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A review on anaerobic membrane bioreactors: Applications, membrane fouling and future perspectives

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HIGHLIGHTS

► Recent progress in AnMBRs treating various wastewaters is summarized.
► Advances in membrane fouling control strategies in AnMBRs are addressed.
► Research directions regarding AnMBR technology are identified.
► AnMBR is a promising technology for wastewater treatment and reuse.

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ABSTRACT

In the last years, anaerobic membrane bioreactor (AnMBR) technology is being considered as a very appealing alternative for wastewater treatment due to the significant advantages over conventional anaerobic treatment and aerobic membrane bioreactor (MBR) technology. Many articles have touted the diverse potential applications of AnMBR in various stream treatment, and membrane fouling issues. In current review, the fundamentals of AnMBR (including advantages and configurations, membrane materials and modules, and history development), application development in various stream treatment, and membrane fouling researches are summarized and critically assessed. The characteristics of AnMBR and aerobic MBR for wastewater treatment are also compared. AnMBR technology appears to be suitable for treatment of various streams, especially for food industrial wastewater and municipal wastewater. AnMBR treatment usually encounters more serious membrane fouling problem. This, however, can be remedied through various conventional and novel membrane fouling control or cleaning measures. Based on the review, future research perspectives relating to its application and membrane fouling research are proposed.

Contents

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1. Introduction

Anaerobic digestion is one of the most important processes used for various industrial wastewaters as well as sewage treatments because it combines pollution reduction and energy production. Moreover, compared to the aerobic counterparts, the costs of aeration and sludge handling in anaerobic treatment are dramatically lower as no oxygen is needed and sludge yield is lower. Notwithstanding these advantages, the widespread application of conventional anaerobic biological systems has been limited. This is mostly due to the biomass retention dilemma. Biomass retention is one of the most important aspects of anaerobic technology providing sufficient solid retention time (SRT) for the methanogens. On one side, the net biomass production is low, up to ten times less than that of aerobic treatment. On the other side, the relatively poor settling properties of the biomass in conventional anaerobic biological treatment systems would result in the loss of biomass to effluent. This situation corresponds to the poor biomass retention in the conventional anaerobic biological system. While biofilm or granule formation offers the strategy for biomass retention in modern high-rate anaerobic reactors (HRARs), it usually requires a long start-up period, and is a complex process that involves physico-chemical as well as biological interactions, and has proven to be much more problematic under conditions of high or low temperature, or for the treatments of low strength and/or salinity wastewater [1,2].

Over the past 15 years, the use of membranes in aerobic biological waste treatment processes has been well established. A complete retention of all microorganisms in the bioreactor can be achieved in membrane bioreactors (MBRs) by the use of microfiltration (MF) or ultrafiltration (UF) modules. The MBR technology also offers advantages in terms of reduced footprint, capacity of handling wide fluctuations in influent quality and improved effluent quality. Since membranes work well with aerobic processes, it should be possible to extend to anaerobic processes. This is of particular interest for anaerobic processes that depend on the retention of a large population of slow growing microorganisms.

Due to its unique advantages, anaerobic membrane bioreactor (AnMBR), which combines anaerobic process and membrane technology, is attracting remarkable interest in both research community and industrial sectors. A careful literature review shows that more than 250 peer-reviewed English papers regarding AnMBR technology have been published, and more than 100 out of them have been published just in the last 6 years. Although much research has been conducted on this topic, studies have generally been limited to single treatment system, and there are still some challenging issues regarding AnMBR systems, particularly membrane fouling problem. Therefore, it is necessary to summarize and compare the results obtained in the literature in order to provide an overview of the findings. Several review papers (most were recent) were available in the literature, which focused on applications on various wastewater treatment [3] or only on industrial wastewater treatment [4], parameters governing permeate flux [5], effect of operational conditions [6], and anaerobic bioprocess [7] in AnMBR systems, respectively. While these reviews extended our understanding of AnMBR, they didn't comprehensively address application concerns and membrane fouling issues, nor did they cover the updated studies simultaneously. With the rapid development of AnMBR technology, a detailed and comprehensive analysis of past academic research progress could be valuable.

The objective of this review was then to conduct a comprehensive literature survey to the recent (mainly from 2006 onwards) application progress and membrane fouling issues regarding AnMBR technology. Accordingly, fundamental aspects of AnMBR would be introduced and discussed. The developments in applications, researches on membrane fouling mechanisms, factors and control measures would also be reviewed and discussed. Finally, the main conclusions and the future perspectives were presented.

2. Fundamentals of AnMBR

2.1. Advantages and disadvantages

An AnMBR can be simply defined as a biological treatment process operated without oxygen and using a membrane to provide solid–liquid separation. The advantages offered by this process over conventional

Table 1
Comparison of conventional aerobic treatment, anaerobic treatment, aerobic MBR and AnMBR.

Feature	Conventional aerobic treatment	Conventional anaerobic treatment	Aerobic MBR	AnMBR
Organic removal efficiency	High	High	High	High
Effluent quality	High	Moderate to poor	Excellent	High
Organic loading rate	Moderate	High	High to moderate	High
Sludge production	High	Low	High to moderate	Low
Footprint	High	High to moderate	Low	Low
Biomass retention	Low to moderate	Low	Total	Total
Nutrient requirement	High	Low	High	Low
Alkalinity requirement	Low	High for certain industrial stream	Low	High to moderate
Energy requirement	High	Low	High	Low
Temperature sensitivity	Low	Low to moderate	Low	Low to moderate
Start up time	2–4 weeks	2–4 months	<1 week	<2 weeks
Bioenergy recovery	No	Yes	No	Yes
Mode of treatment	Total	Essentially pretreatment	Total	Total or pretreatment

anaerobic systems and aerobic MBR are widely recognized [3,8–10]. Table 1 presents the comparison of conventional aerobic treatment, anaerobic treatment, aerobic MBR and AnMBR. It is apparent from Table 1 that AnMBR technology combined the advantages of anaerobic treatment and MBR technology. Among these, the ones most often cited are: total biomass retention, excellent effluent quality, low sludge production, a small footprint and net energy production.

AnMBR systems were essentially implemented based on two configurations: external/side-stream configuration and submerged/immersed configuration. Generally, the external configuration provides more direct hydrodynamic control of fouling, and offers the advantages of easier membrane replacement and high fluxes but at the expense of frequent cleaning and high energy consumption (of the order 10 kWh/m^3 product) [11]. Moreover, high cross-flow velocity has been reported to have a negative impact on biomass activities in AnMBR systems [12–14]. Compared to external configuration, submerged configuration directly places the membrane into the liquid. A pump or gravity is used to drag the permeate through the membrane. Several distinct advantages of submerged configuration are their much lower energy consumption and fewer rigorous cleaning procedures, as well as the milder operational conditions due to the lower tangential velocities.

Table 2
Main membrane materials and modules used in AnMBR studies.

Membrane material	Module configuration	Nominal pore size/μm	Manufacturer	Reference
PVDF	Hollow fiber	0.04	GE, USA	[37]
PVDF	Hollow fiber	100 kDa	Koch, USA	[38]
PVDF	Flat sheet	70 kDa, 140 kDa	SINAP, China	[9,30,39]
PVDF	Tubular	0.03	Norit X-Flow, Inc. Netherlands	[40]
PVDF	Tubular	0.1	PCI Membrane Systems, Inc. USA	[41]
PES	Flat sheet	20–70 kDa	SINAP, China	[10]
PES	Tubular	20 kDa	Weir Envig, Paarl, South Africa	[42]
PE	Flat sheet	0.4	Kubota Corporation, Japan	[43]
PE	Hollow fiber	0.4	Mitsubishi Rayon, Japan	[44]
PP	Hollow fiber	0.45	Sumitomo Electric Fine Polymer Inc., Japan	[25]
PSF	Tubular	0.2	Triqua, Netherlands	[26,27]
Ceramic	Tubular	40 kDa	Aquatech Memtuf©, Korea	[32]
Ceramic	Tubular	0.2	Atech Innovations, Germany	[45]
Metal	Tubular	1.0	Fibertech Co., Ltd, Korea	[21]

2.2. Membrane materials and modules

The membrane materials can be classified into three major categories: polymeric, metallic and inorganic (ceramic). Ceramic membranes can be backwashed effectively providing high resistance to corrosion, abrasion, and fouling as well as increased concentration polarization control [15,16]. Ghyoot and Verstraete [14] found that a commercial ceramic MF membrane reached 200–250 L/m^2/h (LMH), which was 10-fold higher than the flux achieved with a polymer UF membrane, with both membranes producing permeate of similar quality for filtration of anaerobic sludge. In this respect, ceramic membranes appeared to be most widely used in early studies regarding AnMBR [14,17–19]. Meanwhile, metallic membranes have also been used in the AnMBR system, showing better hydraulic performance, better fouling recovery, and higher strength endurable impact force and tolerance to oxidation and high temperature compared to polymeric membrane [20,21]. However, ceramic or metallic membranes are much more expensive than polymeric membranes. As economics of a system was gradually becoming a great concern (this is particular true for commercial applications), polymeric membranes gained more interests in both research community and commercial applications in recent years. The preferred polymeric membrane materials are polyvinylidene difluoride (PVDF) and polyethersulfone (PES), which account for around 75% of the total products on the market including 9 out of the 11 most commercially important products [22]. Other polymeric materials, such as polythylene (PE) [23], polypropylene (PP) [24,25] and polysulfone (PSF) [26,27], are also used for some cases of AnMBR applications.

Most membrane modules used in AnMBRs are implemented by using MF or UF membranes, with the configuration of either hollow fiber, flat sheet (plate or frame) or tubular. Due to their high packing density and cost efficiency, hollow fiber membrane modules are most popularly used in SMBRs. However, flat sheet membrane modules also retained significant interests, especially from research community [9,28–31], for their advantages of good stability, and the ease of cleaning and replacement of defective membranes. A tubular membrane module is made up of several tubular membranes arranged as tubes. The main advantages include low fouling, relatively easy cleaning, easy handling of suspended solids and viscous fluids and the ability to replace or plug a damaged membrane, while the disadvantages include high capital cost, low packing density, high pumping costs, and high dead volume. Its applications in AnMBR can be found in many literature studies [32–36]. Table 2 summarized the main membrane materials and modules used in AnMBR studies. As can be seen from Table 2, most membranes used have pore size ranged from 0.03 to 1.0 μm, which obviously lower than the size of the most flocs or microorganisms in AnMBR, and therefore can almost completely retain biomass.

Lin et al. [30] reported that membrane costs accounted for 46.4–72.3% fraction of total capital costs of a full scale AnMBR treating municipal wastewater under different assumptions, indicating significant costs of introduction of membrane modules into the anaerobic system due to the relative high costs of membrane. Since membranes only serve for solid and liquid separation, and an improved effluent quality might not always be required, developing low cost filters applied in AnMBRs would be very desirable. The low-cost filters investigated include non-wovens, meshes and filter cloths as summarized by Meng et al. [46]. Although some applications were based on aerobic MBRs, it has been confirmed that these filters can also be applied in AnMBRs [47,48]. These filters generally have large pore size or porosity, and therefore could obtain a high initial flux even at a very low pressure [49], but should have a shorter lifetime as compared with polymeric membranes due to their lower tensile and tear strength. Moreover, application of these filters in AnMBRs encountered the severe fouling problem [48], and this was mainly caused by the inadequate fouling controls, and also can be attributed to their rough surface and the too large pore size. It has been reported that precoating the filter cloth with powdered activated carbon (PAC) could mitigate membrane fouling [50]. It also indicates that the severe fouling of low-cost filters can be resolved by modifying the filters to improve the surface roughness, hydrophilicity, surface charge and so on [46]. Meanwhile, self-forming dynamic membranes, which employed cheap coarse pore-sized materials such as Dacron mesh [51,52], non-wovens [53], stainless steel mesh [54], etc., as filtration media, have been extensively investigated. The sludge cake layer and gel layer that dynamically formed on the filtration medium were found effective in enhancing the solid–liquid separation, and the effluent quality could be kept at a stable level with undetectable suspended solid (SS) concentration [51,52], suggesting a promising material for separation in AnMBRs.

Another promising membrane process would be forward osmosis (FO). FO is an osmotic process that uses a semi-permeable membrane to effectively separate water from dissolved solutes by using high concentration draw solution. Because FO uses the osmotic pressure differential across the membrane, rather than hydraulic pressure differential as the driving force for transport of water through the membrane, it provides recognized advantages including operating at low or no hydraulic pressures, high rejection of a wide range of contaminants, and lower membrane fouling propensity as compared to conventional pressure-driven membrane processes [55], and has emerged as an alternative membrane process to the conventional membrane processes in the recent years [55]. Holloway et al. [56] used FO process for concentration of anaerobic digester centrate, and found that high water flux (initial flux = 10.5 LMH) and high nutrient rejection (> 90% nitrogen and phosphorus rejection) could be achieved, showing the potential of using FO process in AnMBR. However, the high costs of FO membrane or process must be reduced to improve its economic feasibility. Also, effect of salt accumulation on biological activity should be addressed.

2.3. History and commercial development

The AnMBR concept appears to be firstly reported by Grethlein [57] who used external cross-flow membrane to treat septic tank effluent, and achieved increased biomass concentration with 85–95% biochemical oxygen demand (BOD) reduction and 72% nitrate removal simultaneously. With 3 decades development, the advantages of the AnMBR systems have been well proven in the literature. Recognizing the value of AnMBR, both the private sectors and the governments have made considerable investments in promoting AnMBR systems. The notable efforts were the development of commercially-available AnMBR systems known as the "Membrane Anaerobic Reactor System (MARS)" [58] and "Anaerobic Digestion Ultrafiltration (ADUF)" [59] in the 1980s. These systems have been tested and operated in pilot- and full-scale, and mostly used for industrial wastewater treatment. During the same period, Japan government initiated a national project "Aqua-Renaissance '90" which led to the development of a wide variety of AnMBR systems [60–62]. These commercially-available AnMBR systems were mostly implemented based on external configuration. By the 2000s, studies on the AnMBR focused on system performance, filtration characteristics, characterization of membrane foulants, and membrane fouling control.

The success of submerged aerobic MBRs in the early 2000s highly encouraged the exploration of submerged AnMBRs (SAnMBRs) for wastewater treatment. In the last decade, Kubota Corporation developed a SAnMBR named "KSAMBR" process, which has been successfully applied in a number of full-scale food and beverage industries [31]. Using the similar technology, ADI Systems Inc. developed ADI-AnMBR system specific for food wastewater treatment. The largest AnMBR installation up to date in the world was completed by ADI, which produced effluent free of suspended solids (SS) and with 99.4% COD removal, allowing 100,000 gal/d of wastewater to be easily discharged into the municipal system [63]. Later in 2010s, the submerged AnMBR treatment was significantly studied with attempts made to improve energy efficiency, extend the application scope and solve technical problems such as membrane fouling.

3. Applications in various wastewater treatment

3.1. Treatment of various wastewaters

A detailed review shows that more and more attention and efforts of individuals and research organizations have been dedicated in AnMBR research, especially in the last 6 years. This situation may be attributed to two trends of wastewater treatment. On one side, the industrial sectors have been facing with stringent requirements on its increasing water use efficiency and closing industrial process water cycles and the same trend will continue in the future. Meanwhile, the extreme conditions of wastewater are likely to become more common in these years and in the future. On the other side, while the costs for conventional technologies are slowly rising with labor costs and inflationary pressures, the costs for all membrane equipment have been falling steadily during the last decade. Moreover, biogas recovery associated with AnMBR treatment can create benefits which will significantly offset the operational costs. On a capital and operational cost basis for any given project, the likelihood of AnMBR becoming a favored option is increasing with time. In this section, these AnMBR applications will be reviewed together with their state of the art in the wastewater treatment.

3.1.1. Synthetic wastewater treatment

It is a common operation using synthetic wastewater to test new concepts or study general aspects of membrane fouling [3]. Most recent studies regarding AnMBR used synthetic wastewater as feed. This is reasonable, considering that AnMBR, especially SAnMBR, is kind of a novel solution for wastewater treatment, and membrane fouling is a major issue of AnMBR research. Table 3 presents some recent relevant references regarding AnMBR systems treating synthetic wastewater. Various substrates have been used to make feed, including glucose, starch, molasses, peptone, yeast, and volatile fatty acids. Due to the absence of refractory compounds, the chemical oxygen demand (COD) removal by AnMBR was generally higher than 95%. The applied organic loading rate (OLR) varied depend on research purposes. OLR was generally high when synthetic wastewater was used to test the removal efficiency or processing capacity of an AnMBR. In theory, AnMBR can achieve same high OLR (usually > 10 kg COD/m^3/d) achieved by HRARs, such as upflow anaerobic sludge blanket (UASB) reactors, hybrid UASB reactors, and expanded granular sludge bed (EGSB) reactors. However, most studies regarding AnMBR applied OLR < 10 kg COD/m^3/d. This can be attributed to several aspects associated with the operation of AnMBRs. By far, most of AnMBRs studied used a completely stirred tank reactor (CSTR) configuration due to the ease of use and construction. Such a configuration was usually operated at a lower biomass concentration compared to HRARs, corresponding to a lower OLR. Moreover, for research studies, it may not be necessary to operate AnMBRs with high biomass concentration and OLR since membrane fouling was of the main research focus and high biomass concentration or OLR would hinder the sustainable AnMBRs operation. From these studies, it should be concluded that AnMBR is a promising technology in terms of high organic degradation.

3.1.2. Industrial wastewater treatment

Rapid industrialization has resulted in the generation of a large quantity of effluents which include the major sources of industrial wastewaters from food processing, pulp and paper, textile, chemical, pharmaceutical, petroleum, tannery, and manufacturing industries. Industrial wastewater is usually characterized by high organic strength and/or extreme physical–chemical nature (e.g., pH, temperature, salinity), and containing synthetic and natural substances that may be toxic to and/or inhibit biological treatment processes.

Table 4 summarizes some significant recent examples of AnMBR applied to treat some kind of industrial wastewaters. The most popular application area appears to be food industrial wastewater. A review of literature showed that wastewaters from food industry are generally biodegradable and nontoxic, and have high concentrations of COD and SS [69]. Liao et al. [3] stated that the extensive opportunity for AnMBR is to treat high organic strength and highly particulate wastewater. The characteristics of food industrial wastewater render it much more suitable for AnMBR treatment. Generally, COD removal efficiency achieved was higher than 90%, while the applied OLR was in the range of 2–15 kg COD/m^3/d. As most of the applied AnMBRs used CSTR configurations, the achievable OLR would be lower than the HRAR, but higher than the conventional CSTR digesters. For instance, Kubota Corporation developed a SAnMBR system named "KSAMBR" process, which has been successfully applied in a number of full-scale food and beverage industries [31]. The process has the volume which can be scaled down to around 1/3 or 1/5 of the conventional digesters provided that biomass is 3 to 5 times as concentrated, corresponding to 3 to 5 times OLR based on volume if the same flow rate applied.

Treatment of pulp and paper industry wastewater by AnMBR has been reported at least for 10 times, and recent studies were mostly

Table 3
Summary of AnMBR performance for synthetic wastewater treatment.

Type of wastewater	Scale[a]	Configuration	Characteristics of membrane[b]	Type of reactor[c]	Reactor volume (L)	Operating condition	Influent[d]	Effluent[e]	Reference
Tapioca starch wastewater	L	External	Hollow fiber UF membrane Pore size: 0.03–0.15 μm	AF + M	1	HRT = 10 d Temp = 30 °C OLR = 1.76 kg COD/m^3/d	COD = 20.15	COD = 675–780 (>95%)	[64]
Meat extract + peptone	L	Submerged	Flat-sheet PE membrane, Pore size: 0.4 μm	CSTR + M	3	HRT = 6 h SRT = 150 MLVSS = 2.62 ± 0.13 g/L Temp = 35 ± 1 °C Flux = 10 LMH	COD = 0.45 ± 0.02	COD$_s$ = 18 ± 9 (95%)	[8]
Whey + sucrose	L	Submerged	Flat-sheet membrane, Pore size: 0.4 μm	CSTR + M	11	SRT = 30–40 d MLVSS = 5.5–20.4 g/L OLR = 1.5–13 kg COD/m^3/d Temp = 35 ± 1 °C Flux = 2–5 LMH	–	–	[65]
Glucose + peptone + yeast extract	L	External	Tubular MF membrane Zirconia pore size: 0.14 μm PP pore size: 0.2 μm	CSTR + M	4.5	HRT = 6.5 d Temp = 54–56 °C OLR = 4 kg COD/m^3/d	COD = 27.0 K$_j$–N = 1.288	COD$_t$ = – (78.5–84.4%)	[28]
Glucose	L	External	Hollow fiber PE membrane Pore size: 0.4 μm	CSTR + M	25	MLSS = 3.5 g/L	COD = 0.8	–	[66]
Glucose	L	External	Flat-sheet membrane, Pore size: 0.45 μm	CSTR + M	5	HRT = 12 h SRT = 30 d Flux = 5.3 LMH OLR = 1.1 kg COD/m^3/d pH = 6.8–7.0 MLVSS = 5.132 g/L Temp = 25–30 °C	COD = 0.55	COD = – (99.1%)	[67]
Volatile fatty acid	L	External	Tubular ceramic aluminum oxide (Al_2O_3) membrane Pore size: 0.2 μm	CSTR + M	2	SRT = 120 d MLVSS < 21 g/L OLR = 10–55 kg COD/m^3/d Temp = 55 °C Flux = 20–40 LMH	COD = 10	–	[45]
Maltose + glucose + volatile fatty acid	L	Submerged	Hollow fiber PP membrane Pore size: 0.45 μm	CSTR + M	0.6	HRT = 14 d MLVSS = 19.5 g/L OLR = 2.5 kg COD/m^3/d Temp = 35 °C	COD = 25	COD = 95.1 ± 8.6 (99.6 ± 0.0%)	[25]
Molasses	L	External	Tubular ceramic membrane, Pore size: 0.1 μm	CSTR/ CSTR + M	3/6	HRT = 16/32 h MLVSS = 1.8/10 g/L OLR = 14.9/5.6 kg COD/m^3/d Temp = 55/55 °C pH = 5.5/7.2	COD = 10.2/ 7.5	COD = - (78–81%)	[68]

[a] L = laboratory/bench scale.
[b] PE = polythylene and PP = polypropylene.
[c] CSTR = completely stirred tank reactor and AF = anaerobic filter.
[d] The concentration unit is g/L if not specified and – indicates value not reported.
[e] The concentration unit is mg/L; removal efficiency is presented in parentheses; COD$_s$ = soluble COD, and COD$_t$ = total COD.

conducted by Lin and Liao group [9,39,70–72]. Evaporator condensate (EC), one of the important wastewaters produced from pulp and paper industry, is characterized as high temperature, high organic strength (due mainly to methanol), low SS (<3 mg/L), plus inhibitive materials such as total reduced sulfur (TRS) compounds and turpene oils [62]. Xie et al. [71] used a SAnMBR operated at 37 ± 1 °C to treat kraft EC for 9 months. Under tested OLRs of 1–24 kg COD/m^3/d, a COD removal efficiency of 93–99% was achieved. Wastewater from pulp and paper industry is usually high temperature, therefore, operation at thermophilic temperatures is of great interest because pre-cooling and post-heating used in the mesophilic treatment for subsequent reuse of treated effluent could be avoided. Lin et al. [9] compared two parallel SAnMBR treating kraft EC which were operated at mesophilic (37 °C) and thermophilic (55 °C), respectively, and found that a COD removal efficiency of 97–99% with good methane production was achieved at a feed COD of 10,000 mg/L in both SAnMBRs. The results indicated that both the mesophilic and thermophilic SAnMBRs can be potentially promising technologies for kraft EC treatment in terms of COD removal and biogas production. However, thermophilic SAnMBRs faced challenge of sever membrane fouling because a high temperature induced more release of SMP and disruption of sludge flocs [9]. Thermomechanical pulp (TMP) is produced by refining wood chips at temperatures above 100 °C, and TMP whitewater is warm, normally with temperatures between 50 °C and 80 °C, with a COD of 1000–5600 mg/L [73]. Gao et al. [39] investigated TMP whitewater treatment with a SAnMBR at an average OLR of 2.4 kg COD/m^3/d. Without pH shocks, the steady-state COD removal efficiency was found to be about 90%, yielding an effluent with COD < 300 mg/L. The total cost of AnMBR for treatment of kraft mill effluent was found to be much lower than that for aerobic treatment [61,62]. The capital and operating costs of an aerobic MBR operated at high-temperature (60 °C) for foul condensate treatment were significantly lower than the operational costs of a steam stripping system [74]. AnMBR treating petrochemical effluent has been reported twice in last 6 years, one is laboratory-scale [75] and the other is pilot-scale [76]. Fischer–Tropsch Reaction Water (FTRW) is a typical petrochemical wastewater characterized by high strength and consisting of short chain organic acids other oxygenates with a low pH. It was convincingly proven that anaerobic granules did not readily form with FTRW, and the fixed media systems had effluent quality concerns [76]. AnMBR guaranteed completed biomass retention,

Table 4
Summary of AnMBR performance for industrial wastewater treatment.

Type of wastewater	Scale[a]	Configuration	Characteristics of membrane[b]	Type of reactor[c]	Reactor volume (L)[d]	Operating condition	Influent[e]	Effluent[f]	Reference
Cheese whey	L	External	MF pore size: 0.2 μm	CSTR/ CSTR + M	5/15	HRT = 1 d/4 d SRT = –/29.7–78.6 d MLVSS = –/6.4–10 g/L OLR = –/19.78 kg COD/m^3/d Temp = 37 ± 2/37 °C Flux = 139.5 LMH	COD = 68.6 ± 3.3 BOD_5 = 37.71 ± 2.84 K_j–N = 1.12 ± 0.01 TP = 0.5 ± 1.8 × 10^{-3} TSS = 1.35 ± 0.06 pH = 6.5	COD = – (98.5%) BOD_5 < 100 (99.2%) TSS = – (100%)	[78]
Diluted tofu processing waste	L	External	Hollow fiber MF membrane	CSTR + M	5	HRT = 4 h RNA concentration = 150–200 mg/L Temp = 60 ± 0.1 °C pH = 5.5 ± 0.1 Flux = 4.32 LMH	COD = 26.5 ± 2.2 NH_4^+–N = 0.86 ± 0.12 PO_4^{3-}–P = 0.58 ± 0.06 TSS = 23.5 ± 3.5 pH = 1.0	Carbohydrate content < 2 g/L	[79]
Olive-mill wastewater	L	External	Ceramic tubular UF 25 kDa MWCO	PABR + M	15	HRT = 16.67 h MLSS = 1.05–2.41 g/L Temp = 35 ± 2 °C Flux = 80–450 LMH	COD = 350–500 NH_4^+–N = 15–21 PO_4^{3-}–P = 3–4.5 SS = 1–1.5 pH = 6.5–7.8	COD < 30 (> 95%) TN = 9–9.85 (15–20%) PO_4^{3-}–P < 1 (81%) pH = 6.9–7.3	[35]
Brewery wastewater + surplus yeast	L	External	Ceramic tubular; Pore size: 0.2 μm	CSTR + M	4.5	OLR = 12 kg COD/m^3/d MLVSS = 12 g/L Flux = 4–20 LMH Temp = 30 °C pH = 6.9	COD_s = 21 Particulate COD = 45–50	COD = 190 (99%) TSS = 0 (100%)	[34]
High-concentration food wastewater	P	External	Flat-sheet PES 20–70 kDa MWCO	CSTR + M	400	HRT = 60 h SRT = 50 d pH = 7.0 ± 0.2 OLR < 4.5 kg COD/m^3/d MLSS = 6–8 g/L Temp = 37 ± 0.5 °C	COD = 2–15 SS = 0.6–1.0 Chromaticity color = 6000–1000 pH = 5–6	COD = 141–2388 (81.3–94.2%)	[10]
Distilley produces wastewater	F	Submerged	Kubota flat-sheet membrane	CSTR + M	–	Themophilic range	COD = 101.3 TN = 3.72 TS = 6.0% pH = 4.11	COD = – (75–92%)	[31]
Kraft evaporator condensate	L	Submerged	Flat-sheet PVDF membrane 140 kDa MWCO	UASB + M	10	HRT = 5.8 d SRT = 230 d MLSS = 8.3 ± 1.6 g/L OLR = 3.1 ± 0.8 kg COD/m^3/d Temp = 55 ± 1 °C Flux = 2.4 ± 0.6 LMH	COD_t = 10	COD_s = – (97–99%)	[9]
Kraft evaporator condensate	L	Submerged	Flat-sheet PVDF membrane 140 kDa MWCO	UASB + M	10	HRT = 1.93 d SRT = 230 d MLSS = 8.2 ± 1.5 g/L OLR = 12.2 ± 1.1 kg COD/m^3/d Temp = 35 ± 1 °C Flux = 7.2 ± 0.9 LMH	COD_t = 10	COD_s = – (97–99%)	[9]
TMP whitewater	L	Submerged	Flat-sheet PVDF membrane 140 kDa MWCO	UASB + M	10	MLSS = 5.7 ± 0.8 g/L OLR = 2.4 ± 0.4 kg COD/m^3/d Temp = 35 ± 1 °C Flux = 5.2 ± 0.5 LMH	CODs = 2.78–3.35	COD_s < 300 (90%)	[39]
Petrochemical wastewater	L	Submerged	Kubota flat panel membrane Pore size: 0.45 μm	CSTR + M	23	HRT = 31.5 h SRT = 175 d MLSS > 30 g/L OLR = 14.6 kg COD/m^3/d Temp = 37 °C Flux = 8.5–16 LMH	COD = 19.1 pH = 7.2	COD = 612 (98%)	[75]
Textile wastewater	L	Submerged	Hollow fiber MF membrane Pore size: 0.40 μm	CSTR + M	3.25	HRT = 24 h pH = 6.8–7.2 Temp = 35 °C Flux = 1.8–14.4 LMH PAC dose = 1.7 g/L	COD = 730–1100	COD = – (90%) Color = – (94%) Turbidity = 8 NTU	[80]

[a] L = laboratory/bench scale, P = pilot scale, and F = full scale.
[b] PVDF = polyvinylidine fluoride and PES = polyethersulfone.
[c] CSTR = completely stirred tank reactor, UASB = upflow anaerobic sludge blanket, and PABR = periodic anaerobic baffled reactor.
[d] – indicates value not reported.
[e] The concentration unit is mg/L if not specified; COD_s = soluble COD, and COD_t = total COD.
[f] The concentration unit is mg/L and removal efficiency is presented in parentheses.

and OLR up to 25 kg COD/m^3/d was achieved with effluent COD normally < 500 mg/L with no particulates > 0.45 μm [75]. Moreover, no noteworthy deterioration in membrane performance has been observed over the 320 d operational period when operated at a low membrane flux of 1.5–3.5 LMH [75]. Textile treatment by using AnMBR has been reported only once [77]. A SAnMBR combined with PAC addition could achieve the median removal efficiencies of COD and color with 90% and 94%, respectively [77].

The cases of using AnMBR system for treatment of other industrial wastewaters were very limited. More often, the combinations of anaerobic unit and aerobic MBR were applied. For refractory wastewaters, anaerobic treatment governed by hydrolysis and acidification is usually proposed to ameliorate the biodegradability of wastewater feed to aerobic MBR [81]. Such combined systems have been tested for treatments of textile wastewater [82], pharmaceutical wastewater [83], oil refinery wastewater [84], and coke plant wastewater [85]. Enhanced removal of contaminations was essentially evidenced in these studies as compared to the single unit. Meanwhile, anaerobic treatment without membranes has been applied successfully to treat various industrial wastewaters [86], As long as a wastewater was amenable to anaerobic treatment, in theory, an AnMBR could be used to treat it [3]. In this context, additional attentions should be paid to improve membrane performance and the economic feasibility of this technology.

3.1.3. Municipal wastewater treatment

Historically, anaerobic processes have been mainly employed for industrial or high strength wastewater treatment while less employed for municipal wastewater treatment [19,87]. This may mainly due to 2 issues. The first one is the difficulty in retaining slow-growth anaerobic microorganisms with short hydraulic retention time (HRT) associated with treatment of low strength wastewater like municipal wastewater. The second one is that anaerobic effluents rarely meet discharge standards for wastewater reuse due to the kinetic limitations of anaerobic metabolism [32]. The combination of membrane separation technology and an anaerobic bioreactor may allow for a sustainable municipal wastewater treatment with complete biomass retention, the added benefits of lower sludge production, enhanced high quality effluent, net energy production, and without the extra costs for aeration associated with the aerobic treatment processes [87–90]. AnMBR technology is becoming increasingly popular for municipal wastewater treatment in recent years [48,88,90].

There are many cases in the literature investigating the efficiency of the AnMBR technology for the treatment of municipal wastewater. Table 5 exemplifies recent researches on the application of AnMBR treating municipal wastewater. With respect to the removal of common contaminants, AnMBR systems could typically eliminate around >85% COD, and >99% TSS at selected operational conditions regardless of the configurations. The removals were much higher than those of the conventional UASB sewage treatment which usually resulted in a BOD removal efficiency of 80%, effluent COD of 100–220 mg/L, and effluent total suspended solids (TSS) of 30–70 mg/L [91], and comparable with aerobic MBR treatment. This is probably not surprising, considering that the typical pore sized of the membrane used was in the range of 0.01–0.45 μm (Table 5), the SS, most colloids and some organic matters could be readily retained by the membrane and the cake layer formed on the membrane surface. Due to the complete retention of sludge by the membrane and application of longer SRT (e.g., 217 d [89]), the retained pollutants may be efficiently removed in AnMBRs. COD removal will decrease when membrane pore size increases. This is apparent from Zhang et al.'s study [52] where a reduced COD removal of 57.3 ± 6.1% was observed due to the utilization of dynamic membrane for separation.

In contrast to the high COD and TSS removal, the removal of total nitrogen (TN) or total phosphorus (TP) in the AnMBR systems is usually negligible (Table 5). The low removal of TN and TP is expected because both of TN and TP removal processes required anoxic or aerobic zone. This can be beneficial if the effluent is to be used for agriculture or irrigation purpose. However, in most cases, this means that the downstream treatment is needed if the effluent is to be reclaimed. Coupling AnMBRs with conventional biological nutrient removal treatment technologies will face challenges due to the low COD:N and COD:P ratios typical of AnMBR effluents. Partial nitritation/nitrification would be a promising solution for nutrient removal because ammonium could serve as the electron donor, and no additional carbon source/electron donor is required in such process [93]. FO membrane process could provide another perspective to resolve this challenge since FO process can almost totally reject N and P contaminants. Physical/chemical nutrient removal processes could be other solutions although they are significantly more energy intensive than biological treatment.

The occurrence of trace contaminants such as endocrine disrupting chemicals (EDCs) and pharmaceutically active compounds (PhACs) in treated and untreated municipal wastewater has recently become a significant environmental health concern [94]. It has been reported that removal rate of the EDCs and PhACs during anaerobic digestion is low [95,96]. Ifelebuegu [96] reported the EDCs persisted in the anaerobic digestion process with percentage removal of 21–24% for steroidal estrogens (E1), 18–32% for 17β-estradiol (E2), 10–15% for 17α-ethynylestradiol (EE2) and 44–48% for nonylphenol (NP). It is worth noting that prolonging HRT and bioaugmentation would improve the removal efficiency. Under anaerobic conditions and relatively long HRT (30 d), some PhACs (acetylsalicilic acid (ASA), ibuprofen (IBU), fenofibrate (FNF)) can be significantly degraded [97]. Saravanane and Sundararaman [98] applied an AnMBR system to treat pharmaceutical wastewater containing cephalosporin derivative, and achieved enhanced degradation (attained a removal of 81% at a maximum cephalosporin concentration of 175 mg/L) through bioaugmentation. The principal mechanism of removal of these trace contaminants during the sludge process has been demonstrated to be biodegradation by microorganisms and also sorption onto biomass [96].

With respect to operational conditions, HRT of AnMBR is generally longer than 8 h (Table 5), comparing favorably with conventional anaerobic systems [3], while longer than 4–8 h for aerobic MBRs, which corresponds to a less OLR (<3 kg COD/m^3/d) as compared to aerobic MBRs. The sustainable membrane flux applied in most AnMBR studies appeared to be lower than 15 LMH. In contrast, this value for aerobic MBR ranged from 25 to 140 LMH and 3.7–85 LMH for external and submerged configuration, respectively [69]. The low sustainable membrane flux would be a bottleneck to the practical engineering application of AnMBR. Through the formation process of dynamic membrane on the Dacron mesh (pore size = 61 μm), a high flux of about 65 LMH was achieved at an anaerobic dynamic membrane bioreactors (AnDMBR) [52]. Given the relative high cost of UF or MF membranes, and their low sustainable flux achieved, AnDMBR seems to be a promising solution for municipal wastewater treatment.

It was found that the unit capital costs of SAnMBR treating municipal wastewater was about 800 US$/m^3/d capacity [30], which compares favorably to the literature values for full-scale aerobic MBRs [99]. The total operational cost value was only 1/3 of the aerobic counterpart at the similar capacity [100]. Moreover, operational costs can be totally offset by the benefits from biogas recovery. Cost sensitive analysis showed that membrane parameters including flux, price and lifetime play decisive roles in determining the total life cycle costs of the SAnMBR [30]. SAnMBR can be a promising technology for municipal wastewater treatment, provided that membrane performance is significantly improved.

3.1.4. Other stream treatment

Other streams, which have been used in treatment by AnMBRs, can be mainly classified into two categories: high-solid-content streams and leachate. The former includes wastewater treatment plant sludge, the organic fraction of municipal solid waste, animal processing plant effluents, and manures. It is widely accepted that the hydrolysis or solubilization stage represents the rate-limiting step in the anaerobic degradation of most solid organic materials [47]. Hydrolysis proceeds slowly even at optimal conditions, and thus long SRT are required. For the conventional anaerobic digestion process which does not decouple SRT from HRT, long SRT means a large reactor volume and lower OLR, and thus reduces its competitiveness.

It can be seen from Table 6 which summarizes AnMBR applications in the high-solid-content streams, the applied HRT ranged at 1.5–11.8 d, which was rather higher than the values applied in industrial or municipal wastewater treatment. This indicated that for particulate stream treatment, a relatively long HRT may be necessary to ensure significant

Table 5
Summary of AnMBR performance for municipal wastewater treatment.

Type of wastewater	Scale[a]	Configuration	Characteristics of membrane[b]	Type of reactor[c]	Reactor volume (L)	Operating condition	Influent[d]	Effluent[e]	Reference
Municipal wastewater	L	Submerged	Flat-sheet MF PVDF 140 kDa MWCO	CSTR	60	HRT = 10 h MLSS = 6.4–9.3 g/L OLR = ~1.0 kg COD/m^3/d Temp = 30 ± 3 °C Flux = 11 LMH	COD = 425 ± 47 NH_4^+–N = 32.4 ± 11.6 NO_3^-–N = 1.3 ± 0.4 TP = 4.3 ± 0.5 SS = 294 ± 33 pH = 7.6 ± 0.3	COD = 51 ± 10 (88 ± 2%) NH_4^+–N = 31.1 ± 12.3 (~0%) NO_3^-–N = 1.1 ± 0.6 (~0%) TP = 3.8 ± 0.7 (~0%) SS<0.8 (>99.5%) pH = 7.0 ± 0.2	[30]
Municipal wastewater	L	Submerged	Flat-sheet dynamic membrane, Dacron mesh	UASB	45	HRT = 8 h MLSS = 5.9–19.8 g/L OLR = ~0.9 kg COD/m^3/d Temp = 10–15 °C Flux = 65 LMH	COD = 302.1 ± 87.9 NH_4^+–N = 37.9 ± 8.6 TN = 58.8 ± 10.2 SS = 120 ± 23 pH = 7.3 ± 0.3	COD = 120.8 ± 34.0 (57.7 ± 4.6%) SS = 0–15 pH = 7.2–7.6	[52]
Municipal wastewater	L	External	Tubular UF membrane 40 kDa MWCO	UASB	1	HRT = 3 h SRT = 100 d Temp = 25 °C Flux<7 LMH	COD_t = 646 ± 103 COD_s = 385 ± 63 TSS = 140 ± 18 $MPN_{Fecal\ coliforms}$ = 10^6/100 ml	COD_t = 104 ± 12 (87%) COD_s = 104 ± 12 (73%) BOD = 32 ± 5 TSS<1 $MPN_{Fecal\ coliforms}$ = 0 (100%)	[32]
Dilute Municipal Wastewater	L	External	PVDF; pore size: 0.1 μm, 200 kDa MWCO	CSTR	10	HRT = 12–48 h SRT = 19–217 d OLR = 0.03–0.11 kg COD/m^3/d MLSS = 1–7.3 g/L Temp = 25 °C pH = 6.4 ± 0.2	COD_s = 38–131 pH = 7.5	COD_s = 18–37 (55–69%) NH_4^+–N = 8.9–51.8 NO_3^-–N<0.4 (0%) NO_2^-–N<0.4 (0%) pH = 6.6 ± 0.1	[89]
Domestic wastewater	L	External	Hollow fiber, MF, Pore size: 0.2 μm	CSTR	180	HRT = 6 h MLSS = 14–80 g/L OLR = 2.16 kg COD/m^3/d Temp = 25 °C Flux = 7.5 LMH	COD_t = 540	COD_s = 65 (88%)	[88]
Municipal wastewater	L	Submerged	Non-woven fabric, PET, pore size: 0.64 μm	UASB	12.9	HRT = 2.6 h OLR = 2.36 kg COD/m^3/d Temp = 15–20 °C Flux = 5 LMH	COD = 259.5 ± 343.8 NH_4^+–N = 27.5 ± 13.6 TP = 4.2 ± 1.4	COD = 77.5 ± 29.5 NH_4^+–N = 27.6 ± 12.5 TP = 3.2 ± 1.3	[48]
Domestic wastewater	L	External	UF membrane 100 kDa MWCO	CSTR	50	HRT = 15 h SRT > 140 d MLVSS = 0.5–10 g/L OLR = 2.0 kg COD/m^3/d Temp = 37 °C Flux = 3.5–13 LMH	COD = 685 ± 46.4 TOC = 157 ± 8.6 BOD_5 = 356 ± 18.5 K_j–N = 156 ± 7.8 TP = 11.5 ± 0.6 SS = 380 ± 9.3 pH = 7.2 ± 0.2	COD = 87.8 ± 6.2 (88%) TOC = 19 ± 1 BOD_5 = 31.2 ± 2.2 (90%) K_j–N = 38.8 ± 2 TP = 11 ± 0.55 SS = 0 pH = 7.7 ± 0.2	[90]
Municipal wastewater	L	External	Flat-sheet, CA, Pore size: 0.2 μm	CSTR	15	HRT = 16.67 h MLSS = 1.05–2.41 g/L Temp = 35 ± 2 °C Flux = 80–450 LMH	COD = 350–500 NH_4^+–N = 15–21 PO_4^{3-}–P = 3–4.5 SS = 1–1.5 pH = 6.5–7.8	COD<30 (>95%) TN = 9–9.85 (15–20%) PO_4^{3-}–P<1 (81%) pH = 6.9–7.3	[29]
Domestic wastewater	L	External	PTFE Teflon membrane pore size: 0.45 μm	CSTR	850	HRT = 14.4 h OLR = 0.8 kg COD/m^3/d Temp = 22 °C	COD_t = 620–650 (637) TOC = 180–230 (207) NH_4^+–N = 56–61 (58) K_j–N = 70–78 (74) TP = 10–12 (11)	TOC = 17 (>90%) K_j–N = 67 TP = 10	[92]
Municipal wastewater	L	External	PVDF; pore size: 0.1 μm, 200 kDa MWCO	CSTR	10	HRT = 48 h MLSS = 1.01 ± 0.29 OLR = 0.03 ± 0.01 kg COD/m^3/d SRT = 19 d Temp = 32 °C	COD_s = 84 ± 21 NH_4^+–N = 27.3 ± 13.5 PO_4^{3-}–P = 6 ± 2.3 NO_3^-–N = 0.3 ± 0.2 TSS = 120 ± 60 pH = 7.5 ± 0.1	COD_s = 25 ± 12 (58 ± 14%) NH_4^+–N = 8.9–51.8 NO_3^-–N<0.4 (0%) NO_2^-–N<0.4 (0%) pH = 6.6 ± 0.1	[41]

[a] L = laboratory/bench scale.
[b] PVDF = polyvinylidine fluoride, PET = polythylene terephthalate, CA = cellulose acetate, and PTFE = polytetrafluoroethylene.
[c] CSTR = completely stirred tank reactor and UASB = upflow anaerobic sludge blanket.
[d] The concentration unit is mg/L if not specified; COD_s = soluble COD, and COD_t = total COD.
[e] The concentration unit is mg/L; and removal efficiency is presented in parentheses.

hydrolysis of solid matters. The applied HRT, however, was significantly lower than the applied SRT with a range of 20–70.5 d (Table 6). These studies confirmed the proposed advantage of AnMBR, which decouples SRT from HRT, over conventional anaerobic digestion process. The applied HRT appears to be efficient for hydrolysis and methanogenesis processes. It is evident from the study of Trzcinski and Stuckey [101] who reported that no SS, soluble COD and VFA accumulation occurred inside AnMBR during treatment of municipal solid waste. The applied OLR was usually higher than 1 kg COD/m^3/d, and some cases higher than 10 kg COD/m^3/d, demonstrating the capacity of AnMBR to handle certain variation of OLR. The COD removal was generally higher than 90%.

AnMBRs have been used to treat landfill leachate and municipal solid waste leachate. Landfill leachate is a high organic matter and ammonium nitrogen strength wastewater formed as a result of percolation of rain-water and moisture through waste in landfills. The chemical composition of landfill leachate is dependent upon the age and maturity of the landfill site. The typical recent applications regarding AnMBR treatment are summarized in Table 7. It can be seen from Table 7, under selected operational conditions, high COD removal of about 90% could be achieved. The applied OLR was generally higher than 2.5 kg $COD/m^3/d$. Marisa and Beal [103] reported that COD removal was 90.4% for an AnMBR and 21.5% for an anaerobic filter (AF) when treated the same landfill leachate, indicating membrane separation significantly improved COD removal. During these treatments, the inhibition of microbiological activity by landfill leachate was observed, and thus resulted in a reduced COD removal [104]. This effect, however, can be mitigated by using diluted landfill leachate as feed [104]. Also, the treatment efficiency can be improved by prolonged SRT and PAC addition [105].

3.2. Applications in biogas production and energy recovery

One notable advantage of the anaerobic process is the biogas recovery. Continuous biogas production could be observed in AnMBR systems for various wastewaters treatment. The observed methane yield ranged 0.23–0.33 $LCH_4/g\ COD_{removal}$ has been reported [30,78,107–110], which is generally lower than the theoretical yield (0.382 $LCH_4/g\ COD_{removal}$ at 25 °C). The lower observed methane yield would be attributed to high methane solubility [111] and some inhibitors associated with anaerobic process [112]. Lettinga et al. [113] observed more than 50% methane escape with treated effluent by UASB and attributed it to dilute nature of the sewage. Meanwhile, methane solubility is significantly affected by operational temperature. Methane is approximately 1.5 times more soluble at 15 °C compared to 35 °C, for a typical biogas methane content of 70%. Capturing dissolved methane is of great interest (particularly for dilute wastewater), since the loss of dissolved methane with the effluent would offer significant challenge for energy recovery, and also cause greenhouse gas emission. Several processes have been recently proposed for this purpose, including stripping of AnMBR effluent through post-treatment aeration [114], methane recovery using a degassing membrane [115], and the use of a down-flow hanging sponge (DHS) reactor [116].

The composition of the biogas produced from AnMBR appears to be: 70–90% methane, 3–15% carbon dioxide and 0–15% nitrogen [30,39,71,107–109]. The methane rich biogas can be used for digester heating, electricity generation or even recycled for fuel production. It was reported that 2.02 $kWh/kg\ COD_{Removed}$ can be produced from an AnMBR treating synthetic wastewater, which is approximately 7 times more than is required to operate the system [75]. In general, methane fermentation is a complex process divided up into four phases: hydrolysis, acidogenesis, acetogenesis/dehydrogenation, and methanation. Each phase is carried out by different consortia of microorganisms which place different requirements on the environment [117]. Several factors significantly affect methane production. High temperature is known to benefit the maximum specific growth and substrate utilization rates, and thus increase methane production. However, temperature changes or fluctuations were found to affect the biogas production negatively in SAnMBR [118]. Furthermore, thermophilic processes are more sensitive to temperature fluctuations and require longer time to adapt to a new temperature. Meanwhile, the thermophilic process temperature results in a larger degree of imbalance and a higher risk for ammonia inhibition [119]. It's well-known that methane formation takes place within a relatively narrow pH interval, from about 6.5 to 8.5 with an optimum interval between 7.0 and 8.0. The process is severely inhibited if the pH decreases below 6.0 or rises above 8.5 [119]. Characteristics of the organic compounds also exert significant influences on methane production. Organic wastes rich in carbohydrates, such as biowaste and corn silage, can improve the biogas production and the proportion of CH_4 [120].

3.3. Operational conditions

The main operational conditions related to AnMBR applications include hydrodynamic conditions, HRT, SRT, pH and temperature. For AnMBRs with external configuration, employing high liquid cross-flow velocity (CFV) along the membrane surface is a common operation to reduce the particle deposition over the membrane surface. However, high shear conditions have also been reported as detrimental for anaerobic biomass activity and/or responsible for the physical interruption of syntrophic associations—a key factor in the anaerobic degradation of organic matter [13]. Typically, CFV values of 2–3 m/s are sufficient to prevent the formation of reversible fouling while have no obvious effect on microbial activity in external configuration [69]. For submerged configuration, biogas sparging is the most common way to provide shear conditions [9,67,109,121,122]. However, to the best of our knowledge, no studies have assessed the effects of biogas sparging rate on microbial activity or organic removal performance in AnMBRs.

It can be seen from Tables 3 to 7, the applied HRT in AnMBR varied from 2.6 h to 14 d, while the typical HRT for high strength wastewater treatment and dilute wastewater treatment was 1–10 d and 0.25–2 d, respectively. Elongating HRT could generally improve pollutants removal, but only to a limited extent. For example, Hu and Stuckey [109] observed a marginal decrease in COD removal (approximately 5% overall) when they lowered the HRT from 48 h to 24, 12, 6, and 3 h during treatment of simulated dilute wastewater. SRT remains one of the main operational parameters determining both treatment performance and membrane fouling. In contrast to UASB reactor, AnMBR enables completed retention of biomass, and thus provides easier control of SRT. Trzcinski and Stuckey [105] investigated the performance of two SAnMBRs treating municipal solid waste leachate at psychrophilic temperature with SRT of 300 and 30 d, respectively. It was found that longer SRT was associated with higher soluble COD removal [105]. In contrast, Baek et al. [89] found that the decrease in SRT from 213 to 40 d didn't affect treatment performance or membrane fouling. This suggests that the relationship between SRT and treatment performance or membrane fouling is complex, and highly depends on the applied HRT and the feed characteristics. In general, AnMBR operation with relatively long HRTs and SRTs was favorable, to enhance methane recovery, treatment performance and reduce sludge production [108].

Most AnMBR systems operate at near neutral pH since anaerobic digestion takes place within pH 6.5–8.5 with an optimum interval between 7.0 and 8.0 [119]. Such a pH range was usually achieved through neutralization, which could require the excessive use of chemicals because some streams have extreme pH values and hydrolysis, acidogenesis phases would decrease pH values. In this respect, equalization at a desired pH appears to be a prospective solution although related research was very limited in AnMBR systems. Anaerobic digestion is strongly influenced by temperature and can be grouped under one of the following categories: psychrophilic (0–20 °C), mesophilic (20–42 °C) and thermophilic (42–75 °C) [86]. Higher temperatures are known to improve methanogenesis, moreover, for several industries, including the pulp and paper and textile industries, generate high temperature wastewaters. Therefore, operation at thermophilic temperatures is of great interest because pre-cooling and post-heating used in the mesophilic treatment for subsequent reuse of treated effluent could be avoided. Several applications operated at thermophilic temperatures were available in the literature [9,68,123,124]. However, a deterioration of membrane flux always occurred due to sludge deflocculation and EPS released caused by high temperature [9]. Therefore, justification and selection of the operational temperature is important to achieve optimal performance. However, for most streams including municipal wastewater, operation at ambient temperature or low temperature is essential for economical implementation of AnMBRs treating them. Psychrophilic AnMBR treatment has

Table 6
Summary of AnMBR performance for high-solids-content waste streams treatment.

Type of wastewater	Scale[a]	Configuration	Characteristics of membrane[b]	Type of reactor[c]	Reactor volume (L)	Operating condition	Feed[d]	Efficiency[e]	Reference
Municipal sewage sludge	L	Submerged	Tubular stainless steel metal membrane pore size: 1.0 µm	CSTR + M	100	(Run2) HRT = 2 d SRT = 20 d SS = 18–55 g/L Temp = 35 °C pH = 6 Flux = 0.25 LMH	SS = 5–30 COD_s = 7	Most favorable fermentation efficiency was attained	[21]
Waste activated sludge	L	External	Hollow fiber PE membrane pore size: 0.4 µm	UASB + M	8	HRT = 6 d SRT = 80 d Temp = 37 °C	–	VS destruction > 52.1%	[44]
Municipal (solid) waste	L	Submerged	Cylindrical woven nylon mesh; pore size: 30, 40, and 140 µm	CSTR + M		HRT = 1.5 d; SRT = 20 d OLR = 3.75 kg VS/m^3/d	–		[47]
Municipal solid waste	L	Submerged	Kubota PE flat sheet membrane; Pore size: 0.4 µm	CSTR/ CSTR + M	10/3	HRT = –/1.6–2.3 d Temp = 35 ± 1 °C Flux = 0.5–0.8 LMH	COD = –/4–26 Initial TSS = 40/3.31 g/L	COD = 4000–26,000/400–600 (>90%)	[43]
Municipal sewage sludge	P	External	Vibrating unit; Teflon, UF; Pore size: 0.05 µm	CSTR + M	550	HRT = 1.7–11.8 d SRT = 70.5 d Temp = 35 °C TSS = 1.8% Flux = 60,7–83.3 LMH	TSS = 0.6%	TSS = 0; Average TS and VS reductions were 51% and 59%, respectively, by the digester	[36]
Wastewaters containing suspended solids	L	Submerged	Tubular PSF MF membrane	CSTR + M	3.8	TSS = 40 g/L OLR = 10 kg COD/m^3/d Temp = 30 °C Flux < 4 LMH	COD = 10	COD = 150–200	[26]
Slaughterhouse wastewater	L	External	MF 100 kDa MWCO	CSTR + M	50	HRT = 1.66 d MLVSS < 10 g/L OLR = 8.23 ± 2.5 kg COD/m^3/d Temp = 37 °C Flux < 3 LMH	COD_t = 10.174 ± 3.31 pH = 7.53–7.7	COD_s = 338 ± 60 (94 ± 2.12%)	[102]
Slaughterhouse wastewater	L	External	MF 100 kDa MWCO	FBR/ CSTR + M	25/50	HRT = 1.25 d MLVSS = 8.257–10.1 g/L OLR = 12.7 ± 1.71 kg COD/m^3/d Temp = 37 °C Flux < 3 LMH	COD_t = 10.58 ± 0.99 pH = 7.53–7.7	COD_s = 196 ± 4 (98.75 ± 0.44%)	[102]
Swine manure	L	External	Tubular PES UF membrane; 20 kDa MWCO	CSTR + M	6	(Run 1) HRT = 6 d; pH = 7.5 OLR = 1 kg VS/m^3/d Temp = 37 ± 1 °C Flux < 0.3 LMH	Biomass concentration = 6 g VS/L	COD_s = 200–250 (86%), COD_t = – (96%)	[42]
Sand-separated dairy manure	L	External	Tubular PVDF membrane; Pore size: 0.03 µm	CSTR/ CSTR + M	100/100	HRT = 9/9 d; SRT = 28 d Mesophilic condition OLR = 3.3/2.4 kg VS/m^3/d	COD = 44.9 ± 12.1/ 31.8 ± 6.56 TS = 4.54 ± 0.69%/ 3.36 ± 0.64% NH_4^+–N = 1.24 ± 0.47/1.3 ± 0.09 TP = 0.34 ± 0.03/ 0.32 ± 0.08	COD = 3440 ± 700 (92 ± 1.8%) TS = 800 ± 150% (81.7 ± 5.3%) NH_4^+–N = 1330 ± 90 (−16.6 ± 31.9%) TP = 14 ± 5 (95.8 ± 1.5%)	[40]

[a] L = laboratory/bench scale and P = pilot scale.
[b] PVDF = polyvinylidine fluoride, PE = polythylene, PSF = polysulfone, and PES = polyethersulfone.
[c] CSTR = completely stirred tank reactor, UASB = upflow anaerobic sludge blanket and FBR = fixed bed reactor.
[d] The concentration unit is g/L if not specified; – indicates value not reported; COD_s = soluble COD, and COD_t = total COD.
[e] The concentration unit is mg/L and removal efficiency is presented in parentheses.

recently drawn significant attentions [105,110]. It was found that both psychrophilic and mesophilic treatment achieved comparable COD removal efficiency close to 90%, although the former corresponded to a little higher membrane fouling rate due to volatile fatty acid (VFAs) accumulation [110]. This result highlights the possible role of membrane filtration in performance stability across temperature fluctuations. In order to widely apply the AnMBR technology, one key challenge is to overcome the problems caused by the local climate change conditions within approximately 0 to 25 °C. However, no studies have assessed AnMBR treatment performance of psychrophiles at elevated temperatures.

3.4. Applicability of AnMBRs

In current review, it is proposed that wastewater can be conceptualized as having three axes, including an x-axis (concentration of the constituents), a y-axis (particulate nature of the constituents), and a z-axis (extreme conditions, e.g. extreme pH, temperature, salinity) which represent the principle characteristics of wastewater (Fig.1). Fig.1 classifies wastewaters into 8 zones. For example, municipal wastewater characterized by low organic strength, low particulate content and less extreme properties, will fall in Zone VII. Pulp and

Table 7
Summary of AnMBR performance for leachate treatment.

Type of wastewater	Scale[a]	Configuration	Characteristics of membrane[b]	Type of reactor[c]	Reactor volume (L)	Operating condition	Feed[d]	Efficiency[e]	Reference
Landfill leachate	L	External	UF 100 kDa MWCO	CSTR + M	50	(Run 3) HRT = 7 d pH = 7.5 OLR = 6.27 ± 0.78 kg COD/m^3/d MLVSS < 3 g/L Temp = 37 °C	COD = 41 ± 3.14	COD = 3.77 ± 0.34 (90.7 ± 1.1%)	[106]
Sanitary landfill leachate	L	External	Ceramic tubular membrane pore size: 0.2 μm	CSTR + M	44	HRT = 2.04 d Temp = 34–36 °C OLR = 5.07 ± 2.90 kg COD/m^3/d	–	COD = – (90.4%) turbidity = 2.0 ± 2.0NTU (90.3%)	[103]
Diluted landfill leachate	L	Submerged	Capillary UF membrane Pore size: 0.1 μm	CSTR + M	29	HRT = 2 d pH = 8.18 OLR = 2.5 kg COD/m^3/d MLSS = 10 g/L Temp = 35 °C	COD = 5 NH_4^+–N = 0.382 pH = 8.03	COD = 0.417 (90%), NH_4^+–N = 0.206	[104]
Municipal solid waste leachate	L	Submerged	PE flat sheet membrane Pore size: 0.4 μm	CSTR + M	3	HRT = 1.5 d; SRT = 30 d OLR = 8 kg COD/m^3/d pH = 7.3 Temp = 35 °C	–	COD = 1.0 (79–95%),	[105]
Municipal solid waste leachate	L	Submerged	PE flat sheet membrane Pore size: 0.4 μm	CSTR + M	3	HRT = 1.1 d; SRT = 300 d OLR = 11.7 kg COD/m^3/d pH = 7.3 Temp = 35 °C	–	COD = – (90%)	[105]

[a] L = laboratory/bench scale and P = pilot scale.
[b] PE = polythylene.
[c] CSTR = completely stirred tank reactor.
[d] The concentration unit is g/L if not specified and – indicates value not reported.
[e] The concentration unit is g/L and removal efficiency is presented in parentheses.

paper industrial wastewater will fall in Zone IV due to that it is characterized by high organic strength, low particulate content and extreme properties (e.g. high temperature).

AnMBRs appeared to be suitable to treat all types of wastewaters except for wastewaters falling in Zone VIII which are characterized by high organic strength, low particulate content and favorable properties. Wastewaters falling in Zone VIII are currently treated effectively with various HRARs, such as UASB and EGSB. Biofilm and granule formation in HRARs would enable to decouple SRT from HRT, allowing for high OLR and organic removal with a minimum of SS in the effluent. Compared to AnMBRs, HRARs could achieve comparable or a little worse effluent quality, while their capital and operational costs remain relative low due to no membrane used. The opportunity for AnMBRs to be applied to these wastewaters appears to only exist when very low SS concentration in effluent and/or short start up period are required.

Low organic strength will favor AnMBRs treatment as compared to HRARs. Biomass loss and bad effluent quality are two major problems associated with HRARs treating low organic strength wastewaters. AnMBRs can solve these problems because membrane totally retains biomass and enhanced effluent quality achieved. AnMBRs treating low organic strength wastewaters have recently drawn considerable attention [8,30,109,125]. Increase in particulate content in wastewater will increase the applicability of AnMBRs. Retention of particulates in HRARs would be very problematic, and long SRT and HRT were usually required for sufficient hydrolysis. This will significantly increase the capital costs. Complete retention of particulates can be achieved in AnMBRs, which may allow greater treatment efficiency by allowing more complete hydrolysis of slowly degraded compounds. Application of AnMBRs is more likely restricted to conditions or applications where granular sludge technology may or will encounter problems. This likely is the case when extreme conditions prevail, such as high temperatures and high salinity, or wastewaters with refractory and/or toxic compounds, since biofilm and granule formation can be severely affected. Following the current trend of increasing water use efficiency and closing industrial process water cycles, these extreme conditions are likely to become more common in the future [126]. It is expected that AnMBRs will get more opportunities in these wastewaters treatment.

4. Membrane fouling issues

Membrane fouling remains the critical obstacle limiting the more widespread application of AnMBR in wastewater treatment. Membrane fouling could decrease system productivity, cause frequent cleaning which might reduce the membrane lifespan and result in higher replacement costs, and increase the energy requirement for sludge recirculation or gas scouring. Membrane fouling results from interaction between the membrane material and the components of sludge suspension. Though the membrane used in aerobic MBR can be generally used in AnMBR system, the sludge suspension in AnMBR system is significantly different from that in aerobic compartment, presenting certain unique impacts on membrane fouling characteristics. A set of techniques or approaches are now available to characterize membrane fouling [127], which allows for better understanding of membrane fouling in AnMBR system. To date, there have been a considerable number of published papers on AnMBR system, perusal of the literature shows that there is a lack of a comprehensive review regarding membrane fouling specific for AnMBR system.

4.1. Membrane fouling classification

Membrane fouling can be traditionally classified into reversible and irreversible fouling based on the cleaning practice, although their definitions were not consistent in the literatures. Here, we adopt the classification proposed by Meng et al. who further defined reversible fouling into removable fouling and irremovable fouling. Accordingly, removable fouling refers to fouling that can be removed by physical means such as backflushing or relaxation under cross flow conditions, while irremovable fouling refers to fouling needed to be removed by chemical

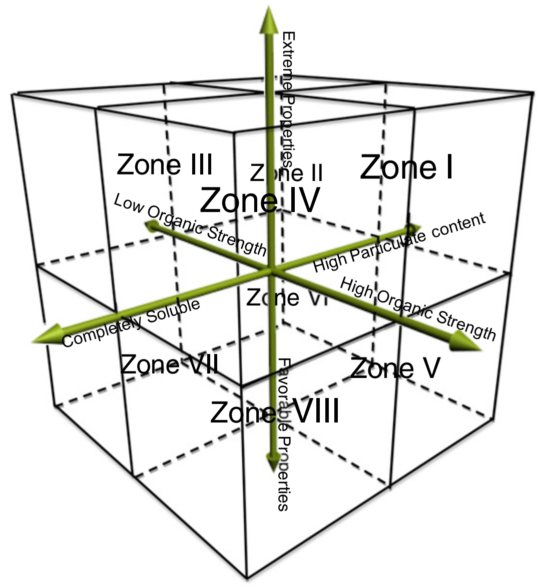

Fig. 1. Conceptualization of wastewaters according to their principle characteristics.

cleaning. The irreversible fouling is a permanent fouling which cannot be eliminated by any cleaning approaches. In general, removable fouling occurs due to loose external deposition of material. In contract, irremovable fouling is caused by the pore blocking and strongly attached foulants during membrane filtration. Formation of a strong matrix of fouling layer with the solute during a continuous filtration process will result in removable fouling being transformed into an irremovable fouling layer. Considering the nature and the causes of irremovable fouling, many efforts have been performed to investigate cake layer. During the long-term operation of a SAnMBR, Jeison and van Lier [122], and Gao et al. [128] observed that cake formation and consolidation, which could not be removed by the back-flush cycles or relaxation, provide the dominant mechanism of membrane fouling as compared to internal pore fouling. Meanwhile, Di Bella et al. [129] reported that the cake layer formed in an aerobic MBR had a mainly removable nature. Although cake formation is a complex process and involved a lot of influencing factors, it might be concluded that, on average, cake layer formed in AnMBR has relatively lower removability than that in aerobic compartment due to the different sludge properties.

Membrane fouling can also be classified into biological, organic and inorganic fouling in viewpoint of the foulant components [3,46]. Biological fouling is specifically related to the interaction of biomass with the membrane. Membrane fouling appears to start from pore clogging caused by cell debris and colloidal particles. Passive adsorption of colloids and organics has been observed even for zero-flux operation, before the biomass deposition initiates [129]. Gao et al. [128] reported that about 65% of the particles based on number in the top cake layer in a SAnMBR had a size smaller than 0.3 μm that was the same to the pore size of the used membrane. These particles/flocs would penetrate into and block the membrane pores easily. Biofouling also includes the accumulation and adsorption of extracellular polymeric substances (EPS) and soluble microbial products (SMP) on membrane and pore surfaces as these substances were biologically secreted. Meanwhile, some studies have been performed to investigate the microbial community and its role in membrane fouling in AnMBRs. Gao et al. [130] and Lin et al. [131] found that there were significant differences in microbial communities between sludge on membrane surfaces and in bulk sludge in an external and submerged AnMBR, respectively, suggesting that some bacteria selectively adhered and grew on the membrane surface.

Organic and inorganic fouling usually respectively refers to macromolecular species (biopolymers) and scalants. Lin et al. [9] also observed the supernatant COD was consistently higher than the effluent COD for a SAnMBR. The significantly higher content of organics in the supernatant was believed to be biopolymer matters, which may act as a "glue", facilitating a cake layer formation. Analysis through Fourier transform infrared (FTIR) spectroscopy and confocal laser scanning microscopy (CLSM) demonstrated that the foulants on the membrane surface in SAnMBR were rich in proteins and polysaccharides [9,128,132], indicating that organic fouling was originally caused by SMP or EPS. As for inorganic fouling, struvite ($MgNH_4PO_4 \cdot 6H_2O$) appeared to one of the main inorganic foulants identified earliest in AnMBR systems [13,28,133,134]. Other inorganic foulants can include $K_2NH_4PO_4$ and $CaCO_3$ [90]. Precipitation of inorganic foulants much depends on the presence of cations in the influent and sludge suspension, which is the origin of inorganic elements in cake layer. Through charge neutralization and bridging effect, metal clusters and metal ions in the influent could be caught by the flocs or biopolymers and then enhanced filtration resistance [135]. Lin et al. [70] reported that the cake layer in a SAnMBR was formed by organic substances and inorganic elements such as Ca (4.45% dry weight content), Mg (1.94%), Al (1.72%), Si (1.46%), K (0.15%), etc. Herrera-Robledo et al. [32] found that cake sludge in AnMBR was mainly composed of volatile solids (85%) and the rest was related to mineral matter. Similar results have also been reported by other researchers [33,48].

It should be borne in mind that biological, organic and inorganic fouling take place simultaneously, and the interaction of them usually increases filtration resistance. For example, Choo and Lee [13] reported that deposition of the microbial cells together with struvite played a significant role in the formation of the strongly attached cake layer limiting membrane permeability. From one point of view, membrane fouling is generally characterized by initial pore clogging followed by biocake formation and consolidation regardless of aerobic and anaerobic MBRs. However, the forms and significance of membrane fouling in AnMBR would be of some differences. For instances, Gao et al. [128] found that cake thickness in a SAnMBR could be 1900–2100 μm, which was much higher than 20–200 μm reported on aerobic MBR systems [136,137]. For the same membrane, the maximum sustainable membrane flux was found to be 11 and 25–30 LMH for SAnMBR [30] and aerobic SMBR [138] for municipal wastewater treatment, respectively. Also, considering the relatively high concentration of carbonate and biocarbonate [139], and the production of high ammonia and phosphate concentrations in anaerobic digestion, AnMBRs may be more susceptible to inorganic fouling than aerobic MBRs. The unique characteristics of membrane fouling suggest that more attention should be paid on its control in AnMBR.

4.2. Membrane fouling mechanisms

Based on their relative contributions of foulant components to the total membrane fouling, several membrane fouling mechanisms, including pore plugging/clogging by colloidal particles, adsorption of soluble compounds and biofouling, deposition of solids as a cake layer, cake layer consolidation [122] and the spatial and temporal changes of the foulant composition [140] during the long-term operation, have been proposed.

The current trend in AnMBR design is to operate at constant flux. When operated in this mode, a three-stage trans-membrane pressure (TMP) profile characterized as an initially short term rapid TMP rise (stage 1) followed by extended slow TMP rise period (stage 2) and a transition to a rapid TMP rise (stage 3), which typically occurred in aerobic MBR operation [141], can also be observed (Fig. 2a) in AnMBRs [9,27,70,142,143]. The possible mechanisms for each stage are illustrated in Fig. 2b, c, and d according to previous studies [133,142]. Under suction drag and gas scouring in SMBR, there are two opposite forces that control the deposition of sludge components on membrane surface: permeation drag, which is generated by permeate flux, increased with operation TMP, and back transport, consisted of Brownian diffusion, inertial lift and shear induced diffusion [144]. Initially, the colloids and soluble products can be readily deposited onto the membrane surfaces by permeation drag, and not readily detached by shear force due to its low back transport velocity [145]. Their higher deposition tendency over large flocs has been verified in SAnMBR systems [9]. These colloids and soluble products usually have size lower than the pore size of the used membrane, and would readily penetrate into and block the membrane pores, and therefore caused significant membrane fouling, which would be responsible for the first TMP jump in Fig. 2a (Fig. 2b). The deposited colloids and soluble products were also considered to play the role of conditioning the membrane surface, facilitating followed cake formation [11]. The gradient developing sludge cake would prevent the further penetration and blocking of membrane pores by the colloids and soluble products, and itself corresponds to the slow TMP increase (stage 2, Fig. 2c). To date, the interpretation to the second TMP jump is still debated. The most popular interpretation is local flux theory which was firstly introduced in AnMBR system by Cho and Fane [142]. They attributed the second TMP jump to the changes in the local flux due to uneven distribution of foulants and EPS causing local flux to be higher than the critical flux [142]. However, even for membrane with relatively uniform distribution of foulants, the second TMP jump was also observed in a SAnMBR [9,146]. Hwang et al. [140] recently reported that the sudden jump of TMP was closely related to the sudden increase in the concentration of EPS at the bottom of cake layer, which might be attributed to the death of bacteria in the inner of cake layer. As membrane fouling is a really complex process, combination of the two explanations appears to be more extensive and reasonable than single interpretation alone.

Sludge cake consolidation (compression) is inevitable as TMP increases. After cake was formed on membrane surface, cake consolidation is a kind of sludge dewatering process. Activated sludge is made up of microbial organisms and colonies, embedded in a matrix of EPS [147] which carry charged functional groups, including carboxyl, hydroxyl and phosphoric groups, leading to the presence of large concentrations of counter-ions within the matrix of EPS for reasons of electro-neutrality [148]. These counter-ions closely associated with the matrix of EPS in cake layer will not readily go through the membrane, thus, the difference in salt concentration between two sides of the membrane will result in an osmotic gradient. Chen et al. [149] recently reported that osmotic pressure accounted for the largest fraction of total operation pressure during cake layer filtration, indicating that osmotic pressure generated by the retained ions was one of the major mechanisms responsible for membrane fouling problem in SAnMBR once a cake layer was formed.

4.3. Parameters affecting membrane fouling

Membrane fouling results from the interaction between membrane and sludge suspension. In this regard, all the parameters related to membrane and sludge suspension would have effects on membrane fouling. These parameters can be generally classified into four categories: feed characteristics, broth characteristics, membrane characteristics and operational conditions. The effects of these parameters on membrane fouling are summarized in Table 8 mostly based on the recent literature related to AnMBR. Among them, some parameters, such as SMP, EPS, particle size distribution (PSD) and hydrodynamic conditions, have direct effects on membrane fouling, and therefore were considered as the major parameters affecting membrane fouling. In contrast, some others, such as HRT, OLR, SRT and pH indirectly affect membrane fouling through the change in the broth characteristics. A comprehensive assessment of the major parameters and indirect affecting parameters in AnMBR appears to be not necessary, since membrane fouling mechanisms are generally similar in MBR systems, and previous reviews of MBR fouling have warranted separate presentations [11,46]. However, it should be noted that, for AnMBR treatment, the change will be the relative importance of these parameters under specific conditions. Table 9 compares some facets of these major parameters in aerobic MBR and AnMBR. It is expected that the higher MLSS, OLR, residual COD and SMP production in AnMBR will cause more serious membrane fouling. Moreover, the extreme conditions (pH and temperature) related to AnMBR treatment will induce decreased PSD of sludge liquor, which in turn negatively affect membrane fouling. For instance, under similar operational conditions, Martin-Garcia et al. [150] found that SMP in AnMBR supernatant was 500% higher than that in aerobic MBR supernatant. Considering different broth characteristics and operational conditions, it may be not surprising to conclude that, on average, AnMBR treatment would result in more serious membrane fouling problems. This comparison suggests that more attention should be paid on membrane fouling control in AnMBRs.

4.4. Membrane fouling control

The purpose of membrane fouling study is to develop strategies for membrane fouling control and membrane cleaning. Based on the parameters affecting membrane fouling, these strategies in AnMBR systems can be classified into five groups: (1) pretreatment of feed, (2) optimization of operational conditions, (3) modifying activated sludge, (4) modification of membrane and optimal design of membrane module, and (5) membrane cleaning.

4.4.1. Pretreatment of feed

Feed characteristics may exert significant impacts on membrane fouling. Some industrial streams contain trash which can plug the coarse bubble diffusers used to scour the membranes. The extreme pH conditions in some industrial wastewaters not only damage biologic performance, but also affect membrane permeability and lifespan. It has been reported that cake layer on membrane surface was rich of elements Mg, Al, Ca, Si, and Fe [70]. These components apparently originated from the inorganic matters in the feed. Interaction of biopolymer matters and these elements were reported to have significant impacts on the formation and compactness of the cake layer [13,70]. Excess quantities of these materials should be removed through wastewater pretreatment programs (i.e., filtration [92], pH adjustment [78], establishment of local wastewater limits). Kim et al. [28] used a dialyzer/zeolite (D/Z) unit to selectively remove NH_4^+ in the influent, in which substantial NH_4^+ removal (in excess of 90%) was achieved, leading to the significant reduction in struvite precipitation on the ceramic membrane in the AnMBR.

4.4.2. Optimization of operational conditions

The main operational parameters include hydrodynamic conditions, flux, HRT, SRT, biomass concentration, pH and temperature. Increasing the gas scouring intensity and time in SAnMBRs and the flow velocity of mixed liquor in sidestream AnMBRs could certainly achieve better hydrodynamic conditions for membrane fouling control. However, it could also disrupt sludge flocs, producing small size particles and releasing more EPS which negatively impact membrane fouling [9,45]. Jeison et al. [45] introduced the concept of “shear rate dilemma” to describe the dual effects of shear during AnMBR operation. There exists a practical limit above which only a minor benefit is provided. Pilot testing is required to find optimal hydraulic conditions. A well known strategy for membrane fouling control is to operate membrane at sustainable flux. Detailed discussion on critical flux and sustainable flux can be found elsewhere [163]. Other above mentioned parameters will directly affect broth properties. In this regard, control strategies should focus on modifying broth properties by adjusting these parameters. The relationship between these parameters and broth properties can refer to Table 8.

4.4.3. Modifying broth properties

Addition of the additives, such as adsorbent agents, coagulants, carriers, suspensible particles and other chemical agents, can modify the properties of the broth in AnMBRs. Suited additives for fouling mitigation can act through a number of different phenomena such as adsorption of SMP, coagulation, cross-linking between flocs, and a combination of

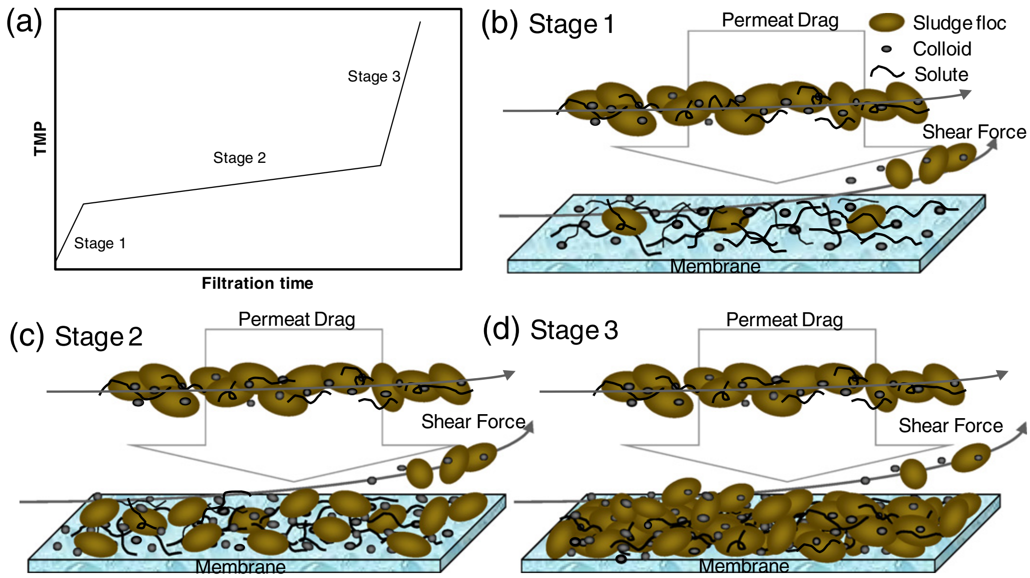

Fig. 2. Schematic illustration of the three-stage TMP profile and its fouling mechanisms.

these [164]. Meanwhile, some novel measures have been developed for the optimization of mixed liquor in recent years, providing various options for membrane fouling control in AnMBRs.

Powdered activated carbon (PAC) is the most widely used "flux enhancers" in MBRs. The first study testing the effect of PAC on fouling mitigation in AnMBR appeared to be reported on 1999 [165], and it was found that both the fouling and cake layer resistances decreased continuously with increasing the PAC dose up to 5 g/L. The enhanced membrane performance in AnMBR due to PAC addition has been confirmed by many studies [23,121,134,165]. The mechanism of PAC for fouling mitigation was supposed to be adsorption of the solutes and colloids in the supernatant [121], and enlarged floc size due to incorporation of PAC to the bioflocs [166]. However, an overdose of PAC could increase membrane fouling because excess PAC itself could be a foulant [121,167]. Other adsorbent agents like zeolite, bentonite, vermiculite and *Moringa oleifera* were also used to mitigate membrane fouling in AnMBRs [168,169]. These additives have high adsorption and ion-exchange capacity, and therefore are capable to reduce soluble organics and NH_4^+ in the supernatant. Improved effluent quality and membrane performance were usually observed in these studies. To date, coagulants including aluminum sulfate, ferric chloride, polyaluminum chloride (PACl), polyferric sulfate (PFS), polyacrylamide (PAM) and chitosan have been tested in aerobic MBRs [170–172], and alleviated membrane fouling due to increased floc size and decreased soluble organics in supernatant were generally observed. Addition of chemical coagulants to the wastewater may cause side effects by producing by-products and/or increasing the volume of sludge in the reactor [173]. An alternative technology for creating coagulation inside the system, suggested by the Bani-Melhem and Elektorowicz [174,175], is to introduce electrokinetic processes into the MBR. The design of electro-kinetic process was based on applying an intermittent direct current (DC) field between immersed circular perforated electrodes around an immersed membrane filtration module. Such a system not only significantly reduced the fouling rate, but also enhanced the removal of COD and PO_4^{3-}–P up to 96% and 98%, respectively [175]. These studies demonstrated the great potential of utilization of coagulants in AnMBRs.

Recently, Chae et al. [176] investigated potential use of fullerene C_{60} nanoparticle addition for membrane biofouling control. It was found that C_{60} significantly impeded bacterial surface attachment. Magnesium or titanium oxide and copper-based nanoparticles [176] could be other potential additives for fouling mitigation. This study provided a novel option for membrane fouling control in AnMBRs. Another notable novel measure is the use of ozone for sludge modification. It was found that ozonation enlarged suspended flocs by reducing zeta-potential and increasing hydrophobicity, thus enhancing flocculability of the particles in the mixed liquor, and mitigate membrane fouling [177,178]. For this measure, an optimal dosage is critical because overdosing would break the flocs and release colloidal and soluble organics, and therefore exacerbate fouling [177]. More recently, a granular AnMBR seeded with granular sludge from a UASB was developed by Martin-Garcia et al. [150]. As compared to parallel operated flocculated AnMBR, colloids and SMP in the supernatant were significantly reduced in the granular AnMBR. This study showed that development of granular sludge in AnMBR could increase filtration ability of broth supernatant, and present an effective strategy for membrane fouling control.

4.4.4. Membrane optimization

Surface modification for hydrophilic improvement of membrane is a common strategy for membrane fouling control since the most salient property of the membrane materials is their surface properties. Surface modification with aims to implant polar organic functional groups onto the membrane surface could be achieved by means of plasma treatment, surface grafting, surface coating and surface blending, etc. Plasma treatment appears to be an efficient technique to create hydrophilic functional groups on the membrane surface. It was found that membrane hydrophilicity significantly increased after NH_3 and CO_2 plasma treatments, and new membranes presented better filtration performances and flux recovery than those of unmodified membranes [179,180]. To date, plasmas including air, O_2, N_2, CO_2, H_2O, and NH_3 plasmas, have been explored [179–183]. The unique advantage of plasma treatment is that the surface properties and biocompatibility can be enhanced selectively while the bulk attributes of the materials remain unchanged. Whereas, the complex chemical reactions and the large number factors affecting treatment efficiency involved in plasma treatment make it difficult to extend such technology on large-scale. Surface graft polymerization is another attractive method to improve membrane hydrophilicity. By performing UV photo-induced graft polymerization of acrylic acid and acrylamide on a PP MF membrane surface, Yu et al. [184] observed the decreased water-contact angle and increased zeta potential (absolute value) with increased grafting degree. A PP membrane modified by ozone treatment followed by graft polymerization with 2-hydroxy-ethyl methacrylate (HEMA) has been applied in

Table 8
Description of the effects of fouling parameters on membrane fouling in AnMBRs.

Fouling parameters	Description of the effects on membrane fouling	Wastewater	Ref.
Operational conditions			
HRT	HRT↓→biomass concentration↑, PN/PS in SMP↑→dTMP/dt↑	Synthetic low-strength wastewater	[125]
	HRT↓→EPS↑, SMP↑→cake resistance↑	Acidified wastewater	[25]
	HRT↓→biopolymers↑, floc size↓→specific cake resistance↑	Synthetic municipal wastewater	[151]
OLR	OLR↑→VFA concentration↑, predominant VFA type changed	Synthetic coke wastewater	[68]
SRT	SRT↑→sludge activity↓, SMP↑→dTMP/dt↑	Synthetic low-strength wastewater	[125]
	SRT↑→MLVSS↑, floc size↓→irreversible fouling↑	Synthetic low-strength wastewater	[67]
Hydrodynamic conditions	Gas sparging rate↑→critical flux↑	Kraft evaporator condensate	[71]
	Gas sparging time↓→TMP↑	Saline sewage	[23]
	CFV↑→shear force↑, floc size↓→critical flux firstly↑ then↓	Acidified synthetic wastewater	[45]
	Gas sparging was ineffective in increasing the critical flux	Acidified wastewater	[152]
	CFV↑→SMP↑, floc size↓→flux↓	Diluted anaerobic sludge	[153]
Permeate flux	Permeate flux↑→long-term operation period↓	Swine wastewater	[154]
	Permeate flux↑→cake formation rate↑	Kraft evaporator condensate	[146]
	Permeate flux↑→fouling rate↑	Domestic wastewater	[155]
Temperature	Temperature↓→COD_{sup}↑→stable flux↓	Municipal solid waste leachate	[105]
	Temperature↑→COD_{sup}↑, floc size↓, PN/PS of EPS↑→filtration resistance↑	Kraft evaporator condensate	[9]
	Temperature↑→viscosity↓, COD removal↑→flux↑	Food wastewater	[10]
Biomass characteristics			
MLSS	MLSS↑→initial and stabilized flux↓, optimal MLSS = 15–18 g/L	Diluted anaerobic sludge	[153]
	MLSS↑→TMP↑	Food industry wastewater	[156]
	MLSS↓→solids deposition rate↓	Dilute municipal wastewater	[89]
PSD	Amount of small flocs↑→filtration resistance↑	Kraft evaporator condensate	[9]
	Floc size↓→specific cake resistance↑	Synthetic municipal wastewater	[151]
	$D_{0.1}$↑→cake formation rate↓	Kraft evaporator condensate	[146]
SMP	SMP↑→filtration resistance↑	Kraft evaporator condensate	[9]
	High-MW protein and carbohydrate material↑→internal fouling↑	Low-strength synthetic feed	[8]
	Low flux was attributed to high amounts of SMP	Medium strength wastewater	[157]
EPS	PN/PS ratio↑→fouling rate↓	TMP whitewater	[39]
	EPS↑→cake resistance↑	Acidified wastewater	[25]
	EPS the foulant layer contributed to membrane fouling	Particulate artificial sewage	[130]
Microbial community	Some bacteria play a pioneering role in cake formation	TMP whitewater	[131]
	Relative abundance of bacteria was different in cake layer and suspension	Artificial sewage	[130]
Membrane characteristics			
	MWCO↑, surface roughness↑→flux decline↑, recoverable flux rate↓	Food wastewater	[10]
	Pore size↑→attainable flux↓	Synthetic wastewater	[158]
	Fouling of PEI membrane was faster than PVDF membrane coated with PEBAX	Artificial sewage	[130]

an AnMBR, and the results showed that the membrane permeability was significantly enhanced [24]. However, there are the two major problems remained: the difficulty in obtaining optimal value of grafting chain length and grafting density for membrane permeability and membrane antifouling characteristics, and the high costs of employing high-energy induced methods, such as UV irradiation [184], gamma irradiation [185], and chemical reaction [24]. Surface coating via adsorption of surfactants was also explored. A significant example is the investigation of Kochan et al. [186], where different UF flat-sheet membranes, PVDF, PES, PSF and cellulose acetate (CA) were coated by branched poly(ethyleneimine) (PEI), poly(diallyldimethylammonium chloride) (PDADMAC) and poly(allylamine chloride) (PAH) and filtrated with sludge supernatant, and it is found that coating led to lower fouling rates during filtration. The disadvantages of this measure would be the low physical tolerance (to, e.g., desorption, cross-flow and gas scouring) and the chemical stability of the coating layer under MBR conditions. To overcome above disadvantages, a self-assembly technique was employed to create thin film composite nanofiltration membranes (TFC NF), which was achieved by coating commercial PVDF UF membrane with the amphiphilic graft copolymer PVDF-graft-polyoxyethylene methacrylated (PVDF-g-POEM) [187]. The new TFC NF membranes exhibited no irremovable fouling in 10 d dead-end filtration of model organic foulants (bovine serum albumin (BSA), sodium alginate and humic acid) at concentrations of 1000 mg/L and above. Meanwhile, TiO_2 embedded polymeric membranes prepared by a self-assembly process have recently drawn considerable attention. When applied for activated sludge filtration, it was found that adsorbed foulants on the TiO_2 embedded membrane surface were more readily dislodged by shear force than those on neat polymeric membranes due to the increased hydrophilicity of the membrane [188,189]. In general, the above modified membranes can be used in aerobic systems as well as anaerobic systems.

4.4.5. Membrane cleaning

Membrane fouling can never be completely avoided, while the fouled membranes can be regenerated by physical, chemical, and biological schemes. Physical cleaning techniques for MBRs include mainly membrane relaxation and membrane backflushing. Detailed discussion on these conventional physical cleaning measures can be found in the previous review paper [11]. In recent years, a novel on-line physical cleaning method, ultrasonication, has been developed and extensively investigated in MBRs, especially in AnMBRs [44,66,190,191]. Wen et al. [191] showed that ultrasound can effectively control cake formation on the membrane surface. The mechanism of ultrasonication for membrane fouling control was considered to be cavitation and acoustic streaming induced by ultrasonic waves preventing the cake formation and enhancing membrane filtration rates [192]. Meanwhile, it was found that ultrasonic irradiation could negatively affect anaerobic bacterial activity [190] and cause membrane damage [191]. These effects, however, can be significantly reduced by properly selecting ultrasonic intensity and working time and keeping a certain thickness of cake layer on the membrane surface [191].

When the above-mentioned cleaning methods are not effective enough to reduce the fouling to an acceptable level, it is necessary to clean the membranes chemically. Many chemical cleaning agents, such

Table 9
Some facets of membrane fouling propensity in aerobic MBRs and AnMBRs.

Parameters	Aerobic MBR	AnMBR	Potential effects on membrane fouling
MLSS	Lower MLSS maintained in bioreactor	Similar or higher MLSS maintained in bioreactor	MLSS is positively correlated to membrane fouling [156,159]
OLR	Low	High	Filtration resistance increases with organic loading [160]
SMP	Depends	Flocculated AnMBR supernatant was characterized by a SMP concentration ca. 500% higher than the aerobic MBR [150]	High SMP content results in serious membrane fouling [161] Soluble SMP amount in the mixed liquor was the most important property influencing the fouling propensity of sludge [162]
PSD	More often, aerobic MBRs were used for municipal wastewater treatment. Mild nature of municipal wastewater favored flocs growth.	More often, AnMBRs were applied for industrial wastewater treatment. Extreme conditions of industrial wastewater were frequently encountered, which would cause dispersed flocs	Flocs size significantly affected cake formation, filtration resistance [131,145]

as sodium hypochlorite (NaClO), hydrochloric acid (HCl), nitric acid, citric acid, sodium hydroxide (NaOH) and EDTA, have been frequently employed for membrane cleaning in AnMBRs [30,33,122,132]. Efficient chemical cleaning requires the selection of cleaning agents that target dominant compounds responsible for fouling and that do not adversely affect the membrane itself. In general, oxidizing and alkaline agents, such as NaClO and NaOH, are used to remove the microorganisms and organic foulants. Acidic agents are effective in breaking metal-associated structures including metal organic foulant complexation and inorganic scales. Coordination agents like citric acid and EDTA can remove metallic foulants as well, due to their outstanding binding ability with metal ions. It is evident that a combination of cleaning agents, such as NaClO and NaOH, is more efficient than single-agent methods [193]. The typical cleaning protocol used in AnMBRs comprises a weekly clean in place (CIP) with 500 mg/L NaClO and 2000 mg/L citric acid, and a cleaning out of place (COP) with 1000 mg/L NaClO and 2000 mg/L citric acid, conducted twice yearly [30]. The above mentioned cleaning agents are usually corrosive or caustic, and may damage membranes. In this respect, mild and environmentally friendly cleaning agents, such as purified enzymes and surfactants, have been employed to extract biologically derived foulants from polymer membranes. Allie et al. [194] demonstrated the feasibility of using both proteases and lipases to clean their UF membranes fouled by abattoir effluent. te Poele and van der Graaf [195] obtained 100% flux recovery for UF membranes by using new enzymatic cleaning protocol. Application of these agents in AnMBRs should be further investigated.

5. Conclusions and perspectives

A critical analysis of literature reveals that much progress has been achieved in applications and research of AnMBR technology. AnMBR technology features many advantages over aerobic treatment and conventional anaerobic methods, and the developments in membrane materials and modules added to its advantages. The review also demonstrates some advances in commercial AnMBR systems. AnMBRs appear to be suitable to treat most of the streams, and high treatment efficiency and high quality effluent was generally achieved, suggesting that AnMBR technology was a prospective for wastewaters treatment and subsequent reuse. Membrane fouling remained the major obstacle limiting the widespread application of AnMBR. The literature results in membrane fouling classification, mechanisms, affecting parameters and control strategies were thereby summarized and updated.

All in all, the current review demonstrates the strong possibility and need to enhance the use of AnMBR treating various streams. Despite the rapid development of AnMBRs in recent years, there are remaining several barriers or challenges that limit their widespread practical application. Thus, further breakthroughs in these challenges should be pursued in future works as summarized below:

- The literature reviewed revealed that most of the research reported on AnMBR treating wastewater is confined to bench-scale experiments. Many times, results from bench testing could not simply transfer to full-scale practical application. Further research is needed to support its wide implementation at industrial scales.
- Membrane fouling and its consequences in terms of operating costs and plant maintenance remain the critical limiting factors affecting the widespread application of AnMBRs for wastewater treatment. Although intensive efforts have been dedicated to the study on membrane fouling mechanisms and control, it is still necessary to develop more effective and easier methods to control and minimize membrane fouling especially in full-scale applications.
- The majority of membranes used in AnMBRs are UF and MF membranes, which represent significant costs of the whole AnMBR system. Thus, adopting low costs filters in AnMBR for separation should be a good solution to reduce the costs of AnMBR. Efforts aimed in better understanding on the filter properties as well as the influencing factors would enable the optimization of their performance in AnMBRs.
- AnMBRs based on pressure-driven membrane processes for treatment of wastewaters, especially municipal wastewater encountered two major challenges: membrane fouling, and low N and P removals. The recent progress in FO membrane process has provide a promising perspective to resolve the above challenges since FO process has a lower fouling propensity and can almost totally reject N and P contaminants. Continued efforts should be devoted to develop FO AnMBR system, and investigate its fouling behaviors and application in wastewater treatment. In addition, high costs of FO membrane should be reduced.
- Biogas recovery represents one of the major advantages of AnMBR. More engineering research needs to be directed toward biogas (mainly methane) recovery measures. Development of effective and economical methane recovery process would further improve economic feasibility of AnMBR for real wastewater treatment.
- It is operationally and economically advantageous to adopt anaerobic–aerobic processes in wastewater treatment. Such a process would combine the benefits of membrane separation, anaerobic digestion (i.e. biogas production) and aerobic degradation (i.e. better COD and VSS removal). Attention should be paid on the research and application of the combined process.
- There is a short of fundamental information on the operational issues, cost issues, energy issues, and manufacture cost of AnMBR systems for various wastewaters treatment. Well-controlled pilot or full scale AnMBR studies are needed to address these issues.

Above perspectives were proposed to the potential development of the AnMBR technology in the future. With more efforts being conducted in both pilot- and full-scale AnMBR systems, the prospects of developing technologically acceptable and economically feasible AnMBR treatment alternatives over conventional methods are pleasant.

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