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Comparison of fouling between aerobic and anaerobic MBR treating municipal wastewater

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Abstract

The key driver for anaerobic membrane bioreactors (AnMBR) for municipal wastewater treatment is enabling the transition to energy neutral wastewater treatment. However, municipal wastewater delivers a comparatively constrained methane yield, which means energy conservation must be prioritised to achieve the proposed energy neutral ambition. A critical focus on membrane fouling is therefore warranted, as membrane operation represents the primary energy demand in MBRs. This review seeks to quantify the characteristics of the prevailing AnMBR biological suspension and to ascertain whether knowledge transfer exists between fouling characteristics in aerobic and anaerobic MBRs for municipal applications. Analysis of literature data revealed that the level of extractable extracellular polymeric substrate is slightly higher in aerobic MBRs than in anaerobic MBRs. However, AnMBR comprises considerably higher soluble microbial product concentrations, which have been widely reported to increase fouling propensity in aerobic systems. More distinct is the difference in the colloidal and fine solids fraction (between 1 and 10–15 μm), which is likely to dominate fouling in anaerobic systems and limit knowledge transfer from aerobic MBRs. Literature data on energy production was compared to that employed for membrane operation, and evidences that despite the challenging character of the particle matrix, energy neutral operation is achievable for AnMBR applied to municipal wastewater treatment.

Key words: biogas, biomass characteristics, energy demand, membrane bioreactor

INTRODUCTION

Anaerobic membrane bioreactors (AnMBRs) hold significant potential to reduce the overall energy demand of municipal (domestic) wastewater treatment through obviating the need for aeration, increasing energy recovery from methane production and producing lower secondary sludge (Martin-Garcia *et al.* 2011). Several reviews have been published on AnMBR over the past ten years. Liao *et al.* (2006) focused on AnMBR technology for high strength industrial wastewater treatment. More recent reviews have since updated the position of AnMBR for industrial treatment (Skouteris *et al.* 2012; Lin *et al.* 2013), which include a particularly insightful narrative on the thirty years of AnMBR commercial development for industrial application (Lin *et al.* 2013).

Development of AnMBR for municipal wastewater treatment has been comparatively limited due to the past perception that municipal wastewater could not be treated anaerobically since the low organic concentration was insufficient to support microbial growth (Lester *et al.* 2013). Integrating

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membrane technology into anaerobic systems helps respond to this challenge by decoupling hydraulic retention time (HRT) from solids retention time (SRT), thereby diminishing washout. The capability of AnMBR to achieve chemical oxygen demand (COD) and suspended solids compliance to international discharge standards has been experimentally demonstrated on municipal wastewater (Martin Garcia *et al.* 2013). The impact of reactor configuration and operating conditions on organic biodegradation, coupled with a description of potential methods for integrating AnMBR technology into the flowsheet for municipal wastewater, is presented in recent reviews (Smith *et al.* 2012; Ozgun *et al.* 2013).

While the capability of AnMBRs to treat municipal wastewater has been demonstrated, the principal driver for their implementation is in enabling the transition toward energy neutral wastewater treatment. McAdam *et al.* (2011) demonstrated this conceptually for anaerobic treatment of temperate settled wastewater, evidencing around 0.28 kWh m^{-3} of additional energy production (including recovery of the dissolved methane fraction) (Cookney *et al.* 2016). However, through comparison of the typical energy demand of membrane operation in commercial aerobic MBR technology (0.19–0.70 kWh m^{-3}; Judd 2011; Krzeminski *et al.* 2012), it can also be evidenced that energy conservation must be prioritised to achieve the proposed energy neutral ambition. Consequently, membrane fouling can be considered one of the critical barriers in achieving commercially viable AnMBR for municipal wastewater treatment, as this governs the energy demand (Judd 2011).

The mechanisms underpinning membrane fouling in aerobic MBR for municipal wastewater have been studied extensively and summarised (Le-Clech *et al.* 2006; Meng *et al.* 2009). The factors that influence membrane fouling in both aerobic and anaerobic MBR can be generally defined by the membrane characteristics, biomass properties, reactor and membrane operational conditions. Very few studies have directly compared aerobic and anaerobic MBR experimentally, for either municipal wastewater or blackwater (Baek & Pagilla 2006; van Voorthuizen *et al.* 2008; Martin-Garcia *et al.* 2011). These studies acknowledge an important distinction in fouling behaviour between aerobic and anaerobic systems, however, an in-depth characterisation and comparison of foulants has not yet been completed. What is clear from the literature is that size distribution and organics concentration in the AnMBR mixed liquor determines the characteristics and mechanisms of fouling. Providing greater resolution on the characterisation can thus inform the most effective fouling control strategies, as compared with the more extensively studied aerobic MBRs (AeMBRs). This review therefore proposes to complement existing knowledge through characterisation of biomass properties within AnMBRs for municipal wastewater treatment, evaluating their impact on membrane fouling and comparing to those in AeMBR, so as to ascertain the main factors that determine differences in fouling behaviour and characteristics between these two systems. Since membrane fouling strongly influences energy demand, published data on both energy production from AnMBR for municipal wastewater treatment and membrane specific energy utilisation was further analysed to ascertain the position of the existing literature relative to the aspiration of 'energy-neutral' wastewater treatment.

BIOMASS CHARACTERISTICS

Differences in characteristics between aerobic and anaerobic sludge are most readily attributed to the different mechanisms involved in the biological process. Aerobic biological suspensions mainly comprise microorganisms, decay products and influent solids forming microbial aggregates which are held together by high molecular weight polymers secreted by bacteria. This allows them to exist at high population densities (Laspidou & Rittmann 2002) in virtue of their high biomass yield and growth rates. It has been reported that the structure, morphology and surface properties of such suspensions can be altered by changes in physiological state of the biomass induced by changes in bioreactor operational parameters such as SRT and food to microorganism (F:M) ratio (Liao *et al.* 2001).

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On the other hand, anaerobic degradation of wastewater with dissolved, colloidal and particulate organic matter involves several sequential steps such as hydrolysis, acidogenesis, acetogenesis and methanogenesis (van Lier *et al.* 2008). Hydrolysis is thought to be an extracellular reaction where solids are converted to simple monomers by extracellular enzymes secreted by hydrolytic and fermentative bacteria (Sanders *et al.* 2000; Vavilin *et al.* 2008). As a result, and due to the low hydrolysis rates and biomass yield of anaerobic bacteria in high rate anaerobic reactors, the reactor solids inventory is considered to be mainly constituted by influent particles (Soto *et al.* 1993) that are of reduced particle size (Elmitwalli *et al.* 2001) and density (Lant & Hartley 2007). Therefore, as opposed to aerobic systems, sludge properties are probably more dependent on influent characteristics than on bioreactor operational parameters.

Mixed liquor suspended solids

Although sludge flocs are not considered the main contributors to fouling under low flux operation, mixed liquor suspended solid (MLSS) concentration has been shown to negatively affect membrane fouling in AnMBRs (Table 1). For instance, Jeison & van Lier (2006) found that changing biomass concentration had a greater impact on the formation of a cake layer than varying gas sparging intensity. Le Clech *et al.* (2003) also reported that the effect of MLSS was higher than gas sparging in an aerobic MBR. However, in their study, critical flux increased at higher biomass concentrations of 12 g MLSS L^{-1} as compared to the lowest MLSS concentration of 4 g L^{-1}. Robles *et al.* (2013a) also indicated that higher MLSS concentration led to lower membrane permeability, which supports the observations of Jeison & van Lier (2006). Martin Garcia *et al.* (2013) compared a flocculated AnMBR configured as a continuous stirred tank reactor (CSTR) and a granulated AnMBR configured as an upflow anaerobic sludge blanket (UASB) reactor for settled municipal wastewater treatment, which presented MLSS concentrations for membrane filtration of 7.7 and 0.1–0.6 g L^{-1}, respectively. Higher fouling rates and lower critical fluxes have been determined in CSTR configured AnMBR with higher MLSS concentrations.

The contradicting influence that MLSS concentration has on membrane hydraulic performance between aerobic and anaerobic MBRs can be attributed to differences in the relationship between biomass and dissolved/colloidal compounds in the mixed liquor. In aerobic MBRs, it has been widely reported that higher levels of soluble microbial products are found at lower MLSS when short sludge retention times are applied (Massé *et al.* 2006; Liang *et al.* 2007) while in anaerobic systems, soluble microbial products (SMP) tend to accumulate together with biomass (Harada *et al.* 1994) or at high sludge ages.

Particle size distribution

According to Lant & Hartley (2007), anaerobic sludge particle sizes are one order of magnitude lower than aerobic floc (activated sludge floc), even though the range of particle sizes covers three orders of magnitude as opposed to only one for aerobic biomass. Data collated from the literature identified similar median particle size ranges of 5.2–220 and 0.8–138 μm in aerobic and anaerobic MBRs respectively (Table 2), however, a significant difference is the presence of a population of fine colloidal matter in AnMBR which has been shown to negatively affect membrane performance. This was confirmed by Martin-Garcia *et al.* (2011) who found a higher colloidal content in a flocculated AnMBR (CSTR) and granulated AnMBR (UASB), when compared directly with AeMBR. The authors attributed the higher degree of dispersive growth to higher fouling propensity.

Particle size analysis in AnMBRs (Table 2) indicates that externally configured (side-stream) membrane systems yield considerably lower particle sizes when compared to immersed systems using gas sparging (Imasaka *et al.* 1989; Bailey *et al.* 1994; Hu & Stuckey 2006; Jeison & van Lier 2006)

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Table 1 | Overview of operating conditions and membrane performance in immersed anaerobic MBRs treating municipal wastewater (real and synthetic)

Source	Reactor type/ Sludge	Reactor size RE. l	Reactor size FI.	Material/ Geo./ Pore size μm	MLSS g L^{-1}	SGD $m^3 m^{-2} h^{-1}$	J_c L $m^{-2} h^{-1}$	$J_{c,20}$	Flux L $m^{-2} h^{-1}$	TMP mbar	Fouling rate mbar min^{-1}	Fouling rate kPa h^{-1}	T_{op} h	Ref.
Cru[a]	UASB/F	17.7[d]		PE/HF/0.03	16–21.5	0	–	–	5,10	<700	0.033,0.058	0.200,0.350	336,144	Wen *et al.* (1999)
Cru[b]	CSTR/F	1,300	800	PVDF/HF/0.05	10–30	0.23	–	–	9–13.3	–	–	–	14,400	Robles *et al.* (2013b)
Cru[b]	CSTR/F	–	50	PVDF/HF/0.04	21.3	1.3	10–12	10.3–12.3	10	<140	0.002–0.075	0.012–0.45	60	Ruigómez *et al.* (2016b)
Cru[b]	CSTR/F	–	50	PVDF/HF/0.04	21.3	0	12–14	12.3–14.4	10	<140	0.001–0.027	0.007–0.16	>200	Ruigómez *et al.* (2016b)
Cru[a]	CSTR/F	550	80	PVDF/HF/0.04	12.8[f]	0.146	–	–	17	<215	0.002	0.010	2,160	Dong *et al.* (2015)
Cru[a]	CSTR/F	550	80	PVDF/HF/0.04	5.6[f]	0.146	–	–	17	<88	0.001[j]	0.004[j]	2,160	Dong *et al.* (2015)
Cru[a]	CSTR/F	550	80	PVDF/HF/0.04	11.3–23	0.146	21–27	20–25	17	<250	0–0.004	0.001–0.025	4,272	Dong *et al.* (2016b)
Cru[a]	CSTR/F	60[d]		PVDF/FS/140[f]	6.4–9.3	0.12	–	–	12	–	–	–	–	Lin *et al.* (2011)
Cru[c]	IAFMBR	7.6[d]		–/HF/0.4	–	0	–	–	11.3	<300	0.014	0.086	348	Gao *et al.* (2014b)
Cru[c]	IAFMBR	7.6[d]		–/HF/0.4	–	0	–	–	11.3	<300	0.009[k]	0.052[k]	576	Gao *et al.* (2014b)
Cru (GA)	CSTR/F	350	–	PES/FS/0.038	15	–	7	7	7	<280	<0.002	0.001–0.011	2,352	Martinez-Sosa *et al.* (2011)
Black	UASB/F	5	–	PVDF/Tub/250[e]	–	–	–	–	10[g]	–	–	–	–	van Voorthuizen (2008)
Black	CSTR/F	4[d]		PVDF/Tub/250[e]	–	–	–	–	8[g]	–	–	–	–	van Voorthuizen (2008)
Set	UASB/F	160	150	PVDF/HF/0.045	0.5,5–7[f]	0.2, 0.4–1.0	–	–	2.5,10–15	50, <550	–	–	4,320	Gouveia *et al.* (2015a)
Set	UASB/G	326	175	PVDF/HF/0.045	2, 12–14[f]	0.07–0.13	–	–	12, 15.7	–	0.001–0.013	0.004–0.079	–	Gouveia *et al.* (2015b)
Set	UASB/G	94	31	PVDF/HF/0.04	0.1–0.6	1.2	4.3–13.4	4.6–14.2	6[g]	<100	0.001	0.004–0.013	>300	Martin Garcia *et al.* (2013)
Set	CSTR/F	900	300	PVDF/HF/0.04	7.7	0.4	1.9–9.7	2.0–10.3	6[g]	<700	0.033	0.198	<200	Martin Garcia *et al.* (2013)
Set	CSTR/G	40[d]		–/HF/0.1	6.5–6.8	–	–	40	–	–	0.004–0.007	0.025–0.042	–	Fawehinmi (2006)
Set	CSTR/F	5	1	PES/PF/0.45	8.0–13.6	–	–	–	–	<300	0.005–0.014	0.031–0.083	360–960	Huang *et al.* (2013)

(*Continued.*)

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Table 1 | Continued

Source	Reactor type/ Sludge	Reactor size RE. l	Reactor size FI.	Material/ Geo./ Pore size μm	MLSS g L^{-1}	SGD $m^3 m^{-2} h^{-1}$	J_c L $m^{-2} h^{-1}$	$J_{c,20}$	Flux L $m^{-2} h^{-1}$	TMP mbar	Fouling rate mbar min^{-1}	Fouling rate kPa h^{-1}	T_{op} h	Ref.
Set	SAF–MBR/F	0.245	0.245	PVDF/HF/0.1	–	0	–	–	9	<400	<0.001–0.035	0.002–0.213	9,500	Yoo *et al.* (2014)
Set	SAF–MBR/F	990	770	PVDF/HF/0.03	0.6–1.2[f]	0	–	–	4.1–7.5[g]	<500	–	–	11,640	Shin *et al.* (2014)
Syn	UASB	–		PVDF/FS/0.22	–	1.8	–	–	25	<300	0.055–0.530[l]	0.330–3.180[l]	–	Wu *et al.* (2009)
Syn	UASB/G	4.7[d]		PE/0.1	–	0	–	–	–	–	–	–	480	Chu *et al.* (2005)
Syn	UASB/F	30[d]		PVDF/FS/0.22	0.14	0–1.2	–	–	25	<300	0.417–1.250	2.5–7.5	4–12	An *et al.* (2010)
Syn	CSTR/F	5	1	PES/PF/0.45	5.6–10.5	–	–	–	5.3–7.9	<300	<0.006	0.002–0.036	840–2,400	Huang *et al.* (2011)
Syn	CSTR/F	3[d]		PE/FS/0.4	4.3–4.8	3	–	–	20	<270	0.003–0.005[k]	0.018–0.031[k]	384	Hu & Stuckey (2007)
Syn	CSTR/F	3[d]		PE/FS/0.4	6–19,12–16	3	–	–	10	120–230	–	–	360	Akram & Stuckey (2008)
Syn	CSTR/F	7[d]		PES/FS/0.2	–	7.24	–	–	7	–	<0.001[m]	<0.001[m]	720	Smith *et al.* (2013)
Syn	AFBR+ AFMBR	0.245	0.245	PVDF/HF/0.1	0.09–0.13[f]	0	–	–	9	<100	–	–	2,400	Bae *et al.* (2014)
Syn	ABR+ AFMBR/F	3.93	2.0	PVDF/HF/0.1	–	0	–	–	7	<30[h], < 320[i]	0.444[i]	2.667[i]	–	Kim *et al.* (2011)
Syn	ARMBR	4[d]		PE/FS/0.22	5.5	0	–	–	11	<50	–	–	2,400	Kim *et al.* (2014)

Acronyms: ABR + AFMBR: anaerobic baffle reactor + anaerobic fluidised bed membrane bioreactor; AFBR: anaerobic fluidised bed reactor; ARMBR: anaerobic rotate disk MBR; BW: backwash; Cru: Crude; CSTR:continuous stirred tank reactor; F: flocculent sludge; FI: filtration; FS: flat sheet; G: granular sludge; GA: glucose addition; HF: hollow fibre; IAFMBR: integrated anaerobic fluidised–bed; MWCO: molecular weight cut off; PE: Polyethylene; PES: polyethersulfone; PF: plate and frame; PVDF: polyvinylidene fluoride; RE: anaerobic reactor; SAF–MBR: staged anaerobic fluidised MBR; Set: Settled; SGD_m: specific gas demand; Syn: Synthetic; T_{op}: time of performance; Tub: tubular; UASB: upflow anaerobic sludge blanket.

[a]Crude: after screening.

[b]Crude: pre–treatment including screening, degritter and grease removal.

[c]Crude: from septic tank.

[d]Membrane submerged in the anaerobic reactor.

[e]MWCO (kDa).

[f]reported MLVSS.

[g]Net flux.

[h]With GAC.

[i]Without GAC.

[j]With $FeCl_3$ addition.

[k]With GAC (granular activated carbon) or PAC (powered activated carbon) addition.

[l]With addition of PAC (powered activated carbon)/zeolite/polyamide/polyaluminum chloride.

[m]Backwash 4 min every 4 h.

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Table 2 | Particle size distribution of sludge/biomass in aerobic and anaerobic MBRs in contact with the membrane

Source	Reactor Type	Geo./Mode/Config.	Temp °C	SRT d	Average d_{p50} μm	Ref.
Anaerobic MBRs						
Industrial[a]	CSTR	PF/pumped/Side	53–55	–	3–16	Choo & Lee (1998)
Industrial[b]	CSTR	Tubular/pumped/Side	52	–	0.8,4.0	Imasaka *et al.* (1989)
Settled	UASB	HF/gas sparged/Sub	–	30,200	60.3,1.4	Martin-Garcia *et al.* (2011)
Settled	CSTR	HF/gas sparged/Sub	–	30,200	42.8,7.6	Martin-Garcia *et al.* (2011)
Settled	CSTR	PF/gas sparged/Sub	25–30	30,60	28,21	Huang *et al.* (2013)
Synthetic	UASB	Tubular/pumped/Side	30	–	36,16[d]	Bailey *et al.* (1994)
Synthetic[c]	UASB	FS/pumped/Side	–	–	75,13[d]	Cho & Fane (2002)
Synthetic	UASB	Tubular/pumped/Side	25	–	465,138[e]	Ozgun *et al.* (2015a)
Synthetic	UASB	Tubular/pumped/Side	25	–	80–137[f]	Ozgun *et al.* (2015b)
Synthetic	UASB	Tubular/pumped/Side	15	–	25–88[f]	Ozgun *et al.* (2015b)
Synthetic	CSTR	–/pumped/Side	35		13	Elmaleh & Abdelmoumni (1997)
Synthetic	CSTR	Tubular/pumped/Side	25	90–360	50.6,25.3[d]	Ho & Sung (2009)
Synthetic	CSTR	Tubular/gas sparged/Sub	30,55	–	70–90[g]	Jeison & van Lier (2006)
Synthetic	CSTR	FS/gas sparged/Sub	35	250	23.5	Akram & Stuckey (2008)
Synthetic	CSTR	FS/gas sparged/Sub	35	250	9.6–16.3[h]	Akram & Stuckey (2008)
Synthetic	CSTR	FS,HF/gas sparged/Sub	35	–	60–65	Hu & Stuckey (2006)
Synthetic	CSTR	PF/gas sparged/Sub	25–30	30–∞	24–31	Huang *et al.* (2011)
Aerobic MBRs						
Industrial + Crude	CSTR	HF/gas sparged/Sub	>10	–	31.8–35.6	Sun *et al.* (2011)
Crude	CSTR	MT/pumped/Side	20	60	50–300	Defrance *et al.* (2000)
Crude	CSTR	HF/gas sparged/Sub	9–21	5,20,40	14,48,31	Huang *et al.* (2001)
Settled	CSTR	HF/gas sparged/Sub	20	10,30	120–220, 70–100	Massé *et al.* (2006)
Settled	CSTR	HF/gas sparged/Sub	–	30,200	32.4,14.0	Martin-Garcia *et al.* (2011)
Synthetic	CSTR	HF/gas sparged/Sub	–	20,40,60	5.2–6.6	Lee *et al.* (2003)
Synthetic	CSTR	HF/gas sparged/Sub	–	30	57.6	Zhou *et al.* (2014)

Acronyms: CSTR: continuous stirred tank reactor; FS: flat sheet; HF: hollow fibre; MT: multiple tube; PF: plate & frame; UASB: upflow anaerobic blanket reactor.
[a]Alcohol–distillery wastewater.
[b]Concentrated thermophilic fermentation broth of evaporator condensate discharged from a kraft pulp mill.
[c]Not mention synthetic, but assume according to the influent characteristics.
[d]Particle size before and after shear induction.
[e]UASB sludge particle size before and after membrane addition.
[f]Particle size in the UASB effluent.
[g]Estimate from graph.
[h]With addition of PAC (powered activated carbon).

which can be attributed to the shear imposed by the pumping demanded to maintain cross-flow velocity (Imasaka *et al.* 1989; Jeison *et al.* 2009). Analogous effects have been demonstrated following conversion of conventional anaerobic reactors to AnMBR (Ho & Sung 2009; Ozgun *et al.* 2015a) and it is suggested that such particle reduction can also decrease biomass activity (Brockmann & Seyfried 1997).

Apart from hydrodynamic conditions, operating temperature has been shown to influence biomass characteristics. Ozgun *et al.* (2015b) observed a reduction in median particle size from 80–137 to 25–88 μm when the temperature was reduced from 25 to 15 °C during the treatment of synthetic municipal wastewater. This was corroborated by Robles *et al.* (2013a), following the transition from mesophilic to psychrophilic conditions. The authors proposed that the smaller particle sizes led to lower cake layer porosities, and higher cake layer tortuosity, which resulted in considerable cake layer resistance. Transients in particle size distributions have been noted in both anaerobic

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and aerobic MBR systems following changes to SRT. Huang *et al.* (2011, 2013) reported a decrease in median particle size for AnMBR when SRT increased from 30 to 60 d, which is similar to the observations of Martin-Garcia *et al.* (2011). Similar effects of particle size reduction in aerobic MBR in response to an increase in SRT have been observed (Huang *et al.* 2001; Martin-Garcia *et al.* 2011).

Organic fouling by EPS

Extracellular polymeric substances (EPS), composed mainly of polysaccharides, proteins and lipids, have been widely reported as being responsible for organic fouling in both aerobic (Pollice *et al.* 2005; Fan *et al.* 2006; Rosenberger *et al.* 2006) and anaerobic MBRs (Harada *et al.* 1994; Cho & Fane 2002; Hu & Stuckey 2006; van Voorthuizen *et al.* 2008). For example, Rosenberger *et al.* (2006) revealed high polysaccharide concentrations in the sludge supernatant corresponded to high membrane fouling rate in aerobic MBR. In AnMBR, Huang *et al.* (2011) indicated higher protein and polysaccharides accumulated at long SRT, which resulted in increased fouling. These biopolymers have been fractionated according to whether they are found in the sludge supernatant as SMP or bound to the sludge flocs and are thus extracted from the cell walls (eEPS). Although the term SMP implies that these substances are of bacterial origin, they may also be the result of recalcitrant or partially transformed influent organics. This is particularly the case for anaerobic systems at lower temperatures where lower biodegradation rates apply, which increase SMP concentration above those of aerobic biomass (Lettinga *et al.* 2001). However, independent of their origin, analysis of eEPS and SMP has contributed to further characterisation both of the solid and the colloidal/soluble fractions of biological suspensions respectively.

Bound/extractable EPS

The surface properties of the sludge are primarily determined by eEPS. The hydrophobicity and surface charge have been correlated with the total EPS concentration and the ratio of proteins to carbohydrates in both conventional activated sludge and MBR systems (Liao *et al.* 2001). For instance, a higher ratio of proteins to carbohydrates in activated sludge has been reported to enhance bioflocculation through the reduction of surface charge and increase in hydrophobicity (Liao *et al.* 2001). On the other hand, high proportions of carbohydrates in the eEPS are associated with a more dispersed sludge structure due to the greater repulsion between sludge particles and interaction with the aqueous phase resulting from the higher negative surface charge and reduced hydrophobicity (Liao *et al.* 2001).

Literature data regarding surface properties of anaerobic sludge do not correlate as consistently as those from aerobic systems, and no conclusions can be drawn regarding comparative concentrations of eEPS and fractions thereof, to allow comparison between aerobic and anaerobic sludges with respect to surface charge and hydrophobicity. Morgan *et al.* (1990) reported that aerobic sludge was more negatively charged, contained higher levels of total eEPS and lower ratios of proteins to carbohydrates than anaerobic sludge. Similarly, analysis of eEPS literature data from MBR operated with municipal wastewater (Crude, Settled and Synthetic) demonstrates a considerably higher protein to carbohydrate ratio in AnMBR when compared to AeMBR (Table 3). In terms of specific eEPS concentration, the distinction between aerobic and anaerobic systems is less clear. Comparison between anaerobic and aerobic MBRs operated with complete retention of solids and fed with settled sewage (Baek & Pagilla 2006) showed levels of EPS to continuously decrease. In a further study on AnMBR, Fawehinmi *et al.* (2004) observed an increase in specific resistance to filtration as the eEPS content of crushed granular sludge increased from 20 to 130 mg g VSS^{-1}. Cerón-Vivas *et al.* (2012) demonstrated the highest eEPS

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Table 3 | Concentration and composition of EPS in aerobic and anaerobic MBR sludge/biomass

Source	Reactor type/sludge	Config.	Temp. °C	SRT d	HRT h	MLSS g L^{-1}	EPS_{tot} mg g VSS^{-1}	EPS_p mg g VSS^{-1}	EPS_c mg g VSS^{-1}	Ratio EPSp/EPSc	Ref.
Anaerobic MBRs											
Crude[a]	CSTR/F	Sub	20	70	24.5	18–28	98[h]	74	24	7.0 (3.1)[o]	Robles *et al.* (2013a)
Crude[a]	CSTR/F	Sub	33	70	5.5–16.5	14–32	155[h]	121	34	16.4 (3.6)[o]	Robles *et al.* (2013a)
Crude[a]	CSTR/F	Sub	17–29	28.6–41.1	12.1–28.4	10–25	161[i]	126	35	3.6	Giménez *et al.* (2014)
Settled	CSTR/G	Sub	12	120[f]	6	5.9	11–32[i,n]	10.1–31.5[n]	0.9[n]	10.7–33.3	Fawehinmi (2006)
Settled	CSTR/F	Sub	25–30	30–90	10	8.0–13.6	42–50[i]	32–40	10	4–5	Huang *et al.* (2013)
Synthetic	UASB/G	Sub	11–25	–	3.5–5.7	–	5–7.7[i,j]	0.6–1.6	4.4–6.1	0.18–0.25	Chu *et al.* (2005)
Synthetic	UASB/G	Sub	11–25	–	3.5–5.7	–	17.4–20.4[i,k]	5.8–7.0	11.6–13.4	0.48–0.55	Chu *et al.* (2005)
Synthetic	UASB/G	Sub	21–24	–	8	–	219[i,l]	108–244	27–60	4.1	Cerón-Vivas *et al.* (2012)
Synthetic	UASB/F	Side	25	–	6	–	4[i]	3.7	0.3	12.3	Ozgun *et al.* (2015a)
Synthetic	CSTR/F	Sub	15	300	16	–	57–81[h,k]	32–46	25–35	1.3	Smith *et al.* (2013)
Synthetic	CSTR/F	Sub	25–30	30	8–12	5.6–7.1	55.5[i]	38–48	10–15	2.5–2.85	Huang *et al.* (2011)
Synthetic	CSTR/F	Sub	25–30	60	8–12	5.7–8.9	55.5[i]	37–45	11–18	2.0–2.85	Huang *et al.* (2011)
Synthetic	CSTR/F	Sub	25–30	∞	8–12	6.5–10.5	49.5[i]	35–42	11	2.5	Huang *et al.* (2011)
Synthetic	IAFMBR[d]/F	Sub	35	–	6	–	60[i,m]	45[m]	15[m]	3	Gao *et al.* (2014b)
Synthetic	IAFMBR[d]/F	Sub	35	–	6	–	37.5–42.5[i,m]	30–35[m]	5–10[m]	4.3	Gao *et al.* (2014b)
Synthetic	IAFMBR[d]/F	Sub	35	–	6	–	70[i,k,m]	52[m]	18[m]	2.8	Gao *et al.* (2014b)
Aerobic MBRs											
Crude[b]	CSTR	Sub	16–22	∞	13.3	5.2–10.5	25.5–79.8[i]	20.4–64.5	3.4–34.0	2.3	Bella *et al.* (2011)
Crude[b]	CSTR	Sub	–	40–45	16	10–31.2	34–157[i]	–	–	–	Holba *et al.* (2012)
Crude[b]	CSTR	Sub	–	30–75	42	2.2–5.6	38–114[i]	–	–	–	Holba *et al.* (2012)

(*Continued.*)

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Table 3 | Continued

Source	Reactor type/sludge	Config.	Temp. °C	SRT d	HRT h	MLSS g L^{-1}	EPS_{tot} mg g VSS^{-1}	EPS_p mg g VSS^{-1}	EPS_c mg g VSS^{-1}	Ratio EPSp/EPSc	Ref.
Crude[b]	CSTR	Sub	11–23.8	5–12	6	10–11.2	90–140[h]	–	–	–	Fan *et al.* (2006)
Crude[b]	CSTR	Sub	13–26	10	10	3.8–4.2	162.7[i]	58.7	42.9	1.4	Liu *et al.* (2012a)
Crude[b]	CSTR[e]	Sub	13–26	10	10	3.8–4.3	165.2[i]	58.6	41.8	1.4	Liu *et al.* (2012a)
Crude[c]	CSTR	Sub	–	–	–	6.0	6.9[h]	3.5	3.4	1.0	Gabarrón *et al.* (2013)
Settled	CSTR	Sub	–	10–30	–	12–18	81–115[i]	57–88	24–29	2.4–3.2	Trussell *et al.* (2007)
Settled	CSTR	Sub	20	10	16	1.9	45–70[i]	–	–	2–4	Massé *et al.* (2006)
Settled	CSTR	Sub	20	53	16	6.0	20–40[i]	–	–	2–4	Massé *et al.* (2006)
Synthetic	CSTR	Sub	–	20,40,60	7.8	2.8–5.5[g]	63–70[i]	30–36	30–35	1.0	Lee *et al.* (2003)
Synthetic	CSTR	Sub	–	10–80	–	1.7–3.7	–	<2	3–6	0.33–0.67	Duan *et al.* (2015)

Acronyms: CSTR: continuous stirred tank reactor; F: flocculent sludge; G: granular sludge; GAC: granular activated carbon; IAFMBR: integrated anaerobic fluidised–bed membrane bioreactor; SAF–MBR: staged anaerobic fluidised membrane bioreactor; UASB: upflow anaerobic sludge blanket.

[a]Crude: pre-treatment including screening, degritter and grease removal.

[b]Crude – after screening.

[c]Data from full scale plant, Crude – after coarse screen, grit chamber, buffering and fine screen.

[d]With 40 g GAC addition.

[e]With biofilm carrier.

[f]Not report SRT, but no sludge is wasted during the tests except sampling. SRT can be assumed the same as experiment duration time.

[g]Reported MLVSS.

[h]Reported as extracted EPS.

[i]Reported EPS.

[j]EPS of sludge on granules.

[k]EPS of sludge on membrane.

[l]EPS in the UASB effluent.

[m]No MLSS or MLVSS, data is reported with initial unit mg L^{-1}.

[n]Use MLVSS/MLSS ratio = 0.63 to convert the unit from mg g SS^{-1} to mg g VSS^{-1}.

[o]Values in the bracket were calculated directly from the average value.

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values above 200 mg g VSS^{-1} in a synthetic wastewater. Analysis of the arising cake layer revealed that the specific eEPS deposited on the membrane surface was twice that found in the granular sludge (Chu *et al.* 2005) with higher eEPS protein to carbohydrate ratio on the membrane, whilst Lin *et al.* (2009) reported a lower eEPS protein to carbohydrate ratio in the fouling layer when compared to the bulk material. Regardless of the major fouling component, the discrepancies between biomass and cake layer eEPS composition found in these studies suggest that soluble or colloidal compounds are also responsible for the increase in membrane resistance in AnMBRs.

Soluble-colloidal EPS: SMP

It has been reported that soluble organic matter in the effluent from the biological treatment processes is predominantly SMP, which comprises the soluble cellular components released during cell lysis, lost during synthesis, or otherwise secreted for some purpose (Laspidou & Rittmann 2002). Soluble microbial products are classified according to their origin as products associated with biomass growth and are produced at a rate proportional to substrate utilisation and non-growth associated products related to cell lysis. In conventional (i.e. non-membrane based) systems, the concentration of SMP normalised to influent COD is higher in aerobic than anaerobic processes (Barker & Stuckey 1999). This can be explained by the lower biomass uptake and decay rates of anaerobic microorganisms compared to aerobic biomass.

The SMP concentration within aerobic and anaerobic MBR is higher than for conventional processes (Aquino & Stuckey 2006; Massé *et al.* 2006). Ozgun *et al.* (2015a) directly compared the SMP of conventional UASB with UASB coupled with membrane and reported that SMP increased over three times after membrane addition, from about 37 to 120 mg L^{-1}. This is due to an increased retention of the high molecular weight organic fraction by the membrane (Massé *et al.* 2006) as well as an increase in net SMP productivity through endogenous decay and cell lysis which are enhanced by long SRT and high organic loading rates (Harada *et al.* 1994). Anaerobic MBRs have considerably higher SMP concentrations than are present in aerobic MBR (Table 4). Normalising literature SMP_{COD} data to influent COD (w/w), suggests arising SMP_{COD} ranges from 10 to 48% and from 9 to 59% for aerobic and anaerobic MBRs, respectively. This has been confirmed through comparative experimental study of anaerobic and aerobic MBR systems for municipal wastewater treatment, which has evidenced SMP concentrations up to five times higher in anaerobic MBR (Martin-Garcia *et al.* 2011). The higher colloidal content within AnMBR, compared with that of AeMBR, may reflect the higher levels of free bacteria in mixed liquor, together with the lower biodegradation rates or SMP biodegradability expected under anaerobic conditions (Ince *et al.* 2000). There is also evidence of high molecular weight polymeric material of up to 1,000 kDa being retained by the cake layer in AnMBRs, which presumably decreases permeability and potentially increases rejection capability (Harada *et al.* 1994). Feed temperature is also a factor, where decreased temperature from 25 to 15 °C in an AnMBR increased SMP_{COD} from 50 to 150 mg L^{-1} (Ho & Sung 2010).

FOULING CONTROL STRATEGIES

While much of the early research on AnMBR for industrial applications sought to pursue externally configured MBR, the development of AnMBR for municipal applications has been generally more focused on immersed MBR technology, which is presumably due to the lower energy penalty that this configuration can achieve (Judd 2011).

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Table 4 | Concentration and composition of soluble microbial products in aerobic and anaerobic MBRs

Source	Reactor type[a]/ Sludge	Config.	Temp °C	SRT d	HRT h	MLSS g L^{-1}	SMP^{b}_{COD} mg L^{-1}	SMP_c mg L^{-1}	SMP_p mg L^{-1}	Ratio SMP_p/SMP_c	Ref.
Anaerobic MBRs											
Crude	UASB/F	Side	22	180	6	0.4	–	99.2	133.2	1.3	Herrera-Robledo *et al.* (2011)
Crude[c]	IAFMBR[j]	Sub	35	–	6	–	–	3–5	14–15	3.5–3.8	Gao *et al.* (2014b)
Black	UASB/F	Sub	37	–	12	–	327 (0.29)	81	70	0.9	van Voorthuizen *et al.* (2008)
Black	CSTR/F	Sub	37	–	12	–	269 (0.24)	45	69	1.5	van Voorthuizen *et al.* (2008)
Settled	UASB/G[k]	Sub	10–25	250[l]	16	0.1–0.6	198 (0.59)	18	50	2.8	Martin Garcia *et al.* (2013)
Settled	CSTR/G	Sub	12–35	120[l]	6	5.9–6.8	180 (0.40)	8	59	7.4	Fawehinmi (2006)
Settled	CSTR/F	Sub	–	100	16	7.7	598 (1.77)	47	108	2.3	Martin Garcia *et al.* (2013)
Settled	CSTR/F	Side	25	233	24	7.1	51 (0.61)[n]	–	–	–	Baek & Pagilla (2006)
Settled	CSTR/F	Sub	25–30	30–90	10	8.0–13.6	–	18–19	40–50	2.2–2.5	Huang *et al.* (2013)
Synthetic	UASB/G	Sub	21–24	–	8	–	–	0.5–1.2	1.1–3.3	2.6	Cerón-Vivas *et al.* (2012)
Synthetic	UASB/F	Sub	35	–	12	0.14	–	1.2–16.6	7.4–15.1	0.7–6.5	An *et al.* (2010)
Synthetic	UASB/F	Side	25	–	6	0.5	122 (0.23)	25–40	60–80	2.1	Ozgun *et al.* (2015a)
Synthetic	UASB/F	Side	–	–	4–12	–	40 (0.11)	–	–	–	Salazar-Peláez *et al.* (2011)
Synthetic	CSTR/F	Sub	35	150	6	3.7[m]	180[o] (0.40)	31	58	1.9	Aquino *et al.* (2006)
Synthetic	CSTR/F	Sub	35	250	15	12–16	1,787 (0.45)	–	–	–	Akram & Stuckey (2008)
Synthetic	CSTR/F	Sub	35	250	6	6–19	228–360[o] (0.09)	–	–	–	Akram & Stuckey (2008)
Synthetic	CSTR/F	Sub	25–30	30,60, ∞	8–12	5.6–10.5	–	4.5–14	7–18	0.9–2.0	Huang *et al.* (2011)
Synthetic	CSTR/F	Side	35	190[l]	48–120	15[m]	1,200 (0.24)	80	400	5	Harada *et al.* (1994)
Aerobic MBRs											
Crude[d]	CSTR	Sub	13–26	10	10	3.8–4.2	–	3.5 (1.8)[p]	3.6 (1.9)[p]	1.0 (1.1)[p]	Liu *et al.* (2012a)

(*Continued.*)

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Table 4 | Continued

Source	Reactor type[a]/ Sludge	Config.	Temp °C	SRT d	HRT h	MLSS g L^{-1}	SMP^{b}_{COD} mg L^{-1}	SMP_c mg L^{-1}	SMP_p mg L^{-1}	Ratio SMP_p/SMP_c	Ref.
Crude[e,f]	CSTR	Sub	10–25	–	–	–	–	3.6	2.4	0.7	Lyko *et al.* (2007)
Crude[g]	CSTR	Sub	11–12	5–12	6	9.7–11.9	–	17–38	10–58	1.2	Fan *et al.* (2006)
Crude[e,h]	CSTR	Sub	–	14–38	12–41	2.2–13.5	–	3–18	<5	0.5–1.7	Shen *et al.* (2012)
Crude[e,i]	CSTR	Sub	–	–	–	6.0	–	3.7	2.7	0.7	Gabarrón *et al.* (2013)
Settled	CSTR	Sub	–	10–30	–	12–18	37–82 (0.11–0.24)	12–26	10–79	0.5–3.0	Trussell *et al.* (2007)
Settled	CSTR	Sub	–	10–30	–	12–18	33–166 (0.10–0.48)	16–27	12–140	0.6–5.4	Trussell *et al.* (2007)
Settled	CSTR	Sub	20	10–110	16	1.9–7.2	45–110 (0.12–0.30)	37.2	8.9	0.2–0.6	Massé *et al.* (2006)
Settled	CSTR	Sub	–	100	16	8.7	99 (0.29)	18	18	1.0	Martin-Garcia *et al.* (2011)
Synthetic	CSTR	Sub	28	10–40	10	3.1–7.8	–	8–12	5–9	1.4	Liang *et al.* (2007)

Acronyms: CSTR: continuous stirred tank reactor; F: flocculent sludge; G: granular sludge; IAFMBR: integrated anaerobic fluidised–bed membrane bioreactor; UASB: upflow anaerobic sludge blanket.
[a]Report SMP from UASB effluent, CSTR and IAFMBR from mixed liquor.
[b]Values in brackets correspond to normalised SMPCOD with respect to influent COD.
[c]Crude: from septic tank.
[d]Crude: after sand settler and screening.
[e]Data from full scale plant.
[f]Crude: after coarse screen, grit chamber, grease trap and fine screen.
[g]Crude: pre-treatment with screening.
[h]Crude: after screening.
[i]Crude: flow after coarse screen, grit chamber, buffering and fine screen.
[j]With 40 g GAC addition.
[k]Reported SMP in the membrane tank.
[l]No reported SRT, but no sludge is wasted during the tests except sampling. SRT can be assumed as same as experiment duration time.
[m]Reported MLVSS.
[n]Normalised against influent soluble COD.
[o]With addition of PAC (powered activated carbon).
[p]Value in bracket is with bio-carrier.

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Specific gas demand and operation flux

The operational costs related to membrane operation in immersed MBRs are mainly determined by the relationship between the specific gas demand (SGD_m) and operating flux, with the SGD_m being the gas flow rate per unit membrane area (the specific aeration demand, SAD_m, for aerobic systems). This reflects the relationship between the convective flow towards the membrane produced by permeate suction and the back transport induced by the gas sparging and tangential shear at the boundary layer (Liu *et al.* 2003).

In AeMBRs, sustainable or critical fluxes have been reported to increase by increasing gas intensity up to a certain threshold value beyond which no further increase in flux is observed for flat sheet (Guglielmi *et al.* 2008), hollow fibre (Guglielmi *et al.* 2007) and multi-tubular membranes (Le-Clech *et al.* 2006). For instance, Chen *et al.* (2016) reported an increase in critical flux from 23 to 47 LMH for an AeMBR when SAD_m increased from 4.5 to 9.0 $m^3\,m^{-2}\,h^{-1}$. Analysis of full scale immersed AeMBRs indicated an operational SAD_m range of 0.21–0.88 $m^3\,m^{-2}\,h^{-1}$, corresponding to fluxes between 24 and 31 LMH (Verrecht *et al.* 2008).

Increasing membrane flux has been shown to increase fouling rates and decrease the duration of the slow fouling phase in aerobic MBRs under conditions of sub-critical flux operation, prior to the widely reported 'TMP jump' (Pollice *et al.* 2005; Le Clech *et al.* 2006). For example, under sub-critical conditions, a decrease in flux from 10 to 2 LMH caused an exponential decrease in the fouling rate, from 19.8 to 0.46 kPa h^{-1}, and prolonged the time before the TMP jump to up to 8 days (Brookes *et al.* 2006). Results from a pilot scale study suggest that the duration before the TMP jump decreases linearly at fluxes close to the critical flux and that an asymptote exists at a certain flux below which operation can be extended to long filtration cycles (Guglielmi *et al.* 2007).

Research into immersed AnMBR for municipal wastewater treatment (Table 1) has employed a wide range of SGD_m up to 7.2 $m^3\,m^{-2}\,h^{-1}$ (Smith *et al.* 2013). Similar to AeMBR, optimum hydrodynamic conditions have been identified by increasing gas sparging until a threshold is reached, or the sustainable or critical flux is identified (Martin-Garcia *et al.* 2011, 2013; Robles *et al.* 2012). Robles *et al.* (2012) observed a linear increase in critical flux from 12 to 19 LMH when SGD_m was increased from 0.17 to 0.50 $m^3\,m^{-2}\,h^{-1}$.

Operational fluxes for AnMBR (Table 1) between 2.5 and 25 LMH have been reported with the higher fluxes achieved using synthetic wastewater (Wu *et al.* 2009; Gouveia *et al.* 2015a). Similar to AeMBRs, an increase in membrane flux, albeit below the critical flux, also leads to an increased fouling rate in AnMBRs (Vallero *et al.* 2005). An increase in gas sparging intensity appears to be effective in extending membrane operation in both AeMBRs and AnMBRs, but does not obviously enhance permeability. For instance, Weinrich & Grélot (2008) reported sustained permeability for an AeMBR of 600 LMH bar^{-1} for around two months at a flux of 25 LMH and SAD_m of 0.20–0.35 $m^3\,m^{-2}\,h^{-1}$. Robles *et al.* (2013b) reported sustained permeability above 100 LMH bar^{-1} in an AnMBR for over 30 days while operating at a sub-critical flux of 13.3 LMH and SGD_m of 0.33 $m^3\,m^2\,h^{-1}$. Three years of AnMBR operation has been reported, by adopting an operating flux of 12–14 LMH, which sustained TMP between 0.35 and 0.6 bar, without extra physical and chemical cleaning (Gouveia *et al.* 2015b). Overall, the permeability is still lower than for AeMBRs which for full scale municipal wastewater treatment plants is between 150 and 250 LMH bar^{-1} (Judd 2011), even when the quantity of gas provided to the membrane is up four times higher and fluxes between two and three times lower.

Physical and chemical cleaning: reversible, irreversible and irrecoverable fouling

Membrane fouling can be generally classified as external fouling (cake and gel formation) and internal fouling (pore clogging) (Judd 2011; Wang *et al.* 2014). In AeMBRs, cake and gel layer fouling are

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Table 5 | Hydrodynamic conditions and specific energy demand of immersed anaerobic MBRs treating municipal wastewater (real and synthetic)

Source	Reactor type/ Sludge	Material/ Geo./ Pore size μm	Temp °C	MLSS g. L^{-1}	Flux LMH	Filtration cycle			Others	SGD $m^3.m^{-2}.h^{-1}$	U_g m h^{-1}	Gas sparging cycle			Energy consump. kWh m^{-3}	Ref.
						Fl min	R min	BW min				Gas	On min	Off min		
Cru[a]	CSTR/F	PVDF/HF/0.05	33	23	12–19	250s	50s	30s[h]	40 s V + 30 s D[j]	0.17–0.5	–	B	Con.		0.10–0.46	Robles *et al.* (2012)
Cru[a]	CSTR/F	PVDF/HF/0.05	20–33	23	10–13.3	250s	50s	30s[h]	40 s V + 30 s D[j]	0.23	–	B	Con.		0.19–0.26	Robles *et al.* (2013a)
Cru[a]	CSTR/F	PVDF/HF/0.05	17–29	10–25	7–11	250s	50s	30s[h]	40 s V + 40 s D[j]	0.23	–	B	Con.		0.23–0.37	Giménez *et al.* (2014)
Cru[b]	CSTR/F	PVDF/HF/0.04	18.9	21.3	10	Con.	–	–	–	1.3	–	B	Con.		2.10[m]	Ruigómez *et al.* (2016b)
Cru[b]	CSTR/F	PVDF/HF/0.04	18.9	21.3	10	Con.	–	–	–	0	–	–	–		2.30[m]	Ruigómez *et al.* (2016b)
Cru[b]	CSTR/F	PVDF/HF/0.04	23	–	17	8	2	4[i]	–	0.146	–	B	Con.		0.10	Dong *et al.* (2016a)
Cru[b]	CSTR/F	PVDF/FS/140[d]	30	6.4–9.3	11	9	1	–	–	0.11	–	B	Con.		0.10	Lin *et al.* (2011)
Cru[c]	IAFMBR	–/HF/0.4	15–35	–	11.3	Con.	–	–	–	0	–	–	–		–	Gao *et al.* (2014)a, 2014b)
Cru (GA)	CSTR/F	PES/FS/0.038	20	9.5–14.7	7	10	0.5	1	–	–	62	B	Con.		–	Martinez-Sosa *et al.* (2012)
Black	UASB/G	PVDF/Tub/250[d]	37	–	10[f]	8	1	1	–	–	8–16	–	Con.		–	van Voorthuizen (2008)
Black	CSTR	PVDF/Tub/250[d]	37	–	8[f]	8	1	1	–	–	40	–	Con.		–	van Voorthuizen (2008)
Set	UASB/G	PVDF/HF/0.04	10–25	0.1–0.6	6[f]	10	–	1	–	1.17	148	N	10s	10s	0.88	Martin Garcia *et al.* (2013)
Set	UASB/G	PVDF/HF/0.04	10–25	0.1–0.6	6[f]	10	–	1	–	1.17	148	N	1	10	0.16	Martin Garcia *et al.* (2013)
Set	UASB/F	PVDF/HF/0.045	18	5.95[e]	10–14	7.5	5s	15s	5 s R	0.32–0.48[k]	40–60	B	Con.		0.26–0.39	Gouveia *et al.* (2015a)
Set	UASB/G	PVDF/HF/0.045	18	2[e]	8	15	10s	1	10 s R	0.07[k]	9	B	Con.		0.09	Gouveia *et al.* (2015b)
Set	CSTR/F	PVDF/HF/0.08	10–25	7.7	6[f]	10	–	1	–	0.39	277	N	10s	10s	0.29	Martin Garcia *et al.* (2013)
Set	SAF–MBR/F	PVDF/HF/0.03	8–30	–	5.1–6.2[f]	30	5	–	–	0	–	–	–		0.23[n]	Shin *et al.* (2014)
Syn	UASB/G	PVDF/Tub/100[d]	21–24	–	4.2	10	1	–	–	5.29	–	N	1	10	1.13	Cerón-Vivas *et al.* (2012)
Syn	UASB/G	PVDF/Tub/100[d]	21–24	–	5.2	4	1	–	–	5.29	–	N	1	4	2.27	Ceron-Vivas *et al.* (2012)
Syn	CSTR/F	PS/Tub/0.2	55	25–50	16–23	10	–	0.5	–	0.29–1.02[l]	12–42	B	Con.		0.12–0.63	Jeison & van Lier (2006)
Syn	CSTR/F	PS/Tub/0.2	30	25–50	5–21	10	–	0.5	–	0.85–1.70[l]	35–70	B	Con.		0.40–3.35	Jeison & van Lier (2006)
Syn	CSTR/F	PE/FS/0.4	35	–	5–8	Con.	–	–	–	3	–	B	10	5	2.2–3.6	Vyrides & Stuckey (2009)

(*Continued.*)

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Table 5 | Continued

Source	Reactor type/ Sludge	Material/ Geo./ Pore size μm	Temp °C	MLSS g. L^{-1}	Flux LMH	Filtration cycle			Others	SGD $m^3.m^{-2}.h^{-1}$	U_g m h^{-1}	Gas sparging cycle			Energy consump. kWh m^{-3}	Ref.
						FI min	R min	BW min				Gas	On min	Off min		
Syn	CSTR/F	PE/FS/0.4	35	–	5–8	Con.	–	–	–	3	–	B	Con.		3.4–5.4	Vyrides & Stuckey (2009)
Syn	CSTR/F	PE/FS/0.4	–	–	7.2	Con.	–	–	–	2.4	–	B	Con.		3.0	Fox & Stuckey (2015)
Syn	CSTR/F	PES/FS/0.2	15	–	7	30 (240)[g]	–	0.5 (4)[g]	–	7.24	14	B	Con.		9.6	Smith *et al.* (2013)
Syn	CSTR/F	PES/FS/0.2	3–15	–	1.2–12	5	–	1	–	5.8	–	B	Con.		6.5–64.8	Smith *et al.* (2015)
Syn	ARMBR	PE/FS/0.2	30	5.5	11	8	2	–	–	0	–	–	–		0.10[n]	Kim *et al.* (2014)
Syn	ABR + AFMR	PVDF/HF/0.1	35	–	7–10	Con.	–	–	–	0	–	–	–		0.058[n]	Kim *et al.* (2011)

Acronyms: ABR + AFMBR: anaerobic baffle reactor + anaerobic fluidised bed membrane bioreactor; ARMBR: anaerobic rotate disk MBR; AT: ambient temperature; B: biogas; BW: backwash; Cru: Crude; CSTR: continuous stirred tank reactor; F: flocculent sludge; FI: filtration; FS: flat sheet; G: granular sludge; GA: glucose addition; HF: hollow fibre; IAFMBR: integrated anaerobic fluidised-bed; MWCO: molecular weight cut off; N: Nitrogen; PE: Polyethylene; PVDF: Polyvinylidene fluoride; R: relaxation; SAF–MBR: staged anaerobic fluidised membrane bioreactor; Set: Settled; Syn: Synthetic; Tub: tubular; UASB: upflow anaerobic sludge blanket.

[a]Crude: pre–treatment including screening, degritter and grease removal.

[b]Crude: after screening.

[c]Crude – from septic tank.

[d]MWCO (kDa).

[e]Reported MLVSS.

[f]Net flux.

[g]values in the bracket are the second filtration conditions.

[h]backwash 30 s every 50 min.

[i]backwash weekly with chemicals.

[j]40 s ventilation every 50 min, 30 s degasification every 250 min.

[k]ZW–10 module, use membrane cross sectional area of 74 cm2 (Martin Garcia *et al.* 2013).

[l]Assume 50% of the membrane package ratio and membrane mount area is double size of the membrane fibre cross sectional area.

[m]Energy demand provided in the paper.

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Table 6 | Overview of operating conditions and membrane performance in side-stream anaerobic MBRs

Source	Reactor type/ Sludge	Temp °C	Material/Geo./ Pore size μm	Mode	MLSS g L^{-1}	CFV[a] m s^{-1}	U_g^b m s^{-1}	J_c L m^{-2} h^{-1}	Flux L m^{-2} h^{-1}	TMP kPa	Fouling rate mbar min^{-1} [c]	kPa h^{-1} [c]	L m^{-2} h^{-1} d^{-1} [d]	T_{OP} h	Filtration cycle FI (min)	R (min)	BW (min)	Ref.
Ind[e]	CSTR	53–55	–/–/20[i]	Pumped	0.3–3.0[j]	0.2–1.0	–	–	10–70	100–200	–	–	–	4,800		Dep +relax [k]		[1]
Cru	UASB/F	4	PAN/Tub/–	Pumped	–	0.4	–	–	10.5	<50	0.067	0.400	–	<125	Con.	–	–	[2]
Cru	UASB/F	4	PAN/Tub/–	Pumped	–	0.4	–	–	10.5	<50	0.040	0.238	–	<210	10	20s	–	[2]
Cru	UASB/F	4	PAN/Tub/–	Pumped	–	0.4	–	–	10.5	<15	<0.001	0.007	–	2,150	10	–	20s	[2]
Cru	UASB	22	PVDF/Tub/100[i]	Pumped	–	2.25	–	–	45–50	–	–	–	–	–	Con.	–	–	[3]
Cru	CSTR	37	–/–/100[i]	Pumped	0.5–10[j]	3	–	–	9–13	100–200	–	–	0.056	4,080	Con.	–	–	[4]
Set	UASB/G	–	PVDF/Tub/0.03	Pumped	–	0.4–2.0	–	4–41	–	–	–	–	–	–	Con.	–	–	[5]
Set	UASB/G	–	PVDF/Tub/0.03	Gas–lift	–	–	0.02–0.14	4	11–12	–	1–2	6–12	–	–	Con.	–	–	[5]
Set	CSTR/F	–	PVDF/Tub/0.03	Pumped	–	0.4–2.0	–	4–19	–	–	–	–	–	–	Con.	–	–	[5]
Set	CSTR/F	–	PVDF/Tub/0.03	Gas–lift	–	–	0.02–0.14	4	11–12	–	8–25	48–150	–	–	Con.	–	–	[5]
Set	CSTR/G	12–35	PE/Tub/0.1	Pumped	5	0.7	–	–	9–20	–	0.001–0.002	0.004–0.011		–	Con.	–	–	[6]
Syn	UASB/F	–	PVDF/Tub/100[i]	Pumped	–	2	–	–	–	103.4	–	–		<6.7	Con.	–	–	[7]
Syn[f]	UASB	–	PVDF/–/0.22	Pumped	0.3–0.55	0.93	–	50	30	<40	0.008–0.042	0.05–0.25	–	>400	Con.	–	–	[8]
Syn	UASB/F	25	PES/Tub/0.03	Pumped	0.5	1	–	–	12.3	8.7	–	–	–	–	Con.	–	–	[9]
Syn	UASB/F	25	PES/Tub/0.03	Pumped	0.3–0.5	1	–	41–70	12.3	<150	0.001–0.003	0.008–0.021	–	720	3	–	20s	[10]
Syn	UASB/F	15	PES/Tub/0.03	Pumped	0.3–0.5	1	–	34–41	12.3	<450	0.003–0.009	0.017–0.055	–	720	3	–	20s	[10]
Syn[g]	CSTR	55	Ceramic/Tub/0.2	Pumped	–	1–1.5	0.1	–	20	–	–	–	–	–	5	–	20s	[11]
Syn[h]	CSTR	35	Ceramic/Tub/0.2	Pumped	–	2	–	–	14–19	40–50	–	–	–	1,680	Con.	–	–	[12]

(*Continued.*)

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Table 6 | Continued

Source	Reactor type/ Sludge	Temp °C	Material/Geo./ Pore size μm	Mode	MLSS g L^{-1}	CFV[a] m s^{-1}	U_g^b m s^{-1}	J_c L m^{-2} h^{-1}	Flux L m^{-2} h^{-1}	TMP kPa	Fouling rate mbar min^{-1} [c]	Fouling rate kPa h^{-1} [c]	Fouling rate L m^{-2} h^{-1} d^{-1} [d]	T_{OP} h	Filtration cycle FI (min)	Filtration cycle R (min)	Filtration cycle BW (min)	Ref.
Syn	CSTR/F	35	PS/PF/3000[i]	Pumped	15[j]	0.8	–	–	21–75	49	–	–	1.9–9.2	168–240	Con.	–	–	[13]
Syn	CSTR	35	Ceramic/0.2	Pumped	1.6–22	2	–	–	25–65	35	–	–	–	120	Con.	–	–	[14]
Syn	CSTR/F	25	PTFE/Tub/1	Pumped	6–12	0.1–0.2	–	–	5	7–55	–	–	–	–	Con.	–	–	[15]
Syn	CSTR/F	35	PVDF/HF/0.03	Pumped	6	0.1–0.3	–	10–12.5	6	<5	0.200	1.2	–	2,160	9	1	–	[16]

Acronyms: BW: backwash; Cru: Crude; CSTR: continuous stirred tank reactor; F: flocculent sludge; FI: filtration; G: granular sludge; Ind: Industrial; MWCO: molecular weight cut off; Tub: Tubular; PAN: polyacrlonitrile; PE: polyethylene; PF: flat plate; PES: polyethersulfone; PTFE: poly–tetrafluoroethylene; PVDF: Polyvinylidene fluoride; R: relaxation; Set: Settled; Syn: Synthetic; UASB: upflow anaerobic sludge blanket.

Reference: [1] Choo & Lee (1996); [2] An *et al.* (2009); [3] Herrera-Robledo *et al.* (2011); [4] Saddoud *et al.* (2007) [5] Martin-Garcia *et al.* (2011); [6] Fawehinmi (2006); [7] Salazar-Peláez *et al.* (2011); [8] Cho & Fane (2002); [9] Ozgun *et al.* (2015a); [10] Ozgun *et al.* (2015b); [11] Jeison *et al.* (2009); [12] Cadi *et al.* (1994); [13] Harada *et al.* (1994); [14] Beaubien *et al.* (1996); [15] Ho & Sung (2009); [16] Wei *et al.* (2014).

[a]Liquid cross flow velocity.

[b]Gas scouring rate.

[c]Fouling rate under constant flux.

[d]Permeate drop down (fouling rate under constant transmembrane pressure).

[e]Alcohol–distillery wastewater.

[f]Not mention synthetic, but assume according to the influent characteristics.

[g]VFA mixed.

[h]Synthetic wastewater containing starch.

[i]MWCO (kDa).

[j]Reported MLVSS.

[k]Depressurisation (down to about 0.3bar) + relaxation (flow stopping).

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generally considered to govern membrane fouling (Liu *et al.* 2012a). Dominant cake layer resistance has similarly been reported by a number of authors in AnMBR for municipal wastewater (Liu *et al.* 2012b; Martinez-Sosa *et al.* 2012; Ozgun *et al.* 2015b), which is therefore comparable to the general trend for AeMBR. Chu *et al.* (2005) reported that 90% of total resistance for an AnMBR corresponded to cake resistance while only 9% was internal fouling. Ozgun *et al.* (2015b) reported that the relative contribution of cake layer resistance in their AnMBR study decreased with a decrease in operating temperature, which suggests transients in the bulk particulate and colloidal characteristics will influence cake deposition.

Resistance to filtration due to membrane fouling can also be classified as reversible, irreversible or irrecoverable, depending on whether it can be removed physically during operation (by relaxation or backflushing), chemically, or if it remains after chemical cleaning (Judd 2011). In immersed AeMBRs, physical cleaning procedures like relaxation and backwashing have shown to be effective in extending membrane operation compared to continuous filtration reducing the chemical cleaning frequency (Judd 2011). By applying physical cleaning procedures, it is possible to apply fluxes that result in an increase of resistance to filtration as long as the cake layer deposited onto the membrane surface can be removed by relaxation or backflushing. For instance, operational cycles of 10 minutes of filtration followed by 1 minute relaxation at fluxes between 22.3 and 28.5 LMH resulted in fouling rates during filtration cycles of 1.39 and 1.8 mbar min^{-1} respectively, while the irreversible fouling rates were two orders of magnitude lower (Guglielmi *et al.* 2007). Similarly, in AnMBRs, relaxation and backwashing have also been applied and proved to be effective for membrane fouling mitigation (Tables 5 and 6). Smith *et al.* (2013) also showed that a 4 min backwash every 4 h reduced membrane fouling from 0.045 to below 0.001 mbar min^{-1}, with similar observations made for side-stream AnMBR (An *et al.* 2009). A four times decrease in fouling rate was observed following implementation of backwash at double the permeate flux, which indicates backwash flux specification is also important.

Once a tenacious cake layer is formed, it is hard to remove just with physical cleaning which is evidenced by cake layer growth on membranes exposed to high shear flow (Imasaka *et al.* 1993) or high shear in combination with relaxation and depressurization (Choo & Lee 1996). Permeability recovery can be variable following chemical cleaning in both aerobic and anaerobic MBRs, which is illustrative of irrecoverable fouling. Gouveia *et al.* (2015a) reported a chemical cleaning protocol for an AnMBR comprising 1,000 mg L^{-1} sodium hypochlorite (NaClO), which achieved recovery of only 61% of initial permeability. However, more typically only around 7–20% of the total resistance is regarded irrecoverable following incorporation of chemical cleaning policies in AnMBR (Chu *et al.* 2005; Ozgun *et al.* 2015b; Dong *et al.* 2016a).

ANAEROBIC MBR FOR INDUSTRIAL VERSUS MUNICIPAL WASTEWATER TREATMENT

Industrial wastewaters are generally of higher organic strength and comparatively smaller flows than for sewage (van Lier *et al.* 2008). Organics in industrial wastewater are predominantly soluble and readily assimilable (Liao *et al.* 2006); whereas sewage comprises a considerable particle fraction, with a proportion of the organics known to be recalcitrant (Lettinga *et al.* 2001). Due to the smaller flows and higher organic strength, higher methane yield (m^3CH_4 m^{-3} permeate) in industrial applications is sufficient for heating the influent to mesophilic temperatures, which improves biodegradation kinetics, and enables application of higher organic loading rates than for sewage (Lettinga *et al.* 2001; van Lier *et al.* 2008). The combination of a more complex organic feed composition (coarser particle fraction), comprising a large recalcitrant contribution, coupled with insufficient heat to improve biodegradation kinetics, implies that sewage presents the higher fouling potential. The higher methane yield indicates more electrical power is available to industrial AnMBR making

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Table 7 | Biogas production from anaerobic MBRs treating municipal wastewater (real and synthetic)

Source	Volume l		Inf. COD (BOD/OD) mg L^{-1}	SO_4^{2-} mg L^{-1}		COD[a] %	OLR[b] kg COD $m^{-3}d^{-1}$	Temp °C	HRT[c] h		CH_4 %	CH_4 production (Normalised to STP) l CH_4 $gCOD_{rem}^{-1}$[d]	l CH_4 m^{-3}	kWh m^{-3}	Dissolved CH_4 (Normalised to STP) l CH_4 m^{-3}	kWh m^{-3}	Total CH_4[e] kWh m^{-3}	Ref.
Cru[f]	17.7[n]		98–2,600	–	–	97	0.5–12.5	14–25	4–6		53–66	0.02–0.06	25–82	0.10–0.33	20.0[x]	0.080	0.18–0.41	Wen *et al.* (1999)
Cru	34		58–348	–	–	(77–81)	0.3–0.9	4	5.5–10	–	59–64	0.04–0.07[s]	5.5–10.8	0.02–0.04	27.9[x]	0.111	0.13–0.15	An *et al.* (2009)
Cru[g]	1,300	800	445	297	5	87	0.9–3.0	33	6–21		55	0.06[s]	22.8	0.10	17.2[w]	0.069	0.160	Giménez *et al.* (2011)
Cru[g]	1,300	800	445	297	5	87	0.9–3.0	33	6–21		55	0.26[s,t]	–	–	17.2[w]	0.069	>0.160	Giménez *et al.* (2011)
Cru[g]	1,300	800	468–598	300[o]	–	90–94	0.6–1.9	17–29	12–29		–	0.001–0.05	0.7–22.1	0.003–0.088	7.1–10.2[w]	0.03–0.04	0.03–0.12	Giménez *et al.* (2014)
Cru[g]	1,300	800	650	315	–	–	–	17–33	–	–	–	–	1.3–23.6[s]	0.005–0.095	6.7–13.0[w]	0.03–0.05	0.03–0.15	Pretel *et al.* (2014)
Cru[f]	550	80	304–388	50–55	–	88–92	1.1	23	8.5		35	0.04–0.07	14–19	0.06–0.08	13.3[w]	0.053	0.11–0.13	Dong *et al.* (2016b)
Cru[f]	60		342–527	–	–	90	1.0	30	10		75–85	0.22[s]	83	0.331	22.3[x]	0.089	0.420	Lin *et al.* (2011)
Cru[h]	5.8		247–449	–	–	51–74	1.2–1.4	15–35	6		–	0.13–0.17[s]	23.6–43.4	0.09–0.17	–	–	>0.09–0.17	Gao *et al.* (2014a)
Cru[i]	350	–	630 (0.63)	–	–	82,90	0.6–1.1	20,35	14.3		88,80	0.23–0.27[s]	127–149	0.506–0.594	29,21[x]	0.12,0.08	0.62–0.68	Martinez-Sosa *et al.* (2011)
Set[j]	160	150	892 (0.64)	47	–	74–90	0.8–2.6	18	13–17	–	80–83	0.14–0.26	57–285	0.23–1.14	28.6[x]	0.114	0.31–1.25	Gouveia *et al.* (2015a)
Set[k]	160	150	892 (0.64)	47	–	73–90	2.0–4.7	18	7–13	–	83–86	0.09–0.23	86–231	0.35–0.92	28.6[x]	0.114	0.46–1.04	Gouveia *et al.* (2015a)
Set[j]	326	175	978 (0.48)	47	–	75–90	0.6–3.2	18	10–15	–	81–83	0.11–0.19	25–191	0.16–0.76	28.6[w]	0.114	0.21–0.88	Gouveia *et al.* (2015b)
Set	42.5	30	265	–	–	93	0.2–0.6	14	12		–	0.004	1.0	0.004	13.0[w]	0.052	0.056	Cookney *et al.* (2016)
Set	5	1	427	–	–	84–86	1.0	25–30	10		–	0.04–0.10	14–38	0.06–0.15	–	–	>0.06–0.15	Huang *et al.* (2013)
Set[l,m]	0.25	0.25	154 (0.57)	–	–	84	4–6 (1.3)[q]	25	1	1.3	–	–	–	0.03[v]	–	0.05–0.06[v]	0.08–0.10[v]	Bae *et al.* (2013)
Set[l]	0.25	0.25	154 (0.57)	63	0	84	3.5 (1.2)[q]	25	1	1.3	40 (54)[r]	0.05	9.1	0.04	15.8[w]	0.063	0.099	Yoo *et al.* (2012)
Set	990	770	233 (0.50)	41	7	91–93	2.5–3.0	9–25	2	2.6	–	0.09–0.13	17–31	0.07–0.12	12–27[w]	0.05–0.11	0.17–0.18	Shin *et al.* (2014)
Syn	4.7[n]		383–849	–	–	85–96	1.6–4.5	11–25	3.5–5.7		63–72	0.06–0.12	32–66	0.13–0.27	23.5[x]	0.09–0.10	0.22–0.35	Chu *et al.* (2005)
Syn	30[n]		390	–	–	89[p]	0.78	35	12		81.2	–	26.3	0.105	20.8[x]	0.083	0.189	An *et al.* (2010)
Syn	5[n]		440	–	–	92	0.7	15	16		–	0.05–0.13	20–53	0.08–0.21	29.1[w]	0.09–0.14	0.20–0.33	Smith *et al.* (2013)
Syn	3[n]		460	–	–	90–95	0.2–3.7	35	3–48		60–70	0.20–0.29[s]	83–125	0.33–0.50	16.7[x]	0.067	0.40–0.57	Hu & Stuckey (2006)

(*Continued.*)

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Table 7 | Continued

Source	Volume l		Inf. COD (BOD/OD) mg L^{-1}	SO_4^{2-} mg L^{-1}		COD[a] %	OLR[b] kg COD $m^{-3}d^{-1}$		Temp °C	HRT[c] h		CH_4 %	CH_4 production (Normalised to STP) l CH_4 $gCOD_{rem}^{-1}$[d]	l CH_4 m^{-3}	kWh m^{-3}	Dissolved CH_4 (Normalised to STP) l CH_4 m^{-3}	kWh m^{-3}	Total CH_4[e] kWh m^{-3}	Ref.
Syn	2	–	400	–	–	98	0.8–1.6		35	6–12		80–90	0.08–0.12[s]	30–47	0.12–0.19	–	0.055	0.18–0.24	Wei *et al.* (2014)
Syn	4	–	500	60–90	–	>90	1.0–2.0		25	6–12		70–75	0.19–0.20[s]	89–93	0.35–0.37	21[w]	0.084	>0.44–0.46	Ho & Sung (2009)
Syn	3.93	2	513	–	–	99 (88)	4–6	–	35	2–3	–	86	0.18	92[u]	0.368	40[w]	0.159	0.527	Kim *et al.* (2011)
Syn	4[n]		342	–	–	96	–		30	–		68	0.14[s]	–	0.144[v]	19.0[x]	0.076	0.220	Kim *et al.* (2014)

Acronyms: Cru: Crude; Eff: Effluent; FI: filtration section; Inf: Influent; RE: Anaerobic reactor; Set: settled; STP: standard temperature and pressure, 0 °C and 1 bar; Syn: Synthetic.
[a]The values in bracket means the biological reactor removal.
[b]Organic loading rate of anaerobic reactor.
[c]Split tables showed the HRT of biological reactor and membrane tank, combined tables showed the HRT of whole AnMBR.
[d]LCH_4 $gCOD^{-1}$ removed based on CODt influent and CODt permeate.
[e]Assume 1 m^3 CH_4 can generate 10 kWh of energy and CHP efficiency is 40% CH_4.
[f]Crude: after screening.
[g]Crude: pre-treatment including screening, degritter and grease removal.
[h]Crude: from septic tank.
[i]Crude wastewater with glucose addition.
[j]With recirculation.
[k]Without recirculation.
[l]Settled sewage go through 10 μm cartridge filter.
[m]Settled sewage go through 1 mm screen.
[n]Membrane submerged in the anaerobic reactor.
[o]Reported as SO_4^{2-}–S.
[p]TOC removal.
[q]OLR for AFBF (OLR of AFMBR).
[r]CH_4 composition in SAF (membrane tank).
[s]Directly reported from literature.
[t]Calculate the methane yield on CODt used for methanogenesis bacterium by subtracting the CODt removed for sulphate reduction bacterium (Lens *et al.* 1998; Giménez *et al.* 2012).
[u]Only consider the methane from AFBR.
[v]Directly reported the energy production.
[w]Literature reported the dissolved CH_4 (directly test or estimate from Henry's law).
[x]Use Henry's law to for dissolved CH_4 calculation (assume saturation index is 1.00) (Giménez et al. 2012).

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sustainable membrane operation easier to achieve with adoption of high cross-flow velocities in side-stream (Lin *et al.* 2013), and the application of conventional CSTR configured AnMBR with high biomass concentrations (Skouteris *et al.* 2012) commonly reported that can still deliver energy neutral membrane operation. This practice is not readily translatable into sewage applications, subsequently demanding the development of engineered solutions to facilitate energy neutral operation (Ozgun *et al.* 2015a; Wang *et al.* 2018).

ESTABLISHING THE ENERGY PROFILE OF ANMBR FOR MUNICIPAL APPLICATION

Towards energy neutral wastewater treatment

The key facet that drives commercial interest for AnMBR is in the potential to achieve energy neutral wastewater treatment through a reduction in net energy demand for treatment, coupled with an

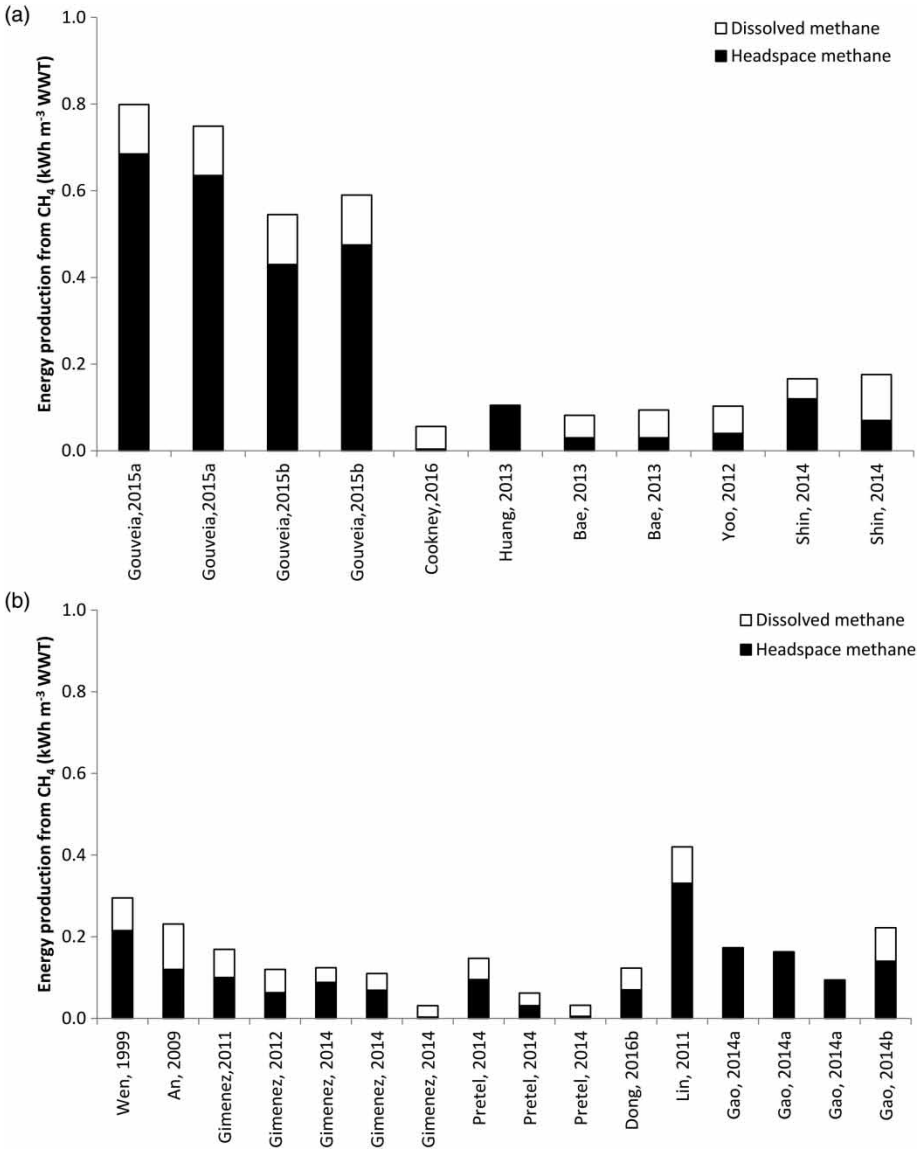


Figure 1 | Combined headspace and dissolved methane production rates from anaerobic MBR treating: (a) settled sewage and (b) crude sewage. Where dissolved methane data was not provided, the dissolved fraction was estimated using Henry's law. Energy data was normalised assuming 40% CHP conversion efficiency.

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increase in total biogas production, without having to rely on sludge imports (McAdam *et al.* 2011). Several flowsheets have been proposed for how to incorporate AnMBR technology. Upstream, the decision must be made as to whether primary sedimentation is introduced to divert particulate COD toward anaerobic digestion, leaving only settled wastewater to be processed by AnMBR (McAdam *et al.* 2011) or whether the full organic load (crude wastewater) is to be treated within the AnMBR (Giménez *et al.* 2011; Robles *et al.* 2013a, 2013b). Downstream of the AnMBR, the decision to have biological or physical separation processes for polishing and nutrient removal is a further consideration (McAdam *et al.* 2011; Sutton *et al.* 2011). The merit of these various approaches is beyond the scope of this review. Nevertheless, it is implicit in each of these cases that the AnMBR itself needs to be energy neutral, or better, energy positive (to provide residual energy for the ancillary processes), if overall energy-neutrality is to be achieved.

Typical headspace methane yield from the literature is between 0.02 and 0.27 L CH_4 . g COD^{-1} for both settled and crude wastewater (Table 7). This is below the expected stoichiometric conversion of 0.35 L CH_4 . g COD^{-1}. The difference between measured and expected values can be explained by: (i) the fact that yield does not take into consideration the solids which are retained (and accumulate) within the bioreactor (Uemura & Harada 2000; Lester *et al.* 2013); (ii) differences in HRT or OLR between literature studies which can influence specific yield, and (iii) the dissolved methane fraction which is released with the AnMBR effluent. Several studies have now demonstrated this dissolved fraction can comprise over 50% of the dissolved methane balance (Cookney *et al.* 2012), with losses exacerbated at lower temperature (Cookney *et al.* 2016). Downstream membrane technology has been demonstrated that can recover this fraction (Bandara *et al.* 2011; Cookney *et al.* 2016) in sufficient concentration for reuse in power generation. Collectively the average dissolved and headspace methane yield from the literature provides around 0.34 kWh m^{-3} of permeate produced (0.80 kWh m^{-3} maximum), which is the typical membrane operation energy consumption of full-scale aerobic MBR (0.19–0.70 kWh m^{-3}). Consequently, AnMBR should be delivered at equivalent or better energetic performance to aerobic MBR to achieve 'energy-neutral' conditions. Comparison of specific energy production with membrane energy demand estimated from published AnMBR literature (Figure 1) demonstrates that several AnMBR studies can achieve membrane operation within 'energy-neutral' conditions (Figure 2). Since the shear promoted at laboratory scale is generally a conservative estimate of that attained at scale (Delgado *et al.* 2004), such evidence is encouraging. While

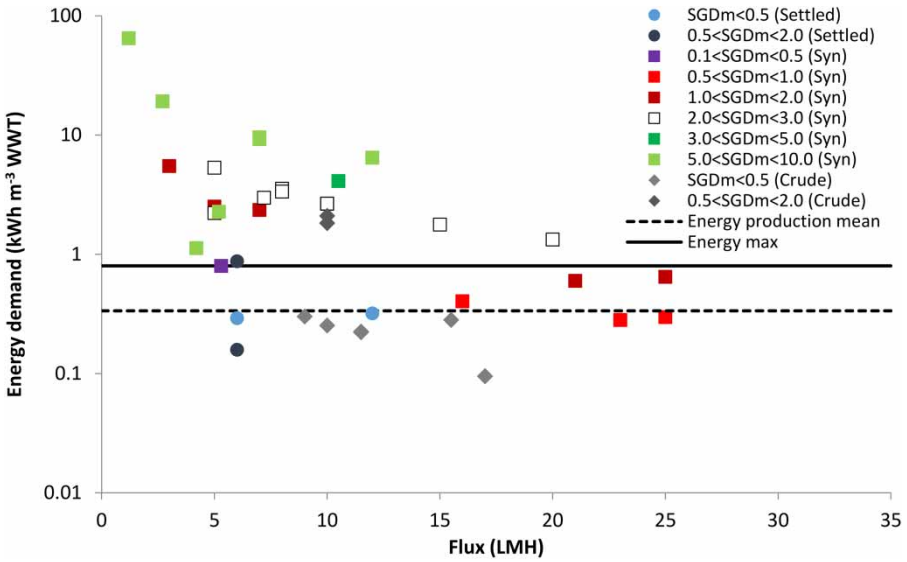


Figure 2 | Comparison of membrane energy demand to the average (0.34 kWh m^{-3}, dashed line) and maximum (0.80 kWh m^{-3}, continuous line) energy production reported in the literature to date for anaerobic MBR treating settled wastewater.

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many studies have sought to optimise hydrodynamics to lower energy (Martin Garcia *et al.* 2013; Seib *et al.* 2016; Wang *et al.* 2018), an explicit focus on achieving energy neutral membrane operation would facilitate further improvements toward this goal.

Emerging engineered solutions to constrain energy demand within AnMBR

Several authors have now directly compared different AnMBR reactor configurations and have identified that when configured as a UASB, membrane fouling and energy demand is constrained (van Voorthuizen *et al.* 2008; Martin-Garcia *et al.* 2011). The authors ascribed the enhanced performance to the lower particle concentration within the downstream membrane tank (Martin-Garcia *et al.* 2011). Martin-Garcia *et al.* (2011) also screened a host of membrane geometries and were able to demonstrate a specific gas demand of 0.3 kWh m^{-3} for immersed hollow fibre membranes compared to 3.7 kWh m^{-3} for externally configured membranes. It has been shown that further reduction in specific energy demand can be attained for immersed membranes through inclusion of novel gas sparging strategies (Cerón-Vivas *et al.* 2012; Martin Garcia *et al.* 2013; Wang *et al.* 2018).

Shin *et al.* (2016) introduced the anaerobic fluidised bed MBR which incorporates granulated activated carbon (GAC) into the reactor to scour the membrane. Significantly, the authors demonstrated sustainable fluxes could be achieved with an immersed membrane without the need for gas sparging, provided a critical upflow velocity could be achieved to sufficiently fluidise the GAC (Wang *et al.* 2017). Kim *et al.* (2011) reported a specific energy demand of 0.028 kWh m^{-3} for this configuration, which evidences a substantive energy reduction when compared to classically commercially configured aerobic MBR technology. One of the future challenges with GAC is to identify a compatible membrane that can withstand the abrasion introduced by the GAC (Shin *et al.* 2016; Wang *et al.* 2017). Ruigómez *et al.* (2016a, 2016b) integrated rotating membrane technology into a CSTR configuration as an alternative method of shear (Zsirai *et al.* 2016). The authors demonstrated an improvement in critical flux with membrane rotation when compared to conventional gas sparging; specific energy demand for disk rotation is around 0.104 kWh m^{-3} (Kim *et al.* 2014).

Within the context of scaling-up AnMBR, membrane cost must also be considered as the sustainable fluxes identified at present of around 10 L m^{-2} h^{-1} (Table 1) are markedly below those nominally achieved in commercial aerobic MBR. For large scale installations, investment decisions are strongly influenced by initial capital investment. As such, there is a need to establish new paradigms that can access cost efficient water productivities whilst still delivering to the energy neutral objective. One innovative example is anaerobic dynamic membrane bioreactors (Alibardi *et al.* 2014; Ersahin *et al.* 2016) which employ woven and non-woven filter substrates as a low cost alternative (less than 13€ m^{-2} (Ersahin *et al.* 2016)) to conventional polymeric membranes. Mesh sizes of 10–200 μm have been tested which provide substantial clean water permeability. The developed cake provides the rejection (Alibardi *et al.* 2014) with >99% COD rejection noted for several prospective applications, coupled with sustainable fluxes, which evidences their considerable potential to making AnMBR both capitally and energetically efficient (Alibardi *et al.* 2014; Ersahin *et al.* 2016).

CONCLUSIONS

The present literature review reveals significant differences with respect to biomass characteristics and fouling behavior between aerobic and anaerobic MBRs which can be summarized as follows:

1. The ratio of proteins to carbohydrates in eEPS is higher in AnMBRs than in AeMBRs. However, the total eEPS appears slightly higher in aerobic MBR. There is a general lack of knowledge regarding the relationship between eEPS concentration and composition and surface properties such as charge and hydrophobicity for AnMBR.

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2. Although similar median particle sizes have been reported in aerobic and anaerobic MBRs, the presence of a population of fine solids with particle sizes ranging from 1 to 10–15 microns have been widely reported in the latter and associated to biomass of high fouling propensity.
3. The SMP_{COD} concentration contained within bulk sludge is an order of magnitude higher in AnMBR than in AeMBR. The main operational parameters that have shown to enhance SMP production in AnMBRs are low temperature and extended SRT.
4. The effect of turbulent gas sparging on membrane performance in immersed AnMBRs seems to be limited as compared to aerobic systems, indicating that fouling is more determined by sludge properties than by membrane operational conditions
5. AnMBR fluxes are between one-third and one-half of those reported in AeMBRs. Reported permeabilities in AnMBR are around 50% below those of AeMBR and employ specific gas demands between 50 and 300% higher.
6. Although as with aerobic systems the predominant fouling mechanism in AnMBRs has been reported to be cake filtration, contradictory results with respect to the effectiveness of membrane backwashing at reducing membrane fouling and permeability recovery after chemical cleaning have been reported and would require further research.
7. Whilst membrane fouling remains a critical challenge, several research groups have already demonstrated the potential to achieve energy self-sustained conditions at pilot scale. Whilst AeMBR is comparatively mature, continued investment in commercial module and aeration (gas scouring) engineering is still enabling radical reductions in specific energy demand to be realised at scale. Consequently, the potential to achieve energy neutral wastewater treatment at a commercial scale appears inherently viable.

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Comparison of fouling between aerobic and anaerobic MBR treating municipal wastewater

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