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Current status and future perspectives

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DOI
10.1080/10643389.2020.1780879

Publication date
2020

Document Version
Final published version

Published in
Critical Reviews in Environmental Science and Technology

CRITICAL REVIEWS IN ENVIRONMENTAL SCIENCE AND TECHNOLOGY
https://doi.org/10.1080/10643389.2020.1780879

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Anaerobic membrane bioreactors for sludge digestion: Current status and future perspectives

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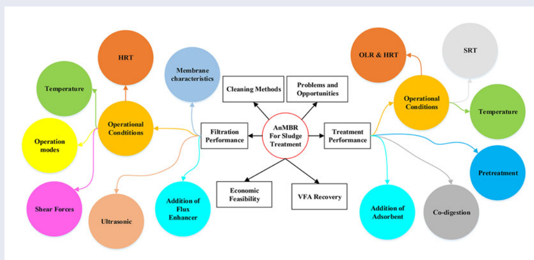
ABSTRACT



Excess sewage sludge in wastewater treatment plants (WWTPs) is regarded the key energy source for achieving energy neutral WWTPs. The anaerobic digestion process transforms sludge-organic matter into methane, which subsequently can be used for heat and electricity production. Conventional anaerobic digesters (ADs) have been used for sludge treatment for many decades, requiring high energy and providing poor effluent quality. Anaerobic membrane bioreactor (AnMBR) technology exhibits a promising option for treatment of high solids concentration streams including sludge. AnMBRs result in an increase in digestion efficiency and enhancement in effluent quality at small footprints. AnMBRs have the potential to reduce capital and operational costs, and produce more energy in comparison to conventional ADs. Thus, energy neutral or positive operation can be achieved with AnMBRs. Besides, nutrient recovery or direct use of permeate will become more feasible in AnMBRs compared to use of sludge supernatant in ADs. However, membrane fouling can limit the feasibility of AnMBRs for sludge treatment, which requires further research. This review paper critically evaluates the current status of AnMBR technology for municipal sludge treatment discussing the effect of different factors on treatment and membrane filtration performances. Furthermore, future research opportunities to enhance applicability of this technology are addressed.

KEYWORDS Anaerobic membrane bioreactor; energy; fouling; sewage sludge; sludge treatment

1. Introduction

Treatment of municipal wastewater in conventional activated sludge processes generates large volumes of biosolids due to settling processes and due

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to microbial anabolism during the conversion of organic pollutants. The amount of sludge produced from wastewater treatment plants (WWTPs) varies widely based on applied treatment technology and operational conditions. Almost 10 million tons of dry sludge per year is generated in the EU. This quantity is based on a daily dry solids (DS) production of 4–87 g $DS{\cdot}pe^{-1}{\cdot}d^{-1}$ (Eurostat, 2015). Moreover, WWTPs in the United States generate more than 6.5 million tons of dry sludge per year, followed by China and Japan with around 3 and 2 million tons, respectively, according to UN-Habitat's statistics (LeBlanc et al., 2008).

Municipal wastewater sludge contains organic matters such as proteins, fats and grease, and cellulose; nutrients such as nitrogen, phosphorus, potassium, heavy metals such as iron and chromium, and organic micropollutants (OMPs) such as polycyclic aromatic hydrocarbons and polychlorobiphenys. Primary sludge extracted from primary settling tanks, and secondary sludge from activated sludge processes have distinct compositions. Primary sludge has a higher content of fats, grease and cellulose; and a lower content of proteins, phosphorus and nitrogen in comparison to secondary sludge (Fytili & Zabaniotou, 2008; Smith et al., 2009). Microorganisms included in the secondary sludge are primarily bacteria, protozoa, rotifers and filamentous organisms.

Sludge stabilization is generally required for reducing the environmental risks related to sludge disposal, such as uncontrolled methane emissions and human health concerns due to pathogens. Biological stabilization, especially anaerobic digestion, is one of the most commonly used methods for decreasing the organic matter content of wastewater treatment sludge worldwide (Luduvice, 2007). The anaerobic digestion process transforms organic matter into methane, reduces final solids and pathogenic microorganisms in the absence of molecular oxygen. Several types of anaerobic digesters (ADs), from standard rate (unheated) digesters to egg shaped high rate digesters (heated and mixed), have been developed to improve the organic matter removal efficiency and methane production (Appels et al., 2008; van Lier et al., 2015). The treatment bottleneck for anaerobic digestion happens during the hydrolysis stage, where reaction rates limit the overall process performance, especially for low temperature ($<20\,^{\circ}C$) operation conditions (Ozgun, Tao, et al., 2015; Ozgun et al., 2019). Various pretreatment methods have been adopted to improve hydrolysis and to increase the overall digestion efficiency and methane production (Appels et al., 2008; Anjum et al., 2016; Carrère et al., 2010). Energy costs associated with pretreatment may possibly be met with the energy of additional methane produced during anaerobic sludge stabilization (Ruffino et al., 2015). Methane can be used for heat and electricity production, and can serve as an important asset for improving the energy efficiency of WWTPs.

Energy from methane can potentially increase the energy autonomy of municipal WWTPs. A current sustainability goal of municipal wastewater treatment is to achieve energy neutral and ultimately net energy positive operation (Guven et al., 2019; McCarty et al., 2011; Ozgun, Gimenez, et al., 2015), which will have a positive impact on operating costs. Currently, management of excess sludge including treatment and disposal is a big challenge in WWTPs and up to 50% of operating costs can be related to treatment and disposal of excess sludge (Appels et al., 2008; Campos et al., 2009). For this reason, the efforts are focused on reducing the costs of sludge management in wastewater treatment sector. Anaerobic digestion is already beneficial in this regard, by providing potential for energy production and associated operational costs reduction. However, due to large reactor footprints, low hydrolysis and biogas production rates, difficulties in stable process operation, and prethickening requirements to reduce reactor sizes and to increase digestibility (Dagnew et al., 2012), there is still room for improvement of conventional anaerobic sludge digestion processes.

Anaerobic membrane bioreactors (AnMBRs) have been developed to improve the efficiency of conventional anaerobic processes and to overcome the problems related with biomass washout due to poor granulation, e.g., for treatment of wastewaters with extreme properties such as high temperature, high salinity, etc. (Dereli et al, 2012; Ozgun, Dereli, et al., 2013; Wang et al., 2017). AnMBRs can be operated at long solids retention times (SRTs) independent from the hydraulic retention time (HRT), whereby slow growing active methanogenic biomass is kept inside the reactor by physical membrane separation (Skouteris et al., 2012). AnMBRs are resistant to inhibitory or toxic substrates, have high transformation capacity of carbonated organic compounds into biogas, have small footprints, and provide high effluent quality (Dvořák et al., 2016; Ozgun et al., 2013).

Anaerobic sludge digesters are generally flow through mechanically-mixed reactors with theoretically equal HRT and SRT. In order to keep the SRT and HRT high enough to maintain methanogenic biomass inside the reactor and to achieve sufficient volatile solids (VS) reduction, sludge digesters are built with significantly high reactor volumes. As a result, sludge digesters in municipal WWTPs are generally low loaded reactors with typical organic loading rates (OLRs) of 1–3 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$ (Verstraete & Vandevivere, 1999). AnMBR is a promising technology for the treatment of wastewaters/slurries with high particulate matter content, such as municipal sludge. Since SRT can be decoupled from HRT, AnMBRs can be operated at lower volumes resulting in a decrease in capital costs and heat losses. Thus, based on 20 years' project life estimations, that was conducted in the study of Pillay et al. (1994), capital and total project costs of AnMBR can be decreased by 27 and 12% compared to conventional AD. AnMBRs can efficiently increase SRT of digesters, enabling better conversion efficiencies to methane, which can be used to generate extra energy to balance operational energy consumption. Net energy demand can be decreased by about 37.3% with AnMBRs in comparison to conventional ADs. Therefore, energy neutral or positive operation can be achieved with AnMBRs (Yu et al., 2016). At the same time, AnMBR produces suspended solids (SS) and pathogen free permeate (Hafuka et al., 2019; Liao et al., 2006). Thus, nutrients recovery or direct use of permeate for irrigation purposes can become more feasible compared to conventional AD supernatants. However, membrane fouling can be one of the most critical limitations that may affect the feasibility of the operation of AnMBR. An important question is whether AnMBRs can achieve these targets and maintain enough level of filtration efficiency at the same time, in order to provide a feasible process for sludge digestion.

Although municipal sludge digestion with AnMBRs presents many opportunities and has the potential to serve as a competitive technology for achieving energy sustainability of WWTPs, there is still little information about this topic in the literature. There is no single source addressing questions linked to performance, operational conditions and challenges related to AnMBRs for sludge treatment. Thus, the aim of this review is to critically evaluate the current research situation of AnMBRs for wastewater sludge treatment, present the performance of AnMBRs treating sludge, and evaluate the effects of different parameters for bioreactor operation and membrane filtration. Moreover, problems encountered, and future opportunities are also addressed throughout the study.

2. Historical development, materials, and configurations

2.1. Historical development

The research on the AnMBR process began in 1970s by Grethlein (1978) and the attention toward AnMBRs has started to increase sharply since 2005. The reasons for the recently increased scientific interests in AnMBRs might be ascribed to decreasing membrane prices, the development of new membranes with good anti-fouling properties, the gradual increase in energy prices, water reuse initiatives, and the more stringent discharge standards (Lin et al., 2013; Ozgun, Dereli, et al., 2013). The number of scientific articles about AnMBRs has showed an exponential increase since 2005 and it is noted that 93% of the whole literature was developed in the last decade, as shown Figure 1. The number of articles, that focused on sludge treatment, were 54 out of 817. Figure 1 was drafted by using

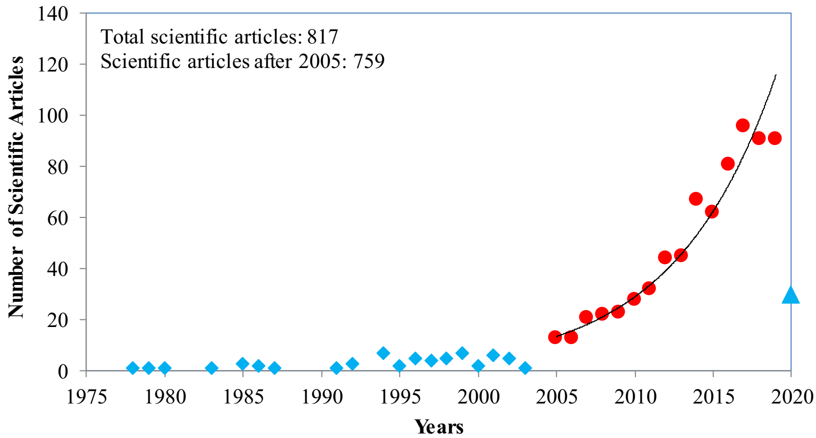

Figure 1. Number of scientific articles about AnMBRs (published between 1978 and 2020).

SCOPUS database and searching with keywords “AnMBR”, “anaerobic”, “membrane” and/or “filtration” in the article title. The results were further checked manually to refine the relevance.

The first study on AnMBR application for sewage sludge digestion was conducted in the end of 1980s, which was reported by Bindoff et al. (1988). After this study, till 2000s, only a few studies were conducted in the focus of coupling AD with membranes for digestion of sewage sludge (Kayawake et al., 1991; Pillay et al., 1994), waste activated sludge (WAS) (Takashima et al., 1996) and primary sludge (Ghyoot & Verstraete, 1997) (Figure 2a). In 2000s, the research started to focus on using AnMBR for volatile fatty acids (VFAs) recovery from sludge (Figure 2b). VFAs can be used as carbon source for biological nitrogen and phosphorous removal processes in wastewater treatment systems (Kim & Jung, 2007; Kim & Somiya, 2001a; Kim, Somiya, et al., 2002). Jeong et al. (2007) used recovered VFAs from WAS fermentation to produce hydrogen gas by using a photosynthetic reactor (Figure 2c).

Since 2008, AnMBR applications have attracted remarkable attention mainly for methane production by WAS digestion. Different operational conditions of AnMBRs were examined to specify the optimum values and figure out the limitations of the technology (Dagnew et al., 2012; Hafuka et al., 2019; Meabe et al., 2013; Wandera et al., 2018). Addition of adsorbents and sludge pretreatment were examined to improve the digestibility and/or filterability of sludge (Joshi & Parker, 2015; Martin-Ryals et al., 2017; Yu et al., 2015). Furthermore, some researchers coupled ultrasound application with AnMBR (Xu et al., 2011, 2013). Sludge was also co-digested with some other wastes (e.g., coffee grounds, coffee processing wastewater) in AnMBR in the studies of Qiao, Takayanagi, Shofie, et al. (2013) and Chen et al. (2019).

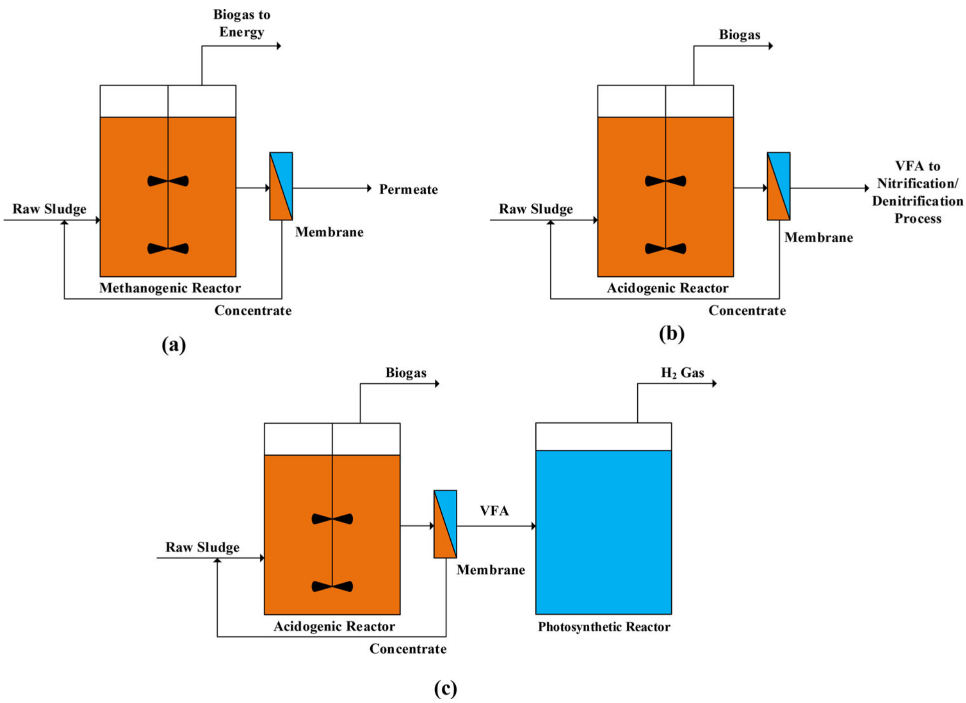

Figure 2. Alternative products obtained with AnMBR during sludge treatment: (a) Biogas, (b) VFA and (c) H_2 gas.

2.2. Materials

Different membrane materials; such as, organic (polymeric), inorganic (ceramic) and metallic membranes, have been widely used in AnMBR applications. Polymeric membranes were commonly used in AnMBRs for sludge digestion, which might be related to the lower costs of polymeric membranes compared to metallic and ceramic ones (Lin et al., 2013). Polyvinylidene difluoride (PVDF) and polyethersulfone (PES) are the most preferred polymeric membrane materials, which account for approximately 75% of total market products (Santos & Judd, 2010). Ceramic membranes provide high tolerance to corrosion and abrasion, whereas some authors claim that ceramic membranes had less irreversible fouling comparing to polymer ones (Lin et al., 2013; Murić et al., 2014). Metallic membranes have some advantages over polymeric membranes, such as higher strength, durability and resistance to oxidation and high temperature (Kim & Jung, 2007). Dynamic membranes (DMs) were also used in AnMBR applications for sludge treatment (Ersahin et al., 2012; Kooijman et al., 2017; Liu et al., 2016; Pillay et al., 1994; Yu et al., 2014, 2016). DM is a secondary filter layer constituted on a porous support material, such as mesh or filter cloth, during filtration of a solution containing SS (Ersahin et al., 2013). DM layer has an important role in particles rejection in anaerobic dynamic membrane bioreactors (AnDMBRs) (Ersahin et al., 2014). Up to now, PVDF (Zheng et al., 2018), PES (Ghyoot & Verstraete, 1997; Martin-Ryals et al., 2020), polysulfone (Joo et al., 2016; Kim et al., 2017; Liew Abdullah et al., 2005), polythene (Li et al., 2015; Qiao, Takayanagi, Shofie, et al., 2013; Xu et al., 2010, 2011), polytetrafluoroethylene (Hafuka et al., 2016, 2019), ceramic (Kayawake et al., 1991; Kim & Chung, 2012; Kim et al., 2005; Kim & Somiya, 2001a, 2001b; Kim, Somiya, et al., 2002; Meabe et al., 2013) and stainless steel (Kim & Jung, 2007) membranes, and DMs using different support materials such as nylon mesh (Joo et al., 2016), Dacron meshes (Yu et al., 2014, 2016), silk filter (Liu et al., 2016), woven fabric filter (Kooijman et al., 2017) and woven fiber filter (Pillay et al., 1994) have been used for solid-liquid separation in AnMBRs treating sludge. Most of the membrane types applied in AnMBRs for sludge filtration were ultrafiltration (UF) and microfiltration (MF) membranes with a configuration of tubular, hollow fiber, flat sheet or monolithic. Zhao et al. (2019) tested flat sheet forward osmosis (FO) membrane in an AnMBR for WAS filtration and used magnesium chloride as a draw solution.

2.3. Configurations

AnMBR configurations can be classified into two major categories: External cross-flow and submerged ones. The membrane module can be located in the bioreactor or in an external chamber. These configurations can be applied on single phase or two-phase: acidogenesis and methanogenesis (Liao et al., 2006). Single phase AnMBRs with external membrane configuration have been used in most of the studies, which is probably due to easiness of cleaning or replacement of the membrane as well as obtaining high fluxes (Lin et al., 2013). Some studies have applied submerged configurations in single phase AnMBRs (Wandera et al., 2018; Yu et al., 2016). External (Joo et al., 2016) and submerged (Martin-Ryals et al., 2020) configurations have also been examined in two-phase AnMBRs. Jeong et al. (2007) tested a membrane module at both external and submerged configurations. Biogas can be recirculated to control membrane fouling in different AnMBR configurations (Dagnew et al., 2013; Wandera et al., 2018). Further details regarding the implications of different reactor configurations on filtration performance are discussed in section 3.2.1.

3. Factors affecting treatment and filtration performances of AnMBRs for sludge treatment

Efficient sludge treatment and optimization of AnMBR need better understanding of factors affecting the biological and treatment performance (Figure 3).

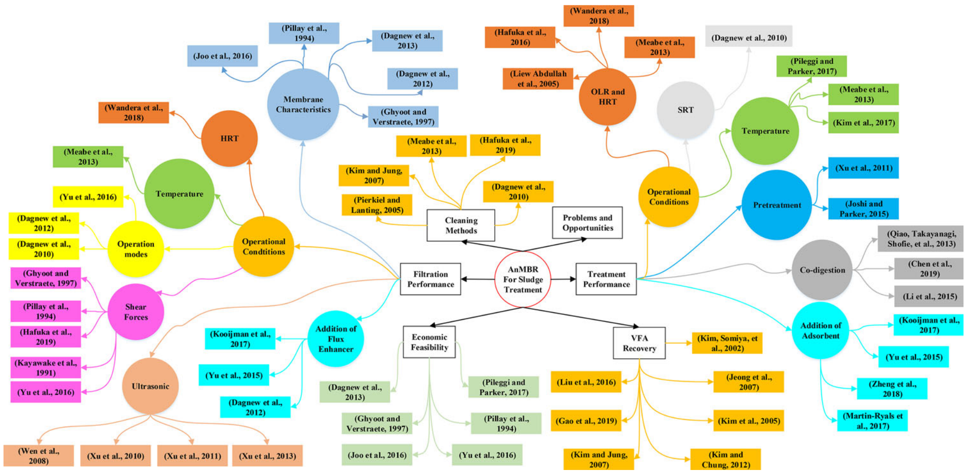

Figure 3. Mapping content of the study.

3.1. Factors affecting treatment performance

In this section, the influence of various factors including design parameters and operational conditions such as temperature, SRT, HRT and OLR, sludge pretreatment, co-digestion as well as addition of adsorbents on biological performance of AnMBRs are further discussed. Table 1 presents the treatment performance of different AnMBR systems for sludge treatment described in the literature.

3.1.1. Operational conditions

3.1.1.1. Temperature. Biochemical reaction rates and degradation efficiency of organic matter are dependent on temperature. Hydrolysis of particulate matter is the rate limiting step in sewage sludge digestion. Increasing operational temperature improves the hydrolysis rate and increases the solubilization of the organic compounds (Appels et al., 2008). Sludge digestion at thermophilic temperatures (~55 °C) have some advantages over mesophilic digestion (~35 °C); such as higher reaction rates, higher organic load capacity and higher destruction of pathogens. On the other hand, the process becomes more vulnerable to instability and inhibition as a result of VFA accumulation due to increased rate of acid production (Kim, Ahn, & Speece, 2002). Pileggi and Parker (2017) investigated the treatment of mixed primary sludge and thickened WAS, and evaluated the performance of AnMBR at ambient, mesophilic and thermophilic temperatures at the same OLR. Specific methane production (SMP) increased by increasing the temperature. SMP at 25 °C, 35 °C, and 55 °C were 0.19, 0.28, and 0.34 Nm^3 $CH_4 \cdot kg$ VS_{fed}^{-1}, respectively. In the study of Kim et al. (2017), the temperature was elevated from mesophilic to

Table 1. Treatment performance of AnMBRs for sludge treatment.

Sludge type	Reactor type/module configuration	Reactor volume (L)/temperature (°C)	OLR (kg COD·m^{-3}·d^{-1})	HRT (d)	SRT (d)	Influent TS (g·L^{-1})	VS removal (%)	COD removal (%)	Biogas production rate (m^3·m^{-3}·d^{-1})	CH_4 content (%)	Reference
Heat treated liquor of sewage sludge	CSTR/Submerged	200/37	4–16	0.5–2	—	—	—	79.3–83.2	2.13–7.1	63–66	Kayawake et al. (1991)
Sewage sludge	CSTR/External	1800/—	—	14	26	—	—	—	—	—	Pillay et al. (1994)
WAS	CSTR/External	5/35	—	30	100	20.7	—	—	—	57	Takashima et al. (1996)
WAS	CSTR/External + alkaline heat post-treatment	5/35	—	30	100	20.7	—	—	—	71	Takashima et al. (1996)
Primary sludge	Up-flow anaerobic bioreactor/External	120/35	0.88–1.2	20	—	2.2–35	25–57	29–54	0.15–0.26*	63–64	Ghyoot and Verstraete (1997)
sewage sludge	CSTR/External	50/—	0.1–10	7.8–943.4	16.1–1250	—	—	96.5–98.8	—	66.3–76.3	Liew Abdullah et al. (2005)
Sewage sludge	CSTR/External	550/35	—	1.7–11.8	4.2–70.5	—	59	—	—	—	Pierkiel and Lanting (2005)
WAS	CSTR/External + US	—	—	—	—	3.5	—	—	—	—	Wen et al. (2008)
WAS + Thickened WAS	CSTR/External	540/35	1.34	15	30	19.4	48	—	—	—	Dagnew et al. (2010)
WAS	CSTR/External	8/37	—	6	80	—	44.2–63.2	—	—	—	Xu et al. (2010)
WAS	CSTR/External + US	8/37	—	6	80	—	52.1–72.8	—	—	—	Xu et al. (2010)
WAS	CSTR/External	2.4/35	1.5–3.7[a]	9	40	6.2–18.8	42.8–50.7	—	—	—	Xu et al. (2011)
WAS	CSTR/External + US	2.4/35	1.5–3.7[a]	9	40	6.2–18.8	45.7–51.3	—	—	—	Xu et al. (2011)
Thickened WAS	CSTR/External	530/35	1.47–1.59*	15	30	16–20	44.6–47.6	48.2–49.1	—	—	Dagnew et al. (2013)
Sewage sludge	CSTR/External	25/35	4.8	5–7	30, 50	32.7	—	99	1.63*	67.5	Meabe et al. (2013)
Sewage sludge	CSTR/External	25/55	4.8–6.4	3–7	50	32.7	—	94	1.76*	67.5	Meabe et al. (2013)
WAS	CSTR/External + US	2.4/35	3.7[a]	3	—	20	—	—	—	—	Xu et al. (2013)
WAS + coffee grounds	CSTR/Submerged	7/55–57	2.2–11.8	7–70	7–100	100, 150	—	60.1–67.4	—	—	Qiao, Takayanagi, Shofie, et al. (2013)
WAS + coffee grounds	CSTR/Submerged	7/55–57	2.2–33.7	7–70	7–100	100, 150	—	46.5–66.8	0.53–5.8	51.5–60.1	Qiao, Takayanagi, Niu, et al. (2013)
WAS	CSTR/Submerged	5/35	—	3	20	7.06	—	49	—	—	Joshi and Parker (2015)
WAS + pretreatment (20 min US)	CSTR/Submerged	5/35	—	3	20	7.06	26[b]	58	—	—	Joshi and Parker (2015)
WAS + pretreatment (60 min US)	CSTR/Submerged	5/35	—	3	20	7.06	48[b]	63	—	—	Joshi and Parker (2015)
WAS + coffee grounds + milk waste	CSTR/Submerged	7/55–57	3.98–14.6	8.5–30	17–60	69.6	—	94–95.1	1.51–3.91	61–61.9	Li et al. (2015)
WAS + coffee grounds + milk waste + sulfate addition	CSTR/Submerged	7/55–57	4.06–15.2	8–30	16–60	69.6	—	93.9–95.8	1.39–4.68	61.1–61.3	Li et al. (2015)
MBR excess sludge	CSTR/External	20/35	1.3–2.2[c]	34, 67	700	15	—	98	0.1–1.3	64.5–68.5	Hafuka et al. (2016)
Sewage sludge	Two-phased CSTR/External	2[d], 4[e]/35	13.7*	—	4[d], 20[e]	9.8	<46	97	—	55–60	Joo et al. (2016)
WAS	CSTR/Externally submerged	67/35	—	5	20	—	50.8[b]	—	0.15	72	Yu et al. (2016)
Sewage sludge	Two-phased CSTR/External	2[d], 4[e]/35[d], 55[e]	1.1*	—	4[d], 20[e]	2.85	—	97.9	—	55–65	Kim et al. (2017)
WAS	CSTR/External	7/35	—	18	24	48.2	32	—	—	—	Kooijman et al. (2017)
WAS	CSTR/External + coagulant + flocculant aid	7/35	—	18	24	48.2	24	—	—	—	Kooijman et al. (2017)
Primary sludge + TWAS	CSTR/External	500/25.7–54.5	2.1–3.7	6.9–14.5	22–39.4	25–30	49–64	—	—	—	Pileggi and Parker (2017)
Primary sludge	Two-phased CSTR/Submerged	1.5,12/37	1.86	2,16	2,30	24.2–34	—	52.4	—	71.2	Martin-Ryals et al. (2020)

(*continued*)

Table 1. Continued.

Sludge type	Reactor type/module configuration	Reactor volume (L)/temperature (°C)	OLR (kg $COD \cdot m^{-3} \cdot d^{-1}$)	HRT (d)	SRT (d)	Influent TS ($g \cdot L^{-1}$)	VS removal (%)	COD removal (%)	Biogas production rate ($m^3 \cdot m^{-3} \cdot d^{-1}$)	CH_4 content (%)	Reference
Primary sludge	Two-phased CSTR/Submerged	1.5[d], 12[e]/37	1.76–2.44	2[d], 16[e]	2[d], 30[e]	24.2–34	—	65.5–81	—	68–76.4	Martin-Ryals et al. (2020)
Thermal hydrolyzed sewage sludge	CSTR/Submerged	9/37	4.13–20.43*	3–30	6–60	47–82.6	48.4–66.7*	39.9–67.9*	0.88–3.43	—	Wandera et al. (2018)
coffee processing wastewater + WAS	CSTR/Submerged	6/55	0.87–9.18	5–50	—	41.8	—	92	—	72–82.4	Chen et al. (2019)
WAS	CSTR/External	10/35	0.15–0.55[a]	15–61	53–250	5–10	—	23–56	0.02–0.05	—	Hafuka et al. (2019)
Thermal hydrolyzed sewage sludge	CSTR/Submerged	7/37	1.39–5.72	5–20	10–40	4.5–4.9	46–63	79–96.3	0.43–1.3	70–78	Wandera et al. (2019)
WAS	CSTR/Submerged	1/35	1.1–1.2[a]	17	40–50	6	57.3	—	0.15–0.18	60–70	Zhao et al. (2019)

[a]kg $VS \cdot m^{-3} \cdot d^{-1}$.
[b]VSS removal.
[c]kg $COD \cdot m^{-3}$ and feeding of sludge was twice a week.
[d]Acidogenic reactor.
[e]Methanogenic reactor.
*Calculated by using the reported data.

thermophilic condition in only methanogenic reactor of two-phase AnMBR treating sewage sludge. They observed an improvement in the methane production, which increased from 0.48 to 0.57 m^3 CH_4·kg VS_{fed}^{-1}. They reported that the increase in methanogenic activity after elevation of the temperature resulted in enhanced organic removal efficiency in the methanogenic reactor.

Meabe et al. (2013) investigated the performance of an AnMBR treating sewage sludge under mesophilic and thermophilic conditions at 50 d of SRT and 7 d of HRT. A similar digestion efficiency was observed in each condition, and around 72% of chemical oxygen demand (COD) was converted to biogas. However, permeate quality deteriorated under thermophilic condition due to the increased solubilization rate, which led to an increase in the concentration of soluble COD (sCOD), VFA and ammonia passing through the membrane. This resulted in a brown color in the permeate, while the permeate of the AnMBR operated under mesophilic condition was light yellow. In addition, Meabe et al. (2013) claimed that, operating the system at an SRT of 50 d allowed the biomass in the system to acclimate, which led to high biodegradation efficiency. They reported no inhibition due to increase of ammonia concentration during their study.

3.1.1.2. Sludge retention time. AnMBR technology has an advantage over conventional anaerobic sludge digesters due to the decoupling of SRT and HRT. Therefore, AnMBR process achieves similar or even better (Dagnew et al., 2010) digestion performances with significantly lower (<50%) reactor volumes compared to the conventional counterparts. The independent increase in SRT apart from HRT in AnMBRs allows increased retention of active biomass and particulates in the bioreactor, resulting in better hydrolysis of particulate organic matter (Liao et al., 2006).

In order to explore how decoupling of SRT and HRT affects treatment performance, Dagnew et al. (2010) compared the treatment performance of a pilot-scale AnMBR and two bench-scale conventional ADs (continuous stirred tank reactors [CSTRs]) for the treatment of a mixture of WAS and thickened WAS. The AnMBR was operated at SRT of 30 d and an HRT of 15 d and the ADs were operated at HRT = SRT of 15 d and 30 d. An increase was observed in VS destruction from 35.3% at an SRT of 15 d to 44% at an SRT of 30 d for conventional ADs. The AnMBR exceeded the performance of each AD system by achieving 48% of VS destruction. SMPs were obtained as 0.21, 0.28, and 0.32 m^3 CH_4·kg VS_{fed}^{-1} for (AD with SRT of 15 d), (AD with SRT of 30 d), and AnMBR, respectively. The highest SMP belonged to the AnMBR, and the lowest SMP was obtained in the AD unit with the shorter SRT.

3.1.1.3. Organic loading rate and hydraulic retention time. AnMBRs treating sewage sludge were operated at a wide range of OLRs (Table 1). However, reported OLRs were significantly lower than the OLRs reported by the studies on AnMBRs treating industrial wastewater (Dereli et al., 2012), which indicates that there may still be room for optimization of the process performance. Liew Abdullah et al. (2005) investigated the treatment of sewage sludge by an AnMBR at different OLRs (0.1–10 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$). It was observed that biogas yield increased from 0.28 to 0.81 $m^3{\cdot}kg\ COD^{-1}{\cdot}d^{-1}$, after increasing OLR from 0.1 to 10 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$, respectively. Increase in OLR resulted in a slight decrease of COD removal efficiency, which was still higher than 96% thanks to the membrane keeping all the particulate organics in the reactor. It was recommended that the SRT should be increased in order to make the system more tolerant to higher OLRs (Liew Abdullah et al. 2005).

For the same substrate concentration, lowering the HRT of an AnMBR will be accompanied by the increase in OLR. Higher OLR and lower HRT lead to a reduction in reactor volume, thus, reduce the capital costs. However, lowering HRT under certain values may cause some operational problems and may affect the system performance (Kim, Somiya, et al., 2002; Meabe et al., 2013; Wandera et al., 2018). Hafuka et al. (2016) investigated the treatment of aerobic membrane bioreactor excess sludge in an AnMBR operated at 35 °C and infinite SRT. It was reported that a decrease in HRT from 67 to 34 d (increasing OLR from 1.3 to 2.2 kg $COD{\cdot}m^{-3}$) led to an increase in the biogas yield from 0.03 to 0.08 $m^3{\cdot}kg\ COD_{fed}{}^{-1}$. Low biogas yields reported in this study were attributed to the relatively poor digestibility of the feedstock. In the study of Wandera et al. (2018), the performance of an AnMBR and a CSTR were compared for treatment of thermally hydrolyzed sewage sludge at different HRTs of 20 d, 10 d and 5 d, under mesophilic conditions. SRT of the AnMBR was set as double the HRT value. It was reported that both reactors showed a declining trend in terms of TS and VS removal efficiency with decreasing HRT; however, TS and VS removal efficiencies of AnMBR were still higher than the CSTR. Consequently, higher biogas production was observed in the AnMBR (1.12–3.13 $m^3{\cdot}m^{-3}{\cdot}d^{-1}$) in comparison to the CSTR (0.63–2.7 $m^3{\cdot}m^{-3}{\cdot}d^{-1}$). Lowering HRT to 3 d led to a serious foaming in the AnMBR, which caused blockage in the gas pipe. Meabe et al. (2013) investigated the applicability of a thermophilic AnMBR operated at an HRT of 5 d and SRT of 50 d. The system could be operated successfully, however, by decreasing the HRT to 3 d, VFA accumulation was observed in the reactor, reaching to a concentration of 2000 $mg{\cdot}L^{-1}$. In contrast, for a mesophilic AnMBR, increase in viscosity of sludge and deterioration in permeate quality in terms of COD and VFA concentrations was observed at an SRT of

30 d and an HRT of 7 d in the study of Meabe et al. (2013). Pushing the HRT to much lower levels than acceptable values applied for conventional anaerobic digestion processes seems to create operational problems in AnMBRs.

3.1.2. Pretreatment

The complexity of microstructure and refractory components of sewage sludge hinder the hydrolysis process. Pretreatment can be used before anaerobic digestion of sludge to disintegrate biomass and to make the organic matter more accessible to microbes, thus, it accelerates conversion of organic solids into methane. Several disintegration methods such as thermal, biological, mechanical, chemical processes, and combinations of these processes have been developed and are currently in use prior to full scale conventional ADs (Zhen et al., 2017).

Ultrasonication (US) may enhance the biodegradability of sludge due to sludge disintegration and transformation of organic matter into soluble compounds (Tiehm et al., 1997). Xu et al. (2011) investigated the operation of two parallel AnMBRs treating WAS (one was integrated with US equipment) with external membrane configurations under similar operational conditions. It was concluded that integrating US and AnMBR process slightly enhanced anaerobic digestion performance with 0.6%–3.1% increase in VS removal efficiency, due to an increase in hydrolysis rate of organic matter.

Combined methods can be applied to increase the efficiency of pretreatment. Joshi and Parker (2015) investigated the effect of pretreatment by using a combination of ultrasound and hydrogen peroxide (H_2O_2) for the digestion of WAS in an AnMBR. Two sonication durations of 20 and 60 minutes were tested in combination with a H2O2 dose of 50 g H_2O_2·kg TS^{-1}. COD destruction in the AnMBR after 20 and 60 min of US were higher by 9% and 14%, respectively compared to control reactor without pretreatment. However, it was observed that permeate COD concentrations were not significantly affected by pretreatment.

3.1.3. Co-digestion

Anaerobic co-digestion of complementary feedstocks is a commonly applied practical solution to balance the nutrients, avoid toxicity, and improve the digestibility of mono-substrates (Mata-Alvarez et al., 2014). Since conventional ADs present in many WWTPs are generally underloaded systems, the extra capacity can be utilized by adding different substrates such as industrial waste, manure, organic fraction of solid waste to increase the methane production (Cavinato et al., 2013). However, there is no study that compared the performance of AnMBRs for single sludge digestion and sludge co-digestion, while, only impact of sludge addition on co-digestion performance was studied.

Qiao, Takayanagi, Shofie, et al. (2013) investigated the treatment of a mixture of coffee grounds and WAS (15% of TS in the mixture) in an AnMBR under thermophilic conditions. They reported 67.4% of COD removal efficiency at an OLR of 11.8 kg COD·m^{-3}·d^{-1}. However, AnMBR treating only coffee grounds under the same conditions failed because of nitrogen and micronutrients deficiency, which resulted in VFA accumulation. In case of co-digestion with sludge, the OLR of the AnMBR could be increased up to 33.7 kg COD·m^{-3}·d^{-1}. Chen et al. (2019) investigated the applicability of a thermophilic AnMBR treating mixture of coffee processing water and WAS (2.8% of wet weight) at different HRTs (5–50 d) and OLRs (0.87–9.18 kg COD·m^{-3}·d^{-1}). It was concluded that optimal performance was achieved at a HRT of 10 d with 82.4% of COD conversion to methane, and methane yield of 0.307 m^3 CH_4·kg^{-1} $COD_{removed}$.

Li et al. (2015) investigated the co-digestion of coffee grounds, milk waste and WAS (7.9% of wet weight in the mixture) in an AnMBR under thermophilic conditions and studied the impact of sulfate addition on treatment performance. Without sulfate addition, inhibition and failure of operation were observed due to accumulation of VFA, especially propionic acid, in the reactor at an OLR of 14.6 kg COD·m^{-3}·d^{-1}. However, after addition of sulfate, propionic acid was degraded, the system achieved stable conditions at an OLR of 15.2 kg COD·m^{-3}·d^{-1} without accumulation of VFA, and no inhibition was observed due to an increase of free hydrogen sulfide (H_2S) concentration.

3.1.4. Addition of adsorbents

Addition of adsorbents in AnMBRs could decrease colloidal and soluble organic compounds in the bioreactor. Thus, reduction of organic fouling, and membrane flux enhancement can be achieved. Besides, addition of adsorbents may affect the anaerobic digestion process due to the changes in environmental conditions such as pH. Therefore, proper selection of appropriate type and dosage of adsorbent is very important (Ozgun, Dereli, et al., 2013; Yu et al., 2015). Yu et al. (2015) investigated the effect of two polymeric flocculants, polyaluminum chloride (PAC) and polyacrylamide (PAM), addition on AnMBR performance. For improvement of sludge filterability, it was observed that PAM could be added up to 100 mg·L^{-1} without any inhibition. The addition of PAC with dosage more than 500 mg·L^{-1} resulted in an inhibition in biogas yield due to limited nutrients. PAC addition reduced bioavailable P, which was transformed to aluminum-associated P, so limited P remained available for bacterial metabolism. Besides, the contact between microorganisms and the substrate could be disrupted due to their entrapment inside the flocculated aggregate, thus, the reaction rate could be slowed down, which is consistent with the findings of Kooijman et al. (2017). A cationic flocculant aid in combination with $FeCl_3$ was added into an AnMBR treating WAS. It was observed that the viscosity of the filtrated sludge reduced after the addition of the flocculant, however, VS destruction efficiency decreased from 32% to 24%.

Zheng et al. (2018) investigated the impact of PAC addition on microbial activity characteristics in an AnMBR. The structure of microbial community changed after PAC addition due to the limited bioavailability of phosphorus. Besides, bacterial diversity was affected as a result of increasing aluminum concentration that could negatively affect certain microbial communities. Martin-Ryals et al. (2017) investigated the effect of ion-exchange resins on the performance of two-phase AnMBR treating primary sludge after organic shock loadings. Deterioration in permeate quality and reduction in methane production were observed after organic shock loading. However, rapid improvement in permeate quality was observed after the addition of ion-exchange resins. Addition of resins shortened the recovery period in comparison to a control reactor with no ion-exchange resins addition.

3.2. Factors affecting filtration performance

Stability in flux or transmembrane pressure (TMP) is the most critical parameter for the applicability and economic feasibility of AnMBRs treating sludge for a long-term operation. There are several factors such as reactor configurations, membrane characteristics, operational conditions, and addition of flux enhancers, which contribute to the instability of flux and TMP. Therefore, a better understanding of the mechanisms of membrane fouling and its effect on flux and TMP is required. Table 2 presents the filtration performance of AnMBR applications for sludge treatment in the literature.

3.2.1. Reactor configurations

As shown in Table 2, only a few studies were performed in submerged membrane configurations with the membrane immersed either directly into the reactor or externally for sludge treatment, which might be related with the high potential of fouling due to high mixed liquor suspended solids (MLSS) concentrations (Melin et al., 2006). Submerged configurations were reported to be more suitable for low organic loads such as municipal wastewater in the study of Musa et al. (2018). Externally submerged configurations require additional construction work for building an external

Table 2. Membrane performance of AnMBR for sludge treatment.

Reactor type/module configuration	Membrane configuration	Membrane type	Material	Pore size (μm)	Filtration area (m^2)	Operation duration (d)	Flux ($L \cdot m^{-2}.h^{-1}$)	TMP (bar)	Cross-flow velocity ($m \cdot s^{-1}$)	Cleaning type	Gas sparging rate ($L \cdot min^{-1}$)	Reference
CSTR/Submerged	Tubular	—	Ceramic	0.1	1.06	35–40	2.5–8.3	0.27	0.2–0.3	Nitrogen gas backwashing	0–50	Kayawake et al. (1991)
CSTR/External	Tubular	MF	Flexible woven fiber + limestone coating	—	0.75*	—	—	—	—	Mechanical cleaning	—	Pillay et al. (1994)
CSTR/External	–	UF	—	30[a]	0.0177	124	1–13	—	—	—	—	Takashima et al. (1996)
Up-flow anaerobic bioreactor/External	Tubular	MF	Ceramic	0.1	0.05	40	120–275	2	4.5	Tap water rinsing	—	Ghyoot and Verstraete (1997)
CSTR/External	Tubular	UF	Polysulfone	0.1	0.024	44	6.9–62.1	1.5–2	—		—	Liew Abdullah et al. (2005)
CSTR/External	Tubular	UF	Titanium dioxide	0.1	1.4	7	145.8	4.8–5.5[b]	5	Chemical cleaning every day	—	Pierkiel and Lanting (2005)
CSTR/External	Vibrating configuration	UF	Polymeric Teflon	0.05	1.6	56	66.7–83.3	3.45[b]	—	Chemical cleaning every 30 days	—	Pierkiel and Lanting (2005)
CSTR/External + US	Hollow fiber	—	Polyethylene	0.4	—	7	24	—	0.75	Tap water backwashing and US	—	Wen et al. (2008)
CSTR/External	Tubular	UF	—	120[a]	0.2	180	40	28.2	1.2	Mechanical Washing (sponge balls) and chemical cleaning	—	Dagnew et al. (2010)
CSTR/External + US	Hollow fiber	—	Polythene	0.4	0.05	77	—	—	1	Chemical cleaning	—	Xu et al. (2010)
CSTR/External + US	Hollow fiber	—	Polythene	0.4	0.012	390	1.3–7	—	1	Water rinsing	—	Xu et al. (2011)
CSTR/External	Tubular	UF	Neutral surface charged	0.04	0.2	160	32.3	0.29	1	—	—	Dagnew et al. (2013)
CSTR/External	Tubular	UF	Negative surface charged	0.04	0.2	160	38.6	0.29	1	—	—	Dagnew et al. (2013)
CSTR/Externally submerged	Hollow fiber	UF	—	0.02	1.07	160	11.2	<10	—	—	—	Dagnew et al. (2013)
CSTR/External	Tubular	UF	TiO_2/ZrO_2 ceramic	300[a]	0.0226	238	7	0.05–0.5[c]	3	Tap water rinsing and chemical cleaning	—	Meabe et al. (2013)
CSTR/External	Tubular	UF	TiO_2/ZrO_2 ceramic	300[a]	0.0226	384	7	0.17–0.6[d]	3	Tap water rinsing and chemical cleaning	—	Meabe et al. (2013)
CSTR/External + US	Hollow fiber	—	Polythene	0.4	0.012	54	3.5	—	1	—	—	Xu et al. (2013)
CSTR/Submerged	Flat sheet	MF	Chlorinated polyethylene	0.2	0.116	155	2–7.6	—	—	Chemical cleaning	5	Qiao, Takayanagi, Shofie, et al. (2013)
CSTR/Submerged	Flat sheet	MF	Chlorinated polyethylene	0.2	0.116	263	—	—	—	—	5	Qiao, Takayanagi, Niu, et al. (2013)
CSTR/Submerged	Hollow fiber	—	—	—	0.047	70	2.75	—	—	Backwashing and chemical cleaning	2	Joshi and Parker (2015)
CSTR/Submerged	Flat sheet	MF	Chlorinated polyethylene	0.2	0.116	373	—	—	—	—	5	Li et al. (2015)
CSTR/External	Tubular	MF	Polytetrafluoroethylene	—	0.06	92	4.2–25	0–25	0.7	—	—	Hafuka et al. (2016)
Two-phased CSTR/External	Mesh screen	—	Naylon	100	—	190	10–15	1.47–3.92	—	Tap water rinsing and chemical cleaning	—	Joo et al. (2016)
	Spiral-wound	UF	Polysulfone and thin film composite type with spiral-wound	0.03	—							
CSTR/Externally submerged	Flat sheet	DM	Dacron mesh	39	0.38, 0.46	200	15	0–30	—	Physical cleaning	37.5[e]	Yu et al. (2016)
Two-phased CSTR/External	Mesh screen	—	Naylon	60	—	>155	3.8–22.62–4		—	Tap water rinsing and chemical cleaning	—	Kim et al. (2017)
	Spiral-wound	UF	Polysulfone and thin film composite type	0.03	—							
CSTR/External	—	DM	Woven fabric	—	0.025	217	0.1	0.15	0.044	—	—	Kooijman et al. (2017)
CSTR/External + coagulant + flocculant aid	—	DM	Woven fabric	—	0.025	83	0.1	0.3	0.044	—	—	Kooijman et al. (2017)
CSTR/External	Tubular	—	PVDF negatively charged	40	0.4	>730	—	—	—	Water flushing, abrasion and chemical cleaning	—	Pileggi and Parker (2017)
Two-phased CSTR/Submerged	Filter cartridges	—	—	10	0.22	130	—	—	—	—	—	Martin-Ryals et al. (2020)
	Hollow fiber	—	PES	0.2	0.15	65						

(*continued*)

Table 2. Continued.

Reactor type/module configuration	Membrane configuration	Membrane type	Material	Pore size (μm)	Filtration area (m^2)	Operation duration (d)	Flux ($L \cdot m^{-2} \cdot h^{-1}$)	TMP (bar)	Cross-flow velocity ($m \cdot s^{-1}$)	Cleaning type	Gas sparging rate ($L \cdot min^{-1}$)	Reference
CSTR/Submerged	Flat sheet	MF	—	0.2	0.116	155	4.4	0.06–0.18	—	—	5	Wandera et al. (2018)
CSTR/Submerged	Flat sheet	MF	—	0.2	0.116	125	—	—	—	—	2.6[e]	Chen et al. (2019)
taCSTR/External	Hollow fiber	MF	Polytetrafluoroethylene	—	0.015	248	0.42–2.92	<0.12	0.4	Physical and chemical cleaning	—	Hafuka et al. (2019)
CSTR/Submerged	Flat sheet	MF	—	0.22	0.116	170	3.6–10.5	0.04–0.11	—	Water flushing and chemical cleaning	—	Wandera et al. (2019)
CSTR/Submerged	Flat sheet	FO	Cellulose triacetate	—	0.005	105	0.3–1	—	—	Physical and chemical cleaning	—	Zhao et al. (2019)

[a]kDa, molecular weight cut off.
[b]Inlet pressure.
[c]Mesophilic.
[d]Thermophilic.
[e]$m^3 \cdot m^{-2} \cdot h^{-1}$.
[*]Calculated by using the reported data.

chamber in addition to gas sparging pumps to control membrane fouling, and flameproof piping. Thus, the capital and operational costs will increase and threatens the economic feasibility of this configuration. External cross-flow configurations seemed to be more commonly applied for sludge digestion (Table 2). External cross-flow configurations can be operated at high fluxes with better control of fouling and easier membrane replacement (Lin et al., 2013). Higher range of fluxes (0.1–275 $L \cdot m^{-2} \cdot h^{-1}$) was reported in the literature for external cross-flow configuration compared to those reported for submerged one (0.3–10.5 $L \cdot m^{-2} \cdot h^{-1}$), as shown in Table 2. However, it would require increased energy for cross-flow pumps to maintain a high shear force over the membranes (Maaz et al., 2019). Membrane life-time is a critical issue in cross-flow configuration since some inert materials such as sand particles in primary sludge can act as abrasives on membrane surface during sludge recirculation (Siembida et al., 2010), which may deteriorate membrane integrity and life time. Thus, maintaining adequate cross-flow velocity is important in order not to damage the membrane. External AnMBRs can also be operated with gas-lift mode, at which biogas can be used for the transfer of sludge from bioreactor to the membrane module instead of liquid recirculation (Ersahin et al., 2016; Torres et al., 2011).

3.2.2. Membrane characteristics

Membrane material and pore size are important factors that affect fouling affinity. For similar operational conditions, Ghyoot and Verstraete (1997) obtained a flux of 209 $L \cdot m^{-2} \cdot h^{-1}$ with ceramic MF membrane, which was ten times higher than the flux obtained with polymer UF membrane in a 90-minute filtration trial. Dagnew et al. (2012) investigated the applicability of neutral and negatively charged tubular membranes in an AnMBR with total solids concentration of 6 and 18 $g \cdot L^{-1}$. At low flux (8 $L \cdot m^{-2} \cdot h^{-1}$) conditions, no effect was observed on membrane fouling. However, neutral charged membrane exhibited a significant increase in fouling at higher flux (30 $L \cdot m^{-2} \cdot h^{-1}$), while negatively charged membrane showed stable TMP profile.

Membrane configuration is another factor that can affect permeate flux. Dagnew et al. (2013) compared the filtration performances of hollow fiber and tubular membranes in an external configuration. A higher critical flux was obtained with the tubular membrane (>30 $L \cdot m^{-2} \cdot h^{-1}$) in comparison to the hollow fiber one (>18 $L \cdot m^{-2} \cdot h^{-1}$). Pillay et al. (1994) investigated the applicability of tubular woven fiber in a cross-flow AnDMBR configuration for primary sludge treatment for 50 h as a full-scale trial. The system could be operated without any clogging, and very slow decrease was observed in permeate flux during the operational period. It was claimed that AnMBR process could be operated for long time period without a necessity of membrane cleaning. Joo et al. (2016) investigated the use of 100 μm nylon mesh screen prior to UF membrane to decrease solids load and improve permeability of the UF membrane. Solids concentration after the mesh screen decreased from 15–30 to 3–4 $g{\cdot}L^{-1}$ and permeate flux increased from 10 to 15 $L{\cdot}m^{-2}{\cdot}h^{-1}$.

3.2.3. Operational conditions

3.2.3.1. HRT. HRT is an important factor for membrane fouling as decrease in HRT is followed by an increase in solids concentration in the reactor (Pillay et al., 1994). In addition, decrease in HRT results in more water passing through the membrane unit, and unless effective membrane filtration area is increased, flux needs to be increased. Consequently, TMP would increase causing a negative impact on fouling. Few researchers have studied the effect of HRT on filtration performance in AnMBRs for sludge treatment. AnMBR system could be operated without any change in flux and TMP at HRTs of 20 and 30 d (SRT as twice as HRT) in the study of Wandera et al. (2018). However, sudden increase in TMP and decrease in permeability were observed during operation at HRT of 10 d, which required membrane replacement. Although solids concentration in the sludge decreased, it was hard to operate the AnMBR at an HRT of 3 d because of foaming and subsequent increase in TMP (Wandera et al., 2018). An increase in TS concentration from 2.6% to 5.5% was observed by a decrease in HRT from 26 d to 14 d (Pillay et al., 1994). Several researchers confirmed the fact that increase in solids concentration had a negative effect on membrane flux (Dagnew et al., 2012; Ghyoot & Verstraete, 1997; Meabe et al., 2013; Pillay et al., 1994).

3.2.3.2. Temperature. Temperature is another factor that affects the filtration performance of membranes as it affects the viscosity of the filtered liquor and the solubilization rate of various compounds. Meabe et al. (2013) compared filtration performance of an UF membrane under mesophilic and thermophilic conditions. It was reported that increasing the temperature affected the rheological properties of the sludge. Due to higher solubilization rate, the amount of small size particles increased significantly under thermophilic conditions, showing wider particle size distribution, and lower viscosity values compared to the mesophilic one. Although better filtration performance was observed in the thermophilic reactor due to lower viscosity, the small particles increased the pore blocking in the membrane, which increased the irreversible fouling and required more frequent chemical cleaning.

3.2.3.3. Operation mode. Operation mode is an important factor to achieve sustainable and stable operation in AnMBRs. Frequent backwashing and/or relaxation can be used for fouling control, in addition to gas sparging and periodical chemical cleaning. Yu et al. (2016) investigated the operation of AnDMBR under two different operational modes: Continuous filtration and a filtration (10 min)/relaxation (2 min) cycle. Almost 4 times longer period of operation without physical cleaning was obtained at the filtration/relaxation mode in comparison to continuous filtration. Dagnew et al. (2012) monitored TMP for WAS filtration at a flux of 30 $L \cdot m^{-2} \cdot d^{-1}$ at continuous and intermittent filtration modes. Lower extended TMP (<0.4 bar) was obtained at intermittent mode compared to increasing TMP (0.3–0.8 bar) obtained at continuous mode. Dagnew et al. (2010) compared the permeate flux of negatively charged and neutral membranes at continuous and intermittent operation modes. Permeate flux during intermittent mode of operation for negatively charged and neutral membranes were higher by 74.7% and 58.8%, respectively, in comparison to continuous mode.

3.2.3.4. Shear force. Controlling cake layer deposition on the surface of membrane is important for the stability of AnMBR operation. High cross-flow velocity and/or gas sparging rate have been used as strategies to prevent cake layer formation in AnMBRs (Liao et al., 2006; Ozgun, Dereli, et al., 2013). Ghyoot and Verstraete (1997) stated that increase in cross-flow velocity had a minor effect on permeate flux. This result is not consistent with the results of Pillay et al. (1994) in which permeate flux exhibited sensitivity to changes in cross-flow velocity. AnMBR system could be operated for 248 d without membrane cleaning at low flux (0.42 to 2.92 $L \cdot m^{-2} \cdot h^{-1}$) and moderate cross-flow velocity (0.4 $m \cdot s^{-1}$) (Hafuka et al., 2019). Cross-flow velocity became less effective at high solids concentration, which increased the viscosity, leading to a decrease in the turbulence of the membrane unit (Dagnew et al., 2012). Biogas sparging in AnMBRs can be used not only for scouring cake layer but also for mixing bulk sludge (Wandera et al., 2018; Yu et al., 2016). Yu et al. (2016) applied intermittent gas sparging in an AnDMBR in order to control the DM layer formation on a mesh surface. Kayawake et al. (1991) reported that permeate flux was doubled after gas sparging application during filtration of heat-treated liquor obtained from sewage sludge.

3.2.4. Ultrasonication

US can be used to prevent cake layer formation on membrane surface through cavitation and acoustic streaming mechanisms (Lamminen et al., 2004). Xu et al. (2010) coupled AnMBR with US that was operated at different power intensities (0.12–0.18 $W{\cdot}cm^{-2}$) and durations (3–5 $min{\cdot}h^{-1}$). They compared membrane filtration resistances of US-AnMBR coupled system and single AnMBR process. Coupling US to AnMBR was effective for fouling control, showing lower filtration resistance in comparison to the single AnMBR. Xu et al. (2013) observed effective removal of cake layer and partial control of gel layer fouling after US integration. It was reported that US was effective to remove organic gel layer fouling, especially proteins and humic acids.

Viscosity of the liquor is a limiting factor for US to control membrane fouling (Williams & Wakeman, 2000). Xu et al. (2011) reported that higher ultrasound energy was required to obtain stable long-term filtration performance at higher MLSS concentration. However, increasing US power intensity may cause damage in the membrane. Wen et al. (2008) observed damage in the membrane, which affected permeate quality after increasing US power intensity from 0.12 to 0.20 $W{\cdot}cm^{-2}$. It was suggested that this damage might be a result of collisions of micro particles on membrane surface and chemical oxidation of hydroxyl radicals generated during acoustic cavitation.

3.2.5. Addition of flux enhancers

Flux enhancers have attracted attention for fouling control in AnMBRs. Membrane fouling control via flux enhancer addition acts through phenomena such as adsorption of soluble microbial products and coagulation of small sized particles (Ozgun, Dereli, et al., 2013). However, few researchers reported the effect of flux enhancers specifically for sludge treatment by AnMBRs. Kooijman et al. (2017) reported that addition of cationic flocculant was not effective on fouling control, and caused pore clogging. This is in contrast with the results of Dagnew et al. (2012), in which improvement in membrane performance and reduction in filtration resistance were observed after addition of cationic polymer. Yu et al. (2015) investigated the filtration performance of an AnMBR by adding two polymeric flocculants: PAM and PAC. Improvement in sludge filterability was observed after addition of each flocculant but through different mechanisms. PAM enlarged floc size, which would increase the filterability of sludge. However, applied shear forces during membrane filtration broke apart the formed large flocs, which weakened the effect of PAM. PAC adsorbed the soluble microbial products in the reactor, which is considered as one of the main contributors in membrane fouling. Moreover, adding the flocculant at the startup phase of operation was recommended to have more stable extended operation (Yu et al., 2015).

4. VFA recovery

VFAs can be recovered from sewage sludge in AnMBR systems and recovered VFAs can be used for enhancement of denitrification and biological phosphorus removal, biofuel production, i.e., ethanol, butanol, and bioplastic production (Kondaveeti & Min, 2015; Lee et al., 2014). Alternatively, hydrogen gas can be produced from VFA in photosynthetic reactor (Jeong et al., 2007). VFA production can be achieved by inhibiting the methanogenic activity in the reactor with the control of operational conditions such as pH (Liu et al., 2012, 2016). Limited studies focusing on VFA recovery from sludge were conducted in the literature (Table 3).

Integration of membranes retains the microorganisms and solids inside a fermenter, resulting in a better conversion of organic matter to VFAs (Jeong et al., 2007). Liu et al. (2016) compared the performance of a fermenter before and after coupling a DM for VFA production. They observed that VFA yield increased by 233.3% and the total recovered VFA concentration increased from 990 to 3220 $mg{\cdot}L^{-1}$ after addition of the membrane. Efficient retention of proteins and polysaccharide by 70% and 40%, respectively and low retention of sCOD by 30% were achieved in the AnDMBR. A similar observation was indicated in the study of Gao et al. (2019), in which rejection rates of sCOD, proteins and polysaccharides by 12%, 13%, and 74%, respectively were achieved in an AnMBR. The higher rejection of polysaccharides in comparison to proteins was attributed to the significant enrichment of Christensenella minuta from glycolytic genus Christensenella, which can consume different sugars for acid production.

Kim, Somiya, et al. (2002) investigated the treatment of coagulated primary sludge for VFA recovery by an AnMBR at different HRTs (8 h to 4 d) and SRT of 10 d. By increasing HRT to 12 h, VFA recovery reached to a maximum concentration, around 1200 $mg{\cdot}L^{-1}$. Increasing HRT beyond 12 h had a negative effect on VFA recovery since some soluble VFAs were converted to gaseous or mineral products. Liu et al. (2019) investigated the applicability of AnDMBR to recover VFAs from the supernatant of sewage sludge, applying different OLRs of 5, 11, and 17 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$ and HRTs of 7.74, 3.52, and 2.28 d, respectively. VFA concentration in the permeate was in a range of 8500–8600 $mg{\cdot}L^{-1}$ at OLRs of 5 and 11 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$. However, VFA concentration decreased to 6280 $mg{\cdot}L^{-1}$ after increasing OLR to 17 kg $COD{\cdot}m^{-3}{\cdot}d^{-1}$, which decreased the conversion rate of organic matter in the reactor and reduced the concentration of VFAs in the permeate.

Jeong et al. (2007) investigated the recovery of VFAs from WAS in an AnMBR, and use of VFAs to produce hydrogen gas in a photosynthetic reactor. 1100 $mg{\cdot}L^{-1}$ of VFA could be recovered by using an external

Table 3. AnMBR performance for VFA recovery.

Sludge type	Reactor type/module configuration	Reactor volume (L)/temperature (°C)	OLR (kg COD·m^{-3}·d^{-1})	HRT (d)	SRT (d)	Influent TS (g·L^{-1})	Operation duration (d)	VFA production (mg·L^{-1})	Reference
Coagulated raw sludge	CSTR/External	76/35	0.6–7.5	—	10	5.6	—	801–1197	Kim and Somiya (2001a)
Coagulated raw sludge	CSTR/External	—/35	—	4	20	—	100	—	Kim and Somiya (2001b)
Coagulated raw sewage sludge	CSTR/External	76/35	0.6–7.5	0.3–4	10	5.6	180	801–1197	Kim, Somiya, et al. (2002)
Coagulated raw sludge	CSTR/External	76/35	—	—	—	—	0.69	—	Kim et al. (2005)
Primary sludge	CSTR/Submerged	100/35	—	1–4	10–20	8.14–12.35	280	2451–4105	Kim and Jung (2007)
WAS	CSTR/External	—/35	—	3	—	—	90	1100	Jeong et al. (2007)
WAS	CSTR/Externally submerged	—/35	—	3	—	—	90	550	Jeong et al. (2007)
Coagulated raw sludge	CSTR/External	65/—	—	4	20	—	160	—	Kim and Chung (2012)
Sewage sludge	CSTR/Submerged	14/35	—	5.4	—	12–15[a]	18	3220	Liu et al. (2016)
Dewatered sewage sludge	CSTR/Submerged	3.6/25	—	—	7.6	—	18	1300–1825	Gao et al. (2019)
Supernatant of sewage sludge	CSTR/Submerged	14/37	5–17	2.28–7.74	—	4.1	70	6280–8590	Liu et al. (2019)

[a]TSS.

AnMBR configuration and the hydrogen production in the photosynthetic reactor was found to be 160 mL H_2·g VFA^{-1}.

Ozone has an effect not only on membrane fouling but also on treatment performance and microbial activity. Kim and Chung (2012) applied intermittent ozonation before the membrane unit in an external AnMBR fermenting coagulated primary sludge for recovery of VFAs. Reduction in concentration of refractory substances and improvement in solubilization of solids were observed after ozone application. Acetic acid fraction in VFAs increased from 52.2% to 65.7% without significant microbial inhibition, showing improvement in acetate production after ozonation.

To increase the permeate recovery, filtration performance should be improved. Table 4 shows the filtration performances obtained in different AnMBRs applied for VFA recovery. Different strategies such as air/ozone backwashing and cross flow velocity were reported in the literature to control fouling and enhance permeate recovery (Kim et al., 2005; Kim & Jung, 2007). Kim and Jung (2007) applied intermittent air and air/ozone reciprocal backwashing modes during the recovery of VFA. Higher flux recovery was achieved with air/ozone backwashing mode (1.7 times) in comparison to single air backwashing. Cross-flow velocity is also important to avoid fouling and obtain stable flux. Kim et al. (2005) applied different cross-flow velocities (0.1–1.2 m·s^{-1}) to a membrane having a pore size of 1 μm. A positive correlation between cross-flow velocity and permeate flux was observed, while a negative relationship between cross-flow velocity and cake layer resistance was obtained. However, applying high cross-flow velocities resulted in associated energy costs, which needs to be taken into consideration.

5. Cleaning methods

Membrane cleaning is very important in terms of efficient AnMBR operation. Physical and/or chemical membrane cleaning can be applied depending on the nature of fouling. Physical cleaning can be applied to remove the reversible fouling that is mainly caused by cake layer formation since it is loosely attached to the membrane surface. Physical cleaning is not effective for preventing irreversible fouling, which requires chemical cleaning due to gel layer formation on membrane surface and membrane pore blocking. Irrecoverable fouling causes permanent increase in filtration resistance especially during long-term operation, which can be removed by neither physical cleaning nor chemical cleaning (Hafuka et al., 2019; Liao et al., 2006; Ozgun, Dereli, et al., 2013; Wang et al., 2014).

Generally, physical cleaning is carried out in-situ via backwashing and/or relaxation. Membranes can be taken out of reactor and physically cleaned

Table 4. AnMBR filtration performance for VFA recovery.

Reactor type/module configuration	Membrane configuration	Membrane type	Material	Pore size (μm)	Filtration area (m^2)	Flux ($L \cdot m^{-2} \cdot h^{-1}$)	TMP (bar)	Cross-flow velocity ($m \cdot s^{-1}$)	Cleaning type	Reference
CSTR/External	Monolith	MF	Ceramic	1	—	—	0.38	0.4	—	Kim and Somiya (2001a)
CSTR/External	Monolith	MF	Ceramic	1	0.12	30–38	0.35	0.4	Physical cleaning (sponge)	Kim and Somiya (2001b)
CSTR/External	Monolith	MF	Ceramic	1	0.12	42–54	0.35	0.4	Intermittent Ozonation	Kim and Somiya (2001b)
CSTR/External	Monolith	MF	Ceramic	1	—	—	0.38	0.4	—	Kim, Somiya, et al. (2002)
CSTR/External	Monolith	MF	Ceramic	0.1–0.5	0.12	17–62.5	0.75	0.4	—	Kim et al. (2005)
CSTR/External	Tubular	MF	Ceramic	1–5	0.035	24–72	0.75	0.4	—	Kim et al. (2005)
CSTR/Submerged	Tubular	MF	Stainless steel	1	0.0097	4.2	0.25	—	Air/ozone backwashing	Kim and Jung (2007)
CSTR/External	Hollow fiber	MF	Polysulfone	0.1	0.0625	0.1852	—	—	—	Jeong et al. (2007)
CSTR/Externally submerged	Hollow fiber	MF	Polysulfone	0.1	0.0625	0.1852	—	—	—	Jeong et al. (2007)
CSTR/External	Monolith	MF	Ceramic	1	—	28.75	0.35	0.4	—	Kim and Chung (2012)
CSTR/External	Monolith	MF	Ceramic	1	—	48.75	0.35	0.4	Intermittent Ozonation	Kim and Chung (2012)
CSTR/Submerged	Tubular	DM	Silk	100	0.0314	1–2	—	—	Physical cleaning	Liu et al. (2016)
CSTR/Submerged	Hollow fiber	MF	PVDF	0.1	0.0751	—	—	—	—	Gao et al. (2019)
CSTR/Submerged	Tubular	DM	Silk	100	—	0.26–1.04	—	—	Physical Cleaning	Liu et al. (2019)

ex-situ to be rinsed with tap water. Liew Abdullah et al. (2005) reported that daily physical cleaning by applying ex-situ water flushing was not effective for obtaining long-term stable flux. Backwashing is also important to minimize fouling rate. Backwashing can be performed by using permeate (Joo et al., 2016) or nitrogen gas (Kayawake et al., 1991) during operation of AnMBRs for sludge digestion.

Different chemical cleaning reagents are used to remove organic and inorganic irreversible fouling. Base solutions e.g., sodium hydroxide (NaOH) and oxidants e.g., sodium hypochlorite (NaClO) and H_2O_2, are generally used to remove organic foulants such as proteins and carbohydrates. Acid reagents such as citric, hydrochloric, phosphoric, and sulfuric acids can eliminate inorganic foulants, e.g., multivalent cations and metal hydroxides. Combining these reagents for chemical cleaning, sequentially or jointly, is required to improve membrane cleaning efficiency (Ozgun, Dereli, et al., 2013; Wang et al., 2014). Hafuka et al. (2019) investigated the applicability of sequential cleaning—i.e., water rinsing, NaOH, HCl, and NaClO, to identify fouling types accumulated on the membrane. It was reported that reversible foulants and irreversible inorganic and organic foulants contributed to overall fouling with a share of 1.2%, 15.9%, and 44.4%, respectively, while irrecoverable fouling accounted for 38.5%. In contrary to NaClO, NaOH was not effective for organic foulant removal.

One type of chemical cleaning is to apply backwashing in conjunction with chemicals such as ozone (Wang et al., 2014). Kim and Somiya (2001b) investigated the effect of intermittent ozone backwashing on the membrane fouling during recovery of VFA. Higher flux recovery and less frequent cleaning interval were achieved with ozone backwashing in comparison to operation without ozonation.

Membrane material and configuration may affect the efficacy and interval of cleaning. Dagnew et al. (2010) applied physical cleaning, washing with sponge balls, followed by a sequence of chemical cleaning including NaOH and citric acid, for negatively and neutral charged membranes. Clean water flux recovery of 93% and 98% were obtained with negatively-charged and neutral membranes, respectively. Iron salts precipitates were identified as the largest contributor of fouling for negatively-charged membranes, which were removed through acid cleaning. Moreover, it was reported that NaOH cleaning adversely affected clean water flux recovery for neutral membranes and resulted in 2% water flux reduction. Pierkiel and Lanting (2005) applied chemical cleaning every 30 d for vibrating membrane configuration to keep the module inlet pressure at 3.45 bar, while daily chemical cleaning was required in a tubular cross-flow membrane configuration to keep the module inlet pressure in the range of 4.8–5.5 bar.

Liu et al (2016) used a DM in rotary configuration in submerged AnMBR for VFA recovery. The agitator speed was maintained at 60 rpm during the fermentation process. In case of fouling and low filtration resistance, they adopted rapid rotation for the membrane installation without permeation while in the case of severe fouling, the membrane was taken out to apply ex-situ water flushing.

Operational temperature affects chemical and physical characteristics of sludge, which, consequently, affects fouling characteristics and cleaning intervals. Meabe et al. (2013) reported that reversible fouling was more important at mesophilic conditions due to higher viscosity, which reduced the effect of shear forces over the membrane surface. In contrast, irreversible inorganic fouling had more contribution in a thermophilic system, where precipitated inorganic crystals (struvite) were observed on the membrane surface. As a result, more frequent chemical cleaning was required at thermophilic conditions compared to mesophilic conditions.

6. Economic feasibility

Financial evaluation is important for the selection of a feasible treatment technology among various processes. Thus, feasibility studies should be carried out for AnMBRs based on energy requirements and operational costs. Energy can be consumed in heating sludge, mixing, permeation and gas/liquid recirculation. Installment of membranes, life time of membranes and frequency of chemical cleaning should be considered in feasibility studies. However, limited information is reported in the literature about financial evaluation of AnMBRs for sludge treatment.

Energy requirement for heating sludge, and heat loss account for more than 90% of overall energy consumption for AnMBR and conventional AD operation, while reactor mixing and pumping consume less than 10% of the overall energy (Dagnew et al., 2013; Pileggi & Parker, 2017). Methane produced by anaerobic digestion is used to generate energy to alleviate the high energy requirements. Despite the additional energy requirements for permeation and gas/liquid recirculation, AnMBRs may have more favorable energy balance in comparison to conventional AD due to higher methane production and lower heat losses. The capability of increasing OLR for AnMBRs to greater values than conventional AD may reduce the effective volume of the reactor, which can potentially decrease the overall heat losses from the reactor. Moreover, the capability to increase SRT, thanks to membranes, can lead to increased destruction of organic matter in the reactor, thereby increasing methane production (Dagnew et al., 2013; Pileggi & Parker, 2017; Yu et al., 2016). Pillay et al. (1994) estimated the capital and operational costs of conventional AD and AnMBR for a project life of 20 years. It was reported that the capital and total project costs of conventional AD can be saved by 27% and 12%, respectively, by integrating external MF membrane unit. Pileggi and Parker (2017) compared AnMBR and conventional AD in terms of energy balance at ambient, mesophilic and thermophilic conditions. Overall, net energy balance for AnMBR at each condition was found better than that for conventional AD, considering the energy required for heating, pumping, mixing, recirculation; energy gained from methane production, and heat losses from reactor walls. Furthermore, positive energy balance was estimated for AnMBR at ambient and mesophilic temperature. This agrees with the results of Yu et al. (2016), in which the energy balance between AnDMBR and conventional AD was compared for WAS treatment. It was reported that net energy demand of AnDMBR was lower than conventional AD by 37.3%, which could save energy of 66×10^4 kWh annually for full-scale WWTP applications.

Membrane configurations affect the overall operational energy consumption and total costs. Dagnew et al. (2013) compared tubular and hollow fiber membranes in AnMBR with conventional AD for WAS treatment. It was reported that hollow fiber membranes have lower capital costs and required less space in comparison to tubular ones, while tubular membranes were better for handling high solid concentrations and being operated at higher temperature. Moreover, higher energy for permeation and sludge recirculation was required for hollow fiber membranes in comparison to tubular one. It was reported that conventional AD required an additional energy of 1.85×10^4 kWh per m^3 of sludge fed, while AnMBR with tubular membrane produced (positive net energy) 0.61×10^4 kWh per m^3 of sludge fed. Compared to tubular membrane, AnMBR with hollow fiber membrane produced 27.3% less net energy (Dagnew et al., 2013). Ghyoot and Verstraete (1997) compared the economic feasibility of polymer UF and ceramic MF applications in AnMBR based on energy requirements and membrane replacement costs. It was indicated that the life time of ceramic membranes was between 3 and 5 years, while this value was between 0.5 and 2 years for polymer membranes. However, it was reported that the unit total cost including energy requirements and membrane replacement costs for ceramic MF was more than twice that for polymer UF. Mesh screen was coupled with UF membrane in external configuration to reduce solids concentration load on UF membrane in the study of Joo et al. (2016). By improving filtration performance of UF, it was estimated that the mesh screen contributed to a decrease of 20% in operational costs.

Biogas recirculation contributes to overall energy consumption, which can be alleviated by optimization. Yu et al. (2016) reported that intermittent biogas recirculation mode (120-min off and 20-min on) could save up to 85.7% of energy consumed by continuous biogas recirculation mode.

7. Challenges and future opportunities

Current literature and experience suggest that even though there seem to be several advantages of using AnMBRs for sludge treatment, there are also problems such as low OLR application and membrane fouling, which can limit the applicability and feasibility of AnMBR processes for full-scale systems. Therefore, further investigation must be performed to reduce fouling and optimize operation conditions in AnMBRs for effective sludge treatment.

One of the major concerns for AnMBRs treating sewage sludge is limited OLR, although applied OLR for AnMBR is higher in comparison to conventional AD. Increase in OLR or decrease in HRT decreases the effective volume of the reactor, which saves energy required for operation and decreases the capital costs of AnMBR. However, inhibition during anaerobic sludge digestion may happen due to VFA accumulation in the reactor, which is perpetuated by increasing OLR. Foaming was detected in the reactor, which blocked the gas pipe when HRT was decreased to 3 d in the study of Wandera et al. (2018). On the other hand, applied OLRs of sewage sludge reported in the literature were significantly lower than OLRs applied for industrial wastewater treatment (Dereli et al., 2012), which indicates that there is still a room for improvement to avoid inhibition and to optimize the performance of AnMBR.

Fouling is considered as the most critical constraint for the feasibility of long-term operation of AnMBRs as it affects the operational costs. Increase in membrane fouling rate leads to TMP rise and a need for more frequent chemical cleaning. Thus, understanding fouling characteristics and reasons behind is significant for reducing fouling. High viscosity of sludge was reported as one of the main reasons causing fouling in a mesophilic AnMBR since it affected the applied shear forces (Meabe et al., 2013). Thus, development of AnMBR handling high solids concentrations is required. Inorganic substances such as struvite should also be taken into consideration due to release of ammonia and phosphate in anaerobic digestion process (Liao et al., 2006). Irreversible inorganic fouling (crystals) was reported to be the major contributor in fouling for sludge treatment at thermophilic conditions, which might be related with struvite precipitation (Meabe et al., 2013). Moreover, clogging was detected in permeate pipe because of struvite deposition, which needed chemical cleaning. Therefore, further investigation for fouling characteristics must be carried out in long-term operation of AnMBRs treating sludge.

Most of the studies reported in the literature have been performed in single-phase AnMBRs for sludge treatment. In a single-phase reactor, all the anaerobic bacteria including acid and methane producing archaea are inside the same volume. However, growth rate of acid producing bacteria is faster than the growth rate of methane producing archaea. This is the reason for the accumulation of VFAs in reactors, when the conversion rate of acidogenic bacteria exceeds that of methanogenic species. When this kind of imbalance occurs, methanogens are further inhibited by decrease in pH levels, and methane production comes to a halt. Therefore, the optimization of the two-phase AnMBR configuration, where acidogenesis happens in the first phase and methanogenesis happens in the second, can be a promising solution to avoid inhibition caused by VFA accumulation in high loaded reactors.

DM technology has been gaining a considerable amount of interest for wastewater treatment (Ersahin et al., 2017; Isik et al., 2019). However, few studies examined the applicability of DM for sludge treatment. DM is a promising solution for solving fouling problem as foulants take part in filtration process. Thus, operating the system with higher solids concentration may be possible. Besides, reduction in capital and operation costs is expected as low-cost support material is used instead of membranes, and only physical cleaning is applied.

Lower COD concentrations can be obtained in AnMBR permeate with 99% of COD removal efficiency, as shown in Table 1, compared to the supernatant of conventional AD. Thus, reduced C/N ratio will result in less growth of heterotrophic denitrifying bacteria, which compete with Anammox bacteria for ammonia (Molinuevo et al., 2009). This would allow a better operation in case of the application of SHARON/Anammox technology for treatment of AnMBR permeate in comparison to conventional AD supernatant.

Sewage sludge contains a considerable amount of OMPs and heavy metals, which are persistent during anaerobic digestion (Barret et al., 2012; Gonzalez-Gil et al., 2016). Integration of membrane to anaerobic digestion process will affect the fate of these pollutants in both permeate and excess sludge. Therefore, further research focusing on the fate and removal of OMPs and heavy metals in AnMBR for sludge digestion is needed.

Nutrient recovery concept has been gaining importance in sludge treatment processes recently. Ghyoot and Verstraete (1997) detected high amount of ammonium ($202\,mg{\cdot}L^{-1}$) and phosphate ($3.7\,mg{\cdot}L^{-1}$) in the permeate with polymer UF membrane treating primary sludge. One of the advantages of AnMBR process is keeping SS and microorganisms in the reactor (Hafuka et al., 2019). This allows the direct use of permeate for irrigation in agriculture applications or more efficient and feasible nutrient recovery in the form of struvite. The recovered struvite from AnMBR will contain less impurities in comparison to that from conventional AD supernatant, which would increase the market value of the recovered struvite. Pretreatment of sludge by ozonation can be a promising technique to oxidize and convert the refractory organic compounds into more easily biodegradable forms, and release of phosphorous. Pretreatment by ozonation may increase the efficiency of sludge digestion enabling higher biogas production and phosphorous recovery from liquid fraction (Chu et al., 2009; Erden & Filibeli, 2011). Furthermore, VFA recovery from sludge fermentation has been examined by a few researchers (Kim & Chung, 2012; Kim & Jung, 2007; Liu et al., 2016). VFAs can be used as a carbon source for nitrification and denitrification in WWTPs, and for bioplastic production (Lee et al., 2014). Further research is required that focus on VFA recovery and optimum operational conditions.

Additional research is required that focus on feasibility and modeling of AnMBR process treating sludge and also on comparison of AnMBR and conventional AD in terms of treatment performance and economic feasibility. Research that addresses the energy and cost aspects of adopting AnMBRs for sludge processing would provide valuable contribution to the current literature.

8. Conclusions

AnMBR technology has a potential to overcome the limitations encountered in the application of conventional ADs for sludge treatment. By decoupling the HRT and SRT, low reactor footprint and high permeate quality can be achieved with AnMBRs. In addition, retention of solids and microorganisms in the AnMBR will result in an increase in digestion efficiency and methane production. Since permeate is free of SS, direct use of permeate for irrigation or nutrient/struvite recovery become more efficient and feasible in comparison to conventional AD supernatant. Moreover, low COD concentrations in the AnMBR permeate would allow better operation of subsequent innovative nitrogen removal processes due to low C/N ratio, compared to conventional AD supernatant which includes higher COD concentrations. However, there are still problems, such as membrane fouling, which hinder the adoption of AnMBR technology for sludge management, as well as a lack of studies demonstrating the economic benefits of using AnMBRs for sludge treatment. In this context, pilot and full scale AnMBR applications should be further investigated as well as lab-scale studies.

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