

CRANFIELD UNIVERSITY

ANTONIO CAMPITELLI

CLOTH FILTERS AS ALTERNATIVE TO THE CONVENTIONAL
MEMBRANES FOR SOLIDS REMOVAL FROM UASB EFFLUENTS

School of Water, Environment and Energy
Research Degree

MSc by Research
Academic Year: 2017- 2018

Supervisor: Prof. Bruce Jefferson and Dr. Ana Soares
Associate Supervisor: Dr. Luca Alibardi
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This thesis is submitted in partial fulfilment of the requirements for
the degree of MSc by Research

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ABSTRACT

Anaerobic Membrane Bioreactors (anMBR) represent one of the most promising technology to achieve energy neutral or positive wastewater treatment with a high effluent quality. Despite this potential, its implementation is still restricted by the high costs of the ultrafiltration membrane. This thesis investigated the feasibility of using a cloth filter as an alternative to the conventional membranes for solids removal from an Upflow Anaerobic Sludge Blanket (UASB) effluent. The advantages of the cloth filter are the low cost of the material, simpler operation and maintenance procedures. However, solids removal efficiencies are lower than those of the ultrafiltration membrane. The removal efficiency depends on the pore size of the cloth and on the influent particle and solid profile load characterisation.

The aim of this study is to assess the efficacy of cloth filters to treat UASB effluent. Effective solids separation was possible with the cloths tested with optimal performance achieved for the 1 and 10 μm non-woven cloths operated under low cross flow or dead end conditions. In these cases, solids removal efficiencies up to 67% were possible whilst maintaining a flux of 1000 LMH. In relation to the UASB operation, the sludge blanket height in the UASB was seen to correlate to the suspended solids removal and resulted in a progressively higher median particle size exiting the reactor. Further, incorporation of the separation in the recycle line deteriorated performance a little indicating that the cloth filters would be best configured post the recycle line.

Keywords: *anaerobic sewage treatment, particle characterisation, cloth filter*

ACKNOWLEDGEMENTS

My first acknowledgement goes to my supervisor Professor Bruce Jefferson who, despite all adversities, always motivated me to improve myself in every way encouraging me to think and to interpret the data as a real researcher. Thanks to Dr. Ana Soares for giving me the opportunity to undertake this exciting experience. Special thanks to Dr. Luca Alibardi for his presence and guidance essentials in the success of this research. In this respect I would like to thank Professor Francesco Fatone and Dr. Anna Laura Eusebi who imparted me the passion for this field and the research.

Thanks also go to my industry sponsors: Scottish Water, Severn Trent Water, Thames Water and Anglian Water for their financial and technical support.

Thanks to all friends of Cranfield especially Sabrina and Samuela for the support and help provided me during this year. I also want to thank my old friends Paolo, Giovanni, Marco, Gianmarco, Alessandro, Antonio, Emanuele, Davide, Lorenzo and Leo for making me feel at “home” during our chats.

Last, but not least, thanks to my all family especially to my parents who taught me to never give up and to pursuit my dreams.

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LIST OF EQUATIONS

LIST OF ABBREVIATIONS

AnDMBR	Anaerobic Dynamic Membrane
AnMBR	Anaerobic Membrane Bioreactor
ASP	Aerobic Sludge Process
CAPEX	Capital expenditure
COD	Chemical Oxygen Demand
CFV	Cross Flow Velocity
DAF	Dissolved Air Flotation
DM	Dynamic Membrane
HRT	Hydraulic Retention Time
LMH	$\text{L m}^{-2} \text{h}^{-1}$
d_{50}	Mean particle diameter
MF	Microfiltration membrane
OPEX	Operational expenditure
PSD	Particle Size Distribution
pCOD	Particulate Chemical Oxygen demand
PVDF	Polyvinylidene fluoride
I	Scatter intensity
sCOD	Soluble Chemical Oxygen Demand
TSS	Total Suspended Solids
TMP	Transmembrane Pressure
UF	Ultrafiltration membrane
UK	United Kingdom
US	United States
df	Fractal dimension
UASB	Upflow Anaerobic Sludge Blanket
v_{up}	Upflow velocity
Q	Wave number

1 INTRODUCTION

1.1 Background

The development and application of engineered biological processes for wastewater treatment represented a turning point in achieving the stringent environmental quality standards imposed by national and international legislations. Aerobic processes and in particular the activated sludge process (ASP) has been and still are the most developed and applied biological process for municipal wastewater treatment due to the ability to achieve high quality effluent. The high performances of the aerobic system are related to their ability to remove organic compounds in the forms of particulate, soluble and colloidal matter, nutrients (ammonium and phosphorus), and guarantee reduced odour emissions (Metcalf & Eddy, 2014). However the high costs of aerobic treatment, mainly associated to the energy consumption of the aeration system (Gude, 2015), have increased the interest towards anaerobic processes due to their lower energy demand, reduced sludge production and the possibility to recover energy from wastewater thanks to the produced biogas (Metcalf & Eddy, 2014). Despite these advantages, anaerobic processes are affected by low temperatures and low strength wastewater (e.g. municipal), and as such have so far found implementation only in tropical regions (Chernicharo *et al.*, 2015). Nevertheless, McAdam *et al.* (2011) demonstrated that the benefits of anaerobic processes can also be achieved at lower temperature, importantly at temperatures that relate to temperate climates. In fact, integrating the anaerobic processes into existing flowsheets has the potential to decrease electricity demand by 60% (McAdam *et al.*, 2011). However, to extend this to the development of energy neutral flowsheets, further process developments are still needed, in particular for low temperature applications.

In recent times, anaerobic reactors have been coupled to membrane systems (Anaerobic Membrane Bioreactor – anMBR) to overcome the limitations posed through low temperatures. Reported trials with anMBR to treat municipal sewage in temperate climates has demonstrated the ability to deliver a valid treatment alternative (Martinez-Sosa *et al.*, 2012; Liao *et al.*, 2006; Wen *et al.*,

1999). The success of this technology is related to the improved biomass retention that is ensured by the UF membrane with consequent increase in biogas production (Skouteris *et al.*, 2012; Judd, 2011; Ross, Bill & Strohwlad, 1994). In addition, the membrane is able to guarantee a high effluent quality (Melin *et al.*, 2006) with no need of further downstream treatments for solids and pathogens removal.

The current limitations for the implementation of anMBR schemes for municipal wastewater treatment are the low permeate fluxes, not exceeding 10-20 LMH, (Judd, 2011), the high capital investments and the costs linked to the maintenance of the membranes and energy demand for gas sparging (Jefferson *et al.*, 2018). These limitations have encouraged the investigation of valid alternatives to the conventional membranes with technologies such as depth filters, microscreens, dynamic membranes (DMs) and flotation system (e.g. Dissolved Air Flotation - DAF)) representing the most promising technologies (Lier *et al.*, 2018). The solids removal efficiencies are expected to be lower than those achievable when using membranes but the lower capital and operative expenditures (CAPEX and OPEX) can attract interest for their development and implementation in full scale applications (Jefferson *et al.*, 2018).

To date there is limited knowledge on the application of these technologies for the treatment of anaerobic effluents. Only two studies assessed the application of DAF (Penetra *et al.*, 1999) and sand filter (Tyagi *et al.*, 2009) for solids removal in UASB effluents while no experiences are reported on the use of cloth filtration with anaerobic effluents. The development of DM filtration systems has mainly focused on the treatment of high solids anaerobic effluents (Ersahin *et al.*, 2017, 2013; Saleem *et al.*, 2017; Alibardi *et al.*, 2016, 2014) with influent TSS concentrations up to 15 g L⁻¹ while only few examples are reported on the application in UASB effluents (Quek *et al.*, 2017; Zhang *et al.*, 2010) The growing interest on dynamic membrane filtration (Hu *et al.*, 2018; Ersahin *et al.*, 2013) suggest that this technology can represent a promising alternative to UF for anaerobic effluent treatment, due to the low cost of the filtration media

(woven or non-woven mesh) and the potential solids separation efficiency it can achieve.

To understand the potential to utilise non membrane technologies there is a need to understand both the solids load profile and particle characteristics (size and shape) coming off the anaerobic reactor as well as the performance of different separation technologies (Hu *et al.*, 2018). For instance, to achieve removal efficiency up to 90% for particles of 20 μm in a DAF process, the aeration system has to deliver bubbles of 40 μm (Edzwald, 1995). Whereas, for microscreens, solids removals of 50-80 % can be ensured with openings of 15-60 μm (Ljunggren, 2006). In separation technology that relies on the formation of a cake (dynamic membranes and cloth filters), the particle size of the incoming solids affects the formation and resistance of the cake layer and, in turn, the quality and flux of the permeate (Mackley and Sherman, 1993; Wakeman and Tarleton, 1991).

The PSD in an anaerobic effluent depends on the feed water characteristics and the reactor operational conditions, which for a UASB reactor is influenced by the solids removal achieved by the sludge blanket layer (Mahmoud *et al.*, 2003). Accordingly, there is an interconnection between the operation of the anaerobic reactor and the identification of the appropriate separation technology that can only be resolved through particle characterisation of the solids exiting the anaerobic reactor (Baker, 2004; Svarovsky, 1990).

The aspiration of this work is to explore such issues in the case of an UASB reactor treating settled sewage in a temperate climate. The work explores both the concentration and nature of the solids exiting the reactor as well as establishing the efficacy of cloth filter materials as a basis for the solids liquid separation process post the UASB.

1.2 Aim and objectives

The overall aim of this thesis is to understand what influences the feasibility of cloth filters for UASB effluent polishing. To deliver against the overall aim the following objectives has been defined:

1. **Objective 1:** To understand the impact on the characterisation of solids exiting from two different UASB configurations working at set operational conditions.
2. **Objective 2:** To understand what conditions have a predominant role in the filtration process of UASB effluents with cloth filter.
3. **Objective 3:** To establish the type of cloth filter that at set filtration operational conditions provided the best solids removal from UASB effluent.

In this way the thesis aims to provide the basis for the treatment of UASB effluents with cloth filters used as alternative to the membrane.

1.3 Thesis Plan

The thesis is structured in the form of a series of formatted papers. All experiments were conducted by Antonio Campitelli at Cranfield University in collaboration with Judit Ribera Pi from Catalonia Polytechnic University (Barcelona, Spain) who contributed to the data collection for Chapter 2. Each chapter was entirely written by the first author Antonio Campitelli and edited by Prof. Bruce Jefferson and Dr. Luca Alibardi.

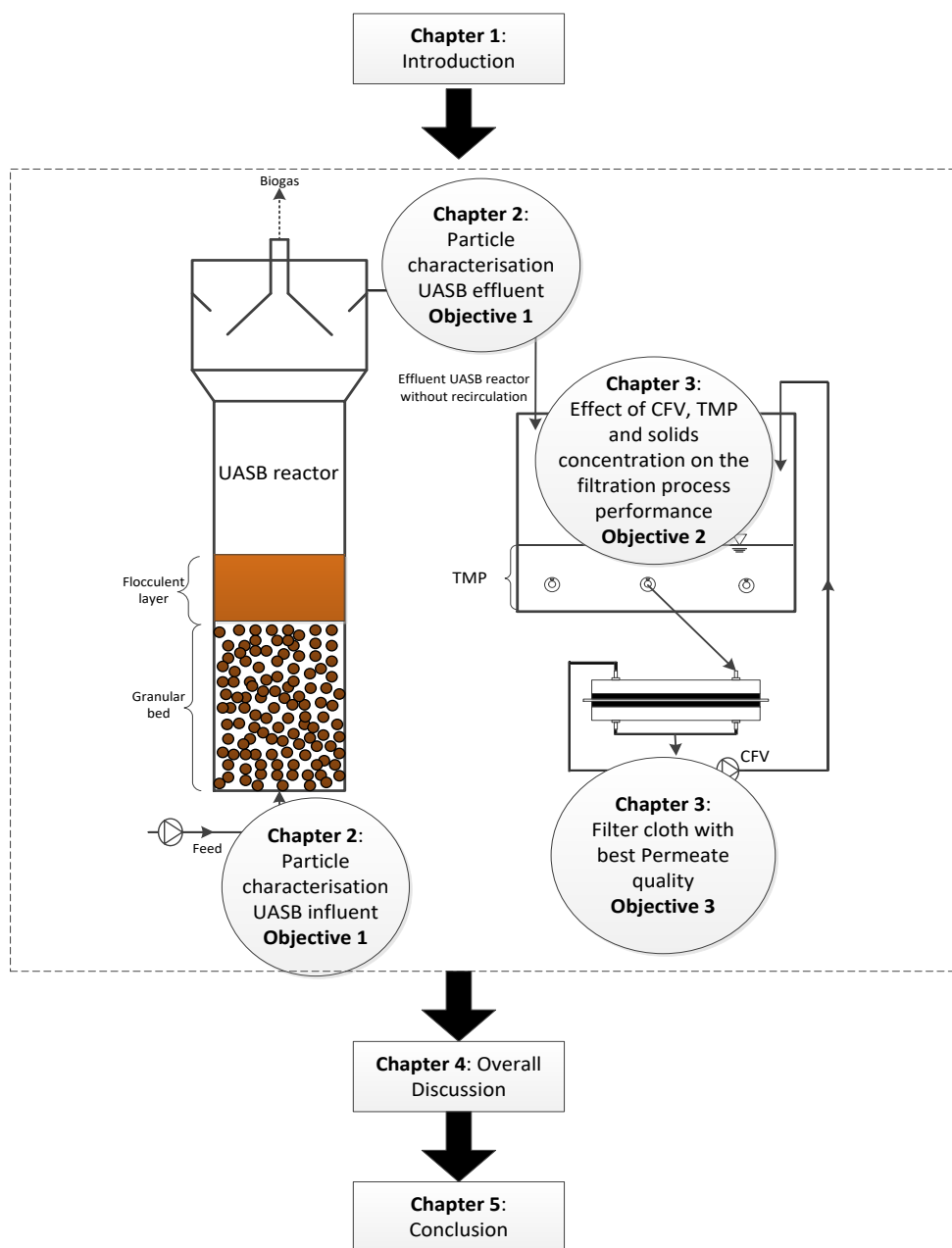
Chapter 2 compares the performance between an UASB reactor coupled with an ultrafiltration membrane in the recycle line (anMBR) and an UASB reactor without separation technology in the recycle line, operating at the same operational conditions. The impact on the solids characterisation was assessed in terms of particulate matter removal efficiency and particles size distribution (objective 1).

Chapter 3 investigates the performance of cloth filters with different pore sizes, structures (woven or non-woven) and materials (polyamide and polypropylene). The cloths were tested at different Cross Flow Velocities (CFV), Transmembrane Pressures (TMP) and solids concentration to understand the role of each of them on the filtration process performance (objective 2). The work then proceeded with the identification of the conditions (CFV, TMP) and

cloth filter (pore size and structure) able to polish the UASB effluents with the best solids removal (objective 3).

Chapter 4 discussed the overall implication of the work in relation to anaerobic sewage treatment with non-membrane separators.

Chapter 5 describes the overall conclusion from the work, providing the key aspects for the implementation of an UASB reactor with cloth filter.



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2 Comparison of a UASB configured with and without a membrane for anaerobic sewage treatment: Impact on solids profiles

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2.1 Abstract

Upflow Anaerobic Sludge Blanket (UASB) reactors are considered a promising technology for the anaerobic treatment of municipal wastewater, even at low temperatures. To strengthen the success of this technology in temperate climates, the combination of the UASB reactor with a filtration membrane unit is considered sensible leading to the generation of an Anaerobic Membrane Bioreactor (anMBR). The significant capital and operational costs of the membrane unit have recently encouraged the research of viable alternatives for solids removal such as depth filters, micro-screens and dynamic membranes and dissolved air flotation. Efficacy of these technologies in removing solids is strictly linked to the amount and nature of the solids to be removed. This study considers this by measuring the nature of the solids exiting the UASB reactor when configured as either a traditional UASB or as an anMBR. In these conditions differences in solids concentration, particles size, structures and removal efficiencies were observed. The anMBR configuration as seen to reduce removal across the UASB linked to the membrane retaining all colloids. Overall, the work suggested that if alternative separation technologies would be used then operating them after the recycle loop may be more appropriate.

2.2 Introduction

The Upflow Anaerobic Sludge Blanket (UASB) reactor has been demonstrated to be a suitable technology for anaerobic treatment of municipal wastewater, especially in tropical regions (Chernicharo *et al.*, 2015). Early studies on UASB also demonstrated its applicability at low temperature (Lettinga *et al.*, 1983).

However, when also fed with low strength sewage (e.g. municipal wastewater), low temperatures have been reported to represent a challenge for the implementation of anaerobic process in UASB reactors (Martin Garcia *et al.*, 2013). This problem can be overcome by coupling the UASB reactor with an ultrafiltration membrane (Anaerobic Membrane Bioreactor - anMBR), as the membrane enables improved microbial activity thanks to an enhanced biomass retention (Ross *et al.*, 1994). AnMBRs have now been demonstrated to be a viable option for the anaerobic treatment of low strength sewage, including settled sewage, under temperate climatic temperatures (Martin Garcia *et al.*, 2013; Martinez-Sosa *et al.*, 2012; Liao *et al.*, 2006; Wen *et al.*, 1999). Inclusion of the technology into sewage treatment flowsheets offers a reduction in the energy demand for treatment, increased energy production which can lead to net energy neutral operation of the entire flowsheet is known to be one of the core technologies when considering the switch to resource factory thinking (Jefferson *et al.*, 2018)

Whilst the membrane affords treatment enhancements it exerts a relatively high impact on the capital and operating expenditure of the overall plant (OPEX and CAPEX) (Judd, 2011; Gander *et al.*, 2000). The impacts are heightened in anMBRs compared to aerobic MBRs as research is indicated a lower design flux and a higher sparging intensity is required (Jefferson *et al.*, 2018; van Lier *et al.*, 2018). To a degree this can be minimised by restricting the solids flux going onto the membrane by only filtering the effluent of the UASB rather than the full biomass concentration of a suspended growth reactor (Wang *et al.*, 2018; Martin Garcia *et al.*, 2013). Further, operation against minimised solid fluxes enables greater adaptability to hydraulic load such that the membranes can be turned up during periods of elevated flow without severe long term deterioration (Wang *et al.*, 2018b). The consequence is that the membrane can be sized against an average flow rather than the peak flow to the site resulting in substantial cost and energy savings (Wang *et al.*, 2018b; Krzeminski *et al.*, 2012). However, even when optimised, the membrane remains a major cost component to the total expenditure of the technology raising questions concerning whether alternative solid liquid separation processes could be

utilised such as micro-screens, depth filters, dynamic membranes or dissolved air flotation systems. All the alternatives are likely to reduce total costs and energy but will generate a poorer effluent quality than a membrane such that it is important to balance the two aspects to understand the true efficacy of the alternatives. To illustrate, development of a resource factory flowsheet using UASBs utilises downstream treatment with degassing membranes for methane recovery and fixed bed adsorbers for nutrient removal and recovery (Jefferson *et al.*, 2018). Effective operation of these technologies needs low solids (sub 40 mg L⁻¹) and are limited on the maximum size of particles to avoid channel clogging (< 10 µm) (Heile *et al.*, 2017; Cookney *et al.*, 2016). However, when used for just treatment of organics in countries such as Brazil, the effluent constraint are less severe with accepted suspended solids above 75 mg L⁻¹ and no specific requirements on the particle size (Chernicharo *et al.*, 2015). Accordingly, different solid liquid separators can be used in different situations.

Effective operation of most solid liquid separation processes, such as sedimentation, flotation and filtration depends on the quantity and character of the solids that require separation (Edzwald, 2015; Svarovsky, 1990). However, to date, there is a genuine paucity of data regarding the particle size distribution and quantity that are likely to exit the UASB reactors beyond simple suspended solids information (Daud *et al.*, 2018). For instance, typical suspended solids concentrations exiting UASBs are 101 mg L⁻¹ treating a crude sewage at low temperature (Barbosa and Sant'Anna, 1989), while around 70 mg L⁻¹ at temperate climate (Agrawal *et al.*, 1997). Whereas in tropical countries the solids concentration varies between 60 and 360 mg L⁻¹ (Chernicharo *et al.*, 2015).

Typical operation of the UASB includes a recycle to maintain a stable upflow flow velocity and the build-up of a sludge blanket that aids solids capture within the main reactor section (Ives, 1968). The device used to separate out the solids from the UASB effluent can be configured within this loop or external to it. In the former case the retained solids build up and are returned to the UASB for inclusion into the blanket layer prior to being desludged. However, it is unclear

how this alters the character of the particles and the separation efficacy of the sludge blanket and hence which configuration is best. Accordingly, the aim of this study is to characterise UASB effluents in terms of the solids loading profile and the particle size distributions to inform the discussion of the efficacy of non-membrane UASB configurations.

2.3 Materials and Methods

2.3.1 Experimental setup

The UASB reactors, operating in parallel, had an overall height of 2 m and diameter of 0.1 m with a solid/liquid/gas separation at the top of the column (Model products, Wootton, UK). Both reactors were inoculated with 7.5 L of granular and 7.5 L of flocculent sludge, which came from a previous experimental run. The sludge was originally collected from full-scale mesophilic UASB treating paper and pulp mill industry effluent. The reactors were left to stabilise for the first two weeks monitoring the biogas production, COD, and TSS. A primary effluent from Cranfield University campus (UK) sewage treatment work was fed to the bottom of the reactors by means of a peristaltic pump (520U, Watson Marlow, Falmouth, UK). The reactors operated at hydraulic retention time (HRT) of 8h and upflow velocity (v_{up}) of 0.4 m h⁻¹. This latter was kept by the internal recirculation through another peristaltic pump (620S, Watson Marlow, Falmouth, UK). One of the UASB reactors was connected to a pilot scale ultrafiltration (UF) module configured within the recycle line while the second had no solid separation device in the recycle line (Figure 2-1).

Influent and effluent samples were collected from upstream and downstream sampling points from both UASB reactors (Figure 2-1). The influent was analysed for a period of one month, while the effluents for a period of two months. Samples of 100 mL were collected every hour for a period of 24 hours by using an autosampler (ISCO 3710). Afterwards, the 24 collected samples were mixed and analysed as soon as possible. Influent and effluents samples were analysed for Total Suspended Solids (TSS), Chemical Oxygen Demand (COD), pH, Alkalinity and Particle Size Distribution (PSD). Comparative data on

the permeate from the membrane was taken from a previous study using the same system (Wang *et al.*, 2019).

2.3.2 Analytical Methods

Total Suspended Solids (TSS), pH and Alkalinity was measured according to standards methods (APHA, 2005). Chemical Oxygen Demand (COD) was analysed with Merck test kits (Merck KGaD, Darmstadt, Germany).

2.3.3 Particle Size Distribution Analysis

In this study, the PSD was measured by means of a laser diffraction instrument (Mastersizer 2000/3000, Malvern Instrument Ltd, Malvern, UK). For each sample, particle size was analysed every minute for a maximum period of 10 minutes. The results were expressed as volume weighted distribution. The number of particles associated to a diameter from 0.45 to 3500 μm was calculated by weighting the solids concentration by the volume fraction in any specific size band. This was then converted to a volume by applying a fixed density term of 1.1 g cm^{-3} before dividing by the average volume of a particle within that size band to reveal the total number of equivalent particles present.

Fractal dimension of the particles was calculated from the PSD measured with the light scattering technique as described in other studies (Jarvis *et al.*, 2008, 2005; Jarvis and Jefferson, 2005). Briefly, the logarithm of the scatter intensity (I) is plotted against the wave number (Q) leading to the generation of a straight-line plot, the gradient of which is defined as the fractal dimension (d_f). All the data is extracted from the particle size measurement such that each PSD has an associated average fractal dimension associated to it.

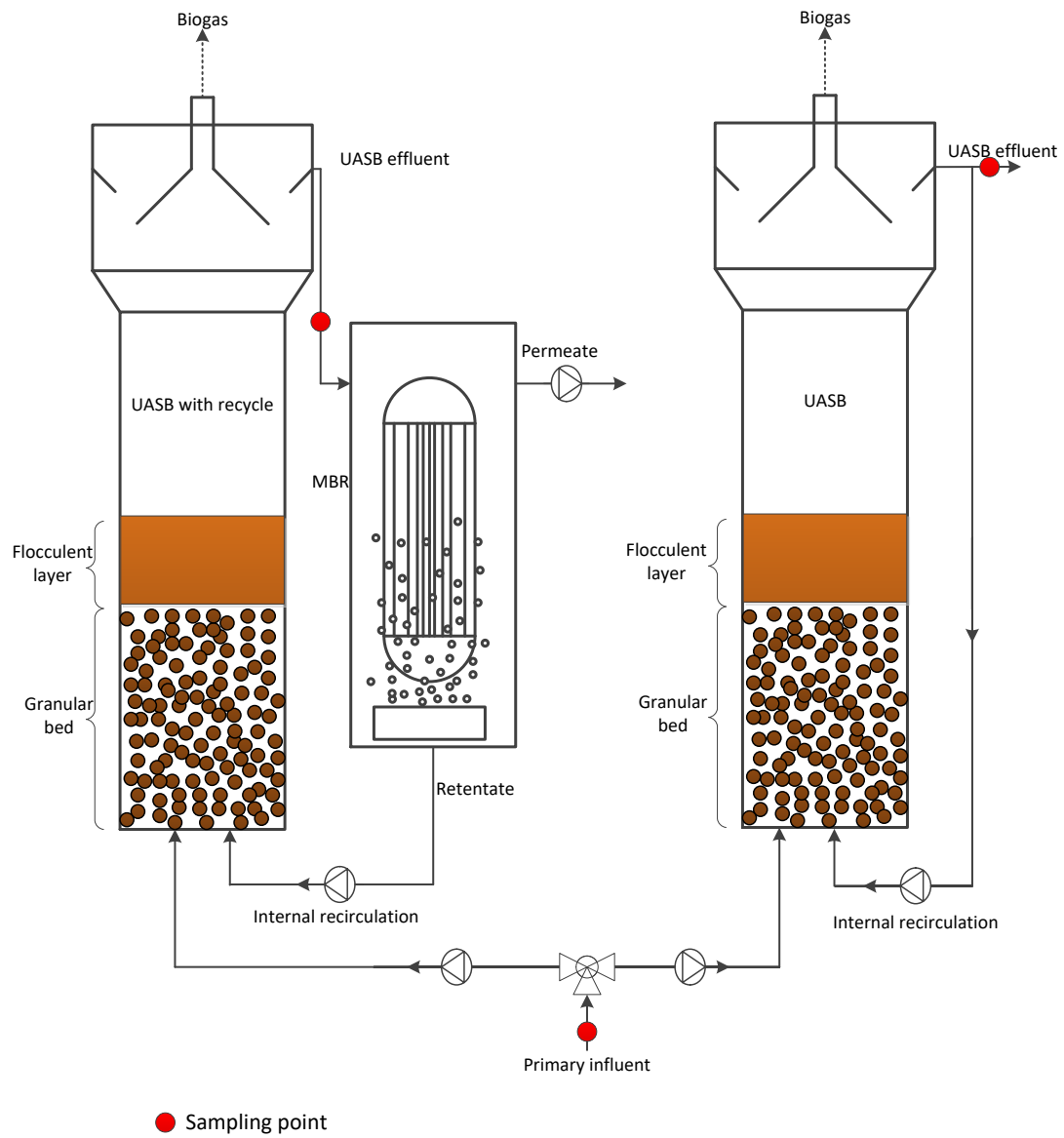


Figure 2-1 Schematic representation of the two UASB reactors with and without recycle

2.4 Results

2.4.1 Comparison of effluent character from the two UASB reactors.

The inclusion of the membrane reactor into the recycle line (anMBR configuration) marginally increased, but not in a statistically significant manner, the concentration of suspended solids and COD compared to the UASB operating with just a recycle line (UASB configuration) (Table 2-1). To illustrate, the suspended solids concentration decreased from an influent level of $88 \pm 20 \text{ mg L}^{-1}$ to $50 \pm 19 \text{ mg L}^{-1}$ for the anMBR configuration and $40 \pm 13 \text{ mg L}^{-1}$ for the UASB configuration. The corresponding COD was $195 \pm 81 \text{ mg L}^{-1}$ for the influent and $123 \pm 35 \text{ mg L}^{-1}$ and $98 \pm 24 \text{ mg L}^{-1}$ for the anMBR and UASB configurations respectively. The difference was predominately observed in relation to the particulate COD with effluents levels of $71 \pm 36 \text{ mg L}^{-1}$ for the anMBR configuration and $48 \pm 22 \text{ mg L}^{-1}$ for the UASB configuration. This compares to a soluble COD of $54 \pm 11 \text{ mg L}^{-1}$ and $50 \pm 11 \text{ mg L}^{-1}$ respectively demonstrating the impact was related to differences in the solids carry over through the UASB reactor. The impact of the membrane on the effluent profile can be observed through comparison to previous trials with the same set up where the effluent COD and SS post membrane were $39 \pm 2 \text{ mg L}^{-1}$ and below the detection limit during a period when the influent levels were $168 \pm 28 \text{ mg L}^{-1}$ and $105 \pm 9 \text{ mg L}^{-1}$ respectively (Wang *et al.*, 2019).

The process stability of the two reactors is confirmed by the consistency of the pH and by the increase of the alkalinity in the effluents (Table 2-1). For instance, pH was 7.9 ± 0.2 in both effluents during the sampling campaign with corresponding alkalinity levels of $286 \pm 49 \text{ mgCaCO}_3 \text{ L}^{-1}$ and $309 \pm 55 \text{ mgCaCO}_3 \text{ L}^{-1}$ for the anMBR and UASB configurations respectively compared to $244 \pm 43 \text{ mgCaCO}_3 \text{ L}^{-1}$ in the influent (Table 2-1). Accordingly, the observed differences in pCOD and SS are associated to changes in the physical functions within the reactor most likely associated to the changes in particle characteristics and the grade efficiency performance of the sludge blanket layer.

Table 2-1 Characterisation and performance of the effluent of UASB reactors with and without membrane in the recycle line.

			Influent	Effluent UASB with membrane in recycle line	Effluent UASB reactor without membrane in recycle line
				Vup=0.4 m h ⁻¹	Vup=0.4 m h ⁻¹
	pH	-	7.7±0.2 (35)	7.9±0.2 (35)	7.9±0.2 (35)
	Alkalinity	mg CaCO ₃ L ⁻¹	244±43 (14)	286±49 (14)	309±55 (14)
	TSS	mg L ⁻¹	88±20 (33)	50±19 (33)	40±13 (27)
	COD	mg L ⁻¹	195±81 (38)	123±35 (38)	98±24 (25)
	PCOD	mg L ⁻¹	138±81 (38)	71±36 (38)	48±22 (25)
	SCOD	mg L ⁻¹	62±24 (38)	54±11 (38)	50±11 (25)
Removal efficiency	TSS	%	-	44±0.2	54±0.1
	COD	%	-	40±0.2	44±0.2
	PCOD	%	-	44±0.2	55±0.3
	SCOD	%	-	12±0.3	14±0.3

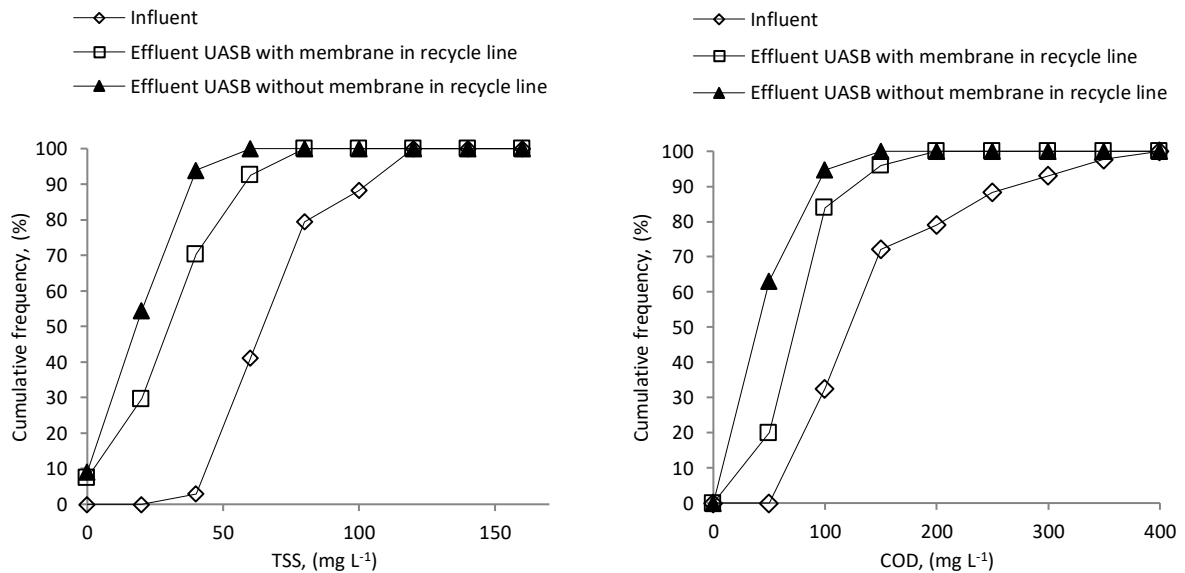


Figure 2-2 Cumulative frequency curves of TSS and COD concentrations in the influent and effluent of both UASB reactors.

2.4.1.1 Comparison of the PSD of influent and effluent of the UASB and anMBR configurations

The particles in the influent wastewater were characterised by a monomodal distribution with a mode size that varied between 60 and 150 μm during the sampling period. The volume percentage above 10 μm is used as surrogates for percentage of particles that should be easily separated by flotation/filtration respectively and that above 100 μm for sedimentation (Parsons and Jefferson, 2006; Bouwer, 1987). Overall, the 10 μm and above fraction of the particles accounted for between 73 and 93 % of the total volume distribution across the trials. In comparison between 5 and 20 % of the particle volume existed below 10 μm indicating only a small fraction that should be difficult to remove if classical separation mechanisms are fully operating within the sludge blanket.

In contrast, the particle size distributions of the effluents from both reactors showed a significantly different profile from the influent with three observable peaks approximately associated to sizes of below 1 μm , nearby 10 μm and above 100 μm (Figure 2-3). For instance, the percentage of the total volume associated with particles above 10 μm varied between 59-96 % for the anMBR configuration and between 31 % and 94 % for the UASB configuration. The relative distribution varied across the trials with periods where a large peak in fines (<1 μm) or large aggregates (>100 μm) could be observed. This reflects dynamic changes inside the bed and is consistent with previous reports associated with periodic release of large bubbles due to the low strength and low temperature of operation (Wang *et al.*, 2019).

The PSD represent a fast method to define the proper separation system for removal of particles with specific size distribution. In presence of high concentrated suspensions, sedimentation systems are suitable for removal of particle with $d_{50} > 100 \mu\text{m}$ (Kavanaugh *et al.*, 1980). In fact primary sedimentation tank can ensure suspended solids removal efficiency from 50 to 70 % (Metcalf & Eddy, 2014). This values can rise up to 90 % utilising filtration technology for removal of particles with $d_{50} < 30 \mu\text{m}$ and mass concentration below 50 mg L^{-1} (Kavanaugh *et al.*, 1980). However the filtration diminishing in

the entrapment of particles between 0.5 and 10 μm , since at these size the diffusion becomes the main removal mechanism (Bouwer, 1987).

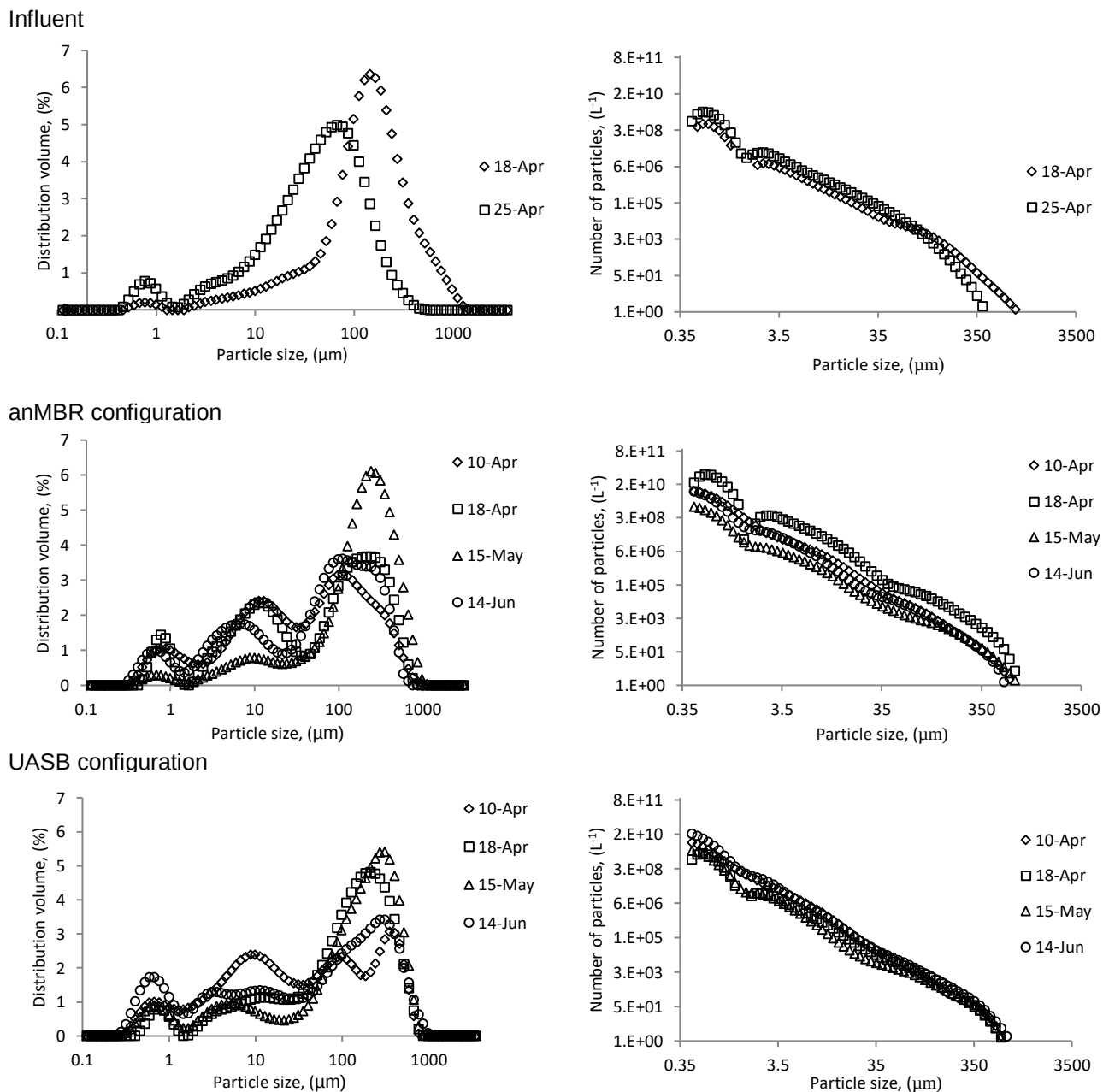


Figure 2-3 Representative PSD curves of the influent and effluent of both UASB reactors during the experimental period. On the left there is the volume distribution, while on the right the correspondent number of particles.

Converting to an equivalent number distribution revealed very high numbers of particles below 1 μm in the influent and both effluents with maximum levels between 2×10^9 and 4×10^{10} particles per L^{-1} (Figure 2-3). The highest number of particles was observed in the anMBR configuration congruent with the membrane retaining colloidal material whilst the equivalent material would be bled off and so would not accumulate as easily in the UASB configuration. The initial month of monitoring was used to average the number size distribution to investigate the overall grade efficiency of the sludge blanket (Figure 2-4). Overall positive removal was observed with both configurations for particle sizes above 50 μm and with that extended to above 10 μm in the case of the UASB configuration. In fact, the removal across the sludge blanket in the UASB configuration was better than that for the anMBR configuration with the latter producing substantial number of particles in the sub 10 μm sizes compared to the influent.

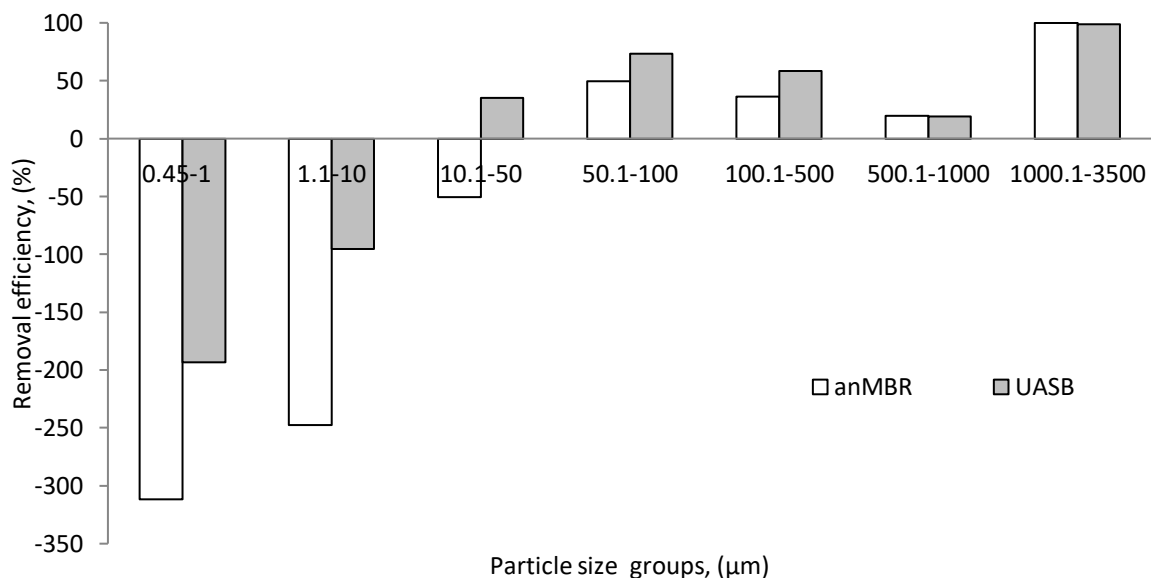


Figure 2-4 Grade efficiency for different size bands operated by the anMBR and UASB configurations.

Comparison of the generated PSDs from the two configurations across the sampling trial revealed a high degree of variation (Figure 2-5). To illustrate, the

d_{50} varied between 3 and 226 μm for the UASB configuration and between 14 and 227 μm for the anMBR configuration. In addition to differences in size, the particles exiting the UASB reactor appear to have different structural character based on the used configuration (Figure 2-6). In particular, the particles related to the anMBR configuration exhibited a larger fractal dimension than those from the UASB configuration. The difference reflects a compaction of the structure in the case of the anMBR consistent with the impact of elevated shear breaking off the extremities of the dendritic structures (Jarvis *et al.*, 2005). However, the fractal dimension was relatively low throughout with a maximum value of 1.8 indicating that the structures were linear and open (Kusnierz and Wiercik, 2016). The same change in temperature of wastewater was observed for both reactors during the experimental period (Figure 2-5). Temperature varied from a minimum of 13 $^{\circ}\text{C}$ in the early stage of the experiment to a maximum of 19 $^{\circ}\text{C}$ at the end of the trial. Temperature has been reported to influence particle characteristics in UASB effluents due to links to various removal mechanisms (Mahmoud *et al.*, 2003). Whilst characterized by high variations, a general trend of increasing d_{50} was observed with increasing temperatures for the anMBR configuration while the opposite was observed for the UASB configuration (Figure 2-5).

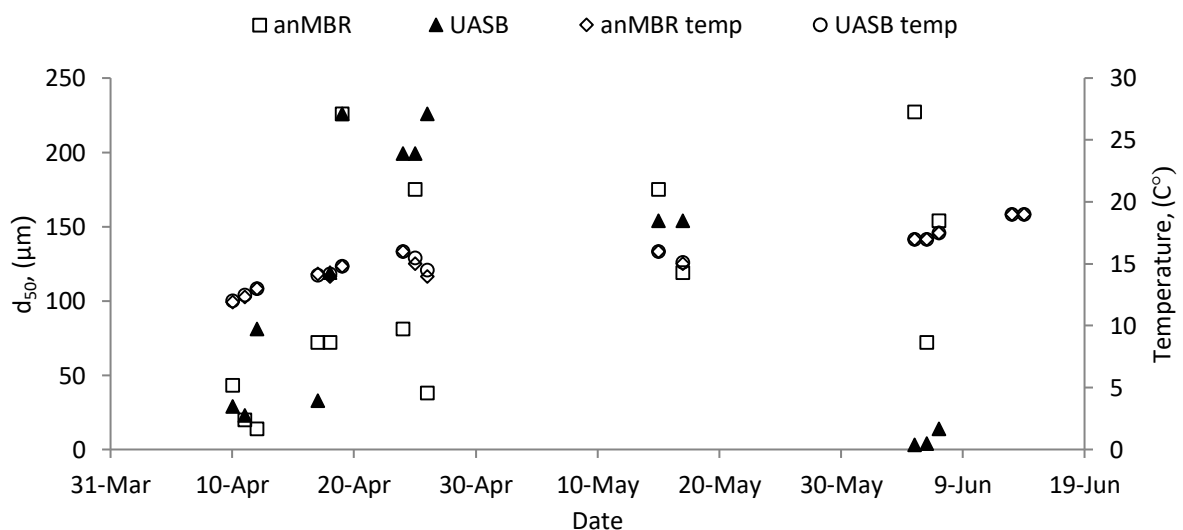


Figure 2-5 d_{50} and temperature of the effluent from the UASB and anMBR configuration across the sampling trial period

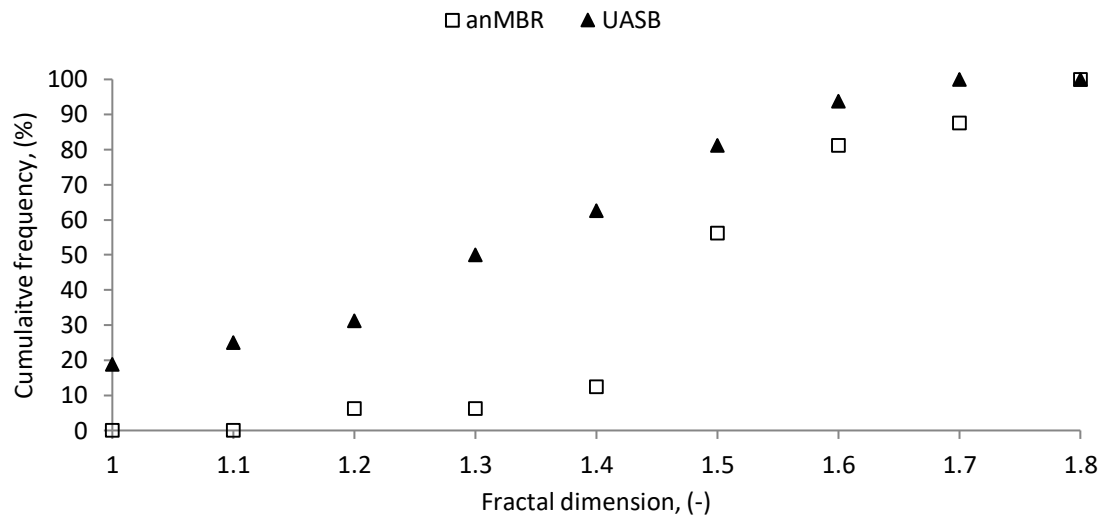


Figure 2-6 Fractal dimension of UASB effluent expressed as a cumulative frequency curve and relative to a period of two months

2.4.2 Effect of the sludge blanket on the PSD

To further explore the importance of the sludge blanket on the management of particles across the UASB reactors, the data was compared to the measured sludge blanket height during the trial using the base of the reactor as a reference point (Figure 2-7, 2-8 and 2-9). Overall the suspended solids removal efficiency increased progressively with an increase in sludge blanket height from a low of 7 % at the minimum sludge blanket height of 60 cm to 74 % when the blanket height was 85 cm (Figure 2-9). Greater variation in blanket height was observed for the anMBR configuration but across periods of similar blanket height the removal efficiency was similar for both configurations indicating no impact on the effectiveness of the blanket. The development of the blanket height followed the expected trend with solids removal rate showing that the height was controlled by the amount of incoming material. For instance, solids removal rate of $0.288 \text{ kgTSS m}^{-3} \text{ d}^{-1}$ and $0.313 \text{ kgTSS m}^{-3} \text{ d}^{-1}$ resulted in sludge blanket growth rate of 32 % and 44 % in anMBR and UASB configurations, respectively (Figure 2-8). This has operational significance as it also determines the frequency that the blanket needs desludging. In the current set up a maximum blanket height of 90 cm is used before desludging such that the mass

and size of the incoming solids controls the frequency of desludge. For instance, previous experience with these pilots indicated that treatment of settled sewage enable desludging roughly every 100 days whereas treatment of crude requires a much shorter period of desludging of 14 days (Martin Garcia *et al.*, 2013). When operating on short cycle times, the structure of the blanket is unlikely to form so stably and so may impact on the removal efficiency that can be achieved, indicating that treating lower solids feed waters is preferred with this configuration of anaerobic reactor.

The impact of the changing blanket height was also observed in relation to the residual particle sizes that exited the reactors with an overall increase in d_{50} as the blanket height increased (Figure 2-7). Comparison of the two configurations revealed a more consistent trend in the case of the UASB configuration were the d_{50} increased from 29 μm at a blanket height of 74 cm to 227 μm at a blanket height of 86 cm. In comparison, for the anMBR configuration, the d_{50} increased from 14 μm at a blanket height of 60 cm to 226 μm at a blanket height of 88 cm.

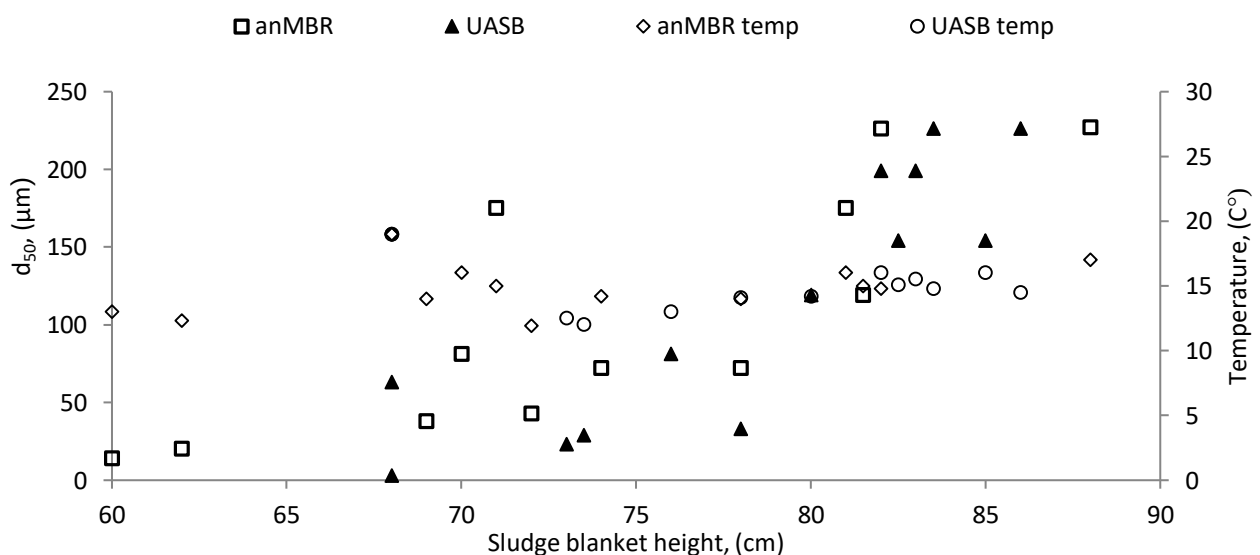


Figure 2-7 d_{50} and temperature of the effluents with the raising of the sludge blanket within UASB reactors.

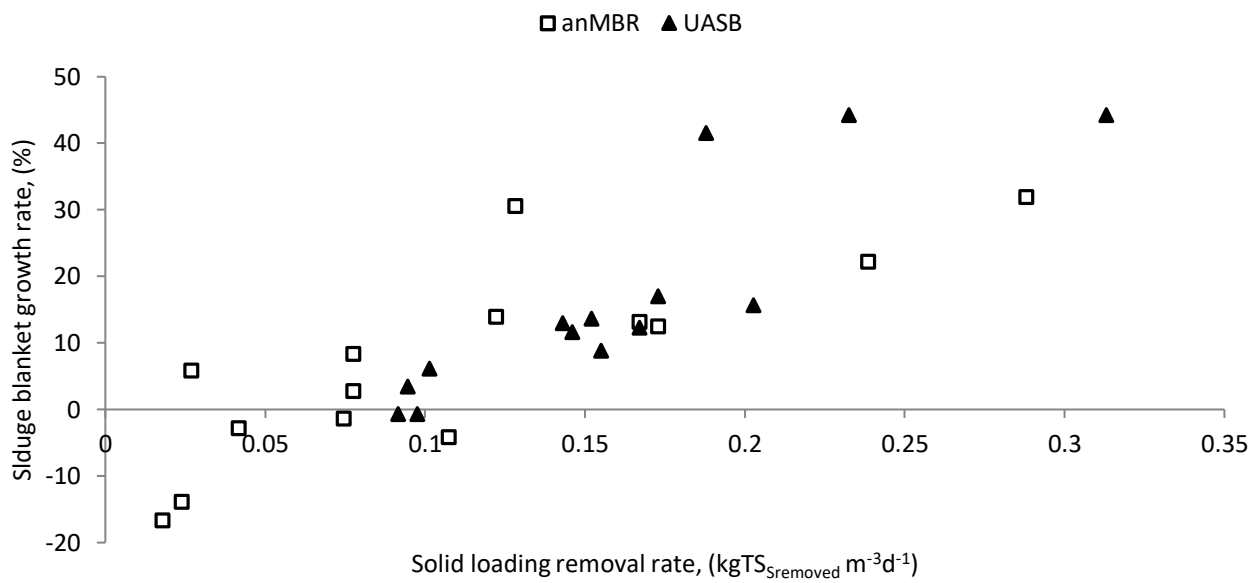


Figure 2-8 Sludge blanket growth rate as a function of the solid removal rate

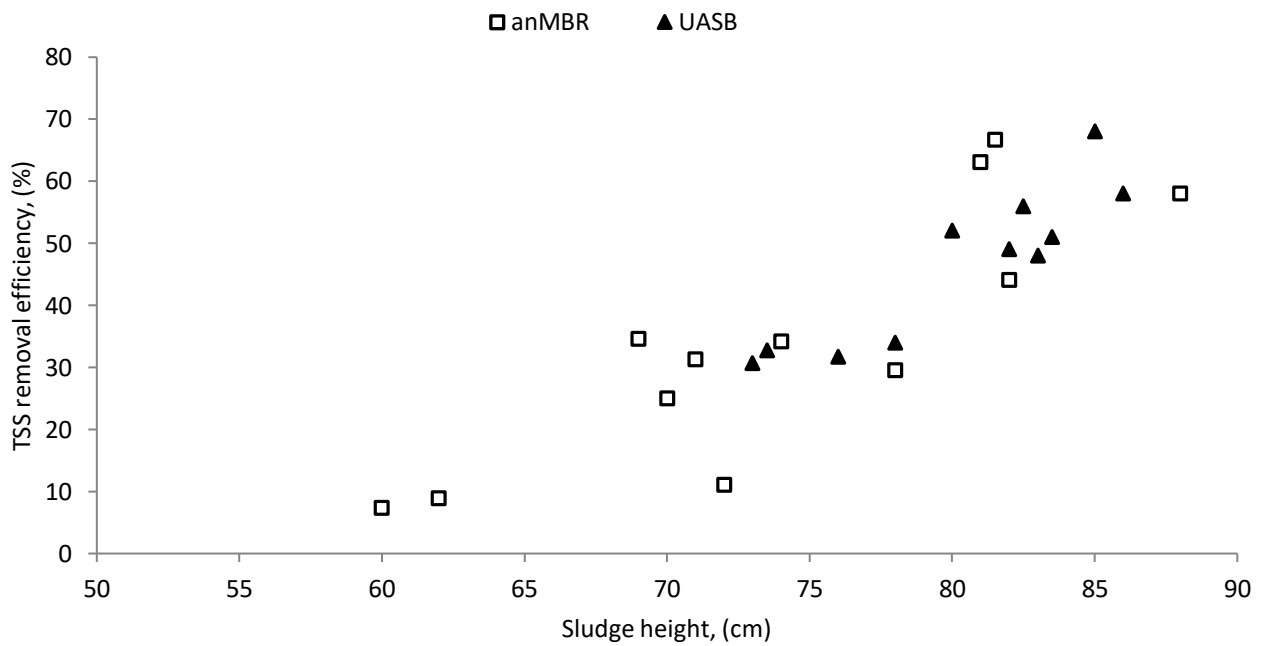


Figure 2-9 TSS removal efficiencies of UASB reactors as a function of the sludge blanket height

2.5 Discussion

This study demonstrated that the particle characterisation of anaerobic effluents from UASB reactors was dependent on the operational conditions of the anaerobic reactors. The stability of the biological processes within the reactors was confirmed by the alkalinity increase and by the neutral pH (Metcalf & Eddy, 2014). In these conditions a sludge blanket growth was observed in both reactors which can be attributed to solids entrapment into the sludge blanket (Figure 2-8). Entrapment, adsorption and settling are the first causes of physical solids removal, especially at ambient temperature (Mahmoud *et al.*, 2003; Lettinga *et al.*, 1980). Therefore, high solids entrapment ensured higher sludge blanket growth rate which in turn generated higher solids removal, peaking at 74% (Figure 2-9). This aspect was also reported in other studies but with particular attention on the temperature and up-flow velocity effect (Wang *et al.*, 2019; Seghezzo *et al.*, 2014). For example Wang *et al.* (2019) observed that higher temperature ($>10\text{ }^{\circ}\text{C}$) improved particle settling characteristics, ensuring stable sludge blanket with better solids removal efficiency ($64 \pm 12\%$) compared to sub $10\text{ }^{\circ}\text{C}$ ($42 \pm 13\%$). In addition, a decrease of the solids removal efficiency was observed when the upflow velocity was increased from 0.4 to 0.8 m h^{-1} . Seghezzo (2014) also observed a better solids removal at lower v_{up} (0.28 m h^{-1}) that improved ($> 90\%$) at higher sludge blanket height (1.84 m) compared to higher than 80% when the v_{up} was 0.85 m h^{-1} . Ozgun *et al.* (2015) reported higher turbidity and smaller d_{50} during the operation of an anMBR at temperature of $15\text{ }^{\circ}\text{C}$ if compared to those of $25\text{ }^{\circ}\text{C}$ similarly to the results observed in the present study for the same reactor configuration. This suggests that the presence of a physical barrier in the recirculation of the UASB effluent recycles into the system a larger amount of solids than in the UASB configuration, due to the high solids rejection capacity of the membrane. These solids are only partially degraded at lower temperatures due to a slower biological activity and a higher viscosity that reduces the possibility of small particles removal by entrapment or adsorption (Mahmoud *et al.*, 2003).

Overall assessment of the sludge blanket revealed it is capable of effectively removing particles over $10\text{ }\mu\text{m}$ in size, consistent with traditional depth filtration

processes where it is dependent on the particle size of the media and on the particles to be removed (Landa *et al.*, 1997; Jackson, 1980). Moreover, as with other depth filtration processes, the effectiveness of the blanket was correlated to the blanket height with an increased solids removal and increase in residual particle size observed as the blanket height increased (Figure 2-7). The observed changes reflect the impact of aggregation processes within the UASB reactor and more specifically below and within the blanket. For instance, it has been suggested that physical aggregation caused by the self-immobilization of bacteria could be the main mechanism of this process (Liu *et al.*, 2003), since the particles are forced to come into contact due to the upflow mode under which UASB reactors operated (McHugh *et al.*, 2003). Accordingly, higher sludge blanket heights offer longer trajectory path lengths for the flow increasing the cases of collision and aggregation such that blanket height has traditionally be defined as the critical parameter in regards to the flocculation efficiency within blanket clarifiers (Ives, 1968). Similar findings have also been reported by Yan *et al.* (1997) in a study investigating the characterisation of the granulation process of a UASB reactor. He observed that a gradual increase of the sludge loading rate during the start-up improved particles aggregation leading to a higher sludge blanket height.

Comparison of the two configurations revealed differences in the particle sizes, structures and removal efficiency across the UASB reactors and clearly indicates a measurable impact of utilising a membrane within the recycle loop. In fact the implementation of the membrane in the recycle line deteriorated the particulate matter removal efficiencies of the anaerobic reactor (Table 2-1). Overall this suggests that if alternative separators are to be used then it may be preferable to configure them to treat the outlet from the overall system and not connect them into the recycle line.

2.6 Conclusion

This study demonstrated the impact of a UASB and anMBR configuration on the solids loading profile and particle size distribution exiting the UASB reactor stage. According to the results the following conclusion can be drawn:

- The inclusion of the membrane within the recycle line impacts the particulate matter removal efficiencies across the UASB reactor resulting in a reduction of SS removal between 10-14 %.
- Particles related to the two effluents showed also different size and structure. In fact the d_{50} trend revealed difference in particle size variation related mainly to the sludge blanket growth. Whereas the fractal dimension exposed that particles from anMBR configuration were characterised by more compacted structure.
- The management of the solids across the UASB reactor is controlled by the characteristics of the sludge blanket layer.
- These results demonstrated that the installation of a separation system in the recycle line exacerbated the removal efficiency of the UASB reactor. Therefore alternative low costs technologies should consider to be implemented externally to the recycle line to treat the UASB effluent.

2.7 References

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3 Understanding what influences the effectiveness of a cloth filtration system treating effluent from an UASB reactor

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3.1 Abstract

This study presented the feasibility of a cloth filtration system for solids removal from UASB reactor treating municipal wastewater. The aim is to understand the influence of different operational conditions on the performance of a bench scale cloth filter, so that to consider this technology as a valid alternative to the conventional membranes. Therefore woven and non-woven meshes with different pore sizes were tested at different Cross Flow Velocities (CFV), TransMembrane Pressures (TMP) and influent solids concentration. The resulted demonstrated the overall efficacy of non-woven cloths, especially with smaller pore sizes operated under low or no cross flow. Grade efficiency analysis revealed the meshes of all pore sizes provided an effective cut of all particles bigger than 50 μm after just 30 minutes of filtration, with removal up to 50 % for smaller particles (below 50 μm).

3.2 Introduction

Membrane Bioreactors (MBR) are an established technology for municipal wastewater treatment that are able to deliver a high quality effluent by means of the combination of ultrafiltration or microfiltration coupled to a biological reactor. The membrane acts as a total barrier to suspended solids such that the Solids Retention Time (SRT) is controlled independently from the Hydraulic Retention Time (HRT) enabling operation at elevated biomass concentration, intensifying the process. In addition, the membrane uses pores that are small enough to clarify and to disinfect the effluent without the need for further downstream treatment (Judd, 2011; Le-Clech, 2010).

In more recent times, anaerobic reactors have been coupled with membranes to form anaerobic MBRs (anMBR) which are able to produce high effluent quality

whilst also generating energy through the production of biogas (Skouteris *et al.*, 2012). Comparison of systems currently under investigation have shown that switching to anMBR based sewage treatment enables net energy neutral operation (Jefferson *et al.*, 2018). Sustainable operation of the membrane in anMBRs is achieved at lower fluxes than typical used in aerobic MBRs such that the membrane becomes the dominant capital and operating expenditure (CAPEX and OPEX) associated with the flowsheet. Therefore alternative separation processes to membranes are being considered such as dynamic membrane, flotation, depth filter and microscreens. In such cases solids removal efficiencies are expected to be lower than those of the membrane but may be sufficient for many required applications. In addition, the alternative technologies are likely to have substantially lower capital and operating costs as well as easier maintenance improving the business case for their implementation (Jefferson *et al.*, 2018). To date there is a genuine lack of reported studies considering alternative separations with rare examples of Dissolved Air Flotation (DAF) fed with pre-coagulated UASB effluent (Penetra *et al.*, 1999), and a sand filter used as a post treatment for the UASB reactor (Tyagi *et al.*, 2009). In both cases a 90 % of TSS removal is reported albeit utilising coagulant in the case of the DAF process.

The most commonly considered separator linked to anaerobic processes is the use of fabric filters where a low cost porous support is used to house the rapid build-up of a cake layer which then delivers the separation performance, collectively known under the name of dynamic membranes. Research into dynamic membrane (DM) systems has been conducted since 1994, mostly applied to the activated sludge process (Ersahin *et al.*, 2012) or aerobic MBRs (Gander *et al.*, 2000). In more recent times this has extended to anaerobic reactors, predominately at lab or small pilot scale (Table 3-1). The underlying support material can be either woven or non-woven filter cloths (Hu *et al.*, 2018). Woven filter cloths are produced through the specific bind of yarns from a range of materials such as polypropylene, polyamide and polyester and leads to the formation of reasonably well defined pore sizes. In comparison, non-woven filter cloths exist as a random assemblage of physically intertwined yarns

made with synthetic materials such as nylon and polyester with much less well defined pore sizes (Svarovsky, 1990). In the majority of cases, the membrane material is loaded with the full biomass audit to enable rapid cake build up with only two studies to date placing the dynamic membrane systems above the main biomass section within an UASB (Quek *et al.*, 2017; Zhang *et al.*, 2010). However, no reported study has looked at the use of cloth filtration as a means of post UASB filtration where only the effluent particles are filtered and such replaces the use of a membrane as such in an UASB-UF configuration of an anMBR (Wang *et al.*, 2019). In such cases, the application is similar to that of tertiary treatment where surface filtration technology is commonly applied. For instance, operation of a Discfilter as tertiary treatment process was shown to ensure an effluent solids concentration lower than 5 mg L⁻¹ whilst operating with an energy demand of 0.034 kWh m⁻³ (Kangseep, 2016). Similarly, recent reports on cloth filters have indicated equivalent effluents solids but even lower energy demands (Cooper-Smith, 2018). Accordingly, it is suggested that cloth filter technology is a viable alternative to membrane for use in an UASB configured anMBR. To test this, two different types of cloth with three different pores size have been tested under different operational conditions when fed with effluent from an UASB treated settled wastewater under ambient conditions.

3.3 Materials and Methods

3.3.1 Experimental setup

Effluent was collected from a pilot scale UASB reactor treating settled municipal sewage under ambient conditions (Chapter 2) and stored in a 110 L reservoir overnight to be used the next day. In this way, the accumulated volume could guarantee the required TMP throughout the tests. Before starting the filtration tests, the effluent in the reservoir was mixed with a stirrer at 50 rpm for 10 minutes to homogenise the sample (Figure 3-1). The stirrer was kept at the same velocity throughout the trials. The UASB reactor had a total volume of 70 L, inoculating with 15 L of granular and flocculent sludge sourced from an UASB treating paper and pulp mill industry effluent. The HRT and up-flow velocity (v_{up})

was fixed at 8 h and 0.4 m h^{-1} , respectively. In these conditions, the effluent of the UASB reactor was characterised by a TSS concentration of $40 \pm 13 \text{ mg L}^{-1}$, and by a COD concentration of $98 \pm 24 \text{ mg L}^{-1}$.

The water was gravity fed to the dynamic membrane module test cell which tested membrane with an area of 30 cm^2 (Figure 3-2). The dimensions of the cell were $29 \times 6 \text{ cm}$ into which six different types of cloth material were tested (Figure 3-3). Different transmembrane pressures (TMPs) and cross flow velocities (CFVs) were used in this study. CFV was maintained by the use of a peristaltic pump. TMPs were maintained by virtue of a pseudo-constant vertical distance between water level in the feed reservoir and the membrane modules. This was obtained by either the addition of UASB effluent in the reservoir when the water level decreased up to 5 % (especially at the beginning when the permeate fluxes were very high) or by lowering the vertical height of the membrane modules. The TMP varied between three levels: 5 cm (0.5 kPa), 20 cm (2 kPa) and 40 cm (4 kPa). Four CFVs were tested: 0.01 m s^{-1} , 0.1 m s^{-1} and 0.6 m s^{-1} as well as 0 m s^{-1} , i.e. dead end, by application of a peristaltic pump (520Du, Watson Marlow, Falmouth, UK) (Figure 3-1). To further verify the impact of pore size, a series of trials were conducted where the different pore sizes were tested in parallel, eliminating impacts due to a changing characteristic of the feed water, which was observed to change daily. The permeate from the membrane modules was collected into cylinders for the flux measurement. Sample for TSS, COD, turbidity and PSD analysis were collected every 30 minutes during the filtration process and subsequently analysed.

In particular, the results of the PSD were expressed in the form of a volume distribution (%), which was converted in number of particles to calculate the efficiency removal of the following classes of particles: a) $0.45\text{--}1 \text{ }\mu\text{m}$, b) $1.1\text{--}10 \text{ }\mu\text{m}$, c) $10.1\text{--}50 \text{ }\mu\text{m}$, d) $50.1\text{--}100 \text{ }\mu\text{m}$, e) $100.1\text{--}500 \text{ }\mu\text{m}$ and f) $500.1\text{--}1000 \text{ }\mu\text{m}$, g) $1000.1\text{--}3500 \text{ }\mu\text{m}$.

3.3.2 Analytical methods

Total Suspended Solids (TSS) was measured according to standards methods (APHA, 2005). Chemical Oxygen Demand (COD) was analysed with Merck test

kits (Merck KGaD, Darmstadt, Germany). Turbidity was measured with the HACH 2100Q portable turbidimeter (HACH Company, Loveland, Colorado, US). Particle Size Distribution (PSD) was analysed with Mastersizer 2000/3000 (Malvern Instrument Ltd, Malvern, UK). Flux was calculated measuring the flow through the cloth by means of graduated cylinders.

Table 3-1 AnDMBRs performance for waste(water) treatment

substrate	support material	pore size μm	filtration area m^2	TMP kPa	CFV m s^{-1}	flux LMH	TSS removal %	Ref
synthetic municipal wastewater	MF membrane PTFE	1	0.09 to 0.12	7 to 55	0.1 to 0.2	5 to 8	n.a.	(Ho <i>et al.</i> , 2007)
synthetic municipal wastewater	Non-woven PET mesh	n.a 20	0.0188 0.0292	n.a n.a	n.a	> 3 0.5-3	>99 >99	(Jeison <i>et al.</i> , 2008)
biodegradable municipal waste	nylon woven mesh	30, 100, 140	n.a.	n.a.	n.a.	44	> 70	(Walker <i>et al.</i> , 2009)
municipal wastewater	PET non-woven fabric	0.64	0.98	>30	n.a	5	n.a	(An <i>et al.</i> , 2009)
municipal wastewater	Dacron mesh	61	n.a.	> 25	n.a	65	>94	(Zhang <i>et al.</i> , 2010)
municipal wastewater	Dacron mesh	61	0.081	>35	n.a.	60	64	(Ma <i>et al.</i> , 2013)

synthetic wastewater	PP woven fabric	10	0.018	36 to 41	*17, 35, 52	2.6	>99	(Ersahin <i>et al.</i> , 2013)
synthetic wastewater	Nylon woven mesh	200	0.004	>20	0.0003 to 0.1	1 to 7.2	99	(Alibardi <i>et al.</i> , 2014)
synthetic wastewater	Nylon woven mesh	200	0.004	5-10	0.03	15-20	>95	(Alibardi <i>et al.</i> , 2016)
waste sludge	Tubular membrane	100	0.0314	1.3 5-6	n.a.	0-20 3.5, 2.3, 1.2	99	(Liu <i>et al.</i> , 2016)
anaerobic sludge	Nylon mesh wound	10, 52, 85, 135, 200	0.00583	5, 10, 18	0.00833	329-9880 to 33-990	>99	(Saleem <i>et al.</i> , 2017)
municipal wastewater	Mesh 500 Mesh 300	28 46	0.005	5 to 40	n.a.	100, 60, 30	86	(Quek <i>et al.</i> , 2017)
synthetic wastewater	PP woven fabric	10	0.014	38, 68	*35	2.2	n.a.	(Ersahin <i>et al.</i> , 2017)

*Gas Sparging Velocity (GSV)

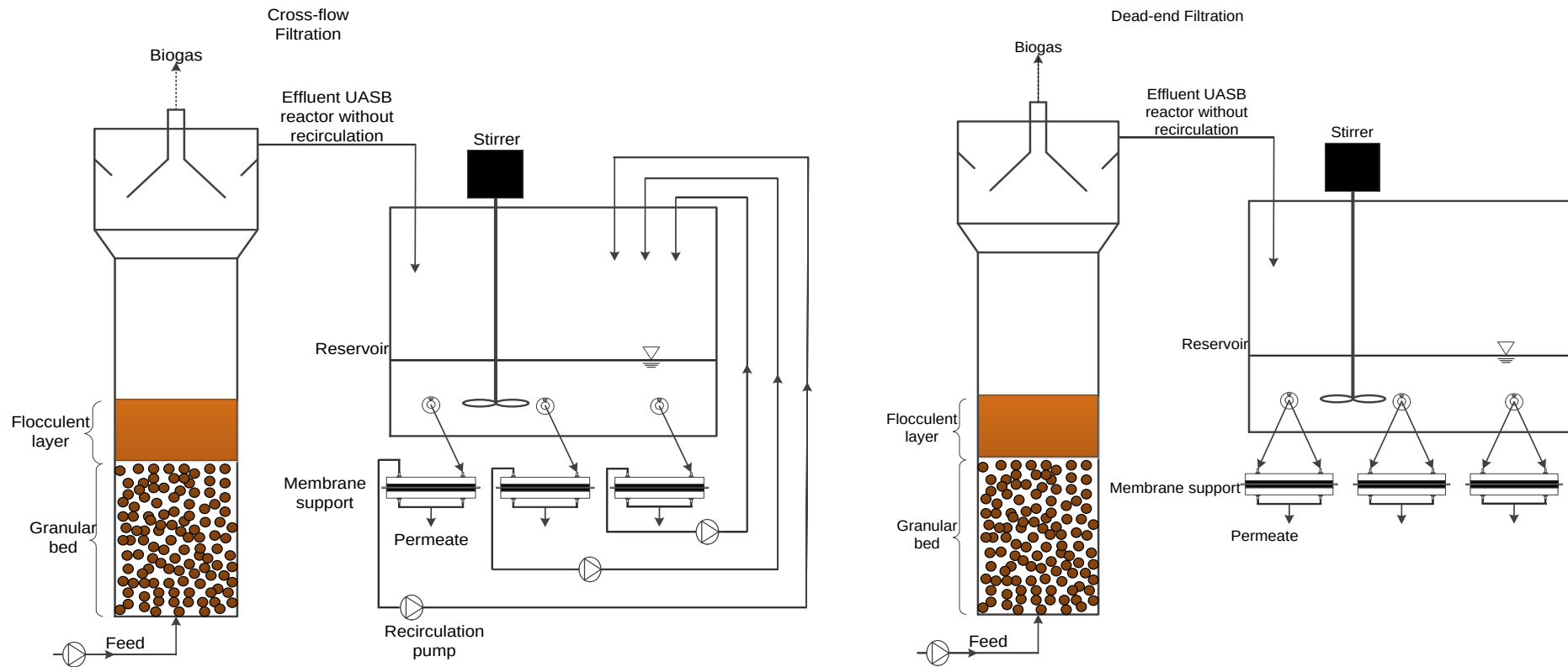


Figure 3-1 Schematic diagram of the filtration process in cross-flow and dead-end mode

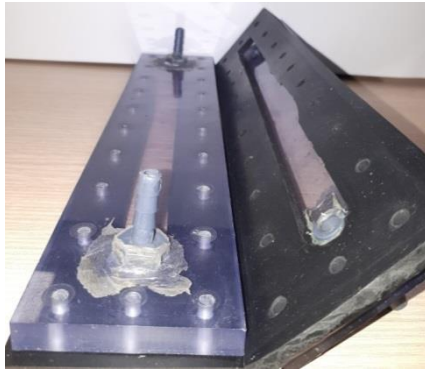
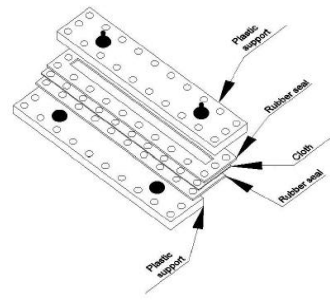


Figure 3-2 Membrane test cell: on the right one of the real module used for this study, on the left it is illustrated the position of each part of the cell test

Monofilament woven mesh made of polyamide

Felt bag non-woven yarns made of polypropylene

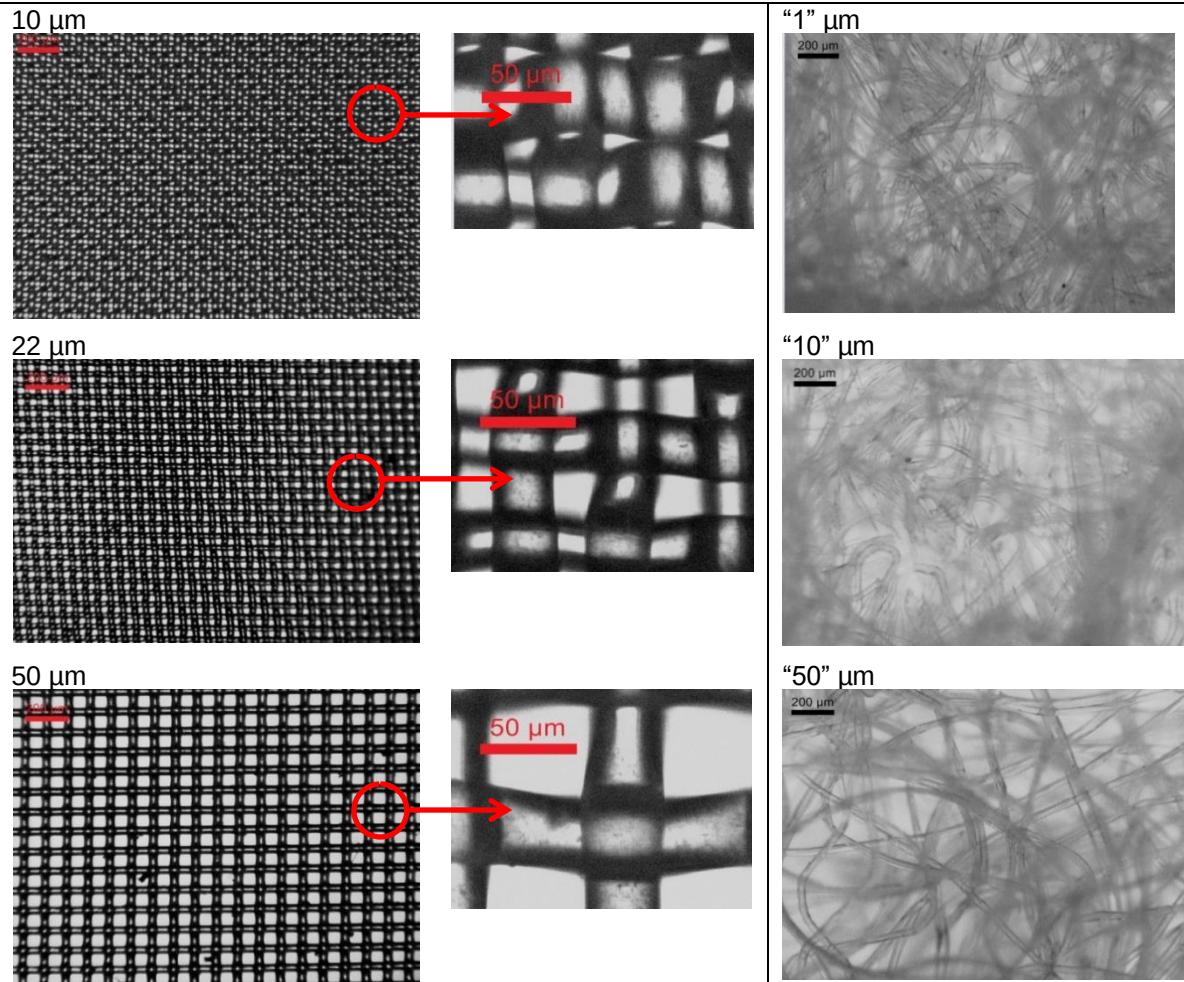


Figure 3-3 Microscope photographs of woven and non-woven mesh at different pore size

3.4 Results

3.4.1 *Effect of the CFV, TMP and pore size on the cloth filtration*

An impact of the mesh pore size was seen in terms of both the initial flux obtained and the relative rate of decrease in flux observed during the 180 minutes trial (Figure 3-4). To illustrate, in the case of the woven mesh operated at a CFV of 0.6 m s^{-1} and a TMP of 2 kPa, the initial flux was 12000 and 10000 LMH for 50 μm and 22 μm pore sized meshes, respectively, and 5000 LMH for the case of the 10 μm mesh (Figure 3-4). The flux decreased to 1520, 88 and 36 LMH for the 50, 22 and 10 μm pore sized meshes respectively after 180 minutes of operation. This represents a flux decline of 87.4%, 99.2% and 99.3% respectively and indicates a similarity between the 22 and 10 μm pore sized meshes.

This flux trend was observed in all tests and for clarity is described in terms of the ratio of the initial (J_0) (after 3 minutes) compared to post 30 minutes (J_{30}) and post 180 minutes (J_{180}) of operation to assess the stability of flux delivery (Figure 3-5). Comparison of the trials revealed that the highest flux ratios occurred at a TMP of 2 kPa. Consistency was also observed for the non-woven meshes in that the highest final flux ratios all occurred in dead end mode with a TMP of 2 kPa (Figure 3-5). In contrast, no observable trend was observed in the case of the woven cloths where the highest flux ratio occurred under a range of different conditions and pore sizes. For instance, the peak final flux ratio of 0.043 was observed for the 10 μm pore sized mesh under operating conditions of a TMP of 4 kPa and a CFV of 0.6 ms^{-1} . Whereas for the 22 μm and the 50 μm the highest final flux ratio occurred at a TMP of 2 kPa and CFV of 0 and 0.01 ms^{-1} , respectively. Further, increasing the CFV did not appear to have observable benefit in relation to the flux ratio in the case of the smaller pore size meshes; specifically, the 1 and 10 μm non-woven mesh and the 10 μm woven mesh.

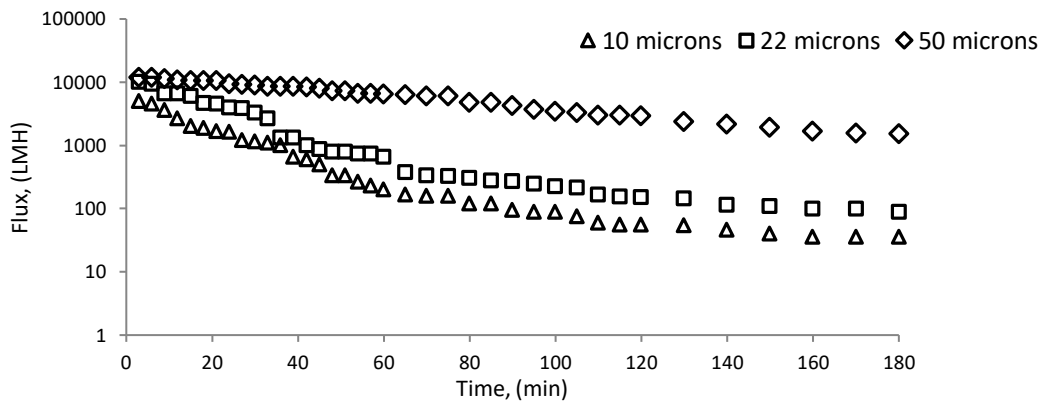


Figure 3-4 Flux variation over time for the cloth with woven fibres and pore size of 10, 22 and 50 μ m, operating at TMP of 2 kPa and CFV of 0.6 m s⁻¹

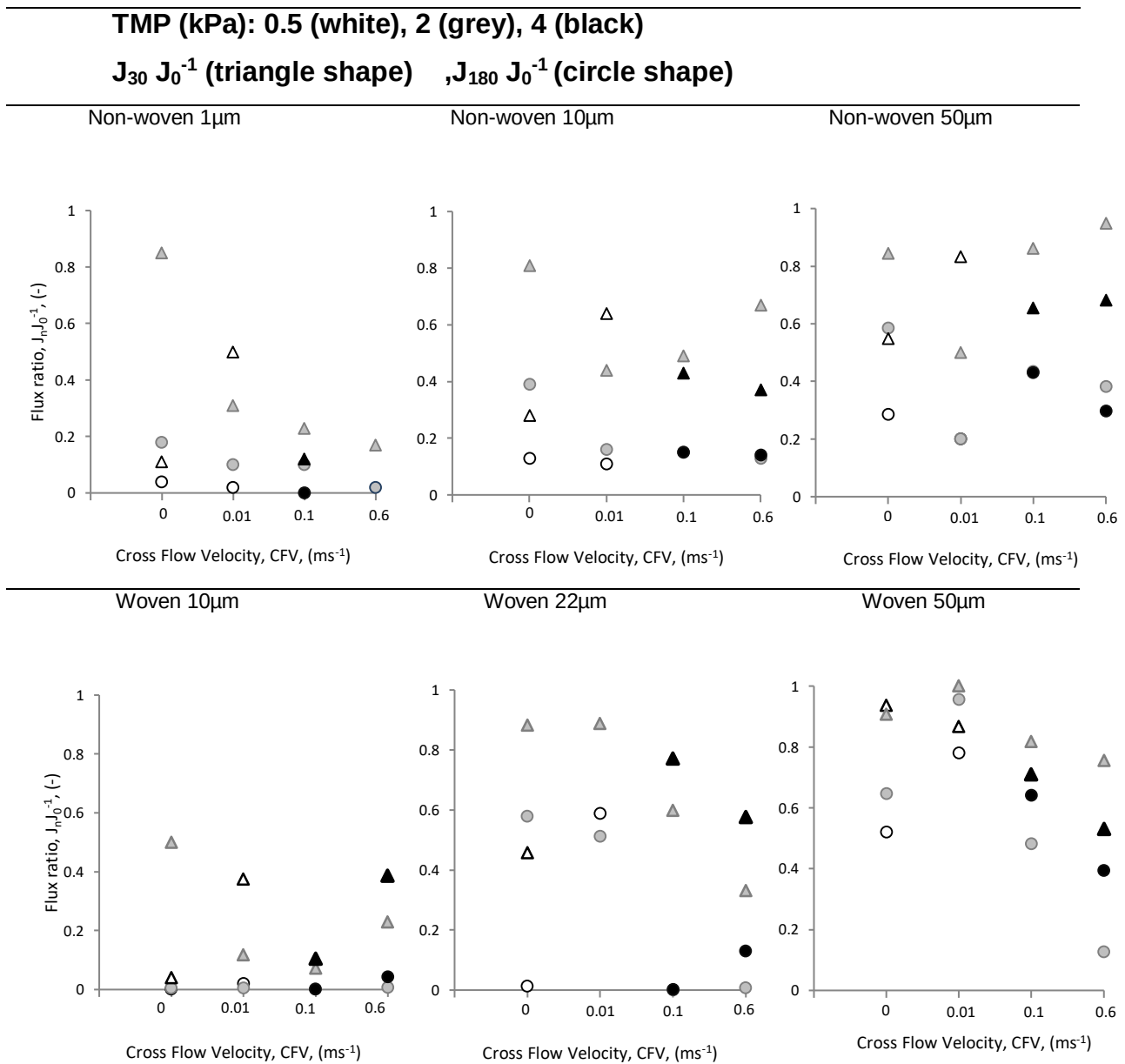


Figure 3-5 Variation of the ratios J_{30}/J_0 and J_{180}/J_0 as a function of CFV and TMP

3.4.2 Impact of TSS on cloth filtration performance

The influent TSS concentration impacted the flux profile across all meshes consistent with the obtained flux being controlled by the size and compaction of the formed dynamic layer. For instance, when the influent solids were 82 mg L^{-1} the final flux for non-woven meshes were 230 LMH, 1000 LMH and 1600 LMH for pore sizes of $1 \text{ }\mu\text{m}$, $10 \text{ }\mu\text{m}$, $50 \text{ }\mu\text{m}$ respectively. In contrast, when the influent TSS concentration was at its lowest value of 32 mg L^{-1} , the final fluxes were 2000 LMH, 5400 LMH, 8600 LMH for the same cloths, respectively. Less consistency was observed for the woven meshes where no observable impact was seen with regards to the impact of influent solids concentration on the flux ratio. In fact, in the case of the $10 \text{ }\mu\text{m}$ pore size the flux ratio varied between 0.015 and an influent solids concentration of 32 mg L^{-1} and 0.043 at an influent solid concentration of 86 mg L^{-1} (Figure 3-6a). In comparison, the flux ratio decreased from 0.39 to 0.11 over the same influent solid concentrations for the $10 \text{ }\mu\text{m}$ non-woven mesh (Figure 3-6a).

The impact of influent solid concentration was also seen with regards to the suspended solids removal efficiency (Figure 3-6b). Clearer trends were observed in the case of the non-woven mesh, congruent to the observation related to flux decline. For instance, the $1 \text{ }\mu\text{m}$, $10 \text{ }\mu\text{m}$ and $50 \text{ }\mu\text{m}$ pore size meshes removed 31 %, 19 % and 12 % of the suspended solids when the influent concentration was 32 mg L^{-1} compared to 66 %, 63 % and 34% when the influent solid concentration was 82 mg L^{-1} . In the case of woven meshes, the suspended solids removal efficiency increased as the influent solids concentration increased but then decreased when the highest concentration was applied (Figure 3-6b). However, with both types of meshes, the best removal was observed for the smallest pore size. For instance, in the case of the woven meshes, removal efficiencies of 25%, 13% and 6% were observed for pore sizes of $10 \text{ }\mu\text{m}$, $22 \text{ }\mu\text{m}$, $50 \text{ }\mu\text{m}$, respectively. In this case the peak removal efficiency of 47%, 41% and 31% were observed for the same pore sizes, significantly lower than those observed for the non-woven meshes. The solids removal efficiency of both cloths depended on the solids accumulation on the cloth surface. In fact, from the mass balance of trials conducted in dead-end

mode and a TMP of 0.5 kPa and 2 kPa, an increase in removal was observed with the increase of solids accumulation (Figure 3-7). For instance, at TMP of 2 kPa, both woven and non-woven meshes ensured higher solids accumulation with smaller pore size resulting in higher solids removal efficiencies but lower fluxes. A similar result was also observed at a TMP of 0.5 kPa for the woven meshes in which the highest solids accumulation occurred with the smallest pore size. For instance, at this TMP solids removal of 25 %, 13 % and 6 % were observed for solids accumulation of 720 mg TSS, 624 mg TSS and 317 mg TSS relative to pore size of 10 μm , 22 μm and 50 μm , respectively. In comparison, a more variable result was observed for the non-woven meshes at the same TMP. For instance, the mesh with pore size of 10 μm and 50 μm resulted in solids removal efficiencies of 55 % and 52 % associated with solids accumulation of 5304 mg TSS and 5069 mg TSS, respectively. In contrast, the mesh with pore size of 1 μm achieved a solids removal of 55 % with solids accumulation of 3060 mg TSS.

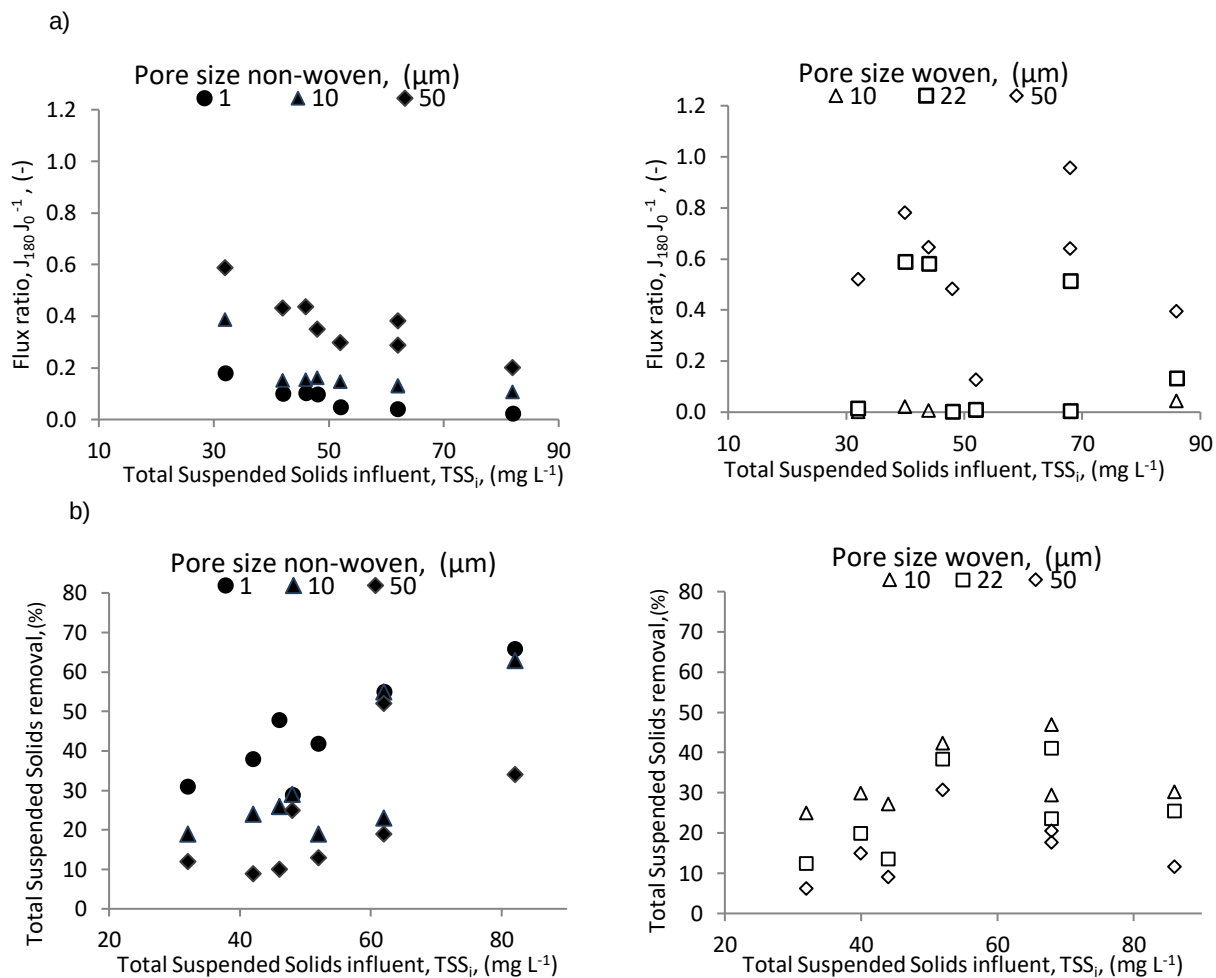


Figure 3-6 Impact of TSS on the cloth filtration performance in terms of flux ratio (a) and suspended solids removal efficiency (b).

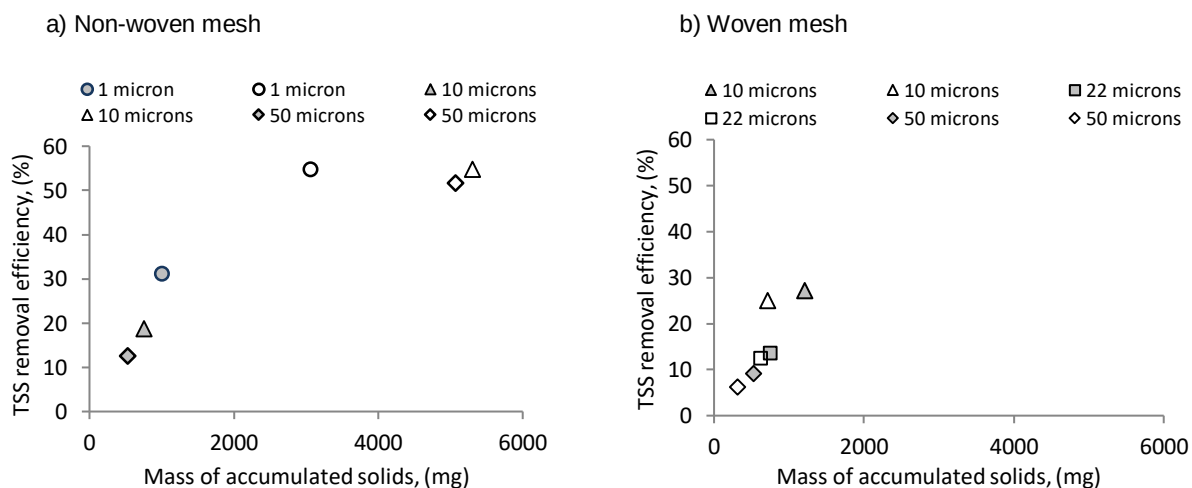


Figure 3-7 TSS removal as a function of mass accumulated onto the mesh for non-woven (a) and woven (b) materials at two different TMP of 0.5 kPa (white) and 2 kPa (grey) during dead end operation.

3.4.3 Performance across the operating cycle

The performance during the 180 minutes operating cycle was then monitored for the best performing systems, which includes the non-woven meshes able to guarantee fluxes above those ensured by conventional membranes (10-20 LMH) and the highest solids removal under dead end operation with pores sizes of 1 μm , 10 μm , 50 μm and a TMP of 0.5 kPa. In addition, the 1 μm non-woven mesh was also monitored for comparison at a CFV of 0.01 m s^{-1} and TMP 0.5 kPa. The only woven mesh tested was the 10 μm pore sized material operating at a CFV of 0.1 m s^{-1} and TMP of 2 kPa (Figure 3-8, 3-9, 3-10, 3-11).

The operating cycles demonstrated a similar relationship between flux decline and solids removal as observed above with the highest solids removal systems resulting in the lowest fluxes. For instance, the 1 μm non-woven mesh operating in dead mode and TMP of 0.5 kPa resulted in a flux of 1200 LMH and solids removal of 67%. Whereas, the equivalent cycle with the 50 μm non-woven mesh resulted in a flux of 3600 LMH and a solids removal of 37 %. Similar solids removal (58 %) was achieved by the cloth of 10 μm non-woven fibres at TMP of 0.5 kPa and in dead-mode after 180 minutes. In this case the flux was 1520 LMH.

The effluent solids concentration decreased progressively throughout the cycle and reached levels of 16, 20 and 30 mg L^{-1} from an influent of 48 mg L^{-1} for the non-woven meshes operated under dead end with pore sizes of 1, 10 and 50 μm respectively (Table 3-3). In the case of the 1 μm mesh, application of a CFV of 0.01 m s^{-1} did not impact the removal profile resulting in an effluent solid concentration of 21 mg L^{-1} (Table 3-3). The highest effluent solid concentration was observed for the woven mesh with a final concentration of 38 mg L^{-1} , albeit from an influent of 60 mg L^{-1} . The associated turbidity removal did not exceed 14% after 180 minutes. Similar result (12%) was achieved by the non-woven mesh of 50 μm . However, the trial with the non-woven mesh of 1 μm ensured the highest turbidity removal (26-27%) after 180 minutes (Table 3-3).

The equivalent effluent COD values for the dead end operation of the non-woven mesh were 27, 31 and 42 mg L^{-1} (Table 3-3) from an influent of 55 mg L^{-1} .

¹ for pore sizes of 1, 10 and 50 μm respectively. These correspond to removal efficiencies of 51%, 44% and 24% as the pore size increases (Figure 3-11). In all cases, the removal efficiency increased over time congruent with the build up of the dynamic layer and the improved solids removal. For instance, in the case of the 10 μm non-woven mesh operated under dead end, the COD removal efficiency increased from an initial value of 12.7 % after 30 minutes to 18 %, 25 %, 33 %, 36 % and 44 % after 60, 90, 120, 150 and 180 minutes of operation respectively. Poorer performance was observed with the 10 μm woven mesh where the respective COD removal efficiencies were 9 %, 15 %, 19 %, 23 %, 27 % and 30 %. Accordingly, the effluent contained a higher residual COD of 48 mg L^{-1} (Table 3-3).

Table 3-2 Selected conditions of the best performing systems

Symbol	Pore size (μm)	CFV (m s^{-1})	TMP (kPa)
◇	10 (woven mesh)	0.1	2
■	1	0	0.5
◆	10	0	0.5
●	50	0	0.5
△	1	0.01	0.5

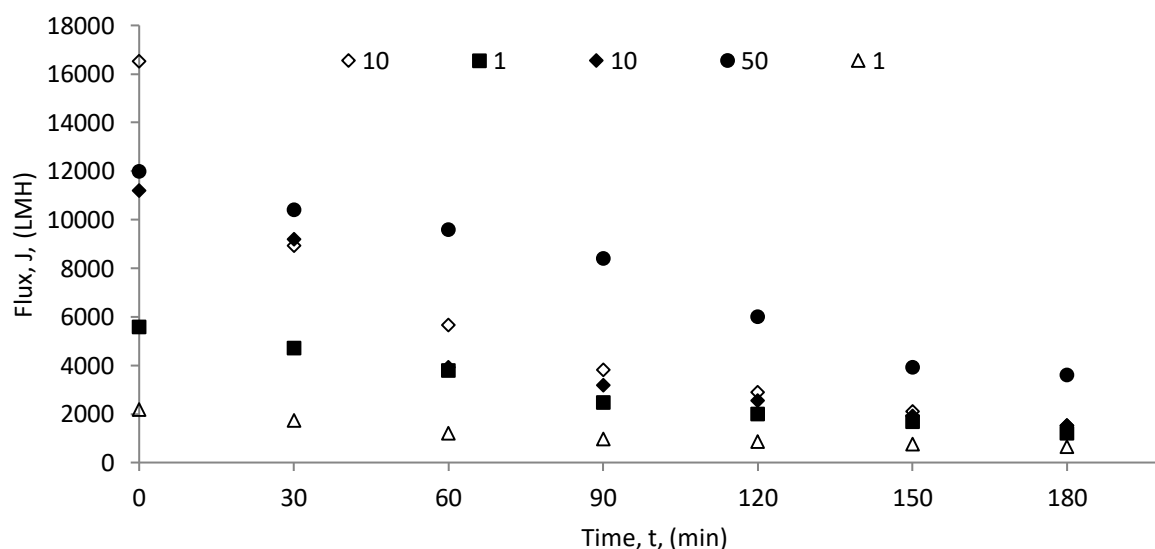


Figure 3-8 Flux variation over time of the best performing systems at the selected conditions

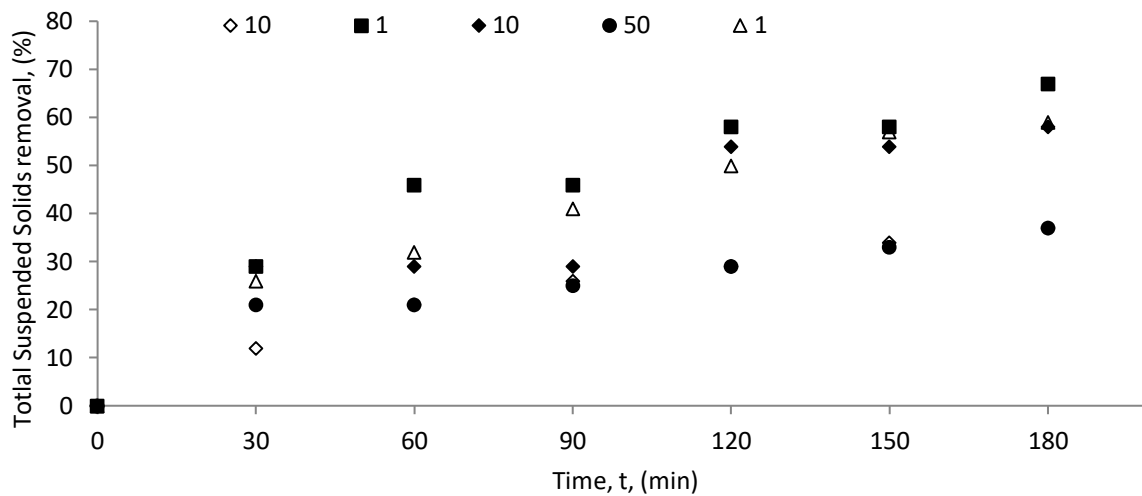


Figure 3-9 Solids removal efficiency over time of the best performing systems at the selected conditions

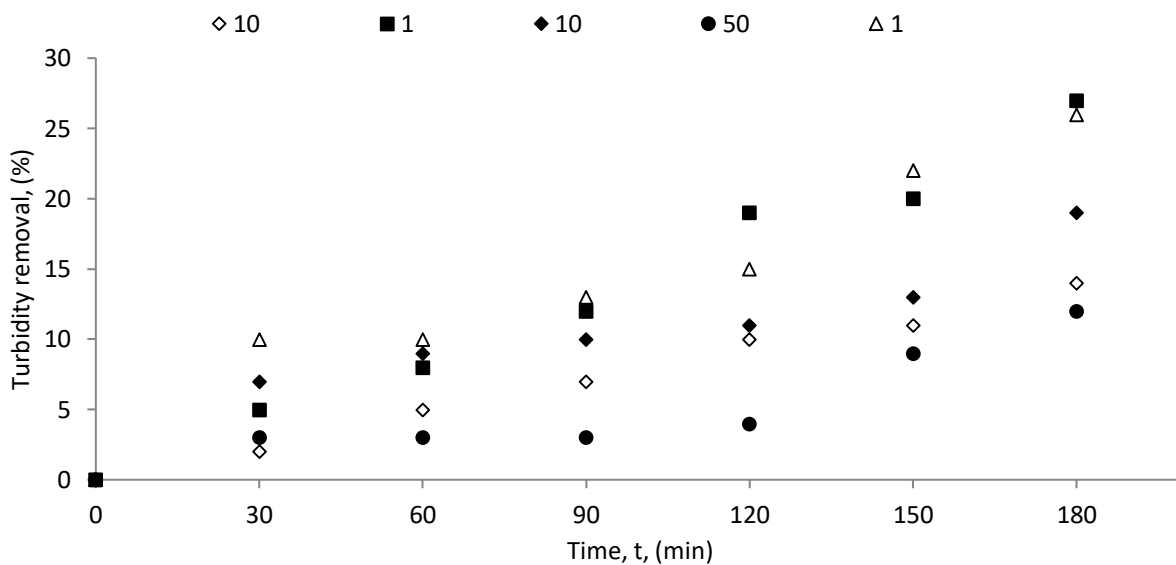


Figure 3-10 Turbidity removal efficiency over time of the best performing systems at the selected conditions

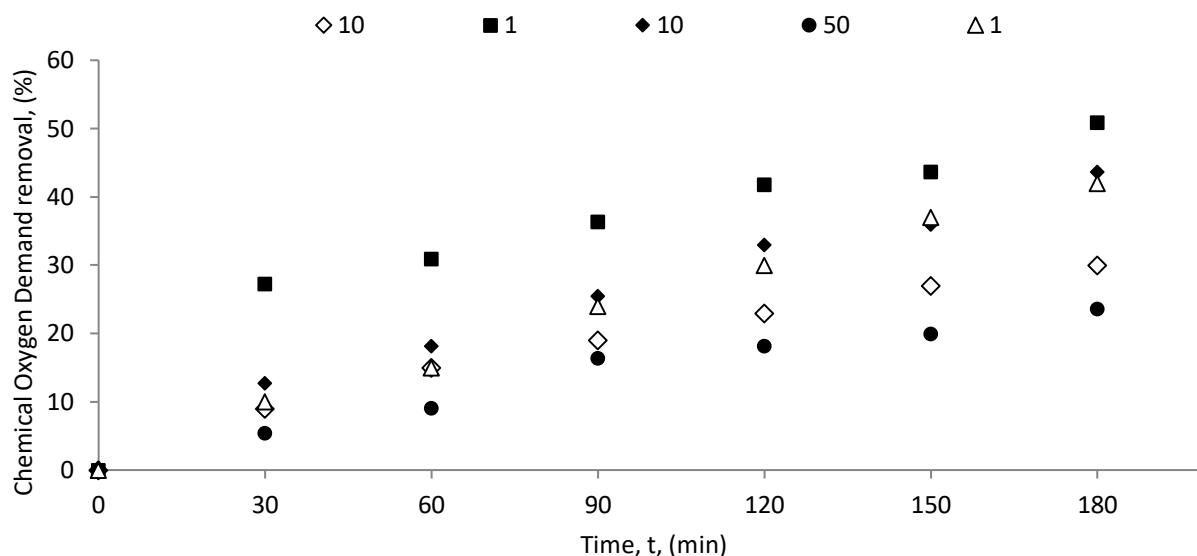


Figure 3-11 COD removal efficiency over time of the best performing systems at the selected conditions

Table 3-3 Characterisation of the permeate from the best performing systems

1µm (non-woven) CFV 0.01m s⁻¹ TMP 0.5 kPa

time	TSS	COD	turbidity	
			in	out
min	mg L ⁻¹	mg L ⁻¹	NTU	NTU
0	52	63	37.1	-
30	38	56	70	63
60	35	54	112	100
90	30	47	158	134
120	26	44	197	165
150	22	40	217	168
180	21	36	208	151

1µm (non-woven) CFV 0m s⁻¹ TMP 0.5 kPa

time	TSS	COD	turbidity	
			in	out
min	mg L ⁻¹	mg L ⁻¹	NTU	NTU
0	48	55	43.1	-
30	34	40	100	95
60	26	38	146	134
90	26	35	153	134
120	20	32	151	123
150	20	31	150	120
180	16	27	142	104

50µm (non-woven) CFV 0m s⁻¹ TMP 0.5 kPa

time	TSS	COD	turbidity	
			in	out
min	mg L ⁻¹	mg L ⁻¹	NTU	NTU
0	48	55	43.1	-
30	38	52	109	106
60	38	50	158	154
90	36	46	155	150
120	34	45	154	148
150	32	44	149	135
180	30	42	146	129

10µm (woven) CFV 0.1m s⁻¹ TMP 2 kPa

time	TSS	COD	turbidity	
			in	out
min	mg L ⁻¹	mg L ⁻¹	NTU	NTU
0	60	69	33.3	-
30	52	63	80	78
60	48	59	160	153
90	44	56	253	235
120	42	53	279	253
150	40	50	276	246
180	38	48	273	236

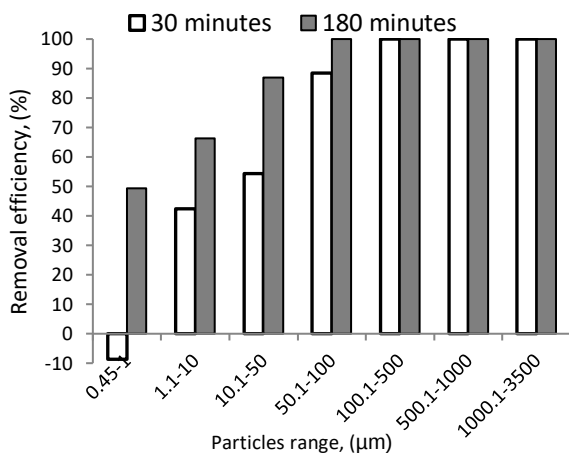
10µm (non-woven) CFV 0m s⁻¹ TMP 0.5 kPa

time	TSS	COD	turbidity	
			in	out
min	mg L ⁻¹	mg L ⁻¹	NTU	NTU
0	48	55	43.1	-
30	34	48	108	100
60	34	45	158	143
90	34	41	159	143
120	22	37	152	135
150	22	35	149	129
180	20	31	143	116

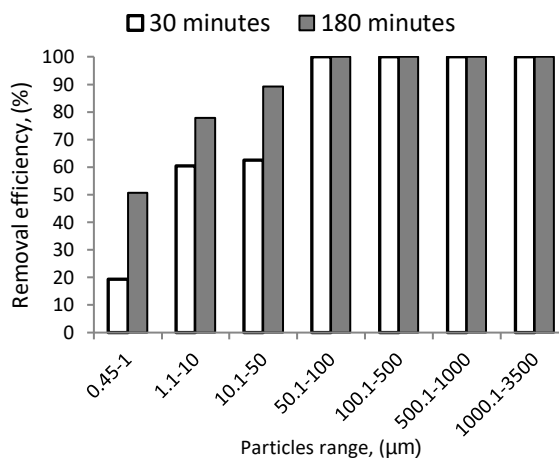
Grade efficiency curves

To further elucidate the performance of the different mesh materials, the particle size distribution was measured to establish the grade efficiency profiles of the different meshes (Figure 3-12). Analysis of the grade efficiency curves revealed 100 % cut of all particles above 50 μm when operating under dead end conditions for both the 1, 10 and 50 μm pore size meshes. Below this, the removal efficiency decreased along with the size band such that at 30 minutes of operation with the 10 μm and 50 μm meshes, the removal efficiency for size bands 0.45-1 μm , 1.1-10 μm and 10.1-50 μm were 12 %, 56 %, 73 % and -1 % 50 %, 51 %, respectively. These levels increased to 41 %, 77 %, 87 % and 14 % 53 %, 54 % after 180 minutes demonstrating the impact of the dynamic layer on the effective particle cut of the system. Comparison to the 1 μm mesh showed enhanced removal with levels at the same size bands after 180 minutes of 51 %, 78 % and 89 % respectively. Application of a cross flow velocity reduced the effective cut of the dynamic layer as demonstrated by the removal efficiencies of 1 μm mesh for the 0.45-1 μm , 1.1-10 μm and 10.1-50 μm size bands after 180 minutes of 49 %, 66 % and 87 % respectively. This reduction increased for higher CFV as demonstrated by the woven mesh of 10 μm pore size. In fact efficiency removal of -31 %, 42 % and 56 % was achieved after 180 minutes. Overall, the different materials demonstrate good removal of large solids but enable transport of smaller particles through the dynamic layer and the mesh beyond the nominal size of the material. The poorer removal at smaller sizes also collaborates the difference in turbidity removal compared to suspended solids removal and reflects the passage of predominately smaller particles. For example at the highest TSS removal efficiency (67 %) the respective turbidity removal was 27 %. Instead, turbidity removal of 12 % and 14 % were achieved for the lowest TSS removal (37 %). This phenomenon occurred in all trials and likely it depended on the particles breakage caused by the stirrer. In fact, the turbidity values in the influent ranged from 3 to 8 times higher than that measured at the beginning (Table 3-3).

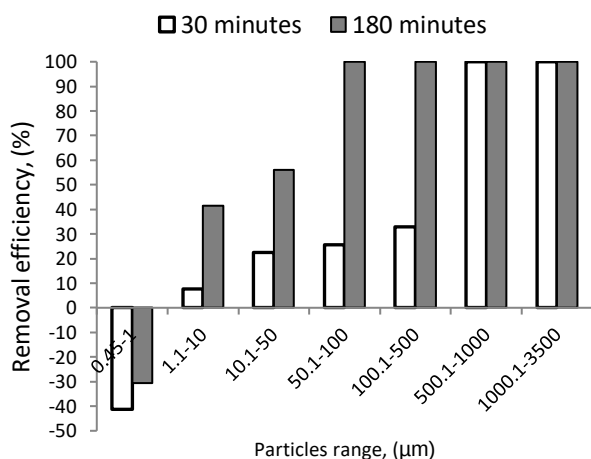
Non-woven mesh: 1 μm , TMP 0.5 kPa, CFV 0.01 m s^{-1}



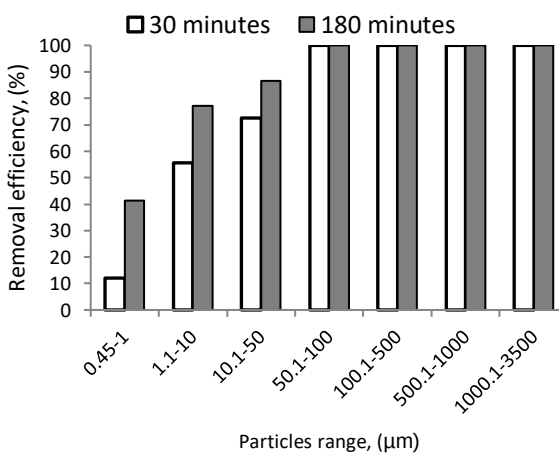
Non-woven mesh: 1 μm , TMP 0.5 kPa, dead-end



Woven mesh: 10 μm , TMP 2 kPa, CFV 0.1 m s^{-1}



Non-woven mesh: 10 μm , TMP 0.5 kPa, dead-end



Non-woven mesh: 50 μm , TMP 0.5 kPa, dead-end

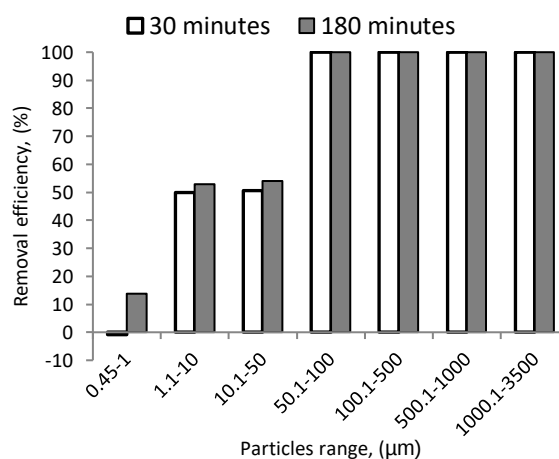


Figure 3-12 Removal efficiency of different groups of particles identified through the PSD analysis

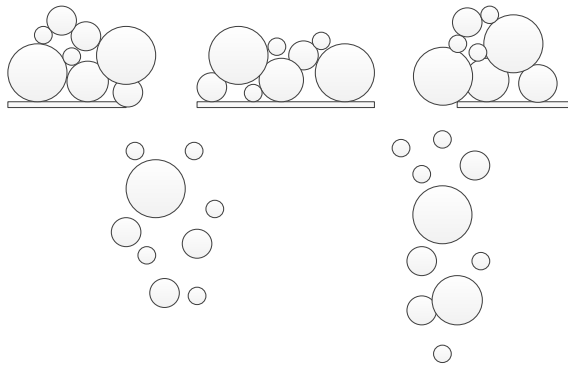
3.4.4 Discussion

The results presented here have demonstrated the ability to polish UASB effluents with cloth materials enabling reasonable solids and COD removals whilst retaining substantially higher fluxes than those obtainable with membranes. For instance, final fluxes were higher than 200 LMH, especially with those operating at steady TMP of 0.5 kPa and CFV of 0.01 m s^{-1} or in dead-end mode. This value was significantly higher than 10-20 LMH achievable with membrane bioreactor for wastewater treatment (Judd, 2011) or in traditional DM operation treating the full biomass audit (Ersahin *et al.*, 2012). In addition, solids removal efficiencies higher than 50 % with solids concentration lower than 30 mg L^{-1} could be achieved after three hours. Whilst such levels are substantially higher than those observed with membranes they represent reasonable levels for the operation of downstream technologies such as zeolite filters and hybrid ion exchange resin beds (Canellas-Garriga, 2018). Accordingly, the use of coarse membrane materials such as non-woven mesh appears a viable way forward to deliver anaerobic sewage treatment.

The main difference between the trials was seen between the non-woven and woven materials in conjunction with changes in solids loading rate and the operating TMP. All features reflect the importance of how the cake layer initially forms on the mesh material in relation to removal of solids and the sustainability of the operating flux. With regards to the material, the cloth with non-woven yarns consisted of a thicker media forming a tortuous path to retain particles and as such provides a degree of depth filtration (Figure 3-13). In comparison, woven cloths offer a truer surface filtration such that the particles are mostly retained on the surface of the media (Figure 3-13) (Baker, 2004). The PSD analysis also showed better removal of smaller particles by cloths with non-woven yarns, especially in dead-end mode with the smallest pore size ($1 \text{ }\mu\text{m}$) (Figure 3-12). Nevertheless, the presence of particles bigger than the cut-off could be related to the irregular form of particles and their orientation nearby the cloth filter surface. In addition, the measurement of negative separation can be attributed by breakage of particles during the filtration process likely due to the stirrer (Persson *et al.*, 2006). However, the improved removal and reduced flux

over time demonstrated the importance of the formed cake on top of the mesh materials. Accordingly, higher solids concentration going onto the cloths resulted in better removals as more substantial cake layers were formed. A similar result was achieved by Grabbe *et al.* (1998) who tested different cloths observing that increasing the solids concentration in the influent from 20 to around 80 mg L⁻¹ removal efficiencies up to 90% was achieved thanks to the progressive formation of a very fine biofilm on the cloth surface (Grabbe *et al.*, 1998). Moreover, under the same conditions, cloths with smaller pore sizes showed higher solids removal and consequently lower fluxes (Figure 3-6a and b) reflecting higher packing levels in the cake at the surface of the mesh. Interestingly, applying or increasing the CFVs did not enhance flux as has been seen elsewhere (Shamsuddin *et al.*, 2016; Bhave, 2000; Stamatakis and Tien, 1993) implying that in our case the particles packing adjustment was more impacted by the shape and size of particles, surface topology and interparticle forces than the hydrodynamics (Mackely and Sherman, 1993). In fact, according their observation, the increase of the CFV from 0 m s⁻¹ to 0.24 m s⁻¹ caused a decrease of the filtrate flux of almost 70% after 50 minutes at a TMP of 5 kPa. Therefore high CFVs would have a selective effect on the particles position that, in turn, would form a cake with reduced thickness than that achieved at low CFVs (Figure 3-14). In addition, high CFVs have been reported to result in an increased force onto the cake resulting in transport of particles through the filter (Al-Malack and Anderson, 1997). However, the impact of the incoming solids concentration also demonstrated to be an important factor in flux decrease. This aligns to previous studies where an increased solids concentration was reported to cause a collapse of the permeate flux approximately half way through the cycle but with an improved quality effluent (Al-Malack and Anderson, 1997).

Woven mesh made of polyamide



Non-woven mesh made of polypropylene

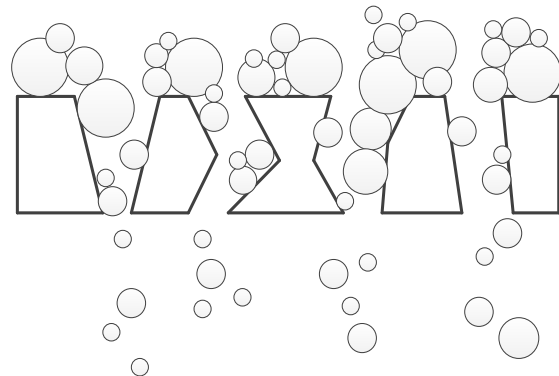
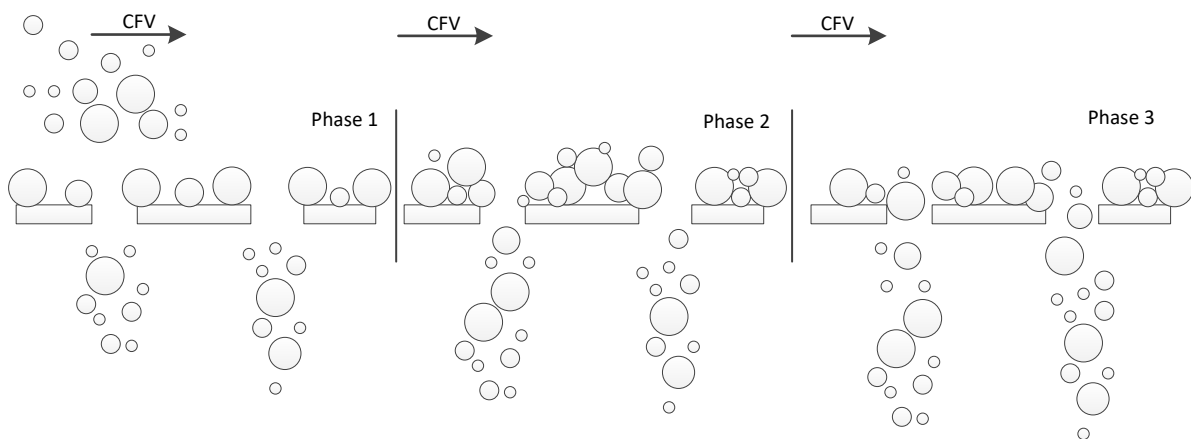


Figure 3-13 Effect of the type of cloth on solids removal from UASB effluent

a)



b)

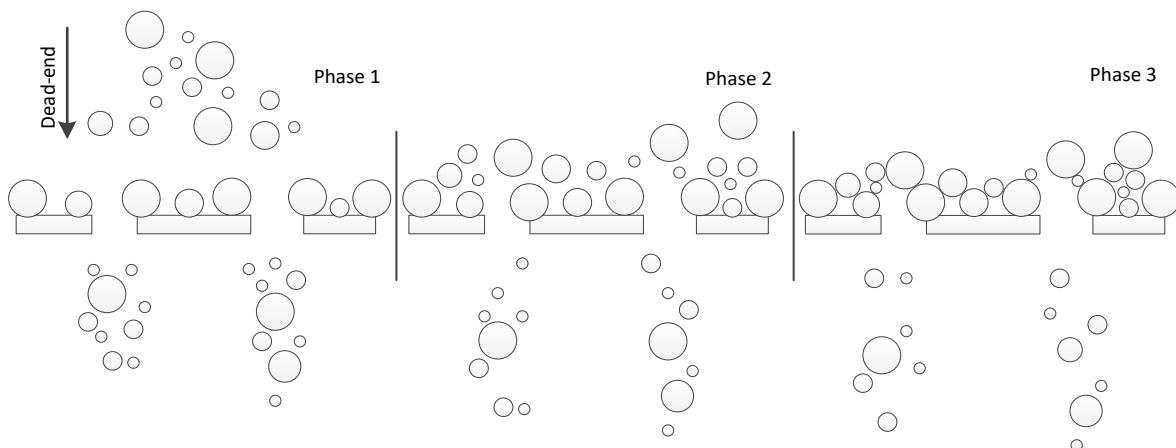


Figure 3-14 Cake formation and on consequent particle entrapment for a cloth filter in presence (a) and not (b) of CFV.

3.5 Conclusion

The study investigated the feasibility of a cloth filter for solids removal from UASB effluent, analysing which operational conditions could improve the solids removal. The following conclusion can be drawn:

- Cloth filters are an appropriate alternative option for post UASB filtration. With non-woven cloth materials provided the most effective filtration systems
- Cloths with non-woven yarns provided the highest solids removal, especially filter with pore size of 1 μm working at TMP of 0.5 kPa and in dead-end mode. In this case the TSS removal efficiency was 67% with solids concentration of 16 mg L^{-1} and permeate flux of 1200 LMH.
- PSD analysis demonstrated that the cloths were able to effective removal all particles above 50 μm . Improved removal of smaller particles is observed with smaller pore sizes, dead end operation and once the cake has been established.
- Higher CFV and TMP did not improve fluxes through the woven and non-woven cloth such that operation is best achieved under low or non cross flow operation.

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4 Implications of the work

The thesis focused on solids removal from UASB effluents using a cloth filter as an alternative to a conventional ultrafiltration membrane. The reason for this is that the membrane exerts a high CAPEX and OPEX as well as a high energy demand, potentially restricting the attractiveness of the technology. The overall operating costs are mainly related to fouling control through the gas sparging system which has been reported to represents more than two-third of the total energy demand (Pretel et al., 2014). Recent works has looked to minimise such features through optimised use of sparging and increasing the net flux (Wang *et al.*, 2018a, 2018b). However, energy demands for anMBRs remain above 0.4 kWh m⁻³ which is substantially higher than that reported for alternative separator technology (Table 4-1).

The current study explored the use of cloth filters polishing a UASB effluent resulting in the following key observations:

1. Cloth filtration is an effective way of managing downstream solids removal post a UASB. The most effective filtration trials utilised non-woven cloth materials operating at TMPs of 0.5 kPa and no cross flow (dead end). Under these conditions a solids removal of up to 67% was possible leading to an effluent with solids below 30 mg L⁻¹ whilst maintaining a flux above 1000 LMH (Chapter 3).
2. In general, the filtration tests performed with the UASB effluent showed that higher CFV and TMP did not provide benefit with regards to flux delivery over the entire cycle. Whereas the solid concentration in the UASB effluent had an impact on the solids separation and associated permeate flux from the cloth filter (Chapter 3).
3. Characterisation of the UASB effluent revealed the solids concentration varied between 40±13 mg L⁻¹ when configured as a UASB to 50±19 mg L⁻¹ when configured as an anMBR. The associated particle size distribution of the particles exiting the UASB contained three major peaks at sizes of around 1, 10 and 100 µm (Chapter 2).

4. Two features were shown to significantly impact the concentration and the nature of the solids exiting the UASB:
 - a. The first was the sludge blanket, the height of which correlated with solids removal with a maximum removal of 74% observed (Chapter 2).
 - b. The second was that inclusion of the separator in the recycle line impacted the nature of the solids exiting the UASB by altering the number and nature of the particles entering the UASB. This was observed in terms of an increase in solids exiting the UASB which were slightly larger in size and more rounded (Chapter 2).

Comparison to UF membranes revealed significantly greater fluxes but a reduction in solid separation. However, the measured suspended solids concentration ($<30 \text{ mg L}^{-1}$) and particle sizes ($<50 \text{ }\mu\text{m}$) are compatible with downstream nutrient recovery technologies such as fixed bed adsorbers meaning the technology has potential for inclusion in resource factory flowsheets (Canellas-Garriga, 2018) as well as general discharge. However, the resultant particle size exceeds the desired limit ahead of membrane degassing technology where a limit of $10 \text{ }\mu\text{m}$ is required to avoid channel clogging (Heile *et al.*, 2017; Cookney *et al.*, 2016). In such cases a double filtration unit would need to be considered or an alternative degassing technology. However, the current study demonstrates that cloth filter technology appears promising as an alternative to membranes for use in anaerobic sewage treatment. Broader consideration can also be given to include other separator technologies to further contemplate the potential efficacy of cloth filtration technology (Table 4-1).

Table 4-1 Performance and energy demand of different separation systems for wastewater treatment

	Solids removal	COD removal	Particle size in the effluent	Energy demand	Discharge/ resource factory
	%	%	µm	kWh m ⁻³	-
UASB	54 [*]	44 [*]	<1000 [*]		D
Membrane	>99 ¹	83 ¹	<0.01 ¹	0.4-1.6 ¹	RF
Depth filter	89 ²	79 ²	0.1-10 ³	0.043-0.049 ¹⁰	RF
Dissolved Air Flotation	>90 ⁴⁻⁵	80-90 ⁴	<10 ⁵⁻⁶	0.1-0.3 ⁷	D
Cloth filter	67 [*]	51 [*]	<50 [*]	0.034 ⁹	RF

^{*}Data related to the current study [#]Energy demand without considering cost for pumping water.¹(Wang *et al.*, 2019) ²(Tyagi *et al.*, 2009), ³(Yao *et al.*, 1971), ⁴(Penetra *et al.*, 1999), ⁵(Edzwald, 1995), ⁶(Henderson *et al.*, 2008), ⁷(Jarvis *et al.*, 2009), ⁸(Persson *et al.*, 2006), ⁹(Kängsepp *et al.*, 2016), ¹⁰(Anglian Water, 2012)

Prior to the current study, no one has reported the consideration of cloth filters post a UASB for treatment of settled sewage. However, the technology has been reported for other applications, notably tertiary treatment, where it has been demonstrated to be a low energy, efficient solids separation process. For instance, a number of studies investigated a similar technology, disc filters, reported solids removal up to 90% with concentration lower than 5 mg L⁻¹ in the tertiary treatment of pre-coagulated denitrified influent with energy requirement of 0.034 kWh m⁻³ (Kängsepp *et al.*, 2016). However the presence of coagulants could comprise the feasibility of further treatment for nutrients recovery. Further, Persson *et al.* (2006) demonstrated that a disc filter with 10 µm openings could ensure separation efficiencies above 90% for particles greater than 10 µm with energy demand of 0.009 kWh m⁻³ without considering the energy for the water pumping. Importantly, if this is replicated for UASB effluents, then membrane degassing would be possible with a single stage filtration system. Specifically Persson *et al.* (2006), through the PSD analysis, has been able to define the separation efficiency of different ranges of particles achievable at different pore size filter. For instance with 10 µm openings removal above 60 % and 20 % were achieved for particles of around 10 µm and 5 µm respectively. Whereas

with filter of 18 μm pore size, the separation efficiencies of the same particles decreased almost to 0 % and -15 % respectively.

Jefferson (1997) and (Henderson et al., 2008) also demonstrated the importance of the PSD in relation to the removal of different types of particles with DAF. For instance Jefferson (1997) reported grade efficiencies of 90 %, 80 % and 60 % in relation to particle number at particle sizes of around 20 μm , below 10 μm and nearby 2 μm respectively. Whereas the other study (Henderson et al., 2008) demonstrated that the dissolved air flotation offered grade removal efficiency above 90 % of 10-20 μm particles on algae flotation once the attachment efficiency is maximised. Equivalent removals of 5 μm and 1 μm algae were around 60 % and 20 % respectively.

Previous trials with DAF post UASB revealed COD removals that exceeded 90% (Table 4-1; Penetra *et al.*, 1999). However the success of DAF technology is related to a pre-coagulation process to ensure the majority of the particles exceed 10-20 μm . However, chemical addition reduces the sustainability of the process and the potential ability to recover nutrients where resource recovery is desired and it is hence more appropriate when discharge is the main target. In addition, the process exerts a reasonably high energy demand due to the need to supersaturate part of the flow for bubble generation with reported energy demands up to 0.3 kWh m^{-3} (Jarvis et al., 2009).

Reported removal performance of a depth filter post a UASB indicates removals above or equal to 80 % for COD and SS using a 120 cm deep bed of small media with a size between 0.15-4.75 mm (Tyagi *et al.*, 2009). Estimation of the grade efficiency of a standard embodiment of the technology through application of a standard collection collision efficiency model revealed a completed removal of all particles bigger than 40 μm decreasing to a minimum removal of around 30% at a particle size of 1 μm and a general reduction in removal up to 10 μm (Yao *et al.*, 1971). In terms of energy, Persson et al (2008) reported energy demands of 0.003 kWh m^{-3} (without considering the energy for the pumping water) which is thought to represent around 3-5% of the total

demand. Other reported energy data supports this with total demands between 0.043 and 0.049 kWh m⁻³(Anglian Water, 2012).

4.1 References

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5 Conclusions

This thesis investigated the possibility to remove solids from a UASB reactor by means of a cloth filter as alternative to the conventional ultrafiltration membrane. The study allowed to draw the following conclusions:

1. The sludge blanket layer showed a predominant impact in the particle size, structure and solids removal of the UASB reactor. In fact the increase of the sludge blanket improved the removal efficiency (up to 74% with sludge blanket height of 85 cm) and enabled greater particles size (up to 226 μm at sludge blanket height of 86-88 cm). However the presence of a membrane in the recycle line caused less consistency in the particle size increase and worse efficiency removal (10-14% lower than that achieved with other UASB configuration), suggesting implementation of separation system out of the recycle line (objective 1).
2. Higher CFV and TMP did not ensure benefits relative to the permeate flux. Non-woven mesh showed more consistency with regard to the flux and solids removal efficiency at different CFVs, TMPs and solids concentration. Higher solids removals were associated to higher incoming solids concentrations with a consequent permeate flux reduction (objective 2).
3. Non-woven mesh of 1 and 10 μm demonstrated to be a valid alternative to the membrane for solids removal from UASB effluent when operated under dead end conditions and moderate TMPs. In fact the solids removal was sufficient to ensure effluents with solids concentration ($< 30 \text{ mg L}^{-1}$) suitable for downstream treatments (e.g. nutrients recovery by ion exchange) (objective 3).

6 Future work

- The measurement of the profile of the UASB effluent should be extended to include a full year cycle to understand the impacts of seasonal variation and to ensure the limiting conditions are understood. This should be considered alongside regular monitoring of the sludge blanket which should go through multiple cycles of growth and desludge to fully explore the interaction between the sludge blanket and the effluent characteristics.
- The impact of the sludge blanket requires more detailed consideration to establish how operation of the UASB can impact the character and removal profile across the blanket. In particular it is recommended to operate across a range of upflow velocities to establish the optimum value of the v_{up} that ensures the most effective blanket in terms of solids concentration and particle size in the effluent.
- The inclusion of a separator in the recycle line of the UASB was shown to impact solids removal and change the particle size of the exiting solids. However, little is currently known about the significance of this process and so it is recommended that more detailed studies focus on characterisation of the recycle line with different levels of concentration occurring within that flow line. This should be mapped to equivalent characterisation of the incoming flow.
- Whilst the current study revealed the efficacy of using cloth filters at laboratory scale, it is important that scaled up trials are conducted to validate the findings and establish operating limits. In addition, the PSD analysis of the influent and permeate it is recommended to understand the grade removal of different particle size to establish key design features based on identified downstream options.