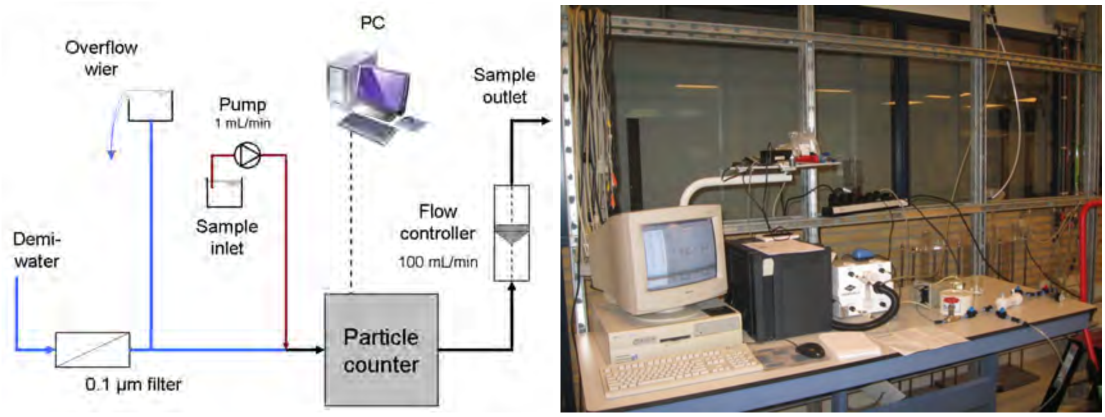

Figure 3.8: Scheme of the particle counting set up, in the range 0.1-5.0 µm

3.4 Biodegradability assessment

Biodegradability of the samples was assessed by oxygen uptake rate (OUR) measurement, which have been proposed by various authors as biodegradability indicator (Spanjers and Keesman 1994, Scott and Ollis 1995, Xu and Hasselblad 1996, Kujawa-Roeleveld 2000, Vollertsen and Hvitved-Jacobsen 2002, Capodici et al. 2010). The OUR, i.e., the oxygen consumption per unit volume per unit time, has been widely recognized as an important parameter in activated sludge and wastewater characterization as well as in biomass viability monitoring (Spanjers et al. 1998, Vanrolleghem and Spanjers 1998). The OUR was determined during respirometric experiments using a “static gas/static liquid” type of batch respirometer (Spanjers et al. 1996). The experimental set-up consisted of an aeration tank (volume of 35 L) for sample aeration connected to a flow-cell measuring unit equipped with an oxygen probe (WTW CellOx 325). The dissolved oxygen (DO) concentration was measured by the oxygen probe, equipped with a temperature sensor, and oximeter (WTW Oxi340), while the oxygen concentration was registered on a recorder. The samples used in the tests were aerated until concentration of dissolved oxygen reached stable conditions before running the respirometric test. The reactor was constantly aerated to maintain a dissolved oxygen concentration at elevated level, i.e., about 6.6-8.9 mg/L. Experiments were conducted at ambient (15.5±2.9 ºC) temperature.

Activated sludge was pumped to the air-tight oxygen measuring cell. After aeration and circulation were stopped, the sample in the vessel was kept agitated by magnetic stirrer. As no mass transfer occurred, the respiration rate could be directly deduced by measuring the decrease in DO. The measured data were recorded every second to calculate the OUR, as the slope of the linear decrease of DO concentration versus time. When DO concentration reached about 0.5-1.0 mg/L, and the endogenous respiration rate was reached, the air and flow valves were opened again.

The raw wastewater batch experiments were commenced by transferring about 300 mL of sludge to the measuring unit, immediately followed by starting the respiration measurement. After the endogenous sludge respiration was measured, a sample of wastewater, with known composition, was added and the respiration rate was recorded until the endogenous respiration rate was again reached and observed to be constant. Spiked and unspiked tests were performed to estimate organism response. The unspiked tests allowed obtaining the actual oxygen uptake rate of the mixed liquor sample from the full-scale system. The spiked tests allowed to measure, via OUR response, how the organisms in the mixed liquor respond to spiked substrate, i.e., wastewater (Spanjers and Vanrolleghem 1995, Dassanayake 2007).

To make comparison between the results from different experiments possible, the specific oxygen uptake rate (SOUR) was calculated. SOUR was expressed in milligrams oxygen per gram biomass per hour as follows: $SOUR = OUR / \mathrm{VSS}$. The VSS is the volatile suspended solids concentration of the sample that was used to perform the respirometric test. The SOUR normalizes the response to the ‘mass of organisms’ and allows comparison of oxygen response for different mixed liquors for each gram of ‘organisms’ (Dassanayake 2007).

3.5 Physicochemical analyses

3.5.1 Solids

The total suspended solids (TSS) and volatile suspended solids (VSS) measurements were conducted in accordance with the procedures described in APHA Standard Methods for the examination of water and wastewater (Eaton et al. 2005). TSS and VSS were measured for both, raw wastewater and activated sludge. The mixed liquor suspended solids (MLSS) and mixed liquor volatile suspended solids (MLVSS) are referred to TSS and VSS in the activated sludge mixed liquor, respectively.

3.5.2 Sludge volume index (SVI)

As the undiluted MBR sludge did not settle during 30 min of SVI test, a dilution was necessary to study sludge settling properties. The results were considered valid when settled sludge volume was equal to or less than 200 mL. The diluted sludge volume index (DSVI) test was carried out in accordance with the protocol given by Koopman and Cadee (1983) and Jenkins et al. (2003).

3.5.3 Analytical methods

Chemical oxygen demand (COD), total organic carbon (TOC), ammonia (NH_4-N), total nitrogen (TN) and total phosphorus (TP) concentrations were determined by photometrical methods with standard test kits Merck KGaA-Photometric method (Merck, D). To carry out the reaction and to determine the concentration, a thermoreactor TR 620 and a photometer NOVA 60, both Spectroquant series, were used respectively. Besides, in the MBRs where COD measurements were included in regular on-site monitoring of the treatment plant, filed monitoring values were often used. These COD measurements were determined by a composite flow-proportional sampling method.

In case of experiments described in section 6.2, the total organic carbon of permeate (TOC_{perm}) and supernatant (TOC_{sup}: supernatant of activated sludge centrifugated at $5000 \times g$ for 10 min at 4 °C) was measured with a Hach Lange IL-550 TOC/TN analyzer.

3.5.4 COD fractionation

COD fractionation was carried out according to the procedure of Nieuwenhuijzen (2002). The samples were filtered through different pore size cellulose nitrate membranes: 5.0, 1.2, 0.45 and 0.1 µm. Filtrate of each fraction was analyzed in terms of COD concentration with a standardised Merck test kit. The fractions were chosen for analyzing the effect of the different types of raw wastewater constituents and the role of each fraction in membrane fouling. The suspended and supra-colloidal fractions (>5.0 µm and between 1.2-5.0 µm, respectively) are related to particles that can be removed by physical or chemical methods, like sedimentation, filtration or flotation. Besides in municipal wastewater, the organic matter is distributed in fractions ranging from 1 nm up to 63 µm (Sophonsiri and Morgenroth 2004). The colloidal fraction can be removed by means of membrane filtration. The semi-dissolved fraction and dissolved or soluble fraction (0.1-0.45 µm and <0.1 µm, respectively) are postulated to have important impact on the filterability and consequently fouling. In addition, researchers often refer to dissolved fraction as the fraction smaller than 0.45 µm or 0.1 µm. Therefore, for reason of comparison, both of the classes were used.

3.5.5 Volatile fatty acids (VFAs)

Volatile fatty acids (VFAs) were determined by gas chromatography using Interscience FocusGC, equipped with a flame ionisation detector (FID) and a fused silica capillary column. The column had a length of 30 m, internal diameter of 0.25 µm and film thickness of 0.15 µm. The temperature of the injector and detector were maintained at 200°C and 230°C, respectively. The HP-Innowax column temperature was 130°C. Helium was used as a carrier gas. Every GC-vial was filled with 1.5 mL diluted calibration sample or diluted influent sample. Samples and the calibration were acidified by the addition of 10 µL of formic acid. The sample injection volume was 0.5 µL. To correct for injection volume error all dilutions were made with an internal standard solution, which contains approximately 300 mg/L pentanol. The instrument detection level for VFAs is about 5 mg/L (Siedlecka et al. 2008). The VFA concentrations were converted to COD equivalents by using VFA-to-COD stoichiometric conversion factors: 1.066 for acetic acid, 1.514 for propionic acid, 1.818 for butyric and iso-butyric acids, 2.039 for valeric and isovaleric acids, and 2.207 for caproic acid. The sum of measured acetic, propionic, butyric, isobutyric, valeric and isovaleric acids was regarded to be the total VFA concentration.

3.5.6 Biopolymer clusters (BPCs)

Biopolymer clusters (BPCs) are non-filterable organics present in the liquid phase of the MBR activated sludge and in the cake layer on the membrane surface. They are much larger than SMP and may range from 2.5 to 60 µm. They are neither microbial mass nor EPS. It is postulated that BPCs play a role in the formation of the sludge fouling layer on the membrane surface (Sun et al. 2008, Wang and Li 2008). The procedure to determine BPC concentration consists of centrifugation of the activated sludge sample during 30 min at 3200 rpm and measurement of the supernatant TOC afterwards. The difference between the supernatant TOC and the TOC determined in the permeate, is considered as the BPC concentration (Sun et al. 2008, Lin et al. 2009).

3.5.7 Extracellular Polymeric Substances (EPS)

The term EPS is used as a general term for different classes of macromolecules such as polysaccharides, proteins, nucleic acids, (phospho-)lipids and other polymeric compounds which can be found at or outside the cell surface and in the intercellular space of microbial aggregates (Le-Clech et al. 2006). EPS can be divided in two different classes, soluble EPS or SMP and bound or extractable EPS (eEPS)

In case of experiments described in *Chapter 5*, SMP were extracted from a 100mL activated sludge sample by means of centrifugation (5000G for 10min, at 4°C) followed by a filtration step on a pre-rinsed filter (Whatman Grade 1, rinsed with 100mL of Milli-Q water) (Le-Clech et al. 2006). The filtrate was analyzed for SMPs.

The eEPS was extracted by a heat treatment step (80°C, 10min) (Le-Clech et al. 2006) followed by a centrifugation (5000G for 10min, at 4°C) and a filtration step on a pre-rinsed filter (Whatman Grade 1, rinsed with 100mL of Milli-Q water). The SMP and eEPS fractions were then analyzed for proteins (PN) and polysaccharides (PS) according to a corrected Lowry method (Frølund et al. 1996) and the method of Dubois et al. (1956), respectively.

3.5.8 Soluble Microbial Products (SMP)

Soluble Microbial Products *(*SMP) are the soluble components which are released during substrate metabolism, biomass growth and cell lysis (Laspidou and Rittmann 2002). The concentrations of SMP were determined for both activated sludge and raw wastewater samples. In the SMP analysis, discussed in *Chapter 4*, the following materials were used: Thermo Electron Genesys 6 UV-Visible spectrophotometer, Vortex mixer Genie 2 G-5680 and 4 cm cuvettes.

For protein analysis the modified method of Frølund et al. (1996), based on the method of Lowry et al. (1951) and improved by Rosenberger and Kraume (2003) and Te Poele (2006) was applied. For the calibration Bovine Serum Albumin (BSA) (Acros Organics, Geel, Belgium), in a concentration range between 0 – 25 mg/l was used. Afterwards the concentration was calculated using the BSA calibration curve and the measured difference between the sample and the blank, i.e., demi-water.

Polysaccharides analyses were made according to the modified method of Rosenberger and Kraume (2003), based on the method of Dubois et al. (1956), and improved by Te Poele (2006). For the calibration D-glucose (J.T.Baker Co.), in a concentration range between 0.5-10 mg/l, was used. The concentration was calculated using the polysaccharides calibration curve and the measured difference between the sample and the blank.

3.5.9 Image analysis

A fully automated image analysis procedure, ACTIAS (ACTivated sludge Image Analysis System), was used for the characterization of the activated sludge composition (Jenné et al. 2007, Van den Broeck 2011, Van den Broeck et al. 2011).

All sludge samples were diluted to 1.0 gMLSS/L prior to microscopic analysis. The dilution was made with permeate to maintain the same environmental matrix, since a dilution with demineralized water could cause sludge deflocculation. The reason for diluting the sludge sample to 1.0 gMLSS/L is twofold: (*i*) at high sludge concentrations sludge flocs touch each other and no distinction can be made between neighbouring flocs, which would lead to an overestimation of object sizes; (*ii*) working at a normalized sludge concentration enables comparison of different sludge samples, i.e., with different biomass concentrations.

Activated sludge images were captured manually from two 10 µL drops on a carrier slide using a light microscope (Olympus BX 51) with phase contrast illumination (Ph1) and a total magnification of 100 times. The microscope is equipped with a 3CCD color video camera (Sony DXC-950P), which is connected to a computer. Microscopic images (90/sample) were digitized and stored as JPG (768x576 pixels) using Zeiss KS100.3 acquisition software. These images were subsequently processed by the developed image analysis procedure which is embedded in the MATLAB Image Processing Toolbox 4.2 (The Mathworks Inc., Natick, MA).

Two parameters were selected to represent the sludge flocculation state, i.e., activated sludge mean particle size and the surface fraction of activated sludge particles equal to 1 pixel (with 1 pixel=1.675 µm x 1.675 µm = 2.80 µm^2 (square) for this microscope-camera configuration). These parameters can be described as follows:

A_{mean} is the average surface of a sludge particle, calculated from all particles present in all processed images [µm^2/particle]:

$$A_{mean} = \frac{\sum_{i=1}^{all} A_i}{\#\, particles} \tag{3-4}$$

with A_i the surface of particle i [µm^2].

%1pixel is the total surface of particles with a size equal to 1 pixel (2.80 µm^2) divided by the total surface of all particles, calculated from all particles present in all processed images [%]:

$$\%1pixel = \frac{\sum_{i=1}^{all} A_i \mid A_i = 1pixel}{\sum_{i=1}^{all} A_i} \cdot 100\% \tag{3-5}$$

with A_i the surface of particle i [µm^2].

3.5.10 Hydrophobicity

A procedure based on the MATH-test (Rosenberg et al. 1980) was used to determine sludge relative hydrophobicity (RH). Sludge samples were diluted and washed twice with a phosphate buffered saline (PBS-buffer) to 2.5 gMLSS/L. This diluted sample is measured as the initial value at 650 nm (Dubois et al. 1956) in a spectrophotometer (DR5000, Hach Lange) with the filtrate from this sample as a blank. Next, 3 mL of this dilution sample is shaken vigorously with an equal amount of n-hexadecane for 2 min. After that, the sample is allowed to separate again for 5 min. The absorbance in the aqueous phase is then measured at 650 nm (Abs_f) and compared to the absorbance of the dilution sample (blank). The RH can then be calculated as follows:

$$RH = 1 - \left(\frac{Abs_f}{Abs_i} \right) \cdot 100\% \tag{3-6}$$

3.5.11 Surface charge (SC)

Activated sludge SC was measured with a colloid titration according to the work of Mikkelsen (2003) and Wilén et al. (2003). For colloid titration, polybrene (PB) and polyvinyl sulphate potassium salt (PVSK) were used as cationic and anionic reactants, respectively. The endpoint was determined from the colour change (blue to pink) of toluidine blue (TB). 1 mL of a 1.0 gMLSS/L activated sludge sample was diluted to 100 mL with permeate. To this solution, 5 mL of 0.001N PB and 5 mL of 0.05 g/L TB was added and stirred with a magnetic stirrer. Backtitration with PVSK was performed in triplicate. Backtitration of 100 mL permeate with PB and TB served as blank. The SC was expressed as milli-equivalents per gram MLSS of negative colloidal charge [meq/gMLSS].

3.5.12 Floc stability

The activated sludge dissociation constant (DC), being a measure for floc stability, was determined with a procedure similar to the method described in Zita and Hermansson (1994). First, activated sludge samples were diluted with permeate to 1.0 gMLSS/L. Next, 1L of the dilution sample was gently stirred (Heidolph RZR2051, 200 rpm) for 15 min in a glass beaker. After 15 min of sedimentation, 200 mL of the supernatant was taken and the turbidity was measured on a spectrophotometer (DR5000, Hach Lange) at 650 nm with the filtrate from this sample as a blank. Then, 200 mL of Milli-Q was added to the beaker and the sample was gently stirred again for 15 min. This procedure was repeated five times. The DC is defined as the slope of the accumulated turbidity versus the number of washing steps (Zita and Hermansson 1994).

3.6 X-ray analysis

The X-Ray powder Diffraction (XRD) and semi-quantitative X-ray Fluorescence (XRF) analysis were performed to precisely determine composition of the road salt. The analysis was carried out by Ruud Hendrikx from the Department of Materials Science and Engineering of the Delft University of Technology.

The XRD patterns were recorded in Bragg-Brentano geometry in a Bruker D8 Advance diffractometer equipped with a Vantec position sensitive detector and graphite monochromator. Data collection was carried out at room temperature using monochromatic Co Kα radiation ($\lambda = 0.179026$ nm) in the 2θ region between 10° and 110°, step size 0.0426 degrees 2θ. Step time 2 s. The samples were placed on a Si {510} substrate and rotated during measurement. Data evaluation was done with the Bruker program EVA. In the figures the measured XRD patterns are shown in black. The coloured red lines show the peak positions and intensities of the identified phases, such as found using the ICDD pdf4 database (ICDD). All patterns are background-subtracted, meaning the contribution of air scatter and possible fluorescence radiation is subtracted. The XRF analysis was conducted with Philips PW2400 X-ray wavelength dispersive Fluorescence Spectrometer and data evaluation was done with UniQuant 5.0 software.

3.7 Statistical analysis

Statistical analyses were carried out in order to investigate and estimate relationships between the filterability of the sludge and other analyzed parameters. The Pearson product momentum correlation coefficient was used to estimate linear correlations. Pearson's correlation coefficient (r_p) ranges from -1 to +1, where -1 is a perfect inverse (negative) correlation, 0.0 indicates no correlation, and +1 is a perfect direct (positive) correlation. The Pearson coefficient between 0.4 and 0.7 indicates moderate correlation between two parameters. The Pearson coefficient between -0.4 and 0.4 stands for weak correlation and the interrelation can be ignored in this situation (Moreau 2010, Gil 2011). Correlations were considered statistically significant at the 95% confidence interval ($p < 0.05$). Data were analyzed using the statistical program SPSS 19.0 (SPSS Corporation).

CHAPTER **4**

Activated sludge filterability

4 Activated sludge filterability

4.1 Chapter outline

This chapter discusses the results of the activated sludge filterability assessment carried out in the full-scale municipal MBRs. Section 4.2 deals with a general introduction and a brief description of the experiments and methodology used in this research. The results of the conducted experimental campaigns are presented in section 4.3, 4.4 and 4.5. The development of the filterability along the MBR process flow is discussed in section 4.6. Seasonal fluctuations of activated sludge filterability are analysed in section 4.7. Section 4.8 deals with a temperature effect on the activated sludge filterability.

4.2 Introduction

An extensive two-year long measurement campaign consisting of filterability tests at three research locations was performed. The experiments were carried out during summer and winter periods in order to follow seasonal variations and to identify differences in activated sludge characteristics and MBR operation between the seasons. The aim of the performed filtration characterisation was to elucidate the impact of activated sludge filterability and its seasonal changes, on operation and performance of municipal MBRs. Measurements representative for the summer period were performed in the months of June–August and for the winter period in the months of January–March. During those measurement periods, the DFCi was placed at the wastewater treatment plant for a period of 4 – 5 days. The DFCm (section 3.3) was exploited to determine the filterability of activated sludge samples. In this comparative analysis, three full-scale MBR plants located in the Netherlands were under investigation. The investigated plants, namely MBR Heenvliet, MBR Varsseveld and MBR Ootmarsum, are described in detail in Table 3.1. Activated sludge filterability tests and physical-chemical analyses were carried out. The filterability of the activated sludge was monitored in different compartments of the MBR, e.g. membrane tank, aerobic, anaerobic and anoxic. These results became an important reference point for further studies on the impact of sludge filterability on MBR functioning. The results presented in this chapter permit an assessment of the filterability influence on operation and performance (*Chapter 6*) as well as on energy efficiency and economy (*Chapter 8*) of the full-scale municipal MBR plants based on the longer time scale. Furthermore, relations between sludge filterability and membrane configurations as well as between sludge filterability and design layout of MBR plants could be discussed (*Chapter 7*).

4.3 Assessment of activated sludge filterability in full-scale municipal MBRs

The DFCm measurement campaigns as well as the number of filtration characterization experiments carried out under standardised conditions, as defined in the standard measuring protocol (J = 80 L/m^2.h^1, CFV = 1 m/s), are summarized in Table 4.1.

Table 4.1: Summary of the DFCm measurement campaigns and number of filtration characterization experiments under standard conditions (J = 80 L/m^2.h^1, CFV = 1 m/s)

#	Campaign location	Campaign period	Number of DFCm tests
1	Heenvliet	Jun-2008	23
2	Varsseveld	Jun-2008	16
3	Ootmarsum	Jun-2008	16
4	Heenvliet	Jan-2009	36
5	Varsseveld	Feb-2009	31
6	Ootmarsum	Feb-2009	26
7	Heenvliet	Jul-2009	41
8	Varsseveld	Aug-2009	30
9	Ootmarsum	Aug-2009	25
10	Heenvliet	Feb-2010	49
11	Varsseveld	Feb-2010	34
12	Ootmarsum	Mar-2010	27
13	Leuven[1]	Apr-2010	28
14	Leuven[1]	Aug-2010	25
15	Terneuzen[2]	2009/2010	17
16	Heenvliet[3]	2010/2011	32
	Total number of DFCm test		**456**

Almost half (49 %) of the analyzed activated sludge samples had a moderate filterability, whereas 34 % of the samples were characterised as poorly filterable. The smallest fraction of the measured samples, 17 %, was classified as the ones with a good filterability. The high diversity of sludge filterability highlights the importance of the activated sludge filterability studies and efforts towards identification of the filterability influencing parameters.

[1] Described in *Chapter 6*
[2] Described in *Chapter 8*
[3] Described in *Chapter 5*

4.3.1 Filtration characterisation at MBR Heenvliet

4.3.1.1 Plant description

The wastewater treatment plant of Heenvliet, operated by Waterboard Hollandse Delta (WSHD), is a plant that has been retrofitted from an original CAS system with a capacity of 8,950 population equivalents. The existing plant was upgraded with the membrane bioreactor, in order to meet more stringent effluent requirements and to provide a required capacity of 13,000 population equivalents. As the result, the new hybrid configuration is composed of two subsystems, the existing conventional line and the MBR system, which is in operation since May 2006 (Mulder et al. 2008).

A hybrid system consists of two subsystems (Figure 4.1), which are operated in series or in a parallel mode. In the series mode, the conventional system is followed by the MBR process line, thus water flows first through the conventional treatment process and then through the membrane treatment. In the parallel mode both systems work separately, part of the incoming water is treated in conventional system and part in the membrane bioreactor.

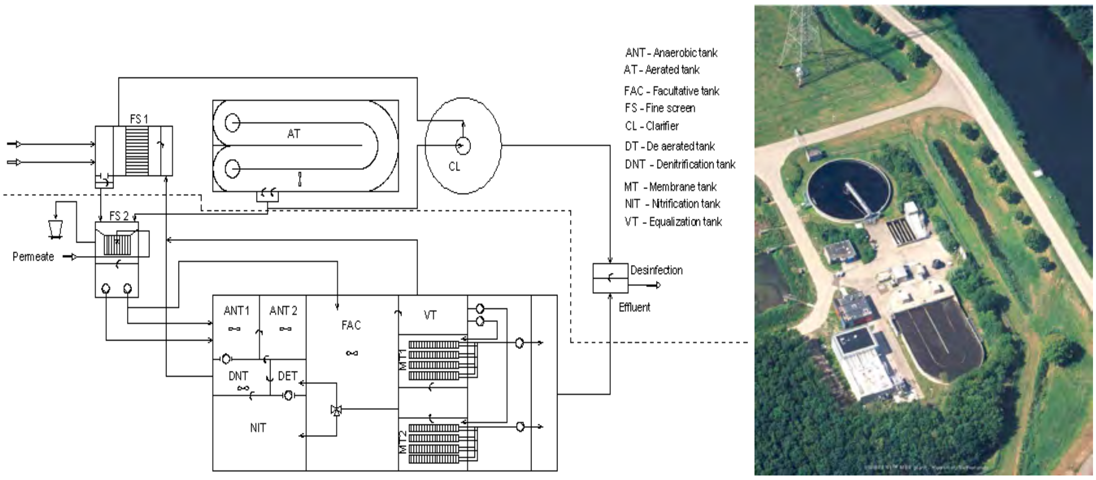

Figure 4.1: Scheme and aerial view of the Heenvliet WWTP (Keppel & Seghers)

In the series mode, at the dry weather flow, the entire incoming water goes through the 6 mm pre-treatment of the conventional system and is directed into the sequentially aerated activated sludge carrousel. The overflow water from the CAS carrousel goes partly to the MBR and partly, in case of rainy conditions, to the final clarifier. The part directed to the MBR is first pre-treated in the 3 millimetre punch-hole system. Afterwards wastewater is treated in several biological compartments after which it enters the membrane tank where the separation process is performed. In this way the membrane capacity is utilised as much as possible, and usually no water is discharged by clarifier overflow (Mulder et al. 2007). During dry weather the influent flow can even be lower than the design capacity and the permeate production of the MBR. In this case, the level in the secondary clarifier will be lower in order to create an extra buffer volume for storm weather event.

In case of the storm weather event, immediately following the low flow, the buffer volume can be used to accommodate higher flow without using the clarifier or even to collect the first peak of the flow for later treatment. When the flow increases, the permeate production rate of

the MBR will increase until it reaches its maximum, i.e., 100 $m^3.h^{-1}$. The amount of water exceeding maximal capacity of the MBR is leaving the plant via the secondary clarifier. In this way, even at peak situation, a substantial amount of the total flow is treated by the flat sheet membranes. The WWTP of Heenvliet, similar to most WWTPs treating combined sewage, has a bypass for extreme rain events. Hence, in case of extremely long and heavy rain periods when the capacity of the WWTP is exceeded, water is discharged directly.
Schematic representation of the in-series operation, originally published in a report of the Foundation for Applied Water Research (STOWA), is presented in Figure 4.2. The in-series mode has been in operation from the start-up of the hybrid configuration until March 2009, when the plant was switched to work in the parallel mode (Figure 4.3).

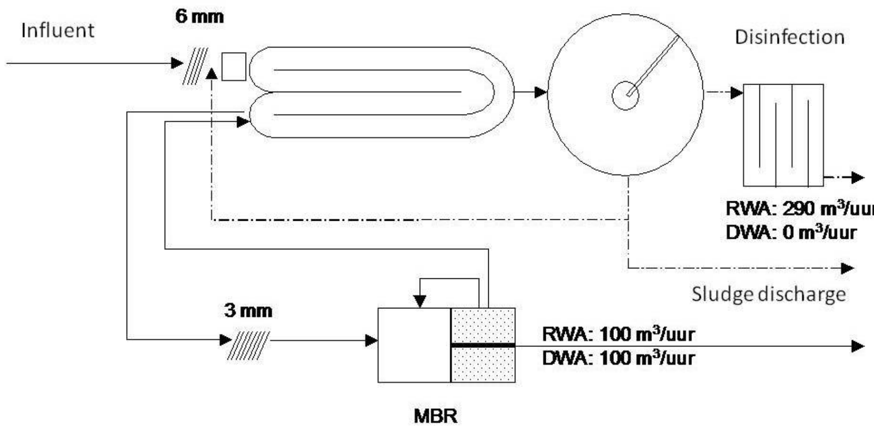

Figure 4.2: Schematic representation of in-series operation of hybrid MBR Heenvliet (STOWA 2009)

In the parallel mode, incoming water is split between CAS and MBR before the pre-treatment stage. Thus, the inflow to CAS goes through 6 mm bars before reaching carousel, whereas the MBR influent has a more rigorous 3mm pre-treatment. The MBR is always treating a fixed part, i.e., about 25% of the total influent flow. While operating in the parallel mode, both systems operate fully independent of each other.

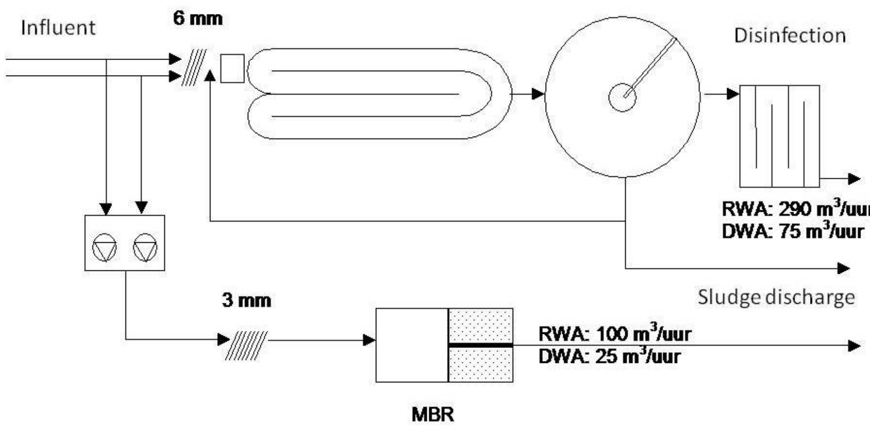

Figure 4.3: Schematic representation of parallel operation of hybrid MBR Heenvliet (STOWA 2009)

The membrane bioreactor system consists of pre-treatment, biological section with sub-compartments for nutrient removal and two membrane tanks. The detailed and schematic representation of the MBR system is presented in Figure 4.4. The biological sub-compartments are, in accordance with the flow, as follow: anaerobic tank nr 1, anaerobic tank nr 2 (where phosphorous is released), anoxic tank (denitrification zone), aerobic tank (nitrification zone) and a facultative tank which works mainly as denitrification zone but aeration for nitrification is also possible. After being submitted to several biological treatments, activated sludge enters the pre-aeration tank which is used to return the sludge and feed the two separated membrane tanks. The liquid-solid separation process is performed in the membrane tanks and the extracted permeate is stored in a clean-water tank (partially used for membrane cleaning). The effluent from the MBR is mixed with effluent from the CAS final clarifier before discharging to the channel.

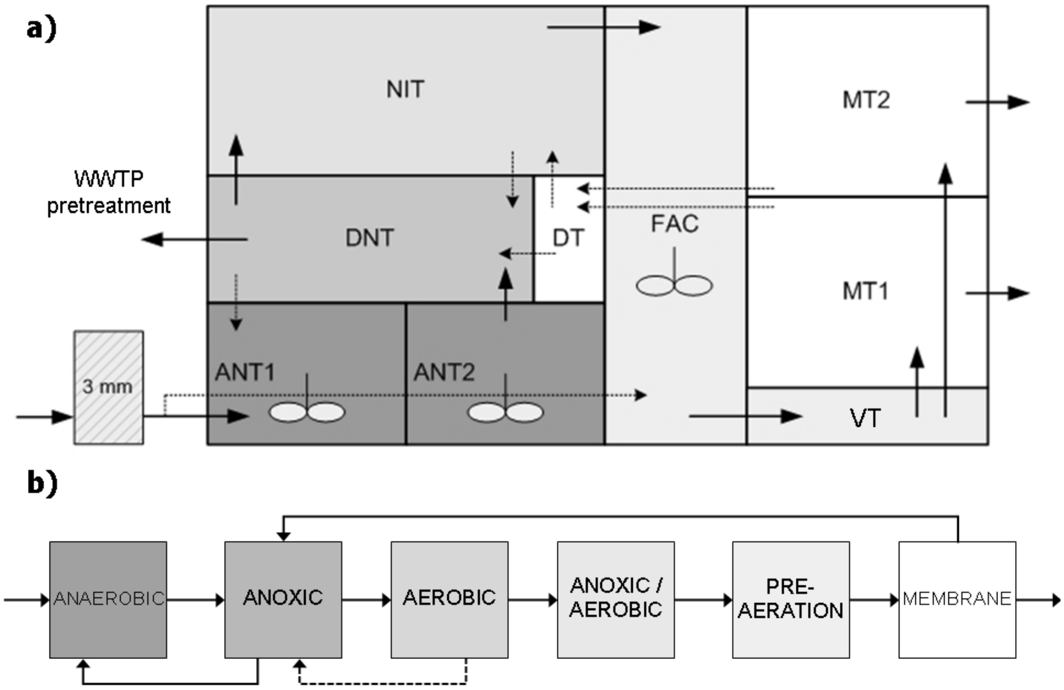

Figure 4.4: Detailed (a) and schematic (b) view of the membrane bioreactor system in Heenvliet (STOWA 2009)

Description and operational parameters of the MBR Heenvliet are presented in detail in Table 3.1.

4.3.1.2 Filtration characterisation

Filtration characterisation experiments were practised on-site during four measurement campaigns, two in the summer and two in the winter:

- 9th – 13th of June 2008,
- 26th – 29th of January 2009,
- 20th – 24th of July 2009, and
- 22nd – 25th of February 2010.

Activated sludge was sampled from each compartment of the MBR plant: anaerobic, anoxic (DNT), aerobic (NIT), facultative tank (FC), pre-aeration tank (VT) and two membrane tanks (MT). All activated sludge samples were submitted to filtration tests. A representative filtration characterization curve of the analysed samples for each experimental campaign is plotted in Figure 4.5. The results, expressed as added filtration resistance (R_{add}), obtained during research periods clearly demonstrate different filtration characterisation behaviour of the activated sludge samples in the summer and in the winter periods. The observed variations in a measured resistance for the membrane tank #2 (MT2) can be ascribed to the limited accuracy of the DFCm when working with sludge with a very good filterability (Figure 4.5a).

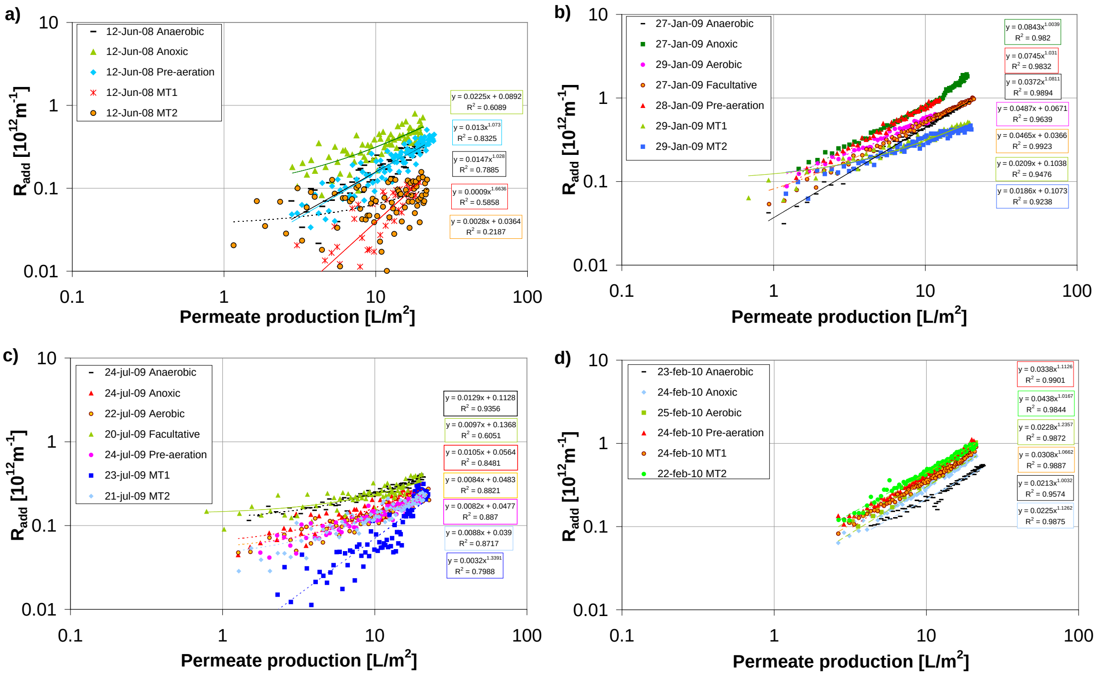

Figure 4.5: Output of DFCm measurements from (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010 measurement campaigns at MBR Heenvliet – representative filtration characterization curves

The filterability results of the samples taken from each tank are presented in Figure 4.6. Filterability varies between the research campaigns and can be considered most of the time as moderate, with poor filterability only in winter seasons.

During the first experimental campaign, conducted in June 2008, the filterability varied between the compartments and the sampling days and was within a range of $0.02{\cdot}10^{12}$ m^{-1} and $0.4{\cdot}10^{12}$ m^{-1}. Hence, the filterability of activated sludge could be considered as good to moderate depending on the sampling location and time of the sampling. The filterability was moderate in anaerobic, anoxic, pre-aeration and membrane tank #1 with average ΔR_{20} values of $0.23{\cdot}10^{12}$ m^{-1}, $0.44{\cdot}10^{12}$ m^{-1}, $0.35{\cdot}10^{12}$ m^{-1} and $0.11{\cdot}10^{12}$ m^{-1}, respectively. Good sludge filterability was measured in aerobic, facultative and membrane tank #2 with average ΔR_{20} values of $0.03{\cdot}10^{12}$ m^{-1}, $0.03{\cdot}10^{12}$ m^{-1} and $0.05{\cdot}10^{12}$ m^{-1}, respectively.

Also during the second experimental campaign, carried out in January 2009, the filterability varied between the compartments and the sampling days. The filterability was classified as moderate to poor as the measured ΔR_{20} values were within a range of $0.1 \cdot 10^{12}$ m^{-1} and $1.9 \cdot 10^{12}$ m^{-1}. Moderate filterability quality was measured in aerobic, membrane #1 and #2 tanks with average ΔR_{20} values of $0.95 \cdot 10^{12}$ m^{-1}, $0.29 \cdot 10^{12}$ m^{-1} and $0.31 \cdot 10^{12}$ m^{-1}, respectively. Poor sludge filterability was measured in anaerobic, anoxic, facultative and pre-aeration compartments with average ΔR_{20} values of $1.0 \cdot 10^{12}$ m^{-1}, $1.6 \cdot 10^{12}$ m^{-1}, $1.2 \cdot 10^{12}$ m^{-1} and $1.5 \cdot 10^{12}$ m^{-1}, respectively. A clear deterioration of activated sludge filterability was observed along the whole campaign as the ΔR_{20} values increased from day-to-day.

During the third experimental period, July 2009, a stable, yet with small fluctuations, and similar filterability was measured. The activated sludge filterability can be considered moderate in each of the compartments. The average ΔR_{20} values of $0.22 \cdot 10^{12}$ m^{-1}, $0.22 \cdot 10^{12}$ m^{-1}, $0.17 \cdot 10^{12}$ m^{-1}, $0.18 \cdot 10^{12}$ m^{-1}, $0.13 \cdot 10^{12}$ m^{-1}, $0.18 \cdot 10^{12}$ m^{-1} and $0.14 \cdot 10^{12}$ m^{-1} were determined in the anaerobic, anoxic, aerobic, facultative, pre-aeration and both of the membrane tanks, respectively.

During the fourth measurement campaign, performed in February 2010, filterability varied between $0.5 \cdot 10^{12}$ m^{-1} and $1.1 \cdot 10^{12}$ m^{-1} depending on the compartment that was sampled. Although the filterability was relatively stable a slight improvement was observed during the campaign. The average ΔR_{20} values of $0.5 \cdot 10^{12}$ m^{-1}, $0.7 \cdot 10^{12}$ m^{-1}, $1.1 \cdot 10^{12}$ m^{-1}, $1.1 \cdot 10^{12}$ m^{-1} and $0.8 \cdot 10^{12}$ m^{-1} were measured in the anaerobic, anoxic, aerobic, facultative, pre-aeration and both of the membrane tanks, respectively. Therefore the filterability was classified as moderate in case of samples originating from anaerobic, anoxic and both of the membrane tanks, whereas poor classification was assigned to aerobic and pre-aeration samples.

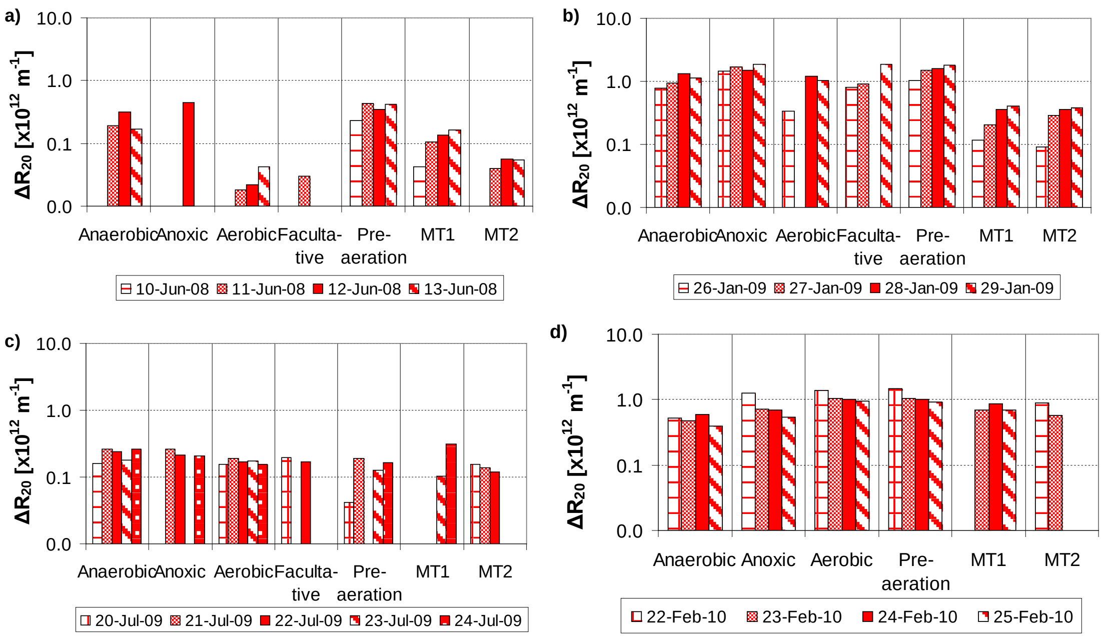

Figure 4.6: Filterability evolution along MBR process line during experimental periods at MBR Heenvliet: (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010

Monitoring of filtration characteristics demonstrated also clear differences between the investigated compartments (Figure 4.7). The worst activated sludge filterability was measured in one of the first biological sections of the MBR – namely anaerobic and anoxic. Different behaviour was only observed once, namely during the 'winter 2010' campaign, when the samples originating from the pre-aeration zone were classified as the worst filterable samples. The development of filterability in MBR compartments is further discussed in section 4.6.

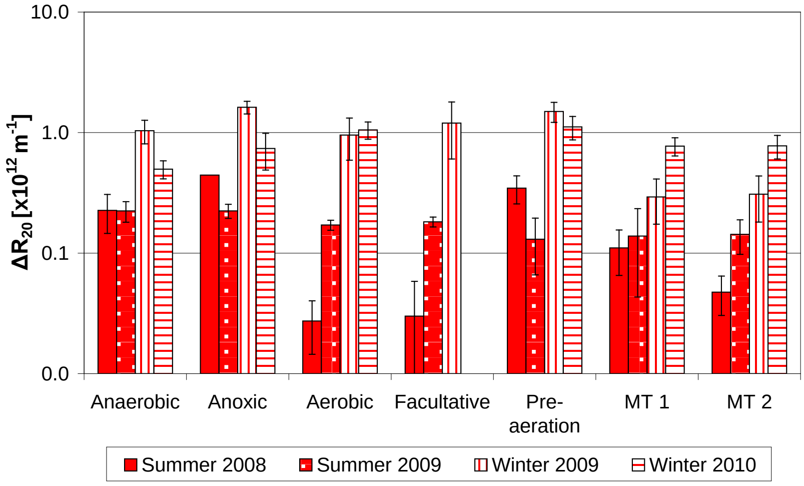

Figure 4.7: Summary of filterability measurements at MBR Heenvliet

Summarising the results from four investigated periods one can observe that filterability of activated sludge differs between summer and winter periods. The results of the DFCm are corrected for temperature and for viscosity changes due to temperature. Therefore, observed seasonal fluctuations exceed the temperature effect on viscosity. The seasonal filterability fluctuations in the membrane tank are presented in Figure 4.8. During summer periods, filterability measured in the membrane tanks was closer to the good/moderate limit and during winter conditions was closer to, or even exceeds, the moderate/poor limit. In addition, clear filterability deterioration can be observed after measurement campaigns conducted in 2009. This is most probably related with the operational change that took place in March 2009 at MBR Heenvliet: the MBR was switched from in series to parallel mode of operation. In a hybrid MBR operated in series, the CAS system precedes the MBR and creates a buffer zone that provides the required time for the microorganisms to adapt to new conditions and consequently more stable conditions for the activated sludge are achieved (Krzeminski et al. 2012c). During parallel operation, the MBR system acts as a separate and stand-alone plant. Therefore, instead of being supplied with the more stable activated sludge from CAS system, the MBR was fed with the raw wastewater with constantly changing parameters. Consequently, activated sludge filterability worsened compared to the earlier period. Further discussion on impact of MBR configuration is presented in *Chapter 7*.

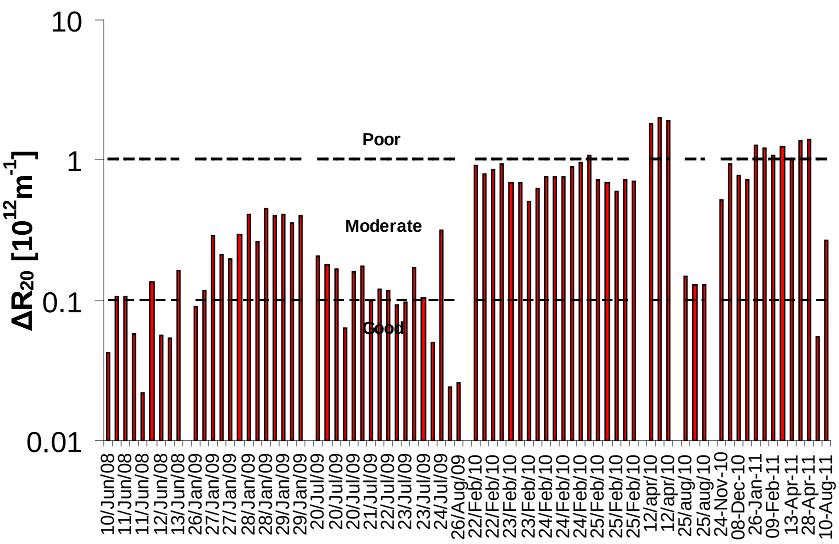

Figure 4.8: Filterability fluctuations in membrane compartments at MBR Heenvliet

The comparison of measurement campaigns clearly shows better filterability of activated sludge, i.e., smaller increment of additional resistance, in the summer than in the winter period. To better visualize the seasonal variations, the representative filtration characterization curve of the membrane tank samples for each experimental campaign is plotted in Figure 4.9.

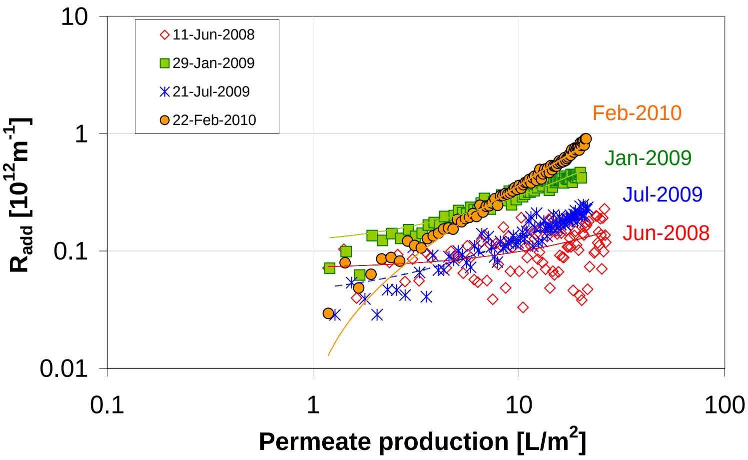

Figure 4.9: Output of DFCm measurements for samples originating from the membrane tanks during four experimental campaigns at MBR Heenvliet – representative filtration characterization curves

The seasonal variations, characterised by a temperature fluctuations, are likely impacting activated sludge filterability. The discussion on the relation between activated sludge filterability and its temperature is presented in section 4.8 and in *Chapter 5*.

4.3.2 Filtration characterisation at MBR Varsseveld

4.3.2.1 Plant information

The wastewater treatment plant of Varsseveld, operated by Rijn and IJssel Waterboard (WRIJ), is the first full-scale plant in the Netherlands that has been retrofitted with the MBR technology for treatment of domestic wastewater. After the upgrade, the treatment plant consists of rigorous pre-treatment, sand and oil trap, oxidation ditch of activated sludge, scum and foam remover, four membrane tanks and sludge thickener (Figure 4.10).

Figure 4.10: Scheme and aerial view of the Varsseveld WWTP (Waterboard Rijn & IJssel)

The incoming water is submitted to a three-step pre-treatment. At first, it passes through the fine bar screens with a bar spacing of 6 mm for fine screening. Then the wastewater enters the aerated sand and fat removal chamber from where the settled sand is transported to a centrifuge to separate it from water that is returned to the micro-sieves. Finally, the partially pre-treated influent flows through 1 mm micro-sieves which separate floating material and organic matter from liquids.

The biological section, designed for biological phosphorous and nitrogen removal, is an oxidation ditch with fine bubble aeration, consisting of anoxic (denitrification zone) and aerobic (nitrification zone) sections (Figure 4.11). After being submitted to several biological treatments, activated sludge is fed to a membrane section for the liquid-solid separation process. The stand-alone MBR Varsseveld has four separate process lines with hollow fibre membranes that are symmetrically loaded via activated sludge supply pumps. The number of membrane compartments in actual operation is based on the activated sludge level in the aeration tank. When the activated sludge level is below 88 % then only one membrane tank is in use. When the level rise up to 90 % then another membrane tank is started and in total two tanks are in use. Finally, when the level exceeds 90 % of the total aeration tank volume, all four tanks are in use.

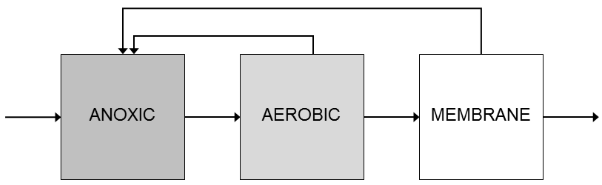

Figure 4.11: Schematic view of the membrane bioreactor system in Varsseveld

Description and operational parameters of the MBR Varsseveld are presented in detail in Table 3.1.

4.3.2.2 Filtration characterisation

Filtration characterisation experiments were practised on-site during four measurement campaigns, two in the summer and two in the winter:

- 23rd – 26th of June 2008,
- 9th – 13th of February 2009,
- 17th – 21st of August 2009, and
- 1st – 5th of March 2010.

Activated sludge was sampled from each compartment of the MBR: anoxic, aerobic and four membrane tanks. All activated sludge samples were submitted to filtration tests. A representative filtration characterization curve of the analysed samples for each experimental campaign is plotted in Figure 4.12. The results, expressed as added filtration resistance, obtained during research periods clearly demonstrate different filtration characterisation behaviour of the activated sludge samples in the summer and in the winter periods.

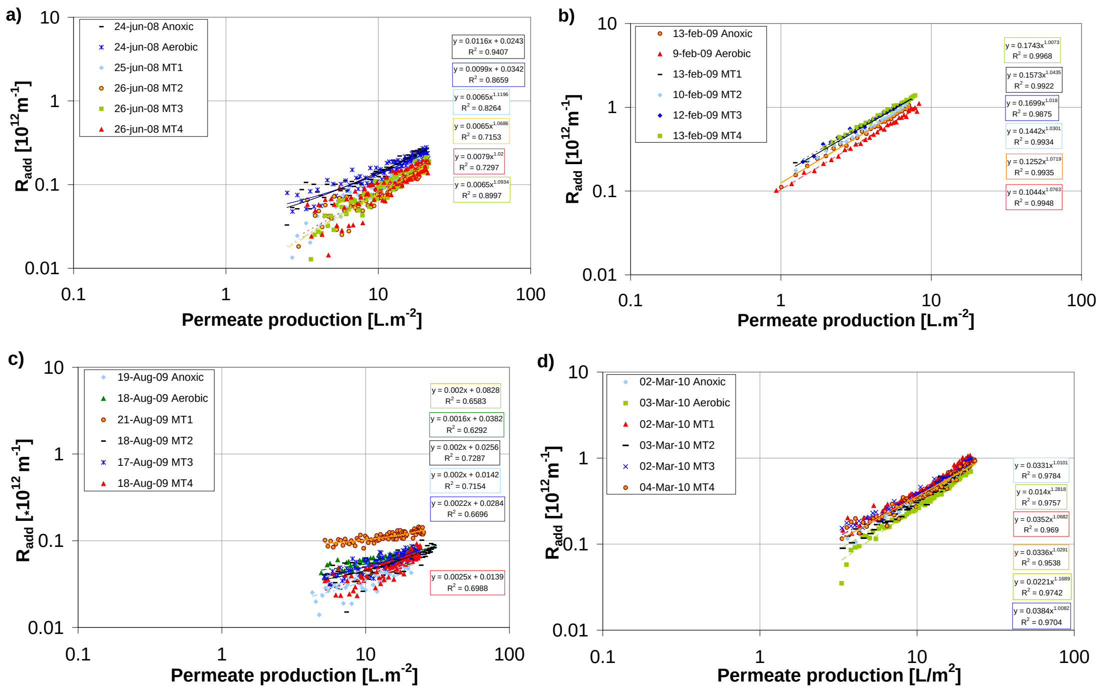

Figure 4.12: Output of DFCm measurements from (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010 measurement campaigns at MBR Varsseveld – representative filtration characterization curves

The filterability results of the samples taken from each tank are presented in Figure 4.13. Filterability vary between the research campaigns and was qualified as good to moderate and as moderate to poor in the summer and in winter periods, respectively.

During the first experimental campaign, conducted in June 2008, the filterability was nearly uniform along the compartments and the sampling days and was within a range of $0.15 \cdot 10^{12}$ m^{-1} and $0.22 \cdot 10^{12}$ m^{-1}. Hence, the filterability of activated sludge could be considered as moderate regardless of the sampling location and time of the sampling. The average ΔR_{20} values of $0.18 \cdot 10^{12}$ m^{-1}, $0.21 \cdot 10^{12}$ m^{-1} and between 0.17-$0.18 \cdot 10^{12}$ m^{-1} were measured in anoxic, aerobic and four of the membrane tanks, respectively.

During the second experimental campaign, carried out in January 2009, the filterability varied between the compartments and the sampling days. The filterability was classified as poor as the measured ΔR_{20} values were within a range of $2.2 \cdot 10^{12}$ m^{-1} and $4.4 \cdot 10^{12}$ m^{-1}. The average ΔR_{20} values measured in aerobic and anoxic sections were $3.0 \cdot 10^{12}$ m^{-1} and $2.7 \cdot 10^{12}$ m^{-1}, respectively. The average ΔR_{20} values of $3.7 \cdot 10^{12}$ m^{-1}, $3.4 \cdot 10^{12}$ m^{-1}, $3.1 \cdot 10^{12}$ m^{-1} and $3.5 \cdot 10^{12}$ m^{-1} were measured for samples originating from the membrane tank #1, #2, #3, and #4 respectively.

During the third experimental period, i.e., July 2009, very similar and stable filterability was measured. The activated sludge filterability can be considered as good as an average ΔR_{20} value of $0.04 \cdot 10^{12}$ m^{-1} was measured in each of the compartments.

During the fourth measurement campaign, performed in February 2010, the filterability varied between $0.53 \cdot 10^{12}$ m^{-1} and $0.96 \cdot 10^{12}$ m^{-1} depending on the compartment that a sample was taken from. Although the filterability was relatively stable a slight deterioration was observed during the campaign. The average ΔR_{20} values of $0.68 \cdot 10^{12}$ m^{-1}, $0.60 \cdot 10^{12}$ m^{-1}, $0.74 \cdot 10^{12}$ m^{-1}, $0.75 \cdot 10^{12}$ m^{-1}, $0.76 \cdot 10^{12}$ m^{-1} and $0.76 \cdot 10^{12}$ m^{-1} were measured in the anoxic, aerobic, and one to four membrane tanks, respectively. Therefore the filterability was classified as moderate for all of the samples.

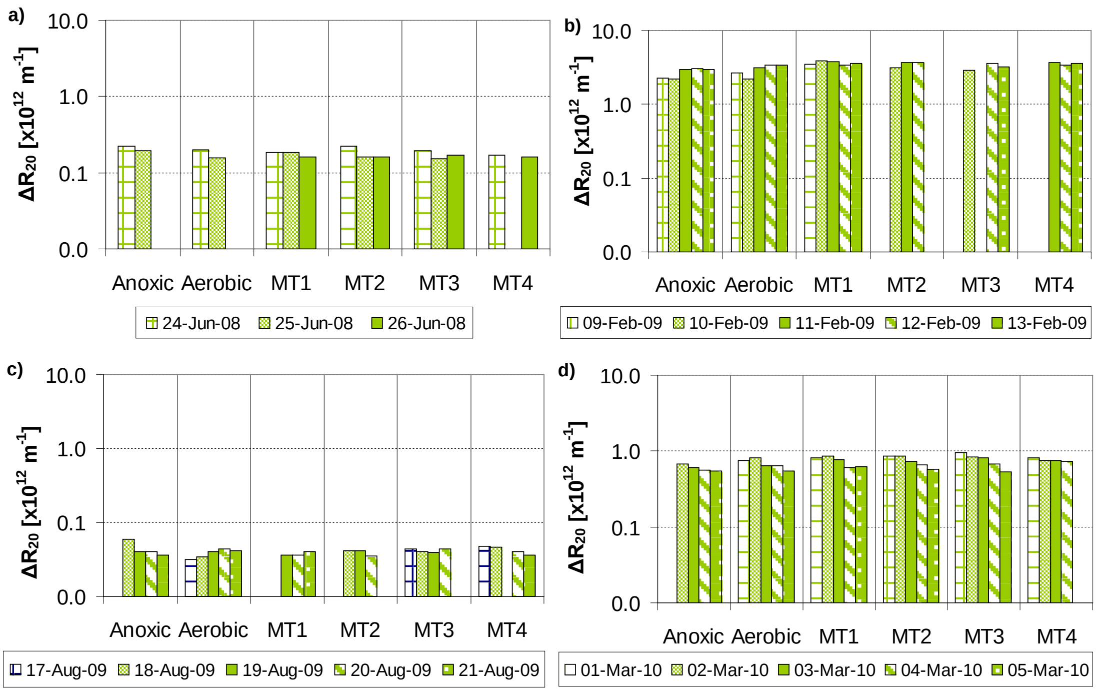

Figure 4.13: Filterability evolution along MBR process line during experimental periods at MBR Varsseveld: (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010

Activated sludge filtration characteristics were also analysed to monitor filterability development along the MBR and to identify potential differences between the investigated compartments. For the MBR Varsseveld, no clear dependency between the filterability rate and the sampling point, and thus the process flow, along the MBR could be found (Figure 4.14). Especially in the summer period, activated sludge filterability was stable and very similar along the whole MBR. Nonetheless, a small improvement in activated sludge filterability along the process line was observed during summer. Contrary, in winter period, filterability changes were more visible along the MBR as the quality of the sludge deteriorated with the process flow to be the worst in the membrane tanks. The development of filterability in MBR compartments is further discussed in section 4.6.

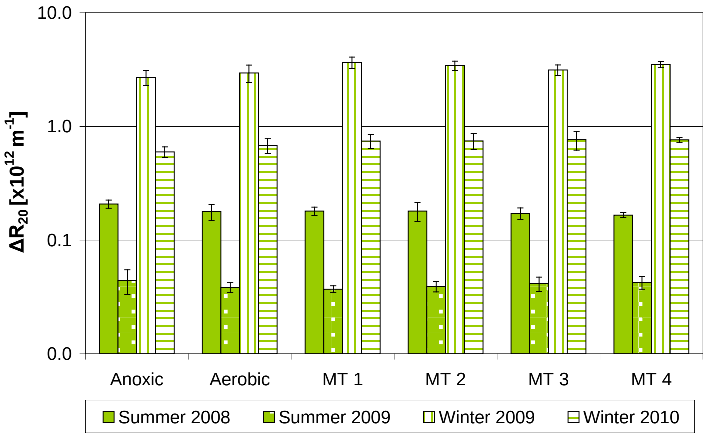

Figure 4.14: Summary of filterability measurements at MBR Varsseveld

Summarising the results from four investigated periods one can observe that filterability of activated sludge differs between summer and winter periods. The seasonal filterability fluctuations in the membrane tank are presented in Figure 4.15. During summer periods, filterability measured in the membrane tanks was classified as good or moderate and during winter conditions as moderate or poor. During both summer and winter campaigns, lower ΔR_{20} values were measured during the second campaign, i.e., ‘summer 2009’ and ‘winter 2010’, respectively. The improvement between the same seasons can be probably explained by the higher sludge temperatures during the improved filterability periods: 22.0°C vs. 23.4°C in the ‘summer 2009’ and 9.7°C vs. 11.7°C in the ‘winter 2010’. On the other hand, it could indicate kind of sludge development and an ongoing filterability improvement. In addition, clear filterability differences can be observed between the same seasons. Apparently, besides the seasonal fluctuations, other factors like characteristic of the incoming influent have influenced the activated sludge filterability (see section 6.3). For example, in winter 2009 filterability quality was qualified as poor while in winter 2010 filterability was assessed to be moderate. This is most probably related with the drastic temperature change that occurred in winter 2009 at MBR Varsseveld (Figure 9, section 6.3). Further discussion on the impact of temperature shock and influent characteristic on filterability is presented in *Chapter 6*.

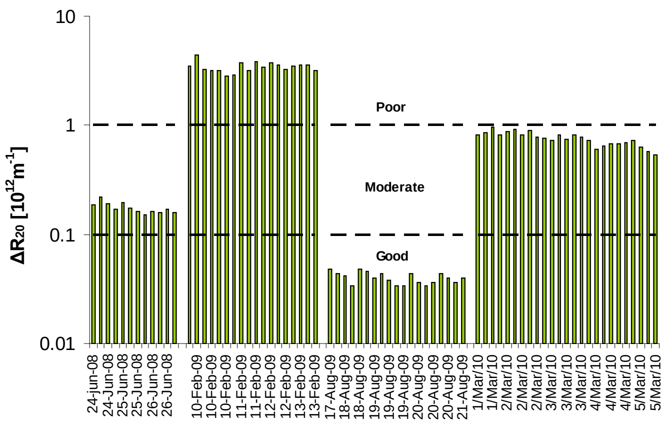

Figure 4.15: Filterability fluctuations in membrane compartments at MBR Varsseveld

The comparison of measurement campaigns clearly shows better filterability of activated sludge, i.e., smaller increment of additional resistance, in the summer than in the winter period. To better visualize the seasonal variations, the representative filtration characterization curve of the membrane tank samples for each experimental campaign is plotted in Figure 4.16.

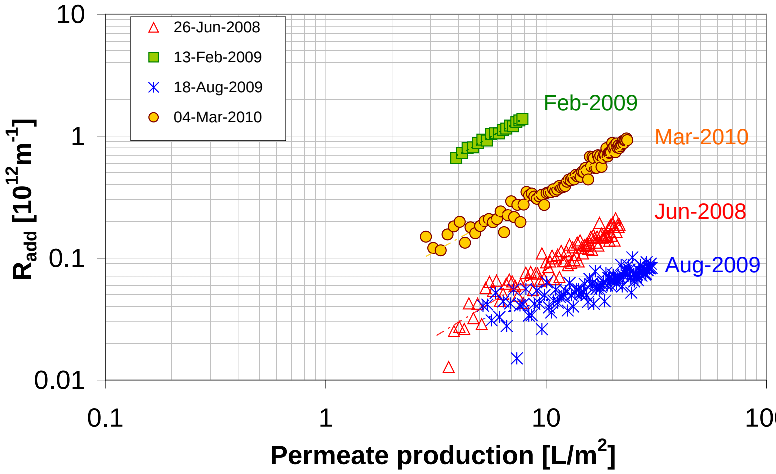

Figure 4.16: Output of DFCm measurements for samples originating from the membrane tanks during four measurement campaigns at MBR Varsseveld – representative filtration characterization curves

The seasonal variations, characterised by temperature fluctuations, are likely impacting activated sludge filterability and significantly contribute to filterability differences that can be observed between the measurement campaigns. The discussion on the relation between activated sludge filterability and temperature is presented in section 4.8 and in *Chapter 5*.

4.3.3 Filtration characterisation at MBR Ootmarsum

4.3.3.1 Plant information

The wastewater treatment plant of Ootmarsum, operated by Regge and Dinkel Waterboard (WRD), has been renovated in 2007 into a so-called hybrid MBR where a conventional activated sludge system is operated parallel to a membrane bioreactor system. In order to prevent system overloading and overflows during heavy rainfall, the local rainwater collection system was disconnected from the plant. Thus, the hydraulic capacity of the total plant remained the same, i.e., 650 $m^3.h^{-1}$. After the upgrade, the hybrid treatment plant consists of an MBR alongside the conventional activated sludge system (Figure 4.17). During the dry weather conditions half of the flow (up to 150 $m^3.h^{-1}$) is treated in the membrane bioreactor. During the rainy periods, excess rainwater is treated in the conventional activated sludge system (Futselaar et al. 2007).

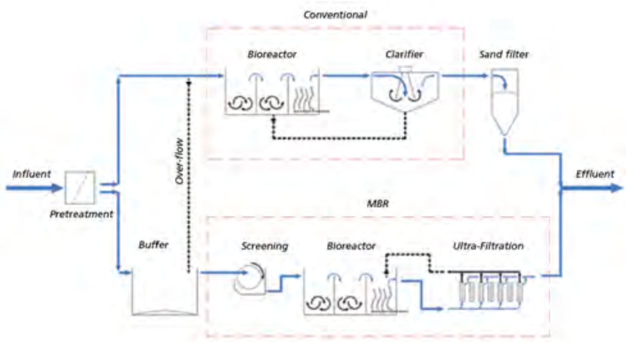

Figure 4.17: Scheme (Norit) and aerial view of the Ootmarsum WWTP (Grontmij)

The incoming water is submitted to a two-step pre-treatment. At first it passes through the fine bar screens with a bar spacing of 6 mm and then through a hydro cyclone sand removal. After that, the inflow is equally divided between the conventional system and the membrane bioreactor. The part treated by the MBR is firstly directed to the buffer tank from where it is conveyed to the MBR in controlled quantities. The buffer tank serves also as a primary settling tank. Hence, if the supply of sewer is too large, like in case of prolonged rain, the clarified water overflows and it is directed to the conventional part of the WWTP. The settled and concentrated wastewater from the bottom is pumped to an additional drum sieve pre-treatment of the MBR. The peak flows during rain weather are split hydraulically and biologically.

The membrane bioreactor system composes of a 0.75 mm fine-screen pre-treatment, a biological section with sub-compartments, and membrane lines (Figure 4.18).

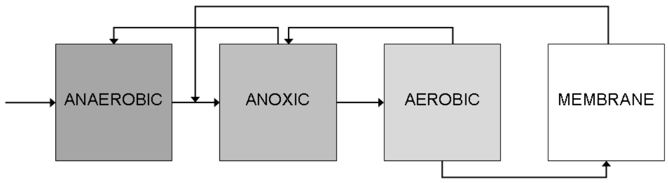

Figure 4.18: Schematic view of the membrane bioreactor system in Ootmarsum

The biological sub-compartments are, in accordance with the flow, as follows: anaerobic tank for selection of phosphate accumulating organisms (PAO), selector for filamentous microbe control, anoxic tank for denitrification, aerobic tank for nitrification and carbon removal. After being submitted to several biological treatments, the activated sludge is fed to external multi tube membranes for the liquid-solid separation process. The membranes are placed in six separate lines, equipped with symmetrical loading via activated sludge supply pipes. Activated sludge is internally re-circulated from the membranes – via the aerobic tank – to the selector tank, from the aerobic tank to anoxic and from anoxic to the anaerobic. Finally, before discharge, the MBR permeate and CAS effluent are mixed together and directed to an ecological filter in order to ‘ecologically activate’ the effluent after the purification process. This system is an example of application of a ‘water harmonica’ concept, i.e., natural and ecological link between treated wastewater and receiving surface water (van den Boomen et al. 2012). In this so-called ‘biozone’ water passes through a system of water bodies and reed zones (HRT of about 4 days) in which various types of plants and small organisms are present.
Description and operational parameters of the MBR Ootmarsum are presented in detail in Table 3.1.

4.3.3.2 Filtration characterisation

Filtration characterisation experiments were practised on-site during four measurement campaigns, two in the summer and two in the winter:

- 16th – 19th of June 2008,
- 2nd – 6th of February 2009,
- 10th – 14th of August 2009, and
- 22nd – 25th of March 2010.

Activated sludge was sampled from each compartment of the MBR: anaerobic, anoxic, aerobic, and from the membrane lanes. The membrane samples were collected at the beginning of the membrane lines just before entering membrane modules. All activated sludge samples were submitted to filtration tests. A representative filtration characterization curve of the analysed samples for each experimental campaign is plotted in Figure 4.19. The results, expressed as added filtration resistance, obtained during research periods clearly demonstrate different filtration characterisation behaviour of the activated sludge samples in the summer and in the winter periods.

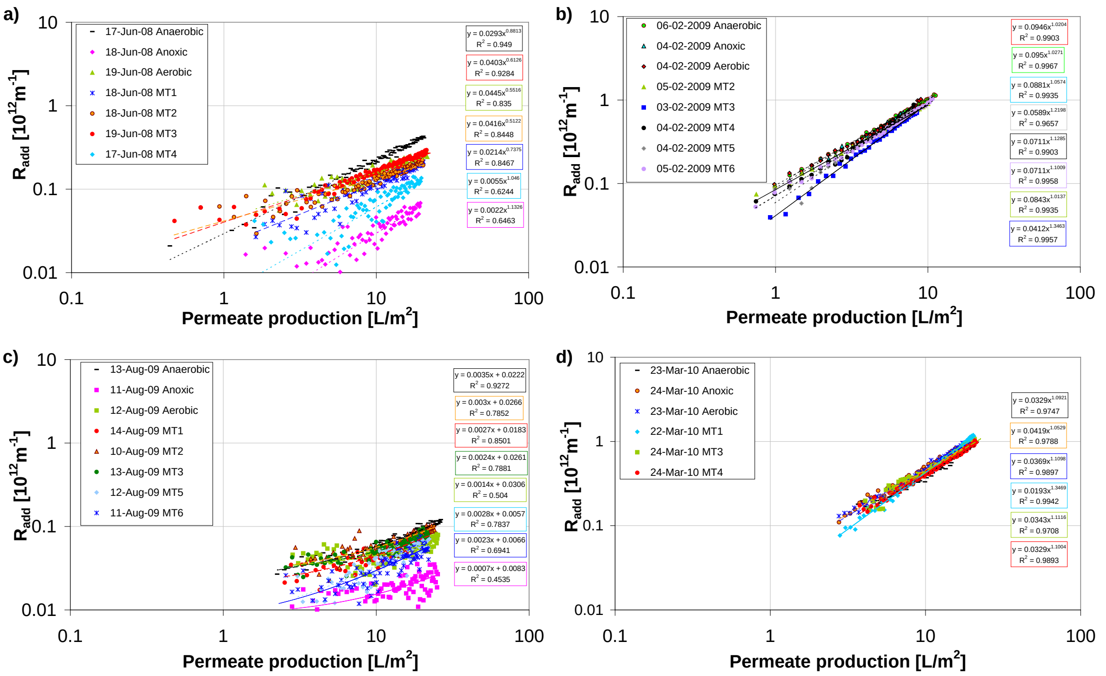

Figure 4.19: Output of DFCm measurements from (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010 measurement campaigns at MBR Ootmarsum – representative filtration characterization curves

The filterability results of the samples taken from each MBR section are presented in Figure 4.20. Activated sludge filterability varies between the measurement campaigns and thus between the seasons. Filterability was qualified as mainly moderate during the first summer period, as good during the second summer period, as poor during the first winter period and, finally, as moderate during the second winter period (Figure 4.21).

During the first experimental campaign, conducted in June 2008, the filterability varied between the compartments and the sampling days and was within a range of $0.06 \cdot 10^{12}$ m^{-1} and $0.50 \cdot 10^{12}$ m^{-1}. Hence, the filterability of activated sludge could be considered as good to moderate depending on the sampling location and time of the sampling. The filterability was moderate in anaerobic and aerobic with average ΔR_{20} values of $0.43 \cdot 10^{12}$ m^{-1} and $0.17 \cdot 10^{12}$ m^{-1}, respectively. Also samples originating from the membrane lines #1, #2, #3 and #4 were classified as moderate with average ΔR_{20} values of $0.18 \cdot 10^{12}$ m^{-1}, $0.18 \cdot 10^{12}$ m^{-1}, $0.19 \cdot 10^{12}$ m^{-1} and $0.14 \cdot 10^{12}$ m^{-1}, respectively. Good sludge filterability was measured in anoxic compartment with average ΔR_{20} values of $0.06 \cdot 10^{12}$ m^{-1}. However, only one sample coming from anoxic tank was measured during this campaign. Furthermore, clear deterioration of activated sludge filterability was observed along the whole campaign as the ΔR_{20} values increased on a daily basis.

Also during the second experimental campaign, carried out in January 2009, the filterability varied between the compartments and the sampling days. The filterability was classified as poor regardless of sampling location as the measured ΔR_{20} values were between $1.6 \cdot 10^{12}$ m^{-1} and $3.2 \cdot 10^{12}$ m^{-1}. The average ΔR_{20} values of $1.9 \cdot 10^{12}$ m^{-1}, $2.1 \cdot 10^{12}$ m^{-1} and $2.1 \cdot 10^{12}$ m^{-1} were calculated for samples from anaerobic, anoxic and aerobic compartments, respectively. The average ΔR_{20} values for the samples coming from membrane lane #2, #3, #4, #5 and #6 were $1.9 \cdot 10^{12}$ m^{-1}, $2.4 \cdot 10^{12}$ m^{-1}, $2.2 \cdot 10^{12}$ m^{-1}, $2.5 \cdot 10^{12}$ m^{-1} and $2.0 \cdot 10^{12}$ m^{-1}, respectively.

During the third experimental period, executed in July 2009, the filterability was similar and stable with ΔR_{20} values between $0.01 \cdot 10^{12}$ m^{-1} and $0.08 \cdot 10^{12}$ m^{-1}. The activated sludge filterability can be considered good in each of the compartments. The average ΔR_{20} values of $0.08 \cdot 10^{12}$ m^{-1}, $0.02 \cdot 10^{12}$ m^{-1}, $0.03 \cdot 10^{12}$ m^{-1} were determined in the anaerobic, anoxic and aerobic compartments, respectively. The membrane samples were collected from lines #1, #2, #3, #5 and #6 with the average ΔR_{20} values of $0.04 \cdot 10^{12}$ m^{-1}, $0.06 \cdot 10^{12}$ m^{-1}, $0.03 \cdot 10^{12}$ m^{-1}, $0.05 \cdot 10^{12}$ m^{-1} and $0.04 \cdot 10^{12}$ m^{-1}, for the respective membrane lane.

During the fourth measurement campaign, performed in February 2010, the filterability varied between $0.7 \cdot 10^{12}$ m^{-1} and $1.3 \cdot 10^{12}$ m^{-1} depending on the compartment that a sample was taken from. Although the filterability was relatively stable, in general, a slight improvement was observed during the campaign. The average ΔR_{20} of $0.8 \cdot 10^{12}$ m^{-1}, $1.0 \cdot 10^{12}$ m^{-1} and $0.9 \cdot 10^{12}$ m^{-1} were measured in the anaerobic, anoxic and aerobic compartments, respectively. The membrane samples were collected from lines #1, #3 and #4 with the respective average ΔR_{20} values of $0.99 \cdot 10^{12}$ m^{-1}, $0.92 \cdot 10^{12}$ m^{-1} and $0.92 \cdot 10^{12}$ m^{-1}. Therefore, filterability was classified as moderate in case of samples originating from anaerobic, aerobic and all three membrane lanes that were investigated during this campaign. Poor classification was only assigned to samples from anoxic section of the MBR.

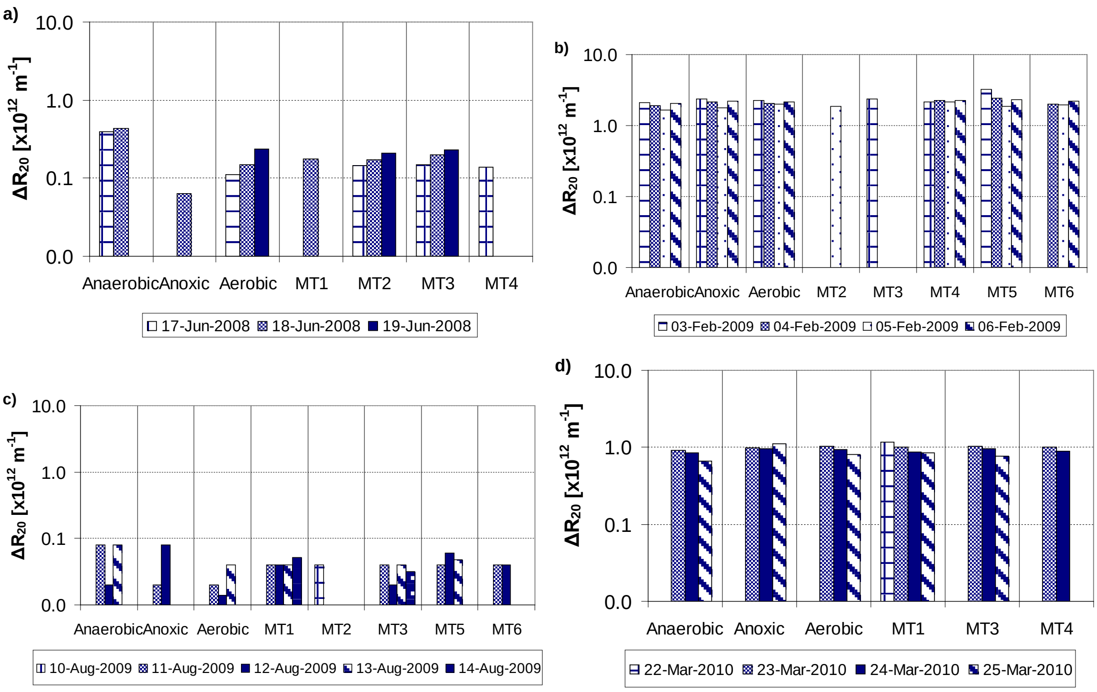

Figure 4.20: Filterability evolution along MBR process line during experimental periods at MBR Ootmarsum: (a) summer 2008, (b) winter 2009, (c) summer 2009 and (d) winter 2010

Monitoring of filtration characteristics demonstrated also clear differences between the investigated compartments (Figure 4.21). It was generally observed for the summer experiments that the best filterability was measured in anoxic tank and the worst in the anaerobic tank. Results from the first winter period were ambiguous: both, best and worst filterability was measured in the membrane lines. Furthermore, during the second winter campaign in 2010, the best filterability was measured in the anaerobic compartment and the worst in the anoxic compartment. This is the opposite of the first summer result. Furthermore, a clear pattern of filterability development along the MBR process flow was not observed. However, during the winter 2009 campaign, a small deterioration of activated sludge filterability along the process line was observed. The development of filterability in MBR compartments is further discussed in section 4.6.

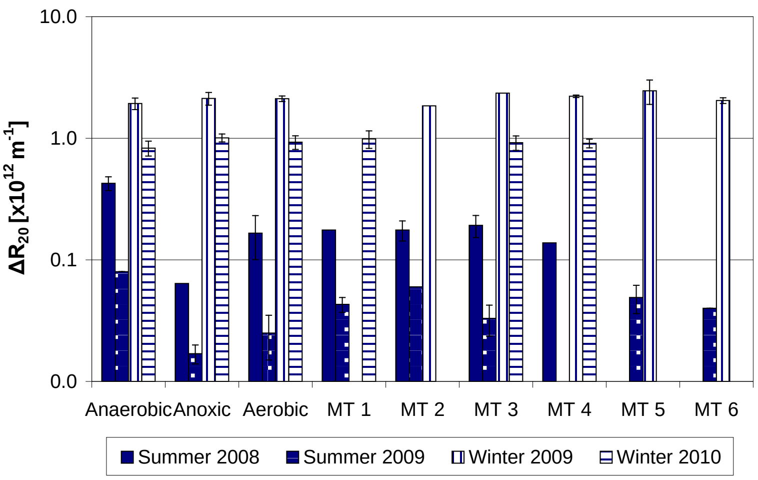

Figure 4.21: Summary of filterability measurements at MBR Ootmarsum

Summarising the results from four investigated periods one can observe that the filterability of activated sludge differs between summer and winter periods. The seasonal filterability fluctuations in the membrane tank are presented in Figure 4.22. During summer periods, filterability measured in the membrane tanks was classified as good or moderate and during winter conditions as moderate or poor. Likewise in case of MBR Varsseveld, lower ΔR_{20} values were measured during the second measurement campaigns both in summer and winter campaigns, i.e., ‘summer 2009’ and ‘winter 2010’, respectively. Also in this case, the improvement between the same seasons can be probably explained by the higher sludge temperatures during the improved filterability periods: 18.4°C vs. 21.1°C in the ‘summer 2009’ and 8.1°C vs. 11.0°C in the ‘winter 2010’. On the other hand, it could indicate kind of sludge development and an ongoing filterability improvement. In addition, certain differences in filterability can be observed between the same seasons itself. Apparently, besides the seasonal fluctuations, other factors like incoming influent composition have influenced the activated sludge filterability (see section 6.3). For example, in winter 2009, the filterability was qualified as poor, whereas in winter 2010 the filterability was assessed as moderate. The explanation of poor filtration behaviour is most probably related to abnormal chemical composition of the incoming wastewater (Figure 13, section 6.3). Further discussion on impact of wastewater composition and toxicity effects on filterability is presented in *Chapter 6*.

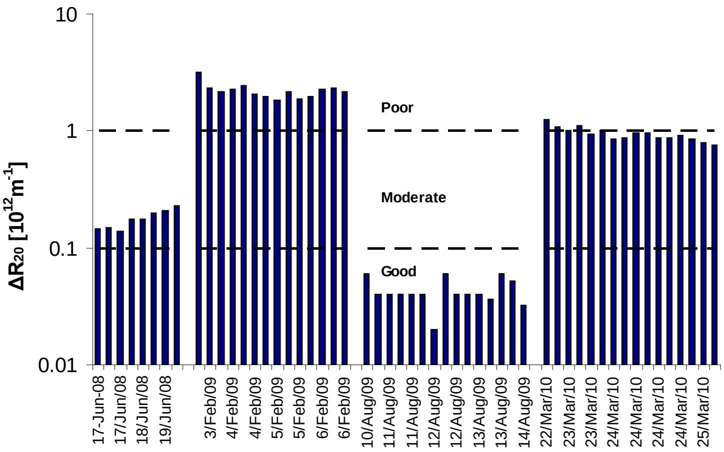

Figure 4.22: Filterability fluctuations in membrane compartments at MBR Ootmarsum

The comparison of measurement campaigns clearly shows better filterability of activated sludge, i.e., smaller increment of additional resistance, in the summer than in the winter period. To better visualize the seasonal variations, the representative filtration characterization curve of the membrane tank samples for each experimental campaign is plotted in Figure 4.23.

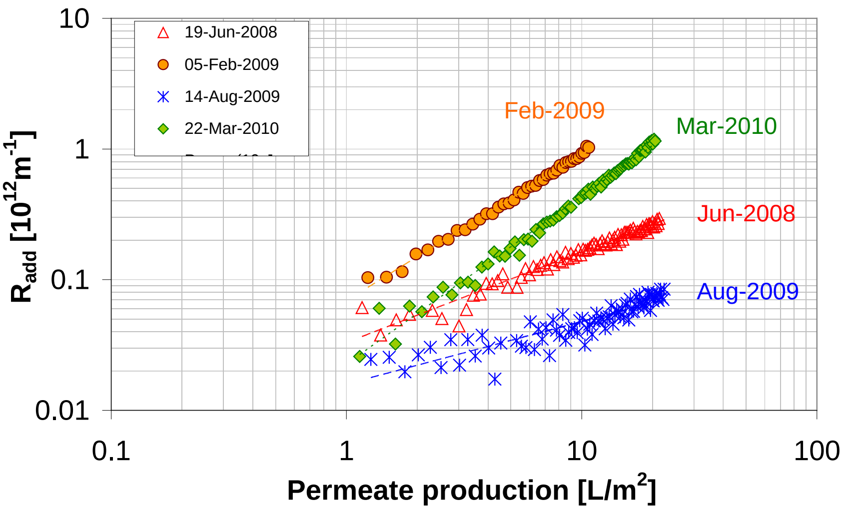

Figure 4.23: Output of DFCm measurements for samples originating from the membrane tanks during four measurement campaigns at MBR Ootmarsum – representative filtration characterization curves

The seasonal variations, characterised by a temperature fluctuations, are likely impacting activated sludge filterability and contribute to changes in filterability between the measurement campaigns. The discussion on the relation between activated sludge filterability and its temperature is presented in section 4.8 and in *Chapter 5*.

4.4 Specific cake resistance and compressibility

The DFCm output, besides the ΔR_{20} value, provides information about the membrane cake layer build up as previously mentioned in *Chapter 3*. The compressibility coefficient ‘s’ expresses the compressibility potential of the cake layer. The product of the specific cake layer resistance, $\alpha_R \cdot c_i$, provides information about substances accumulating in the cake layer and the specific cake resistance caused by these substances. The results of the $\alpha_R \cdot c_i$ and s are presented in Figure 4.24 and Figure 4.25. The $\alpha_R \cdot c_i$ results coming from the four experimental campaigns carried out at three MBR plants are plotted with the respective filterability results in Figure 4.24.

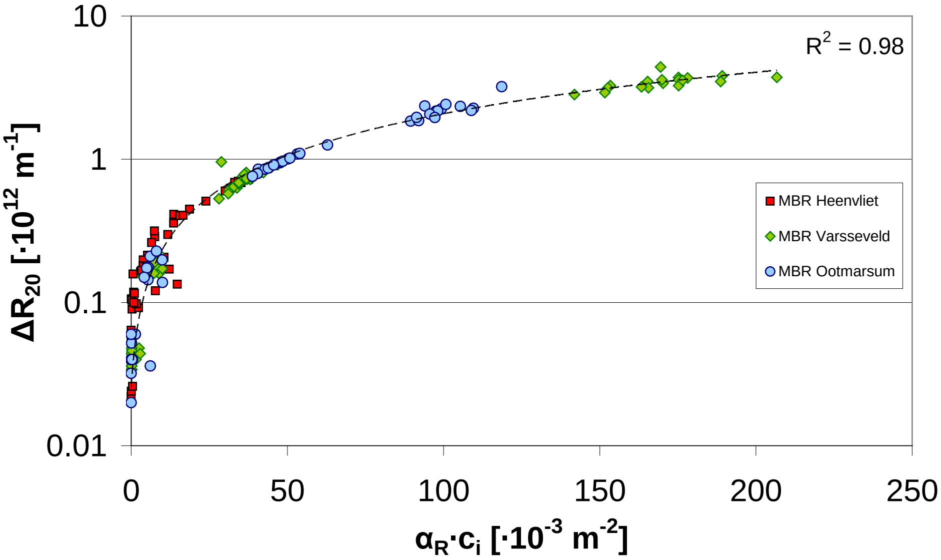

Figure 4.24: Filterability vs. mass involved in the cake layer build up

Likewise in case of the predecessors (Geilvoet 2010, Moreau 2010, Lousada-Ferreira 2011), strong relation is observed between the filterability and the mass involved in the cake layer build up. It is due to the fact that in MBRs and in the DFCm experiment in particular, the cake layer formation is the main fouling mechanism. Therefore, the total cake layer resistance is predominantly determined by the concentration of substances accumulating in the cake layer (Geilvoet 2010). In other words, ΔR_{20} gives an indication on the amount of (submicron) particles involved in the membrane fouling. Furthermore, the difference between the summer and winter seasons is clear. The average $\alpha_R \cdot c_i$ during summer campaigns was $3 \pm 4 \cdot 10^{-3}$ m^{-2} compared to average $67 \pm 54 \cdot 10^{-3}$ m^{-2} during winter campaigns.

Contrary to $\alpha_R \cdot c_i$ product, the compressibility shows no relation with the activated sludge filterability (Figure 4.25). In the majority of the cases, when a well filterable sludge and ΔR_{20} values below $0.7 \cdot 10^{12}$ m^{-1} were measured, the compressibility coefficient was equal to 0. In other words, activated sludge with a good or, to some extent, moderate filterability most probably creates an almost incompressible cake layer at the membrane surface. In contrast, when ΔR_{20} values were above $0.7 \cdot 10^{12}$ m^{-1} the compressibility coefficient was mostly above

0. Therefore, poor filterable activated sludge is likely to form a compressible cake layer as observed by Moreau (2010). Furthermore, as the s coefficient results are within 0.0-0.3 range, the cake layer can be considered to be loose and not compressible. This is in accordance with the findings of the previous researchers who reported the s coefficient to be primarily in the range of 0.0-0.3 (Geilvoet 2010, Moreau 2010, Lousada-Ferreira 2011).

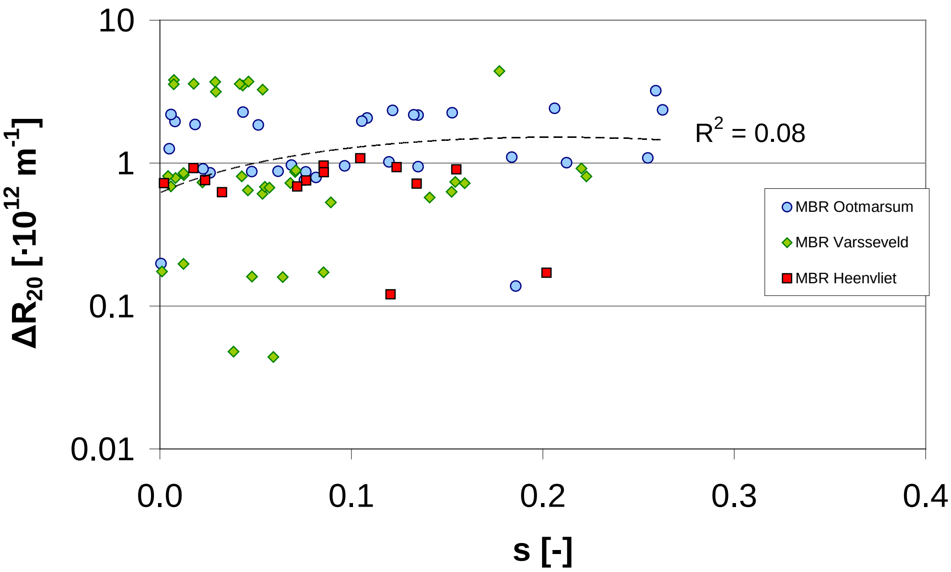

Figure 4.25: Filterability vs. compressibility of the cake layer

4.5 Activated sludge composition monitoring

The differences between the activated sludge samples are generally more visible in a winter period than during a summer period. Hence, the image analysis was performed during one of the winter campaigns (February–March 2010) in order to further investigate the differences between the activated sludge samples. Nevertheless, the municipal MBRs had a very similar type of activated sludge morphology, as confirmed by the microscopic activated sludge images (Figure 4.26) and by the filtration characterisation tests performed on samples originating from the membrane tanks. Moreover, the observed similarities were coherent with the results of the diluted SVI test. The DSVI of the same activated sludge samples was in the range of 155 mL/g and 157 mL/g for Heenvliet and Ootmarsum samples, respectively.

In order to create a good activated sludge floc, a filamentous bacteria and a floc former need to be present. The presence of filamentous microorganism is necessary to form a stable floc. In the absence or deficiency of filamentous microorganisms, small, compact and weak floc – called a pin floc – will form, resulting in poor settleability, poor filterability and turbid effluent (Meng et al. 2006, Bitton 2011). If the filaments are present in excess, they interfere with sludge settling, compaction and concentration of solids. They create bridging lattice, which prevents floc particles to agglomerate and may form thick and stable foam. This foam disturbs the biological process and consequently can lead to a decrease in the filterability (Gil et al. 2011). In this study, slightly more filamentous bacteria were observed in the samples collected from MBR Heenvliet compared to the samples from the other two MBR plants. In consequence flocs are likely to be stronger in the sludge samples from MBR Heenvliet. However, filamentous bulking sludge, i.e., excessive growth of filamentous bacteria (Martins et al. 2004), was not observed in any of the case.

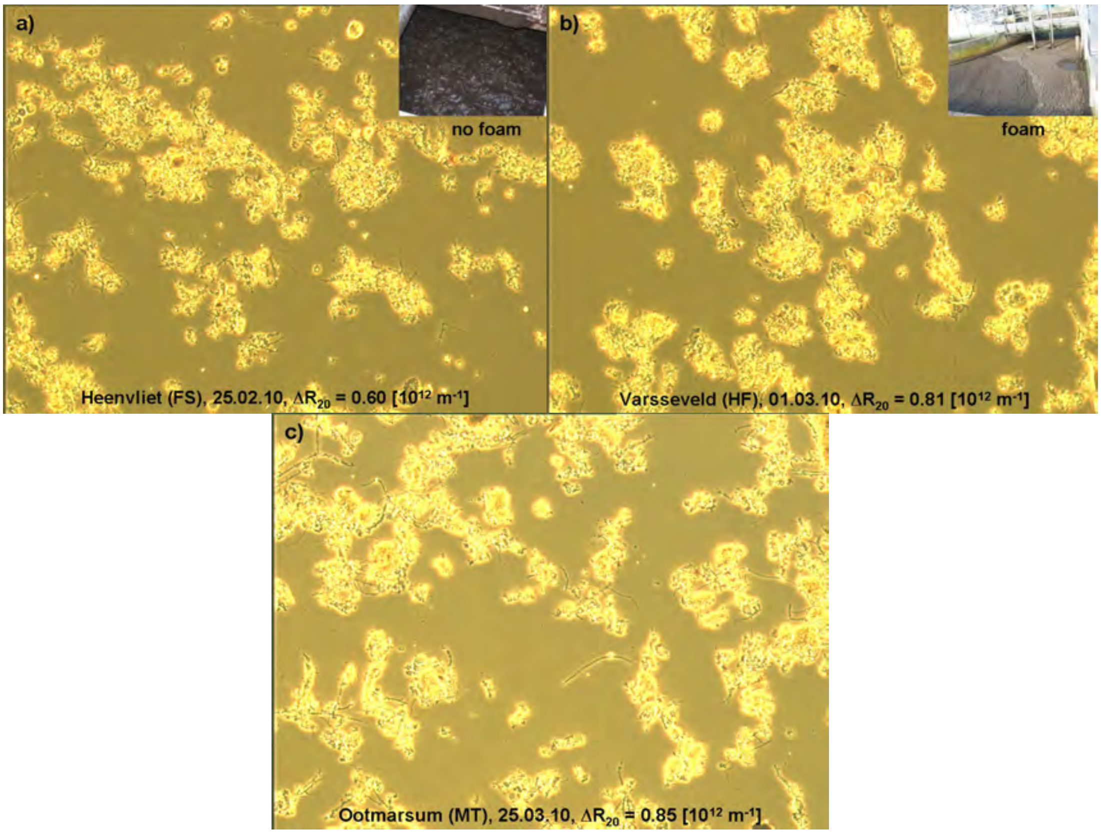

Figure 4.26: Microscopic images (x100, Ph1) of activated sludge from the membrane compartments of: (a) MBR Heenvliet, (b) MBR Varsseveld and (c) MBR Ootmarsum

Thereafter, the relation between the activated sludge filterability and the presence of filamentous bacteria was further investigated (Figure 4.27). The relative abundance of filamentous organisms was quantified according to the rating scale described by Jenkins et al. (2003): none (0), few (1), some (2), common (3), very common (4), abundant (5), and excessive (6). All analyzed samples during the 2010 winter period were classified as common.

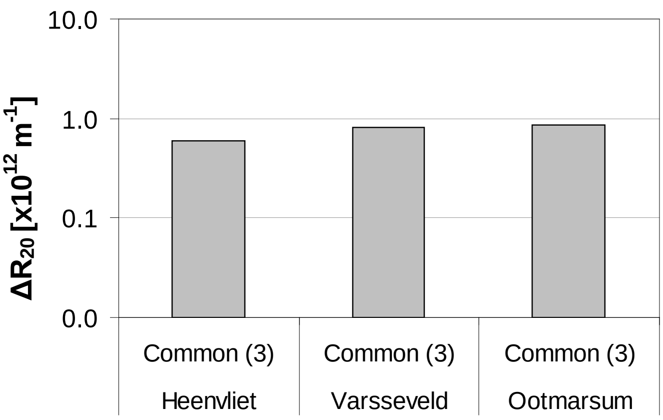

Figure 4.27: Activated sludge filterability in relation to the presence of filamentous bacteria

Furthermore, similar composition of the activated sludge was observed for samples that originate from the aerobic and the four different membrane compartments of the MBR Varsseveld (Figure 4.28). This similar activated sludge composition is in coherence with the uniform filterability measured along the MBR in Varsseveld.

Similar type of morphology and comparable filterability results of the activated sludge samples originating from different MBRs, during the winter period, does not exclude sludge differences between the seasons. Especially as significant differences in activated sludge filterability, expressed by the ΔR_{20} parameter, were observed between the seasons (section 4.7). Therefore, the activated sludge morphology might be different between the MBR plants, in other periods.

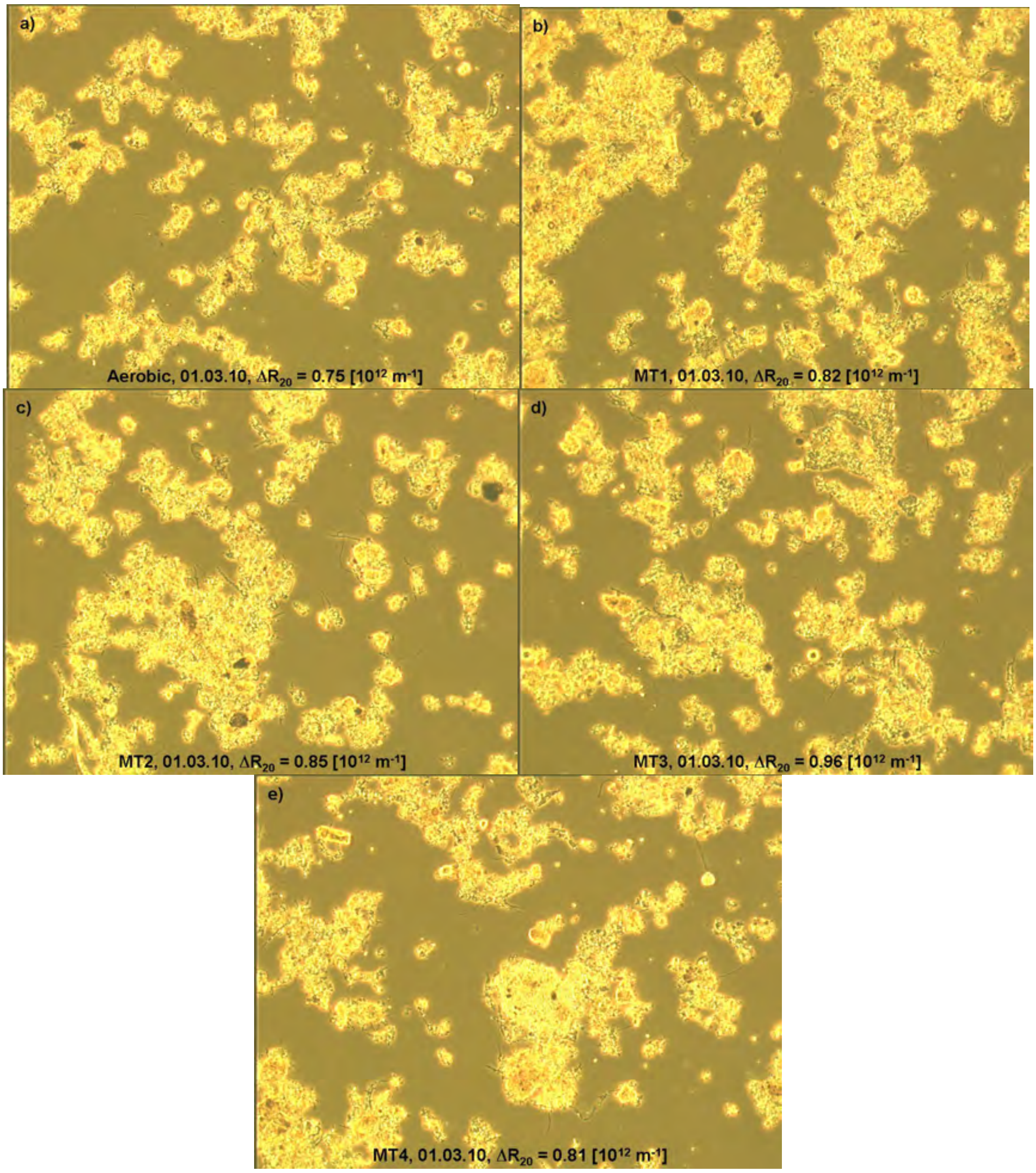

Figure 4.28: Microscopic images of activated sludge from the (a) aerobic, (b) 1st membrane, (c) 2nd membrane, (d) 3rd membrane and (e) 4th membrane compartment of MBR Varsseveld

A more thorough analysis aiming at sample determination is difficult when it is based only on the undiluted activated sludge images. Analysis of the diluted sludge sample is necessary if more detailed information on sludge flocculation state is required. Diluting the sludge sample is necessary to enable distinction between neighboring flocs, not possible at high sludge concentrations due to contact between the sludge flocs, without the risk of an overestimation of object sizes. Furthermore, dilution allows to work at a normalized sludge concentration enabling a comparison of sludge samples with different biomass concentrations (Van den Broeck et al. 2011).

4.6 Development of filterability in MBR compartments

In order to study the influence of operational processes on activated sludge filterability the development of filterability in MBR compartments was monitored. The effect of different compartments and the order of the tanks on activated sludge filterability were studied. This analysis aimed to improve the understanding on how to promote good filterability in the membrane sections. The activated sludge samples were collected from all available compartments of the investigated MBRs. In this way, the potential differences between the investigated compartments could be identified and the impact of the biological process occurring along the process flow in the MBR could be assessed.

Each MBR comprises of various compartments where biological treatment and membrane separation takes place. The order of the compartments agrees with its functionality in the treatment, i.e., removal of phosphorous, nitrogen and carbon. Thus, keeping in mind that different MBRs can have their own particular design, the order of the tanks can be different too. The order of the tanks in each location is presented schematically for MBR in Heenvliet, Varsseveld and Ootmarsum in Figure 4.4, Figure 4.11 and Figure 4.18, respectively. Typically, the anaerobic compartment for conversion of carbon and PAO selection is followed by the anoxic section where denitrification process takes place. In the subsequent – aerobic – tank a removal of organic carbon and nitrification process is carried out. The membrane section, last compartment of the MBR, is designed for a solids-liquid separation. Therefore, good filterability in the membrane section is of major importance for good filtration performance and overall MBR performance. In certain cases, the functionality of some tanks, e.g. the facultative tank of MBR Heenvliet, can be manipulated depending on the actual nutrient load in the tank and actual oxygen demand.

According to Lousada-Ferreira (2011) two types of filterability trends between the tanks are typically observed: homogeneous or heterogeneous. Homogeneous filterability is when sludge filterability, expressed by ΔR_{20} parameter, is uniform in all MBR tanks. In other words, similar filterability is measured irrespectively to the sludge collection point. The heterogeneous filterability describes the situation when activated sludge filterability is not uniform and ΔR_{20} results vary between the MBR compartments.

Moreau (2010) proposed and Lousada-Ferreira (2011) proved, that high recirculation rates between the membrane tanks and the aerobic tank provides good homogenisation of the activated sludge in the process. Moreover, reduced permeate flux was reported at MBRs operated with high return ratios (Wisniewski and Grasmick 1998). In addition, contradicting results are published in regard to impact of return ratios on the floc sizes. Some researchers observed floc breakage at high return ratios (Wisniewski and Grasmick 1998) and some observed no significant changes in the floc sizes (Lousada-Ferreira 2011). In contrast, heterogeneous filterability occurs when the return rate is low (Lousada-Ferreira 2011). At low return rates and when the MLSS concentration in the membrane tanks is above the critical MLSS concentration of 10.5 g/L, sludge filterability improves. It is likely due to a potential retention of fouling particles in activated sludge matrix (Lousada-Ferreira 2011), which in turn results in better sludge filterability and improved filtration conditions in the membrane tank. Therefore, according to Lousada-Ferreira (2011), low return rates are preferable to promote good filterability in the separate membrane tank and thus for MBR operation.

Monitoring of filtration characteristics demonstrated clear differences between the MBRs in regard to filterability development in the investigated compartments (Figure 4.29). Mainly heterogeneous filterability between the compartments was found in case of MBR Heenvliet, whereas mostly homogenous filterability in all tanks was observed in MBR Varsseveld and MBR Ootmarsum. Results of filterability development in MBR compartments from four experimental campaigns, two in summer and two in winter, are presented in Figure 4.29a-b and in Figure 4.29c-d.

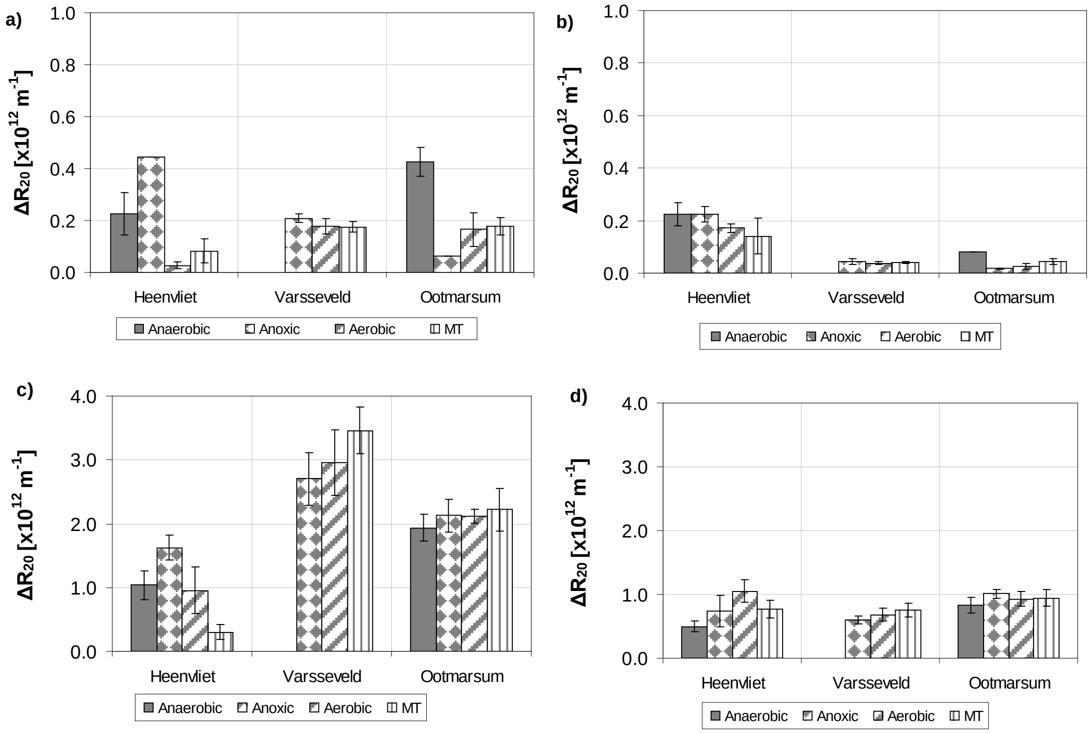

Figure 4.29: Summary of filterability measurements in different MBR compartments: a) in summer 2008, b) in summer 2009, c) in winter 2009, d) in winter 2010. Note different ΔR_{20} scale for summer and winter campaigns

A heterogeneous filterability was found in case of MBR Heenvliet as activated sludge filterability clearly demonstrated differences between the investigated compartments. The recirculation rates were 2.6, 4.7, 1.8 and 3.4 during the summer 2008, winter 2009, summer 2009 and winter 2010 experimental campaigns, respectively (Table 4.2).

Table 4.2: Recirculation ratios during experimental campaigns at MBR Heenvliet and MBR Varsseveld

MBR Heenvliet			MBR Varsseveld		
Date	Recirculation ratio	Recirculation ratio - average	Date	Recirculation ratio	Recirculation ratio - average
09-Jun-08	2.4	2.6	24-Jun-08	8.4	9.0
10-Jun-08	2.7		25-Jun-08	9.4	
11-Jun-08	2.9		26-Jun-08	9.1	
12-Jun-08	2.7		09-Feb-09	10.1	8.8
13-Jun-08	2.2		10-Feb-09	3.9	
26-Jan-09	1.7	1.8	11-Feb-09	7.4	
27-Jan-09	1.8		12-Feb-09	12.4	
29-Jan-09	1.9		13-Feb-09	10.4	
29-Jan-09	2.0		17-Aug-09	14.0	13.3
20-Jul-09	4.9	4.7	18-Aug-09	13.8	
21-Jul-09	4.8		19-Aug-09	13.5	
22-Jul-09	4.4		20-Aug-09	12.3	
23-Jul-09	4.3		21-Aug-09	12.8	
24-Jul-09	5.0		01-Mar-10	5.1	9.0
22-Feb-10	3.2	3.4	02-Mar-10	10.3	
23-Feb-10	3.0		03-Mar-10	10.5	
24-Feb-10	3.8		04-Mar-10	11.4	
25-Feb-10	3.7		05-Mar-10	7.8	

Therefore, heterogeneous filterability results coincide with low return rates as observed before (Moreau 2010, Lousada-Ferreira 2011). Filtration characterisation results from Heenvliet show slight improvement along the process flow that in consequence resulted in better filterability, when samples originated from the aerated membrane tanks. In fact, the best filterability was usually found in the membrane tank and worst in anaerobic or anoxic compartment. For example, in winter 2009, ΔR_{20} value of samples from the membrane tank was about 35% lower compared to samples from the anaerobic compartment (Figure 4.29c). On the other hand, a different behaviour was observed, during the 'winter 2010' campaign, when filterability deteriorates with the process flow (Figure 4.29d). However, similarly to other campaigns, the filterability in the membrane tanks has improved compared to the previous section. This can be explained by the critical MLSS concentration concept proposed by Lousada-Ferreira (2011). In all investigated cases, MLSS concentration of the activated sludge in the membrane tank was above the 10.5 g/L, i.e., critical MLSS concentration. In all of these cases but one, improvement in membrane tank sludge filterability was observed. Conclusively, a general pattern in the filterability development could not be found. This lack of consistent trend in the filterability distribution and the observed differences are possibly associated with the change in operation of the MBR. In March 2009, between 2nd and 3rd

campaign, the MBR was switch from in-series to parallel operation of the MBR system. Subsequently, differences in MLSS concentrations between the aerobic and the membrane sections were lower than during in-series operation. In consequence, differences in the sludge filterability became less apparent (Figure 4.29b) and close to nearly homogeneous filterability along the MBR process (Figure 4.29b).

In the case of MBR Varsseveld very similar filtration resistance of the activated sludge along the whole MBR was observed indicating a non-dependency between the filterability rate and the sampling point. The heterogeneous sludge filterability along the process during 'winter 2009' campaign is probably caused by unexpected events at the plant and related process disturbance that happen at that time (*Chapter 6*). A slight, yet within the accuracy of the DFCm, improvement in activated sludge filterability along the process line was found during summer campaigns. Contrary, in winter period, filterability changes were more visible along the MBR as the quality of the sludge deteriorated with the process flow to be the worst in the membrane tanks. However, comparing all the results it can be concluded that activated sludge filterability was stable and similar along the entire MBR. Therefore, the filterability can be considered uniform and homogenous in all compartments. This nearly uniform activated sludge filterability likely arises from the high recirculation rates in the MBR, which results in consistent activated sludge distribution (Table 4.2). MBR Varsseveld operates with return ratios between 8.8 and 13.3 which are relatively high even as for submerged systems, which are typically operated with a recirculation ratio of up to 4-6 (Bentum and Schyns 2008, Moreau 2010, Lousada-Ferreira 2011). Nevertheless, the recirculation factor is comparable with other MBRs equipped with GE-Zenon membranes, e.g. in Trento and Schilde (Moreau 2010, Lousada-Ferreira 2011). The build up of the MLSS concentrations along the MBR process was observed. The MLSS increased by maximum 2 g/L to reach concentrations between 8.4 g/L and 11.5 g/L in the membrane tank.

Also the MBR Ootmarsum results have shown rather uniform and homogenous filterability between the different MBR sections (Figure 4.29b-d). The development of filterability in compartments of the sidestream MBR Ootmarsum is most likely explained by high sludge recirculation rates generally applied in sidestream MBRs. The exact data are not available but it is known that, compared to submerged systems, sidestream tubular MBRs operate at much higher recycle ratio of approximately 10 to 20, and up to 33 to provide high circulation velocity of the sludge stream along the membranes (Bentum and Schyns 2008, Lousada-Ferreira 2012). Outlying results obtained during the 'summer 2008' campaign, might be explained by the limited number of measurements carried out with anoxic sludge samples and by exceptionally poor filterability in the anaerobic zone. The activated sludge MLSS concentrations were similar between the various tanks, except anoxic tank where significantly higher concentrations were measured, e.g., 16 g/L in the anoxic tank versus 7-8 g/L in the other tanks or 10 g/L in the anoxic versus 6-7 g/L in the other tanks.

As general conclusion it can be stated that for all investigated plants the worst activated sludge filterability was usually measured in the first biological sections of the MBR – namely the anaerobic and/or the anoxic zones. This is most probably related to the incoming influent composition and to the fact, that the treatment process and biological degradation have not been completed yet and a significant amount of the potential foulants coming from the influent is present in mixed liquor. Moreover, activated sludge filterability was relatively uniform over all compartments of the MBR Varsseveld and MBR Ootmarsum. The homogenous filterability is most likely caused by high recirculation rates as stated by Moreau (2010) and Lousada-Ferreira (2011). The observed seasonal changes in filterability development might be related to temperature fluctuations, changes in influent composition or to differences in occurring flocculation-deflocculation processes. Furthermore, an overall and common pattern of filterability changes along the process flow could not be identified. Therefore, based on these results, identification of the best order of the MBR compartments with respect to sludge filterability is not possible. However, it can be concluded that several arrangements of the MBR compartments can be considered as good depending on the water quality targets. This is in accordance with the findings of Moreau (2010). Notwithstanding, the concept of critical MLSS concentration, and its influence on sludge filterability, was clearly observed in case of MBR Heenvliet. In the MBRs operated with low return ratios, MLSS concentrations in the separate membrane compartments should exceed the critical MLSS concentration of 10.5 g/L. In that way improvement of the sludge filterability in the membrane tank is possible. In the MBRs operated with high return ratios, solids concentration in the membrane section is often comparable, likewise the filterability, to the solids concentration in the preceding aerobic tank. Furthermore, besides little concentration differences, the MLSS concentration in the membrane section usually does not exceed the critical MLSS concentration. However, the results show that development of filterability can vary, both between the seasons and the plants.

4.7 Seasonal fluctuations in activated sludge filterability

The results of activated sludge filterability largely vary between summer and winter campaigns (Figure 4.30). Filterability of the activated sludge from the biological part of the MBRs was more prone to seasonal fluctuations, while filterability of samples from the membrane compartments was more stable and in often better, as indicated by lower ΔR_{20} values.

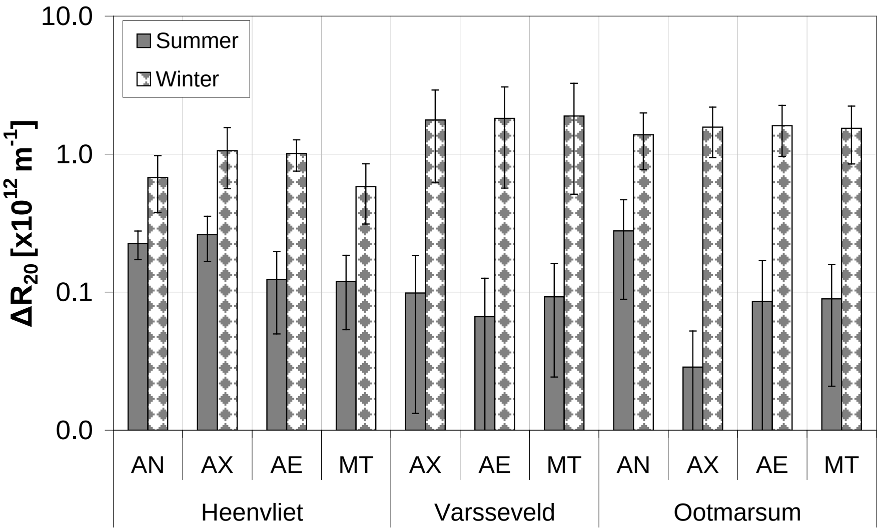

Figure 4.30: Seasonal filterability fluctuations in investigated MBRs

Similar behaviour can be found when filterability of the samples collected from the membrane compartments is individually analysed in more details. Clearly, during summer conditions, filterability was better than during winter conditions (Figure 4.31). The average ΔR_{20} values measured during the summer and the winter periods were in the range of $0.04\text{-}0.2 \cdot 10^{12}$ m^{-1} and $0.3\text{-}3.5 \cdot 10^{12}$ m^{-1}, respectively.

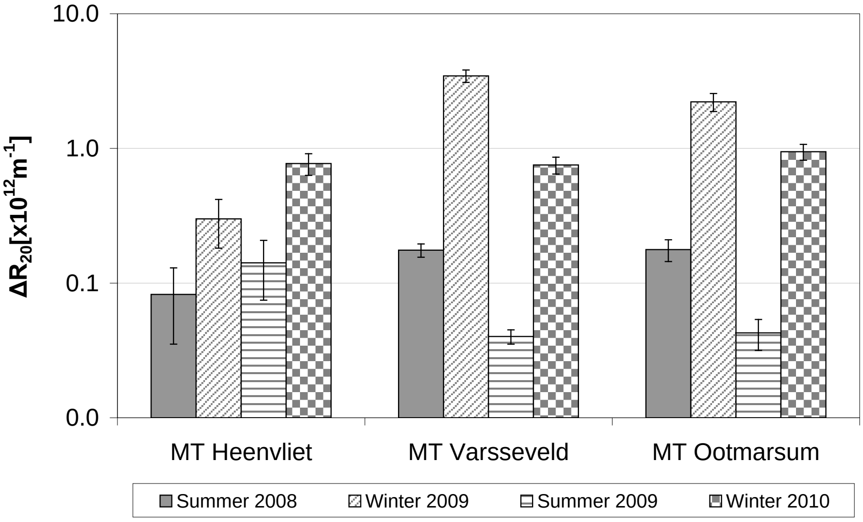

Figure 4.31: Seasonal filterability fluctuations in membrane compartments of investigated MBRs

Therefore, filterability measured in summer was classified as good or moderate, whereas in winter it was classified as moderate or poor (Figure 4.32a). This is in accordance with the findings of Moreau (2010) who found similar distribution of filterability quality between summer-winter seasons (Figure 4.32b). The degree of filterability fluctuation was depending on the location and plant configuration. The impact of MBR configuration on activated sludge filterability is discussed in the *Chapter 7* of this thesis.

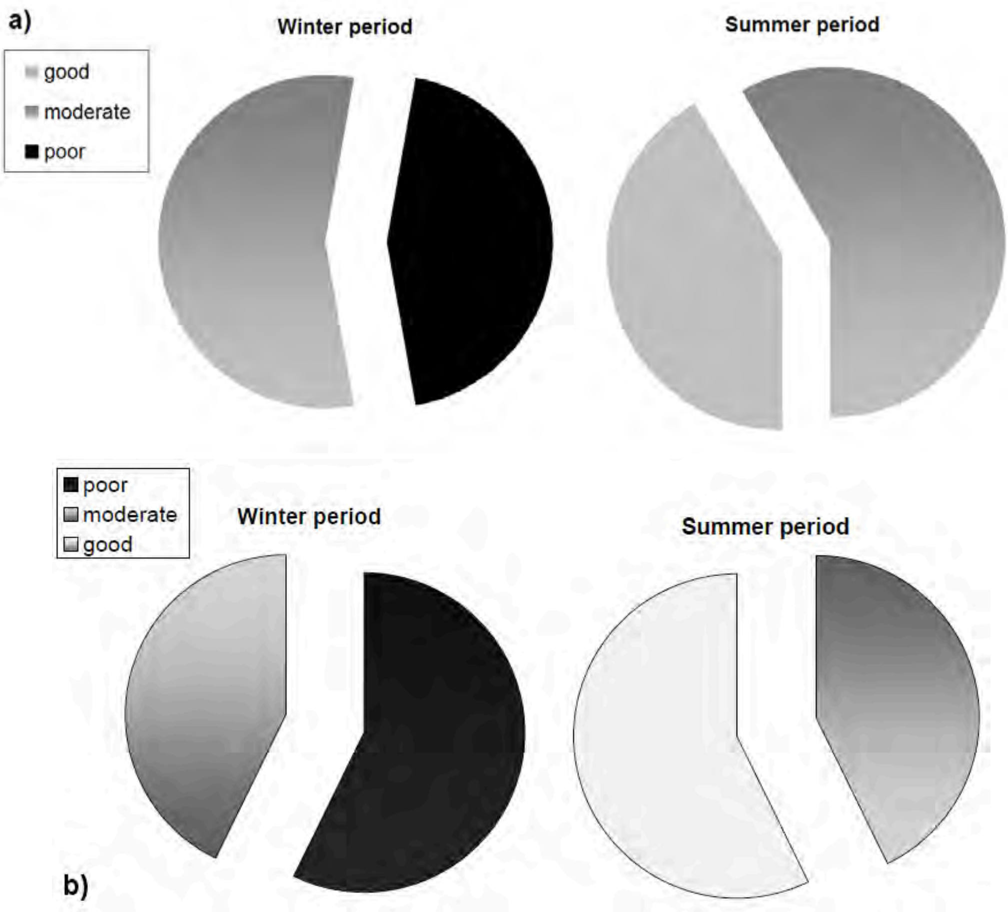

Figure 4.32: Filterability quality of full-scale MBRs during different periods of the year: a) this research, b) research of Moreau (2010)

The seasonal variations in filterability but also associated seasonal changes in membrane fouling rates and membrane permeability are known facts, well reported in literature dealing with MBR operational experiences (Rosenberger et al. 2006, Wang et al. 2006, De Wilde et al. 2007, Drews et al. 2007, Mulder et al. 2007, Lyko et al. 2008, Wedi and Joss 2008, Miyoshi et al. 2009, van der Graaf et al. 2009, Zwickenpflug et al. 2009, Moreau 2010, Wang et al. 2010, Van den Broeck et al. 2011). Few factors are currently reported to be responsible for seasonal fluctuations and the associated changes. Variations in the influent composition, differences in biomass characteristic, changes in microbial growth and bacterial activity or changes in apparent viscosity are the ones most often mentioned. Nevertheless, the most obvious, and probably the most important, is the temperature and its impact on the abovementioned factors. From our results clearly follows that, fluctuation patterns of the filterability are coherent with the seasonal temperature fluctuations. The activated sludge temperature during winter campaigns was on average 10.3 °C lower than during summer campaigns (Table 4.3).

Table 4.3: Average temperature of activated sludge during research campaigns

	Average activated sludge temperature [°C]			
	Summer 2008	Winter 2009	Summer 2009	Winter 2010
MBR Heenvliet	18.7±1.0	12.6±0.3	21.0±0.5	8.8±0.6
MBR Varsseveld	22.0±0.3	10.4±1.6	23.3±0.6	11.7±1.0
MBR Ootmarsum	18.4±0.3	8.1±0.2	21.1±0.3	11.0±0.4

Although temperature effect on activated sludge filterability is known (Lyko et al. 2008), until now it is not well understood. Some researchers (Wang et al. 2006, De Wilde et al. 2007, Wedi and Joss 2008) explain temperature effect through the apparent viscosity of activated sludge, while others (Moreau et al., 2009) found it insignificant. Thus, changes in filterability are not explained solely by changes in activated sludge viscosity (Jiang et al. 2005). Also the relation between total suspended solids and apparent viscosity (Rosenberger et al. 2002) is insufficient to explain the observed differences. Other factors, like submicron particles release due to flocculation-deflocculation process (van der Graaf et al. 2009, Geilvoet 2010), different influent composition in the winter through the lower degradation process in the sewage (Lousada-Ferreira 2011), lower degree of degradation of pollutants during biological treatment process at lower temperatures (Geilvoet 2010) or lower microorganisms’ activity attributed to the winter conditions, are likely playing a role in this phenomenon. Therefore, many phenomena related with the worsening of the activated sludge filterability in winter must be taken into account.

4.8 Temperature effect

As previously stated (section 4.7) low temperatures of the inflow can negatively affect sludge filterability. The relation of activated sludge filterability and temperature was investigated during the four main experimental campaigns carried out at each location. A negative seasonal effect of decreasing temperature on activated sludge filterability was observed during all experiments. At the time of summer measurements, the temperature of the activated sludge was between 17^{o}C and 25^{o}C and the activated sludge presents good to moderate filterability. While at the time of winter monitoring activities, the activated sludge had a temperature ranging between 7^{o}C and 13^{o}C and presents moderate to poor filterability. The temperature effect on activated sludge filterability can be clearly observed as filterability improves with temperature increase (Figure 4.33a). This is in agreement with observations of Moreau (2010) as illustrated in Figure 4.33b. Hence, in respect to activated sludge filterability, temperature is clearly a first order influencing parameter.

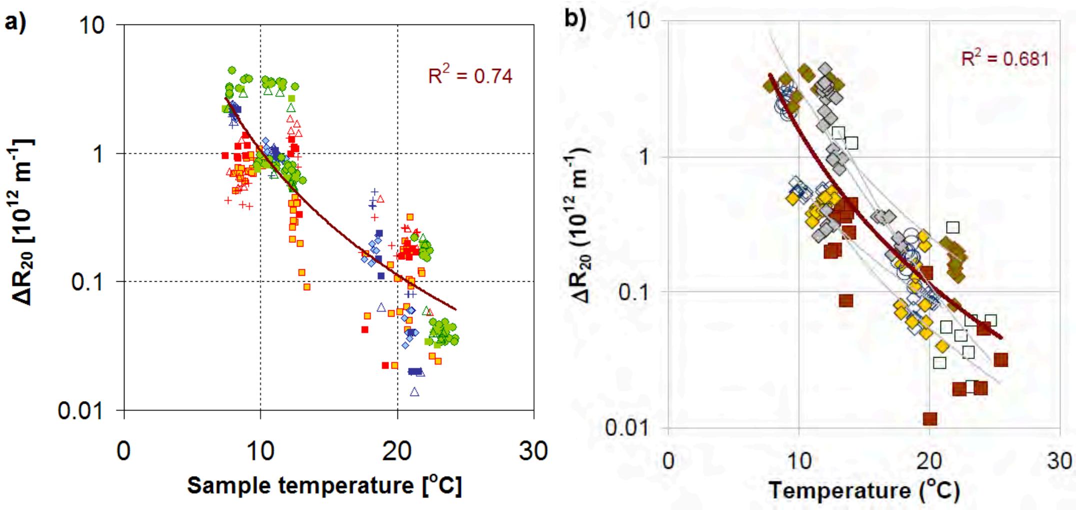

Figure 4.33: Filterability in relation with temperature: (a) this research and (b) research of Moreau (2010)

Furthermore, an additional 10 months-long research campaign was conducted in order to assess the influence of seasonal temperature fluctuations on MBR sludge filterability (*Chapter 5*). Consequently, the seasonal variability of various water quality and physicochemical parameters influencing activated sludge properties could be elucidated. The impact of temperature on activated sludge filterability is thoroughly discussed in *Chapter 5*. The results of activated sludge filterability and their summer-winter differences were also compared with the actual operational permeability of the respective MBR. The observed decrease in permeability in the winter is likely related with the activated sludge filterability deterioration. The relation of filterability and permeability is discussed in *Chapter 6*.

CHAPTER 5

Impact of temperature on raw wastewater composition and activated sludge filterability

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5 Impact of temperature on raw wastewater composition and activated sludge filterability

5.1 Chapter outline

This chapter, based on Krzeminski et al. (2012b), describes the experiments intended to study the influence of seasonal temperature fluctuations on raw domestic wastewater composition and MBR sludge filterability. After a short introduction presented in section 5.2, a brief description of the research approach and a full-scale MBR from where samples were collected is provided in section 5.3. The following section (5.4) deals with the results of the filterability, particle size distribution, respirometry, fractionation and set of physicochemical measurements. The results are discussed and evaluated in this section. The chapter is concluded with the summary of the results and main conclusions as presented in section 5.5.

5.2 Introduction

Despite extensive research efforts, the fouling process is still not fully understood (Drews 2010) and remains one of the main focuses of MBR research within the academic community (Santos et al. 2011). The complexity of the problem arises from the multiple inter-related parameters and their role in activated sludge filtration. The efficiency of the filtration process in an MBR is governed by activated sludge filterability, which is determined by the interactions between the biomass, the wastewater and the applied process conditions.

5.2.1 Filterability of activated sludge

Filterability is an inherent property of activated sludge and an indicator of mixed liquor fouling propensity. The activated sludge condition is of importance for a stable and efficient MBR operation (Gil et al. 2011). As a matter of fact, activated sludge properties are strongly influencing its filterability and consequently the filtration process. Hence, good sludge filterability is necessary to achieve an optimal performance of the process and trouble-free operation of an MBR. The main contributors of fouling still need to be investigated and more accurately quantified in order to develop efficient counter measures and fouling prevention protocols. Filterability measurements can significantly contribute to understand MBR performance under specific prevailing process conditions. Filterability measurements can be used to indicate whether a permeability decrease observed at a full-scale MBR should be attributed to poor activated sludge filterability or inadequate operation of the filtration process (Geilvoet 2010, Moreau 2010). However, until now a single universal fouling indicator has not been identified (De la Torre et al. 2010a). The classical parameters, such as polysaccharides and proteins concentration, CST or TTF are only partly characterising sludge properties, indirectly indicating filterability. In-situ or ex-situ filtration tests – such as DFCm – could provide more relevant information (Evenblij et al. 2005, De la Torre et al. 2009, Lesjean et al. 2011).

5.2.2 Temperature influence

Interactions between temperature, biological activity and fouling rate are of a complex nature. Temperature influences microbial community, biological activity rate and sludge morphology. The optimum temperatures for common bacterial activity in activated sludge processes are in the range between 25 ºC and 35 ºC. When the temperature drops to about 5 °C, the autotrophic nitrifying bacteria practically cease functioning and at 2 ºC even the chemo-heterotrophic bacteria mineralising carbonaceous material become essentially dormant (Tchobanoglous et al. 2003). Low temperature periods might cause a change in both the raw wastewater composition and in the physicochemical properties of the activated sludge, affecting subsequent processes (Geilvoet et al. 2006). Furthermore, temperature is an important parameter in MBR operation as it affects the filtration of the activated sludge and has an influence on the membrane performance (Radjenović et al. 2008).

Deterioration of the activated sludge filterability in winter period is a common observation in MBR installations treating domestic sewage (Rosenberger et al. 2006, Mulder et al. 2007, Lyko et al. 2008, Miyoshi et al. 2009, van der Graaf et al. 2009, Moreau 2010, Wang et al. 2010). In fact, fluctuation patterns of the filterability are coherent with the seasonal temperature fluctuations. However, the observed seasonal changes cannot be explained only by changes in the viscosity (Moreau et al. 2009). As it has been reported before, many phenomena related with the worsening of the activated sludge filterability in winter must be taken into account. The possible mentioned (Jiang et al. 2005, Judd 2006, Geilvoet 2010) effects of low temperature are:

- Intensified deflocculation of activated sludge, leading to floc size reduction and release of EPS and submicron particles;
- increased viscosity of the sludge, lowering the shear stress generated by the coarse bubble aeration. In result, deposition of particles on the membrane may increase;
- reduced mass transfer rate due to linear relation of Brownian diffusion and temperature. Consequently, the particle back transport velocity decreases;
- reduced COD biodegradation resulting in higher concentration of particulate and soluble COD.
- low efficiency of volatile suspended solids reduction (Lee et al. 2005).

Besides, temperature acts upon the rate of bio-reactions, bubble size, oxygen solubility in water, settling properties, thickness and/or porosity of cake layer (Chiemchaisri and Yamamoto 1994), biodegradation of particulate organic matter (Lee et al. 2005) and biodegradability of wastewater.

5.2.3 Wastewater composition

The raw wastewater (influent) content is likely to change during winter time, due to the reduced degradation rate of COD and specific substances, e.g. fatty compounds, in the sewage network. In consequence, the quality of the feed water, reported to be a dominant factor in activated sludge filterability (Moreau 2010), will change. In addition, organic loading plays an important role in membrane fouling and sludge filterability (Cho et al. 2005, Ivanovic et al. 2006, Meng et al. 2009, Zhang et al. 2010). Wastewater organic composition is generally characterized by COD, TOC and BOD, which are divided into particulate and dissolved fractions. Nonetheless, organic matters in the wastewater are highly heterogeneous, containing substances of various molecular weights, ranging from the simple compounds to very complex polymers (Henze 1992). Characterising the different particles present in wastewater is helpful in providing information about the chemical composition of organic matter in different particle size fractions (Sophonsiri and Morgenroth 2004). According to Geilvoet (2010), the amount of particles is closely related to the operational and biological circumstances that the activated sludge is experiencing. When in stress, the floc structure is damaged and deflocculation occurs, resulting in release of fine materials and subsequent deterioration of filterability. This stress can be ascribed to continuous variation in the rate, composition or temperature of influent flow.

Although low temperatures likely induce membrane fouling, the impact of temperature on influent composition and sludge filterability was hardly discussed in the past (Jiang et al. 2005, Lei et al. 2009, Gil et al. 2010, van den Brink et al. 2011). Therefore, wastewater characterization is necessary in order to elucidate the seasonal variability of the analytical parameters influencing wastewater treatment processes and activated sludge properties. The aim of this study is to assess the relations between temperature, raw municipal wastewater composition and activated sludge filterability in a full-scale membrane bioreactor.

5.3 Experiments and MBR plant description

In this research, raw wastewater and activated sludge samples were taken from a full-scale membrane bioreactor treating municipal wastewater. The MBR is located at the Heenvliet WWTP. A detailed description of the Heenvliet plant (WSHD, the Netherlands) is presented in Table 3.1. Sampling campaigns were conducted during different seasons of the year (November 2010–August 2011) to assess the influence of temperature and its seasonal fluctuations (Figure 5.1).

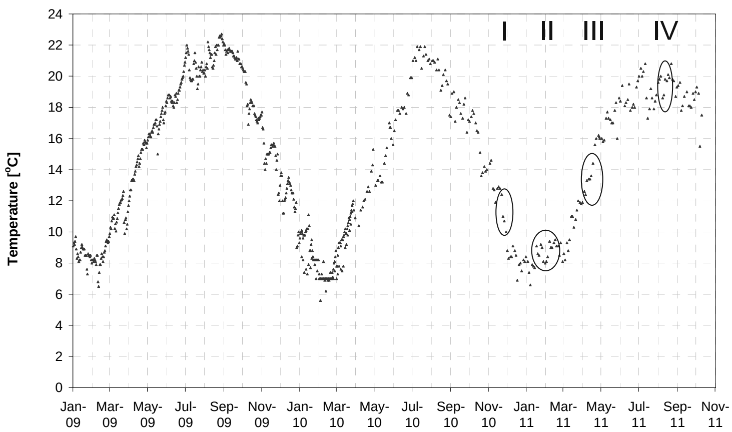

Figure 5.1: Sampling campaigns and temperature fluctuations in Heenvliet WWTP

During this period, both influent and activated sludge were analysed in terms of filterability, respirometry, particle size distribution and physicochemical properties. The influent was collected at the entrance to the MBR after the second sieve (3 mm fine-screen) and the activated sludge samples were taken from the membrane tank. All samplings were carried out at the same time of the day to limit the influence of the diurnal flow variations. Both of the samples were immediately transported to the laboratory and analyzed as soon as possible to keep the characteristics of the influent and activated sludge relatively unchanged.

5.4 Results and discussion

The influence of temperature on raw wastewater composition and activated sludge filterability was studied. The results are discussed below.

5.4.1 DFCm results

An overview of the activated sludge and raw wastewater filterability is presented in Figure 5.2. According to ΔR_{20} classification (Table 3.4), average filterability of the activated sludge was moderate in November-December ($0.7 \cdot 10^{12}$ m^{-1}) and poor in January and April (1.2-$1.3 \cdot 10^{12}$ m^{-1}). The best filterability, yet still moderate, was measured in August ($0.2 \cdot 10^{12}$ m^{-1}). Our results, confirm the observation of other authors that experienced seasonal fluctuations and deterioration of activated sludge filterability when temperature decreases (Rosenberger et al. 2006, Drews et al. 2007, Lyko et al. 2008, Miyoshi et al. 2009, van der Graaf et al. 2009, Gil et al. 2011, Van den Broeck et al. 2011). The filterability results were coherent with the results of the diluted SVI test. The DSVI results also showed a deterioration of the flocculation characteristics under low temperatures in parallel with the filterability results. The DSVI in November-December, January-February, April and August periods was: 116, 150, 119, 100 mL/g, respectively. Higher DSVI values were measured in winter, meaning that the settleability of the sludge has worsened compared to the warm months. Deterioration in DSVI implies low sludge settleability, meaning that flocs are not big or heavy enough to settle.

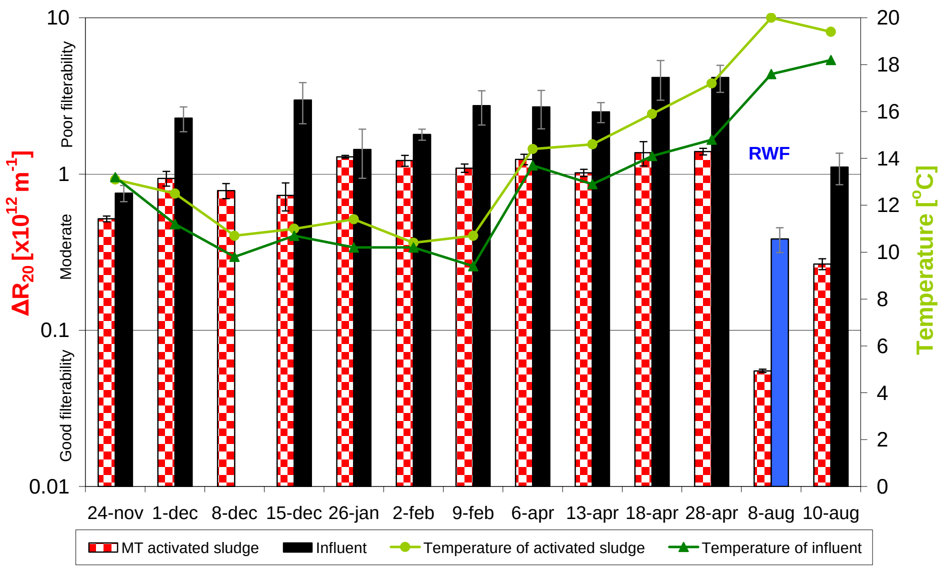

Figure 5.2: Filterability and temperature of activated sludge and influent samples

A strong relation was observed between the filterability and the $\alpha_R{\cdot}c_i$ product, i.e., the mass involved in the cake layer build up (Figure 5.3). This indicates a strong influence of the concentration of substances accumulating in the cake layer on the total cake layer resistance. The higher the $\alpha_R{\cdot}c_i$ product values the worse the filterability of activated sludge and influent. The compressibility coefficient is neither directly related with the activated sludge filterability nor the influent filterability. However, for good sludge filterabilities s coefficient is mainly below 0 and for poor sludge filterabilities is mostly above 0. The values of $\alpha_R{\cdot}c_i$ product and s coefficient are presented in Table 5.1.

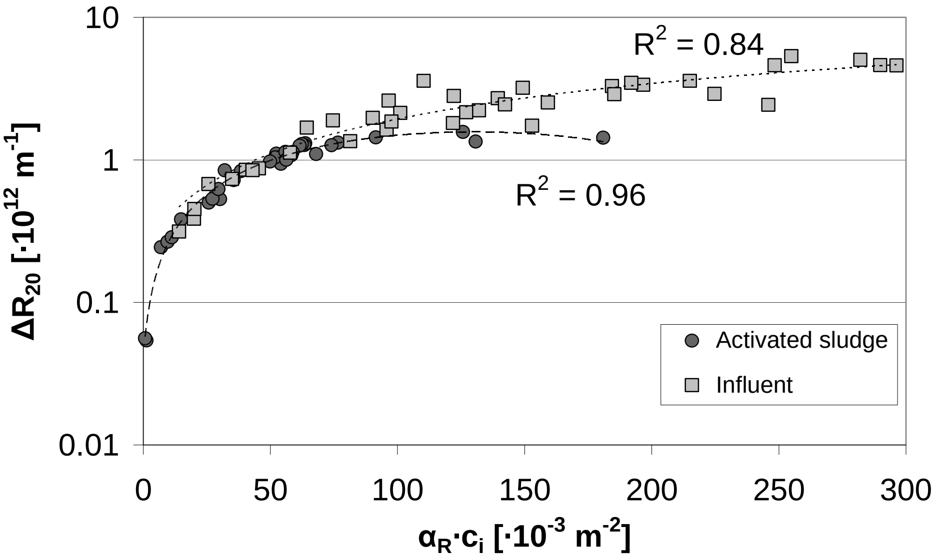

Figure 5.3: Filterability as a function of the mass involved in the cake layer build up for activated sludge and influent cases

Table 5.1: Summary of the $\alpha_R{\cdot}c_i$ product and s coefficient results

	Activated sludge		Influent	
Season / Parameter	**$\alpha_R{\cdot}c_i$ [$\cdot10^{-3}$ m^{-2}]**	**s [-]**	**$\alpha_R{\cdot}c_i$ [$\cdot10^{-3}$ m^{-2}]**	**s [-]**
Nov-Dec 2010	32±11	0.09±0.10	137±80	0.17±0.20
Jan-Feb 2011	59±5	0.10±0.04	106±46	0.17±0.11
Apr 2011	88±41	-	196±71	0.10±0.05
Aug 2011	9±2	-	61±19	0.09

The DFCm results for the raw wastewater were always worse than those for the activated sludge. Best filterability of raw wastewater, indicated by lowest ΔR_{20} results, was measured in August. Nevertheless, filterability of incoming influent was classified as poor, unless rain weather flow conditions occurred resulting in dilution of wastewater and filterability improvement. Results show that fluctuations in filterability along the measurement campaigns were stronger for raw wastewater compared to the more stable behaviour of the activated sludge. Notably, the DFCi and DFCm were defined for the activated sludge and not for raw wastewater analysis (Evenblij 2006, Geilvoet 2010).

Activated sludge composition is generally in agreement with the growth and process conditions prevailing in the aeration tank. Under changing process conditions, e.g. water temperature, sludge characteristics will respond and change accordingly. The timeframe of such response depends on the sludge growth kinetics, the solids residence time, and the extent of the changing parameter. As such, a delay in sludge deterioration can be expected when a temperature drop is imposed to the system. The aforementioned delay likely constitutes an acclimatization period required by microorganisms to adjust to a gradual change in its environment or to the new process conditions. The adaptation, required to achieve pseudo-steady state conditions, is lasting 2-3 times the SRT, which in this case was about 30 days. For example, despite the 5ºC temperature increase between January-February and April campaigns, the filterability of the MBR sludge has not improved yet, even when lower DSVI indicates improved settling properties of the sludge. Likewise, during November-December campaign, the moderately filterable sludge deteriorates significantly within next 40-60 days, even after the average temperature decreased by just 1.0 ºC. The settling properties of the sludge were also worse (i.e., higher DSVI) at the January-February sampling period.

After initial decrease, the SRT was increasing along the measurement campaigns: 25 days in November-December, 23 days in January-February, 26 days in April and 28 in August. Therefore, improvement in filterability might be linked to an SRT increase. However, despite SRT increase of 3 days between January-February and April periods, sludge filterability has not improved but even further deteriorates. Thus, prolongation of SRT does not really explain the filterability improvement in warm period.

5.4.2 Biomass and raw wastewater concentration

TSS and VSS results are shown in Figure 5.4. The MBR was operated in a range of biomass concentration, ranging from 8.3 gTSS/L to 12.9 gTSS/L. The TSS and VSS concentrations in the MBR sludge were lower during winter and higher during summer period, showing a fair correlation to activated sludge filterability. Keeping in mind the controversy about the impact of biomass concentration on sludge filterability (Le-Clech et al. 2006, Van den Broeck et al. 2011), it can be concluded that, in our particular case, the higher the concentration the better the filterability was. Our results are in accordance with the theory of Lousada-Ferreira (2011) postulating that above certain critical biomass concentration the filterability improves. Low TSS during cold season also could be a consequence of reduced SRT. However, reduction in SRT during the cold season was not observed. In fact, SRT was increasing along the measurement campaigns. Therefore, low TSS during the cold season is not explained by the changes in SRT.

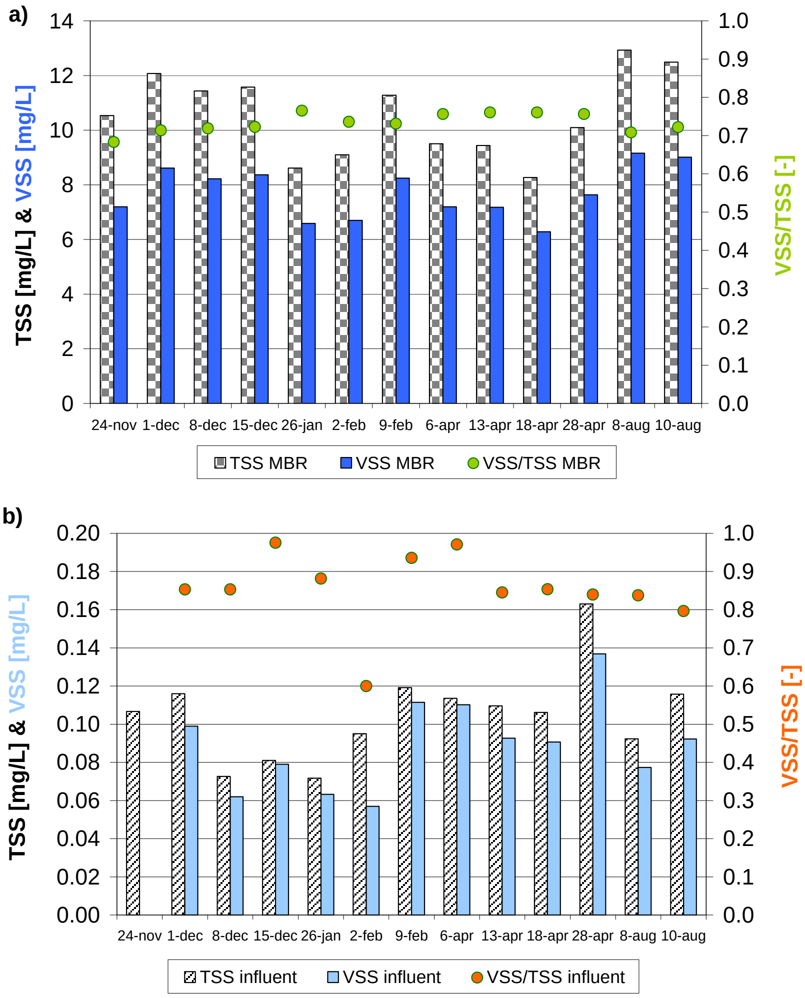

Figure 5.4: Total suspended solids (TSS), volatile suspended solids (VSS) and VSS/TSS ratio of (a) activated sludge and (b) influent samples

The TSS and VSS of raw wastewater varied across the sampling and were ranging from 0.07 gTSS/L to 0.16 gTSS/L and from 0.06 gVSS/L to 0.14 gVSS/L, respectively. Neither their concentrations nor their change seems to be related to the change in the wastewater filterability.

During periods of poor filterability a higher VSS/TSS ratio was observed for sludge but also for some of wastewater samples. TSS represents both organic and inorganic fractions, i.e., total biomass, whereas VSS reflects the organic matters in the activated sludge. The observed variation in the VSS/TSS indicates a change in biomass components. Greater VSS/TSS ratio a indicates larger percentage of organic content, reduced biomass mineralization (Mikosz 2011) and most likely the increase in the loading rate. Contrary, lower VSS/TSS ratios are a sign of accumulation of an inert material (Ouyang and Liu 2009), higher stabilization rate of the sludge and reduction of biological activity of sludge (Benedek et al. 1972). Furthermore, better mineralized sludge is expected when temperature is high. Accordingly, lower values of VSS/TSS ratio were measured during periods of improved, yet moderate filterability, i.e., in August.

5.4.3 Wastewater composition

Filterability of both, activated sludge and raw wastewater, deteriorated with the increase in influent COD (Figure 5.5 and Figure 5.6). This indicates a correlation between filterability and the organic load for the studied samples. The food to microorganism ratio (F/M) varied between the campaigns: 0.05 kgBOD/kgMLSS.day in November-December, 0.07 kgBOD/kgMLSS.day in January-February, 0.04 kgBOD/kgMLSS.day in April and 0.02 kgBOD/kgMLSS.day in August. Thus, lowest organic loading correlates with best filterability but also with warmest period. Contradicting results are obtained for the periods of worst filterability. Although during the January-February campaign (ΔR_{20}=1.2) the highest organic loading was observed, during April period the loading was lower yet filterability (ΔR_{20}=1.3) was as bad as in January-February. Therefore, changes in the organic loading are not solely responsible for the changes in measured filterability, which are most likely consequence of temperature variations. Furthermore, if an MBR has been fed with an average influent COD, and suddenly, influent COD increases, the microorganisms react to this alteration and a change in floc structure or even deflocculation can be expected (Geilvoet 2010). In results, the floc structure is damaged, floc size decreases and fine materials, e.g. submicron particles or EPS, are released from activated sludge matrix into free water. In consequence, deterioration of the filterability is observed (Geilvoet 2010). Also temperature causes a change in filterability which might be ascribed to a change in flocculation properties. Temperature also causes changes in wastewater composition. Yet, this does not mean that changes in wastewater composition do change the filterability of the sludge.

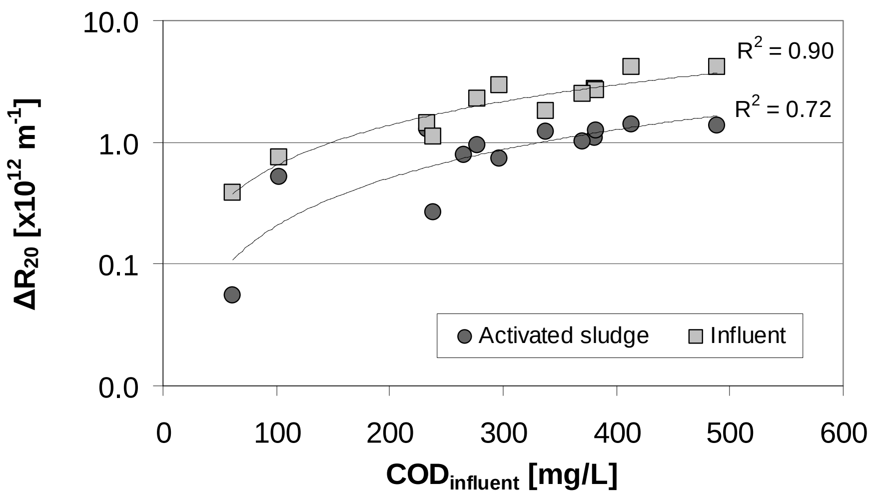

Figure 5.5: Filterability of activated sludge and influent vs. influent COD concentration

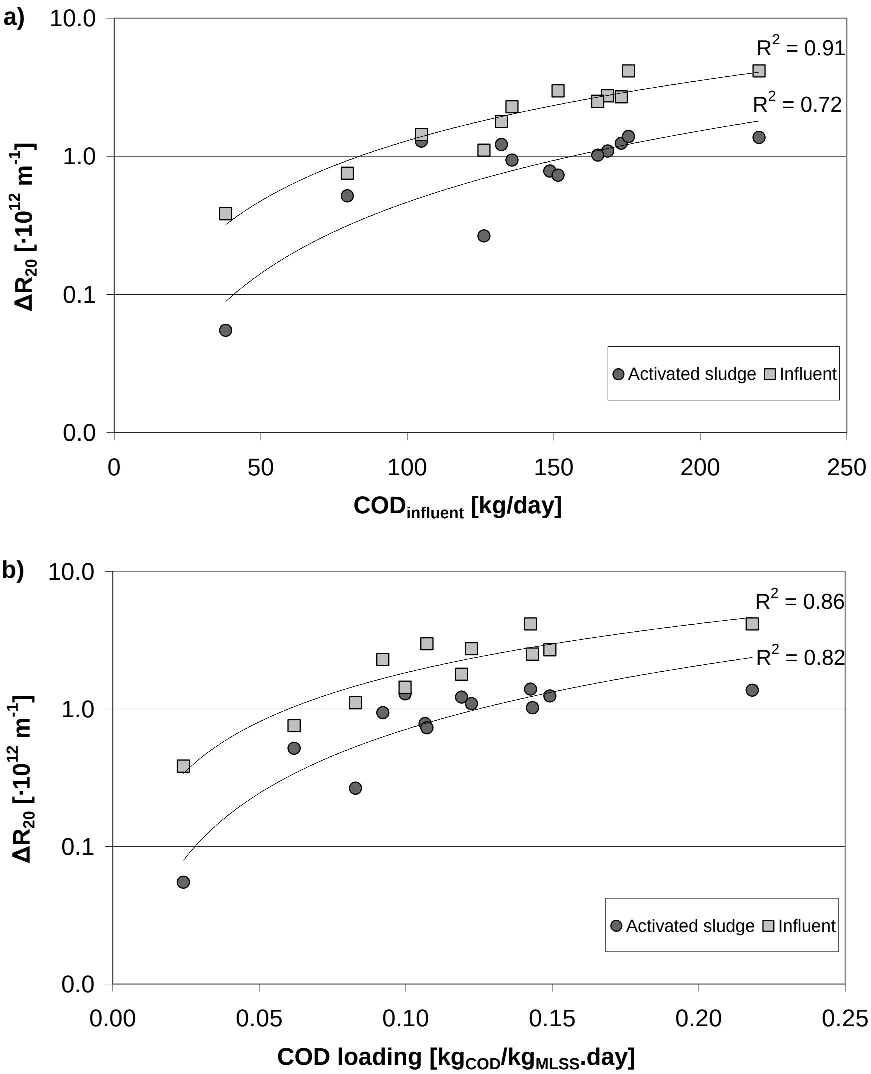

Figure 5.6: Filterability of activated sludge and influent vs. (a) influent COD load and (b) COD loading

The characteristics of wastewater (TSS, VSS, TP, TN, NH4, COD and TOC) and activated sludge (TSS and VSS) are listed in Table 5.2. The influent and permeate characteristics together with removal efficiency of the MBR during different periods (based on the data collected by plant operators) are presented in Table 5.3: .

Table 5.2: Summary of the results from wastewater and activated sludge characteristics

Parameter		**TSS**	**VSS**	**TP**	**TN**	**NH_4**	**TOC**	**$COD_{influent}$**	**$COD_{5.0}$**	**$COD_{1.2}$**	**$COD_{0.45}$**	**$COD_{0.1}$**	**TSS_{sludge}**	**VSS_{sludge}**
Sampling day		g/L	g/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	g/L	g/L
1	24.11.2010	0.107±0.025	N.M.	2.4±0.1	33±1	9±1	112±9	102±11	80±1	55±4	52±2	57±17	10.5±0.4	7.2±0.3
2	01.12.2010	0.116±0.012	0.099±0.047	7.0±0.1	58±1	42±2	205±2	278±5	273±21	193±7	175±1	170±6	12.1±0.4	8.6±0.3
3	08.12.2010	0.073±0.011	0.062±0.009	6.2±0.1	49±1	35±0	182±0	265±1	217±10	158±6	145±1	170±3	11.4±0.1	8.2±0.1
4	15.12.2010	0.081±0.006	0.079±0.008	7.2±0.1	50±4	43±3	189±2	297±1	228±3	189±7	186±0	197±1	11.6±0.2	8.4±0.2
5	26.01.2011	0.072±0.003	0.063±0.004	5.6±0.1	37±3	32±5	171±12	233±1	173±1	146±3	132±3	197±1	8.6±0.2	6.6±0.2
6	02.02.2011	0.095±0.019	0.057±0.007	7.9±0.1	57±2	40±3	181±1	338±23	253±4	232±0	182±0	197±10	9.1±0.1	6.7±0.1
7	09.02.2011	0.119±0.003	0.112±0.002	8.4±0.2	59±0	52±0	194±9	381±7	303±1	249±7	208±3	284±14	11.3±0.1	8.2±0.1
8	06.04.2011	0.114±0.015	0.110±0.014	10.3±1.3	71±3	54±4	200±7	382±11	305±1	248±8	223±1	217±7	9.5±0.2	7.2±0.1
9	13.04.2011	0.110±0.015	0.093±0.006	8.8±0.1	68±0	48±6	191±3	370±8	306±17	280±3	246±11	233±1	9.4±0.3	7.2±0.2
10	18.04.2011	0.106±0.001	0.091±0.003	9.4±0.3	70±1	53±1	223±2	489±1	388±3	324±8	284±3	277±1	8.3±0.4	6.3±0.3
11	28.04.2011	0.163±0.008	0.137±0.002	9.8±0.3	72±1	54±0	204±0	413±7	338±11	286±3	260±8	253±1	10.1±0.4	7.6±0.1
12	08.08.2011	0.092±0.001	0.077±0.003	2.0±0.1	20±1	11±0	141±9	61±1	53±1	53±7	52±6	36±3	12.9±0.1	9.2±0.2
13	10.08.2011	0.116±0.003	0.092±0.001	5.7±0.1	39±1	32±0	135±1	239±1	189±1	162±8	193±4	140±0	12.5±0.1	9.0±0.1

Legend: N.M. – not measured

Table 5.3: Influent and permeate characteristics and removal efficiency

MBR performance	Influent				Permeate								Removal efficiency			
	COD	**BOD**	**TKN**	**TP**	**COD**	**BOD**	**TKN**	**NH_4-N**	**NO_x-N**	**TN**	**TP**	**COD**	**COD**	**BOD**	**TKN**	**TP**
Period	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	%	%	%	%
Nov-Dec 2010	278±111	109±43	32±10	5.0±1.5	21±8	1.0±0.1	0.9±0.3	0.1±0.1	4.6±1.1	5.5±1.3	0.5±0.6	275±106	92±4	98.9±0.4	97.0±0.6	92±10
Jan-Feb 2011	313±170	138±79	41±21	5.8±2.8	28±9	1.2±0.1	1.1±0.3	0.2±0.1	6.1±2.9	7.2±3.2	0.8±0.7	285±161	90±3	98.8±0.9	96.8±1.5	78±28
Apr 2011	500±52	220±42	66±5	9.3±1.1	30±4	1.1±0.1	1.3±0.1	0.2±0.1	4.3±0.4	5.6±0.5	1.2±1.5	470±55	94±1	99.5±0.2	98.0±0.4	87±18
Aug 2011	267±46	210±14	38±4	5.7±0.6	24±0	1.2±0.2	1.2±0.4	0.4±0.4	5.5±1.6	6.7±1.2	4.3±4.7	243±46	91±2	99.0±0.1	96.8±0.8	24±75

5.4.4 Volatile fatty acids (VFAs)

The total influent VFAs concentration was highest in January-February (season average of 347 mgCOD/L), lower in November-December (166 mgCOD/L) and lowest in April and August (~40 mgCOD/L) (Figure 5.7). Two to four times higher concentrations of individual VFAs were measured during colder periods compared to warmer periods (Figure 5.8). The acetic acid was predominant volatile fatty acid in this study. Due to potential dilution of the sample in result of rain weather conditions or overnight storage of the sample for the VFAs analysis, the results of the 24th November and 2nd February samples, respectively, are doubtful and do not agree with the COD measurements. Nevertheless, the difference in VFAs concentrations between colder and warmer periods remains clear.

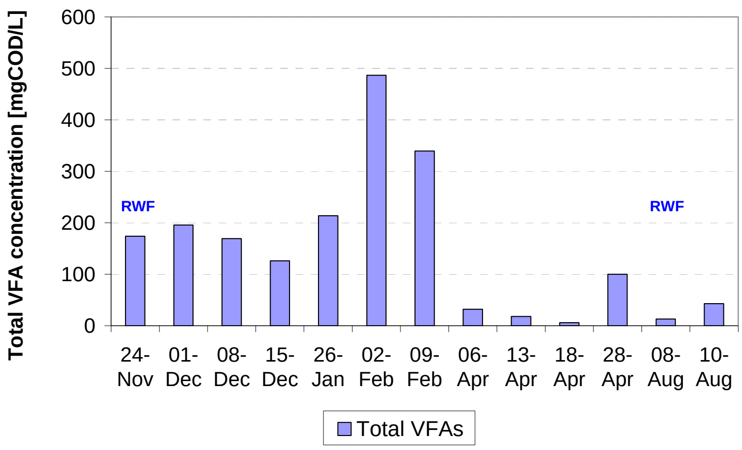

Figure 5.7: Total VFAs concentration in the influent

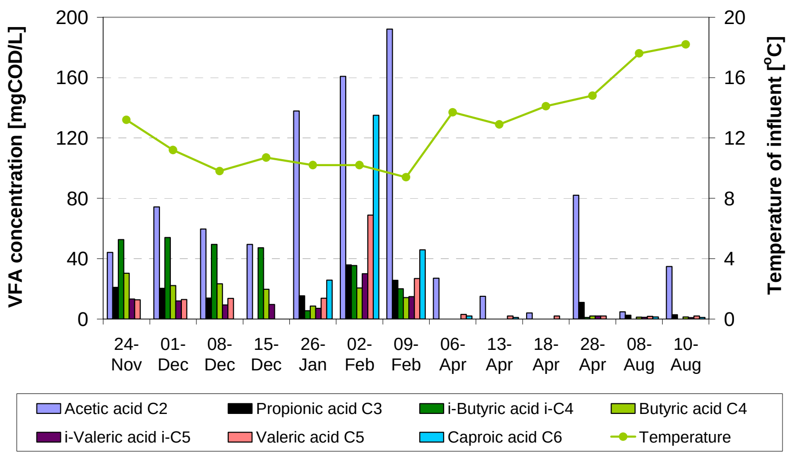

Figure 5.8: Concentration of individual VFAs in the influent during sampling

VFAs are likely produced by microbial oxidation of dissolved and particulate organic carbon in the sewerage anaerobic environment (Szpindor and Łomotowski 1999). The latter resulting from long retention times in the sewer system and transportation in pressurized sewers. Under low temperature conditions, subsequent VFA conversion might be retarded, which is in accordance with the generally observed trend that in winter sewer VFAs are accumulating (Choi et al. 2003, Siedlecka et al. 2008). Elevated VFA concentrations in the wastewater during winter periods might also be explained by potential discharges upstream of the WWTP masking low bioactivity in the raw wastewater. Changes in VFAs conversion rates between the seasons may also explain the concentration differences between warm and cold periods, with high conversion rates in the warm summer period. However, no relation could be observed between VFAs concentration and filterability of influent or activated sludge as well as between VFAs and SOUR.

5.4.5 Fouling indicators: SMP and BPC

SMP measurements provided non-conclusive results. On the one hand, there was a correlation between SMP and resistance tendency in the influent. On the other hand, a clear relationship between temperature and SMP concentration in both influent and sludge samples was not found. Many factors, such as polysaccharides, proteins, but also CST or TTF, were claimed to be directly correlated to irreversible fouling. However, recent studies showed that none of these parameters are universal indicators of fouling phenomena in the membrane filtration process (De la Torre et al. 2010b, Lesjean et al. 2011).

Although a straightforward correlation between SMP and temperature in the activated sludge was not found, it can be observed that with a temperature decrease, polysaccharides concentration increases and proteins concentration decreases (Figure 5.9b). The latter was also observed by Miyoshi et al. (2009), claiming that proteins have higher potential for causing irreversible fouling. Hence, less fouling caused by the proteins at low temperatures can be expected. Keeping in mind that SMP are related with feed COD, it is possible that the SMP fluctuation is also influenced by influent COD and not only temperature. In fact, except December data, influent SMP was generally linked with the influent COD changes (Figure 5.10a). Less apparent relation between influent COD and SMP was found in case of activated sludge (Figure 5.10b).

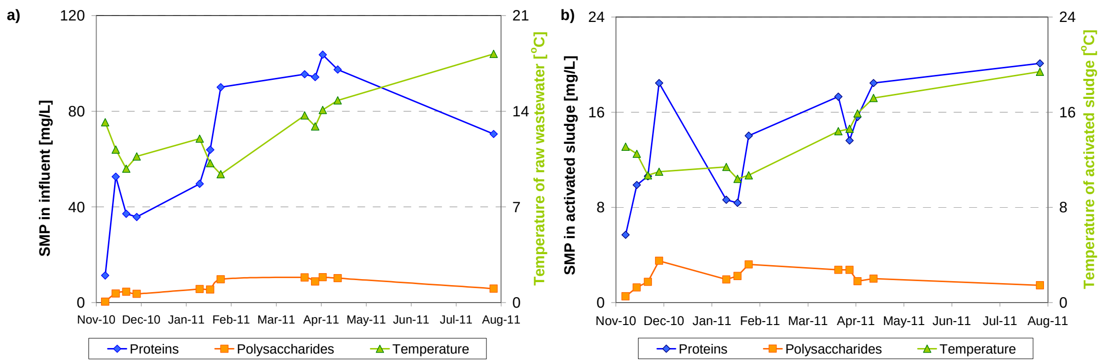

Figure 5.9: SMP content in (a) the influent and (b) activated sludge under different temperatures

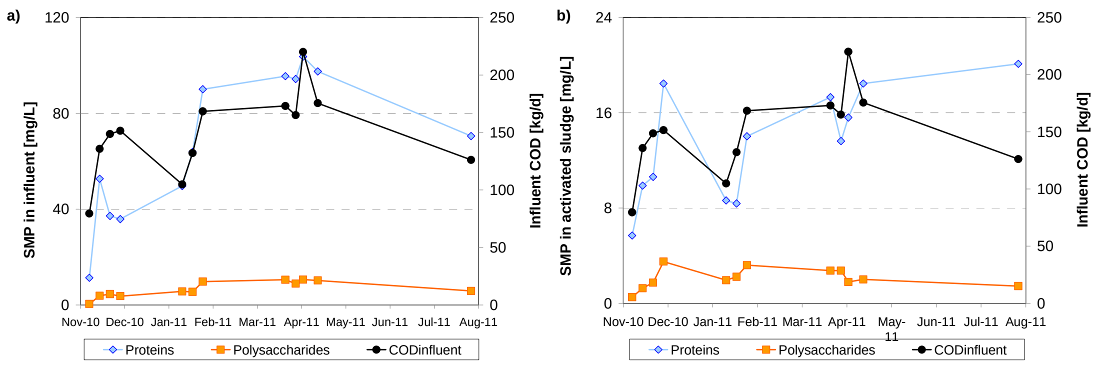

Figure 5.10: SMP content in (a) the influent and (b) activated sludge in relation to influent COD load

Analysing results on filterability and SMP concentration on subsequent measuring days in a specific season, viz. November/December - January/February - April, ΔR_{20} increases with higher proteinaceous matter concentration in the influent (Figure 5.11). For polysaccharides, the correlation with filterability was not clear, in coherence with the results shown by Drews et al. (2008).

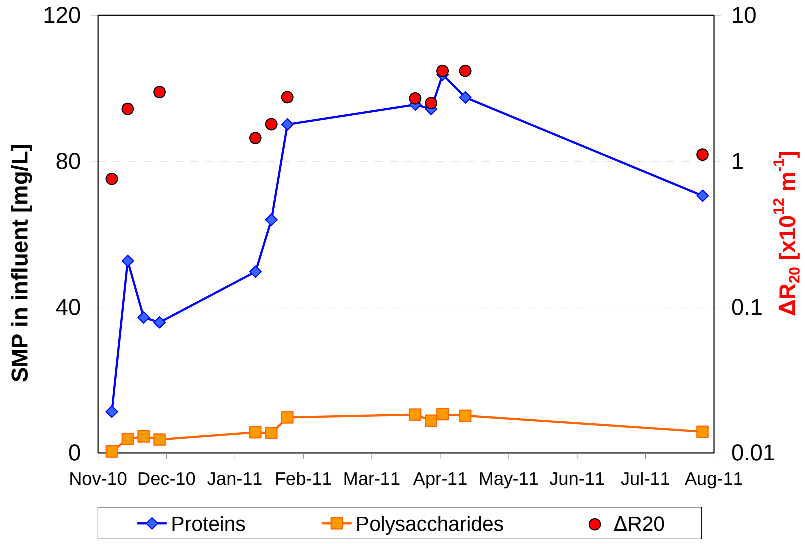

Figure 5.11: Resistance and SMP content in the raw wastewater

Furthermore, no relationship between the proteins/polysaccharides ratio and filtration resistance was found, a phenomenon that was observed by Arabi and Nakhla (2008), who claimed that the increase in this ratio leads into fouling. SMP are not only soluble components but also colloidal and by clustering they lead to BPC formation which are larger biodegradable molecules that do not pass the membrane (Drews 2010). Therefore, they might have an effect in fouling layer formations as reported by Wang and Li (2008).
The contribution of colloidal and soluble fractions, expressed as TOC_{sup}, was found to have a major impact (Gil et al. 2011). The supernatant organic concentration slightly decreases in warmer periods (Figure 5.12). This trend is in accordance with the results showed by Wu et al. (2011) and can result in a better flocculation and filtration quality of the activated sludge during the warmer seasons. The observed peak in the TOC concentration, measured on 8th of August when 20 °C were reported, might not be representative since the sample was collected during rainy weather conditions.

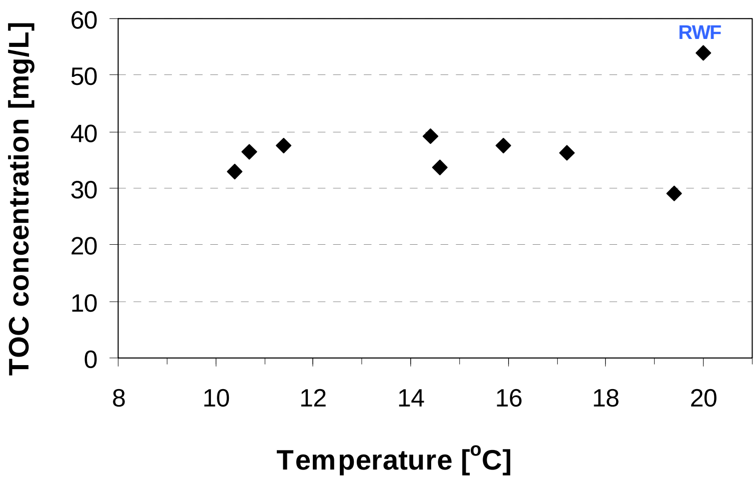

Figure 5.12: Temperature dependency of TOC in the mixed liquor supernatant

The BPC concentration in the mixed liquor varied between 24.0 and 46.9 mgTOC/L, with an average of 31.2±6.5 mgTOC/L. When data were analysed season-wise, and despite the limited results under summer conditions, a moderate relation is observed between BPC and activated sludge filterability (Figure 5.13). In addition, results shown by Sun et al. (2011) could demonstrate a more direct effect of BPC on membrane fouling, in terms of liquid-phase effect on the fouling propensity of the sludge, i.e., worse membrane filtration ability. Incoherencies between these results might be due to the different fouling tests, e.g. UF versus MF membrane, that are used to obtain the membrane resistance data and results in different fouling mechanisms.

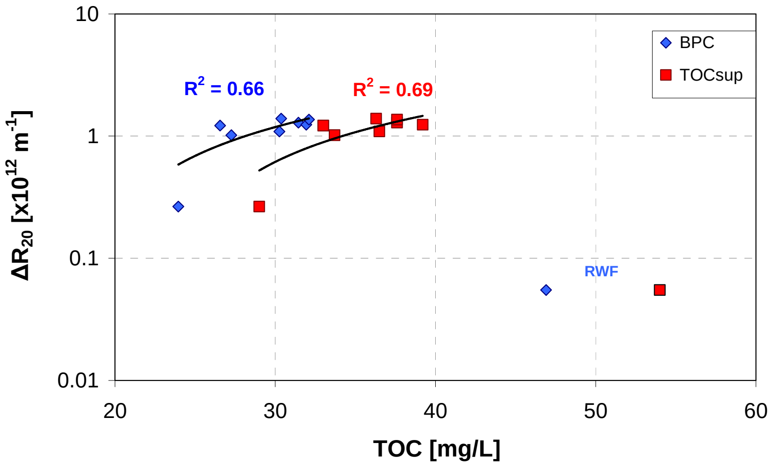

Figure 5.13: BPC and TOC_{sup} effect on resistance

5.4.6 COD fractionation

Results show that on average 63% of the COD measured at different seasons was present in the soluble fraction, i.e., smaller than 0.1 µm, which is in accordance with the findings of Sophonsiri and Morgenroth (2004). The highest COD values in the smallest fraction were observed during the lowest sampling temperature (9.4 °C) on 9th of February and on 18th of April, when worst filterability was measured. In terms of TOC, as depicted in Figure 5.14, concentration in the influent was also higher in winter than in summer and decreases when temperature increases. These results are coherent with the findings of Wu et al. (2011) who reported increase in sludge supernatant organic concentration at low temperature. This indicates that sludge organic content increases, not only due to EPS release during de-flocculation, but also due to introduction by the influent.

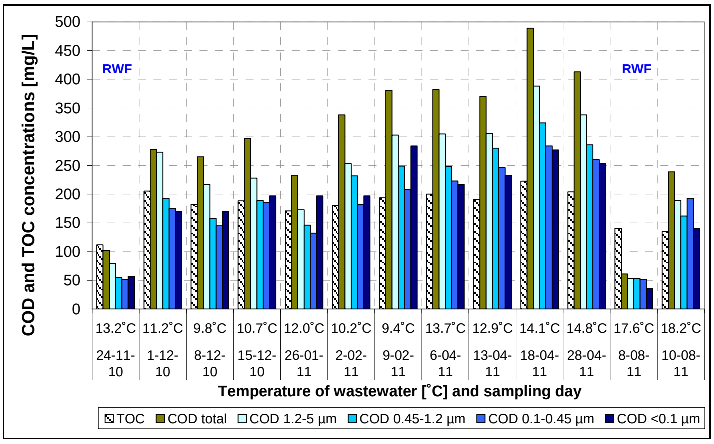

Figure 5.14: Influent TOC concentration and COD concentrations in different fractions of the influent

Results also show that the relative COD concentration of the 0.1 µm fraction increases as the filterability of influent and activated sludge deteriorates (Figure 5.15). Therefore, an increase in membrane resistance is in accordance to the increase in COD concentration of the soluble fraction. It is important to mention that, when comparing Figure 5.15 with Figure 5.6, one can note a similar tendency on both of the plots. This is due to the fact that soluble fraction (Figure 5.15) holds most of the total influent COD (Figure 5.6).

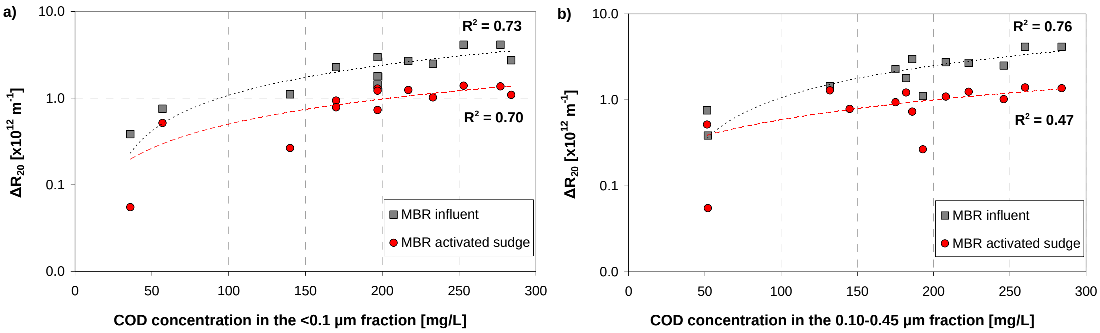

Figure 5.15: Filterability of influent and activated sludge vs. COD concentration of the (a) soluble (<0.1 μm) and (b) colloidal (0.1-0.45 μm) fractions in the influent

Our current fractionation results agree with those observed by van den Brink et al. (2011) for the sludge, who found a different pattern in particle size distribution, and more particles in the smaller fractions, when the temperature of the mixed liquor decreases. Apparently, the soluble fraction of the raw wastewater might play an important role in the resistance increase and subsequently in the membrane fouling. However, as postulated in a recent research (Xiao et al. 2012), also the colloidal fraction, and more specifically the hydrophobic components, could play an major role in the membrane fouling evolution. In addition, it seems to be clear that general parameters like COD, and also EPS, cannot quantify the specific foulants.

5.4.7 Particle size distribution (PSD)

The results of particle counting in the range of 2-100 μm based on activated sludge samples collected from the MBR membrane tank are shown in Figure 5.16. For the sake of legibility only one most representative test for each campaign is plotted. In addition, the results of total number of particles have an average standard deviation of 3.1% proving the light blocking method particle counting of MBR sludge to be reliable and to provide reproducible results.

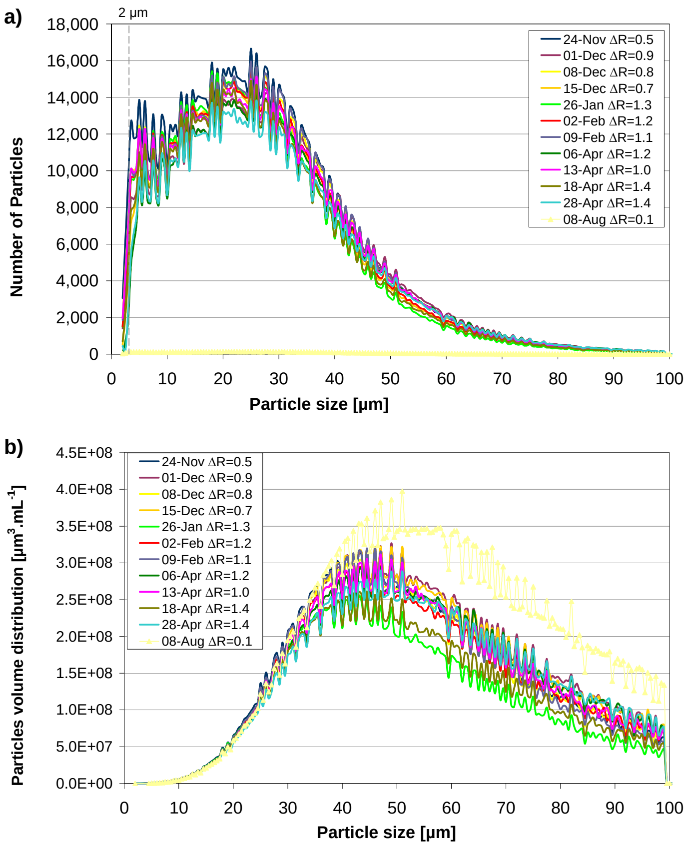

Figure 5.16: Particle size distribution as (a) particle number and (b) particle volume functions

In general, no significant differences were observed between the samples from different seasons in terms of particles number and particles volume distribution. The only considerable difference, observed on 8th of August, was related to the heavy rain weather conditions and was also clearly reflected in nearly all measurements performed that day. Furthermore, the flocs had a similar size of 44-51 µm. In addition, the total number of larger particles does not influence activated sludge filterability (Figure 5.17). Lack of link between filterability and number of particles in the 10-100 µm range was also reported by Lousada-Ferreira et al. (2011). Therefore, it can be assumed that the size of particles, at least in our case, had no evident impact on filterability.

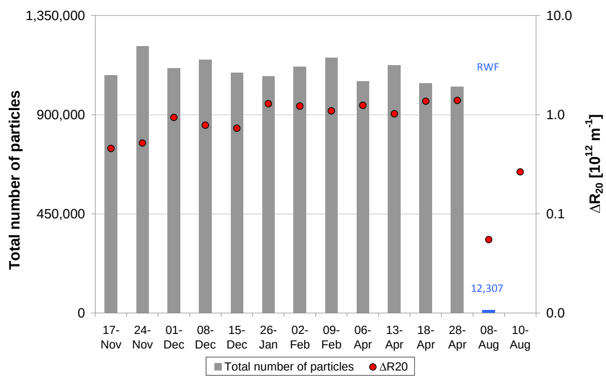

Figure 5.17: Total number of particles versus activated sludge filterability

The particle counting experiments in the range of 0.4-5.0 μm were negatively affected by the technical problems with the particle counting equipment. Consequently, the obtained dataset was smaller than originally planned and the analysis had to be based on a limited number of the tests. Despite these problems, interesting and valuable observations were made. Highest number of particles was measured in the lowest size ranges meaning that the majority of the particles are smaller than 1 μm. The particles in the size range of 1 to 5 μm represents less than 0.5% of the total number of particles in the range of 0.4-5 μm. The number of particles gradually decreases with the increasing particle size. However, a small peak with an increment in the particles number was detected between 1.5 and 3.0 μm. The particles number within this peak was on average 2.5±0.3 times higher than in adjacent size range.

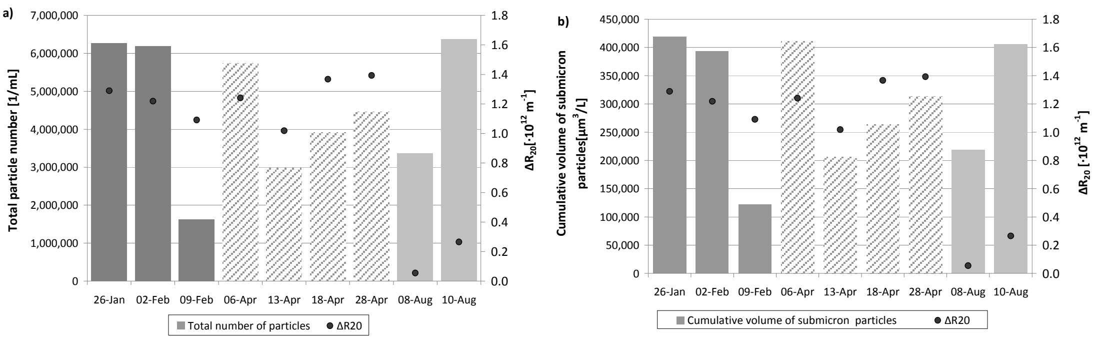

Figure 5.18: Activated sludge filterability versus (a) total particle number size and (b) cumulative volume of submicron particles

Although, the total number of submicron particles cannot be directly linked to activated sludge filterability, when results are compared season-wise, a similar pattern was observed. During each campaign lower number of submicron particles indicates better filterability (Figure 5.18). The trend of filterability change, but not the degree of this change, was mostly coherent with the change in particles number. Similar relations were also observed between filterability and cumulative volume of submicron particles. Accordingly, when the volume of submicron particles in the activated sludge bulk was relatively low a better filterability was measured, whereas increase in submicron particles volume resulted in filterability deterioration. Hence, both total particle number and total submicron particle volume were linked qualitatively but not quantitatively to activated sludge filterability.

5.4.8 Biodegradability

The biodegradability of the samples was estimated by means of respirometry measurements (Table 5.4).

Table 5.4: Results of the respirometric tests

Period	Temp. range	OUR_{sludge}	$SOUR_{sludge}$	$OUR_{influent}$	$SOUR_{influent}$
[-]	[˚C]	[mgO_2/L.h]	[mgO_2/gMLVSS.h]	[mgO_2/L.h]	[mgO_2/gMLVSS.h]
Nov-Dec 2010	10.7-13.1	22.1±2.8	2.7±0.2	50-121	789-1420
Jan-Feb 2011	10.4-11.4	15.4±1.0	2.2±0.1	45-84	695-1125
April 2011	14.4-17.2	23.3±0.9	3.3±0.3	86-144	951-1589
August 2011	19.4-20.3	22.5±2.5	2.5±0.3	42-77	537-834

Legend: OUR – oxygen uptake rate; SOUR – specific oxygen uptake rate

Low OUR and SOUR values were measured for activated sludge indicating low biological activity of the MBR sludge (Henze 2002). The amount of available ‘food’ for bacteria was limited as the measured respiration was mostly endogenous. Moreover, the OUR and SOUR values were lowest during coldest period, i.e., January-February 2011 (Table 5.4), indicating low biological activity of activated sludge under low temperatures (Figure 5.19). The bioactivity could be affected by low temperatures, lower hydrolysis rate, different COD content or due to poor quality of activated sludge (Hasar et al. 2002). Besides, the OUR of activated sludge was basically the same in the other periods and within the range of 22.1-23.3 mgO_2/L.h. In addition, low OUR levels signalize that less easily biodegradable carbon source was available.

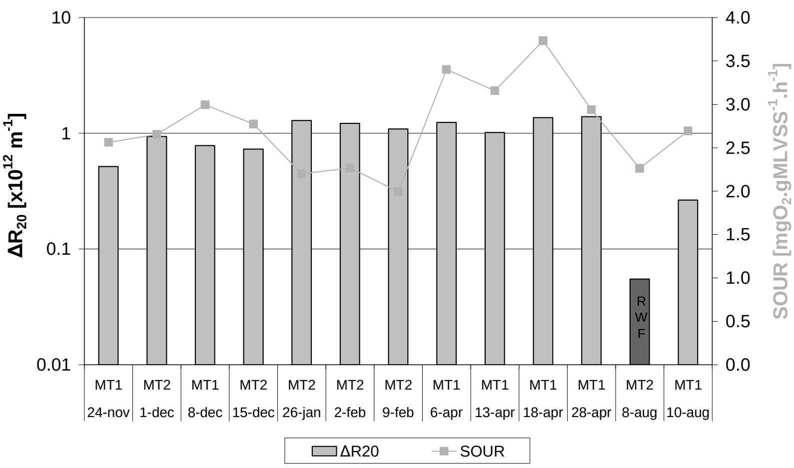

Figure 5.19: Respiration rate of activated sludge in relation to filterability

The specific oxygen uptake rates were similar, yet slightly higher in April when less MLVSS was measured in the samples. Nevertheless, all samples presented low oxygen uptake rates, well below the 10 mgO_2/gMLVSS.h, indicating low biological activity (Henze 2002, Lingling et al. 2009). Furthermore, the oxygen uptake rates decreased with temperature decrease and increased with temperature increase. Hence, it could be concluded that the bioactivity of activated sludge was lowering with the temperature decrease to reach a minimum in winter. However, the August data were not fully supporting this statement. Despite highest temperature, SOUR values were lower compared to the ones from April. Perhaps with the temperature above the certain threshold other factors like food source availability, microbial community composition, or loading became predominant in terms of biomass viability. It was also probably due to limited data points for August and significantly higher VSS of activated sludge in that period that resulted in lower OUR values. Another explanation of the low oxygen uptake rate values could be the rain weather conditions encountered during one of the summer samplings. If the data from that day would be omitted from the analysis, the average OUR_{sludge} would increase to 24.3 mgO_2/L.h indicating relatively high bioactivity of the August samples.

The raw wastewater samples were characterized by considerable fluctuations in hydraulic and organic loading. The aforementioned changes where clearly reflected in the SOUR results of the wastewater samples (Table 5.4). The injection of the same volume of raw wastewater gave different response in summer and winter periods. Faster oxygen uptake, i.e., higher SOUR values, observed in summer indicates faster biodegradation of wastewater in the mixed liquor. In contrast, slower oxygen uptake, thus lower SOUR values, measured in winter was associated with the slower biodegradation of wastewater. Seasonal differences in biodegradation rates were in agreement with the results of filterability monitoring: better in summer and worse in winter. However, during the summer campaign low SOUR values were measured together with good filterability. As mentioned before, this could be explained by the dilution of the raw wastewater due to rain event during one of the samplings. Considering moderate correlation between the SOUR results and influent COD (Figure 5.20), it could have been also caused by the change in organic loading - reflected in the low VSS/TSS ratio - and subsequently different composition of incoming wastewater.

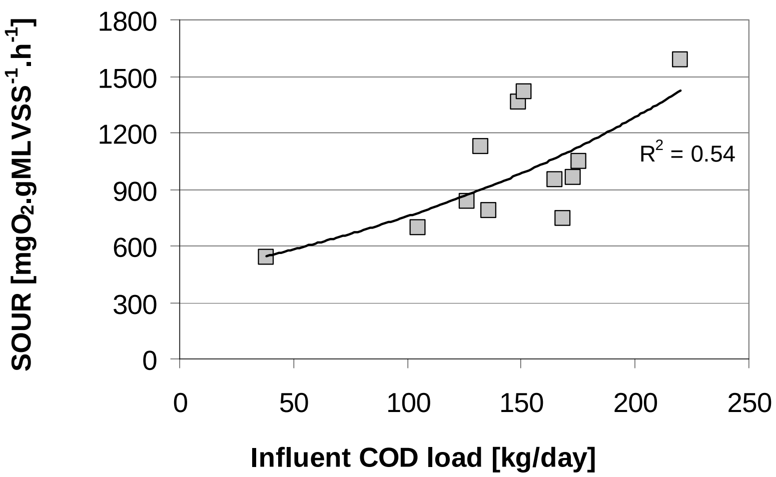

Figure 5.20: Influent SOUR vs. influent COD load

5.5 Summary and conclusions

The effect of seasonal temperature changes on characteristics of the activated sludge and raw wastewater was studied in a full-scale membrane bioreactor treating municipal wastewater. Based on the results discussed in the paper, the following conclusions can be drawn.

- Typical seasonal fluctuations and deterioration of activated sludge filterability, coupled with worsened settling properties, during low temperature periods were observed.
- Filterability of activated sludge is always better that of influent.
- Filterability deteriorated at lower biomass concentrations and with increase in VSS/TSS ratio.
- Filterability is reciprocally correlated with the incoming organic load, as increase in the influent COD loading resulted in filterability deterioration.
- The particles in the range of 10-100 μm had no evident influence on activated sludge filterability. The majority of the constituents was smaller than 1 μm. When compared season-wise, changes in number and volume of submicron particles were linked to the filterability changes.
- Soluble fraction of the raw wastewater contains majority (63 %) of the COD and plays an important role in the membrane resistance increase and filterability deterioration.
- Biological activity of the biomass was decreasing with temperature decrease to reach a minimum during winter period. Deterioration of filterability under low temperatures was linked with slower biodegradation of wastewater in the mixed liquor. The biodegradability of wastewater was worse at low temperatures, except when high amounts of easily biodegradable VFAs were available.

CHAPTER **6**

Impact of activated sludge and influent characteristics on sludge filterability and MBR operation

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6 Impact of activated sludge and influent characteristics on sludge filterability and MBR operation

6.1 Chapter outline

This chapter discusses the influence of both activated sludge and influent characteristics on activated sludge filterability and operation of full-scale MBRs. The results of an extensive survey on activated sludge characteristics affecting sludge filterability are presented in section 6.2. Impact of influent characteristics on sludge filterability and MBR operation is discussed in section 6.3. Section 6.4 deals with operation and performance of full-scale MBRs. The chapter is concluded in section 6.5 with the summary of the results and main conclusions.

6.2 Impact of activated sludge characteristics on filterability

6.2.1 Introduction

Despite extensive research efforts membrane fouling in MBRs is still far from being fully understood (Drews 2010) mainly due to the inherent interactions between activated sludge, influent and applied process conditions. Activated sludge filtration in an MBR is indeed a physical process of which the efficiency is dictated by activated sludge filterability, which, on its turn, is dictated by the interactions between the biomass, the wastewater and the operational conditions. These interdependencies add complexity to the problem of fouling and often lead to conflicting conclusions in literature. As a result, controversial results are commonly encountered, e.g., for the influence of biomass concentration, relative hydrophobicity and EPS on membrane fouling.

6.2.1.1 Biomass concentration or mixed liquor suspended solids (MLSS)

The impact of biomass concentration on membrane fouling has been examined extensively and in the early days of MBRs it was often considered as one of the main foulant parameters. However, with an increasing number of publications, contradictory results are being presented. In the review paper of Le-Clech et al. (2006), it is noticed that some authors report an increasing biomass concentration as a positive impact on membrane fouling, whereas others report a negative correlation while yet another group finds the influence of MLSS insignificant. More recently, also the existence of a critical MLSS concentration of 10.5 g/L is suggested (Lousada-Ferreira et al. 2010). The lack of a clear correlation between MLSS concentration, which is basically the simplest parameter to measure, and any other foulant characteristic indicates that the MLSS concentration alone is a poor indicator for biomass fouling propensity (Le-Clech et al. 2006).

6.2.1.2 Relative hydrophobicity (RH)

Membranes used in MBRs are typically made hydrophilic to improve their water permeability. Low hydrophobicity of sludge flocs is thus expected to cause high fouling due to stronger interactions with the membrane surface. Moreover, a decreasing activated sludge hydrophobicity results in floc deterioration (Liu and Fang 2003) which on its turn again leads to severe membrane fouling (Van den Broeck et al. 2010). In contrast, Meng et al. (2006) found that the RH of activated sludge is positively correlated with membrane fouling.

6.2.1.3 Extracellular polymeric substances (EPS)

EPS have recurrently been labelled as the main cause for membrane fouling in membrane bioreactors. EPS are, by definition, of microbial origin and can be divided into two subclasses, i.e., extractable EPS and soluble microbial products which are bound to the sludge flocs or are free in the bulk solution, respectively. The composition and the amount of EPS, and thus their fouling propensity, are highly dependent on the condition of the activated sludge microbiology. An activated sludge under stress is known to produce more and/or other EPS (Drews et al. 2006). Nonetheless, EPS is generally measured as such and no distinction is made between different fractions or compositions of EPS, e.g., sugars are usually measured as an equivalent of d-glucose according to the method of Dubois et al. (1956). Although some sugars cause more membrane fouling than others (Al-Halbouni et al. 2009, Okamura et al. 2009), most often no further classification is made between different types of polysaccharides. Since the actual fouling relevant fraction of EPS is still unknown, a standard method to measure this fraction does not yet exist (Drews 2010) which leads to conflicting results for the reported impact of EPS on membrane fouling by different authors.

There are two main reasons why results in MBR research are not congruent, i.e., (*i*) there is a lack of standardized methods for crucial analyses, like EPS and all its sub-fractions and (*ii*) most of the studied sludge characteristics are interconnected in one way or another to other sludge characteristics. For example, bound EPS serve as a matrix for microbial aggregates and are as such linked with bioflocculation, and thus with sludge morphology. Furthermore, some components present in the eEPS (e.g., proteins) strongly contribute to sludge floc hydrophobicity. Moreover, according to Drews (2010), depending on the applied method, the group of eEPS overlaps with other polymer fractions. This further stresses the need for (*i*) standardised methods to be used for all MBR-related research and (*ii*) a broader approach for MBR-data interpretation trying to unravel some of the interdependencies of sludge characteristics in relation with membrane fouling. This chapter focuses on the latter aspect.

The goal of this research is to unveil some of the black box features of activated sludge filterability. The obtained relationships between the sludge characterising parameters can help in pinpointing the main causes of fouling and can assist in finding the correct remedial actions to improve process efficiency.

6.2.2 Experiments and MBR plants description

Ten different MBRs in Belgium and the Netherlands were selected for the sampling campaign. The selected set of MBRs includes plants treating municipal and a variety of industrial wastewaters, of different scales and equipped with different membrane types (Table 6.1).

Table 6.1: Type of wastewater, scale and membranes of the sampled MBRs

#	Location	Feed water	Scale	Type of membrane module (pore size, manufacturer)
1	Heenvliet	Municipal	Full-scale	Flat sheet (0.08µm, Toray)
2	Tervuren	Municipal	Full-scale	Flat sheet (0.4µm, Kubota)
3	Schilde	Municipal	Full-scale	Hollow fiber (0.04µm, Zenon)
4	KU Leuven	Municipal	Pilot-scale ($15m^3$)	Flat sheet (0.08µm, Toray)
5	Boortmalt	Industrial – food	Full-scale	Hollow fiber (0.04µm, Koch – Puron)
6	Rendac	Industrial – rendering	Full-scale	Hollow fiber (0.04µm, Zenon)
7	Agristo	Industrial – food	Full-scale	Hollow fiber (0.04µm, Koch – Puron)
8	Undisclosed	Industrial – chemistry	Pilot-scale ($9m^3$)	Flat sheet (0.04µm, Microdyn-Nadir)
9	Kloosterboer	Industrial – food	Full-scale	Hollow fiber (0.4µm, Mitsubishi – Sterapore)
10	Cargill	Industrial – food	Full-scale	Micro-tubular (0.03µm, Norit – X-flow)

All MBRs were sampled twice, both in winter and summer, except MBR Cargill, which was only sampled in winter due to technical problems during the summer sampling campaign. A sample of 30 L was taken directly from the membrane tank, whenever possible. If this was not feasible, the sample was taken from the recirculation flow from the membrane tank back to the bioreactor, i.e., the carbon and/or nutrient removal tank. The sludge samples were placed in the sludge reservoir of the DFCm installation within two hours after sampling. According to the DFCm, the sludge reservoir is continuously aerated to ensure aerobic conditions (as in the membrane tanks) and to keep the sample well-mixed during the course of the experiment. Prior to the start of each experiment, a 1 L sample was taken from the DFCm reservoir for all other sludge analyses, to ensure the exact same sample composition and condition.

6.2.2.1 Delft Filtration Characterization method (DFCm)

The Delft Filtration Characterization method was exploited to determine the filterability of activated sludge samples. A detail description of the DFCm can be found in section 3.3.

6.2.2.2 Activated sludge characteristic and analytical methods

Activated sludge samples were subjected to a large set of standardized measurements, previously described in detail in *Chapter 3*. The selection of parameters was based on state-of-the-art literature on membrane fouling in MBR. All samples were analysed in terms of sludge composition by image analysis (section 3.6.9), relative hydrophobicity (section 3.6.10), floc surface charge (section 3.6.11), floc stability (section 3.6.12) and EPS (section 3.6.7). Furthermore, MLSS, MLVSS and TOC in permeate and supernatant were measured in accordance with the procedures described in section 3.6.3.

6.2.3 Results and discussion

The influence of biomass concentration, relative hydrophobicity, sludge morphology, EPS, sludge dissociation constant being a measure for floc stability, TOC and surface charge on sludge filterability (ΔR_{20}) was studied. The results are discussed below.

6.2.3.1 DFCm results

The DFCm results on activated sludge filterability are presented in Figure 6.1. There is a large variation between the sludge filterabilities in the different plants. If results are compared plant-wise, in general, summer sludge samples exhibit better sludge filterability compared to winter samples, with the exception of MBRs 7 and 8. Given that results from the DFCm are corrected for temperature, this seasonal variation exceeds the mere temperature effect on viscosity.

The presented results, and especially those for the municipal installations 1 to 4, confirm the observation of several authors that a seasonal decrease in reactor temperature negatively affects membrane fouling rates (Rosenberger et al. 2006, Drews et al. 2007, Lyko et al. 2008, Miyoshi et al. 2009, Moreau 2010).

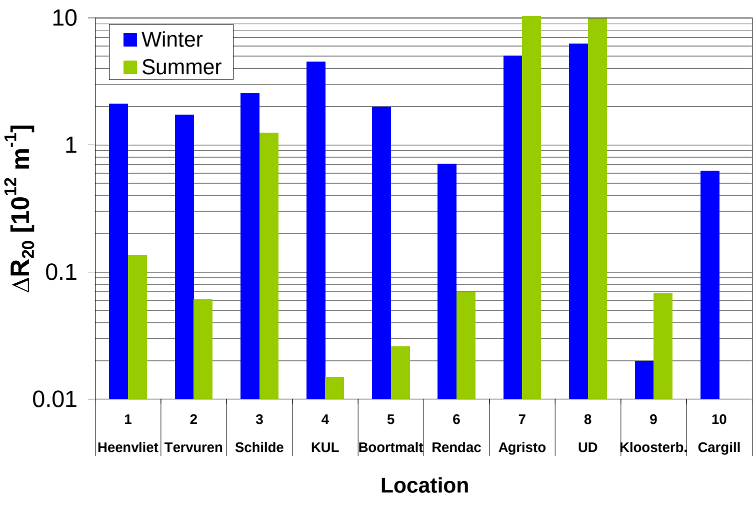


Figure 6.1: Activated sludge filterability results, expressed as ΔR_{20}, from the DFCm

No correlation could be found between the membrane configuration and activated sludge filterability, which is in accordance with the findings of Moreau (2010). The dissimilarities are evidently due to different operational conditions applied in the different plants, e.g., type of wastewater, HRT, SRT, seasonal period, etc.

6.2.3.2 Biomass concentration

MLSS and MLVSS results are listed in Table 6.2. The ten plants cover a wide variety of biomass concentrations, ranging from 4.6 gMLSS/L to 19.4 gMLSS/L. Plant-wise comparison indicates that most of the sampled MBRs operate at a constant biomass concentration.

The MLSS increase in MBR 5 is due to an urgent process water requirement at this location; a surplus of permeate was extracted from the sludge to be used in the production process. Nonetheless, the sludge filterability was good at this location. These results confirm the controversy about the impact of biomass concentration on sludge filterability; also in this sampling campaign, the results of MLSS and MLVSS could not directly be correlated to sludge filterability (Figure 6.2).

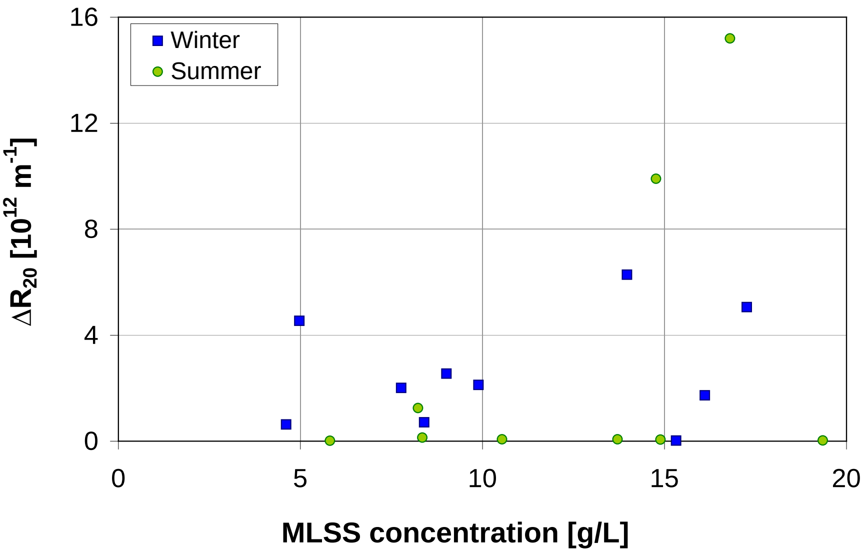


Figure 6.2: Activated sludge filterability versus MLSS concentration

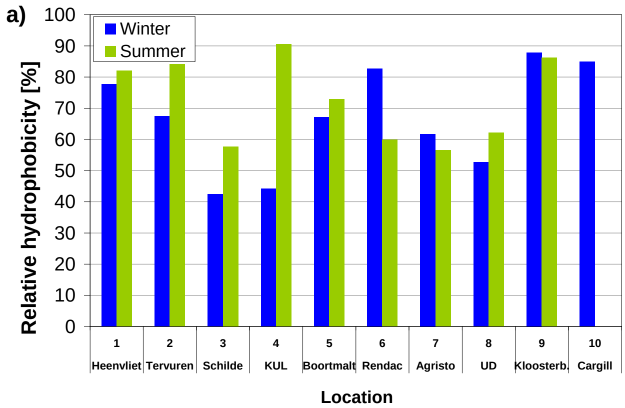
Table 6.2: Summary of the results from activated sludge characteristics

Location		MLSS	MLVSS	SC	DC[a]	TOC_{sup}[b]	TOC_{perm}[b]	SMP PS	SMP PN	eEPS PS	eEPS PN
1	Winter	9.89±0.11	7.45±0.12	-0.453±0.023	0.0113	40.0	7.9	1.15±0.07	2.74±0.23	26.52±1.56	44.26±0.59
	Summer	8.35±0.01	6.07±0.01	-0.373±0.162	0.0041	30.9	8.8	0.30±0.20	0.68±0.25	22.93±1.55	40.76±0.25
2	Winter	16.11±0.12	9.00±0.05	-0.347±0.201	0.0191	36.0	17.3	0.83±0.00	2.69±0.07	19.41±1.29	33.97±0.44
	Summer	14.89±0.10	6.72±0.04	-0.400±0.069	0.0074	15.5	5.8	0.19±0.01	0.21±0.02	23.50±0.01	36.00±0.11
3	Winter	9.01±0.09	6.18±0.08	-0.293±0.122	0.0158	33.7	11.1	1.68±0.01	2.09±0.27	34.89±0.86	71.68±0.74
	Summer	8.23±0.10	5.21±0.06	-0.720±0.106	0.0141	40.3	18.3	1.28±0.11	1.14±0.34	25.93±0.02	48.04±0.37
4	Winter	4.97±0.01	3.21±0.01	-0.507±0.115	0.0550	40.7	21.1	10.21±1.05	6.00±0.09	23.21±0.78	68.01±2.61
	Summer	5.81±0.04	3.41±0.01	-0.307±0129	0.0044	16.3	8.2	1.18±0.24	0.74±0.29	18.23±0.31	43.86±0.27
5	Winter	7.77±0.14	6.85±0.14	-0.707±0.266	0.0757	134.9	71.0	8.87±0.44	27.66±0.80	36.88±0.22	40.92±0.60
	Summer	19.35±0.45	16.80±0.40	-0.773±0.122	0.0599	219.2	93.5	5.81±0.36	12.36±0.08	14.72±0.04	25.54±0.46
6	Winter	8.40±0.04	7.22±0.02	-0.587±0.180	0.0608	55.3	27.2	2.17±0.17	6.89±0.25	21.76±4.31	36.23±0.21
	Summer	10.54±0.21	8.62±0.16	-0.573±0.295	0.0694	55.5	29.9	1.98±0.13	4.86±0.12	24.20±0.18	28.42±0.20
7	Winter	17.26±0.06	7.96±0.02	-0.453±0.061	0.0610	55.2	15.3	2.90±0.07	3.85±0.84	15.70±0.47	35.68±0.25
	Summer	16.80±0.36	6.05±0.11	-0.813±0.092	0.1051	102.5	13.2	15.28±0.27	6.59±0.10	17.94±0.33	23.18±0.77
8	Winter	13.97±0.12	10.00±0.07	-0.200±0.106	0.0625	142.9	48.6	2.77±0.02	7.58±0.06	22.09±0.43	47.82±0.49
	Summer	14.77±0.25	10.60±0.18	-0.653±0.083	0.2642	284.7	51.1	16.19±0.18	17.29±0.21	21.71±0.37	36.13±0.25
9	Winter	15.32±0.09	14.30±0.07	-0.347±0.046	0.0505	119.0	36.2	6.71±0.44	2.98±0.03	17.71±0.07	20.24±0.11
	Summer	13.71±0.14	12.60±0.13	-0.653±0.046	0.0197	40.3	18.3	0.65±0.01	0.83±0.07	19.37±0.02	19.65±0.17
10	Winter	4.61±0.04	4.53±0.02	-0.427±0.101	0.0501	131.2	24.3	30.34±0.27	5.85±0.23	22.97±0.19	30.43±0.24
Unit		g/L	g/L	meq/gMLSS	Abs/washing step	mgC/L	mgC/L	mg/gMLVSS	mg/gMLVSS	mg/gMLVSS	mg/gMLVSS

[a]R^2 of slope > 0.97, [b]maximum allowed standard deviation 2%

6.2.3.3 Relative hydrophobicity

The RH results are presented in Figure 6.3a. In majority of cases, RH is higher in summer samples, especially for MBRs that are subjected to varying influent temperatures, i.e., MBRs treating municipal wastewater: 1, 2, 3 and 4. For the majority of the sampled MBRs, sludge filterability and RH appear to be positively correlated as was expected. Very likely hydrophobic particles interact less with the hydrophilic membrane. However, the RH-value alone renders inconclusive information on the filterability for some samples, e.g., the RH for MBR 6 decreases from 83% in winter to 60% in summer but sludge filterability improves as is reflected by a ΔR_{20} decrease from $0.71 \cdot 10^{12}\,m^{-1}$ to $0.07 \cdot 10^{12}\,m^{-1}$. In the case of MBR 8, the summer sludge sample is more hydrophobic with an RH of 62% versus 53% in winter, although ΔR_{20} increased from $6.3 \cdot 10^{12}\,m^{-1}$ to $9.9 \cdot 10^{12}\,m^{-1}$. For the majority of the sampled MBRs, filterability and RH were rather weakly negatively correlated, however, the RH-value alone does not provide conclusive information on filterability as the determination coefficient R^2 was 0.29. It is obvious that with RH (Figure 6.3b) and biomass concentration (Figure 6.2) as sole input parameters, sludge filterability cannot be estimated.



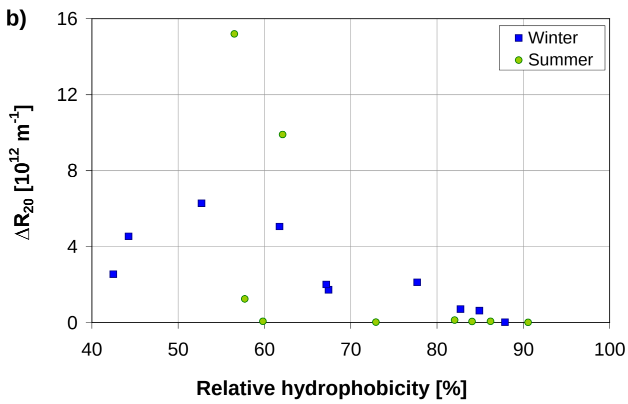


Figure 6.3: (a) Relative hydrophobicity of activated sludge samples. (b) Activated sludge filterability versus relative hydrophobicity

6.2.3.4 Sludge morphology

Sludge morphology was monitored by means of an automated microscopic image analysis procedure, the results are summarized in Figure 6.4a and Figure 6.5a. Since it has been demonstrated that deflocculation deteriorates sludge filterability (Meng and Yang 2007, Van den Broeck et al. 2010), two parameters are selected to represent the sludge flocculation state, i.e., activated sludge mean particle surface A_{mean} and the 1% pixel value being the surface fraction of sludge particles equal to 1 pixel.

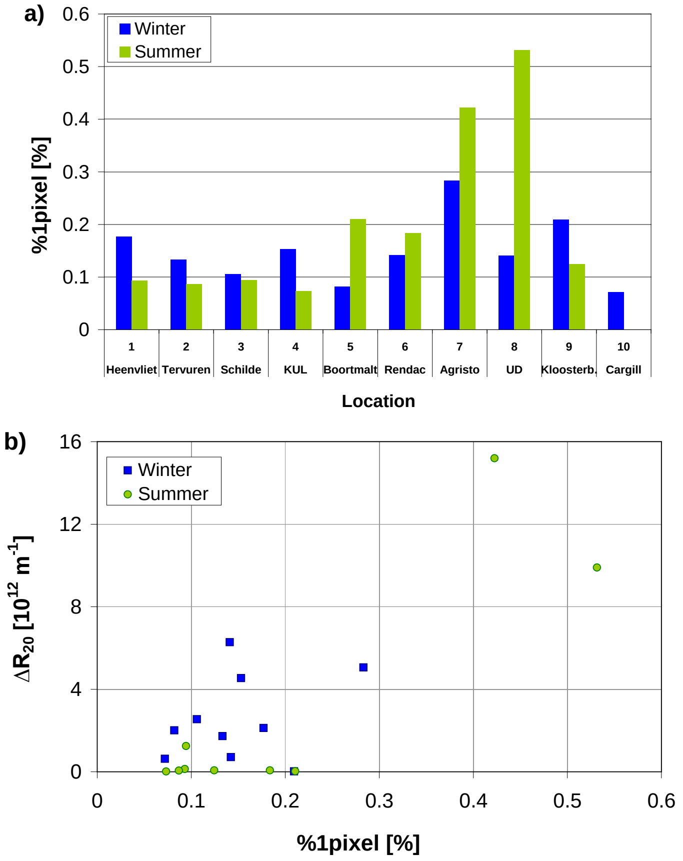


Figure 6.4: (a) Surface fraction of activated sludge particles equal to 1 pixel. (b) Activated sludge filterability versus %1pixel

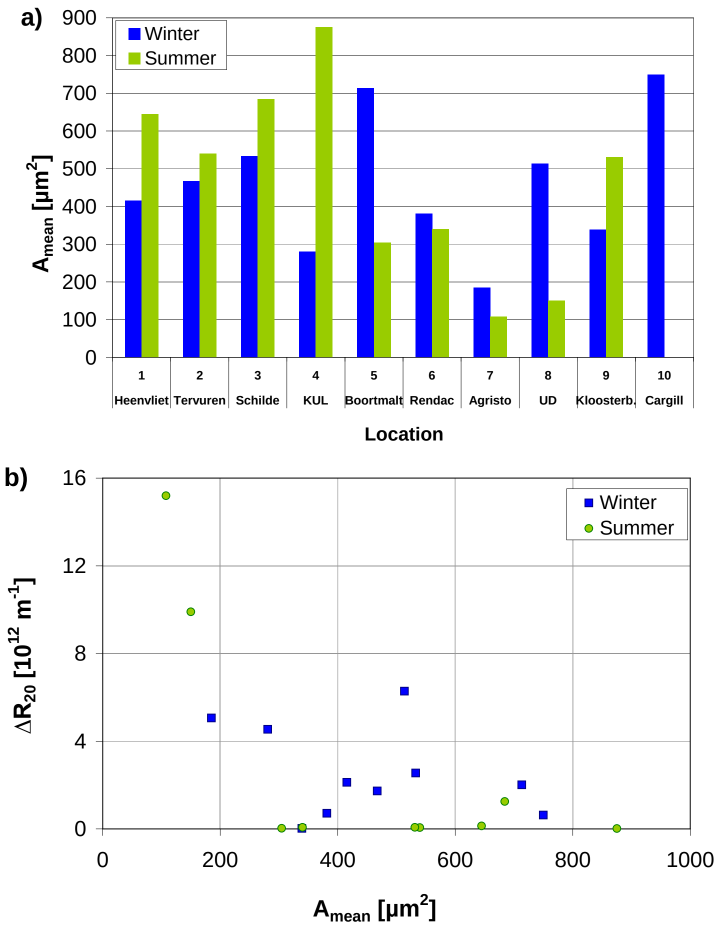


Figure 6.5: (a) Activated sludge mean particle surface. (b) Activated sludge filterability versus mean particle size

The results presented in Figure 6.4a and Figure 6.5a confirm the link between (de)flocculation and filtration performance. E.g., for MBRs 7 and 8 a sludge deflocculation event, reflected by a decrease in A_{mean} and an increase in %1pixel, results in a dramatic deterioration of sludge filterability whilst maintaining similar levels of MLSS. In the case of MBR 8 specifically, sludge morphology parameters explain the deteriorated filterability which, in this case, completely overruled the effect of increased sludge hydrophobicity. Although results confirm the link between deflocculation and filterability deterioration, activated sludge filterability cannot be accurately estimated (R^2=0.62) based on sludge morphology alone (Figure 6.4b and Figure 6.5b).

6.2.3.5 Filterability prediction

Filterability prediction is possible in two ways: (*i*) for primary classification by combining relative hydrophobicity and sludge morphology and (*ii*) for accurate estimation by including extra measurements.

On the basis of the above presented results, it is proposed to combine relative hydrophobicity and sludge morphology into a parameter t, as such yielding reliable information for a rough first estimation of the sludge filterability into two classes, i.e., (*i*) bad sludge filterability and (*ii*) poor to good filterability (Figure 6.6). The parameter t is defined as:

$$t = \frac{\%1pixel}{RH}$$

t > 0.00325 → bad sludge filterability (ΔR_{20} > 3.0)
t < 0.00325 → poor to good filterability (ΔR_{20} < 3.0)

Plotting of the t-parameter against filterability results divides the graph into four sections. The results of this study are placed exclusively in sections 2 and 3 which are defined as t>0.00325, ΔR_{20}>3.0 and t<0.00325, ΔR_{20}<3.0, respectively. Sections 1 and 4, respectively, are defined as t>0.00325, ΔR_{20}<3.0 and t<0.00325, ΔR_{20}>3.0, and have no associated samples. Therefore, the combination of sludge morphology, i.e., %1pixel, and relative hydrophobicity marks a clear threshold for two filterability classes, i.e., good to poor (Section 3) and bad (Section 2) as presented in Figure 6.6.
The theory behind this observation is relatively simple, e.g., small sludge flocs are dramatic for sludge filterability (Van den Broeck et al. 2010), however, if these small flocs are very hydrophobic they tend to foul the hydrophilic surface of the membrane less. Similarly, robust sludge flocs improve sludge filterability, however, if these sludge flocs are very hydrophilic, they interact more with the membrane surface. The latter effect is hardly ever observed, since hydrophilic sludge particles do not flocculate well. Evidently, the proposed threshold value can be refined when additional data are gathered.

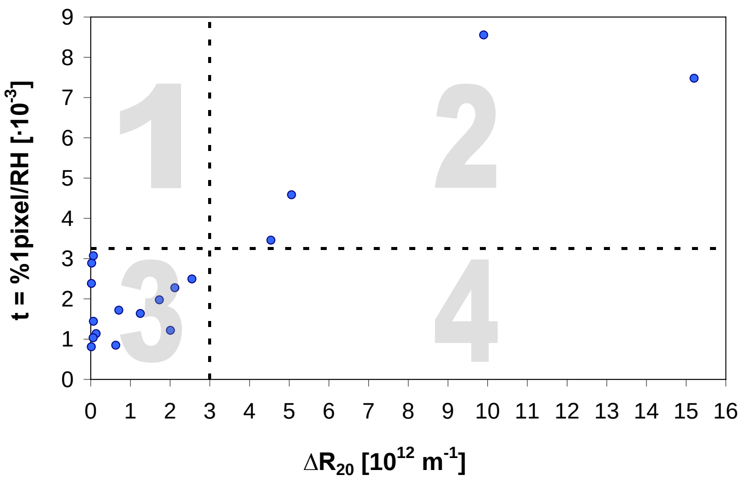


Figure 6.6: Parameter t versus activated sludge filterability (ΔR_{20}). The combination of sludge morphology and relative hydrophobicity marks a clear threshold for two filterability classes, i.e., bad and poor to good

Indicator t permits a classification of poor to good and bad sludge filterability, however, it does not allow a more accurate estimation of the actual sludge filterability value, i.e., ΔR_{20}. The difference between poor, moderate and good cannot be explained by hydrophobicity and sludge morphology alone, more detailed measurements are needed. Therefore, apart from RH and image analysis, all samples were analyzed for eEPS (PN and PS), SMP (PN and PS), TOC in supernatant and permeate, floc surface charge and floc stability (Table 6.2).

To correlate the activated sludge characteristics (input) with the activated sludge filterability (output), a multiple linear regression technique, Partial Least Squares (PLS), was exploited. For sludge samples with a good to poor filterability, i.e., $\Delta R_{20}<3.0$, PLS approach was used to infer a correlation between ΔR_{20} and above listed sludge characteristics in order to predict activated sludge filterability. The model proposed by Van den Broeck et al. (2011) and Van den Broeck (2011) was able to accurately estimate sludge filterability based on eEPS PN, SMP PS, sludge morphology, the dissociation constant and RH. It is important to notice that some of the variables are calculated through a combination of other measurements. The set of essentially needed measurements is, therefore, not that extensive, i.e., SMP PS, eEPS PN, RH, DC and image analysis.

For a given MBR activated sludge sample, the obtained relationships could assist in pinpointing the main causes of fouling and might help in implementing the correct remedial actions to improve activated sludge filterability.

6.2.3.6 Practical implications for MBR operation

As was mentioned in paragraph 6.2.3.1, the sludge filterability of municipal samples is better in summer, compared to samples taken in winter. According to Rosenberger et al. (2006) and Drews et al. (2007) this is probably due to an increased SMP concentration in the mixed liquor supernatant during winter conditions. Our results confirm that SMP PS negatively impact activated sludge filterability, i.e., all municipal MBRs have lower SMP PS concentrations in summer compared to winter. However, these summer samples also indicate an improved sludge flocculation, i.e., sludge morphology and hydrophobicity. Also EPS PN, which contribute to the structural integrity of the flocs and to hydrophobicity, as well as the sludge dissociation constant, i.e., a measure for the stability sludge flocs (Zita and Hermansson 1994, Liao et al. 2002), influence the filterability of activated sludge. It can thus be concluded that the bioflocculation state of activated sludge is a major indicator for activated sludge filterability, not only for municipal MBRs, but for industrial MBRs as well.

To improve the filterability of activated sludge samples with $\Delta R_{20}>3.0$ (bad) it is important to enhance flocculation and increase hydrophobicity. Commercial flocculants can be used as a fast and easy remedy (Huyskens 2012), whilst adjusting process and environmental conditions to favor good bioflocculation and increase sludge hydrophobicity to avoid the excessive cost of constant flocculant dosing. The increase in sludge hydrophobicity can be achieved by increase in SRT, addition of easily assimilable substrates, e.g., glucose (Jorand et al. 1994), and/or at the presence of multivalent cations, e.g., by introduction of cationic polymers (Liu and Fang 2003).

6.3 Impact of inflow characteristics on sludge filterability and MBR operation

Seasonal changes in the composition of raw domestic wastewater and their impact on activated sludge filterability were previously discussed in *Chapter 5*. However, the characteristics of raw wastewater change not only between the seasons, but are also subjected to continuous, e.g., hourly or/and daily, fluctuations. In consequence, the quality of the feed water, reported to be a dominant factor in activated sludge filterability (Moreau 2010), will change. According to Evenblij et al. (2005a) the occurrence of daily fluctuations in filtration behavior is ascribed to changes in influent quantity and quality. In addition, organic loading plays an important role in membrane fouling and sludge filterability (Cho et al. 2005, Ivanovic et al. 2006, Meng et al. 2009, Zhang et al. 2010).

In another research work, Evenblij et al. (2005b) observed variations in the filterability as a result of sudden changes in the influent conditions. Moreover, a continuous deterioration of the filterability with the increasing addition of a carbon source was reported. Furthermore, according to Geilvoet (2010), continuous variation in the flow rate, temperature and/or composition of influent flow may lead to stress conditions, impacting activated sludge quality. Subsequently, the non-stabilised operational and biological circumstances are causing deterioration of filterability.

In literature, the influence of hydraulic loading and temperature were hardly discussed (Syed et al. 2009, van den Brink et al. 2011). Therefore, the research work described in this section (6.3), aims to find a relation between filterability of the activated sludge and the inflow characteristics. In the following sub-sections the effect of influent flow rate, temperature and composition on the activated sludge filterability is discussed.

6.3.1 Hydraulic loading rate effect

Variations in feedwater flow rate, and thus changes in hydraulic load, can cause non-stabilised operation of the MBR (Judd 2011). As distinct from industrial installations, municipal MBRs are exposed to storm flows which change the hydraulic loading, decrease concentrations and can create hydraulic shocks. Short hydraulic shock periods are usually compensated by operation at higher fluxes (up to double the design flux). However, those periods of increased flux should not be longer than 1-2 hours and often needs to be followed by a relaxation period. Therefore, long or prolonged periods with high influent flow rate might be problematic for MBR operation due to potential increment of membrane fouling.

In order to assess the impact of hydraulic shocks on activated sludge filterability, development of the filterability during periods of elevated flow rates was investigated. The MBR Varsseveld is designed for a treatment capacity of 250-300 m^3/h during dry weather flow and is able to treat the flows up to 755 m^3/h during rain weather flow. MBR Varsseveld is connected to a combined sewer system and has a stand-alone configuration. Therefore, incoming stormwater is treated exclusively by the MBR. This provided good conditions for the hydraulic load effect investigation.

The first period suitable for detailed analysis has taken place at the time of ‘winter 2009’ campaign experiments. Due to heavy storm and intensive rainfall the influent flow had increased more than 5 times from a daily average of 130 m^3/h to nearly 700 m^3/h within an hour and reached maximum capacity within 2.5 hours. Before the hydraulic shock had taken place, the activated sludge originating from the membrane tank was considered to have poor filterability and ΔR_{20} of $3.5 \cdot 10^{12}$ m^{-1}. At first, filterability deteriorated during the peak flow, improved after the first peak has passed, was followed by another deterioration of activated sludge filterability after hydraulic load became average again, to finally improve in the coming day (Figure 6.7). Further filterability improvement was not seen probably due to another, yet smaller, rainfall period increasing hydraulic load again. The filterability improvement observed after the first peak also might be attributed to the dilution effect caused by heavy rainfall. The measured changes in filterability are likely caused by the hydraulic load change but also by a snow melt and associated temperature drop of the influent. The temperature and snow melt aspect is discussed in section 6.3.2. Nevertheless, a similar pattern of filterability changes during the period of increased hydraulic loading was observed for samples originating from the aerobic, anoxic and other membrane tanks (Figure 6.7b). We may conclude that distinct changes in the impact of hydraulic loading clearly impacts activated sludge filterability. This is in accordance with Trussell et al. (2006) and Trussell (2008) who found that deflocculation of sludge during wet weather flows caused an increase in colloidal content in mixed liquor and resulted in severe membrane fouling. Furthermore, the change in hydraulic load rate was a reason of experienced hydraulic shock that in result affected the membrane operation. The encountered operational problems will be discussed in the next section (6.3.2).

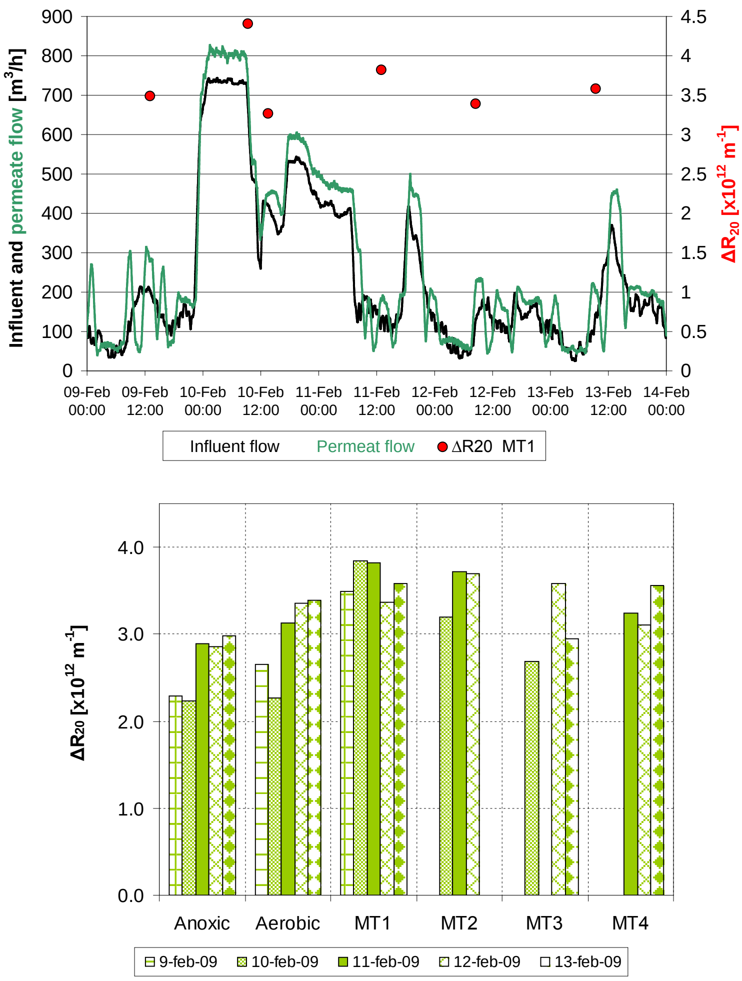


Figure 6.7: (a) Influent and permeate flows versus ΔR_{20} values for membrane tank samples and (b) corresponding activated sludge filterability in various compartments of MBR Varsseveld during 'winter 2009' campaign

Another intensive rainfall period, allowing studying the effect of hydraulic load, was analysed in the course of 'winter 2010' experimental campaign. During this rainfall period, influent flow had increased more than 3 times from 150 m^3/h and reached instantaneously, almost the maximum hydraulic capacity, i.e., values above 700 m^3/h within an hour. The average flow rate during the entire period of hydraulic peak was about 500 m^3/h. The first filterability measurements were performed during the treatment of a storm flow. Therefore, the quality of activated sludge before the period of excessive hydraulic load could not be assessed. Nevertheless, filterability development tendency is straightforward. The worst, yet classified as moderate, sludge filterability was measured during the period of increased flow rate. After the hydraulic peak has passed a continuous improvement in filterability was observed. At first, the filterability improved slowly, whereas in the next 3 days it has improved at a higher rate, reaching a ΔR_{20} of $0.58 \cdot 10^{12}$ m^{-1} (Figure 6.8a). A similar filterability development pattern was observed in all compartments of the MBR (Figure 6.8b). It is of course possible that the filterability simply improved despite the storm flow but it is rather unlikely given the results of the 'winter 2009' campaign. Concluding, also in this case, the impact of hydraulic loading on activated sludge filterability was distinct. However in this case, the positive influence of decreasing hydraulic loading rate on sludge filterability was clearly observed.

The operation of the MBR was not affected seriously during this rainfall event. The fluxes increased from average 24 L/m^2·h up to 44 L/m^2·h to compensate higher influent flow rates. Permeability had improved from 120-140 L/m^2·h·bar to 160-210 L/m^2·h·bar between the periods of elevated and regular hydraulic loads, respectively. Further permeability improvement to values of 180-230 L/m^2·h·bar was achieved after maintenance cleaning was performed.

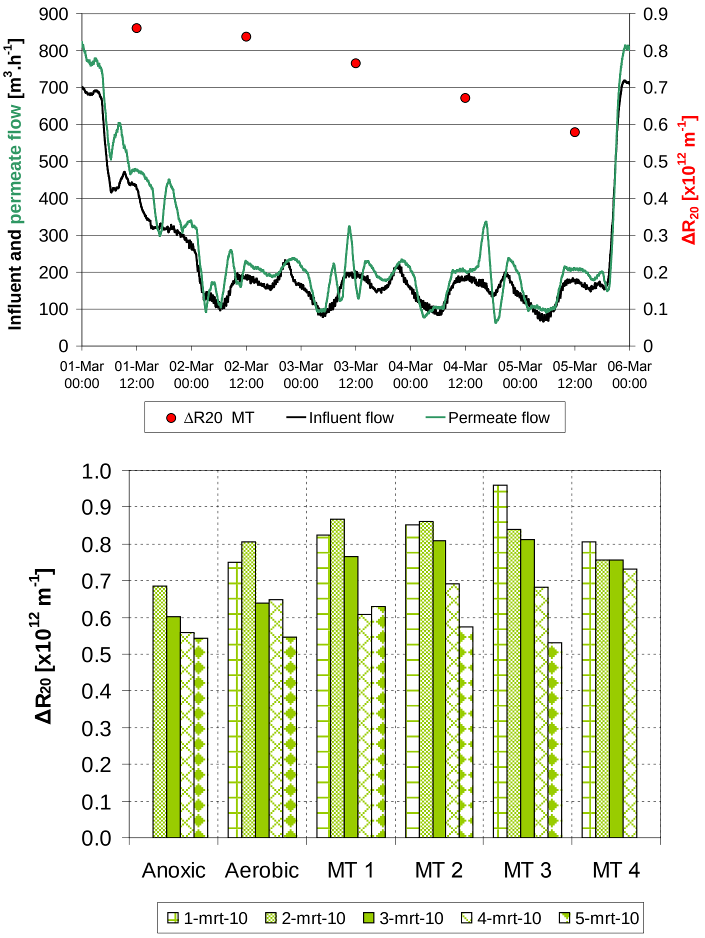


Figure 6.8: (a) Influent and permeate flows versus ΔR_{20} values for membrane tank samples and (b) corresponding activated sludge filterability in various compartments of MBR Varsseveld during 'winter 2010' campaign

In Heenvliet and Ootmarsum MBRs excessive rain weather flow conditions were not encountered. This is mainly due to the fact that experiments were performed during relatively dry periods but also due to hybrid configuration of the plants. Hybrid configuration provides flexibility and allows to buffer or to divide the incoming flow between the MBR and CAS system. For example, Heenvliet MBR can be operated in series or parallel to the CAS system. Both of the configurations are reducing a chance of experiencing the hydraulic load. During in series operation, the MBR is fed with activated sludge from the CAS. Thus, the CAS system acts as a hydraulic buffer and the MBR is less sensitive to hydraulic flow fluctuations. In parallel operation, MBR treats a fixed part, approximately 25%, of the total inflow to the WWTP.

6.3.2 Temperature effect

As has been already discussed in *Chapter 4* and *5*, temperature has a significant influence on activated sludge filterability, seasonal filterability fluctuations and wastewater composition. But temperature and rapid temperature changes in particular, can also demand changes in operation of an MBR plant. The negative impact of quickly decreasing temperature was observed during the 'winter 2009' campaign at the MBR Varsseveld.

During the night between 9th and 10th of February, a rapid temperature drop of 7°C within 7 hours had occurred, seriously affecting the activated sludge filterability (Figure 6.9).

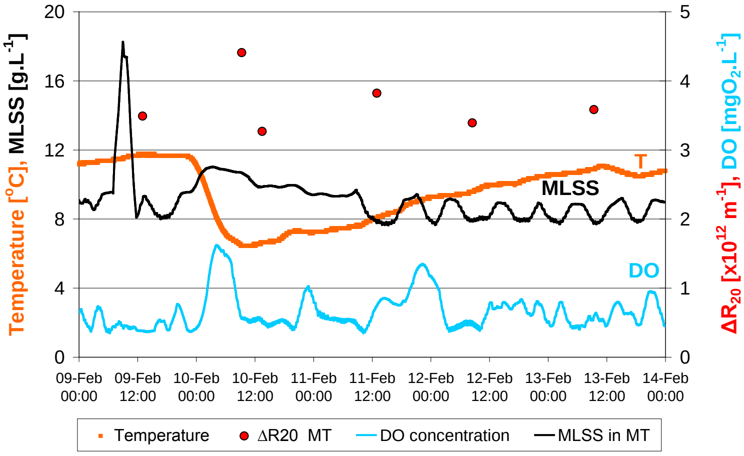


Figure 6.9: Temperature, MLSS, dissolved oxygen profiles and ΔR_{20} values measured at MBR Varsseveld during 'winter 2009' campaign

This abnormal situation was caused by the heavy storm event that happened after a prolonged dry and cold period with snow cover. Consequently, melting snow was transported to the plant together with fresh and cold rainwater. As a result, temperature of incoming wastewater had dropped from about 12°C to 6°C and hydraulic load increased from average 160 m^3/h to quickly reach maximum capacity of 750 m^3/h and last for about 12 hours (Figure 6.7). Furthermore, it is possible that a large amount of salt was present in the influent. The road salt, i.e., sodium chloride, used to de-ice roads at that period and dissolved in melt water was also transported to the plant. The impact of road salt on activated sludge filterability was investigated in the sub-study and is discussed in the following section 6.3.3. Thus, when the cold and possibly salty inflow reached the treatment plant bioreactor, the microorganism population and their activity were likely to be affected. In effect, activated sludge filterability deteriorated in each section and the permeability achieved at the plant was approximately 50% lower than usually (Figure 6.22). The actual permeability decreased from about 200-230 L/m^2·h·bar under undisturbed operation circumstances to values below 100 L/m^2·h·bar during the storm event. At that time, the applied flux had increased from typical 20±6 L/m^2·h to about 35±8 L/m^2·h. Moreover, at high flows, the MLSS concentration in the membrane tanks in MBR Varsseveld is higher than at dry weather flow, due to the lower ratio of permeate and recirculation flows. This might also contribute to the permeability drop at the heavy storm event.

Furthermore, according to the operators, the weekend before the temperature drop had occurred, an increase in MLSS concentration was observed at the plant. The MLSS concentration in the aerobic tank increased from the regular 8.0 g/L on 5th of February up to 13.0 g/L on 8th of February (results not illustrated graphically). Subsequently, concentrations of up to 19 g/L were measured in the membrane tanks (Figure 6.9). This concentration increase is connected with the incoming influent characteristic. In the neighborhood of the WWTP, the ditches alongside roadways were cleaned and accumulated materials from the ditches were removed to provide proper patency and drainage capacity. Subsequently, this mainly organic material was discharged into the sewerage, feeding the plant with more concentrated wastewater, increasing MLSS in the MBR and probably caused an organic overloading condition for the activated sludge microbial community. The high activated sludge loadings could have negatively affected membrane performance (Meng et al. 2009). Moreover, an improvement in sludge quality/filterability under low organic loadings was reported by others (Ivanovic et al. 2006, Moreau 2010). The positive impact of reduced loading on filterability was more visible in summer than in winter (Moreau 2010). On the other hand, Zwickenpflug et al. (2009) did not observe significant effects of organic loading changes on filtration performance compared to seasonal fluctuations. Apparently, seasonal temperature changes or unexpected events have stronger influence on filterability than the sludge loading. Therefore, as previously stated by Moreau (2010), activated sludge loading likely affects sludge filterability but should not be considered a predominant parameter with respect to filterability.

In addition, at that time the oxygen concentration in the aerobic tank was low reaching values of about 0.4 mg/L (Figure 6.9). Prolonged low DO concentrations are reported to be detrimental for activated sludge filterability due to deflocculation processes leading to excessive growth of filamentous bacteria and release of fine particles (Wilén et al. 2000, Gil et al. 2011b).The low DO concentration was most likely a consequence of increased oxygen uptake by the aerobic bacteria consuming available and excessive organic material. During the rest of investigated period, oxygen concentration was in the range of 0.5-1.0 mg/L, while in the comparable period in the previous year the concentration was in the range of 1.0-1.5 mg/L.

Therefore, the combination of unpredictable and uncontrollable circumstances, e.g. heavy rain fall, quick snows melt and organic load increase, likely affected activated sludge filterability and caused serious operational problem for the MBR. In contrast, during the ‘winter 2010’ period, aforementioned circumstances did not occur at the same time and MBR operation was not disturbed.

6.3.3 Influent composition effect

6.3.3.1 Salinity

As already discussed in section 6.3.2 it is hypothesized that road salt could have a detrimental effect on activated sludge filterability. In literature, contradicting results are presented in respect to the impact of salt concentrations on flocculation. While some researchers observed a negative impact (Kincannon and Gaudy 1968, Ng et al. 2005), others found no effect (Wilén et al. 2004) and some reported positive influence of salt concentration, most likely due to reduction of electrical surface charge between the particles, enhancing flocculation (Takeda et al. 1992). Furthermore, as pointed out by Yang et al. (2012), Na^+ may act as a fouling promoter but also as a cleaning agent. Filtration deterioration could be ascribed to ion exchange with the floc sustaining matrix, causing the microbial flocs to disrupt (Evenblij et al. 2005b). It has also been reported in the literature, that high concentrations of salts can affect membrane performance (Reid et al. 2006, Sun et al. 2010), increase fouling propensity (Sun et al. 2010, Remy et al. 2011), disturb biotreatment processes (Kincannon and Gaudy 1968, Lay et al. 2010) and consequently reduce water quality (Reid et al. 2006, Artiga et al. 2008, Reid et al. 2008, Sun et al. 2010). However, the impact of road salt on activated sludge characteristic and on filterability in particular has been largely overlooked in the past. Therefore, in order to elucidate potential impact of the road salt on activated sludge filterability the following sub-study was carried out.

An activated sludge sample of 30 L was taken directly from the membrane tank of the nearby MBR Heenvliet. After immediate transportation, samples were placed in continuously aerated DFCm container. The DFCm was used to measure the activated sludge filterability. A detail description of DFCi and DFCm is provided in *Chapter 3* section 3.3. Prior to the filterability measurement, a sludge sample was taken to perform further sludge analyses. The procedure of MLSS, MLVSS and DSVI analysis is presented in *Chapter 3* section 3.6. The conductivity was measured with the multi-meter (WTW 340i) equipped with a conductivity measuring cell (WTW TetraCon 352).

A road salt sample used during experiments was exactly the same salt used to de-ice roads and was provided by courtesy of the Rijkswaterstaat, an agency responsible for maintenance of the Dutch roads.

Typically, a raw sewage has a salinity of about 0.01% (Kincannon and Gaudy 1966) and sewage following road gritting of about 0.2% (Ludzack and Noran 1965). The concentration of salt in sea water is about 3.3-3.7% (Stewart et al. 1962). The waters

with a salt concentration above 1% are considered to have high salt concentration (Reid et al. 2006).
After performing a reference filterability test without addition of road salt, 10 g of salt was diluted in hot tap water and added to the activated sludge sample. The road salt was added to create following salt content and corresponding concentrations of road salt in activated sludge sample: 10 g (0.03%), 30 g (0.10%), 50 g (0.16%), 150 g (0.48%) and 400 g (1.27%). Corresponding salt injections were also diluted in hot tap water. The cumulative volume of tap water used to dissolve road salt was less than 0.7% of the total sludge volume. After initial manual mixing to distribute salt in the sludge sample, continuous aeration keeps the sample well-mixed during the course of the experiment.
Although the road salt was the same salt as used to de-ice road, unfortunately, the exact composition was unknown. However, sodium chloride is considered the main constituent of the road salt as this is generally used to de-ice pavements and roads in the Netherlands (de Groot 2008). Nevertheless, to precisely determine the salt composition the X-Ray powder Diffraction (XRD) and semi-quantitative X-ray Fluorescence (XRF) analysis were performed by Ruud Hendrikx from the Department of Materials Science and Engineering of the Delft University of Technology.
From both analyses it could be concluded that the road salt sample is rather pure with 98 weight% sodium chloride (NaCl) confirming previous assumption. Apart from potassium, calcium and magnesium, together 1.8 weight% of total mixture, no other compounds in significant quantities were detected. See the Appendix A for detailed results. The measured XRD pattern is presented in Figure A.1. The XRF results are in weight% of total mixture and are presented in Table A.1.

The initial conditions of activated sludge sample prior to sodium chloride injection are presented in Table 6.3.

Table 6.3: Characteristics of activated sludge samples prior to the salinity shock test

Date	**Temperature**	**pH**	**MLSS**	**MLVSS**	**DSVI**	**ΔR_{20}**
-	°C	-	g/L	g/L	mL/g	m^{-1}
07 Apr 2011	18.9	7.7	9.1	6.7	132	0.79
13 Apr 2011	17.1	8.5	9.4	7.2	117	1.02
18 Apr 2011	17.8	7.7	8.3	6.3	115	1.37
28 Apr 2011	18.6	7.8	10.1	7.6	114	1.39

The conductivity and total dissolved salts (TDS) results together with corresponding salt concentrations are presented in Table 6.4.

Table 6.4: Conductivity and total dissolved salts results of the salt injection experiments

Salt addition	Salt concentration		Conductivity		Total dissolved salts	
g	g_{NaCL}/L	%	µS/cm		g/L	
			Average	Standard deviation	Average	Standard deviation
0	0	0.00	1,001	72	0.5	0.04
10	0.3	0.03	1,402	63	0.7	0.03
30	1.0	0.10	2,473	96	1.2	0.05
50	1.7	0.16	3,588	222	1.8	0.1
150	5.0	0.48	7,917	827	4.0	0.4
400	13.3	1.27	16,455	2,965	8.2	1.5

During all experiments, filterability deteriorated when the salt was added (Figure 6.10). The deterioration of activated sludge filterability was directly linked to the increase in salinity, expressed as conductivity, in the sludge sample. The measured conductivity increased from approximately 1000 µS/cm for the samples without the road salt to about 16500 µS/cm for the samples with the highest salt concentrations. The TDS concentrations were in the range of 0.5±0.04 g/L and 8.2±1.5 g/L for the samples without added salt and the sample with the highest salt concentrations, respectively. During the course of experiments the pH remained stable and temperature had increased gradually by maximum 2°C.

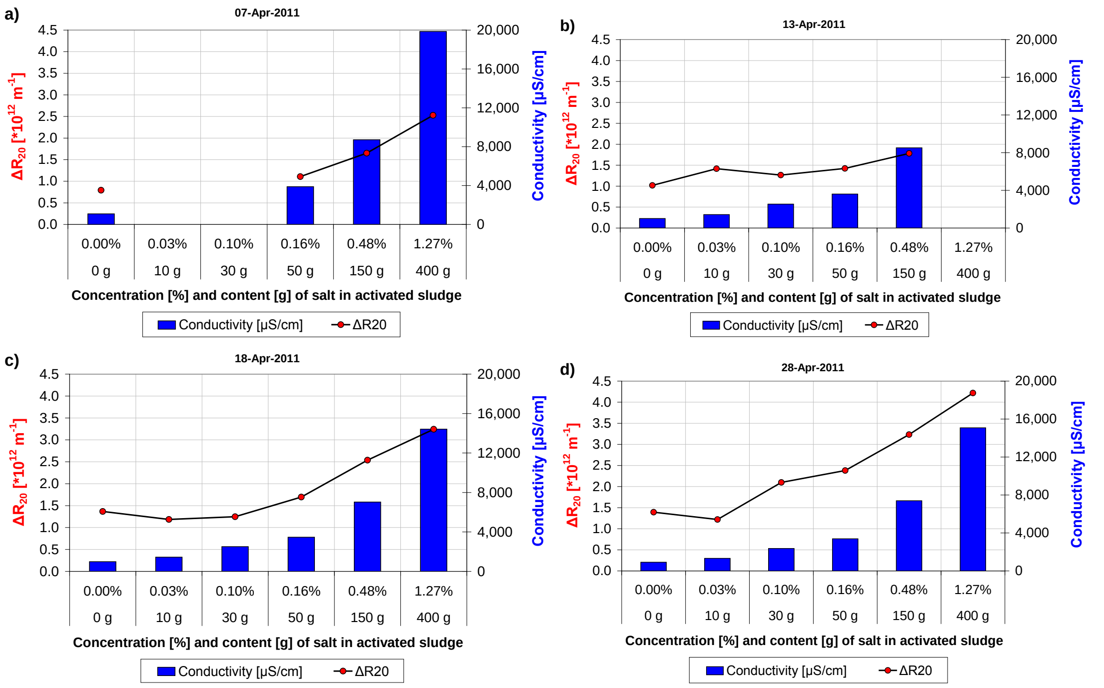


Figure 6.10: Activated sludge filterability versus conductivity

The results of these experiments clearly demonstrate that addition of the sodium chloride to activated sludge results in high ΔR_{20} values, and as such indicating worsening of activated sludge filterability (Figure 6.11).

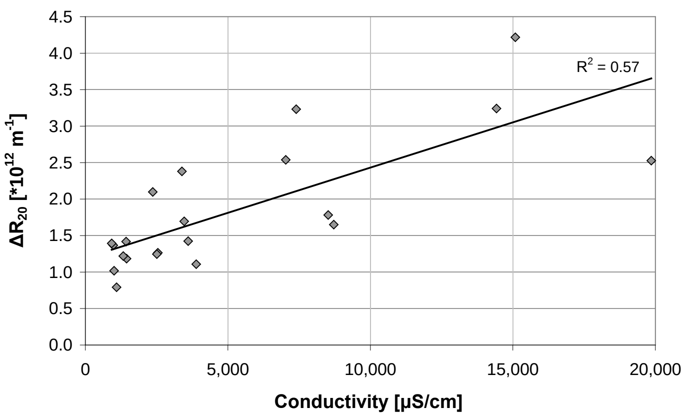


Figure 6.11: Impact of salt concentration on activated sludge filterability

The results are in accordance with the previous findings and suggest that NaCl, the main de-icing agent, can have detrimental effect on activated sludge filterability (Reid et al. 2006). The quality of activated sludge is most probably affected by reduced bacterial activity (Claros et al. 2010), settleability (Biggs et al. 2001, Sobeck and Higgins 2002, Reid et al. 2006), disturbed floc formation (Ludzack and Noran 1965, Reid et al. 2006, Wu et al. 2008), leaching of multivalent cations from the flocs (Ca^{2+}, Mg^{2+}, Fe^{3+}) disturbing the bridging function of these flocs leading to dispersed sludge, and release of EPS and SMP from the floc under high salinity conditions (Reid et al. 2006). Furthermore salinity leads to changes in surface charge, hydrophobicity and bioflocculation (Reid et al. 2006), all being influencing parameters on filterability (Van den Broeck et al. 2011). Therefore, although salt ions are smaller than the membrane pore sizes and they can pass through an MBR membrane, salinity can influence MBR performance indirectly by affecting the biotreatment processes and changes in activated sludge filterability.

6.3.3.2 Toxicity

Peculiar composition of the incoming wastewater can also affect activated sludge filterability and disturb MBR process operation. For example, during the ‘winter 2009’ experiments in Ootmarsum, membrane performance was affected by abnormal chemical composition of the incoming wastewater. The substance present in the inflow was polymeric, greasy, oil like and similar to candle wax. The polymer was supplied via the sewer network and could be discharged to the sewer system by a local industry or local community inhabitants with the toilet-water. As a consequence of the presence of this polymer, the membrane permeability dropped and there was a need to reduce the flux in order to recover expected level of performance. It can be assumed that biological degradation of polymer was limited and that it was retained on the membrane causing additional fouling problems. The permeability dropped from the values above 400 $L/m^2{\cdot}h{\cdot}bar$ to approximately 250 $L/m^2{\cdot}h{\cdot}bar$ under the applied flux of 44 $L/m^2{\cdot}h$. The problem was also observed in the CAS system as the sand filter operation was also affected. In order to retain typical MBR operation, cleanings and drainage procedures had to be performed more frequently. This process disturbance started in January 2009 and lasted till June 2009. Thus, it took approximately 6 months and a number of intensive chemical cleanings to recover to normal MBR operation and performance. These operational problems were accompanied with a serious filterability deterioration, resulting in poor sludge filterability along the whole membrane bioreactor (Figure 6.12).

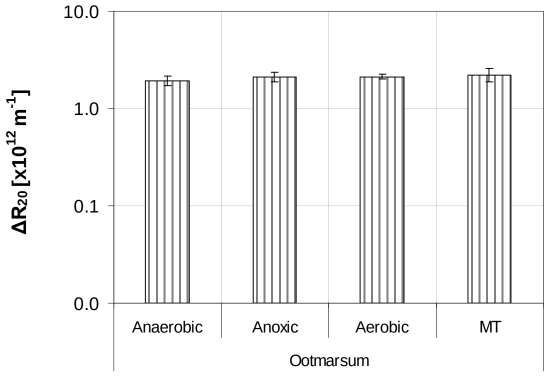


Figure 6.12: Filterability of activated sludge in various compartments of the MBR Ootmarsum during 'winter 2009' campaign

Consequently, exceptional poor filterability results were obtained in winter 2009 in MBR Ootmarsum (Figure 4.29c). During the next experimental campaign in August 2009, thus approximately 2 months after normal operation was restored, the filterability was considered as good and process operation was not hampered anymore. The filterability differences between the two periods were ascribed to the differences in the coefficient $\alpha_R \cdot c_i$, i.e., the product of the specific reference cake layer resistance and the concentration of solids accumulating in the cake layer. The coefficient $\alpha_R \cdot c_i$ was $100 \pm 8 \cdot 10^{-3}$ m^{-2} during the polymer incident period, $47 \pm 6 \cdot 10^{-3}$ m^{-2} during following winter period and about $20 \pm 9 \cdot 10^{-3}$ m^{-2} in the summer periods.

Similar observations were also made by Geilvoet (2010) based on comparable experiences encountered during measurements in Varsseveld. At that time, discharge from the local cheese factory containing synthetic polymer was affecting activated sludge filterability and operation of the MBR. The polymer could not be biologically degraded and was retained on the membrane causing serious membrane fouling. The filterability improved significantly from $5.4 \pm 2.5 \cdot 10^{12}$ m^{-1} to $0.3 \pm 0.2 \cdot 10^{12}$ m^{-1} after the cheese factory was uncoupled from the sewer. Geilvoet (2010) observed that differences in the ΔR_{20} values were attributed to the differences in the coefficient $\alpha_R \cdot c_i$. During the period of disturbed operation the coefficient $\alpha_R \cdot c_i$ was $190 \pm 76 \cdot 10^{-3}$ m^{-2} whereas during undisturbed operation it was $19 \pm 3 \cdot 10^{-3}$ m^{-2}. This is in accordance with our observations as the coefficient $\alpha_R \cdot c_i$ was higher during the period of disturbed operation compared to period without problems with influent composition, i.e., $100 \pm 8 \cdot 10^{-3}$ m^{-2} and $20 \pm 9 \cdot 10^{-3}$ m^{-2}, respectively. Furthermore, also process improvement was observed after the polymer inflow was stopped. The improvement

in permeability from about 300-350 $L/m^2 \cdot h \cdot bar$ and fluxes from 13 $L/m^2 \cdot h$ to permeability of approximately 450 $L/m^2 \cdot h \cdot bar$ and the fluxes of 20 $L/m^2 \cdot h$ under stable operation conditions was reported.
Concluding, based on the two cases, disturbances in incoming wastewater composition are crucial with regard to MBR operation and activated sludge filterability. If a toxic or refractory compound is discharged into the sewer network, and the chemical composition of the influent is drastically different, process disturbances are very likely. Furthermore, duration of the disturbances is directly linked to time that the compound is supplied to and hold-up at the plant. Intensive chemical cleanings are necessary to allow day-to-day operation but effectiveness of cleaning in place cannot be guaranteed. When reactor perturbation sustains, detailed analyses of the cause is indispensible. Depending on exposure time, the recovery period could last from few weeks up to few months.

6.3.3.3 Municipal versus industrial feed wastewater

In order to assess the impact of influent characteristics on sludge filterability, MBRs treating both municipal and industrial wastewaters were analysed. A vast number of industrial cases, e.g., food, chemical and rendering, was investigated and compared with municipal installations (Gil et al. 2011a, Gil et al. 2011b, Van den Broeck et al. 2011, Krzeminski et al. 2012a). Because the composition of any wastewater stream is different, each type of industry has its unique characteristics of wastewater. For example, waste streams of food industry are generally of organic nature and nutrient rich (Wang et al. 2005), whereas chemical ones are often very complex and containing industrial site-specific compounds. Therefore, the stresses on the microorganism are typically much higher for industrial cases compared to municipal ones. Consequently, the industrial waste streams are more difficult to treat, especially by means of biological treatment processes.

Municipal and industrial wastewaters are different in terms of composition and strength. Furthermore, municipal wastewater is generally characterised by low and more or less constant concentration of pollutants, significant seasonal temperature fluctuations, considerable diurnal flow variations and potential of hydraulic shock due to storm flows. Contrary, industrial waste streams are generally characterized by medium to high concentrations of pollutants, relatively high temperature with little variation and frequent potential shocks of toxic components (Judd 2011). Furthermore, in continuous production processes, the produced wastewater flows are mostly stable, opposite to batch-wise production processes, e.g. beer production, when high flow variations are natural, which will then be equalised prior to biological treatment.
Significant seasonal temperature fluctuations and associated low temperatures, which in turn negatively affect the filterability, typically observed in MBRs treating municipal wastewater, were not observed in the majority of industrial MBRs. The

occurrence or the lack of seasonal fluctuations in industrial MBRs depends, of course, on the type of the industry, applied productions processes and specific water consumption of that particular industry. In the majority of investigated industrial MBRs, the absence of seasonal temperature variations likely can be attributed to heat generation during the industrial production processes. The generated heat is thereafter transferred to the wastewater and attenuates the negative effect of low temperatures on filterability. Temperature was in the range 7–24°C and 17–31°C for municipal and industrial MBRs, respectively. During the studied period, influence of temperature on the filterability of the sludge was more profound for municipal MBRs than for industrial MBRs (Figure 6.13).

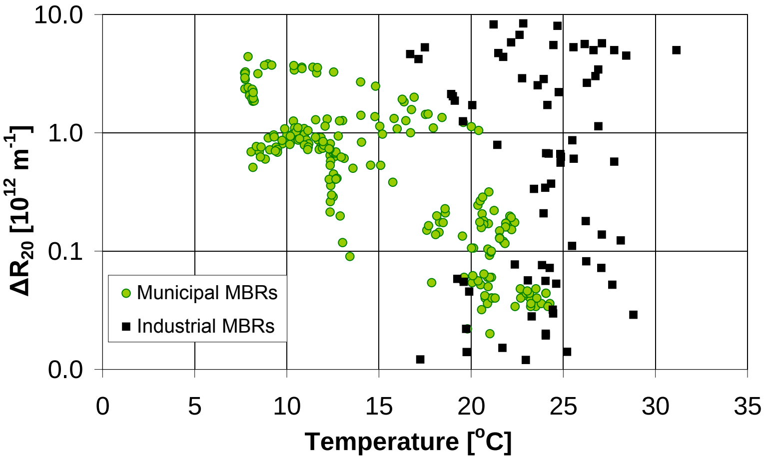


Figure 6.13: Filterability versus temperature in municipal and industrial MBRs

The MLSS concentrations of the municipal and industrial MBRs were in the range of 6-18 g/L and 3-61 g/L, respectively. The MLVSS concentrations were ranging between 5-9 g/L and 2-45 g/L for the municipal and industrial MBRs, respectively. The volatile fraction MLVSS/MLSS, reflecting the amount of organic matter in the activated sludge, was 70±8% and 83±12% for the sludge samples originating from MBRs with municipal and industrial feed, respectively. A high MLVSS/MLSS ratio indicates reduced biomass mineralization (Mikosz 2011), low accumulation or formation of inerts, and most likely the operation at high organic loading rates. Contrary, a low MLVSS/MLSS ratio, indicates accumulation of inert material (Ouyang and Liu 2009), higher stabilization rate of the sludge and/or reduction of the sludge biological activity (Benedek et al. 1972). In both municipal and industrial MBRs, the amount of organic matter was between 0.72-0.82 MLVSS/MLSS, clearly above the 40% of organic matter reported by Viero et al. (2008), who noted that a low fraction of organic matter could cause stress for activated sludge. Therefore, in our case and regardless to wastewater origin, the organic fraction should not be regarded as harmful for activated sludge, and hence for its filterability.

As already reported by Gil et al. (2011a) a slight improvement in filterability at high MLSS concentrations (R^2=0.48) was obtained for the industrial samples. In fact, when

MLSS concentration exceeds 10 g/L, ΔR_{20} values drops to below $0.1 \cdot 10^{12}$ m^{-1}, corresponding to good sludge filterability. This is in accordance with the theory of Lousada-Ferreira (Lousada-Ferreira 2011) postulating that above certain critical biomass concentration the filterability improves. Lousada-Ferreira (2011) postulated that above MLSS of 10.5 g/L, fouling particles become entrapped in the sludge matrix resulting in an activated sludge structure responsible for filterability improvement. Moreover, high MLSS concentrations might result in a highly porous and/or loosely bound cake layer (Le-Clech et al. 2003). However, an improvement in filterability as a result of high MLSS was not observed for all of the municipal MBRs. Sludge filterability improvement was observed in the MBR operated with low return ratios, i.e., MBR Heenvliet, when MLSS concentrations in the membrane tank exceed the critical MLSS concentration of 10.5 g/L (section 4.6). Therefore, it can be assumed that the MLSS concentration does not have a first order influence on activated sludge filterability, especially for the MBRs fed with municipal wastewater. The MLSS results, excluding outlying results from the MBRs operated under non stabilised conditions are presented in Figure 6.14.

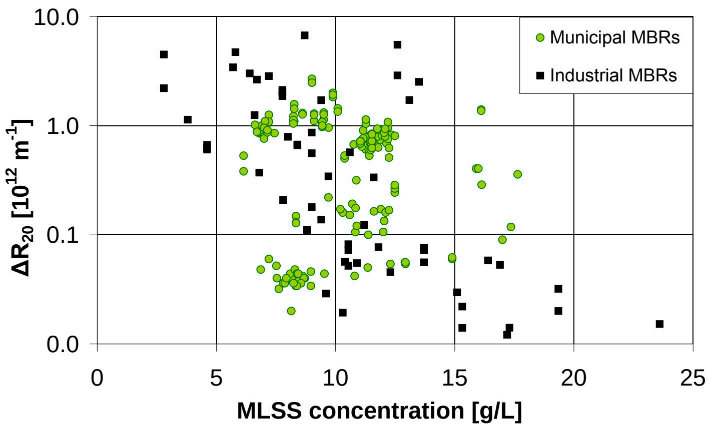


Figure 6.14: Filterability versus MLSS concentration in municipal and industrial MBRs

As mentioned before in section 6.3, deterioration of filterability was observed when the influent organic concentration was increased (Evenblij et al. 2005b). Therefore, poor sludge filterability was expected when high strength waste streams, i.e., with large quantities of contaminants, are treated.

The COD concentrations in the influent of municipal and industrial MBRs were in the range of 100-900 mg/L and 900-4300 mg/L, respectively. The COD load and loading varied significantly between the locations: for municipal MBRs, values between 50-3300 kg/day and 0.02-0.48 kgCOD/kgMLSS.day were measured whereas for industrial MBRs the load was 250-15000 kg/day and specific loading was 0.04-0.22 kgCOD/kgMLSS.day. Figure 6.15 illustrates filterability values versus influent COD concentrations for municipal and industrial MBRs. The relationship between the contaminant concentration and the sludge filterability for the MBRs treating municipal wastewater is not visible surveying all measurements. Likely this can be attributed to the fact that at municipal plants, COD measurements are not performed on a daily basis and average values for the corresponding experimental periods had to be used. In contrast, for the MBR Heenvliet, where COD measurements were performed on the exact same days as filterability tests, the relation is more apparent (Figure 5.5). Strikingly, such relation was not found for the MBRs treating industrial wastewaters (Gil et al. 2011a).

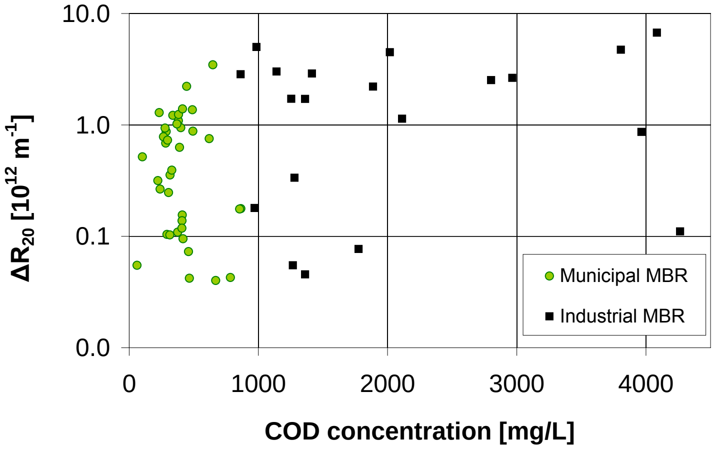


Figure 6.15: Filterability versus influent COD concentration in municipal and industrial MBRs

Figure 6.16a illustrates filterability values versus specific COD loading for municipal and industrial MBRs.

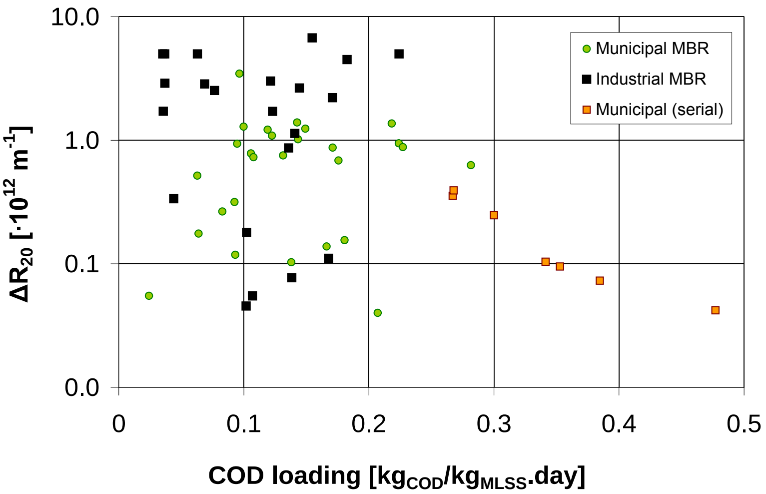


Figure 6.16: Filterability versus specific COD loading in municipal and industrial MBRs

The results and tendencies of the supposed correlation between the specific organic loading in kg_{COD}/kg_{MLSS}.day and ΔR_{20} were similar to the ones based on the influent concentrations. Again, due to the previously discussed reasons, the relationship between the contaminant concentration and the sludge filterability for the MBRs treating municipal wastewater is not visible when looking at all measurements (Figure 6.16). Nevertheless, when COD measurements correspond with filterability tests, a clear relationship is observed between the contaminant specific loading and filterability for installations fed with municipal wastewater (Figure 5.6). The organic loading of the incoming waste streams appears to not have major influence on sludge filterability, except in the case when an MBR is operated in series to the CAS system. In that case, the increase in organic loading has positive influence on sludge filterability most likely due to improved flocculation properties (Wilén et al. 2008). However, keeping in mind the relatively limited number of data points, a general conclusion cannot be drawn without further studies. Nevertheless, under steady state operating conditions, good sludge filterabilities are likely to be found even at high specific organic loadings. The good filterability at high organic loadings could be possibly explained, keeping in mind the differences between various types of industry, by the fact that industrial MBRs are somewhat regularly subjected to waste streams with high COD levels. Likely, additional changes, even if big, in the influent COD loading does not change the filterability significantly. It can be speculated that, activated sludge microbial community at industrial MBRs has possibly acclimated to high COD ranges and potential changes in the organic loading. As has been noted in *Chapter 5*, if an MBR has been fed with an average influent specific COD loading, and suddenly, the influent specific COD loading increases, the sludge reacts to this alteration and a change in floc structure or even deflocculation can be expected (Geilvoet 2010). As a result, the floc structure is damaged, the floc size decreases and fine materials, e.g. submicron particles or EPS, are released from the activated sludge matrix into the free water. In consequence, deterioration of the filterability is observed (Geilvoet 2010). Therefore, providing an adequate acclimatization period for the microorganisms is likely crucial in both municipal and industrial cases. Furthermore, it is highly probable, yet speculative, that if any municipal MBR would be fed with highly concentrated wastewater, and these conditions would be maintained until the biomass acclimatizes to the new F/M ratio conditions, after an initial organic shock and an associated filterability deterioration, the filterability would improve.

Concluding, there is a clear relationship between sludge filterability and temperature as well as between sludge filterability and influent specific COD loading for municipal MBRs. On the other hand, such relations were not observed for the MBRs treating industrial wastewater. With regard to industrial MBRs, a positive correlation was found between MLSS concentration and sludge filterability.

6.4 Operation and performance of full-scale municipal MBRs

6.4.1 Monitoring and analysing of plant operation

Parallel to the filterability tests, plant operations and performances were monitored, analysed and compared with the other investigated plants. For this purpose, the plant process (operational) data were collected for the respective periods. The main goal of this study was to determine the impact of activated sludge filterability on operation and performance of the full-scale municipal MBRs. Furthermore, operational data collected from the investigated WWTP, allowed to undertake necessary operational analysis and comparison studies of the differently designed and configured MBRs.

Several process parameters of each MBR are generally monitored in the wastewater treatment plants. Those full-scale operational parameters are registered every second by on-line sensors. The most important parameters that were investigated are: influent flow, permeate flow, withdrawn sludge flow, flux, TMP, permeability, temperature, pH, MLSS concentration, DO concentration, nitrogen and phosphorous concentrations, and airflow rates. Moreover, several characteristics of influent and effluent were analysed, e.g., COD and BOD. Together with the removal efficiency information, the performance of each MBR was determined.

The filtration characterisation tests were performed for a period of one week, however operational data collected from the plants were analysed for a longer period (three to five weeks). In this way, analysis of MBR operation is based on a longer time scale, providing better background for further comparison studies.

6.4.2 Long-term operation and performance of full-scale MBRs

6.4.2.1 MBR operation

The collected full-scale operational data from the three investigated MBRs allowed to undertake comparative studies of the operational characteristics. Detailed characteristics including membrane performances from the experimental periods at the respective plants are presented in Table 6.5.

Table 6.5: Detailed characteristics from the experimental periods at: (a) MBR Heenvliet, (b) MBR Varsseveld, and (c) MBR Ootmarsum. The values presented in subscript, normal and superscript give minimal, average and maximal values, respectively

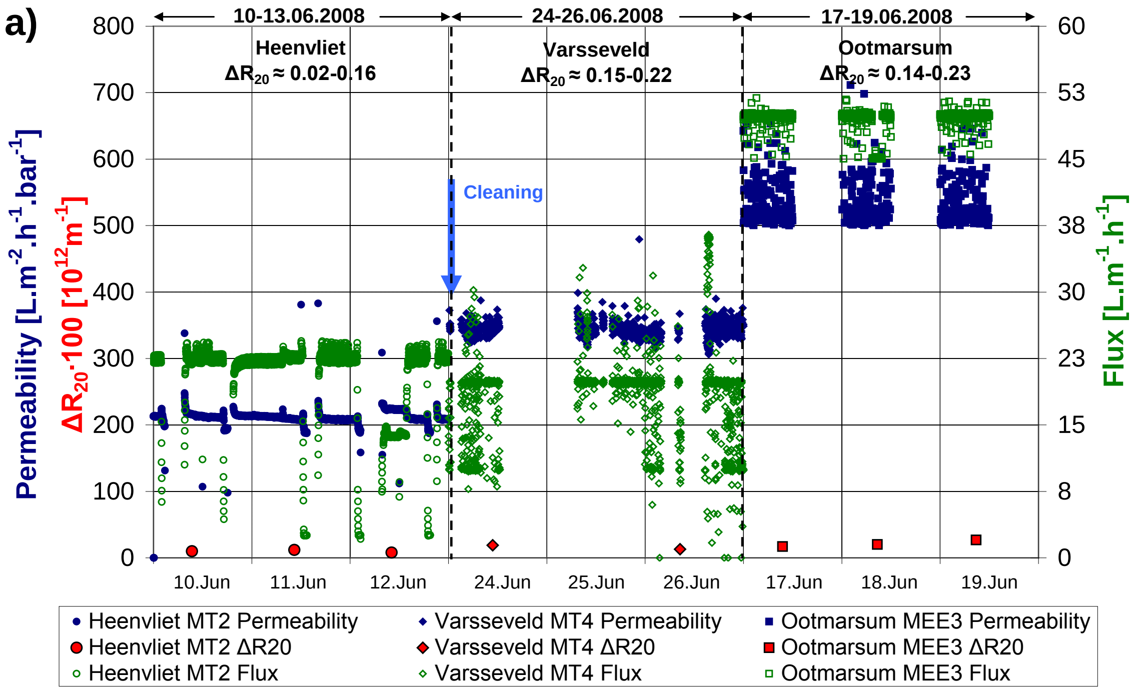
a)		**MBR Heenvliet**			
Parameter/Period		June 2008	January 2009	July 2009	February 2010
Influent	COD [mg/L]	$_{79}$– 441 –^{623}	$_{235}$– 343 –^{420}	$_{222}$– 438 –^{559}	$_{82}$– 305 –^{523}
	TP [mg/L]	$_{2}$– 8.5 –^{11}	$_{5}$– 7.2 –^{10}	$_{4.6}$– 8.6 –$^{11.0}$	$_{2.0}$– 5.2 –$^{8.5}$
	TKN [mg/L]	$_{14}$– 54 –^{71}	$_{35}$– 47 –^{65}	$_{30}$– 55 –^{73}	$_{13}$– 37 –^{51}
Permeate	COD [mg/L]	$_{15}$– 25 –^{36}	$_{19}$– 28 –^{39}	$_{16}$– 25 –^{31}	$_{12}$– 23 –^{30}
	TP [mg/L]	$_{0.3}$– 3.5 –$^{5.5}$	$_{0.9}$– 2.0 –$^{3.2}$	$_{1.0}$– 2.9 –$^{4.5}$	$_{0.1}$– 1.3 –$^{4.4}$
	TN [mg/L]	$_{0.9}$– 2.6 –$^{3.6}$	$_{1.1}$– 2.3 –$^{3.8}$	$_{1.8}$– 4.0 –$^{6.8}$	$_{1.9}$– 7.8 –$^{11.3}$
	TKN [mg/L]	$_{0.5}$– 1.0 –$^{1.5}$	$_{0.6}$– 0.9 –$^{1.2}$	$_{0.4}$– 1.2 –$^{1.9}$	$_{0.4}$– 2.3 –$^{6.8}$
TSS [g/L]		$_{7.4}$– 7.9 –$^{8.8}$	$_{7.5}$– 9.7 –$^{11.2}$	$_{6.3}$– 7.2 –$^{9.0}$	$_{6.8}$– 7.6 –$^{13.5}$
T [°C]		$_{19.6}$– 20.1 –$^{20.5}$	$_{6.3}$– 8.1 –$^{8.7}$	21–22	$_{7.0}$– 9.9 –$^{12.3}$
DO [mgO_2/L]		$_{1.5}$– 3.2 –$^{4.5}$	$_{0.0}$– 2.6 –$^{8.3}$	$_{0.1}$– 0.6 –$^{7.0}$	$_{0.0}$– 0.7 –$^{10.0}$
pH		$_{7.8}$– 8.1 –$^{8.2}$	$_{8.0}$– 8.2 –$^{8.6}$	$_{8.0}$– 8.2 –$^{8.3}$	$_{7.8}$– 8.0 –$^{8.0}$
Influent flow [m^3/h]		$_{0}$– 76 –^{126}	$_{0}$– 120 –^{138}	$_{0}$– 20 –^{54}	$_{0}$– 29 –^{51}
Permeate flow [m^3/h]		$_{0}$– 49 –^{103}	$_{0}$– 70 –^{112}	$_{0}$– 19 –^{81}	$_{0}$– 26 –^{64}
Gross flux [L/m^2.h]		$_{13}$– 20 –^{25}	$_{14}$– 21 –^{39}	$_{16}$– 12 –^{36}	$_{14}$– 17 –^{32}
Permeability [L/m^2.h.bar]		$_{187}$– 210 –^{250}	$_{123}$– 156 –^{205}	$_{269}$– 325 –^{402}	$_{277}$– 414 –^{531}
MBR state		steady	steady	steady	Steady

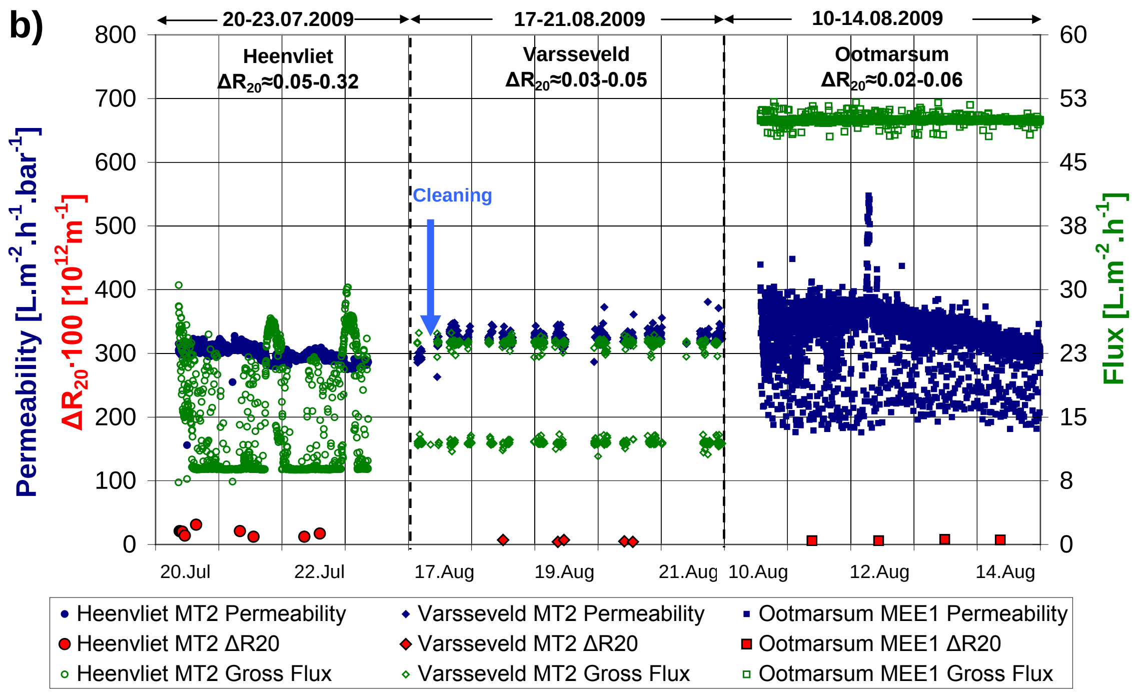
b)		**MBR Varsseveld**			
Parameter/Period		June 2008	February 2009	August 2009	March 2010
Influent	COD [mg/L]	$_{840}$ – 855 – ^{870}	$_{390}$ – 648 – ^{905}	$_{430}$ – 670 – ^{910}	$_{510}$ – 620 – ^{730}
	TP [mg/L]	$_{14}$ – 15 – ^{16}	$_{6.1}$ – 10.5 – $^{14.0}$	$_{5.5}$ – 10.3 – $^{15.0}$	$_{6.4}$ – 8.0 – $^{9.6}$
	TKN [mg/L]	$_{67}$ – 74 – ^{80}	$_{33}$ – 57 – ^{81}	$_{27}$ – 54 – ^{81}	$_{33}$ – 41 – ^{49}
Permeate	COD [mg/L]	$_{25}$ – 27 – ^{29}	$_{16}$ – 23 – ^{30}	$_{25}$ – 27.5 – ^{30}	$_{30}$ – 33 – ^{36}
	TP [mg/L]	$_{0.15}$ – 0.18 – $^{0.21}$	$_{0.8}$ – 1.1 – $^{1.3}$	$_{0.4}$ – 0.6 – $^{0.7}$	$_{0.4}$ – 0.7 – $^{1.0}$
	TN [mg/L]	$_{1.5}$ – 1.8 – $^{2.1}$	$_{4.8}$ – 5.0 – $^{5.2}$	$_{4.1}$ – 5.8 – $^{7.4}$	$_{4.0}$ – 6.3 – $^{8.5}$
	TKN [mg/L]	$_{0.8}$ – 1.1 – $^{1.3}$	$_{1.2}$ – 1.7 – $^{2.2}$	$_{1.7}$ – 1.8 – $^{1.9}$	$_{1.6}$ – 2.1 – $^{2.6}$
TSS [g/L]		$_{8.9}$ – 10.0 – $^{10.5}$	$_{7.4}$ – 8.1 – $^{10.1}$	$_{0.3}$ – 8.0 – $^{10.8}$	$_{7.6}$ – 9.2 – $^{21.4}$
T [°C]		$_{19.7}$ – 21.7 – $^{22.3}$	$_{6.4}$ – 11.3 – $^{12.8}$	$_{18.2}$ – 22.0 – $^{24.0}$	$_{7.9}$ – 11.2 – $^{14.3}$
DO [mgO_2/L]		$_{0.6}$ – 0.6 – $^{3.9}$	$_{0.0}$ – 0.9 – $^{5.0}$	$_{0.1}$ – 0.5 – $^{5.0}$	$_{0.2}$ – 1.1 – $^{4.6}$
pH		$_{7.1}$ – 7.1 – $^{7.2}$	$_{6.9}$ – 7.2 – $^{7.3}$	$_{6.9}$ – 7.3 – $^{8.0}$	$_{7.5}$ – 7.8 – $^{8.0}$
Influent flow [m^3/h]		$_{7}$ – 133 – ^{773}	$_{0}$ – 163 – ^{799}	$_{0}$ – 181 – ^{898}	$_{0}$ – 223 – ^{786}
Permeate flow [m^3/h]		$_{0}$ – 160 – ^{935}	$_{0}$ – 189 – ^{995}	$_{0}$ – 210 – ^{1131}	$_{0}$ – 258 – ^{1074}
Gross flux [L/m^2.h]		$_{10}$ – 20 – ^{40}	$_{12}$ – 22 – ^{45}	$_{12}$ – 24 – ^{45}	$_{14}$ – 25 – ^{45}
Permeability [L/m^2.h.bar]		$_{260}$ – 328 – ^{500}	$_{82}$ – 182 – ^{220}	$_{170}$ – 302 – ^{378}	$_{107}$ – 178 – ^{280}
MBR state		steady	unsteady	steady	steady

c)		**MBR Ootmarsum**			
Parameter/Period		June 2008	February 2009	August 2009	March 2010
Influent	COD [mg/L]	865	$_{350}$ – 445 – ^{580}	$_{640}$ – 783 – ^{1030}	$_{200}$ – 400 – ^{500}
	TP [mg/L]	10	$_{4.7}$ – 6.0 – $^{8.1}$	$_{9.2}$ – 10.0 – $^{11.0}$	$_{2.3}$ – 5.4 – $^{6.9}$
	TKN [mg/L]	78	$_{26}$ – 39 – ^{53}	$_{59}$ – 60 – ^{61}	$_{14}$ – 36 – ^{44}
Permeate	COD [mg/L]	24	$_{21}$ – 26 – ^{30}	$_{22}$ – 25 – ^{28}	$_{26}$ – 26.3 – ^{27}
	TP [mg/L]	1.6	$_{0.1}$ – 0.8 – $^{1.7}$	$_{0.3}$ – 0.4 – $^{0.6}$	$_{2.7}$ – 3.4 – $^{4.8}$
	TN [mg/L]	2.9	$_{1.8}$ – 5.2 – $^{8.8}$	$_{2.1}$ – 3.2 – $^{4.3}$	$_{6.5}$ – 12.4 – $^{16.2}$
	TKN [mg/L]	2.5	$_{1.1}$ – 2.8 – $^{4.6}$	$_{1.5}$ – 2.1 – $^{2.7}$	$_{1.3}$ – 1.6 – $^{2.2}$
TSS [g/L]		$_{18.3}$ – 18.5 – $^{18.7}$	$_{7.0}$ – 7.8 – $^{8.3}$	$_{17.2}$ – 20.4 – $^{21.6}$	$_{6.4}$ – 8.9 – $^{11.2}$
T [°C]		$_{9.0}$ – 9.3 – $^{10.1}$	$_{0.0}$ – 7.6 – $^{8.2}$	$_{3.9}$ – 7.6 – $^{12.5}$	$_{5.7}$ – 7.7 – $^{10.4}$
DO [mgO_2/L]		$_{0.2}$ – 0.6 – $^{2.6}$	$_{0.0}$ – 0.7 – $^{5.7}$	$_{0.4}$ – 1.7 – $^{8.8}$	$_{0.7}$ – 3.8 – $^{7.2}$
pH		7.9	$_{7.6}$ – 7.7 – $^{7.8}$	$_{7.9}$ – 7.9 – $^{7.9}$	$_{7.6}$ – 7.7 – $^{7.8}$
Influent flow [m^3/h]		$_{0}$ – 43 – ^{271}	$_{0}$ – 57 – ^{291}	$_{0}$ – 59 – ^{150}	$_{0}$ – 46 – ^{150}
Permeate flow [m^3/h]		$_{0}$ – 41 – ^{135}	$_{0}$ – 48 – ^{116}	$_{0}$ – 56 – ^{145}	$_{0}$ – 41 – ^{117}
Gross flux [L/m^2.h]		$_{40}$ – 50 – ^{61}	$_{40}$ – 44 – ^{46}	$_{40}$ – 51 – ^{56}	$_{40}$ – 44 – ^{49}
Permeability [L/m^2.h.bar]		$_{350}$ – 458 – ^{630}	$_{125}$ – 284 – ^{448}	$_{125}$ – 257 – ^{550}	$_{125}$ – 263 – ^{580}
MBR state		steady	unsteady	steady	steady

Furthermore, MBRs operation was investigated for different periods and related to the activated sludge filterability measurements as discussed in section 6.4.2.3 and 6.4.3. The results of the operational monitoring comparison, presenting examples of typical full-scale MBR operation during winter and summer seasons are illustrated in Figure 6.17. Additionally, as an example of an a-typical period, operational parameters of the 'winter 2009' period, where special events took place, are presented in Figure 6.22 (section 6.4.4). The graphs illustrate membrane permeability plotted on y-axis (in blue), sludge filterability expressed as, and for better visualization multiplied by factor

100, ΔR_{20} parameter on y-axis (in red), gross membrane flux on second y-axis (in green) and time on x-axis. It is important to mention that, in March 2009, MBR in Heenvliet was switched from in-series to parallel operation. Consequently, among others changes, the permeate production and fluxes were reduced whereas permeability increased. Further discussion on MBR design and configurations is provided in *Chapter 7*.





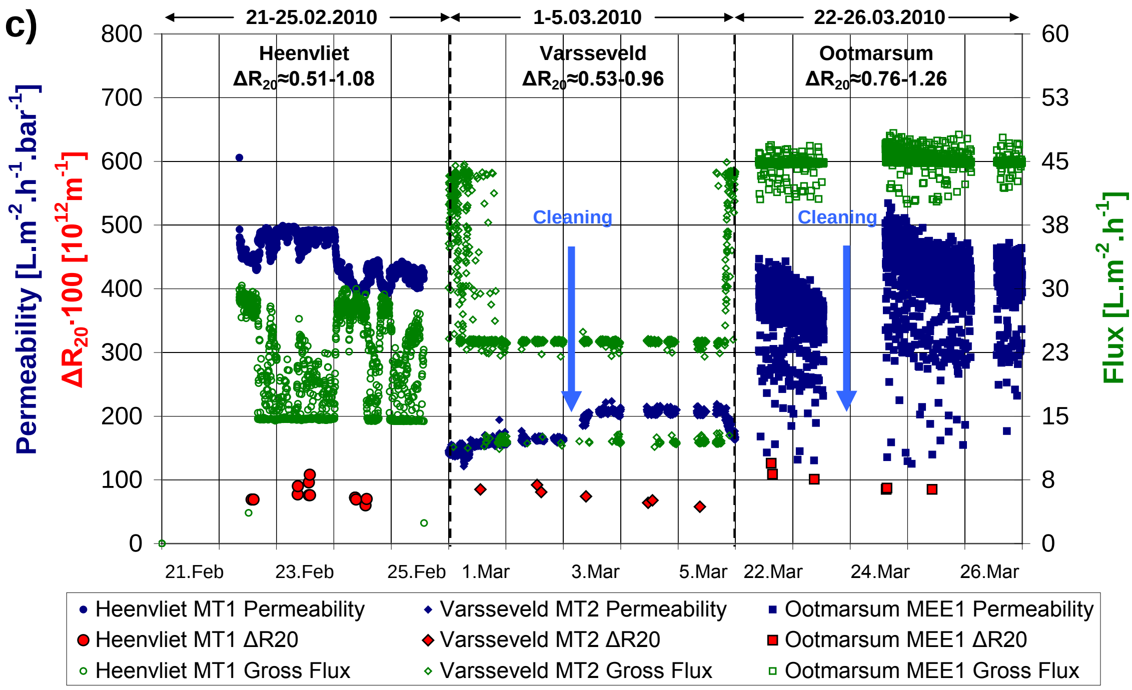


Figure 6.17: Comparison of Heenvliet, Varsseveld and Ootmarsum MBRs operation monitoring – flux, permeability and filterability – during normal operation in: a) summer 2008, b) summer 2009 and c) winter 2010

The operation study reveals a clear contrast between sidestream and submerged configurations. The sidestream MBR, represented by the MBR Ootmarsum, was operated at usually 2 times higher fluxes, i.e., 45 L/m^2.h, compared to the other MBRs, i.e., 15-25 L/m^2.h. Higher fluxes, generally, resulted in a higher permeability. However, different behaviour was observed in summer 2009 and partially in winter 2010. In summer 2009, the average permeability was 6-12% lower then at the submerged MBRs, most likely because the membrane performance was still affected by the toxic discharge in the sewerage (described in section 6.3.3.2). In winter 2010, MBR Heenvliet was operated in parallel to CAS system. At that time, permeate production was reduced by more than a half compared to operation in-series. Thus, lower fluxes were applied, subsequently, leading to higher permeability values than during in-series operation, e.g., in winter 2009.

When comparing submerged MBRs, namely Heenvliet and Varsseveld, it can be seen that Heenvliet plant equipped with flat sheet membranes achieved lower permeability than Varsseveld equipped with hollow fibre membranes, while working with higher fluxes (Figure 6.17a). It is possible that better permeability values at the MBR Varsseveld were influenced by lower MLSS concentrations. While sludge samples from Heenvliet were about 13 g/L, Varsseveld had MLSS concentrations of about 11 g/L. Activated sludge samples, in both cases collected from the membrane tanks, had similar filterability. Under winter conditions both MBRs were operated under comparable operational characteristics: flux of 20 L/m^2.h and permeability of 160-180 L/m^2.h.bar.

The difference between summer and winter period was also observed in the operation of the MBR. Common permeability trends were observed in all treatment plants: low in winter and high in summer. Applied flux was lowered (Ootmarsum) or increased (Heenvliet) in winter periods, or once lowered once increased (Varsseveld), and is mainly dependent on the flow that needs to be treated. Furthermore, when the MBR Heenvliet was operated at a low flux of 10 L/m^2.h in summer 2009 and 15 L/m^2.h in winter 2010, an increase in permeability was observed. This observation is in accordance to Le-Clech (2006) and Zhang et al. (2006). The link between applied flux and sludge filterability is explored in section 6.4.3.

6.4.2.2 MBR performances and removal efficiency

Good removal efficiencies of COD, BOD and TKN were achieved in all of the plants during the 3 years of monitoring. All MBRs removed COD to a similar concentration of about 25 mg/L with removal efficiency between 91-96%. BOD was removed way below the requirements (10 mg/L) with efficiencies of about 99%. BOD concentrations lower than 1.0 mg/L (Ootmarsum and Varsseveld) and 1.7 mg/L (Heenvliet) were accomplished.

TKN was removed with 95-98% efficiency down to concentrations of about 1.0-2.1 mg/L. The nitrogen removal varied between the plants and years with average values of total nitrogen between 3.0 and 7.2 mg/L.

In all cases, complete biological removal of phosphorous was not obtained likely due to insufficient anaerobic conditions in the bioreactor and/or low sludge loading levels. Phosphorous removal efficiency was in range of 67-96%. Best phosphorous removal reaching 0.4-0.7 mg/L in the effluent at an efficiency of 94-96% was attained in MBR Varsseveld when dosage of iron chloride was applied. In MBRs of Heenvliet and Ootmarsum, iron chloride is not added. As a consequence, phosphorous removal was lower (67-85%) and concentrations in the effluent are higher (1.5-2.2 mg/L).

The summary of overall performance of investigated MBRs, in terms of pollutants removal efficiency, is presented in Table 6.6.

Table 6.6: Influent & permeate characteristics and removal efficiency

MBR Performance		**Influent**				**Permeate**					**Removal efficiency**			
		COD	**BOD**	**TP**	**TKN**	**COD**	**BOD**	**TN**	**TP**	**TKN**	**COD**	**BOD**	**TP**	**TKN**
MBR plant	Period	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	%	%	%	%	%
Heenvliet (FS)	2008	366 ± 127	156 ± 60	7 ± 2	44 ± 15	25 ± 9	1.7 ± 4.3	3.0 ± 1.9	2.2 ± 1.3	1.0 ± 0.4	92	98.5	67	98
	2009	374 ± 130	171 ± 67	8 ± 3	48 ± 17	24 ± 7	1.3 ± 0.5	4.2 ± 1.8	1.9 ± 1.4	1.1 ± 0.4	93	99.0	73	97
	2010	326 ± 119	147 ± 61	6 ± 2	41 ± 15	26 ± 6	1.2 ± 0.3	7.2 ± 2.9	1.5 ± 3.7	1.9 ± 1.9	91	99.0	85	95
Ootmarsum (MT)	2008	513 ± 161	212 ± 73	7 ± 2	47 ± 16	23 ± 5	0.9 ± 0.4	3.6 ± 2.0	2.0 ± 2.1	1.5 ± 1.1	95	99.5	72	97
	2009	514 ± 189	222 ± 88	7 ± 3	45 ± 16	24 ± 5	0.8 ± 0.3	3.6 ± 1.7	1.7 ± 1.4	1.7 ± 1.2	95	99.6	74	96
	2010	485 ± 167	195 ± 72	7 ± 3	43 ± 16	24 ± 4	0.7 ± 0.3	5.9 ± 4.6	2.2 ± 1.5	1.4 ± 0.6	94	99.6	69	96
Varsseveld (HF)	2008	693 ± 159	297 ± 90	12 ± 3	60 ± 17	25 ± 5	0.9 ± 0.4	3.9 ± 2.9	0.4 ± 0.3	2.1 ± 2.5	96	99.7	96	96
	2009	752 ± 251	306 ± 130	13 ± 7	59 ± 19	25 ± 6	0.8 ± 0.3	5.8 ± 3.8	0.7 ± 0.7	1.8 ± 0.7	96	99.7	94	96
	2010	765 ± 181	310 ± 84	11 ± 3	61 ± 17	27 ± 6	0.8 ± 0.3	3.6 ± 1.4	0.4 ± 0.3	1.7 ± 0.5	96	99.7	96	97

6.4.2.3 MBR membrane permeability and relation to filterability and temperature

Filterability results were analysed with respect to the membrane permeability achieved by the respective MBR at the time of the DFCm measurements. These analyses allowed to assess the impact of the seasonal temperature and filterability fluctuations on process operation and performance. Significant variations in permeability were observed between the campaigns at the full-scale municipal MBRs (Figure 6.18).

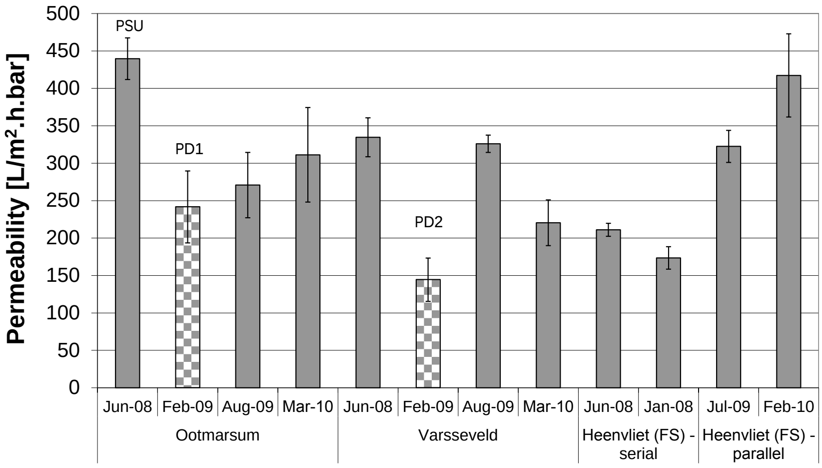


Figure 6.18: Permeability of the full-scale MBRs during different measurement campaigns. PSU – process start-up; PD – process disturbance 1; PD2 – process disturbance 2.

During the analysed period of the MBR operation, two serious process disturbances occurred. Process perturbations and poor sludge filterabilities were encountered during the disturbed periods of operation. The process disturbance 1 (PD1) happened at MBR Ootmarsum during winter period and was a result of peculiar and abnormal chemical composition of the incoming wastewater. The second process disturbance (PD2) also happened during the same winter period but at MBR Varsseveld. The PD2 was caused by heavy rain fall and abrupt change of inflow temperature. In addition, one of the measurement campaigns in Ootmarsum was carried out 7 months after the commissioning period and the MBR process start-up (PSU). The short operation period coincide with the summer period allowing achieving excellent permeability results. Hence, the very high permeability values achieved in June 2008 at MBR Ootmarsum should not be considered as an ultimate permeability reference or a threshold values. It should be rather considered as the initial performance of the new membranes, and as such, not or little influenced by irreversible and/or irrecoverable fouling.

The permeability mostly decreases when filterability deteriorates (Figure 6.19). Moderate to strong statistical correlation between filterability values and permeability data was observed for MBRs equipped with hollow fibre and flat sheet membranes, namely Varsseveld and Heenvliet. The Pearson coefficient (r_p) of -0.67 (p_{value}=0.00) and 0.79 (p=0.00) was noticed for MBR Heenvliet when operated in series and in parallel, respectively. Strongest statistical correlation was observed for hollow fibre MBR, i.e., Varsseveld, with a Pearson coefficient of -0.84 (p=0.00). The correlation between filterability and permeability was found to be weak (r_p=-0.37, p=0.01) for the MBR Ootmarsum equipped with multi tube membranes. The impact of membrane configurations on permeability–filterability correlation is mainly due to the chemical cleanings frequency (Moreau 2010), The cleaning strategies applied in full-scale MBRs are further discussed in section 8.3.15.

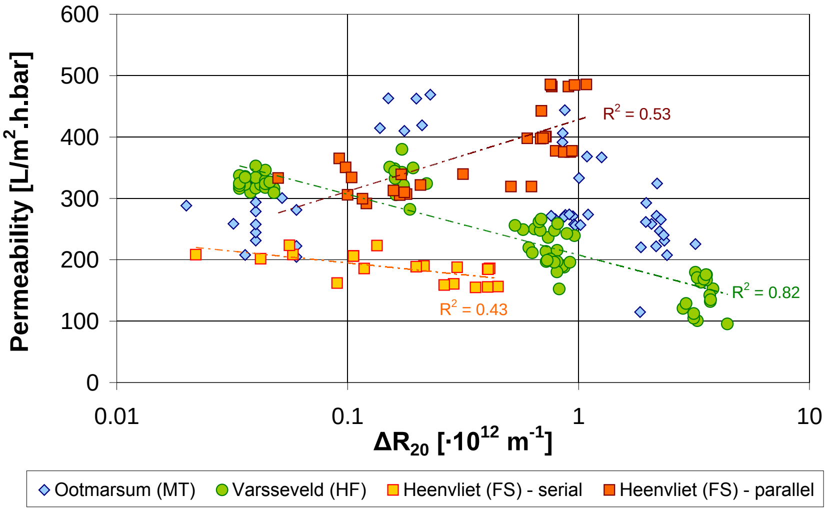


Figure 6.19: Permeability of the full-scale MBRs as a function of sludge filterability

Interesting, yet uncommon, results were observed during parallel operation of MBR Heenvliet. Better permeability values were achieved when sludge filterability was worse. The explanation of this exceptional high permeability is twofold. First of all, prior to the filterability experiments an intensive chemical cleaning of the membranes was performed at the plant. Consequently, the irreversible fouling was probably removed and the membranes were performing better while achieving higher permeability. In addition, chemicals used for the membranes cleaning, i.e., sodium chloride and citric acid, were introduced and treated at the site subsequently leading to deterioration of activated sludge filterability.

These results are partly consistent with and partly opposing to the previous findings of Moreau (2010). Moreau noted significant correlation for hollow fibre configurations which is in line with our findings. Conflicting results were found for flat sheet configurations. Our data demonstrated a moderate to strong correlation whereas Moreau reported no correlation. The lack of correlation in Moreau cases is most likely caused by the limited number of data points and the fact that no distinction between serial and parallel configurations was made. The relation between filterability and permeability for multi tube MBRs was not statistically analysed by the Moreau. In an other study, Gil (2011a) observed no relation between filterability and permeability for any membrane configuration.

Furthermore, permeability is generally worse in winter than in summer periods (Figure 6.20). Decrease in permeability in the winter, between 10% and 50 % compared to summer values, is likely related to the activated sludge filterability deterioration and linked with unexpected operational problems. However, despite the tendency that better permeability is achieved in the summer, no clear pattern between permeability and temperature was found. The uncommon results were observed during the 'winter 2010' campaign when MBR Heenvliet was operated in parallel to the CAS system. At that time, permeability was better than during comparable summer period. As explained before, the recently conducted chemical cleaning of the membranes allowed achieving higher permeability and reducing the effect of low temperatures. Moreover, permeability of different MBRs varies considerably at similar temperatures. At similar temperatures the difference can not be ascribed to the changes in viscosity. Besides, influence of apparent viscosity was reported to be insignificant on membrane permeability (Moreau et al. 2009). The difference in permeability most likely can be ascribed to differences in initial permeability of different membrane configurations (Moreau 2010), time since the last membrane cleaning, differences in achieved permeability between the membrane lines within one plant and/or operational circumstances at individual locations (Gil et al. 2011a). In other words, conservative operation, e.g., frequent cleanings, reduced flux, increased membrane air-scouring, can overcome the negative impact of low temperatures on permeability. Therefore, permeability is considered a weak parameter to monitor MBR performances and to determine the cause of poor filtration process. This is in accordance to the statements of other researchers (Geilvoet 2010, Moreau 2010, Gil et al. 2011a, Van den Broeck et al. 2011, Krzeminski et al. 2012a).

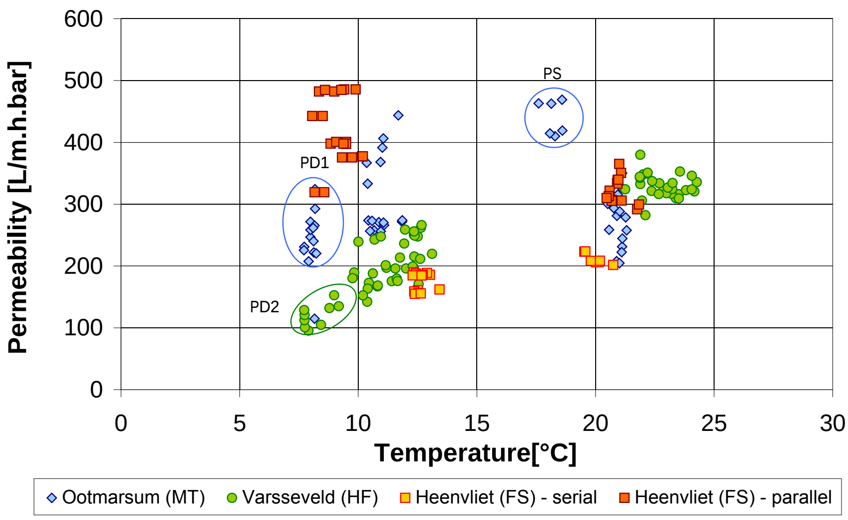


Figure 6.20: Permeability of the full-scale MBRs as a function of sludge temperature

6.4.3 Inter-relations between sludge filterability and MBR operation

In the previous section (6.4.2.3), relation between activated sludge filterability and membrane permeability as well as between filterability and temperature were discussed. The relationship between sludge filterability and MLSS was previously comprehensively investigated by colleague PhD researcher Maria Lousada-Ferreira, and is published elsewhere (2010, 2011). The positive influence of prolonged SRT on activated sludge quality was jointly published by Van den Broeck et al. (2012). Jin et al. (2006) reported greater number of small particles and reduced porosity of the biofilm under DO concentrations below 0.1 mgO_2/L.

The relation between applied fluxes and sludge filterability was equivocal (Figure 6.17). The high operational fluxes applied in MBR Ootmarsum were, on one hand, linked with slight increase in filtration resistance (summer 2008 and winter 2010), and, on the other hand, not accompanied by a higher filtration resistance rate (summer 2008 and summer 2009) expressed by the ΔR_{20} parameter. This could indicate that the flux rate is not directly influencing activated sludge filterability in full-scale MBRs. The flux is proportional to the amount of deposited material on the membrane, hence, to the TMP but not to the filterability. In other words, the increase in flux will result in a TMP increase and, subsequently, in a membrane fouling intensification (Al-Amri et al. 2010). However, the flux increase will neither lead to activated sludge deterioration nor improvement.

On the other hand, the quality of activated sludge does have an indirect influence on the applicable flux rates. Activated sludge with a poor filterability most likely will result in TMP increase due to membrane fouling increment. Consequently, in order to maintain the same process permeability, the flux rates should be increased accordingly at the expense of an increased energy demand. However, due to further TMP increase, more frequent membrane cleanings can be expected. On the other hand, if the flux rates will remain unchanged, overall membrane permeability will be reduced.

Therefore, an MBR can be operated at high fluxes with a poor filterable activated sludge but it will require actions to remedy membrane fouling, e.g., intensive air-scouring, shorter filtration intervals, prolonged relaxation and/or backwash periods, frequent chemical cleanings and/or addition of chemicals like flocculants or activated carbon to the mixed liquor. As a consequence of all these actions, the operational and maintenance costs will increase.

Figure 6.21 presents applied flux in different full-scale MBRs plotted against measured filterability of activated sludge. The statistical analysis shows moderate correlation (r_p=-0.58, p_{value}=0.00) between gross flux and sludge filterability for MBR Ootmarsum equipped with multi tube membranes. Moderate correlation (r_p=0.45, p_{value}=0.00) was observed for MBR Varsseveld equipped with hollow fibre membranes. Also moderate correlation (r_p=0.47, p_{value}=0.01) was found for flat sheet MBR Heenvliet operated in parallel. Statistically insignificant correlation (r_p=0.30,

p_{value}=0.22) was noticed for MBR Heenvliet operated in-series. The results are in contrast to Kornboonraksa and Lee (2009) who observed strong negative correlation of r_p=0.86 between flux and membrane resistance in the hollow fibre pilot plant (Kornboonraksa and Lee 2009).

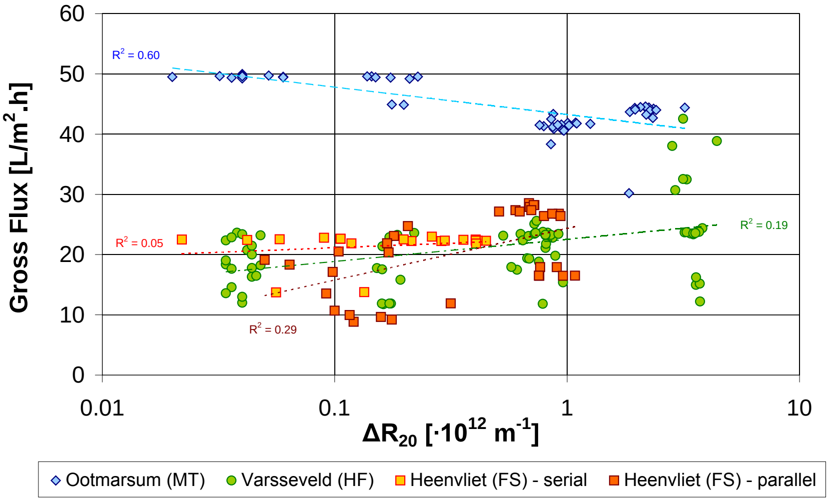


Figure 6.21: Gross flux of the full-scale MBRs as a function of sludge filterability

6.4.4 Operational perturbations

The examples of stable operation of the full-scale MBRs were presented and discussed in section 6.4.2.1. However, membrane bioreactors processes may be disturbed leading to operational problems. Consequently, operation and/or performance of the installation might be affected. Operational problems in MBRs are often related either to composition of the incoming wastewater, to quality of the activated sludge, or to problems related with the equipment failures (Gil et al. 2011b). The examples of a-typical periods encountered in Varsseveld and Ootmarsum during the ‘winter 2009’ campaign are presented in Figure 6.22. The graphs illustrate process permeability plotted on y-axis (in blue), sludge filterability expressed as, and for better visualization multiplied by a factor 100, ΔR_{20} parameter on y-axis (in red), gross flux on second y-axis (in green) and time on x-axis.

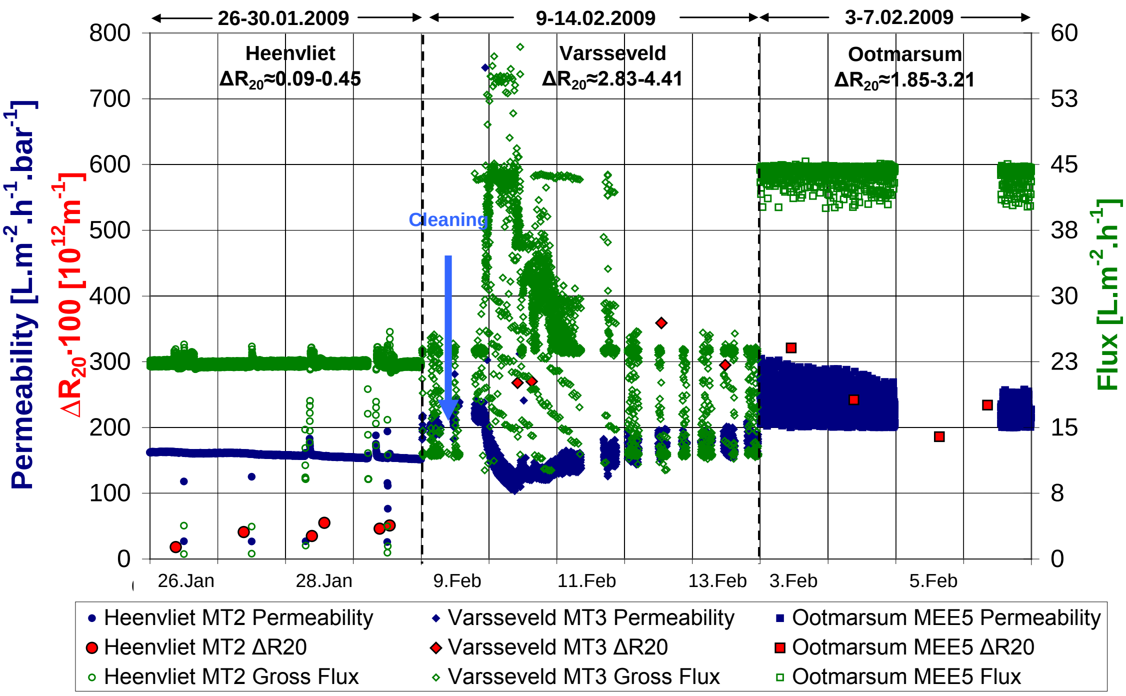


Figure 6.22: Comparison of Heenvliet, Varsseveld and Ootmarsum MBRs operation monitoring – flux, permeability and filterability – during special events in winter 2009

The abnormal circumstances encountered in Varsseveld and Ootmarsum during the 'winter 2009' campaign where partially discussed in section 6.3.2 and 6.3.3.2, respectively. In those two cases, respectively, a rapid drop in the influent temperature combined with hydraulic load shock and composition of the incoming wastewater were most likely crucial factors affecting MBR operation and performance. The first perturbation is clearly visible in Figure 6.22, whereas the latter one is not so clear due to rather a long term effect on MBR operation and performance.

In Varsseveld, due to temperature drop and high influent flow rates, activated sludge filterability deteriorated in each membrane section and the permeability achieved at the plant was approximately 50% lower than usually (Figure 6.22). The actual permeability, depending on the membrane section, decreased from about 200-230 L/m^2·h·bar under undisturbed operation circumstances to values below 100 L/m^2·h·bar during the altered conditions. At that time, the applied flux was increased from typical 20±6 L/m^2·h to about 35±8 L/m^2·h in order to compensate higher flows.

In regard to potential operational problems due to treatment of peak flows, i.e., hydraulic load shocks, two scenarios can be considered. In one, the excessive amount of wastewater is treated by the MBR without a major problem: membrane fluxes increase to compensate higher influent flow rates, whereas permeability decrease lasted only for the duration of the peak flow. The deteriorating sludge filterability (section 6.3.1) indicates higher propensity of the sludge constituents to cause membrane fouling. However, in case of frequently cleaned hollow fibre membranes, like in MBR Varsseveld, significant operational problems were not observed and should not be expected. In the second scenario, the high flow rate is combined with other perturbation circumstances like organic load shock, abrupt temperature change or toxic wastewater composition. All of the above mentioned parameters will likely cause deterioration of activated sludge filterability and lead to reactor perturbations. Despite, that only a limited number of heavy rainfall events in full-scale MBRs observed in this study, the two proposed scenarios are plausible explanations for the most typical operational behaviours related to hydraulic load. Therefore, it can be expected that the excessive amount of the wastewater alone is probably not a serious limiting factor in operation and performance of the full-scale municipal MBRs, unless other undesired circumstances coincide with the higher flow rate.

One of those undesired situations can be operation under quickly changing temperature and/or at low temperature of the incoming wastewater (section 6.3.2). The permeability decreased steadily parallel to the temperature decrease (Figure 6.23). The lowest recorded temperature was 6.4°C. It appears that especially at temperature, below 12°C, the permeability decreases rapidly. Permeability remains at reduced level of 150-200 L/m^2.h.bar for about 36-72 hours, restores somewhat after the chemical cleaning, to fully recover to the permeability levels prior to the event after 7 to 8 days.

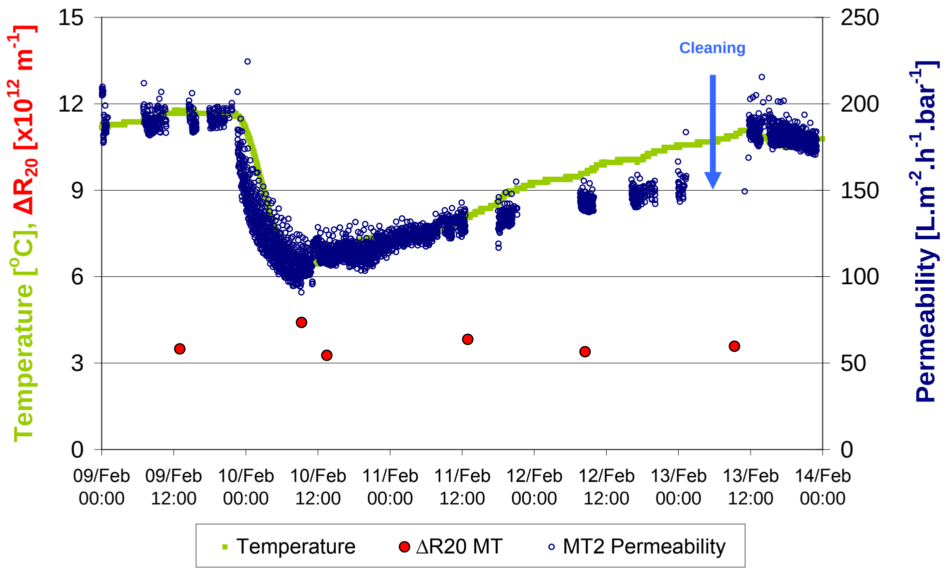


Figure 6.23: Temperature and permeability and filterability, i.e., ΔR_{20}, trends in MBR Varsseveld during special events in winter 2009

Also the presence of a toxic or refractory component in the influent can be detrimental for the activated sludge and the MBR process operation. As has been previously discussed in section 6.3.3 and 6.4.2.3, high salinity or presence of polymers or chemical cleaning agents are likely to cause process disturbances. The presence of the refractory compounds might lead to development of an activated sludge with a high fouling propensity or to accumulation of these compounds on the membrane surface. In consequence, reduced performance and operation upsets of an MBR can be foreseen. In addition, high organic loadings are negatively affecting sludge filterability in municipal MBRs and, thus, can lead to increase in membrane fouling. In consequence, frequent chemical cleanings might be required to ameliorate the reactor performance.

In addition, problems related with the low dissolved oxygen concentrations are reported to be negatively influencing sludge filterability (Gil et al. 2011b). DO levels reported during the ‘winter 2009’ campaign (6.3.2), maintained below 1.0 mgO_2/L for a period of about 2 weeks. In contrast, in the comparable period in the previous year the concentration was in the range of 1.0-1.5 mgO_2/L. The low DO concentration was most likely a consequence of increased oxygen uptake by the aerobic bacteria consuming available and excessive organic material. The aeration for biological purposes in MBR Varsseveld is controlled based on a NH_4/NO_3 concentrations ratio, which calculates an O_2-setpoint, which in turn, determines the required air flows. Due to low DO concentrations, the O_2-setpoint was increased by the control system. Yet, the additional oxygen was not provided. Hence, malfunction of the aeration equipment failure possible lead to the low DO concentrations. At DO concentrations lower than 0.5 mg/L, the nitrification process can be disturbed. Prolonged low DO concentrations are also reported to be detrimental for activated sludge filterability due to deflocculation processes leading to excessive growth of filamentous bacteria and release of fine particles (Wilén et al. 2000, Gil et al. 2011b). Nevertheless, due to the combination of different altering factors, i.e., low temperature, hydraulic and organic load increase, it is not possible to unequivocally determine the effect of low DO on filterability and operation.

6.5 Summary and conclusions

The influence of most important activated sludge characteristics on sludge filterability was analysed. Ten different MBRs in Belgium and the Netherlands, treating both municipal and industrial wastewaters, were sampled in both winter and summer. Based on the results discussed in the section 6.2, the following conclusions can be drawn. The lack of a clear correlation between a single investigated sludge parameter and any other foulant characteristic indicates that every single parameter alone is a poor indicator of biomass fouling propensity, a combination of activated sludge parameters is more reliable to predict sludge filterability. A combination of sludge morphology, characterized by the parameter %1pixel, and RH allows for a first classification of sludge filterability. Deflocculation and low RH have a negative impact on activated sludge filterability whereas good flocculation and high RH improve sludge filterability. For sludge samples that have poor to good activated sludge filterability, the model proposed by Van den Broeck et al. (2011) allows to estimate sludge filterability on the basis of eEPS PN, SMP PS, sludge morphology, the dissociation constant and RH. No correlation could be found between activated sludge filterability and membrane configuration or wastewater to be treated. Sludge filterability is independent of membrane configuration. In general, summer samples had better activated sludge filterability compared to samples collected in winter. This effect exceeds the mere temperature effect because filtration data are corrected for viscosity changes due to temperature. Overall, it can be concluded that the bioflocculation state of the activated sludge is a major indicator for activated sludge filterability.

The relations between the filterability of the activated sludge and the characteristics of the inflow were also discussed in this chapter. Regarding the impact of the inflow characteristics on filterability, low temperatures and sudden changes in the influent concentration and composition are the most common causes leading to poor sludge filterability. Moreover, the combination of undesirable events, e.g., hydraulic and/or organic load shocks, harmful composition of the incoming wastewater, is extremely difficult to overcome without filterability deterioration and operational problems. Furthermore, the impact of the influent COD concentration and loading is much higher in municipal than in industrial MBRs. Under stable operational conditions, the organic loading showed little influence on filterability in industrial plants, whereas, in municipal MBRs, the impact of increased organic sludge loading was more apparent. However, activated sludge loading should not be considered a predominant parameter with respect to filterability in municipal MBRs. The changes in the organic loading are not solely responsible for the changes in measured filterability, which are most likely a consequence of temperature variations or unexpected events. Therefore, temperature can be defined as a major influencing parameter with respect to activated sludge filterability. In municipal MBRs, the incoming organic loading can be considered as of significant influence. For the industrial MBRs, impact of incoming organic loading on filterability can be categorized as of less importance. Other parameters, like complexity of the treated wastewater, MLSS concentration or operational problems due to equipment malfunction, are more influential factors affecting the filterability of the industrial activated sludge.

In regard to operation and performance, the full-scale municipal MBRs achieved good removal efficiencies with only phosphorous removal being a matter of some concern. Without chemical phosphorous removal total phosphorous concentrations in permeate were often above the required discharge limit. Furthermore, all investigated MBRs were operated without major problems and with good or moderate activated sludge filterability unless abnormal events took place. Performance and operation of the full-scale MBR plants can be affected by abrupt temperature drop, high influent flow rates or composition of incoming wastewater. In those conditions, activated sludge filterability deteriorates, achieved permeability is decreased and chemical cleanings need to be performed more often. The membrane flux does not influence activated sludge filterability, whereas sludge filterability does indirectly impact membrane flux in full-scale MBRs. Prolonged operation of MBR with a poor filterable activated sludge should be avoided due to higher operational and maintenance costs. Moreover, substantial differences in filtration performance and operation were observed between summer and winter season. The permeability is generally worse in winter compared to summer periods and decreases when filterability deteriorates. Nevertheless, permeability alone is a weak parameter to monitor MBR performance and to determine the cause of poor filtration process.

Finally, a clear contrast between sidestream and submerged configurations in terms of plant operation was observed. The sidestream system operates at higher fluxes and most of the times achieved higher permeability, irrespectively of the filtration resistance rates.

CHAPTER 7

MBR plant layout and membrane configurations in relation with MBR operation

Parts of this chapter have been published in:
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P. Krzeminski, W. Langhorst, P. Schyns, D. de Vente, R. Van den Broeck, I.Y. Smets, J.F.M. Van Impe, J.H.J.M van der Graaf and J.B. van Lier (2012). The optimal MBR configuration: hybrid versus stand-alone – comparison between three full-scale MBRs treating municipal wastewater, Desalination 284, pp. 341-348

7 MBR plant layout and membrane configurations in relation with MBR operation

7.1 Chapter outline

This chapter discusses the practical knowledge concerning the impact of different MBR plant layouts and membrane configurations on the overall functioning of the MBR plant. Section 7.2 assesses full-scale MBR data comparing the use of flat sheet and hollow fibre membranes and analyses the consequences on operation, process performance, treatment efficiency and operational costs. Section 7.3 deals with an evaluation of a stand-alone MBR in comparison with two hybrid MBR configurations, i.e., in series and in parallel. The effect of design plants layout is discussed in terms of operation, performance and operational costs of the full-scale MBRs. The chapter, based on Krzeminski et al. (2012a, 2012c), is concluded with the summary of the results and main conclusions.

7.2 Impact of membrane configurations on MBR operation: flat sheet versus hollow fibre

7.2.1 Introduction

Three membrane configurations are predominant in the market: hollow fibre (HF), flat sheet (FS) and tubular, also called multitube (MT). In fact, all MBRs are equipped with one of those membrane or their modification (Santos and Judd 2010). The submerged configurations are exclusively fitted with HF or FS membranes, whereas sidestream technologies use mainly MT membranes (Santos et al. 2011). Advantages and disadvantages of each membrane configuration were discussed in the past (Stephenson 2000, Judd 2006, van Bentem et al. 2008, Brannock et al. 2010). However, despite a world-wide experience with full-scale applications, practical operational knowledge concerning the effect of different system configurations and membrane types on performance and operational costs are still lacking (Yang et al. 2006, Meng et al. 2009). Moreover, the selection of system configuration is a crucial step in the design of the MBR-plants and further plant operations. Nonetheless, the impact of membrane configurations on MBR functioning is underestimated. Thus, this information is of major importance to MBR-technology (Judd 2002). A better understanding on the impact of membrane configurations on MBR performances will allow further optimisations of MBR operations and, consequently, MBR cost reduction.

The aim of this research was to assess the impact of using either hollow fibre or flat sheet membranes on operational performances and plant efficiencies. An extensive multi-aspect comparison of four full-scale MBRs was implemented and a major monitoring campaign was carried out to investigate the impact of activated sludge filterability on MBR plant operations and performances. Experiments were performed with activated sludge samples from plants treating municipal and industrial wastewater, thereby including the parameter of wastewater strength into the comparison. This work provides information on the pre-treatment requirements, cleaning strategies and cleaning protocols applied in day-to-day operation of the full-scale plants. Consequently, it gives important information about frequency of necessary chemical cleanings for each configuration and for a trouble-free operation of the system. Furthermore, the results of this research allow the comparison of two systems in terms of energy consumption.

The specific objectives of this study were:

- to asses activated sludge filterability in full scale MBRs equipped with either hollow fibres or flat sheet membranes;
- to analyse and compare flat sheet and hollow fibre configurations in both industrial and municipal scenarios;
- to asses impacts of different membrane configurations on MBR performances.

7.2.2 Experiments description

A measurement campaign was performed for a period of nearly two years. During those measurements, activated sludge samples were collected from four full-scale MBRs and subjected to a filtration characterisation test. Samples were collected directly from the membrane tanks or as close as possible to the membranes. Furthermore, parallel to the filterability tests, plant operation and performance were monitored, analysed and compared with the other investigated plants. For this purpose, design, operational and membrane performance data were collected from each MBR for the respective periods.

7.2.3 MBR plants characteristics

The selected plants, namely MBR Heenvliet, MBR Varsseveld (Table 3.1) and MBR Fujifilm and MBR Rendac (Table 3.2), include MBRs treating municipal and industrial wastewater. The MBRs were equipped with different types of submerged membranes: two with Toray flat sheet membranes and two with Zenon (GE) hollow fibre membranes.

7.2.4 Activated sludge filterability

Experimental results, obtained during filtration characterisation tests of activated sludge, differ between the locations. The difference in activated sludge filterability can be clearly observed between plants treating municipal (locations A and C) and industrial (locations B and D) wastewater. Typical DFCm outputs are presented in Figure 7.1 where results of four filtration experiments, representing each MBR, are plotted. Filtration curves representing industrial MBRs are steeper than filtration curves representing municipal MBRs, which is related with a worse filterability of activated sludge.

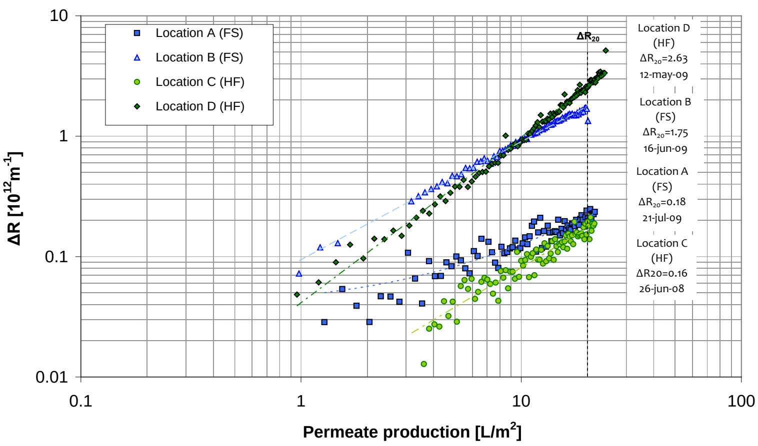

Figure 7.1: Filtration characterisation curves of activated sludge from 4 full-scale MBRs: location A – MBR Heenvliet, location B – MBR FujiFilm, location C – MBR Varsseveld and location D – MBR Rendac.

A strong relation between the filterability and the $\alpha_R \cdot c_i$ product, i.e., the mass involved in the cake layer build up, was observed with a R^2= 0.94. Therefore, likewise in previous cases described in sections 4.4 and 5.4, this indicates a strong influence of the concentration of substances accumulating in the cake layer on the total cake layer resistance. The higher the $\alpha_R \cdot c_i$ product values the worse the filterability of activated sludge and influent. The compressibility coefficient varies between 0 and 0.3. The s coefficient is mostly 0 when the activated sludge filterability is considered as good/moderate and ΔR_{20} values are below $0.5 \cdot 10^{12}$ m^{-1}. For the activated sludge with a poor filterability, the s coefficient is in majority above 0 indicating compressible cake layer. The values of $\alpha_R \cdot c_i$ product and s coefficient are presented in Table 7.1.

Table 7.1: Summary of the $\alpha_R \cdot c_i$ product and s coefficient results

MBR plant	Period	$\alpha_R \cdot c_i$ [$\cdot 10^{-3}$ m^{-2}]	s [-]
Heenvliet	jun/08	2±5	0
	jan/09	10±6	0
	jul/09	4±4	0
	feb/10	39±7	0.08
FujiFilm	apr/09	157±11	0
	mei/09	131±20	0
	jun/09	88±4	0
	okt/09	14±4	0
Varsseveld	jun/08	8±2	0
	feb/09	170±15	0.05
	aug/09	0±1	0
	mrt/10	37±5	0.08
Rendac	mei/09	98±8	0.34
	jun/09	44±1	0.05
	jul/09	3±3	0
	nov/09	212±8	0.24

As a general trend, activated sludge samples collected from municipal MBRs present better filterability (lower ΔR_{20} values) and are less prone to cause membrane fouling problems than the samples collected from industrial MBRs. Filterability results for each MBR, together with the MLSS concentration are shown in Figure 7.2.

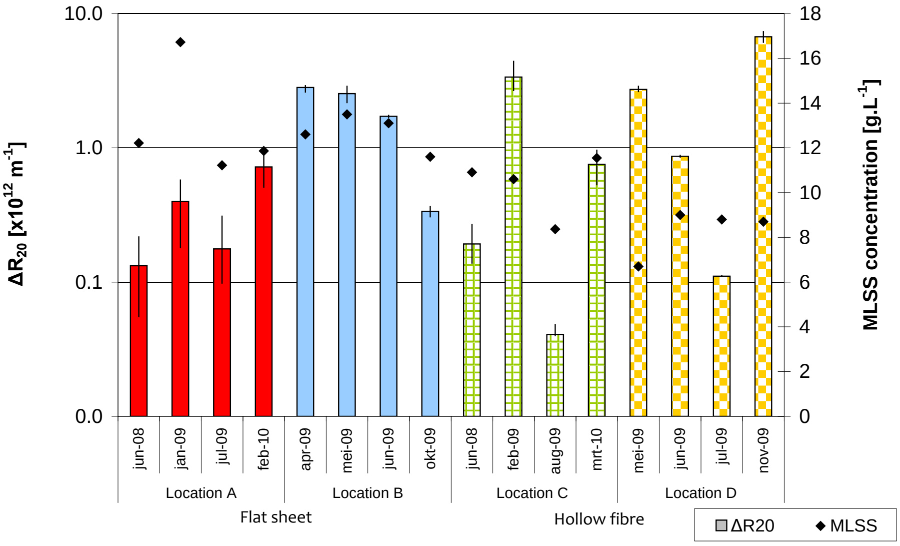

Figure 7.2: Activated sludge filterability (bars) and MLSS concentration (points)

In case of locations treating municipal influent, filterability is at least moderate ($\Delta R_{20} < 1.0$) unless extreme events take place, such as abrupt temperature changes, excessive snow melt or variations in the influent wastewater composition. In addition, significant differences in the temperature of the different activated sludge samples were observed between municipal and industrial MBRs. Seasonal fluctuations have a stronger temperature impact on municipal inflows compared to industrial inflows (Figure 7.3). The significant impact of the temperature on filterability in municipal MBRs was previously observed by Moreau (2010). With regard to the industrial locations, the activated sludge filterability is rather poor ($\Delta R_{20} > 1.0$) and less influenced by seasonal temperature fluctuations due to the constant and relatively high temperature of the industrial wastewaters, which was never below 16°C in winter.

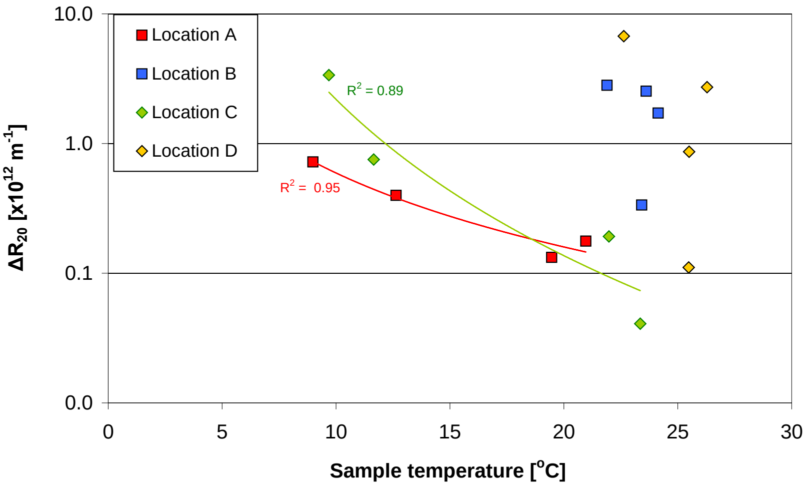

Figure 7.3: Activated sludge filterability and sample temperature

A clear relation between the filterability and MLSS concentration was not observed, which was previously reported by Lousada-Ferreira (2010a, 2011). Although there is no clear relation, higher MLSS concentrations in the membrane tank samples were observed for the flat sheet than for the hollow fibre configurations, i.e., between 11.5-16.7 g/L and 6.7-11.5 g/L, respectively. Moreover, visible differences in activated sludge filterability, in both municipal and industrial locations, were observed between the two investigated configurations (flat sheet - $0.67 \cdot 10^{12}$ m^{-1} and hollow fibre - $1.37 \cdot 10^{12}$ m^{-1}). Besides, in both cases, strong variations in the filterability were observed. Nevertheless, the differences in activated sludge filterability are likely caused by the biological process, the composition of the inflow and possible differences in the recirculation-aeration rates and not by the type of installed membranes. Also Moreau (2010) stated that activated sludge filterability is independent of the membrane configuration.

7.2.5 MBR operation

Detailed operational characteristics of each full-scale MBR are presented in Table 7.2. Hollow fibre configurations are designed to operate at higher fluxes compared to flat sheet configurations. However, the observed differences in the applied fluxes could be caused by lower flows treated by the flat sheet systems at the specific cases. Furthermore, in case of hollow fibre systems, MLSS concentrations were lower (6-10 g/L) than for flat sheet (10-17 g/L) systems. This is in agreement with the fact that, flat sheet MBRs can be, and usually are, operated with higher MLSS compared to hollow fibre MBRs. The flat sheet MBR can be operated with constant MLSS of 15 g/L and with a maximum peak of 18 g/L. On the contrary, the hollow fibre MBRs cannot operate at such high biomass concentrations and frequency of required chemical cleanings increases.

Table 7.2: Operational characteristics of investigated MBRs

Parameter	Unit	Location A	Location B	Location C	Location D
Membrane type	-	Flat sheet	Flat sheet	Hollow fibre	Hollow fibre
Total membrane area	m^2	4115	1680	20160	3520
Hydraulic capacity (RWF)	$m^3.h^{-1}$	100	35	775	120
Permeate production	$m^3.h^{-1}$	100	10	775	70-120
Design Flux	$L.m^{-2}.h^{-1}$	24	21	38	34
Average Flux (net; DWF)	$L.m^{-2}.h^{-1}$	12-24	6	15-25	39
Permeability	$L.m^{-2}.h^{-1}.bar^{-1}$	300	1200	263	76
TMP	bar	0.09	0.009	0.1	0.74
MLSS	$g.L^{-1}$	12-15	10-17	6-10	6-9
Temperature	^oC	14.7	30.8	16.6	29.9
SRT	days	40	160	24-26	29
SADm	Nm^3 air.$h^{-1}m^{-2}$	0.29	0.36	0.20-0.55	0.52
SADp	$Nm^3.m^{-3}$ permeate	12.85	17.1	12.32	15.4
Energy consumption	$kWh.m^{-3}$	0.95	1.54 - 4.45	0.83-0.88	0.11 (MBR) 6.63 (WWTP)

Legend: RWF – rain weather flow

Reported permeability values were similar for the municipal locations: 300 L/m^2.h.bar for location A and 263 L/m^2.h.bar for location C. Very likely, the applied lower flux in location A was mainly responsible for the observed differences between the flat sheet and hollow fibre systems. The observed high permeability value for the industrial location B can be ascribed to a decrease in the permeate extraction from 35 to 10 m^3/h since part of the industrial plant was closed. Consequently, operation under low TMP leads to a high permeability and high SRT. It is important to stress, that permeability, defined as flux over TMP, is thus a function of TMP and is not an independent parameter. Only when clean membrane and clean water are used the permeability could be considered as independent parameter. However, in the real life conditions, the mixed liquor with pollutants is filtered through the fouled membrane. It is also important to note that membrane permeability alone does not provide a clear picture of a current situation. Therefore, permeability is a weak parameter for monitoring of MBR performance (Geilvoet 2010, Moreau 2010, Gil et al. 2011). Additional information about MBR performance is needed to draft the full picture.

Results of activated sludge filterability analysis and relation with permeability are presented in Figure 7.4. It is observed that the permeability values decrease when the filterability increases (location B). If so, activated sludge filterability is not a limiting factor in the filtration process and the operation can be subjected for improvement. On the other hand, stable permeability even with a poor filterability (location D) indicates that a poor filterability is overcome by excessive operational circumstances (cost- and energy inefficient).

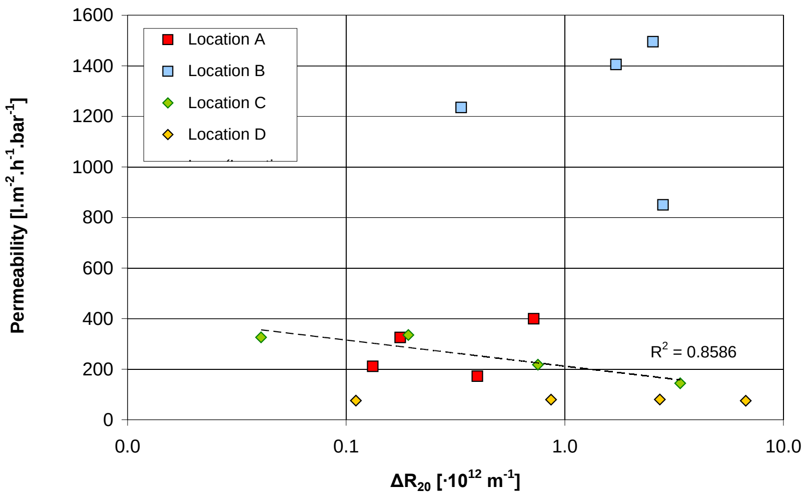

Figure 7.4: Activated sludge filterability and permeability

Effect of seasonal temperature fluctuations on membrane performance was observed when summer and winter permeability in municipal MBRs in location A and C are compared. During winter periods, the permeability decreased about 20-50% mainly due to activated sludge filterability deterioration.

The performed energy analysis revealed that the specific aeration demand per membrane surface area (SAD_m) was in the same range for the municipal facilities. For the industrial plants, specific aeration demand was lower in case of flat sheet installations compared to hollow fibre ones. Additionally, hollow fibre systems show a somewhat lower specific aeration demand per permeate produced (SAD_p) for both industrial and municipal applications. Table 7.2 also shows that municipal MBRs equipped with hollow fibre membranes consumed less energy than the ones with flat sheet membranes during the investigated period (0.88 $kWh.m^{-3}$ vs. 0.95 kWh/m^3). This is in accordance with the literature as hollow fibre systems are reported to require less energy compared to flat sheet systems. One explanation is higher packing density and, in result, in higher surface area to volume ratio that allows HF membranes to achieve the same degree of mixing with lower energy input. As expected, higher energy consumptions are observed for the industrial locations. For location B, the MBR was consuming 1.54 kWh/m^3 at full loading and 4.45 kWh/m^3 after part of the industrial plant was closed down. Table 7.2 lists two values for location D, owing to the fact that this location has retrofitted their actual WWTP including an MBR in the treatment line. The low value of 0.11 kWh/m^3 refers only to the membrane tank. The high 6.63 kWh/m^3 value takes into account both the conventional WWTP and the membrane tank. For comparison, the latter value should be considered since the MBR needs the conventional WWTP to achieve the discharge limits.

7.2.6 MBR treatment performance

During the research period of 2008-2009, COD removal efficiency accomplished the discharge limits in all the locations. Municipal MBRs removed COD to effluent concentrations of 25 mg/L with a removal efficiency between 93-95%. Industrial MBRs removed COD to a lesser degree, but with significantly higher inflow concentrations, i.e., 1566 mg/L for location B and 3973 mg/L for location D. Both systems present high level of robustness as COD was removed with efficiencies of 97-99% down to concentrations in the range of 40-46 mg/L. Slightly better efficiencies were observed for hollow fibre systems probably due to smaller pore sizes, i.e., 0.08 vs. 0.04 μm. BOD was removed below the required value of 10 mg/L to concentrations just above 2.6 mg/L.

TKN removal efficiencies achieved 96-99% and lowered effluent concentrations down to 1.1 mg/L for flat sheet and 2.0 mg/L for hollow fibre configurations. Location D achieved removal efficiencies around 99% with discharge limits around 6.8 mg/L.

The removal of phosphorous was achieved by means of biological uptake (location A) or by means of a combination of biological P-removal and chemical precipitation (locations C and D). Industrial location B does not measure phosphorus removal as its chemical activity is not related to this compound. Differences in applied strategy for phosphorous removal are clearly observed when total phosphorous concentrations in the effluent are analysed. In the case of flat sheet systems, when only biological treatment took place, removal efficiencies around 70% were obtained. In the case of hollow fibre systems, when also chemical treatment was used, efficiencies in the range of 90-96% were achieved. However, high dosage of ferric salt for phosphorous removal purposes may have adverse effects on the membranes and increases the salt content in the permeate. Table 7.3 summarizes the available influent and effluent characteristics data and removal efficiencies achieved in each plant.

Table 7.3: Influent & effluent characteristics and removal efficiency

Parameter	Unit	Location A	Location B	Location C	Location D
Influent characterization					
COD	mgL^{-1}	370	1566	723	3973
BOD	mgL^{-1}	164	-	302	-
NH_4^+-N	mgL^{-1}	-	47	-	713
NO_3^--N	mgL^{-1}	-	6	-	-
TKN	mgL^{-1}	46	-	60	827
TP	mgL^{-1}	7	-	13	17.3
Effluent quality					
COD	mgL^{-1}	25	46	25	40.2
BOD	mgL^{-1}	1.5	-	0.8	2.6
NH_4^+-N	mgL^{-1}	0.1	3.9	-	3.7
NO_3^--N	mgL^{-1}	2.5	5.5	2.7	11.9
TKN	mgL^{-1}	1.1	-	2	6.8
TN	mgL^{-1}	3.6	-	4.8	18.9
TP	mgL^{-1}	2.1	-	0.5	1.7
PO_4^+ - P	mgL^{-1}	1.9	1.0	0.5	-
Removal efficiency					
COD	%	93.2	97.1	96.5	99.0
TKN	%	97.6	-	96.7	99.2
TP	%	70.0	-	96.2	90.2
COD	%	93.2	97.1	96.5	99.0

In general, industrial MBRs demonstrated better overall removal efficiencies due to higher concentrations in the influent but residual effluent pollutant concentrations were slightly higher.

7.2.7 Pre-treatment and cleaning strategies

Pre-treatment is required to remove coarse materials, hairs and other fibrous material in order to protect installed membranes. Hence, importance of proper wastewater pre-treatment in the MBR is essential for the overall process (Frechen et al. 2008). In the municipal MBRs a two step mechanical pre-treatment is installed, i.e., 6 mm screen and 3 mm or 1 mm sieve, whereas in the industrial MBRs one but more rigorous step is installed, i.e., a fine screen of 0.5-0.75 mm. In both municipal and industrial plants, hollow fibre membranes are protected by stricter mechanical pre-treatment. Information about the pre-treatment, cleaning strategies and cleaning protocols applied in the full-scale MBRs with respect to membrane types are summarized in Table 7.4.

Table 7.4: Pre-treatment and cleaning strategies applied in flat sheet and hollow fibre MBRs

Parameter	Unit	Location A	Location B	Location C	Location D
Membrane type	-	Flat sheet	Flat sheet	Hollow fibre	Hollow Fiber
Membrane supplier	-	Toray	Toray	Zenon-GE	Zenon-GE
Pre-treatment	mm	6 mm screens + 3 mm punch-hole	0.75 mm microsieve	6 mm grid removal + 1.0 mm microsieve	0.5 mm microsieve
Filtration duration	min	9	9	6	6
Physical cleaning duration	sec	60 (relaxation)	60 (relaxation)	25 (backwash)	25 (backwash)
Chemical cleaning frequency		Yearly	Yearly	Weekly	Weekly + 2·CIP per year
Cleaning agents	-	1st Citric acid + 2nd NaClO	NaClO	1st NaClO + 2nd Citric acid	1st NaClO + 2nd Citric acid
Chemical cleaning - decision		$TMP_{since\ last\ cleaning}$ > 0,1 bar or TMP > 0,2 bar	Perm < 700	-	$P_{operation}$ < 0,530 bar $P_{backwash}$ >1,5
Chemical cleaning - duration	min	360	120-180	109-145	21 (CIP = 600)

During operation membranes are subject to fouling process that results in permeability decline. To control fouling an appropriate cleaning protocol is required. Distinct differences are observed in applied cleaning methods between the two types of membrane configurations.
The hollow fibre membranes are protected by the membrane air-scouring that can be applied continuously but also periodically, e.g. on intermittent, sequencing or proportional basis. On the contrary, the flat sheet membranes need to be scoured with air continuously. According to the flat sheet membrane manufacturer's, intermittent aeration is not suitable for their membranes. However, other concepts, like proportional and pulse aeration are an option for cleaning the flat sheet membranes at the lower aeration energy cost.
Furthermore, flat sheet membranes are cleaned physically through relaxation periods of 1 min performed every 9 minutes of filtration. Additionally, when further filtration cannot be sustained because of an increase in the TMP intensive chemical cleaning is performed on a yearly basis (once or twice per year). During chemical cleaning most prevalent cleaning agents are used: citric acid and sodium hypochlorite in Location A or sodium hypochlorite alone as is the case in Location B.
Hollow fibre membranes are cleaned physically through the backwash stage (25 sec) performed every 6 minutes of filtration. The physical cleaning is supported by the maintenance chemical cleaning performed every week (can be extended to two weeks). Comparing to the flat sheet cleaning protocol, chemicals are applied in reverse order: first sodium hypochlorite and second citric acid. Hollow fibre membranes are cleaned more frequently both physically (backwash) and chemically. The latter one, results in 2.4 times higher chemicals cost comparing to FS case. However, when cost are normalized for the installed membrane area, the specific chemical cleaning cost (expressed in €/m^2 membrane area/year) is double for FS comparing to HF installation. Hence, costs of chemical cleaning are higher for HF membranes but, at the same time, cleaning of 1 m^2 of the hollow fibre membrane is more cost effective than flat sheet membrane. The higher cleaning cost for HF system can be associated with the more frequent cleanings applied (and as such with the

amount of chemicals consumed) but might be also the effect of the plant scale and bigger membrane area to clean. Nevertheless, each treatment facility can have specific chemical cleaning protocols, especially in industrial locations, i.e., chemical concentrations and cleaning frequencies, as recommended by the membrane suppliers (Le-Clech et al. 2006).
Typical cleaning strategies, differences in the cleaning protocols between the membrane systems and the costs related to the chemical cleaning are further discussed in section 8.3.13.1.

7.3 Impact of MBR configurations on its operation: hybrid versus stand-alone

7.3.1 Introduction

Due to increasing popularity and acceptance as municipal wastewater treatment process alternative, the amount of full-scale MBR plants is continuously increasing (Judd 2011). Growth in plant numbers is accompanied with diversity of configurations and design concepts. The MBR technology is commonly applied to new WWTPs, but they are also introduced in case of upgrades or retrofits of already existing WWTPs (Brepols et al. 2008, Baag 2009, Lesjean et al. 2009). There are several reasons why wastewater treatment plants need to be modernized, e.g., old and out-dated infrastructure, equipment upgrade, more stringent effluent quality requirements and insufficient hydraulic or biological capacity due to increasing pollution load.

There are also different options on how to modernize WWTPs with MBR technology. Whereas optimal design selection is very individual and site specific, two general solutions can be distinguished. One of the options is to completely replace the existing system with MBR technology, with or without reuse of the old infrastructure. In this case all incoming wastewater is treated in a stand-alone MBR (Figure 7.5a), also called separate, complete, full or classic MBR (Giesen et al. 2007, Bixio et al. 2008). Another option is to utilize existing buildings and infrastructure to combine old and new processes into a hybrid system, also known as dual configurations (Bixio et al. 2008, Kraume and Drews 2010). In this hybrid design, part of the wastewater is treated in the CAS process and part is treated in the MBR. As such, CAS treatment is combined with MBR treatment which can be operated either in parallel (Figure 7.5b) or in series (Figure 7.5c).

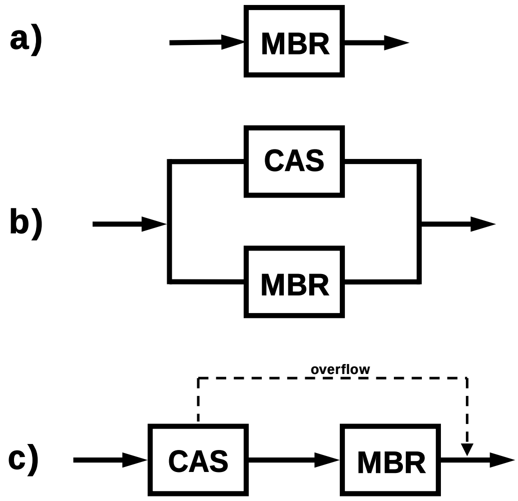

Figure 7.5: Schematic representation of basic configurations: (a) stand-alone, (b) parallel and (c) serial

Advantages of hybrid MBRs compared to stand-alone MBRs are well reported in the literature (Bixio et al. 2008, Brepols et al. 2008, Giesen et al. 2008, Mulder et al. 2008, Bixio et al. 2009) and include lower membrane surface requirement due to treatment of the peak flows outside of the MBR and continuous operation at optimal designed conditions which results in energy efficient operation. Additional advantages are possible in case of plant retrofit, e.g., extended lifetime of the old CAS system, cost effective option of WWTP retrofit and infrastructure utilization reducing the investments costs. One of the obvious disadvantages of the hybrid concept is the bigger footprint of the plant because of the required surface needed for the CAS system and possibility of incidental discharge of suspended solids, bacteria and viruses due to potential overflow or bypassing of the peak flows via the CAS. In addition, when less membranes are installed they need to be frequently used and, in consequence, rest periods of the membranes are limited. Therefore, the life-time of the membranes due to often shorter ‘out of operation’ periods is probably shorter.

Hybrid MBR concepts were successfully established for treatment of domestic wastewater at full-scale in Schilde (De Wilde et al. 2005, Garcés et al. 2007, De Wever et al. 2009, De Wilde et al. 2009), Heenvliet (Mulder et al. 2005, Mulder et al. 2007, Mulder et al. 2008, Mulder 2009, Lousada-Ferreira et al. 2010b), Ootmarsum (Futselaar et al. 2007, Geraats and de Vente 2008, Futselaar et al. 2009a, Futselaar et al. 2009b), Rietliau (Frechen et al. 2009), Viareggio (Battistoni et al. 2006, Fatone et al. 2007), Ulu Pandan (Verrecht et al. 2010), Terneuzen (Mulder et al. 2010), St. Peter ob Judenburg, Brescia and Eitorf (Brepols et al. 2008).

The recent full-scale results and experience significantly broadened the understanding of the associated processes. However, despite the increase in applicability, information regarding the influence of a particular configuration on operation and performance is only scarcely available. Moreover, available information is scattered and hardly published and thus the transfer of knowledge is very limited.

Our present work provides information on design concepts and plant configurations, and their impact on operation, performance, energy consumption and economy of the MBR plant. The specific objective of this study is to evaluate a stand-alone MBR in comparison to a hybrid concept of MBR design, i.e., a combination of a CAS process and an MBR. Finally, advantages and disadvantages of each particular configuration are presented and discussed.

7.3.2 MBR plants description

Three full-scale MBR plants treating municipal wastewater and located in the Netherlands were under investigation. Two of the plants are hybrid installations and one is a stand-alone MBR. The investigated plants, namely MBR Heenvliet, MBR Varsseveld and MBR Ootmarsum, are described in detail in Table 7.5 (see also Table 3.1).

Table 7.5: Characteristics of the plants

Parameter	Unit	Hybrid 1 (Heenvliet)	Hybrid 2 (Ootmarsum)	Stand-alone (Varsseveld)
WWTP configuration	-	MBR+CAS	MBR+CAS/SF	MBR
MBR configuration	-	parallel and serial	parallel	stand-alone
Location	-	Heenvliet	Ootmarsum	Varsseveld
Membrane type	-	Flat sheet (FS)	Tubular (MT)	Hollow fibre (HF)
Membrane supplier	-	Toray	Norit	Zenon-GE
Total membrane area	m^2	4,115	2,436	20,160
Biological Capacity	PE	3,333	7,000	23,150
Hydraulic capacity (DWF)	$m^3.h^{-1}$	38-50	75	250-300
Hydraulic capacity (RWF)	$m^3.h^{-1}$	100	150	755
Average Flux (DWF)	$l.m^{-2}.h^{-1}$	12-24	26-40	15-25

Legend: SF – sand filter; DWF - dry weather flow; RWF - rain weather flow

7.3.3 Data collection, processing and analysis

Three full-scale MBR plants were monitored for a period of 2 years, both in summer and winter period. During this period filterability of activated sludge, as a quality indicator of the MBR filtration process, was quantified experimentally by the DFCm. The filterability results were compared with automated image analysis results and collected process data of the plants. Several process parameters of each MBR are monitored and collected at each location. These data were provided by the regional water authority (i.e. Water Boards) responsible for the managing of respective WWTPs. The most important parameters that were under investigation include influent flow, permeate flow, flux, transmembrane pressure, permeability, temperature, pH, MLSS and DO concentration. In addition, several characteristics of influent and effluent were analysed, e.g., COD, BOD, TKN, TN and TP. Additionally, parallel to the energy consumption study, plant performance was monitored and analysed in respect of their potential indirect relation with energy consumption. Together with the removal efficiency information, the performance of the MBR plants was evaluated in environmental and economical terms based on major performance indicators as presented in the research approach (section 1.3).

The energy consumption data, reported as kWh, are based on the electric power consumed at each investigated location. The specific energy consumption data are reported as specific electricity consumption per volume of treated wastewater and expressed as kWh/m^3. During the energy studies, total and specific energy consumption data were analysed, emphasizing the relation to treated flow, design capacity, membrane area and effluent quality. Additionally, economic studies were performed analysing the cost efficiency in design and operation of the full-scale MBR plants (Krzeminski et al. 2011a).

7.3.4 MBR operation

Differently designed municipal MBRs had a very similar type of activated sludge morphology, as confirmed by the microscopic activated sludge images (Figure 7.6) and by the filtration characterisation tests performed on samples originating from the membrane tanks. However, significant differences in activated sludge filterability, expressed by the ΔR_{20} parameter, were observed between the seasons (Figure 7.7). Also Moreau (2010) and Van den Broeck et al. (2011) observed improved activated sludge filterability in summer as compared to winter samples.

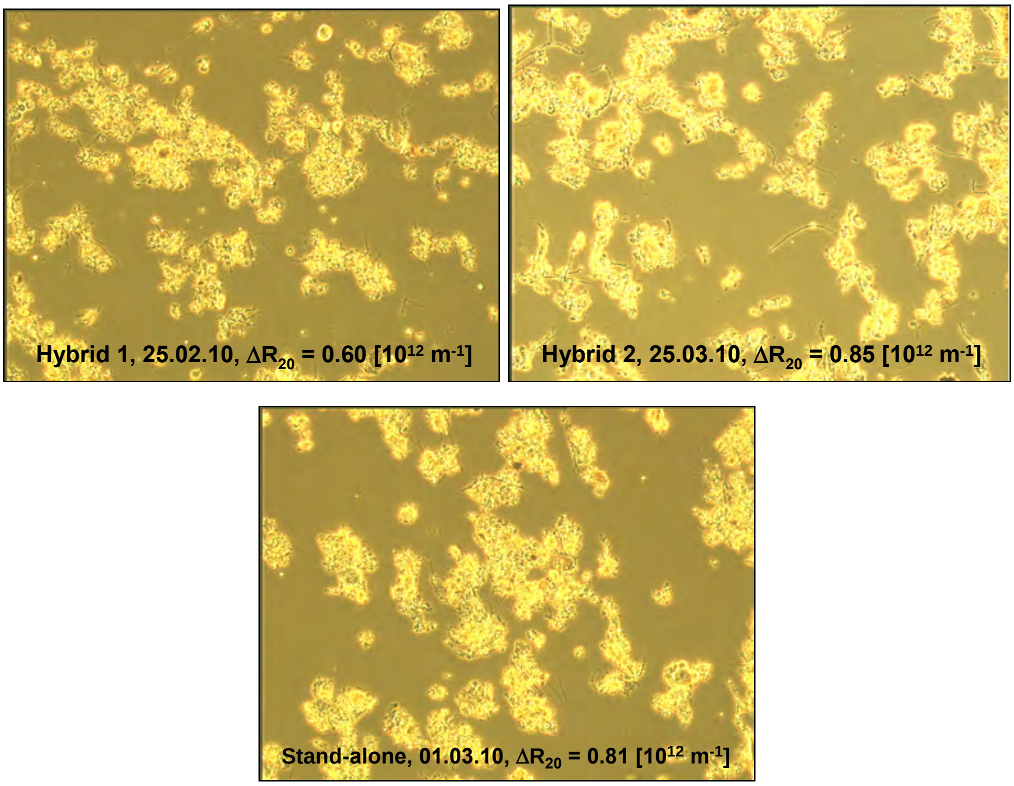

Figure 7.6: Microscopic images (x100, Ph1) of activated sludge from: (a, b) hybrid and (c) stand-alone MBRs

Filterability of activated sludge measured in the three plants was qualified mainly as moderate ($0.1<\Delta R_{20}<1.0$) or good ($\Delta R_{20}<0.1$) during summer periods of 2008 and 2009, respectively. In the case of Hybrid #1 MBR, samples were described as moderately filterable during both experimental campaigns. The results obtained during winter periods show in general moderate activated sludge filterability unless abnormal events appear such as a peculiar chemical composition of the incoming wastewater or a rapid temperature drop due to heavy storm and snow melt. The latter phenomenon was observed in the winter of 2009 and consequently poor filtration behaviour was observed. An abnormal event can occur in both hybrid and stand-

alone configurations and often results in poorly filterable activated sludge. As a consequence, operation of the MBR is hampered and the performance can be affected.
In a stand-alone MBR, activated sludge is submitted to more frequent and rapid changes due to variations in the characteristics of incoming flow and results in unsteady-state operation (Judd 2006). Whereas depending on the operation concept of the hybrid system, i.e., parallel or serial, the probability of an operational upset as a consequence of activated sludge quality deterioration varies. In a parallel system, MBR and CAS are operated as two separate and stand-alone treatment plants. Hence, the likelihood of the operational upset is similar to the one in a stand-alone MBR. Conversely, in a hybrid MBR operated in series, the CAS system precedes the MBR and creates a buffer zone that provides the required time for the microorganisms to adapt to new conditions and consequently more stable conditions for the activated sludge are achieved (Cicek et al. 1998, Le-Clech et al. 2003, Drews et al. 2005). This advantageous effect was most likely the reason for the better activated sludge filterability observed in Hybrid #1 MBR, both in summer and winter, during in series operation in 2008-2009, compared to parallel operation in 2009-2010 (Figure 7.7).

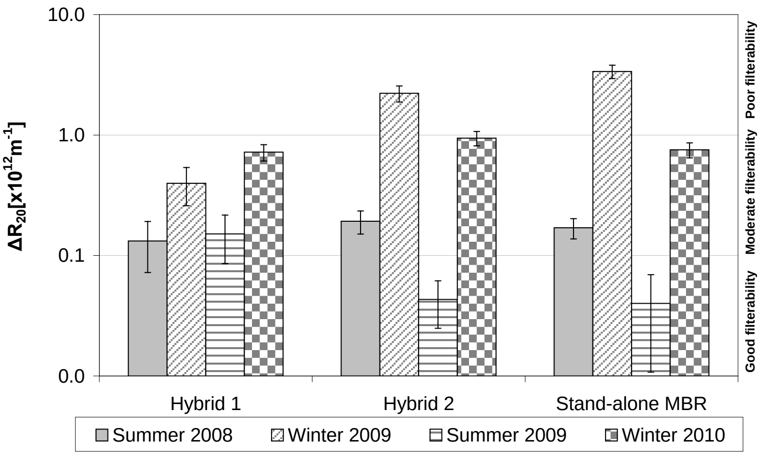

Figure 7.7: Average filterability (ΔR_{20}) of activated sludge samples from hybrid and stand-alone MBRs during the sampling campaigns

Stand-alone and hybrid configurations also differ in an operational strategy concerning excess flow treatment during rain weather conditions, i.e., peak flows, in case of connection to the combined sewer system. In a stand-alone MBR, incoming stormwater is treated exclusively by the MBR and may result in a nearly 3.5 times higher flow compared to the average dry weather flow. Typical flow patterns of the incoming wastewater, expressed as a 1 hour trend line and as a function of plant utilization (% of nominal Dry Weather Flow, i.e., incoming flow during dry weather conditions), in both configurations are presented in Figure 7.8.

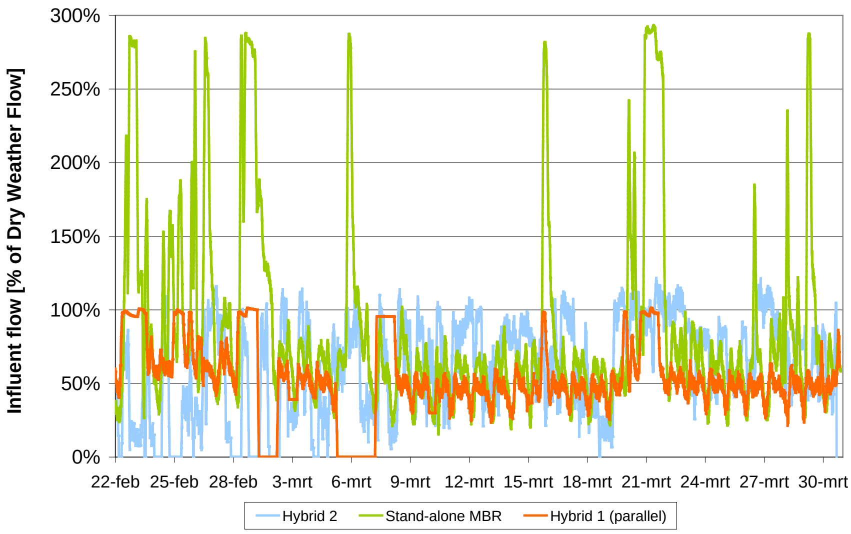

Figure 7.8: Comparison of MBR influent flow in hybrid and stand-alone MBRs

Furthermore, in a stand-alone MBR configuration, overflow or bypassing of peak flows is not possible. Hybrid MBRs on the other hand, most often have an overflow option through the CAS system. Hence, one of the key advantages of the hybrid concept is the possibility of dealing with peak flows that exceed the hydraulic membrane capacity of the MBR. This results not only in lower membrane surface requirements but it also has an influence on the operation of the MBR. It provides operational flexibility for the operators and allows them to react upon certain situations. Therefore, in most cases, it enables stable MBR operation as the plant is less sensitive to abrupt changes, e.g., temperature shifts and hydraulic flow fluctuations. Contrary, in the stand-alone configuration, the whole system is more vulnerable to rapid changes, e.g., intensive rainfall combined with the snow melt results in a severe temperature drop of incoming wastewater and can seriously affect membrane operation for several hours or even a few days. The aforementioned reasons were responsible for an 80% increase in the gross flux and a 40-45% drop of permeability in the winter 2009 period (Figure 7.9). Figure 7.9 compares applied fluxes, activated sludge filterability and process permeability in three MBRs during the consecutive weeks of experimental periods.

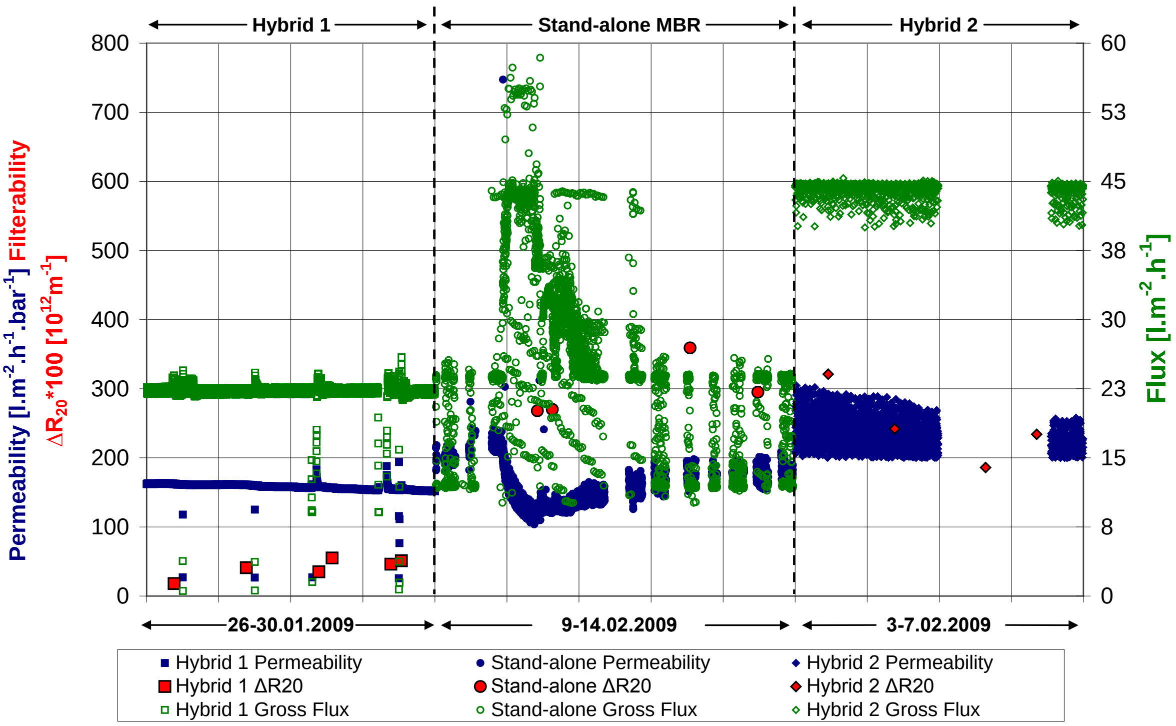

Figure 7.9: Comparison of applied fluxes, activated sludge filterability and process permeability in Hybrid #1, Hybrid #2 and the stand-alone MBR

7.3.5 MBR treatment performance

All three types of investigated MBRs removed COD, BOD and TN far below national and European discharge requirements with efficiencies of about 92-95% and 98-99% for COD and BOD, respectively. Good removal efficiencies of TKN were also achieved in all of the plants, i.e., 96-98% removal to concentrations of about 2.0 mgN/L.

Phosphorous removal efficiency in the hybrid MBRs was in the range of 67-72% and 73-74% in 2008 and 2009, respectively. Phosphorous removal of 94-96% reaching TP concentrations of 0.4-0.7 mgP/L was attained in the stand-alone MBR. However, better performance in the stand-alone MBR is not a result of particular design selection, but an effect of phosphorous removal in combination with the dosage of iron chloride sulphate. In the investigated hybrid MBRs chemicals are not added, which resulted in phosphorous removal of 67-74% and somewhat higher concentrations in the effluent, namely 1.7-2.2 mgP/L.

In case of hybrid MBRs the permeate produced by the MBR is mixed with the effluent of the CAS system before final discharge. Mixing of the CAS effluent and MBR permeate had negligible effect on the quality of the total combined effluent produced in both hybrid MBRs. For example, in case of the Hybrid #2 MBR the COD, BOD and TN concentrations increased up to 30 mg/L, 2.3 mg/L and 3.8 mg/L, respectively; values still below the requirements. Furthermore, in some cases, concentrations can actually be lower than in the MBR permeate as observed for TN (Hybrid #1) and TP (Hybrid #2) in Table 7.6, in agreement with the predictions of Futselaar et al. (2007). Hence, it can be concluded that the selection of a hybrid MBR configuration for communal wastewater treatment plants has no significant impact on effluent quality, especially with respect to the current discharge requirements. Nevertheless, potential differences in the effluent quality could probably be observed in the concentration of total suspended solids and in terms of disinfection, i.e., presence of bacteria and viruses. However, those parameters were not measured during this project.

A summary of the overall performance of the investigated MBRs, in terms of pollutants removal efficiency, with minimum, average and maximum values, is presented in Table 7.6.

Table 7.6: Effluent characteristics and removal efficiencies of the three investigated MBRs

MBR Performance			Permeate from MBR					Total effluent from WWTP					Removal efficiency			
			COD	BOD	TN	TP	TKN	COD	BOD	TN	TP	TKN	COD	BOD	TP	TKN
			mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	%	%	%	%
Hybrid 1	2008	Min	8	1.0	0	0.3	0.5	12	1.0	1.1	0.5	0.8	92	98.5	67	98
		Mean	25	1.7	3.0	2.2	1.0	28	2.0	3.5	2.4	2.0				
		Max	94	44	13	7.6	3.0	55	14	13	5.5	6.0				
		St. dev.	9	4.3	1.9	1.3	0.4	6	1.6	1.7	1.1	1.1				
	2009	Min	6	1.0	0.7	0.2	0.3	5	0.3	0.9	0.5	0.2	93	99.0	73	97
		Mean	24	1.3	4.2	1.9	1.1	28	2.0	3.5	2.3	1.9				
		Max	58	3.9	8.6	5.8	3.0	49	5.0	7.6	5.1	4.2				
		St. dev.	7	0.5	1.8	1.4	0.4	7	0.9	1.0	1.1	0.7				
Hybrid 2	2008	Min	15	0.5	1.4	0.1	0.6	16	0.5	1.9	0.6	0.7	95	99.5	72	97
		Mean	23	0.9	3.6	2.0	1.5	24	1.3	3.7	1.6	1.5				
		Max	46	2.2	8.7	11	7.7	40	3.3	6	6.4	4.4				
		St. dev.	5	0.4	2.0	2.1	1.1	4	0.7	1.1	1.1	0.6				
	2009	Min	15	0.5	1.5	0.1	0.5	19	0.5	1.4	0.2	0.9	95	99.6	74	96
		Mean	24	0.8	3.6	1.7	1.7	30	2.3	3.8	1.2	2.0				
		Max	39	2.1	8.8	5	7.3	71	9.3	11	4.0	6.8				
		St. dev.	5	0.3	1.7	1.4	1.2	10	1.8	1.9	0.8	1.1				
Stand-alone MBR	2008	Min	16	0.5	1.4	0.1	0.8	16	0.5	1.4	0.1	0.8	96	99.7	96	96
		Mean	25	0.9	3.9	0.4	2.1	25	0.9	3.9	0.4	2.1				
		Max	33	1.9	15	1.2	13	33	1.9	15	1.2	13				
		St. dev.	5	0.4	2.9	0.3	2.5	5	0.4	2.9	0.3	2.5				
	2009	Min	12	0.5	2.3	0.1	1.0	12	0.5	2.3	0.1	1.0	96	99.7	94	96
		Mean	25	0.8	5.8	0.7	1.8	28	0.8	5.8	0.7	1.8				
		Max	36	1.6	20	3.2	3.8	36	1.6	20	3.2	3.8				
		St. dev.	6	0.3	3.8	0.7	0.7	6	0.3	3.8	0.7	0.7				

7.3.6 Energy consumption

The specific energy consumption of the MBR in Hybrid #1 varied between 0.8 and 1.8 kWh/m^3 and was on average 1.1 kWh/m^3. For the total plant, thus for the combined MBR and CAS systems at Heenvliet, the specific energy consumption ranged between 0.3 and 1.1 kWh/m^3 and was on average 0.6 kWh/m^3. During the project, Hybrid #1 has been operated both in parallel and in series. When considering those two operational concepts – serial and parallel – a clear difference is observed in favour of the serial concept. The average energy consumption during in series operation was 0.75 kWh/m^3 compared to 1.15 kWh/m^3 during parallel operation, mainly due to a better utilization of the available membrane capacity (Figure 7.11).

It was also observed that the Hybrid #1 MBR consumed less energy for operation and maintenance compared to the stand-alone MBR, although only during in series operation. The specific energy consumption of the stand-alone MBR varied between 0.6 and 1.4 kWh/m^3 and was on average 0.84 kWh/m^3, approximately 12% more than Hybrid #1 MBR during in series operation. However, after 5 years of operational experience with the stand-alone MBR further energy reduction is planned with a goal to reach 0.7 kWh/m^3 during normal MBR operation (Van Bentem et al. 2010).

Figure 7.10 presents the energy consumption distribution of the MBR equipment for the hybrid MBR operated in series (Figure 7.10a) and in parallel (Figure 7.10b) as well as for stand-alone MBR (Figure 7.10c). Observed differences arise rather from the different membrane configurations installed in each plant, namely flat sheet and hollow fibre, and consequently certain operational requirements such as aeration strategy, than from the selected design configuration.

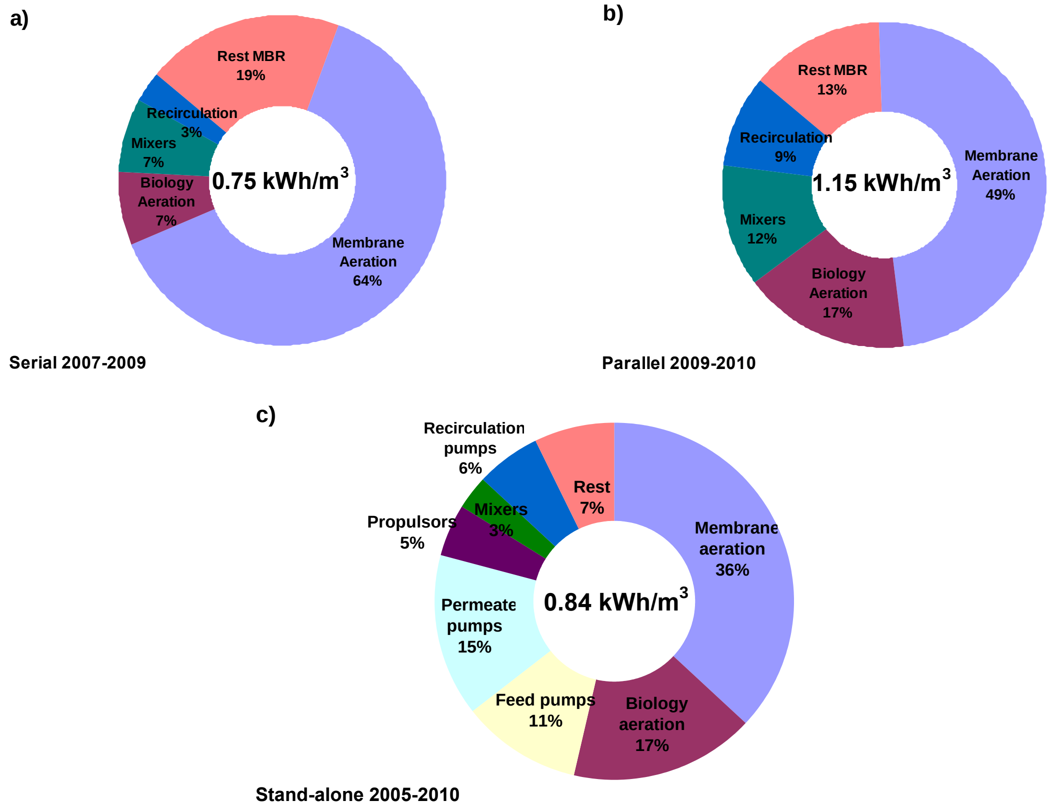

Figure 7.10: Energy consumption distribution of MBR equipment for Hybrid #1 MBR during (a) serial and (b) parallel operation and for (c) stand-alone MBR

According to Figure 7.11a, the specific energy consumption in the Hybrid #1 MBR is higher during the period of parallel operation compared to in series operation. The permeate production was reduced by more than a half and consequently the specific energy consumption of the MBR increased while at the same time the total energy consumption decreased. Therefore, despite sub-optimal flow conditions, operation following the parallel concept can be energy efficient leading to specific energy consumption in the range of 0.8-1.0 kWh/m^3. However, at least 50% overall MBR utilization is required.

The impact of membrane capacity usage on energy consumption follows from the efficiency increase with the treated flow (Figure 7.11). Operation of the underloaded (60% of DWF) stand-alone MBR result in specific energy consumption of 1.05 kWh/m^3, whereas operation at the capacity exceeding 100% result in low specific energy consumption of 0.63 kWh/m^3. Operation of the underloaded (47% of DWF) hybrid MBR results in specific energy consumption of 1.15 kWh/m^3 whereas operation at the about 125% utilization results in specific energy consumption of 0.97 kWh/m^3. Hence, the highest specific energy efficiency is attained when operating the MBR under optimal flow conditions, i.e., close to the design flow at dry weather conditions.

Figure 7.11b shows the flow dependency of differently designed MBR plants, i.e., one stand-alone and two hybrid MBRs. It also shows the added value of CAS implementation in the hybrid concept. The specific energy consumption of the total Hybrid #1 WWTP, is nearly half (40% lower) of the Hybrid #1 MBR whereas the concentrations of TN and TP in the entire WWTP effluent increased by a maximum of 0.5 mg/L and 0.4 mg/L, respectively.

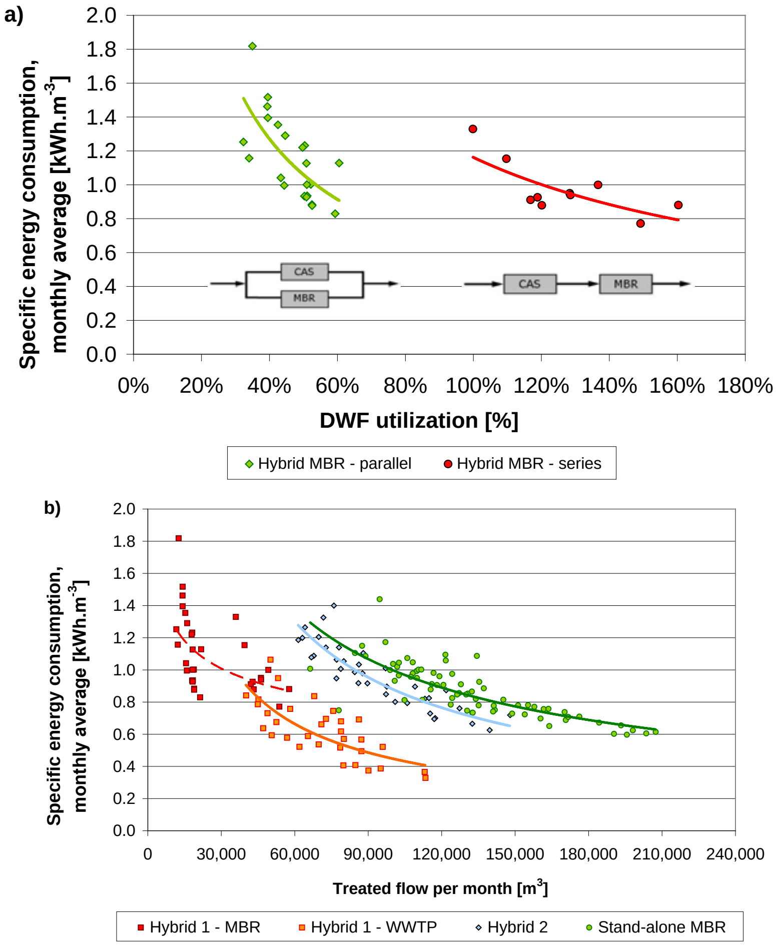

Figure 7.11: Energy consumption in relation to the flow for different design concepts: (a) Hybrid#1 in parallel (2008-2009) and serial (2009-2010); and (b) Hybrid#1 MBR (2008-2010) and Hybrid#1 WWTP (2008-2010), Hybrid#2 (2008-2010) and stand-alone (2005-2010)

Figure 7.12 presents the specific energy consumption as a function of the plant design capacity. In general, the capacity of the plant does not determine the energy efficiency of the installation. The observed improvement for Hybrid #1 is a logical consequence of an operational concept change from serial to parallel where only a small fraction, i.e., 25%, of the influent is treated in the MBR. Higher specific energy consumption values for Hybrid #2 MBR are only partially explained by the energy consumption associated to the sand filter which is incorporated in the CAS process line. Another explanation could be found in excessive aeration and limited possibilities of fine-tuning and reducing blower input for the aeration. The fact that the biological load is lower and the alpha-factor is better than expected during the design also contributes to excessive aeration, which, owing to technical reasons, cannot be lowered. Furthermore, presence of small basins and compartments with numerous, not optimally operated, small mixers and abovementioned aeration issues contribute to the higher energy consumption in Hybrid #2.

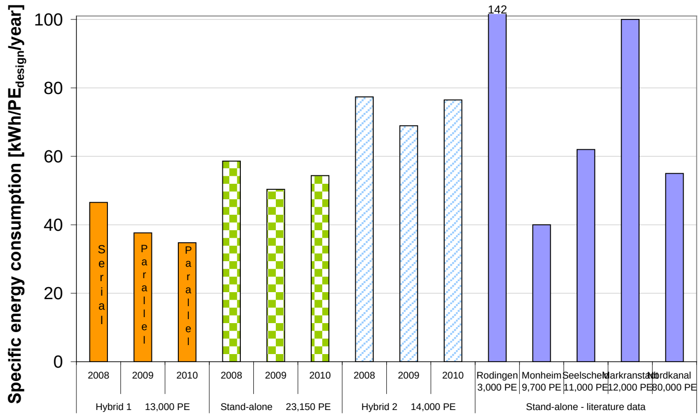

Figure 7.12: Energy consumption as a function of plant design capacity (1 PE_{design} is equal to a pollution load of 54 g BOD/day) for a selection of German MBRs (adopted from Pinnekamp, 2008)

The normalized energy consumption of the entire plant, expressed in kilowatt-hours per removed pollution load, was 81 kWh/$PE_{removed}$, 86 kWh/$PE_{removed}$ and 58 kWh/$PE_{removed}$ for the Hybrid #1, Hybrid #2 and stand-alone MBR, respectively. The $PE_{removed}$ value is the pollution load removed in the WWTP based on WWTP removal efficiencies in the 2008-2009 period and expressed as population equivalents based on 150 g of total oxygen demand (TOD, equal to COD + 4.57·TKN). Therefore, in this particular case, the stand-alone MBR required less energy to remove the same amount of pollutants than the hybrid MBRs, and is as such more energy efficient in this aspect.

The specific energy consumption per area of installed membranes was lower for the stand-alone MBR equipped with hollow fibre membranes. Thus, in terms of membrane surface specific energy consumption (in kWh/m^2), big MBR installations are more energy efficient compared to the small ones. Additionally, operation of sidestream membranes is the most energy demanding. However, because sidestream systems can apply higher fluxes, it needs less membranes than submerged systems and thus requires lower capital costs. When results are compared for similar capacity, sidestream systems require approximately 60-70% less membranes. Therefore, design of hybrid installations with tubular sidestream membranes allows to significantly reduce the required membrane area and possibly, if the price of tubular membranes (€/m^2) is not 2.5-3.3 times more expensive than submerged membranes, to lower capital costs even further.

7.3.7 Operational and capital costs

The selection of a particular configuration has an impact on the capital expenditure (CAPEX) and the operational (OPEX) expenditures. In general, retrofitting an old conventional treatment plant with the hybrid MBR concept is more cost effective than replacing the entire system with a stand-alone MBR (Evenblij et al. 2007, Futselaar et al. 2007, Kraume and Drews 2010). However, reusing old infrastructure while retrofitting an old WWTP into a stand-alone MBR also reduces capital costs (Hashimoto et al. 2009). As has been previously noted, hybrid concepts benefit from lower membrane surface requirement and, consequently, reduced investments costs. Besides, retrofitting a plant allows to utilize old structures to further reduce capital costs. Moreover, installed equipment can be designed for and operated at stable average flows in order to provide optimal work conditions and cost efficient operation.

At the same time investment costs for a hybrid plant in the case of new WWTPs can be higher due to the larger footprint of the plant because of the required land surface needed for the CAS system. In addition, installation of a smaller amount of membranes is also associated with some drawbacks. The lifetime of the membranes might be, depending on operational strategy, shorter due to the continuous operation of the filtration step (Cote et al. 2012). In the hybrid configurations the membranes have often shorter ‘out of operation’ periods and therefore, are likely aging faster compared to the membranes in stand-alone configurations. Necessary adaptation to the treatment of peak flows requires a larger membrane surface which can, if configured so, create multiple process lines which can be operated alternately. In this situation higher number of installed membranes allows resting the membranes more frequently and probably extends their service life (Giesen et al. 2008).

Hence, determination of the optimal plant configuration depends on the particular local situation, i.e., presence and condition of old infrastructure, availability of equalization tanks and space requirement. In the case of a CAS retrofit, a hybrid configuration is usually preferred if the CAS system is still in good condition. However, in case of a new WWTP, the stand-alone concept has the potential to be the most optimal option. Additionally, when retrofitting an old WWTP one should seriously consider utilization of old infrastructure to equalize peak flows.

Average operational costs of the stand-alone MBR were 0.29 €/m^3 of treated wastewater in 2009 (Figure 7.13c). Dosage of iron chloride sulphate for chemical phosphorous removal results in additional chemical costs (9%, 0.026 €/m^3) but also in significantly lower phosphorous concentrations in the effluent. Addition of required capital expenditure costs increases the total costs to 0.45 €/m^3.

Average operational costs of the total Hybrid #2 were 0.24 €/m^3 over a period of 2008-2009 (Figure 7.13b). The average operational costs of the total Hybrid #1 plant, thus, combined CAS and MBR, were 0.13 €/m^3 over a period of 2008-2010 (Figure 7.13a). When only the MBR is considered, the operational costs increase to 0.29 €/m^3. Furthermore, during 22 months of parallel operation average operational costs of the MBR were 0.37 €/m^3 comparing to 0.17 €/m^3 during 14 months of in series operation. Obviously, the MBR in the parallel concept is hindered by operation under sub-optimal flow conditions and is consequently less cost-efficient, approximately by a factor of 2. In the parallel concept, costs of MBR operation are close to, yet still lower, than costs of stand-alone MBR operation. Therefore, both hybrid concepts are associated with lower operational costs compared to stand-alone MBR which is in accordance with Verrecht et al. (2010). Also Bixio et al. (2008, 2009) analysed the potential and economical aspects of two hybrid solutions for WWTP refurbishment in Bulgarian and Cyprus' markets. They reported a possibility of minimal cost reduction of 20-25% if hybrid MBR is selected.

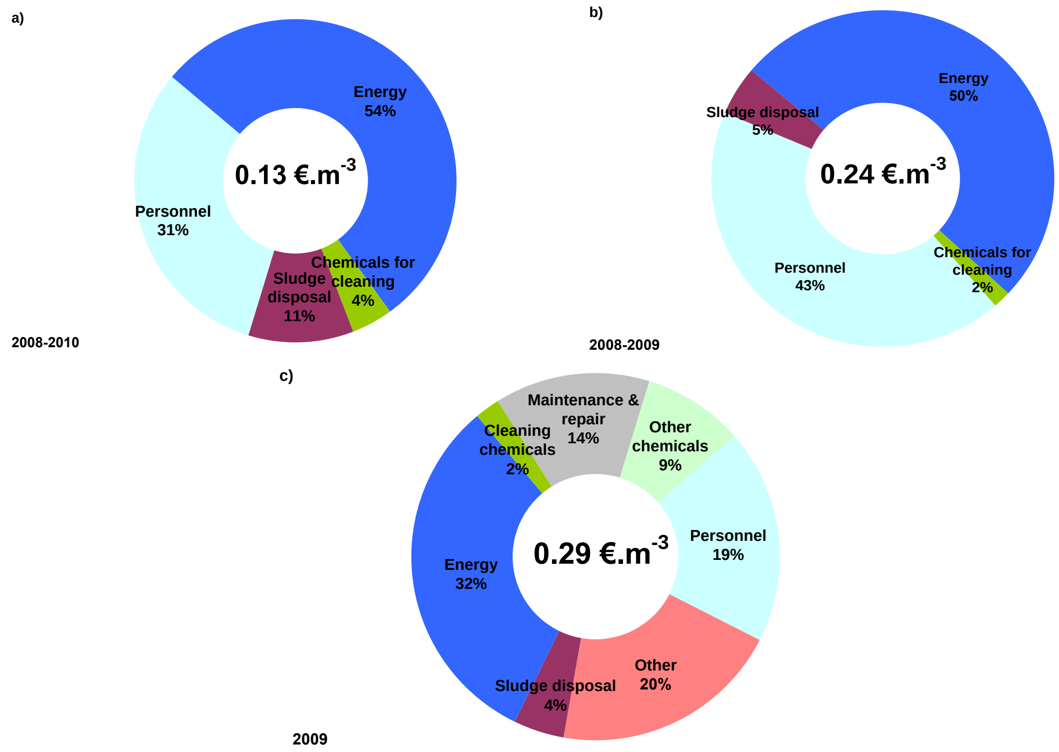

Figure 7.13: Operational cost distribution for (a) Hybrid #1 (b) Hybrid #2 and (c) stand-alone MBRs

The normalized costs of the plant operation, expressed in Euro per removed pollution load per year, were 15.6 €/$PE_{removed}$, 23.6 €/$PE_{removed}$ and 17.2 €/$PE_{removed}$ for Hybrid #1, Hybrid #2 and stand-alone MBR, respectively.

7.4 Summary and conclusions

Four full-scale MBRs treating municipal and industrial wastewater were investigated in terms of impact of membrane configuration on MBR operation. Striking differences were observed between flat sheet and hollow fibre membranes applied in full-scale MBRs.

- In both municipal and industrial plants, hollow fibre membranes were protected by stricter pre-treatment and were cleaned more frequently physically (backwash) and chemically.
- Moreover, hollow fibre configurations were designed to work at higher fluxes, but were operated at lower MLSS concentrations compared to flat sheet configurations.

- Samples collected from municipal MBRs presented better filterability than samples from industrial MBRs. Also seasonal fluctuations have a stronger temperature impact on municipal inflows compared to industrial inflows. Results indicate that this is likely due to differences in the prevailing biological processes or the influent composition and not to the type of installed membranes.
- All investigated MBRs meet the required BOD and COD discharge limits.
- The energy consumption depends more on the influent type than on the membrane configuration, as opposite trends were observed for industrial and municipal locations.

Furthermore, three full-scale MBR concepts in the Netherlands, i.e., one stand-alone MBR and two hybrid MBRs were carefully evaluated. When analysing the performance data it was clear that the principle choice of configuration, i.e., hybrid or stand-alone, largely impacts the overall MBR functioning.

- In the stand-alone and parallel operated hybrid MBRs, activated sludge is more often subjected to unsteady-state conditions that increase the likelihood of an operational upset. A stand-alone MBR is the most vulnerable to rapid changes compared to the hybrid configurations. Hybrid configurations provide operational flexibility. Especially in the serial hybrid concept, the CAS system acts as a hydraulic and biological buffer zone which ensures more stable conditions for the activated sludge in the MBR.
- A sudden perturbation may occur in both configurations, resulting in poorly filterable activated sludge. As a consequence, operation of the MBR is hampered and the performance can be affected.
- Selecting a hybrid MBR configuration and the associated mixing of MBR permeate with CAS effluent has no significant impact on final effluent quality, especially with respect to the current discharge requirements. There is also no substantial difference in stand-alone and hybrid MBR effluent quality.
- The specific energy consumption of the stand-alone MBR was approximately 12% higher than the hybrid MBR during in series operation, yet 27% and 0.3 kWh/m^3 lower than the hybrid MBR during parallel operation. The specific energy consumption is, however, highly dependent on membrane capacity usage; the highest specific energy efficiency being attained when operating the MBR at optimal flow conditions. The MBR in the parallel concept is hindered by operation under sub-optimal flow conditions and is consequently less cost-efficient than serial hybrid MBR, approximately by a factor of 2.
- In case of hybrid MBR, the incoming flow should be, if possible, directed fully to the MBR and the peak flows should be handled by the CAS system. This would increase membrane utilization and improve the energy efficiency of the MBR.
- Hybrid concepts are associated with at least 17% lower operational costs, compared to a stand-alone MBR but the stand-alone MBR required less energy to remove the same amount of pollutants than the hybrid MBRs, and are as such more energy efficient in this aspect.

Concluding, selection of particular membrane configuration has no effect on the permeate quality, negligible effect on activated sludge filterability and apparent effect on operation, e.g. achieved fluxes, biomass concentration, pre-treatment and cleaning strategies. Moreover, it has indirect influence, mainly through associated aeration strategy, on energy demand and consumption of the installation. The selection of specific MBR configuration during the design stage has more significant consequences on the MBR functioning. The activated sludge filterability, operational flexibility and reliability as well as lower operational costs favour hybrid MBR over a stand-alone MBR. Furthermore, the aforementioned features have influence on operation and thus on energy consumption and efficiency. Nevertheless, energy consumption and energy efficiency depends mainly on the membrane utilization and particular configuration of the selected hybrid concept. Even though, determination of the optimal plant configuration depends on the particular local situation such as the presence and condition of old infrastructure, availability of equalization tanks and space requirement. For example, when an old CAS system is available it is more economically feasible to extend the existing plant into hybrid MBR than to stand-alone MBR.

Furthermore, the final choice will depend on local regulations and required effluent quality. If high quality effluent is continuously required a stand-alone MBR should be the one to choose. However, if the high quality effluent is not always required, then the hybrid MBR is an option to consider.

CHAPTER **8**

Energy consumption and energy efficiency of full-scale membrane bioreactors

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8 Energy consumption and energy efficiency of full-scale membrane bioreactors

8.1 Chapter outline

This chapter provides an overview of current electric energy consumption of full-scale municipal MBR installations and available energy reduction opportunities based on literature review and case studies analysis. Moreover, operational processes associated with aspects of energy efficiency are investigated in this study.

In this chapter the energy and economical aspects of the membrane bioreactors are discussed. In section 8.2 literature review and methodology are presented. Section 8.3 discusses the results of the energy analysis in respect to design and operational issues of the full-scale installations. Section 8.4 and 8.5 provides an analysis on energy efficient operation and potential energy savings in the MBRs. The chapter concludes with a recapitulation and some concluding remarks in section 8.6.

The results of this part of the research were presented in Krzeminski et al. (2011b). The chapter is based on Krzeminski et al. (2012d).

8.2 Introduction

Despite continuous technological development, energy demand and related costs issues are, together with the membrane fouling issues, major drawbacks that restrict further expansion. In particular, high aeration rates for frequent membrane cleaning remain a challenge in terms of energy consumption and optimisation of MBRs (Judd 2006, Verrecht et al. 2008). To investigate the specific energy requirements of MBRs, determine realistic operational costs and elucidate where possible future energy consumption reduction can be achieved, extensive research on the energy consumption in full-scale MBR plants was performed.

8.2.1 MBR plants description

Four full-scale MBR installations treating mainly municipal wastewater in the Netherlands were investigated and assessed (Table 3.1). The selected MBRs include plants equipped with flat sheet and hollow fibre membranes submerged in the separate filtration tank along with plant equipped with sidestream externally placed tubular membranes.

The Heenvliet, Varsseveld and Ootmarsum MBRs were monitored in respect to energy consumption, operation and performance. In addition, energy consumption of MBR Terneuzen was investigated (see section 8.3.1).

8.2.2 Data collection, processing and analysis

This analysis was performed based on the data collected and provided by the regional water authority (i.e. Water Boards) responsible for the managing of Heenvliet, Varsseveld and Ootmarsum plants, and also Evides Industriewater operating MBR in Terneuzen. The energy consumption data, reported as kWh, are based on the electric power consumed at each investigated location. The specific energy consumption data are reported as specific electricity consumption per volume of treated wastewater and expressed as kWh/m^3. Additionally, parallel to the energy consumption study, plant performances was monitored and analysed in respect of their potential indirect relation with energy consumption. The performance of the MBR plants was evaluated in environmental and economical terms based on major performance indicators as presented in section 7.4.3:

- effluent concentration of pollutants (mg/L),
- removal efficiencies of pollutants expressed as % of incoming load, and
- energy consumption per volume of treated wastewater (kWh/m^3).

8.3 Results and discussion

8.3.1 Background information

In order to better understand the presented results and discussion, following remarks needs to be pointed out:

- MBR in Heenvliet is a hybrid configuration allowing operation of the MBR in a twofold manner: in parallel or in series to existing conventional activated sludge system. This has certain consequences on operation and thus on energy consumption, energy efficiency and subsequently cost implications. The configuration aspect is discussed in detail in *Chapter 7*.
- Unfortunately, detailed energy analysis of MBR Ootmarsum was not possible. The available energy consumption data are for the entire treatment plant, thus, the MBR plus the CAS with sand filter. Due to lack of installed electricity measurement devices, available energy data are restricted to the total energy for the WWTP and energy for aeration purposes only. Hence, the extensive analysis of the Ootmarsum MBR system was not feasible and only total plant consumption can be considered. Therefore, MBR Ootmarsum was not discussed in terms of energy consumption and for the comparison studies another tubular installation, namely MBR Terneuzen, was introduced.
- In case of MBR Terneuzen, due to recent start-up of the system certain problems occurred. Poor filterability of activated sludge was often observed during first months of operation. In addition, aerators inside the tubular membranes were clogged and required air flow rates increased.
- Aeration for biological reasons is low in case of Terneuzen because of addition of the CAS effluent into MBR. The design population equivalent is based on the influent values and does not include additional volume coming from the CAS effluent. Hence, when energy consumption is presented in kWh/m^3 it is with a benefit for MBR Terneuzen.

8.3.2 Conventional activated sludge systems vs. membrane bioreactors

The energy consumption of membrane bioreactors is often compared with CAS wastewater treatment systems and is reported to be 30-50% (STOWA 2005, Lazarova et al. 2010), 75-90% (Van Bentem et al. 2008, Van Bentem et al. 2010) or 10% to 100% higher to CAS energy consumption (Livingstone et al. 2009). The difference arises from the fact that the authors compared different MBR concepts and CAS plants with specific design and operational characteristic. For example, Mizuta and Shimada (2010) analysed electric energy consumption at 985 Japanese municipal WWTPs and reported consumption of CAS system to be between 0.3-1.9 kWh/m^3. Whereas the former value is beyond the potential of current MBRs, the latter one is easily achievable in most of well operated full-scale MBR. However, also much lower energy consumption values for CAS systems are reported. The CAS energy demand, expressed per volume of treated wastewater, widely ranges, being 0.1-0.2 kWh/m^3 (Gnirss and Dittrich 2000), 0.2-0.3 kWh/m^3 (Ueda et al. 1996), 0.3 kWh/m^3 (Yang et al. 2010), 0.4 kWh/m^3 (Van Bentem et al. 2008), 0.5 kWh/m^3 (Judd 2006), 0.4-0.6 kWh/m^3 (Cornel et al. 2003) and 0.9-2.9 kWh/m^3 for industrial applications (Cummings and Frenkel 2008).

Due to intensive membrane aeration rates required to manage membrane fouling and clogging, MBRs' energy consumption was three times higher even when compared with CAS systems combined with advance treatment techniques (Gnirss and Dittrich 2000). However, the gap was significantly reduced in the past years. Nowadays, the MBR energy requirement is comparable to CAS with tertiary treatment (Brepols et al. 2010b) yet still 10-30% higher (Van Bentem et al. 2008, Van Bentem et al. 2010). It should be noted, however, that a fair comparison of MBR systems with CAS systems is only possible when similar effluent quality is produced. Based on the results of hybrid municipal WWTP Ootmarsum, the quality of MBR permeate and effluent from CAS system combined with a sand filtration could be initially assumed comparable (Table 8.1). Also Xu et al. (2012) reported average quality of sand filter effluent to be about 40 mg/L of COD, 2.5 mg/L of TN and 0.5 mg/L of TP. Therefore, compared to about 25 mg/L of COD, 4.5 mg/L of TN and 1.5 mg/L of TP commonly achieved in municipal MBRs (section 6.4.2), it again could be assumed that a CAS system in combination with sand filtration is capable of providing comparable, or even better, quality of the produced effluent than MBR process. However, despite comparable carbon and nutrients removal, the sand filter effluent quality is not comparable with the MBR permeate due to presence of microorganisms and suspended solids in the effluent. Sand filtration is a well known method for removal of particulate matter including heavy metals and up to 50% removal of pathogenic microorganisms (Høibye et al. 2008). However, the removal of fecal coliform, coliphages, pharmaceuticals and personal care products is limited compared to the one achieved in MBR process (Zhang and Farahbakhsh 2007, Oulton et al. 2010). Thus the presence of viruses and bacteria is more likely in case of sand filtration effluent compared to MBR permeate. Meaning, a direct comparison between MBR and even CAS with a sand filtration is not appropriate and should include disinfection in order to achieve comparable effluent quality (Brepols et al. 2010b). Furthermore, CAS systems are often accompanied by the energy recovery via anaerobic digestion, which might be included in the total energy balance of the plant and leading to low energy consumption. Some of the optimised CAS plants can even go to energy neutral operation when energy consumption is compensated by the energy production.

Table 8.1: Quality of the permeate from the MBR and effluent from the CAS combined with sand filtration at Ootmarsum municipal WWTP

	MBR permeate				CAS + sand filter effluent			
	COD mg/L	BOD mg/L	TN mg/L	TP mg/L	COD mg/L	BOD mg/L	TN mg/L	TP mg/L
2008	23 ± 5	0.9 ± 0.4	3.6 ± 2.0	2.0 ± 2.1	24 ± 5	1.3 ± 0.4	3.6 ± 1.3	1.2 ± 0.7
2009	24 ± 5	0.8 ± 0.3	3.6 ± 1.7	1.7 ± 1.4	26 ± 6	1.7 ± 1.0	4.0 ± 2.1	0.7 ± 1.0
2010	24 ± 4	0.7 ± 0.3	5.9 ± 4.6	2.2 ± 1.5	29 ± 5	1.3 ± 0.4	3.9 ± 1.6	0.7 ± 0.7
Avg.	24 ± 5	0.8 ± 0.3	4.4 ± 3.3	2.0 ± 1.6	26 ± 5	1.4 ± 0.6	3.8 ± 1.6	0.9 ± 0.8

Nevertheless, Krause and Dickerson (2010) and Krause et al. (2010) clearly stated that operation of a full-scale municipal MBR with a total energy demand at the same range as a CAS process having an energy requirement of 0.5 kWh/m^3 is possible provided new MCP and optimized PLC programming is used.

8.3.3 Energy consumption in full-scale municipal MBRs

Detailed energy consumption data for three MBR installations are summarized and presented in Table 8.2.

Table 8.2: Summary of energy consumption data of investigated MBR installations

Location		MBR Heenvliet	MBR Varsseveld	MBR Terneuzen
Period of study		2008-2010	2005-2010	2010
Design dry weather flow [m^3/month]		36,000	180,000	288,000
Treated flow [m^3/month]		27,826	132,054	169,984
Monthly power requirement [kWh]				
	max	33,869	146,051	166,332
	average	22,700	110,486	154,636
	min	14,165	58,408	146,581
Daily power requirement [kWh]		1,788	N.A.	5,888
Yearly power requirement [kWh]		227,001	1,325,833	N.A.
Specific energy consumption [kWh/m^3]				
	max	1.82	1.44	1.28
	average	1.06	0.84	0.97
	min	0.77	0.60	0.76
Specific energy consumption in 2008 [kWh/PE$_{removed}$]		89	67	N.A.

8.3.4 Specific energy consumption per permeate production

The specific energy consumption, expressed in kWh/m^3 of permeate, for each MBR analysed on a long term scale is presented in Figure 8.1.

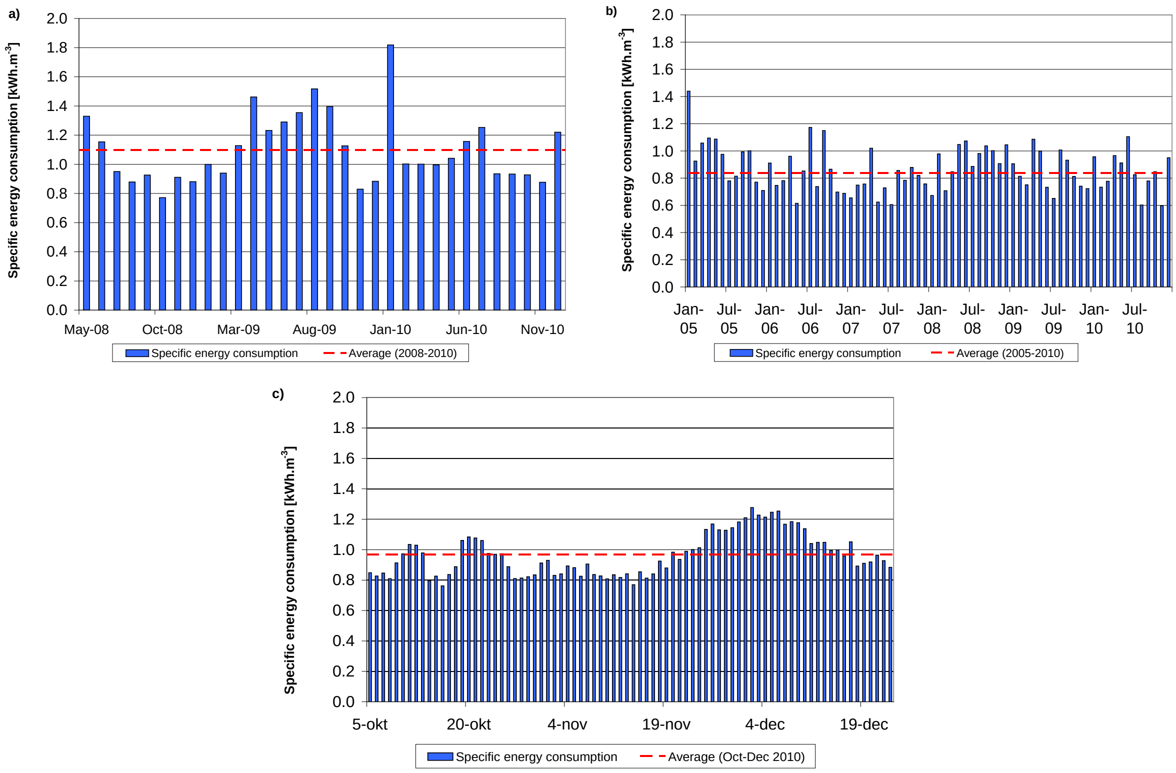

Figure 8.1: Specific energy consumption per volume of treated wastewater for: (a) Heenvliet (FS), (b) Varsseveld (HF), (c) Terneuzen (MT) MBRs

The specific energy consumption of the MBR Heenvliet varied between 0.8 and 1.8 kWh/m^3 and was on average 1.1 kWh/m^3 (Figure 8.1a). For the total plant, thus for combined MBR and CAS systems at Heenvliet, the specific energy consumption ranged between 0.3 and 1.1 and was on average 0.6 kWh/m^3.

The specific energy consumption of the MBR Varsseveld, presented in Figure 8.1b, varied between 0.6 and 1.4 kWh/m^3 and was on average 0.8 kWh/m^3. The total energy consumption was reduced from the initial value 1.1 kWh/m^3 after the start-up of the plant to 0.8 kWh/m^3 after 6 months of operation (Giesen et al. 2006). For the next 3 years energy consumption was slowly but steadily reduced (Van Bentem et al. 2008). After 5 years of operational experience further energy reduction is expected with a goal to reach 0.7 kWh/m^3 during normal MBR operation (Van Bentem et al. 2010).

Figure 8.1c shows the energy results, based on daily values, from the first operational period of the MBR Terneuzen. The energy consumption of the yet not optimized installation varied between 0.8 and 1.3 kWh/m^3 with an average consumption of 0.97 kWh/m^3. It is important to stress that major problems are usually visible during the plant start-up but also with long term experience. Hence, comparison between MBR Terneuzen and other MBRs already operated for many years should be done carefully. Nevertheless, it is expected that after start-up and the optimisation period energy consumption will be reduced to the design values of 0.5-0.6 kWh/m^3 (Mulder 2011). Typical specific energy consumption values for a tubular airlift MBR are reported to be in the range of 0.4–1.0 kWh/m^3 (Van 't Oever 2005, Judd 2006, Helble and Mobius 2009). In 2008, specific energy consumption of only UF installation, i.e., sludge circulation, membrane aeration, permeate and backwash pumps, in Ootmarsum was reported to be lower than 0.4 kWh/m^3 (Borgerink and Schonewille 2008) and in 2009 in the range of 0.2–0.3 kWh/m^3 (Futselaar et al. 2009). This is lower than the currently achieved 0.7-0.8 kWh/m^3 in Terneuzen, however, similar to the design estimation for UF system of 0.3 kWh/m^3 (Van 't Oever and Roman 2011). Detailed distribution of energy consumption components for each MBR is presented in Figure 8.2.

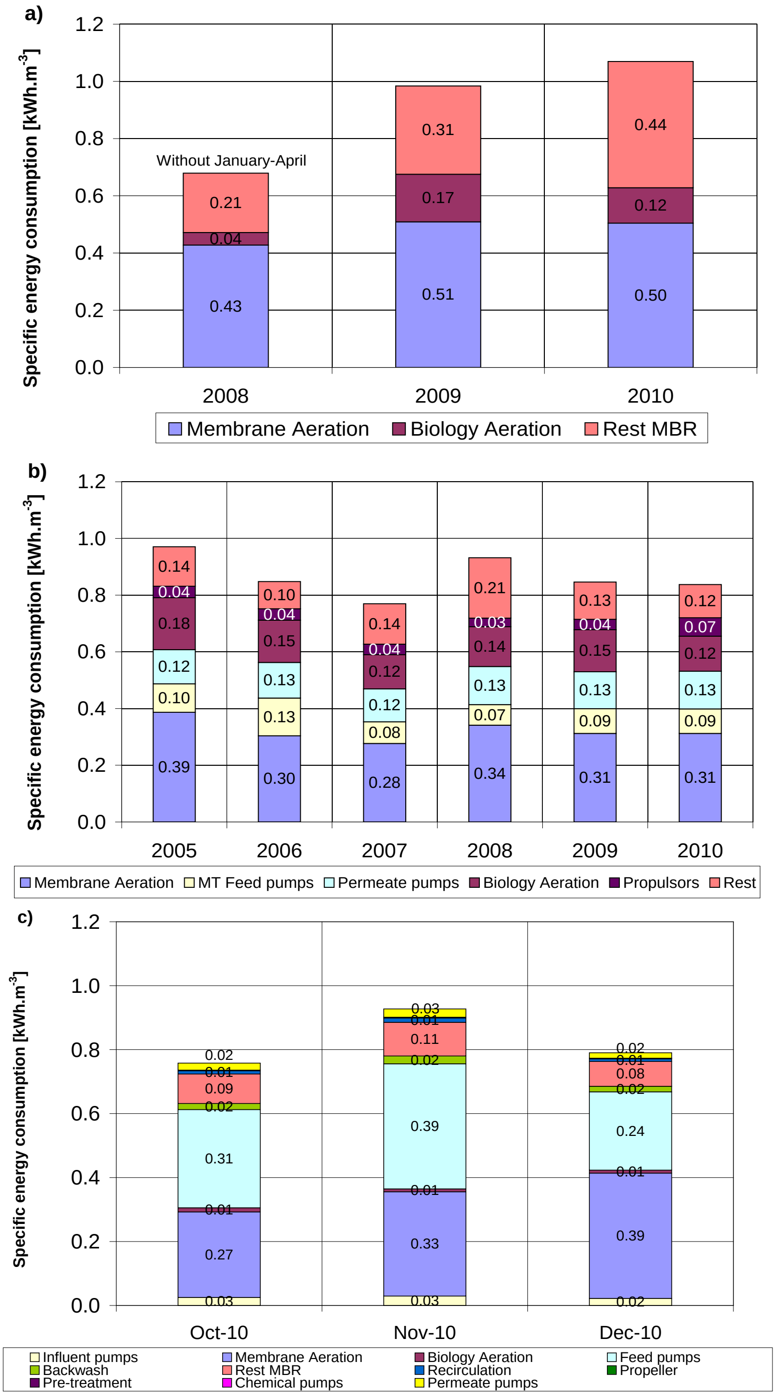

Figure 8.2: Specific energy consumption distribution of equipment for: (a) Heenvliet (FS), (b) Varsseveld (HF), (c) Terneuzen (MT) MBRs

The specific energy consumption of MBR Heenvliet increased from 2008 to 2010 (Figure 8.2a). This can be explained twofold. Firstly, the volume of treated flow in the MBR decreased leading to higher specific energy consumption. Secondly, the 2008 data do not include data from January-April, a period when heating of WWTP building and offices is significantly contributing to higher specific energy consumption.

Figure 8.2b shows an increase in the specific energy consumption in year 2008 very likely due to maintenance works that were performed in the membrane tanks of MBR Varsseveld. At that time, in order to prevent membrane fouling or clogging, process settings for the MBR operation were much more conservative, i.e., higher aeration rates and increased recirculation. In 2009 the settings were optimized again, resulting in lower energy consumption.

In case of WWTPs connected to a combined sewer system, like Varsseveld, the specific energy consumption strongly depends on the weather conditions and amounts of treated flow. The volume of treated flow in the MBR Varsseveld was 10% and 15% lower in 2008 and 2009, respectively, compared to the previous years. Hence, since Varsseveld experienced very dry months, the energy consumption per m^3 was higher leading to a high yearly average in 2009 and 2010. However, compared to other dry months in the past, the plant was actually performing much better in terms of total energy consumption. The energy consumption of the blowers producing air for membrane scouring was based on cyclic aeration: 15 seconds on and 15 seconds off. Implementation of more economical aeration strategy, so called “eco-aeration”, developed by the membrane supplier (Zenon-GE) and based on 10/30 intervals, could potentially save 50% of currently consumed energy (Buer and Cumin 2010). The total specific energy consumption of the MBR could then be about 0.7 kWh/m^3 and as such reaching the set goals (Van Bentem et al. 2008, Van Bentem et al. 2010). Nevertheless, as reported by Judd (2011) certain over aeration might be beneficial from operational point of view, as it reduces the chances of unscheduled manual intervention at the plant caused by fouling or clogging.

Due to recent start-up, only a short period of MBR Terneuzen operation was investigated during this study. Nevertheless, interesting observations in respect to aeration and feed pumps energy consumption were made. Contrary to other MBRs, the feed pumps and not membrane aeration were the most energy intensive process (Figure 8.2c). Because of the aerators clogging the air-scouring rates had to be doubled to provide efficient cleaning of the membranes. In result, energy consumption of the membrane aeration was higher than anticipated and accounted for 0.3-0.4 kWh/m^3. To balance excessive energy input and air-scouring generation, membrane aeration was utilized for the biological aeration and allowed to reduce energy required for biological aeration to just 0.01 kWh/m^3.

8.3.5 Distribution of energy consumption

Various energy users exist in the wastewater treatment plants and different operational processes consume different amount of the energy for their operation. Historically, the cross-flow pumping of the liquid was the most energy intensive process in the membrane bioreactors (Van Dijk and Roncken 1997). Currently, energy needs are heavily associated with the aeration demands which, with a share between 36% and 68% of total energy consumption, are often the most energy intensive process in the MBRs.

The detailed distribution of specific energy consumption in submerged and sidestream MBRs equipped with different membrane types, namely, flat sheet, hollow fibre and tubular are presented in Table 8.3.

Table 8.3: Distribution of specific energy consumption in different full-scale municipal MBRs

Energy users	Unit	Heenvliet – parallel (FS)	Varsseveld (HF)	Terneuzen (MT)
Membrane aeration	kWh/m^3	0.54	0.31	0.33
Biology aeration	kWh/m^3	0.19	0.14	0.01
Permeate pumps	kWh/m^3	N.A.	0.12	0.03
Feed pumps	kWh/m^3	N.A.	0.09	0.39
Recirculation pumps	kWh/m^3	0.11	0.05	0.01
Propellers and mixers	kWh/m^3	0.14	0.07	0.01
Surplus sludge pump	kWh/m^3	N.A.	0.01	N.A.
Rest MBR	kWh/m^3	0.15	0.05	0.16
Membrane related	kWh/m^3	0.65	0.57	0.76
Total MBR	kWh/m^3	1.13	0.84	0.94

Legend: FS – flat sheet; HF – hollow fibre; MT – multi tubular; N.A. – not available

Distinct differences between submerged and sidestream configurations can be observed in terms of energy required for activated sludge pumping, as the feed pumps in tubular system, with consumption of 0.39 kWh/m^3, were at least two times more energy demanding than in flat sheet (0.15 kWh/m^3) and hollow fibre (0.09 kWh/m^3) systems. In case of sidestream systems, energy demand for pumping of activated sludge through the membranes was comparable to energy required for aeration which was 0.34 kWh/m^3. When comparing submerged systems, the specific energy consumption for membrane aeration in flat sheet MBR was 33-37% higher than in hollow fibre system while total specific energy consumption differs only 0.2 kWh/m^3. The energy consumption of membrane related modules, i.e., membrane aeration, supply, recirculation, and permeate extraction was in the investigated MBRs in range of 0.57 – 0.76 kWh/m^3. Nevertheless, the distribution of energy is often very site-specific and depends also on plant design and its operational philosophy and settings. It is remarkable that in MBRs, significant amount of energy is used for the liquid pumping instead of to provide aeration for the biology treatment or for the membrane scouring. Contrary, in the

CAS system, the liquid pumping has only a small contribution to the total energy consumption of the system with about 90% of energy provided to the biological aeration.

Figure 8.3a shows the percentage distribution of the energy consumption in the full-scale flat sheet MBR in Heenvliet. Aeration is the major component of energy consumption as the blowers providing air for the membrane scouring and the biological process contribute to nearly 70% of the total energy demand. The coarse bubble aeration is the largest consumer being 56% and 0.48 kWh/m^3; process aeration energy demand is 11%; mixers and recirculation pumps consumed 9% and 6%, respectively. The rest, 17%, is mainly associated with the pumping, i.e., recirculation, permeate extraction and sludge discharge, the pre-treatment, the mixers and the heaters during winter months.

Figure 8.3b shows the percentage distribution of the energy consumption in the full-scale hollow fibre MBR in Varsseveld. Results show that blowers providing air for the membrane scouring and the biological process contribute to more than 50% of the total energy demand. The coarse bubble aeration is the largest consumer, being 36% and 0.3 kWh/m^3; process aeration energy demand is 17%; permeate and feed pumps consumed 15% and 11%, respectively. Energy consumption related with the membrane operation, i.e., membrane air scouring, feed and permeate pumps, required about 0.5–0.6 kWh/m^3 of treated wastewater. The rest (16%) represent energy consumed by the other installed equipment. The three main contributors are: the pump for internal recirculation from oxic to anoxic zone, about 0.03 kWh/m^3; the mixers in the anoxic tank, about 0.025 kWh/m^3; and the recirculation pump that pumps sludge from the oxic zone to the fine screens, about 0.02 kWh/m^3. Other individual components, with energy usage less than 0.01 kWh/m^3, are the chemical dosing pumps, the waste sludge pumps, the gravity thickener, the thickened sludge pumps, the process water pumps and the heating of the buildings (Van Bentem 2011).

Figure 8.3c shows the percentage distribution of the energy consumption in the full-scale tubular MBR in Terneuzen. Membrane aeration, doubled at the time due to clogging problem of the aerators, is responsible for consumption of 35% of total energy. The airlift system, i.e., feed and permeate pumps, contribute to 46% of total energy consumption, mainly due to high recirculation rate of activated sludge. The rest, 11%, is representing other smaller contributors like: waste sludge pump, iron-chloride dosing pump, online measurements, lights and computers at offices.

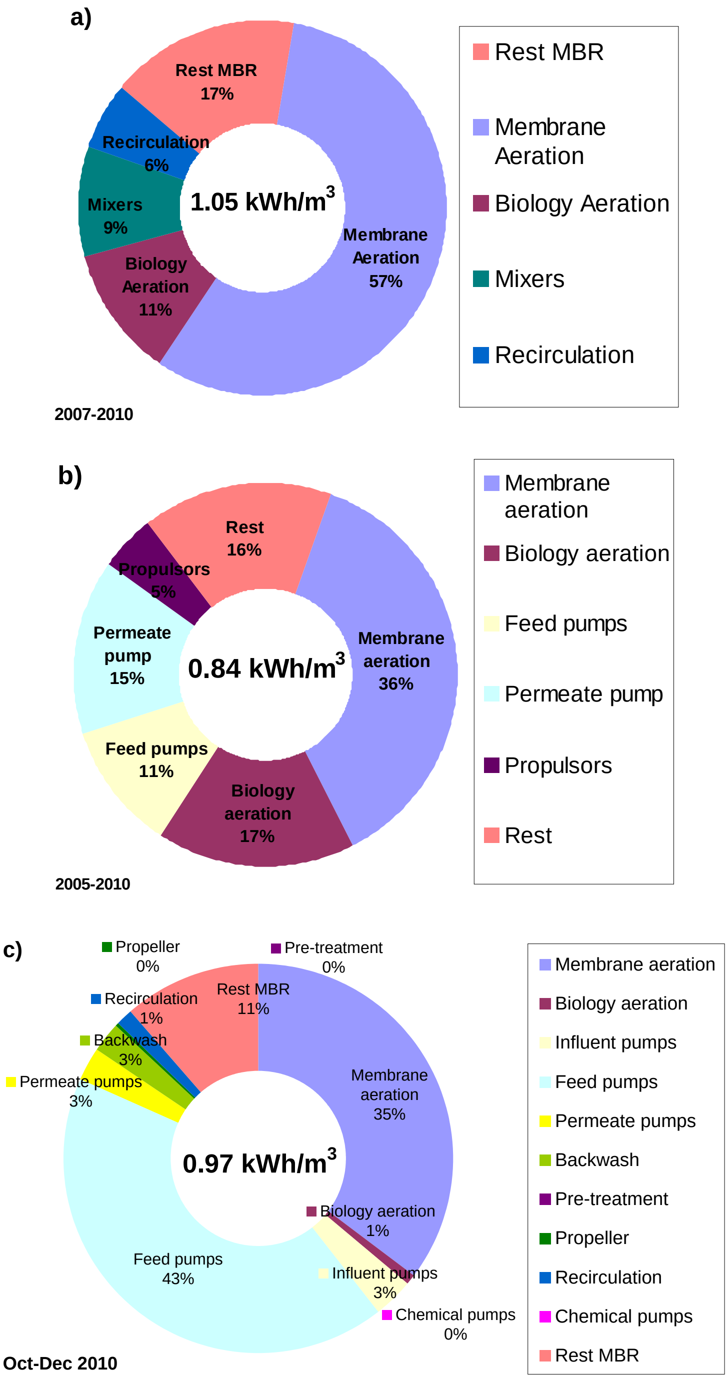

Figure 8.3: Energy consumption distribution of MBR equipment for: (a) Heenvliet (FS), (b) Varsseveld (HF), (c) Terneuzen (MT) MBRs

8.3.6 Energy consumption and flow dependency

Operation at optimal flow conditions, i.e., close to design flow at dry weather conditions (DWF), results in low specific energy consumption of about 0.7-0.8 kWh/m^3 (Figure 8.4).

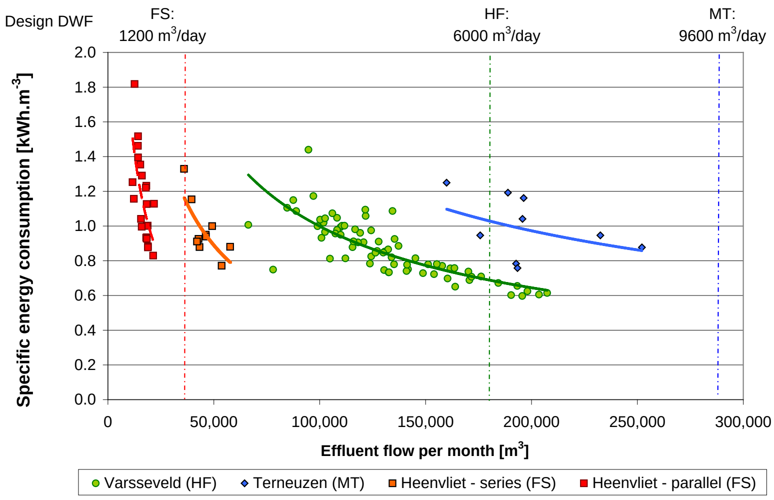

Figure 8.4: Specific energy consumption as a function of treated wastewater at three plants: Heenvliet (2008-2010), Varsseveld (2005-2010) and Terneuzen (2010-2011)

Under these high utilization conditions, reduction in energy consumption was, depending on the plant, between 5% and 20% compared to the average energy consumption (Figure 8.5). It is due to the fact that required membrane aeration rates are not proportional to the volumes of the treated flow. This phenomenon is also partially explained by operation of the process equipment, e.g. pumps and blowers, at or near their best efficient points when the flow increases. Although total energy consumption increased as the flow increases, an improvement in energy efficiency was observed with increase in the volume of treated wastewater. Contrary, sub-optimal operation below the design flow leads to higher specific energy consumption values. In particular, operation below hydraulic utilization of 50-60% of the dry weather flow should be avoided due to associated energy penalty (Figure 8.6).

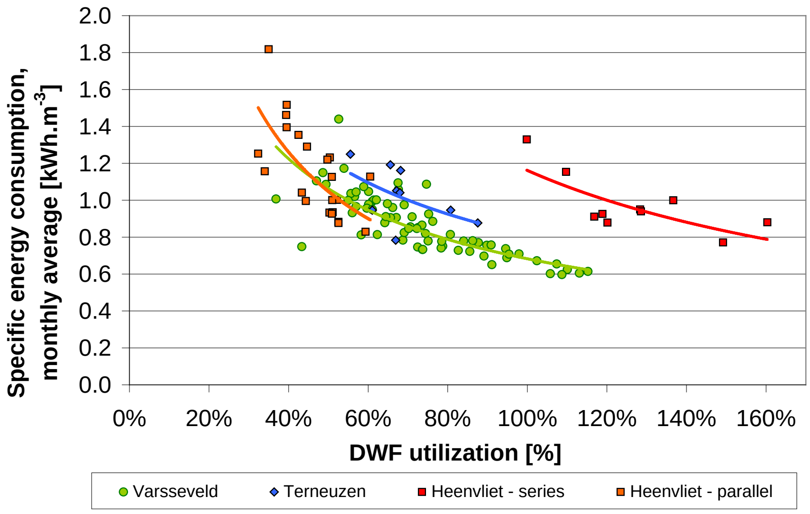

Figure 8.5: Specific energy consumption as a function of hydraulic utilization during dry weather flow (DWF) at three plants: Heenvliet (2008-2010), Varsseveld (2005-2010) and Terneuzen (2010-2011)

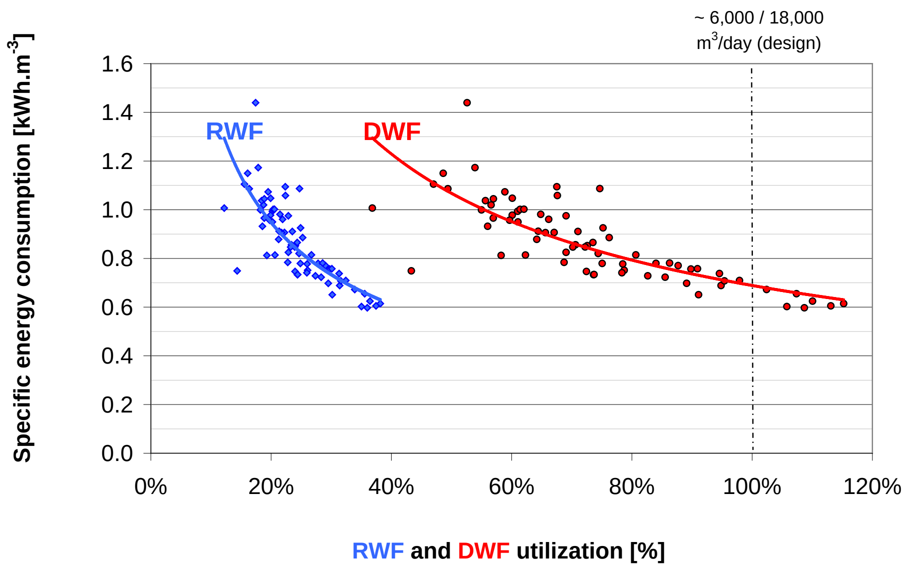

Figure 8.6: Specific energy consumption as a function of hydraulic utilization during dry weather flow (DWF) and rain weather flow (RWF) conditions at MBR Varsseveld during 2005-2010 period

8.3.7 Energy consumption and relation to plant capacity

Analysis of specific energy consumption as a function of plant design capacity (PE_{design}), expressed in kWh per PE_{design}, for Dutch and German municipal MBR plants indicate that, in general, the capacity of the plant does not determine the energy efficiency of the installation, which is in accordance with Brepols et al. (2010b).

Figure 8.7 presents specific energy consumption as a function of plant capacity for Dutch and German municipal MBR plants (adopted from Pinnekamp (2008)). Although, the smallest installations are least energy efficient, the biggest are either not the most efficient ones. Hence, the capacity of the plant does not determine the energy efficiency of the installation. Furthermore, all of the compared MBRs were more energy demanding than the average CAS treatment plant in the Netherlands represented by the benchmark value. General improvement in the range of 11-19% in energy efficiency for Dutch MBRs was observed during 2008-2009 period.

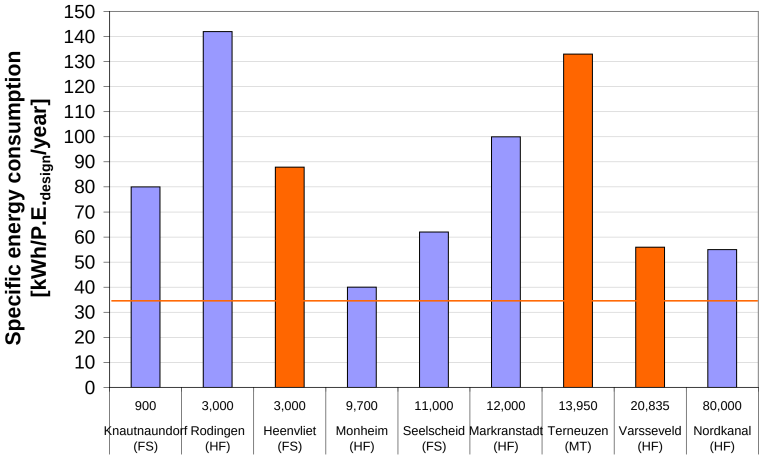

Figure 8.7: Energy consumption as a function of plant design capacity (1 PE_{design} is equal to a pollution load of 60 g BOD/day) for a selection of German MBRs (adopted from Pinnekamp (2008))

On the other hand, the size of the plant does indirectly influence energy efficiency of the MBR plant. Bigger plants needs multiple process lines, hence have the flexibility to match the incoming flow with required number of membrane lines. Concluding, the size of the plant, expressed in design PE, has no direct effect on efficiency, yet has indirect influence on energy efficiency.

8.3.8 Energy consumption per membrane area

The specific energy consumption per area of the membranes installed, expressed in kWh/m^2/year, was lower for hollow fibre installation (Figure 8.8).

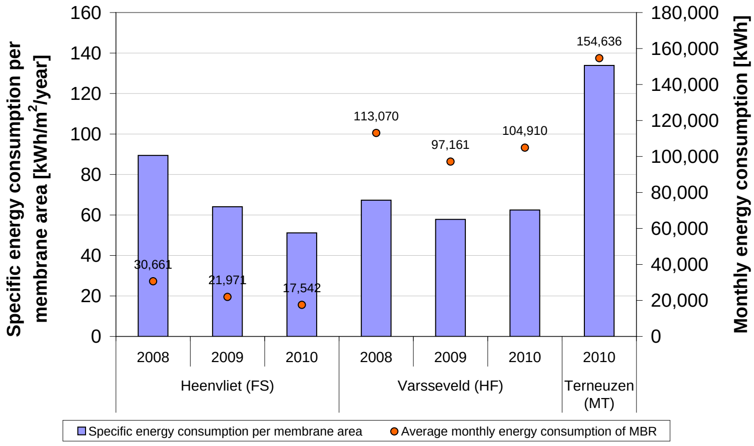

Figure 8.8: Energy consumption as a function of installed membrane area

The observed improvement for MBR Heenvliet is a logical consequence, also reported in the literature by Judd (2011), of an operational concept change from serial to parallel where only a small fraction, i.e., 25%, of the influent is treated in the MBR. In result, since March 2009, two membrane lines were operated alternately to increase membrane utilization and to reduce energy demand for membrane air-scouring. Obviously, the operational power demand increases with the amount of membranes installed in a submerged system. However, when energy usage is normalized for the membrane area their specific energy consumption decreases. Thus, big MBR installations are more energy efficient, in terms of membrane surface specific energy consumption (in kWh/m^2), compared to the small ones. Nevertheless, energy consumption depends more on the particular plant design and influent type than on the membrane configuration (Krzeminski et al. 2012a).

Additionally, operation of sidestream membranes is the most energy demanding. However, sidestream systems can apply higher fluxes, so it needs less membranes than submerged systems and thus requires lower capital costs. When results are compared for similar capacity, sidestream systems require 60-70% less membranes. Therefore, design of hybrid installations with tubular sidestream membranes allows to significantly reduce the required membrane area and, if the price of the tubular membranes is not more than 60-70% expensive than submerged membranes, to lower capital costs even further.

8.3.9 Energy consumption and relation to design configuration and plant layout

Selection of a particular MBR configuration during the design stage has certain consequences on operation and thus on energy consumption and efficiency. The principle choice of configuration, i.e., stand-alone or hybrid, largely impacts the overall MBR functioning (Krzeminski et al. 2011a, Krzeminski et al. 2012c).

A stand-alone MBR is generally more vulnerable to rapid changes, due to frequent variations in the characteristics of incoming flow, compared to the hybrid configurations. Hybrid configurations provide operational flexibility and therefore, in most of the cases, enabling stable MBR operation. In addition when operated in series, they have lower specific energy consumption for operation and maintenance than a stand-alone MBR. Serial hybrid MBRs ensure stable operation at energy and cost efficient conditions.

The specific energy consumption of the stand-alone MBR was approximately 12% higher than the hybrid MBR during in series operation, yet 27% and 0.3 kWh/m^3 lower than the hybrid MBR during parallel operation. The specific energy consumption is, however, highly dependent on the membrane capacity usage; the highest specific energy efficiency being attained when operating the MBR at optimal flow conditions. The MBR in the parallel concept is hindered by operation under sub-optimal flow conditions and is consequently less cost-efficient than serial hybrid MBR, approximately by a factor of 2.

Hybrid concepts are associated with at least 17% lower operational costs, compared to a stand-alone MBR, but the stand-alone MBR required less energy to remove the same amount of pollutants than the hybrid MBRs, and are as such more energy efficient in this aspect.

The relation between energy consumption and design, configuration and plant layout is discussed in detailed in *Chapter 7*.

8.3.10 Energy consumption and relation to operation strategy – focus on alternate membrane operation.

Energy consumption and energy efficiency of an MBR installation can be influenced by the strategy of the MBR operation, e.g. distribution of incoming flow between available membrane lines, operational flux or aeration strategy. Potential impact, and expected energy savings, depends on the possibilities available at particular plant.

When the Heenvliet WWTP is operated in the parallel mode, the MBR has to treat lower amounts of wastewater. Hence, due to energy reasons membranes tanks are working interchangeable and are changed approximately every three days. The membrane tank that is not in use has neither aeration nor feed or outflow (through the recirculation stream) provided. This tank is isolated.

The effect of alternate operation of two membrane lines was analysed based on the case of MBR Heenvliet. In the reference conditions (represented by green columns in Figure 8.9), both of the membrane tanks were operated simultaneously and fed with similar volumes of the activated sludge. During the alternate operation of the membrane tanks (represented by blue columns in Figure 8.9) an extensive relaxation period was introduced and only one line was in use at the time. Every 2 to 4 days the membrane tank in operation was changed to avoid excessive activated sludge settling, that could potentially hamper the re-start of the membrane tank. The flow normally divided between two membranes tanks was feed exclusively to one membrane tank. In consequence, the average operational flux was increased from 8.8 $L/m^2.h$ to 12.7 $L/m^2.h$ compared to reference conditions. The increase was lower than could have been anticipated, mainly due to SBR-like operation of the membrane tanks. Basically, the feed flow to the membrane tank was different from the extraction flow from the membrane tank. The MBR was filled to the 'high' level and then permeate was extracted until certain 'low' level was reached. In this way the downtime of the permeate production was shortened and, with the flux increase, energy consumption of the membrane blowers could have been reduced. Besides limiting over aeration, energy required for the operation of feed and permeate pumps was also reduced with the lower number of pumps in use. In addition, pumps were working for a shorter period of time but often at the higher workload.

Although complete shut down of the membrane aeration is allowed by the supplier, it is advised to keep the activated sludge recirculation in operation during these periods. However, MBR Heenvliet was operated without the sludge recirculation in not utilized membrane tank and only some pH decrease was observed. A positive side effect of implemented relaxations periods and alternate membrane operation was a slight improvement of permeability. Alternate operation of membrane tanks was slightly more energy efficient, as the specific energy consumption was reduced by approximately 0.1-0.3 kWh/m^3 (Figure 8.9).

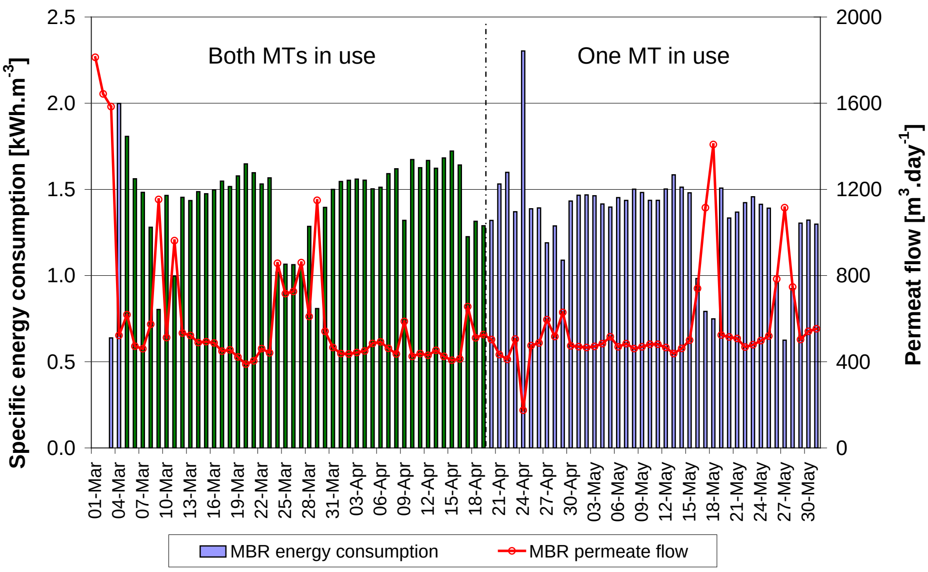

Figure 8.9: Effect of alternate operation of the membrane tanks on the specific energy consumption in MBR Heenvliet during period of March – May 2009

Observed improvement was mainly due to reduction in the membrane air-scouring as mixers and recirculation pumps have consumed similar amount of energy. Furthermore, on average 33% less energy was needed for handling similar flows. During the period examined in detail (March 2009 – April 2010) at least 8% of the energy was saved, which corresponds to financial savings on energy in range of 200-800 €/month (Figure 8.10). The cost of electricity was assumed to be 0.12 €/kWh, same as what is paid in case of MBR Heenvliet.

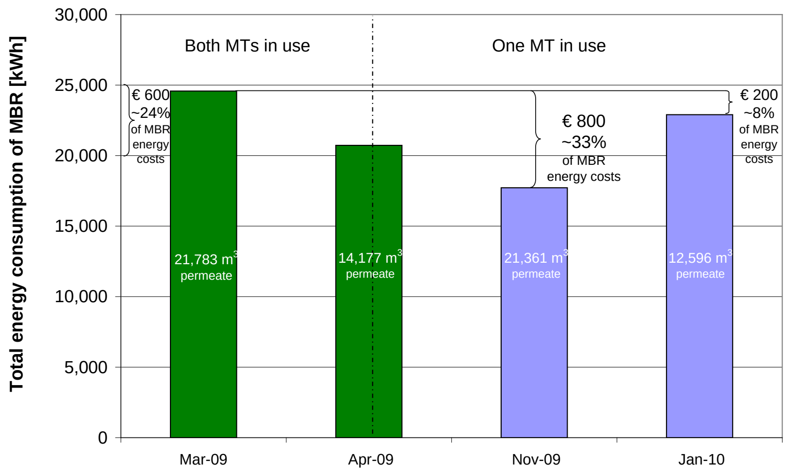

Figure 8.10: Total energy consumption and operational costs savings in relation to permeate production during alternate operation of the membrane tanks in MBR Heenvliet (most and least promising scenario shown). Electricity costs equal to 0.12 €/kWh

8.3.11 Energy consumption and relation with achieved effluent quality

The analysis of the relation between energy input and biological removal performance was performed based on the MBR Heenvliet due to availability of large energy and effluent data set for a period of May 2008 – December 2010. The daily and monthly energy data were used in the study. Influent and effluent data were collected every 2-3 days by the employees of the Hollandse Delta Water Board. Despite, relatively frequent effluent data points and daily energy data used for the analysis correlation between energy input and biological removal performance was not found. No direct relation between total and specific energy consumption and concentration of TSS, COD, BOD, TP, TN and TKN in the effluent was observed. Only a weak correlation between specific energy consumption and BOD removed in the MBR, expressed both in kg/day and mg/L, was observed (Figure 8.11). Removal of higher concentrations of BOD require higher energy input (Figure 8.11a).

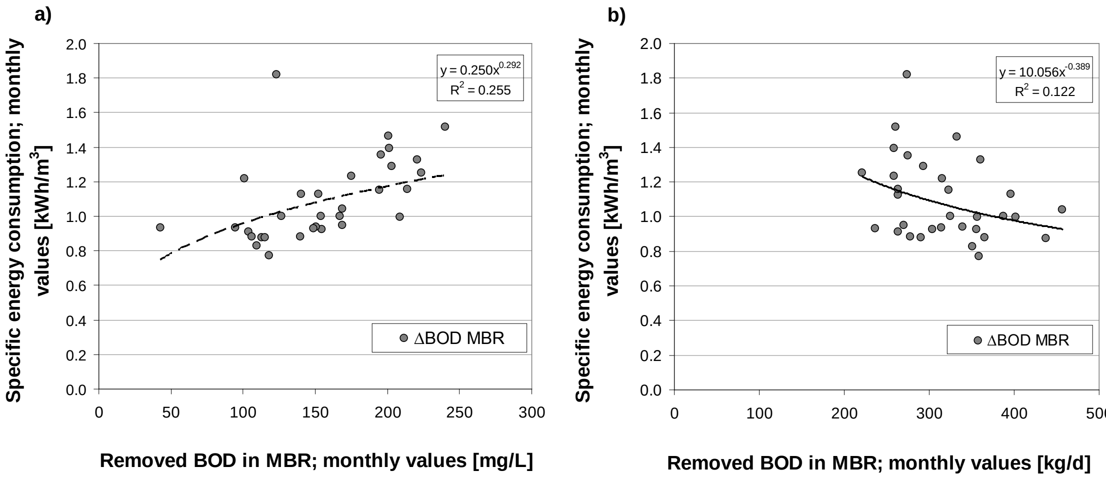

Figure 8.11: Relation of specific energy consumption to BOD removed in MBR Heenvliet during a period of 01.01.2008-31.12.2010. Removed BOD expressed as (a) concentration [mg/L] and (b) load [kg/day]

Moreover, when accounting for the specific energy requirements for process and membrane aeration rates no clear dependency on effluent quality could be determined. The lack of correlation could be partially explained through the fact, that, the aeration system is not adjusted hourly in response to the desired DO level and provide oxygen more conservatively, likely with a certain over-aeration. In result, expected correlation between energy consumption and BOD concentration, both the removed and in the effluent, would not be observed.

Nevertheless, it can be assumed that an energy saving potential exists and certain action towards energy reduction will rather not have a direct impact on effluent quality. This observation is in agreement with Verrecht et al. (2010) who reported a reduction in energy consumption in a small-scale decentralised MBR by 23% without compromising effluent quality, represented by COD and NO_3-N data. However, for a more accurate assessment of the potential energy reduction in a full-scale MBR, more specific measurements and detailed analysis is required. It was also observed that effluent concentrations of analysed parameters, i.e., COD, BOD and TKN, were not dependent on the influent concentrations. Only the effluent TP was slightly affected by the influent concentration (Figure 8.12).

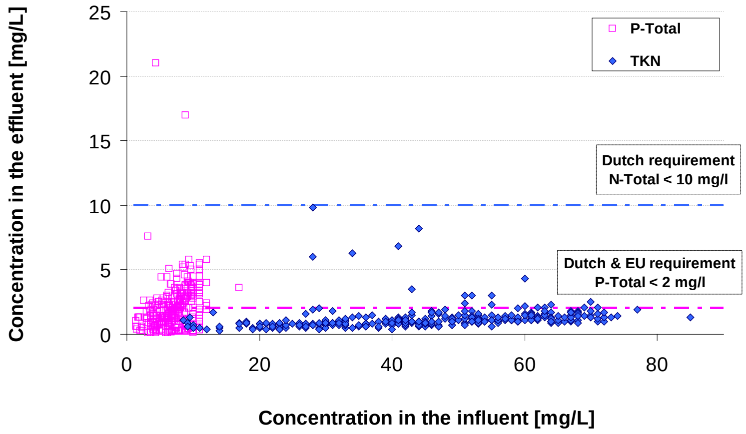

Figure 8.12: Concentration of total phosphorus (TP) and total Kjeldahl nitrogen (TKN) in the influent and effluent of MBR Heenvliet during a period of 01.01.2008-31.12.2010

The analysis was further complicated by the fact that, higher influent BOD concentration and higher BOD concentration removal requirement require more aeration, which require more energy. However, if the flow rate increases towards the treatment design capacity, slight improvements in efficiency are observed, which result in a slightly lower specific energy requirement. Isolating these two effects would require some statistical data massaging. Finally, efforts should be focused on the parallel operation of the CAS and MBR.

8.3.12 Energy consumption and relation with removed pollution load

The total and specific energy input was analysed in terms of pollution load removed in the MBRs. The removed pollution load was expressed as a person equivalent and equal to 150g of TOD as presented in equation 7.2.1. The removed pollution load was calculated as a difference between initial (influent) and final (effluent) pollution load.

$$PE_{150} = \frac{COD\left[\frac{kg}{day}\right] + 4.57 \cdot TKN\left[\frac{kg}{day}\right]}{0.150\left[\frac{kg}{inhabitant \times day}\right]} \tag{7.2.1}$$

$$PE_{150} = \frac{TOD\left[\frac{kg}{day}\right]}{0.150\left[\frac{kg}{inhabitant \times day}\right]} \tag{7.2.2}$$

Despite initial correlation when all plants were studied together (Figure 8.13a), no direct relation was observed when total (Figure 8.13a) and specific (Figure 8.13b) energy consumption was analysed in detail for each plant individually.

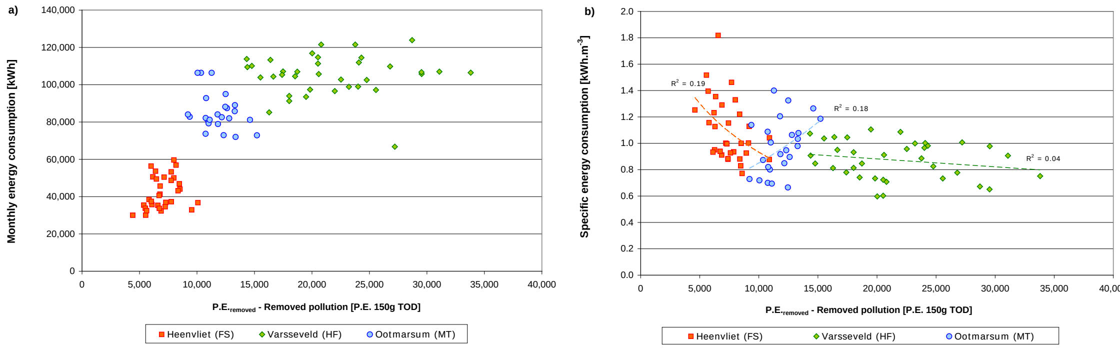

Figure 8.13: Total (a) and specific (b) energy consumption as a function of the removed pollution load. 1 $PE_{removed}$ is equal to a pollution load of 150 g TOD/day

Most probably the available data used for calculations were not sufficiently detailed for this analysis. Due to constant variations in biological and hydraulic loading of the MBR very detailed data are required. The energy and pollution data were monthly averages and not, as it turn out to be a necessity, daily data.

However, when the daily energy data were analysed for case of MBR Heenvliet the relation between removed pollution load and energy input also could not be determined likely because of too general influent/effluent data. The influent and effluent data were based on the data collected during routine samplings by the treatment plants employees. The routine samplings are usually performed with a frequency of approximately 1-3 times per month and in consequence a limited number of available data points.

8.3.13 Energy consumption and relation to activated sludge filterability

The energy consumption data were analysed in respect to activated sludge filterability, measured at Heenvliet during experimental campaigns described in detail in *Chapter 4*. The results from the five campaigns are plotted on Figure 8.14 and Figure 8.15.

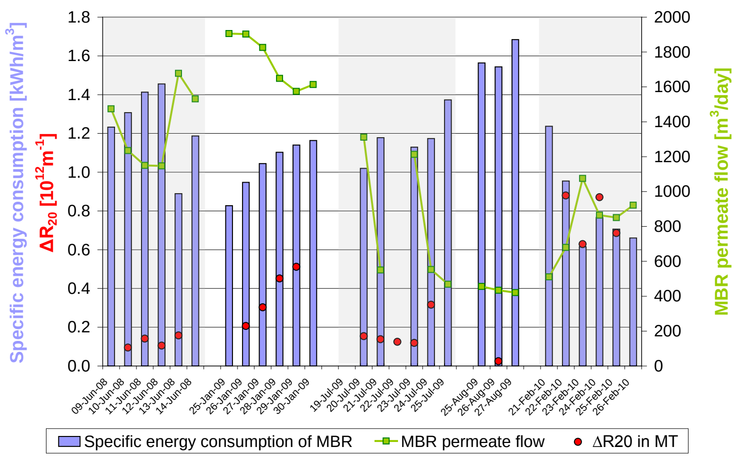

Figure 8.14: Specific energy consumption, permeate flow and filterability of activated sludge from the membrane tank during five measurement campaigns at MBR Heenvliet

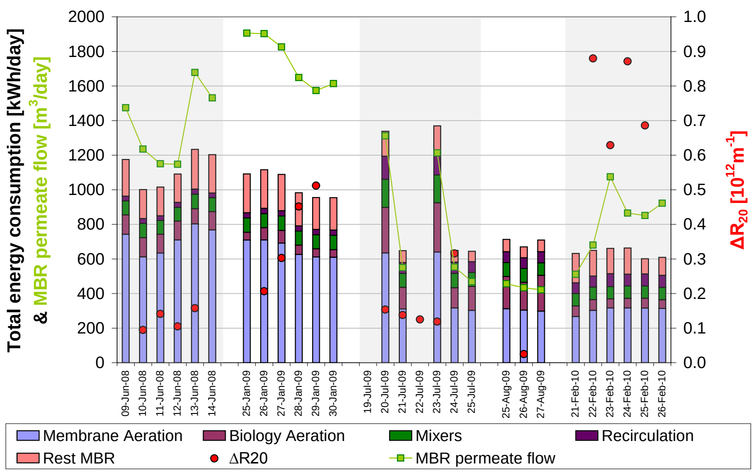

Figure 8.15: Energy consumption distribution, permeate flow and filterability of activated sludge from the membrane tank during five measurement campaigns at MBR Heenvliet

No relation between filterability and specific or total energy consumption was found. However, when filterability results were plotted against energy consumption of specific equipment or process, correlation between activated sludge filterability and biological aeration was observed (Figure 8.16).

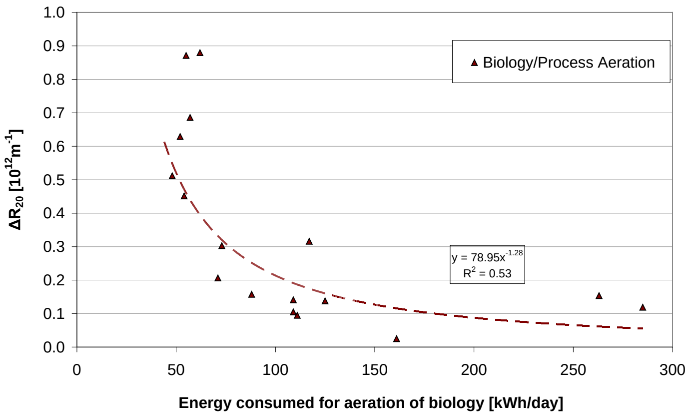

Figure 8.16: Activated sludge filterability and relation with energy consumption for aeration of biology during five measurement campaigns at MBR Heenvliet

Because of continuous and probably excessive airflow rates, correlation between filterability and energy for membrane aeration was not observed. It would require an active operation of the MBR and frequent adjustments of aeration intensity, based on the permeability and TMP trends monitored at the plant by the operators. For example, in case of a gentle increase of TMP the membrane aeration rates could have been reduced to promote energy efficient operation. Contrary, during a period of steep TMP increase, membrane aeration could be more intense in order to protect and clean the membrane effectively. Currently, the adjustments are not made and only when TMP reaches a critical level of 200 mbar, the chemical cleaning is performed to remove the foulants. It is expected, that in case when intensity of aeration for the membrane scouring would be tied to filterability – as the indication of fouling propensity of activated sludge – reduction in energy consumption would be possible, especially during the summer periods when filterability is better.

8.3.14 Operational costs of the full-scale MBRs

MBR cost assessment was performed to estimate the operational cost of full-scale MBR systems treating municipal wastewater. Thus, the analysis focuses only on OPEX and does not include the CAPEX. Economical analysis was based on the data provided by three water boards operating investigated MBR plants.

The early assessment of financial feasibility of MBR operation estimated the cost[1] at 0.58 €/m^3 (Adham and Trussell 2001). Later, the theoretical operational cost of the 38,000 m^3/d MBR equipped with hollow fibre membranes, excluding personnel cost, was reported to be between 0.08-0.12 €/m^3 (Côté et al. 2004). DeCarolis et al. (2004, 2007, 2008) assessed operating costs based on design parameters and data provided by the membrane suppliers. The cost were between 0.41-0.53 for 1,900 m^3/d MBR, 0.35-0.42 €/m^3 for 3,800 m^3/d MBR, 0.28-0.32 €/m^3 for 18,900 m^3/d MBR and 0.27-0.37 €/m^3 for 38,000 m^3/d MBR. The operational costs of the full-scale municipal MBR plants were reported to be: 0.11 €/m^3 for total WWTP, thus MBR and CAS, in Schilde (Garcés et al. 2007), 0.26 €/m^3 for MBR in Nordkanal (Engelhard and Lindner 2006, De Wever et al. 2008), 0.36 €/m^3 for MBR in Varsseveld (De Wever et al. 2008) and 0.06-0.08 €/m^3 for energy & chemicals in MBR Viareggio (Fatone et al. 2007).

The specific O&M costs includes expenditure for energy, chemical, sludge disposal and personnel. Average O&M costs of the stand-alone MBR in Varsseveld were 0.29 €/m^3 of treated wastewater in 2009 (Figure 8.17a). Dosage of iron chloride sulphate for chemical phosphorous removal, results in additional chemical costs (9%, 0.026 €/m^3) but also in significantly lower phosphorous concentrations in the effluent. The average O&M costs of the MBR in Heenvliet were 0.29 €/m^3 over a period of 2008-2010 (Figure 8.17b). When the total WWTP is considered, thus, combined CAS and MBR, the operational costs decreases to 0.13 €/m^3 (Figure 8.17d). This reduction is observed because of high total effluent production of combined CAS and MBR systems and relatively low energy consumptions in the CAS system. The average O&M costs of the total WWTP of Ootmarsum, also combined CAS and MBR, were 0.24 €/m^3 over a period of 2008-2009 (Figure 8.17c).

The energy and personnel costs are most significant part of MBR operation together exceeding the 50% of total O&M costs.

[1] 1 USD = 0.72 EUR where applicable; 24.10.2010

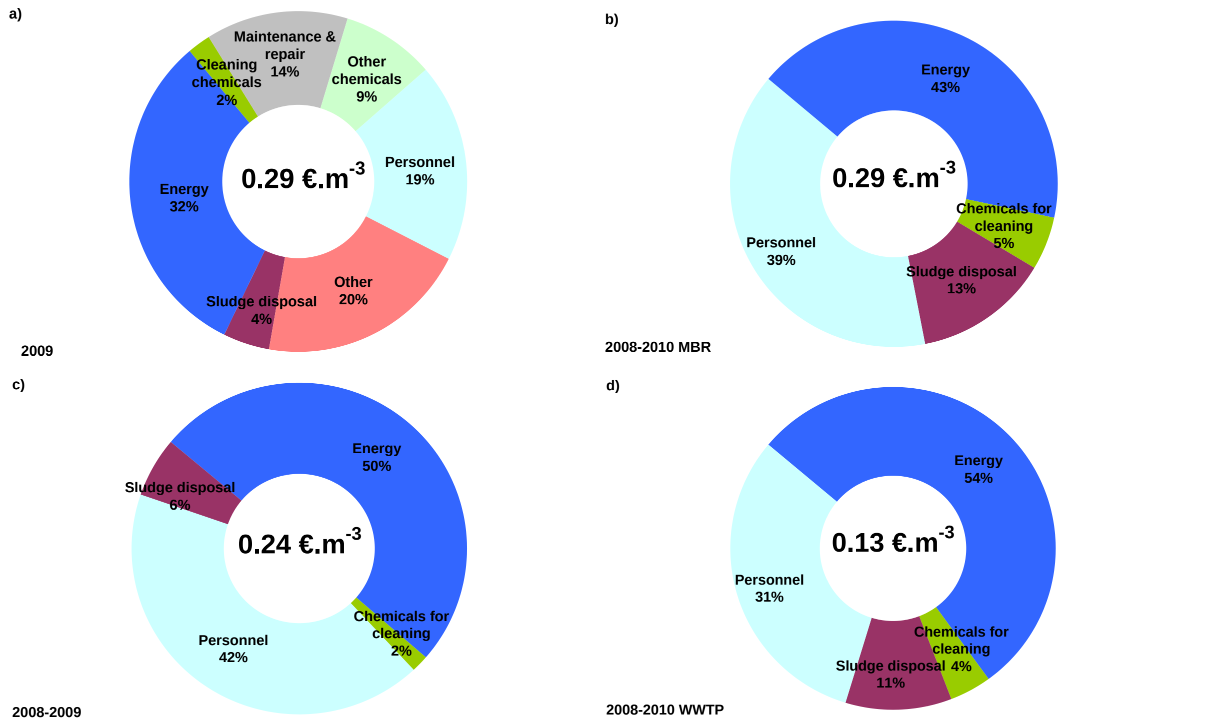

Figure 8.17: Operational cost distribution for (a) MBR Varsseveld, (b) MBR Heenvliet, (c) WWTP Ootmarsum and (d) WWTP Heenvliet

The analysed costs during the period of 2008-2010 were uniformed per removed pollution load and expressed in yearly removed population equivalent pollution. The O&M costs were: in the range of 26-27 €/$PE_{removed}$ for MBR Varsseveld, 21-24 €/$PE_{removed}$ for WWTP Ootmarsum, about 18 €/$PE_{removed}$ for MBR operated in series to the CAS system and between 30 and 41 €/$PE_{removed}$ for parallel operated MBR in Heenvliet.

During the whole operational period of MBR Heenvliet the specific O&M costs varied between 0.11 and 0.90 €/m^3 reaching high averages only during start-up, chemical cleanings or maintenance periods (Figure 8.18). In addition, specific costs during in series operation in 2008 and calculated at 0.17 €/m^3 are by 23% and 21,452 € lower compared to parallel operation in 2010. During the parallel operation MBR treats only a small fraction, i.e., 25%, of the influent coming to the plant and in consequence the specific cost increase to 0.37 €/m^3. Hence, operation at high membrane utilization results in cheaper treatment and better cost efficiency.
The MBR Heenvliet was designed for a treatment capacity of 100 m^3/h but was treating only approximately 30 m^3/h in the period of 2010-2012. The MBR was designed for larger capacity as growth in the wastewater load was expected yet has never become reality. Hence, the MBR is hydraulically under loaded and, in consequence, the operational costs are relatively high.

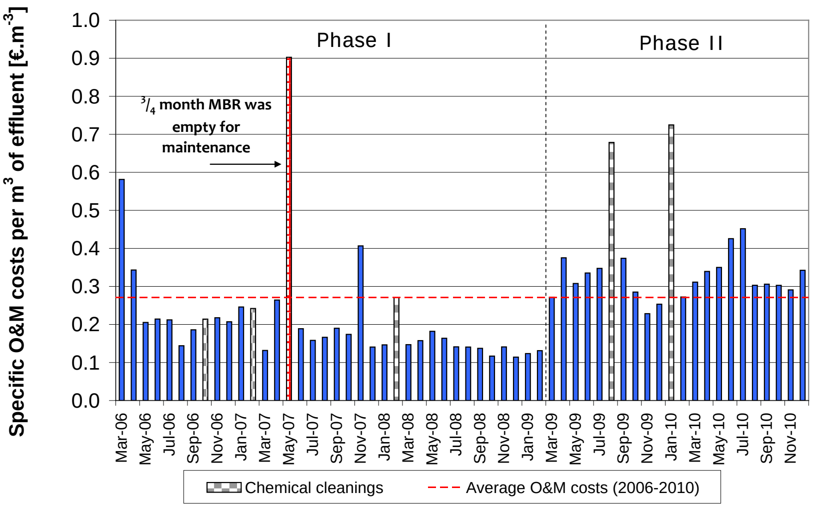

Figure 8.18: Specific operational cost of MBR Heenvliet during a period of March 2006 - December 2010. Phase I indicates the period when MBR was operated in series to CAS system and phase II the period of parallel MBR operation

8.3.15 Cost of chemical cleanings at full-scale MBR facilities

During operation membranes are subject to the fouling process that results in a permeability decline. To control fouling an appropriate cleaning protocol, often composed of numerous individual methods, is required. Typically, the membranes are protected by the membrane air-scouring that creates a shear force and remove some of the potential foulants from the membrane. In addition, relaxation or backwash is incorporated in the filtration cycle providing a physical (or mechanical) cleaning to remove reversible fouling (Geilvoet 2010). Finally, the chemical cleanings complement the overall cleaning approach. Two types of chemical cleanings can be distinguished: maintenance cleaning (on a weekly basis) and recovery cleaning (yearly basis). Maintenance cleaning is used to regularly clean the membrane in order to keep the permeability at a desired level. Intensive recovery cleaning of the membrane is performed when further filtration cannot be sustained because of an increase in the TMP. Nevertheless, each treatment facility can have specific chemical cleaning protocols, especially in industrial locations, i.e., chemical concentrations and cleaning frequencies, as recommended by the membrane suppliers (Le-Clech et al. 2006).

Distinct differences in applied cleaning methods are also observed between the membrane configurations. Hence, a strategy for chemical membrane cleaning is different for hollow fibre, flat sheet and multi-tubular membrane systems. Consequently, differences in the related costs can be expected.

The costs related to the chemical cleaning are the chemicals itself, energy consumed during the cleaning procedure and labour costs. As the personnel related costs are different from one location to another and are not directly related to the cleaning procedure they were excluded from the analysis. The energy demand of chemical dosing pumps is irrelevant with requirement of about 12 kWh per year in case of tubular installation, and approximately 26 kWh per year for flat sheet installations. With the electricity cost of 0.12 €/kWh the cost of energy input would be in the range of 1-5 € per year. Hence, the energy consumed during the cleaning procedure can be assumed negligible. The amount of consumed chemicals depends on the frequency of the cleanings and concentrations of cleaning solutions thus on applied cleaning strategy.

The specific chemical cleaning costs, expressed per installed membrane area per year, are lowest in case of hollow fibre installation and are approximately half of the costs at flat sheet installation (Figure 8.19). At the hollow fibre membrane plants, chemical cleaning methodology is based on the so-called ‘maintenance cleaning in air’ and is performed on a regular basis with a high frequency. The periodical chemical cleaning is carried out weekly or biweekly. The intensive recovery cleaning is carried out when other cleaning methods, including physical cleaning, are not effective. In contrast, at the flat sheet installation only intensive chemical cleaning is performed approximately once or twice a year.

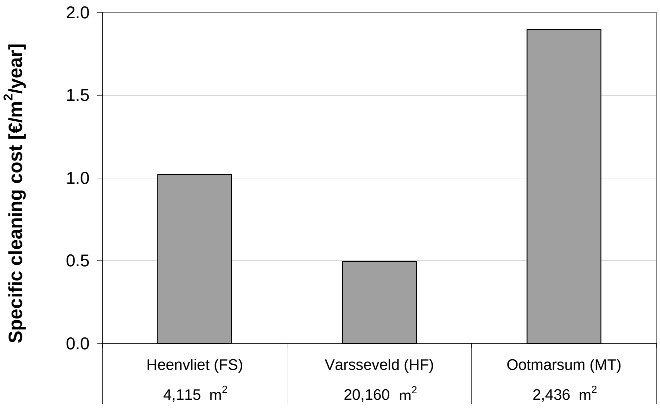

Figure 8.19: Specific cleaning cost, for a period of 2009, per membrane area installed at each MBR

The highest specific cleaning costs are associated with chemical cleanings at the installation equipped with sidestream tubular membranes. However, as mentioned before in section 8.3.8, sidestream systems can apply higher fluxes, so it requires less membranes than submerged systems. In results, specific chemical costs are 2 to 4 times higher than for flat sheet or hollow fibre system, respectively. The periodic chemical cleaning at the tubular membrane plants is carried out every one or two months, and its frequency depends on the applied flux. Furthermore, a combination of forward flushing and periodic back flushing is used to control the cake layer formation inside the membrane tubes and to extend the intervals between maintenance cleanings.

When the yearly costs of chemical cleanings are expressed in relation to volume of the treated flow the tubular membranes are the one requiring lowest maintenance costs (Figure 8.20).

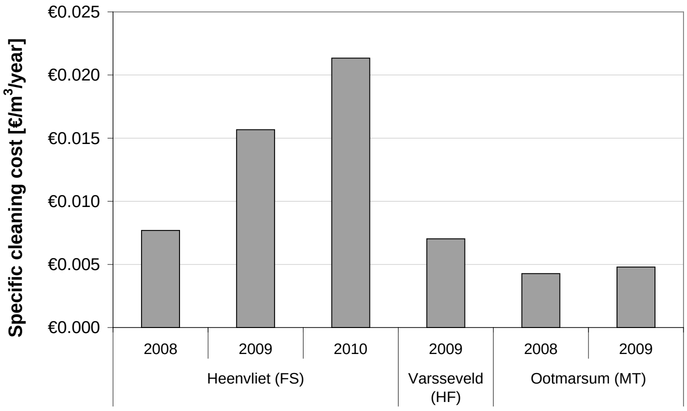

Figure 8.20: Specific cleaning cost per treated flow

The average yearly cost is 0.0045 €/m^3 of permeate and is about 35% lower from 0.007 €/m^3 at hollow fibre membrane plant. The increase in the specific cleaning costs observed at flat sheet installation in Heenvliet was caused by an operational concept change from serial to parallel where only a small fraction of the influent is treated in the MBR. When the MBR was operated in series to the CAS system, and under nominal hydraulic capacity, the specific cleaning cost was comparable, yet 9% higher, to the cost at the hollow fibre plant. After the operational change in the beginning of 2009, the MBR was operated in parallel to the CAS system. Consequently, incoming flow to the MBR was reduced to roughly 25% of the total flow and the specific costs have increased proportionally.

8.4 Energy efficient operation of MBRs

The long term monitoring of the operation and performance of the full-scale MBRs allowed to identify principles of energy efficient MBR operation and made a first attempt to establish a best operational practice:

- Maintain healthy and well filterable activated sludge in order to limit required chemical cleanings and possible operational upsets. The activated sludge condition is crucial for stable and efficient operation. If the sludge quality is good then the membranes are able to be operated in a stable manner at the minimum air flow. A key to good activated sludge filterability lies in the bioflocculation state of activated sludge. One of the options to generate activated sludge with a good filterability is to enhance flocculation by operation under longer SRT (Van den Broeck et al. 2012). Also operation under low sludge loading rate (Geilvoet 2010) and with low recirculation rate leading to activated sludge upconcentration in the membrane tank is reported to improve filterability (Moreau 2010). Process disturbances should be avoided through careful monitoring of plant operation (Gil et al. 2011). Short remedy to poor filterability problems could be found in flux enhancers but, on a long term, a cause of poor filterability should be identified and eliminated (Van den Broeck 2011, Van den Broeck et al. 2011). Of course, the influent composition remains the most important, yet not controllable, factor influencing activated sludge filterability (Moreau 2010).

- In case of hybrid MBRs, to operate in series to the CAS system and benefit from provided hydraulic and biological buffer zone to ensures more stable operating conditions.
- Preferably continuously operate at optimal flow, i.e., design flow at dry weather flow, and at least 60% utilization of the membrane capacity through the proper system design, introduction of internal and, if possible, external equalization. Internal equalization, suitable for daily variations compensation, is commonly introduced in a form of dynamic water level in a bioreactor. The limit of the internal flow equalization in the membrane tank can be different for each manufacturer and, for example, Toray membranes must have at least 0.5 m of liquid above the membranes. External equalization, suitable during storm events, requires a separate basin ahead of the MBR.
- In case when continuous operation at optimal flow is not possible, operational settings should adapt to low or no flow conditions, e.g., relation of air-scouring rate to volume of the incoming flow or putting unnecessary equipment into temporary idle mode. Another option could be alternate operation of the membrane lines when the number of membrane lines in use is matched with the incoming flow in order to achieve a benefit of higher operational flux. In this way up to 60% energy can be saved mainly by lowering membrane aeration in stand-by trains (Kueppers 2011).
- Implement new aeration strategies for the membrane air-scouring in place of continuous aeration, e.g., intermittent aeration, sequencing aeration or proportional aeration with air scouring rate adjusted to the flow conditions. Alternatively, when continuous aeration is required by the membrane supplier, lower aeration rates according to new recommendations. For example, latest findings of Toray allow reducing membrane air scouring from initial 0.90 $Nm^3/m^2.h$ to 0.35 $Nm^3/m^2.h$. The expected energy savings is foreseen to be in the range of 25-35% of the total energy. According to the flat sheet membrane manufacturer's intermittent aeration is not suitable for their membranes, while proportional and pulse aeration are an option for cleaning the flat sheet membranes. The pulse air should start before and finishes after the relaxation period begins. A certain overlap is needed for a better membrane cleaning with the air-scouring.
- Utilize elevated dissolved oxygen concentration from the membrane air-scouring for biological treatment purposes by introduction of return sludge to the oxic part of bioreactor. However, utilization of membrane aeration is often perceived as difficult to implement or not suitable for the particular case.
- Operate at lower MLSS in the bioreactor and higher in the membrane tank in order to improve oxygen transfer efficiency, through the alpha factor increase, in the biological section. For example, the MBR Glanerbrug, in operation since May 2011, was designed in the way that the MLSS content in the biology is lower than in the membrane tanks. Tubular membranes are supplied with the low concentrated, 3-4 g/L, activated sludge coming from ordinary CAS system. The increase in MLSS is

achieved by the internal activated sludge recirculation and limited addition of fresh activated sludge.

8.5 Energy saving potential in MBRs

Opportunities to reduce energy consumption in the full-scale MBRs are in general design, operation and equipment related. Based on literature review (Cooper et al. 2006, Fan and Zhou 2007, Giesen et al. 2008, Livingstone et al. 2009, Ovezea 2009, Wallis-Lage and Levesque 2009, Brepols et al. 2010a, Jimenez et al. 2010, Krause and Dickerson 2010, Krause et al. 2010, Lazarova et al. 2010, Lorain et al. 2010, Palmowski et al. 2010, Prieske et al. 2010, Freeman et al. 2011, Lesjean et al. 2011, Veltmann et al. 2011), case studies and process data analysis we have identified following energy saving options.

8.5.1 Energy savings related to the design

Opportunities at the design stage includes the configuration of the system in multiple process trains to provide flexibility in adapting to flow fluctuations, external and/or internal equalization to ensure higher fluxes, addition of primary sedimentation together with a digester and elimination of the equipment with duplicate functions. In addition, air diffusers should be mounted directly to the tank floor in order to maximise the life of the air bubble on its way to the surface. The use of the whole water depth allows more efficient oxygenation of activated sludge. Furthermore, improvement in design of the membrane racks should also be considered of relevance. For example, in case of Heenvliet MBR, there are some losses of membrane scouring air due to insufficient membrane racks design. Lack of side walls between aerators and membranes sheets cause a certain air loss. Hence, the low airflow set point suggested by the membrane manufacturer needs to be slightly higher in practice than theoretical 0.35 $Nm^3/m^2.h$.

It is worthy to mention that, double deck module design, by itself, does not provide energy saving. It benefits from the reduced air volume requirement but, at the same time, suffers from the increase in required pressure. The double deck design benefit is more likely to be observed when air is also used for the biology purposes.

8.5.2 Energy savings related to the operation

Operational opportunities present the most significant potential for energy savings and focus on novel aeration strategies for membrane scouring to replace continuous aeration and providing efficient flow conditions. For instance use of equalization, flux enhancers or increase of operational flux result in operation at constant and reasonable flux. The flux could be also simply maximised from time to time. After few-days long relaxation periods the membranes are able to cope with a higher workload for a certain period of time (days). Another approach is to couple the number of membrane lines in use, airflow and recirculation rates to incoming loading especially at the times of low flow. This approach is known as the treatment on demand. However, in some cases the installed pumps are too big to significantly decrease recirculated flow rate. Thus, a further reduction of the sludge recirculation can be achieved by the introduction of a stop time for the pumps or by the installation of extra cooling fans on the pump in order to lower the frequency. However, attention has to be paid to the fact that by lowering the frequency, the head of the pump will also lower.

Also other options are closely linked with aeration issues, e.g., utilization of membrane scouring aeration for biological process needs or operation at lower MLSS in the bioreactor and higher in the membrane tank to increase alpha factor, and in result to improve oxygen transfer efficiency in biological section. Likewise finer and dynamic dissolved oxygen control to reduce over-aeration in the bioreactor. Finally, optimization of the filtration protocol, i.e., by flux increase, longer filtration and shorter relaxation/backwash periods, at high wastewater temperatures when the fouling potential is usually lower could also improve overall energy efficiency. However, implementation of changes to the operational protocol should be done only during the periods when activated sludge is of good quality and good filterability. Otherwise, the risk of potential membrane fouling or even clogging is likely higher. Another possibility, however rather applied to hollow fibre systems, would be to increase membrane air-scouring rate during the relaxation or backwash period and to decrease it during the filtration step. During relaxation or backwash, the cake deposit is lifted-up from the membrane surface and as such presents the best time to scour the membrane in order to remove attached foulants. Furthermore, low aeration during filtration keeps the flocs intact, limits the release of cellular substances and reduces the fouling tendencies (Lorain et al. 2010).

8.5.3 Energy savings related to the equipment

Along with operational options, certain energy savings opportunities are related to the equipment selection. For example, use of high-efficiency turbo blowers in place of positive displacement blowers can save approximately 15% in blower energy. Change from the constant speed to variable speed, i.e., variable frequency drivers, on main electromechanical equipment improves equipment overall efficiency at different conditions. It would also allow implement novel control strategies for liquid pumping and aeration for membrane and process purposes. Besides, together with aeration control based on TMP-increase up to 15-25% reduction of energy consumption can be achieved depending on the case (Kueppers 2011). The costs of the equipment can be also reduced by stacking the FS membrane modules on top of each other to create, so called, double or triple deck configurations. Operation of pumps, blowers and mixers close to their most efficient point result in lower energy consumption and increase equipment life. Furthermore, according to Prieske et al. (2010) modified shape and location of membrane aerators, i.e., smoother draft tube edge, increase the liquid circulation by 30-50% likely resulting in lower requirements for air-scouring rates.

8.6 Conclusions

Energy demand is one of the major components of O&M costs of MBRs and has become an essential focus point in the full-scale MBR operation. In this study specific energy requirements of full-scale MBRs were linked to operational parameters and reactor performance, subsequently, enabled to improve understanding of the influence of design and operation parameters on specific energy consumption of MBRs. In addition, principles of energy efficient operation and energy savings opportunities were identified. Assuming that the studied MBRs are representative for the particular membrane configuration, i.e., Varsseveld for hollow fibre, Terneuzen for tubular and Heenvliet for flat sheet, and comparable to other similar installations, generic conclusions can be carefully drawn. Hence, based on the research results presented in this chapter, the following conclusions can be made:

- The municipal MBRs are well operated, with good performance, without major problems and, despite often sub-optimal operation, consume on average 0.8-1.1 kWh/m^3, values similar to other comparable installations.
- Operation at optimal flow conditions, i.e., close to design flow at dry weather conditions, results in a low specific energy consumption of about 0.7 kWh/m^3 and improved energy efficiency. Also an increase in the applied flux results in low energy consumption.
- The specific energy consumption of an MBR system is dependent on many factors, like the system design and layout, the volume of treated flow, the membrane utilization and the operational strategy.
- Lack of clear correlation between total and specific energy consumption and TSS, COD, BOD, N-Total and TKN concentrations in the effluent indicate a potential for energy optimization studies without immediate danger of affecting the quality of the produced effluent.
- Aeration is the major energy consumer, often exceeding a 50% share of total energy consumption, with a minimum of 35% for membrane aeration. In consequence, the coarse bubble aeration applied for continuous membrane cleaning remains the main target for energy saving actions, especially for installations with flat sheet membranes.
- Specific energy consumption for membrane aeration in a flat sheet MBR was 33-37% higher than in a hollow fibre system whereas the total specific energy consumption differs only 0.2 kWh/m^3.
- Investigated full-scale MBRs have a potential for further improvement in terms of energy consumption and energy efficiency.
- Energy reduction is possible with implementation of new aeration strategies, flow equalization or adjusting operational settings to incoming flow.
- The analysis of energy consumption and energy efficiency in MBRs should include potential differences between the plants. Therefore, it has to be based on the well defined scope of the study and include the same components of energy consumption, e.g., total WWTP plant or MBR unit or membrane related modules only.

CHAPTER 9

Conclusions, perspectives and recommendations

9 Conclusions, perspectives and recommendations

9.1 Chapter outline

In this chapter the conclusions and main outcomes of the research thesis are summarised (section 9.2) and perspectives are discussed (section 9.3). Section 9.4 deals with the overall evaluation of the research. Section 9.5 presents recommendations for further research.

9.2 Conclusions from the various research steps

MBR have become an increasingly popular municipal wastewater treatment process alternative and the amount and capacity of full-scale MBR plants is continuously increasing worldwide[1]. The MBR technology is now regarded as mature and various authors denominate MBR as the best available technology for industrial, but also municipal, wastewater treatment (Kraume and Drews 2010, Lesjean et al. 2011). However, although MBR technology attracted significant attention for more than a decade, in the author's opinion, MBRs are preferred over other treatment technologies mainly when certain criteria, i.e., high effluent quality, small footprint, easy retrofit and upgrade of old WWTP, apply. This is due to the fact that despite continuous technological development, old problems still remain unsolved. The retention of activated sludge and wastewater constituents on and in the membrane results in a decrease in the membrane filtration performance. The efficiency of the filtration process in an MBR is governed by the activated sludge filterability, which is determined by the interactions between the biomass, the wastewater and the applied process conditions. Due to the interdependency of the aforementioned factors and the dynamic nature of the feed and biomass, membrane fouling is a very complex phenomenon. Implemented strategies for prevention and removal of membrane fouling lead to an increase in the operational and maintenance costs of the treatment system. In particular, the high energy requirements arisen from frequent membrane cleaning by air scouring remains a challenge in terms of energy consumption and overall cost efficiency of full-scale MBRs.
Based on the various studies and results presented in this dissertation, the following conclusions can be drawn.

In Chapter 4, activated sludge filterability was monitored in three full-scale MBRs. The fluctuation patterns of activated sludge filterability correlate with the seasonal temperature fluctuations. Results show that t*emperature is an important influencing parameter with respect to filterability*. Filterability of activated sludge originating from the biological compartments is more prone to temperature fluctuations than sludge from the membrane tank. Our results confirmed the findings of Lousada-Ferreira (2011) that homogeneous and heterogeneous filterability development can occur depending on applied recirculation rates between the membrane tank and aerobic tank. High return rates lead to homogenous

[1] www.thembrsite.com (Last accessed 16.09.2012)

filterability whereas low return rates lead to heterogeneous filterability along the process. *In case of low return rates, thus heterogeneous filterability, and for MBRs with a separate membrane tank, the MLSS concentration should be above a critical MLSS concentration of about 10 g/L to promote filterability improvement in the membrane compartment*.

In Chapter 5, the influence of seasonal temperature changes on the characteristics of the raw wastewater and activated sludge was assessed. It was found that, the raw wastewater has a higher membrane fouling potential than the activated sludge. The soluble and colloidal fractions of the raw wastewater are likely to play an important role in the membrane resistance increase and filterability deterioration. Filterability deteriorated at low temperatures, increased organic loading and increased MLVSS/MLSS ratio. *Deterioration of filterability under low temperatures was also linked with a slower biodegradation of the wastewater in the mixed liquor compared to high temperatures. The parameters usually denoted in literature as fouling indicators, e.g., BPC, SMP and TOC, are not clearly correlated with sludge filterability.*

In Chapter 6, activated sludge characteristics were analysed in respect to the sludge filterability. It was revealed that sludge parameters previously reported as membrane foulants were not clearly correlated with the sludge filterability. In fact, similarly to findings of Chapter 5, every parameter alone is a weak indicator of biomass fouling propensity. *However, a combination of activated sludge parameters, i.e., the sludge morphology and relative hydrophobicity, better indicated sludge filterability than the parameters alone.* With respect to influent characteristics, a number of undesirable events leading to operational problems and affecting sludge filterability were identified: an undesired and harmful composition of incoming wastewater, hydraulic and/or organic load shocks, as well as abrupt temperature changes of the influent. Nevertheless, an MBR is a robust and reliable technology as permeate quality mostly complies with the regulations and was not affected despite encountered operational problems. However, the removal of TN and especially TP is currently not optimised and, from a biological point of view, improvements in terms of nutrient removal are still possible.

In Chapter 7, the impact of MBR plant layout and membrane configurations on the functioning of full-scale MBR plants was analysed. For the cases analysed in this research, *it can be concluded that both MBR plant layout and membrane configurations have some influence on the overall plant functioning, i.e., operation and performance.* Whereas membrane selection does influence mainly operational strategies (pre-treatment, filtration protocol, membrane cleaning and fluxes) the MBR plant layout has more general influence on plant functioning (operational flexibility and reliability, performance and O&M costs). Moreover, the activated sludge filterability was found to be independent of the membrane configuration but not of the MBR plant layout.

In Chapter 8, the principles of energy efficient operation and energy saving opportunities of full-scale MBRs were identified. *The energy efficiency of an MBR is driven by the hydraulic utilization of the membranes and can be improved mainly by implementation of flow equalization, new aeration strategies and adjusting operational settings to incoming flow.*

9.3 Main outcomes and perspectives

9.3.1 Activated sludge filterability and membrane fouling

- The DFCm proved to be capable of providing information whether a supposed decrease in permeability should be attributed to a poor activated sludge filterability or to an inappropriate operation of the MBR filtration process (Chapter 6). This opens possibilities for the optimisation of the MBR operation, e.g., filtration and cleaning protocol.
- There is no single parameter to explain activated sludge filterability and membrane fouling. EPS, SMP, MLSS, TOC and BPC, often reported in literature as the fouling indicators, failed as single universal fouling indicators and/or parameters determining the filterability of activated sludge. This is understandable keeping in mind the inter-relations between various factors influencing the fouling process (see section 2.5.2) and problem complexity. A combination of parameters is more appropriate when it comes to the prediction of the activated sludge filterability (Chapter 5 and 6).
- The colloidal and soluble fractions (<1 μm) play an important role in the membrane resistance increase. Therefore, the amount of submicron particles is very likely an important component of activated sludge filterability.
- Generally, the bioflocculation state of activated sludge is a major indicator for the activated sludge filterability. The flocculation–deflocculation processes are thus of major importance in respect to good activated sludge filterability promotion. Monitoring and promotion of bioflocculation is a key for the membrane fouling reduction. DFCm in combination with ACTIAS image analysis complement each other well and provide valuable information on filterability and bioflocculation of activated sludge.
- Similar activated sludge filterability, as expressed by comparable ΔR_{20} values, may indicate similar activated sludge physical characteristics especially with respect to the settling properties as confirmed by similar DSVI results.
- Subjection of activated sludge to significant changes in operational conditions results in filterability deterioration and, subsequently in operational problems at the MBR plant.
- Analogically, the operation of a conventional activated sludge (CAS) process in front of the MBR acts as a buffer, provides more stable hydraulic and biological conditions for activated sludge and leads to a better sludge filterability.

9.3.2 Operation of a full-scale MBR

- Full-scale municipal MBRs can be operated well and in a stable manner, without major problems unless unexpected events take place.
- MBR is a reliable process; membrane separation allows to achieve a relatively stable quality of permeate despite being dependent on dynamic biomass properties, which may be additionally affected by unexpected and undesired events. The quality of the MBR permeate is comparable for different MBRs and membrane configurations as well as for periods with good and poor filterability of activated sludge. However, this statement should be further confirmed with a dedicated research campaign focusing on permeate quality and as such measuring a broader range of quality parameters.
- Operation and performance of full-scale MBRs are subjected to similar seasonal differences like activated sludge filterability: better in summer and worse in winter periods. The changes in operation and performance are mainly driven by associated activated sludge fluctuations, which in turn are linked with seasonal temperature variations.
- Due to a higher fouling propensity observed at low temperatures, it is recommended to schedule special activities like, start-up periods, extensive chemical cleanings or significant maintenance actions at periods of elevated temperature. In that way, potential operation under unfavourable filterability conditions or even encountering of operational perturbations is reduced. In addition, operation under temperatures below 10-12°C requires special attention from the operators, and may require fine tuning of operational settings to recover expected performances, due to associated rapid permeability decrease.
- Operation of an MBR with poor activated sludge filterability can be achieved but is often accompanied by more conservative process settings for MBR operation, i.e., higher aeration rates, more frequent cleanings, reduced flux and increased recirculation. The penalty of the conservative operational strategy is an increase in O&M costs.
- The selection of a particular membrane and/or MBR configuration has a strong influence on the overall MBR functioning. Comparing to flat sheet MBRs, the MBRs equipped with hollow fibre membranes were operated at higher fluxes and lower MLSS concentrations, protected by stricter pre-treatment and frequently cleaned by means of physical and chemical cleanings. The hybrid MBR operated in series is characterised by a more stable activated sludge, thus better filterability, operational flexibility and lower O&M costs. Keeping in mind the current discharge requirements and likely negligible permeate quality differences between various configurations, the configuration selection should be based on the local situation.
- Unexpected and undesired events may occur at any time and at any configuration leading to operational perturbations.
- For an MBR designed with high recirculation rates, i.e., with homogenous filterability along the process, the order of particular tanks is not of major importance in terms of a good filterability promotion in the membrane compartment and several good

arrangements exist. The order of the compartments depends rather on the desired permeate quality and the required nutrient removal.

- When low recirculation rates are applied, the MLSS concentration in the membrane compartment should be higher than the critical MLSS concentration of about 10 g/L. In that way, an improvement of activated sludge filterability in the membrane tank may be expected subsequently leading to a filtration process with a reduced fouling propensity.
- A full-scale MBRs comparison aiming at the identification of relationship between the filterability and the operation is difficult due to numerous differences between the plants. The differences, among others, are in design layouts, order of the tanks, membrane configurations, membrane protection and cleaning strategies, removal targets and variations of monitored parameters. All of those result in many changing parameters of influence and subsequently result in difficulties to obtain unequivocal conclusions about MBR operation based on a holistic approach. Therefore, in order to determine the influence of certain parameters on operation and filterability, an isolation of relevant parameters is most likely needed by means of lab or pilot studies.

9.3.3 Energy and costs issues

- Despite no direct influence on MBR energy consumption has been detected, the membrane configuration indirectly influences the energy requirements. Each configuration is associated with operational requirements thus; selection of a particular configuration determines an air scouring method to remove fouling, the frequency of the cleanings, the cleaning agents and their concentration, the pre-treatment type and the oxygen transfer via MLSS concentrations.
- Operation below 50-60% of dry weather flow utilization is associated with a significant energy penalty. Therefore, to improve the energy efficiency, MBRs should be operated at optimal flow conditions, i.e., full hydraulic utilization of the membranes. The optimal flow conditions are closely related to the system flexibility, which may be achieved by designing an MBR with multiple membrane lines.
- The selection of the optimal MBR configuration depends on the specific local situation such as the presence of other treatment processes (e.g. CAS system), condition of the existing infrastructure, space requirements, quality requirements for produced water, expected flow pattern and need of handling storm flows, estimation of future flows, and potential interest for high quality permeate.
- The optimal retrofitting of an old CAS system with MBR technology would include the use of existing infrastructure as equalization tank combined with a hybrid MBR concept. During dry water conditions all wastewater would be directed to the MBR and the not equalized peak flows would be treated in CAS.
- In situations when the existing infrastructure is not available, especially when strict legislation applies and high quality permeate is needed, a stand-alone MBR would be preferred. The decision about the construction of an equalization tank should be based on the expected future flows and their pattern and would require detailed cost

calculation. Construction of multiple membrane lines with a possibility of alternate and independent operation should be considered for high operational flexibility and improved energy efficiency.

- The relation of the activated sludge filterability as the fouling propensity measurement with air-scouring to remove membrane fouling could be utilized to improve yield and/or reduce energy consumption of the MBR process. For example, in case of good sludge filterability, aeration rates could be reduced providing energy savings. Alternatively, due to good filtration properties of activated sludge, the filtration protocol could be optimized by extending the filtration cycle and/or reducing relaxation/backwash cycle. This approach also would be beneficial during operation with poor sludge filterabilites. In that situation, filterability measurement could act as an early warning system indicating increasing fouling propensity of biomass. Based on that, counteractions could be planned in advance eliminating the risk of operational problems related to membrane fouling or even clogging.
- When MBR energy aspects are discussed, it is necessary to distinguish clearly between energy consumption and energy efficiency. The energy consumption represents the absolute value of consumed energy whereas energy efficiency indicates how efficient energy is consumed in respect to a certain parameter. Therefore, during an energy analysis, it is important to determine the ultimate goal behind the energy analysis, e.g., lowering the energy costs or improving the energy efficiency. For example, the MBR energy efficiency with regard to the produced permeate can be improved by the treatment of greater volumes of wastewater. However, this will lead to an increase in energy consumption and higher energy costs. Analogically, if smaller volumes of wastewater are treated the energy consumption will be reduced leading to lower energy costs. However, the energy efficiency will be lower.
- Likewise in case of MBR operation comparison, the analysis of energy consumption and energy efficiency in MBRs should include potential differences between the plants. Therefore, it has to be based on the well defined scope of the study and include the same components of energy consumption, e.g., total WWTP plant or MBR unit or membrane related modules only.

9.4 Overview and evaluation

Filterability is the connecting parameter between membrane bioreactor ‘biology’ and membrane operation. In general, sludge filterability is primarily influenced by flocculation-deflocculation processes and abrupt changes in the inflow or biomass characteristics. The activated sludge filterability goes hand in hand with the amount of submicron particles and flocculation properties of the activated sludge, which in turn are influenced by the inflow conditions. **The most important parameter influencing the activated sludge filterability** in the full-scale municipal MBRs **is the wastewater composition.** Some types of wastewater, some toxic components, or some typical components will lead to a poor filterability. As a process parameter the **temperature** of the MBR content, which of course is greatly dependent of the temperature of the treated wastewater, **is a second dominating parameter** influencing the filtration characteristics of the MBR activated sludge.

It is important to note that, different factors have a different degree of influence on the activated sludge filterability. In other words, if the wastewater composition is optimal for MBR treatment all the other influencing parameters are of minor importance. However, if the wastewater composition constraints MBR treatment all the other factors can hardly overcome the composition problem and are not able to positively influence sludge filterability. If the wastewater composition is in between, the other parameters will determine the activated sludge filterability.

Prediction of the activated sludge filterability is not possible based on a single parameter and a combination of parameters is more appropriate.

With regard to the influence of activated sludge quality on the operation and performance of the full-scale MBRs, a good filterability of the activated sludge is a precondition for an efficient and optimal operation of an MBR. In addition, **periods of good sludge filterability may be exploited by the operators to improve the yield and/or energy efficiency.** In case of a poor activated sludge filterability the membrane performance will be hampered due to sub-optimal filtration conditions. An MBR may also be operated with a poor filterable sludge, however, with a cost penalty associated with the membrane fouling counteractions, i.e., conservative process settings, frequent membrane cleanings and increased energy consumption.

Furthermore, the operation of an MBR in winter conditions is more critical in terms of overall performance and requires more awareness from the operators compared to the summer periods. Analogically, MBRs treating industrial wastewater require more attention, to provide optimal operation, than the ones treating municipal wastewater. **Plant layout is an important aspect of MBR functioning and should be carefully evaluated at the design stage.** It has an influence on the operational flexibility, the activated sludge filterability and the O&M costs. The membrane configuration has also an influence on the overall MBR operation and should be chosen prudently, yet a change from one membrane type to another in the course of plant functioning rather would not be beneficial, and is thus not recommended. The quality of permeate is most likely not affected by activated sludge quality and filterability changes.

The energy consumption and energy efficiency of an MBR system is dependent on many factors, like the system design and layout, the volume of treated flow, the membrane utilization and the operational strategy. **The most important operating parameters influencing the MBR energy consumption and energy efficiency are the hydraulic utilization of the membranes and strategy applied for the membrane air-scouring.** Energy reduction is possible with, yet not limited to, implementation of new aeration strategies, flow equalization or adjusting operational settings to incoming flow. Energy has become an essential focus point in the full-scale MBR operation and energy optimization is required to, at least partially, reduce the gap between energy consumption in MBRs and CAS systems.

Finally, a comparison of the full-scale MBR facilities in terms of operation, energy consumption and energy efficiency should be made on the basis of a well defined scope and should include potential differences between the plants. In other words, it should compare things that are comparable.

The main outcomes of this research are schematically illustrated in Figure 9.1.

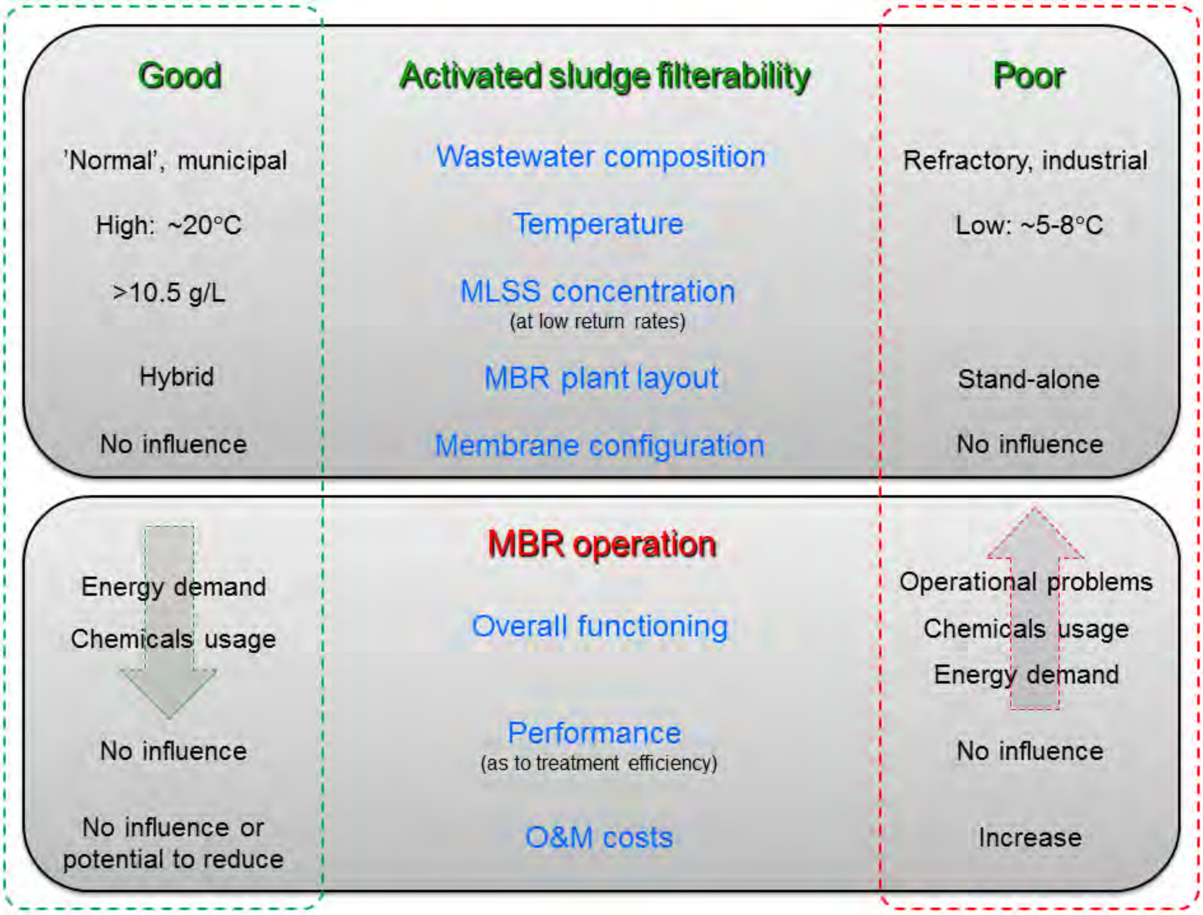

Figure 9.1: Graphical research outcome

9.5 Recommendations for further research

Based on the conducted research described in this dissertation several aspects that deserve attention and require further focus are proposed. The above described recommendations, related to filterability and fouling studies, as well as an optimization of MBR operation and energy efficiency, could be the starting point for further research and, subsequently, lead to improvement of MBR technology.

- Whenever applicable and possible, the DFCm should be combined with microscopic image analysis, e.g. ACTIAS, to provide useful information about the activated sludge composition. The provided information about morphology and bioflocculation state of activated sludge may subsequently help, at least partially, to explain poor activated sludge filterability.
- Chapter 6 presents correlations between membrane fouling and objects smaller or equal to 1 pixel, i.e., 2.8 μm^2. In the microscopic image analysis applied in this

research and with the used microscope and camera configuration, the resolution, or detection limit, is 1 pixel, which corresponds to a square with sides of 1.675 µm. Assuming circular shape of particles, the object would corresponds to an equivalent object diameter of 1.89 µm for the applied magnification of 100x. Therefore, the object diameter appears to be too big to interact with the pore sizes of commonly used MBR membranes. In a recent work of Schultz et al. (2012) filtration behaviour was reported to be correlated with the concentration of particles with a size below 0.2 µm. Hence, information on submicron range is of major importance. Nevertheless, it can be expected that in case of the activated sludge, different particle counting fractions are linked, e.g., increase in one of the fractions would probably also mean an increase in particles in a lower fraction. However, in order to confirm or deny this hypothesis, a microscope with a higher resolution would be required.

- Based on this thesis results, but also on the work of other researchers (Geilvoet 2010, Lousada-Ferreira 2011, Van den Broeck 2011), the influence of submicron particles on filterability and fouling should be further investigated. The analysis of the submicron range could help on identifying the fraction responsible for filterability deterioration, being relevant to membrane fouling. However, such a study would require a well-thought-out and -defined protocol and reliable particle counter, able to provide information about the low range of the submicron particles, i.e., below 0.5 µm, and preferable within the colloidal range of 0.01-1 µm. Such an attempt with an on-line submicron particle analysis was initiated at Berlin Centre of Competence for Water and preliminary lab-tests as well as short-term pilot results were published by Schultz et al. (2012).
- In respect to the membrane fouling studies, standardised methods for determination of activated sludge filterability, relative hydrophobicity, EPS and SMP concentrations are required. Currently, different research groups utilize different measurement protocols leading to the situation that similar studies provide contradicting results. The differences in quantification, extractions and analytical methods make it very difficult to compare different studies in a reliable way.
- The DFCm protocol could be extended by a relaxation step. Implementation of a relaxation step between the filtration cycles would enable estimation of filtration resistance recovery. Subsequently, an indication about irreversible fouling propensity of activated sludge could be obtained likewise in case of VFM (Huyskens 2012). However, as already mentioned by Geilvoet (2010), implantation of the relaxation step would rather provide information on the effectiveness of the physical membrane cleaning than about activated sludge characteristics.
- In order to determine the influence of the temperature on the raw wastewater composition in respect to activated sludge filterability, both readily biodegradable and slowly biodegradable compounds in the influent need to be identified and measured during the year. Based on the obtained results, the impact of temperature changes on the particular wastewater fractions may be assessed.

- Implementation of frequent filterability measurements at an MBR site may deliver potential filtration process optimizations with respect to filtration and relaxation/backwash protocols. Based on obtained filterability results, for example, the filtration protocol could be prolonged or shortened in case of good or poor filterable activated sludge, respectively. The proposed approach could also act as an early warning system for operators and as a membrane aeration energy optimization tool.
- Although optimisation of MBR energy consumption is certainly required, most probably only new developments can significantly reduce current energy requirements of the MBR process. These new developments should concentrate on low-energy membrane cleaning methods or novel fouling mitigation methods like mechanical cleaning with granular medium (Siembida et al. 2010, Rosenberger et al. 2011, Pradhan et al. 2012), membrane vibration (Bilad et al. 2012), electric field (Akamatsu et al. 2010) and others (Drews 2010).

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Appendix **A**

Composition of road salt sample

A. Composition of road salt sample

The salt composition was determined by the X-Ray powder Diffraction (XRD) and semi-quantitative X-ray Fluorescence (XRF) analysis performed by Ruud Hendrikx from the Department of Materials Science and Engineering of the Delft University of Technology.
The measured XRD pattern is presented in Figure A.1; red lines show the peak position and intensity of halite, NaCl, according to the PDF4 ICDD database (ICDD). Apart from potassium, calcium and magnesium, together 1.8 weight% of total mixture, no other compounds in significant quantities were detected. The XRF results are in weight% of total mixture and are presented in Table A.1.

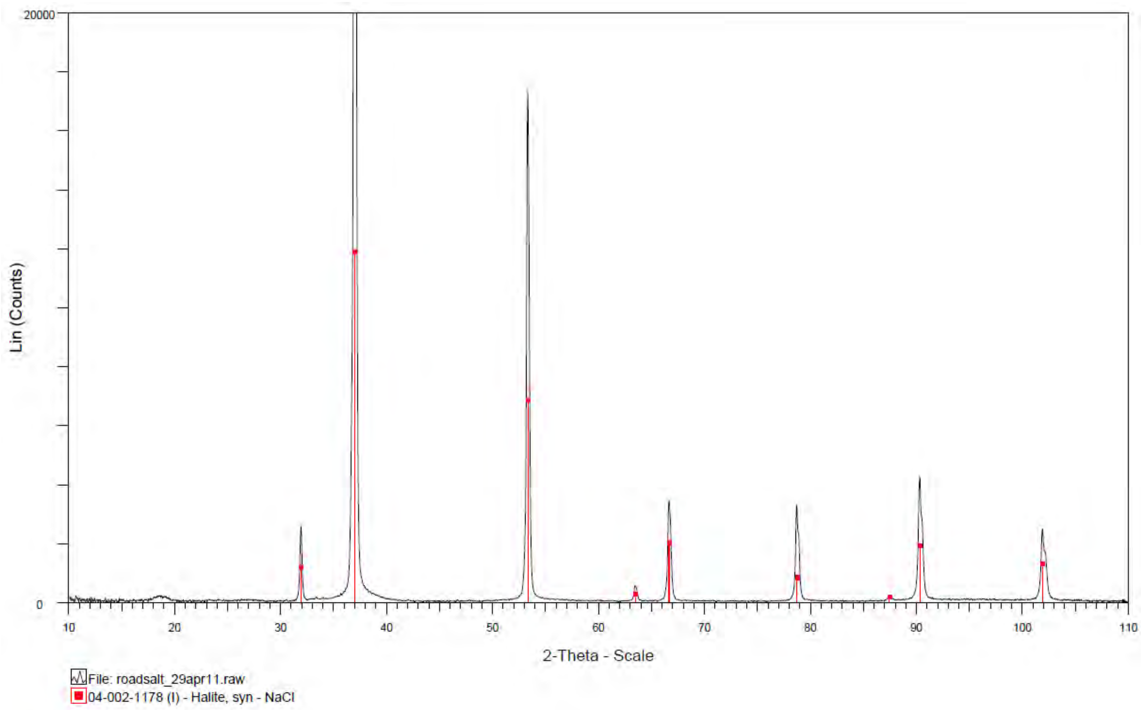

Figure A.1: Results of X-ray powder Diffraction (XRD) analysis of the road salt sample. The intensity of reflected X-rays, expressed in counts, is shown on the x-axis, and increasing values of 2-theta, i.e., diffraction angle in degrees, are shown on y-axis. The XRD provides qualitative analysis to identify present minerals, whereas, XRF quantitatively determines detailed composition of the salt sample as presented in Table A.1

Table A.1: Results of X-ray fluorescence analysis of the road salt sample

Atomic number	Element	w/w%	Standard error	Atomic number	Element	w/w%	Standard error
9	F	<		47	Ag	<	
11	Na	41.6	0.55	48	Cd	<	
12	Mg	0.524	0.058	49	In	<	
13	Al	<		50	Sn	<	
14	Si	0.0151	0.003	51	Sb	<	
15	Px	<		52	Te	<	
15	P			53	I	<	
16	Sx	0.667	0.074	55	Cs	<2e	
16	S			56	Ba	<2e	0.0053
17	Cl	56.42	0.55	57	La	<	0.0072
18	Ar	0.0483	0.0051	58	Ce	<	
19	K	0.154	0.017	59	Pr	<	
20	Ca	0.47	0.052	60	Nd	<	
21	Sc	<		62	Sm	<2e	
22	Ti	<		63	Eu	0.0098	0.0083
23	V	<		64	Gd	0.0103	0.0043
24	Cr	<		65	Tb	<	0.0047
25	Mn	0.0061	0.0014	66	Dy	<2e	
26	Fe	0.0269	0.003	67	Ho	<	0.0075
27	Co	<		68	Er	<	
28	Ni	<		69	Tm	<	
29	Cu	<		70	Yb	<	
30	Zn	<		71	Lu	<	
31	Ga	<		72	Hf	<	
32	Ge	<		73	Ta	<	
33	As	<		74	W	<	
34	Se	<		75	Re	<	
35	Br	0.0206	0.001	76	Os	<	
37	Rb	<		77	Ir	<	
38	Sr	0.0073	0.0008	78	Pt	<	
39	Y	<		79	Au	<	
40	Zr	<		80	Hg	<	
41	Nb	<		81	Tl	<	
42	Mo	0.0053	0.0019	82	Pb	<	
44	Ru	<		83	Bi	<	
45	Rh	<		90	Th	<	
46	Pd	<		92	U	<	

Legend: < - concentration is lower than 50 mg/kg; <2e – weight% lower than two standard errors.

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In 2008, he started a PhD-study at Water Management Department of the Delft University of Technology (the Netherlands) in the framework of Marie Curie Host Fellowship called 'MBR-TRAIN'. This EU funded project aimed at process optimisation and fouling control in membrane bioreactors for water treatment. In addition, he also participated in the Dutch research project 'MBR2+', which was dedicated to identify and further develop the energy saving in design, operation and management of MBR plants. During his work within the MBR-TRAIN and the MBR2+ projects he focuses on activated sludge filterability, operation, performance of the full-scale membrane bioreactors treating municipal and industrial wastewater. Another point of interest, investigated during the study is energy consumption, energy savings opportunities and operational costs of the large scale MBRs.

Paweł presented his work on several international and national conferences organized by IWA, EMS, WEF, AMTA and MBR-Network. In 2012 he won the Best Poster Award at the NMG-BMG Membrane Symposium and Poster Day held at Shell in Amsterdam, the Netherlands.

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List of publications

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