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Perspectives on anaerobic membrane bioreactor treatment of domestic wastewater: A critical review

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ABSTRACT

Interest in increasing the sustainability of water management is leading to a reevaluation of domestic wastewater (DWW) treatment practices. A central goal is to reduce energy demands and environmental impacts while recovering resources. Anaerobic membrane bioreactors (AnMBRs) have the ability to produce a similar quality effluent to aerobic treatment, while generating useful energy and producing substantially less residuals. This review focuses on operational considerations that require further research to allow implementation of AnMBR DWW treatment. Specific topics include membrane fouling, the lower limits of hydraulic retention time and temperature allowing for adequate treatment, complications with methane recovery, and nutrient removal options. Based on the current literature, future research efforts should focus on increasing the likelihood of net energy recovery through advancements in fouling control and development of efficient methods for dissolved methane recovery. Furthermore, assessing the sustainability of AnMBR treatment requires establishment of a quantitative environmental and economic evaluation framework.

1. Introduction

Current domestic wastewater (DWW) treatment schemes are energy intensive, produce large quantities of residuals, and fail to recover the potential resources available in wastewater. In fact, municipal wastewater treatment plants account for approximately 3% of the US electrical energy demand according to the US Environmental Protection Agency's Office of Water (2006). Because of an increased interest in sustainability within water management, DWW treatment practices are being reevaluated with a focus on reducing energy demands and environmental impacts, while recovering resources in the form of water, materials, and energy (Guest et al., 2009). Considering this, it is important to note that the relatively low-strength of DWW (e.g., 5-day biochemical oxygen demand [BOD_5] of 110–350 mg/L in the US) and its production at high per capita flow rates (e.g., 190–460 L/capita $*$ d in the US) (Tchobanoglous et al., 2003) in most of the developed world make sustainable water management particularly challenging.

This focus on sustainable development is driving innovations in anaerobic biotechnology, which has long been considered an option to allow for energy recovery from DWW through the conversion of organic matter to methane-rich biogas. In comparison to aerobic biological DWW treatment, anaerobic processes require less energy input because they do not require aeration, produce a fraction of the residuals, and offer the possibility of operation in energy neutral or even positive configurations due to biogas generation (van Lier and Lettinga, 1999; Zeeman and Lettinga, 1999; Aiyuk et al., 2004; Chu et al., 2005; van Haandel et al., 2006). Conventional wisdom regarding anaerobic treatment assumes, however, that: (1) bioreactors must be heated to mesophilic (30–40 °C) or thermophilic (50–60 °C) temperatures, (2) long solids retention times (SRTs) are necessary, and (3) post-treatment is required to produce an effluent suitable for direct discharge into the aquatic environment. As a result, anaerobic processes have not been utilized widely for full-scale DWW treatment (Aiyuk et al., 2006). Low DWW temperatures in temperate and cold climates have been considered a barrier for anaerobic treatment because the energy requirements associated with heating large quantities of wastewater outweigh the energy recovery potential (Lettinga et al., 2001; Martin et al., 2011). Therefore, low-temperature, ambient or psychrophilic (<20 °C), treatment essentially is the only economically feasible option for anaerobic DWW treatment in temperate and cold climates. Furthermore, high-rate treatment with short hydraulic retention times (HRTs) is necessary to treat the large volumes of dilute DWW, while long SRTs are essential to maintain the slow growing anaerobic microbial populations in the treatment systems. At low temperatures, biomass growth is

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greatly reduced, which increases the need for a long SRT and necessitates the elimination of even minor sludge washout (Lettinga et al., 2001). Two review papers (O'Flaherty et al., 2006; van Haandel et al., 2006) independently concluded that no microbial barriers exist to anaerobic treatment of DWW, even at low temperatures, provided the system is operated at long SRTs and DWW sulfate concenterations are relatively low. The well-established upflow anaerobic sludge blanket (UASB) and expanded granular sludge bed (EGSB) reactor configurations largely meet the requirements necessary for high-rate anaerobic treatment (Seghezzo et al., 1998; Rebac et al., 1999b; Aiyuk et al., 2004). Anaerobic membrane bioreactors (AnMBRs), by coupling membrane filtration with anaerobic treatment, provide an alternative strategy for DWW treatment at low temperatures with the potential for a higher quality effluent.

AnMBRs can provide the same benefits as aerobic membrane bioreactors (AeMBRs), but may do so with reduced energy requirements. AeMBRs have gained considerable popularity in the past decade for the treatment of both high and low strength wastewater as membrane costs have decreased dramatically (Furukawa, 2008). For instance, AeMBRs have been installed in over 200 countries with 4400 total installations by the top three suppliers (Kubota, Mitsubishi Rayon, and Zenon (now GE)) as of 2009 (Judd, 2010). Furthermore, the MBR industry is predicted to have a mean growth rate of approximately 12% from 2000 to 2013 (Judd, 2010). This is largely because AeMBRs have the ability to provide superior effluent quality when compared to conventional aerobic treatment that relies on gravity sedimentation, can reduce the footprint of operation, and have potential in water reuse schemes (Daigger et al., 2005). However, AeMBRs remain energy intensive due to aeration requirements. In addition, membrane fouling continues to be a primary challenge to implementing any MBR system, aerobic or anaerobic, because of its direct effect on capital and operating costs. Consequently, many studies have been conducted to better understand fouling and to assess fouling control strategies in AeMBRs as reviewed by Le-Clech et al. (2006). Significantly less work has been done on fouling in AnMBRs, particularly in applications of low-strength wastewater treatment (Bérubé et al., 2006). Despite this, research on AnMBRs has increased substantially over the past decade because of the interest in reducing energy demands. Two review papers have already appeared on AnMBR treatment of a variety of waste streams. Liao et al. (2006) reviewed AnMBR technology for a wide range of high and low-strength wastewaters including DWW. Bérubé et al. (2006) focused on membrane fouling when considering AnMBRs for low-strength wastewater treatment. However, these reviews did not address operational concerns beyond membrane fouling for AnMBR low-strength wastewater treatment and a substantial amount of AnMBR research has been conducted since they were published.

The objective of the current review is to comprehensively discuss the available literature on AnMBRs for DWW treatment and identify the main research areas that need further attention. Interest in AnMBRs for DWW treatment has grown rapidly during the past few years, as evidenced by a surge of research publications on the topic since 2010 (Baek et al., 2010; Gao et al., 2010; Ho and Sung, 2010; Dagnew et al., 2011; Gimenez et al., 2011; Huang et al., 2011; Kim et al., 2011; Martinez-Sosa et al., 2011; Salazar-Pelaez et al., 2011). Much of the reviewed literature and studies published so far have focused on proof of concept and membrane fouling. However, a broader understanding of AnMBR technology in the context of DWW treatment is needed for successful full-scale implementation. Building on the review paper by Bérubé et al. (2006), which focused on membrane fouling, the current review discusses recent advancements in fouling control, but examines in greater detail other operational concerns that need to be resolved to allow full-scale implementation of AnMBRs for DWW treatment. For instance, the lower limits of HRT and temperature allowing for adequate treatment performance have yet to be established. The complex relationships among HRT, SRT, treatment performance, and membrane fouling are also poorly defined in the current literature. Furthermore, methane solubility, especially at low temperatures, complicates methane recovery. In addition, anaerobic treatment lacks the capacity for substantial nutrient removal, which is an important consideration when direct discharge of treated effluents in nutrient sensitive watersheds is necessary. Thus, coupling AnMBR treatment with downstream treatment is necessary to remove (and ideally recover) nutrients available in DWW. Such post-treatment, if possible, should retain the excellent AnMBR effluent quality with respect to suspended solids. Beyond nutrient removal, the removal of trace contaminants by AnMBR treatment has yet to be evaluated. It is clear that AnMBR research must extend beyond membrane fouling to best determine the circumstances under which AnMBR DWW treatment is practical and economically feasible. Therefore, this review covers recent advancements made in membrane fouling control, the effects of HRT and SRT on treatment performance and fouling, the role of the membrane biofilm in treatment, implications of temperature on AnMBR performance, complications with methane recovery, nutrient removal limitations, the fate of trace contaminants in AnMBR treatment, and finally, pilot-scale studies.

2. Selection of reactor configuration and membrane pore size, material, and configuration

Simply defined, an AnMBR is an anaerobic bioreactor coupled with membrane filtration. The membrane filtration component can exist in three configurations: external cross-flow, internal submerged, or external submerged (Liao et al., 2006). In an external cross-flow configuration, the membrane unit is separate from the bioreactor and the membranes operate under pressure to produce permeate. Suspended anaerobic biomass maintained in the bioreactor is pumped into the membrane unit creating a positive pressure that leads to permeate production. The rejected biomass or retentate is returned to the bioreactor. In an internal submerged membrane configuration, membranes are submerged directly into the suspended biomass in the bioreactor and permeate is produced by exerting a vacuum on the membrane. Alternatively, membranes may be located in an external chamber separate from the main bioreactor, but are still submerged in suspended biomass and are operated under vacuum. In such an external submerged configuration, suspended biomass from the bioreactor is pumped to the external chamber, while retentate is returned to the main bioreactor. This configuration facilitates membrane cleaning and replacement by allowing isolation of the membrane unit in an external chamber. This separation enables anaerobic conditions to be maintained in the main bioreactor during membrane cleaning or replacement.

Regardless of membrane configuration, the anaerobic bioreactor is most commonly a continuously stirred tank reactor (CSTR). Alternatives to a CSTR have also been proposed, such as UASB (Aiyuk et al., 2004; Ho and Sung, 2009), EGSB (Chu et al., 2005), and fluidized bed (Kim et al., 2011) reactors coupled with membrane filtration. These reactor designs allow for considerable biomass retention in the bioreactor, which potentially limits membrane fouling by reducing the amount of biomass in contact with the membranes (Liao et al., 2006). However, biomass growth on the membrane surface, colloidal solids, soluble microbial products (SMP), and extracellular polymeric substances (EPS; which includes extracellular carbohydrates, proteins, lipids, and nucleic acids) are also important contributors to membrane fouling (Bérubé et al., 2006). Therefore, bioreactor designs that limit membrane-biomass contact are not guaranteed to reduce fouling.

The selection of membrane pore size, material, and configuration are important design decisions. Microfiltration and ultrafiltration membranes are most commonly used in MBRs. In addition, there is growing interest in using dynamic or secondary membranes, which rely on the formation of a cake layer for biomass retention rather than on an actual membrane, in both aerobic (Chu and Li, 2006) and anaerobic applications (Zhang et al., 2010). Organic and inorganic membranes have been applied in AnMBR DWW treatment and it has been shown this choice of material can impact the type and extent of membrane fouling, as well as the associated costs (Bérubé et al., 2006). Finally, flat-sheet (Hu and Stuckey, 2007; Huang et al., 2011), tubular (Baek and Pagilla, 2006; Ho and Sung, 2009; Salazar-Pelaez et al., 2011), and hollow fiber (Wen et al., 1999; Chu et al., 2005; Lew et al., 2009; Dagnew et al., 2011; Gimenez et al., 2011; Kim et al., 2011) membranes have been studied for AnMBR DWW treatment. Table 1 presents various operational parameters and treatment performance results obtained in bench-scale AnMBR studies for DWW treatment (studies with simulated and actual DWW are included). A broad range of configurations, membrane materials, operational temperatures, and fouling control strategies have been researched.

3. Membrane fouling control

Membrane fouling continues to be a substantial challenge in advancing AnMBR technology considering membrane material costs and energy demands associated with fouling prevention. Fouling results from the accumulation of inorganic and organic foulants internally in the membrane pores and externally on the membrane surface, which reduce flux, increase TMP, and potentially necessitate chemical cleaning or membrane replacement. The primary foulants of interest in AnMBR systems include suspended biomass, colloidal solids, SMP, EPS, attached cells, and inorganic precipitates such as struvite.

Membrane fouling has been controlled through various strategies, which are linked to the membrane configuration. In external cross-flow configurations, a high cross-flow velocity is maintained to limit inorganic and organic foulant buildup on the membrane. In submerged configurations, fouling control is typically accomplished through biogas sparging, backflushing, and/or membrane relaxation. A consensus has yet to be determined on which strategy is most effective per energy input. For instance, Martin et al. (2011) highlighted the high variability in biogas sparging intensity (specific gas demands of 0.4 to 3.0 m^3/m^2 h) and thus energy demand for fouling control (0.69 to 3.41 kWh/m^3) used in submerged AnMBR studies. When comparing AeMBR and AnMBR studies, lower permeate fluxes are typically observed in AnMBRs potentially as a result of less flocculation and thus increased concentrations of fine particulates and colloidal solids at the membrane surface (Liao et al., 2006; Martin et al., 2011). However, direct comparison studies between AeMBRs and AnMBRs for DWW treatment have indicated similar fouling potential (Achilli et al., 2011) or less propensity for fouling in AnMBRs (Baek and Pagilla, 2006).

Fouling control represents the most intensive energy demand associated with AnMBR treatment, and therefore, reducing this demand is central to maximizing the potential energy recovery. Considering the low organic strength of DWW and correspondingly low potential biogas generation, minimizing energy demands associated with fouling control is likely necessary to achieve energy neutral or positive operation. To this end, Hu and Stuckey (2007) first proposed powdered or granular activated carbon (PAC or GAC) addition to submerged AnMBRs to reduce membrane fouling in conjunction with biogas sparging. Their results suggest that PAC and GAC addition increase membrane flux and enable operation under lower TMP as compared to a control AnMBR in which only biogas sparging was used. However, they did not evaluate the effect of reduced biogas sparging intensity in the presence of PAC or GAC. The PAC or GAC is not used for adsorption and therefore would not need to be regenerated or replaced during operation, however, the initial costs and potential life cycle environmental impacts of the activated carbon must still be considered. More recently, Kim et al. (2011) proposed the use of fluidized GAC through liquid recirculation without biogas sparging for fouling control. Their results show fouling may be controlled with substantially less energy input than biogas sparging requires, however, the long-term effects on the membrane material have yet to be established. This is particularly important as both studies used organic membranes and it has been suggested that aggressive fouling control through the use of PAC, GAC, or other media in contact with the membrane may be better suited for more abrasion resistant inorganic membranes, despite their higher life cycle costs (Ghyoot and Verstraete, 1997). Examining the long-term impact of these aggressive fouling control measures on organic membranes is an important area of research that has received little attention.

4. Effects of HRT and SRT on treatment performance and membrane fouling

HRT and SRT are important operational parameters that impact treatment performance and affect membrane fouling in an AnMBR. In the context of DWW treatment, a low HRT is desirable to reduce AnMBR size and the overall footprint of operation, whereas a high SRT may be required to achieve the necessary treatment performance under the constraints of discharge limits especially for lower temperatures (O'Flaherty et al., 2006). However, increasing the SRT, while keeping the HRT constant, increases the suspended biomass concentration potentially leading to decreased permeate flux (Bérubé et al., 2006; Liao et al., 2006; Huang et al., 2011). Furthermore, increasing the SRT may result in higher SMP and EPS production (Huang et al., 2011), which in-turn play a role in membrane fouling. Therefore, a tradeoff could exist between controlling HRT and SRT for membrane fouling mitigation and obtaining the necessary treatment performance.

The dependence of AnMBR treatment performance on HRT has been evaluated in various studies (Table 1; Fig. 1). Hu and Stuckey (2006) 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 DWW at mesophilic temperature (35 °C). Even at a 3-h HRT, COD removal greater than 90% was achieved. Comparing HRTs of 3.5, 4.6, and 5.7 h, Chu et al. (2005) did not observe a correlation between treatment performance and HRT at temperatures greater than 15 °C. Likewise, Huang et al. (2011) found that treatment performance was independent of HRT when comparing HRTs of 8, 10, and 12 h in an AnMBR treating a simulated DWW at 25–30 °C. Several other studies similarly concluded that HRT had little effect on AnMBR permeate quality (Ho and Sung, 2009; Lew et al., 2009; Baek et al., 2010). Ho and Sung (2009), however, observed an accumulation of soluble COD in an AnMBR operating at 25 °C when the HRT was reduced from 12 to 6 h despite stable permeate COD concentrations. Another study in which HRT was decreased from 12 to 8 and then 4 h observed an increase in permeate COD at the lowest HRT (temperature was not reported in this study), while there was no significant difference between the permeate COD values obtained for HRTs of 12 and 8 h (Salazar-Pelaez et al., 2011). Salazar-Pelaez et al. (2011) also observed an increase in retentate EPS and SMP concentrations at the lowest HRT, which resulted in increased membrane fouling. The authors recommended a lower limit be placed on HRT due to fouling concerns. Furthermore, Huang et al. (2011) noted that combining a short HRT with a long SRT

Table 1
Operational parameters and treatment performance results obtained in published bench-scale AnMBR studies for DWW treatment.

Study	Average influent strength (mg/L TCOD[a])	Temp. (°C)	Bioreactor configuration	Membrane information	Fouling control	SRT (days)	HRT (h)	Average effluent (mg/L TCOD[a]/% removal)
Wen et al. (1999)	100–2600[b]	12–25	UASB with submerged membrane	0.03 μm Polyethylene submerged hollow fiber	Periodic cleaning with 5% NaOCl	150	6 4	19/97 12/97
Chu et al. (2005)	383–849[c]	25 20 15 11	EGSB with submerged membrane	0.1 μm Polyethylene submerged hollow fiber	Backflushing and relaxation; periodic cleaning with 0.03% NaOCl	145	3.5–5.7	93–96 87–92 85–86 76–81
Hu and Stuckey (2006)	460[c]	35	Submerged AnMBR	0.4 μm Submerged hollow fiber / 0.4 μm Polyethylene chloride submerged flat sheet	Biogas sparging	∞	48 24 12 6 3	23/95 ; 25/95 29/94 ; 32/93 38/92 ; 32/93 40/91 ; 40/91 44/90 ; 43/91
Baek and Pagilla (2006)	84 [SCOD][b]	32	Completely mixed anaerobic bioreactor	0.1 μm PVDF[d] external tubular	Cross-flow; weekly cleaning with 0.1% w/w NaOH and disinfectant	∞	48 24 16 12	25/58 37/55 37/56 24/68
Saddoud et al. (2007)	685[b]	37	Jet flow anaerobic reactor	100 kDa external	Cross-flow	∞	15–60	87/88
Ho and Sung (2009)	500[c]	25	Completely mixed anaerobic reactor	1 μm PTFE[e] external tubular	Cross-flow; periodic cleaning with NaOCl	90–360	6–12	<40/>92
Lew et al. (2009)	540[b]	25	Completely mixed anaerobic reactor	0.2 μm external hollow fiber	Periodic backflushing; chemical cleaning with 0.1 M NaOH, 1% H_2O_2, and 1% HCl	∞	4.5–12	65/88
Ho and Sung (2010)	500[c]	25 15	Completely mixed anaerobic reactor	1 μm PTFE[e] external tubular	Periodic backflushing	∞	9	25/95 75/85
Gao et al. (2010)	500[c]	30	Upflow anaerobic reactor	100 kDa external coated PVDF[d] and 30 kDa external polyetherimide	Cross-flow	50	24	<20/>96
Huang et al. (2011)	550[c]	25–30	Completely mixed anaerobic reactor	0.45 μm PES[f] flat sheet	Biogas sparging	30, 60, ∞	8–12	<17/>97
Salazar-Pelaez et al. (2011)	350[c]	–	UASB with external membrane	100 kDa external PVDF[d] tubular	Cross-flow; NaOCl cleaning every 6 h	∞	4–12	70/80
Kim et al. (2011)	513[c]	35	Two-stage fluidized bed/membrane bioreactor	0.1 μm PVDF[d] hollow fiber	GAC fluidization; periodic backflushing and/or NaOCl/NaOH cleaning	∞	4.2–5.9	7/99
Smith et al. (2011)	440[c]	15	Submerged AnMBR	0.2 μm PES[f] flat sheet	Biogas sparging and backflushing	300	16	36/92

[a] TCOD = total COD.
[b] Actual DWW.
[c] Simulated DWW.
[d] PVDF = polyvinylidene fluoride.
[e] PTFE = polytetrafluoroethylene.
[f] PES = polyethersulfone.

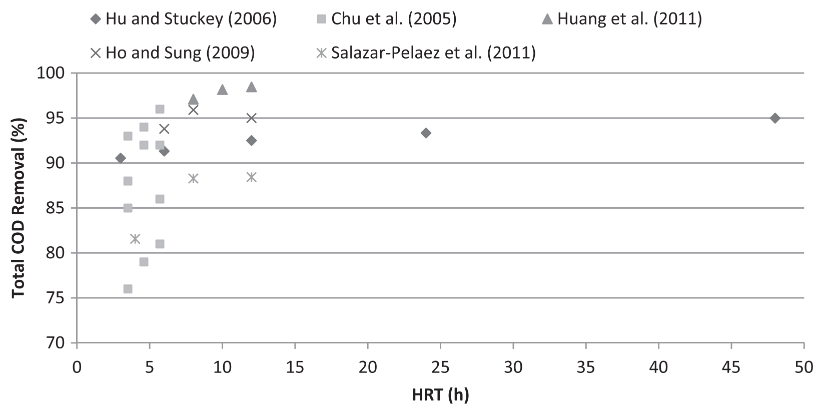

Fig. 1. Total COD removal as a function of HRT observed in Chu et al. (2005)(11, 15, 20, 25 °C), Hu and Stuckey (2006) (35 °C), Ho and Sung (2009) (25 °C), Huang et al. (2011) (25–30 °C) and Salazar-Pelaez et al. (2011) (temperature not specified)..

inevitably leads to increases in suspended biomass concentrations, which positively correlates with membrane fouling rates. Taken together, these studies suggest that adequate AnMBR treatment performance may be obtained at relatively short HRTs even at low temperatures, but that a lower limit on HRT may exist primarily due to concerns with membrane fouling.

Membrane separation enables absolute retention of biomass and thus complete control of SRT. Because of this, SRT is an easily controllable operational parameter affecting both treatment performance and membrane fouling. Baek et al. (2010) operated a bench-scale AnMBR and reduced the SRT through biomass wasting in five steps from 213 to 40 days. The decrease in SRT did not impact treatment performance or membrane fouling. Conversely, Huang et al. (2011) compared performance during operation at SRTs of 30 and 60 days and for a period without biomass wasting and observed better treatment performance at longer SRTs but at the cost of increased membrane fouling resulting from higher suspended biomass concentrations and SMP production. However, a negative correlation between EPS concentrations and SRT was found, which was linked to smaller median particle sizes in the suspended biomass as a function of reduced flocculation in the presence of lower concentrations of EPS. The authors speculated that the decrease in median particle size associated with the increase in SRT likely accelerated membrane fouling. Baek et al. (2010) also observed a decrease in EPS concentrations at higher SRTs but noted that concentrations detected were considerably lower than literature values for AeMBRs possibly indicating relatively less propensity for EPS fouling in AnMBRs. Conversely, other AnMBR studies have pointed to EPS as a major contributor to direct membrane fouling (Chu et al., 2005; Gao et al., 2010). Therefore, EPS may act to reduce membrane fouling by increasing suspended biomass particle size, whereas EPS may directly contribute to membrane fouling when present in excess or when generated directly on the membrane surface by the biofilm or cake layer. These observations suggest that a certain SRT may exist to limit EPS-mediated membrane fouling. However, the role of EPS quantity and characteristics in fouling as a function of SRT as well as other operational variables is not well understood in AnMBRs and controlling SRT is further complicated by its interrelatedness with treatment performance.

5. Role of the membrane biofilm beyond fouling

The biofilm or cake layer that develops on the membrane surface plays a role in membrane fouling, but may also contribute to soluble COD removal and thus final permeate quality in an AnMBR. The latter has received limited attention in the literature on AnMBRs for DWW treatment. Mechanisms of soluble COD removal across the membrane may include microbial activity, adsorption, size exclusion, and charge exclusion. Several AnMBR studies have shown substantial differences between the soluble COD concentrations in the bioreactor and in the permeate (Chu et al., 2005; Hu and Stuckey, 2006; Ho and Sung, 2009, 2010; Baek et al., 2010; Smith et al., 2011). In addition, Ho and Sung (2010) observed an increase in soluble COD removal across the membrane surface with decreasing temperatures. Some researchers have compared soluble COD removal in the bioreactor and soluble COD removal across the membrane and have referred to these as, respectively, "biological" and "physical" soluble COD removals (Ng et al., 2000; Baek and Pagilla, 2006; Ho and Sung, 2009). These definitions are misleading as it is improbable that biological activity does not occur in the membrane biofilm. However, it is important to understand the significance of biological soluble COD removal by the biofilm in total COD removal and relative to other potential non-biological soluble COD removal mechanisms. Ho and Sung (2010) compared specific methanogenic activities in suspended and attached biomass and found that the attached biomass was indeed biologically active although the attached biomass activities were 39% and 22% of the activities of suspended biomass at 25 and 15 °C, respectively. Conversely, Vyrides and Stuckey (2011) compared biological activity of attached and suspended biomass from an AnMBR treating high-salinity wastewater and found that the attached biomass was considerably more active under both high and low salinity conditions. The authors speculated that increases in biological activity result from lower mass-transfer limitations in the biofilm. Furthermore, Vyrides and Stuckey (2009) observed an increase in dissolved organic carbon (DOC) removal when they reduced the frequency of biogas sparging from continuous to intervals of 10 min every 15 min during treatment of high-salinity wastewater. The decrease in biogas sparging increased TMP indicating a thicker biofilm developed on the membrane, which likely caused an increase in adsorption and eventual biodegradation of high molecular weight compounds. Smith et al. (2011) operated two membrane units in parallel in an AnMBR subjected to biogas sparging. One membrane unit was backflushed at a regular interval, while the other one was not backflushed. The differences in fouling control led to a higher amount of fouling in the non-backflushed membrane unit, as indicated by higher TMPs and by visual observations. A positive correlation was observed between membrane fouling and soluble COD removal across the membrane. These results suggest that a tradeoff may exist between increased fouling and increased soluble COD removal across the membrane.

6. Temperature implications on treatment performance

Untreated DWW in the US varies in temperature from approximately 3 to 27 °C, with an average of about 16 °C (Tchobanoglous et al., 2003). Given the relatively low average temperature of DWW, heating of DWW would be necessary in the US and many regions in the world for most of the year if mesophilic treatment were required. Martin et al. (2011) concluded that influent COD concentrations higher than 4–5 g/L, an order of magnitude greater than those present in typical DWW (Tchobanoglous et al., 2003), would be necessary to generate enough biogas to heat a bioreactor to mesophilic temperatures. Therefore, operation at ambient temperatures is essential for economical implementation of AnMBRs for DWW treatment.

Hydrolysis of particulate organics is generally considered to be the rate-limiting step in anaerobic digestion (Lee and Rittmann, 2011) and is of special importance in DWW treatment as particulate organics represent a large fraction of the total COD. Hydrolysis rates decline with temperature (Lettinga et al., 2001), requiring longer SRTs for hydrolysis to occur at psychrophilic temperatures. This creates a limitation for most anaerobic treatment systems as the relatively poor settleability of anaerobic biomass makes it difficult to retain all biomass in systems that rely on gravity separation. Even minor sludge washout in such systems could reduce the SRT to below the limit necessary for acceptable treatment performance (Lettinga et al., 2001). Because membranes enable complete retention of particulates, hydrolysis rates may still be sufficiently high in an AnMBR even at psychrophilic temperatures resulting in acceptable treatment performance. Thus, other pathways may be more critical or rate-limiting in psychrophilic AnMBR operation. Ho and Sung (2009) cited acetogenesis as the rate-limiting step in AnMBRs operating at 25 °C based on an increase in bioreactor soluble COD, but a lack of volatile fatty acid (VFA) accumulation. Rebac et al. (1999a) investigated the effects of temperature on the kinetics of fatty acid degradation using psychrophilically-grown (10 °C) mesophilic seed sludge. They found that, although low temperatures negatively affected degradation rates, specific methanogenic activities at mesophilic temperatures using the psychrophilically-grown biomass were higher than the specific methanogenic activities of mesophilically-grown biomass, indicating that psychrophilic conditions do not inhibit development of methanogenic microbial communities. Overall, low temperatures may reduce maximum specific growth and substrate utilization rates of microorganisms but can also lead to an increase in net biomass yield (O'Flaherty et al., 2006). Furthermore, most biological reactions pertinent to anaerobic digestion such as hydrolysis and various fermentations are less energetically favorable at low temperatures. On the other hand, several reactions are more exergonic at low temperatures because of the increased solubility of hydrogen, including hydrogenotrophic sulfate reduction, hydrogenotrophic methanogenesis, and homoacetogenesis (Lettinga et al., 2001). This implies that hydrogenotrophic methanogenesis may be more important than aceticlastic methanogenesis at low temperatures. Indeed, McKeown et al. (2009) found increased hydrogenotrophic methanogenic activity at psychrophilic temperatures during a long-term study (1243 days) on anaerobic treatment of acidified wastewater. While the temperature of operation certainly impacts various pathways in anaerobic metabolism, there is no evidence that low temperatures are inhibitory to process performance under the appropriate operational conditions.

Despite the lack of microbial barriers at low temperatures, only a few studies have assessed AnMBR performance for DWW treatment at psychrophilic temperatures (Table 1). Even fewer studies have evaluated performance as a function of varying DWW temperature within the low-temperature range. An early study in which a bench-scale UASB system was coupled with membrane filtration treating DWW found that high COD removals (averaging 97%) could be attained at temperatures ranging from 12 to 25 °C for HRTs of 4–6 h (Wen et al., 1999). A dependence on temperature was observed in the UASB with COD removals dropping below 70% at temperatures below 15 °C. More importantly, however, was the finding that total system COD removal (UASB and membrane filtration) was only slightly affected by temperature, and remained greater than 88% at the lowest operational temperature. This result highlights the possible role membrane filtration has in performance stability across temperature fluctuations. In another study, an EGSB coupled with microfiltration treating a simulated DWW was initially operated at 25 °C and subsequently operated at 20, 15, and 11 °C. The temperature was finally increased stepwise back to 25 °C (Chu et al., 2005). COD removal decreased slightly from >90% at 25 °C with decreases in temperature to 15 °C, but sharply declined to 78% when the temperature was further reduced to 11 °C. This low COD removal may have been related to the relatively short HRTs used, i.e., 3.5–5.7 h. Indeed, changes in HRT significantly affected COD removal when the system was operated at 11 °C indicating that adequate treatment performance may be obtained at this temperature at longer HRTs. Ho and Sung compared AnMBR performance treating a simulated DWW using parallel reactors operated at 25 and 15 °C and observed COD removals greater than 95% and 85%, respectively (Ho and Sung, 2010). Although these data suggest some performance dependence on temperature, both reactors were inoculated with sludge from a mesophilic anaerobic digester and an AnMBR previously operated at 25 °C and thus higher performance at 25 °C could have been expected especially considering the relatively short operational period of 112 days. VFA profiles for the AnMBR operated at 15 °C show a long acclimation phase of approximately 60 days after which permeate VFAs remained low suggesting that performance of the two reactors may have converged over a longer operational period. Smith et al. (2011) showed COD removals greater than 90% could be attained in an AnMBR operated at 15 °C.

Low temperatures may impact the choice of an appropriate inoculum for seeding an AnMBR. Kashyap et al. (2003) and O'Flaherty et al. (2006) suggested that psychrophiles from natural habitats be considered for potential use in psychrophilic anaerobic treatment processes citing the large number of psychrophilic methanogens and acetogens isolated from terrestrial ecosystems. In fact, a psychrophilic methanogen was recently isolated from the Zoige Wetland of the Tibetan Plateau that is active at temperature as low as 0 °C (Zhang et al., 2008). Xing et al. (2010) investigated the use of psychrophilic inocula from natural habitats (lake sediments) in anaerobic digestion at 15 °C and determined that inoculation with psychrophilic biomass is feasible. Conversely, Rebac et al. (1999a) concluded that psychrotolerant mesophiles were adequate for psychrophilic anaerobic treatment and that true psychrophiles are not required. Smith et al. (2011) seeded an AnMBR with inocula from two mesophilic and one psychrophilic environment and compared the microbial community structure of suspended biomass and membrane biofilm samples taken after 275 days of operation at 15 °C. The AnMBR microbial communities most closely resembled the mesophilic inocula rather than the psychrophilic inoculum suggesting that psychrotolerant mesophiles dominated in their system (Smith et al., 2011). Nonetheless, AnMBR inoculation with truly psychrophilic microbial communities may be critical for process performance and stability at even lower temperatures. O'Flaherty et al. (2006) commented that better characterization of psychrophiles in anaerobic treatment may require longer term operation (>1000 days) at psychrophilic conditions to allow enrichment of psychrophilic microorganisms relative to psychrotolerant mesophiles. It should be noted that, to the best of our knowledge, no studies have assessed anaerobic treatment performance of psychrophiles at elevated temperatures. This is of particular importance in AnMBR DWW treatment as most temperate climates experience seasonal temperature variation that lead to fluctuations in DWW temperature between approximately 5 and 25 °C. Thus, it is possible that performance may deteriorate at higher temperatures if the AnMBR is seeded with only psychrophilic biomass. A better approach may be to seed the AnMBR with a diverse consortium of mesophilic and psychrophilic microbial communities to maintain performance across seasonal temperature variation.

7. Complications with methane solubility and recovery

Anaerobic treatment enables energy recovery from DWW as long as methane can be easily collected. Capturing methane is also important to mitigate direct greenhouse gas emissions as methane has a global warming potential 25 times that of carbon dioxide (IPCC, 2007). Because of this, methane should not be emitted to the atmosphere during AnMBR operation. Methane in the gas phase can be easily collected, but dissolved methane is more difficult to capture. Specifically, methane is approximately 1.5 times more soluble at 15 °C compared to 35 °C, for a typical biogas methane content of 70%. Because of the relatively low strength of DWW, dissolved methane leaving the treatment process in the liquid phase represents a substantial portion of the total methane generated. Consequently, recovery of dissolved methane is key to approaching energy-neutral AnMBR DWW treatment.

Few AnMBR studies have addressed methane solubility (Hu and Stuckey, 2006; Dagnew et al., 2011; Gimenez et al., 2011) and even fewer have quantified dissolved methane (Kim et al., 2011; Smith et al., 2011). Kim et al. (2011) reported that 30% of the methane generated left their system through the liquid phase during operation at 35 °C. Smith et al. (2011) observed that approximately 50% of the methane generated remained in the liquid phase during operation at 15 °C, highlighting the important role temperature has in methane solubility and, therefore, direct biogas methane recovery. In a recent study by Bandara et al. (2011), dissolved methane in the effluent of a bench-scale UASB reactor treating a simulated wastewater operated at 35, 25, and 15 °C was quantified and recovered through the use of a degassing membrane, a nonporous membrane only permeable to gases. As expected, an increase in the dissolved methane concentration was observed with a decrease in temperature. Dissolved methane concentrations in the UASB effluent were on average 15.8, 20.5, and 26.0 mg/L (converted from concentrations provided in COD based units) at temperatures of 35, 25, and 15 °C, respectively. Comparing these concentrations using concentrations derived using Henry's Law based on the biogas methane content reported suggests a high level of methane oversaturation in the UASB effluent. A number of other studies using anaerobic bioreactors without membrane separation for low-strength wastewater treatment also found methane oversaturation (e.g., Singh et al., 1996; Hartley and Lant, 2006). Pauss et al. (1990) provided a theoretical and experimental evaluation of methane oversaturation for a number of anaerobic bioreactor configurations without membrane separation and cited mass transfer limitations as the cause of the observed oversaturation. The methane oversaturation reported in these non-MBR studies may have resulted from poor gas–liquid phase equilibrium. Utilizing biogas sparging for fouling control in an AnMBR would be expected to result in methane equilibrium by limiting the effect of mass transfer limitations. However, methane oversaturation was also observed in an AnMBR study in which biogas sparging was used (Smith et al., 2011). Since substantial soluble COD removal took place across the biofilm in this study, it was hypothesized that methane oversaturation in the permeate was, at least in part, due to methane generation via biological activity in the biofilm. The presence of TMP across the biofilm likely forces methane generated by methanogens in the biofilm into the permeate stream, regardless of methane saturation. Taken together, the results of several studies indicate methane oversaturation should be expected in AnMBR permeate.

Several methane removal processes have been proposed to capture dissolved methane, including stripping of AnMBR effluent through post-treatment aeration (Hartley and Lant, 2006; McCarty et al., 2011), methane recovery using a degassing membrane (Bandara et al., 2011), and the use of a down-flow hanging sponge (DHS) reactor (Hatamoto et al., 2010). Methane stripping with air is commonly employed on landfill leachate to limit methane release from the liquid to the gas phase in sewer systems. Energy demands associated with methane stripping with air are estimated to be less than 0.05 kWh/m^3 of AnMBR permeate (McCarty et al., 2011). Energy recovery from the resulting mixture of methane and air has not yet been attempted. Foreseeable complications with this practice include the dilution of methane with air and potential explosion hazards resulting from a methane and oxygen rich off-gas. Furthermore, the efficiency of this practice for removing dissolved methane from AnMBR effluent is not well established. The use of degassing membranes represents a more controlled approach by which methane is recovered from AnMBR effluent but not diluted with air. Bandara et al. (2011) observed high recoveries of dissolved methane using degassing membranes with higher efficiencies for lower temperature as a result of increased methane solubility at lower temperatures. However, the degassing technology used was energy intensive; energy requirements for degassing were 300 times the amount of energy embedded in the recovered methane. Therefore, although this technology is worth further investigation, energy requirements must be substantially reduced for economic feasibility. Finally, biological methane oxidation through the use of a DHS reactor has been evaluated (Hatamoto et al., 2010). Using this system, up to 95% of the dissolved methane in the effluent was oxidized by methanotrophs. However, because dissolved methane was oxidized, methane could not be recovered for energy generation using this approach. This technology shows promise for drastically reducing potential greenhouse gas emissions from AnMBR effluent, but at the cost of energy requirements to operate the DHS and lost energy potential in the methane oxidized. Overall, dissolved methane recovery or oxidation is possible through a number of methods although each with substantial drawbacks. Addressing the issue of dissolved methane perhaps represents the greatest barrier to AnMBR implementation.

8. Nutrient removal limitations

Another major barrier to full-scale adoption of AnMBR DWW treatment is the lack of direct nutrient removal capability. Some nutrient removal takes place as a result of biomass growth, but is limited due to the low biomass yields typical for anaerobic microbes. In addition, ammonium and phosphate concentrations increase as a result of ammonification and phosphate release under anaerobic conditions. Challenges exist in coupling AnMBRs with conventional biological nutrient removal treatment technologies due to the low COD:N and COD:P ratios typical of AnMBR effluents. Biological nitrogen and phosphorus removal processes require sufficient amounts of organic electron donor to fuel denitrification and enhanced biological phosphorus removal. Chernicharo (2006) suggests treating only a fraction of the waste anaerobically (50–70%) and using the remaining fraction to support denitrification in downstream biological nitrogen removal. Aerobic or partial aerobic treatment common to many biological nutrient removal systems detracts from the energy savings gained by using anaerobic treatment. In addition, dissolved methane present in AnMBR effluents will be stripped during aerobic treatment, contributing to greenhouse gas emissions. The challenges of capturing or oxidizing methane dissolved in AnMBR effluents and achieving nutrient removal are inextricably linked.

An attractive option for biological nitrogen removal downstream of AnMBR DWW treatment may be partial nitritation/nitrification (i.e., partial oxidation of ammonium to nitrite/nitrate) coupled with anaerobic ammonium oxidation (anammox), a process in which ammonium and nitrite are converted to nitrogen gas by anammox bacteria (Van De Graaf et al., 1996). It has received increasing attention as a cost-effective nitrogen removal strategy in comparison to traditional nitrification/denitrification approaches (Schmidt et al., 2003). Because it is a strictly autotrophic process and ammonium serves as the electron donor, no additional carbon source/electron donor is required to fuel denitrification. Additional benefits of anammox include limited sludge production, low energy input, and almost complete nitrogen removal (some nitrate is produced) (Gao and Tao, 2011). Full-scale anammox treatment has been applied successfully in Europe, Japan, and China where it is being used to treat high-ammonium waste streams, such as industrial wastewaters and anaerobic digestates (Gao and Tao, 2011). However, control of the partial nitritation process can be challenging, startup times are typically long due to the slow-growth of anammox bacteria, and mesophilic temperatures are thought to be necessary. Furthermore, little research has been conducted on anammox treatment of waste streams with relatively low ammonium concentrations such as DWW and, to the best of our knowledge, no papers have reported results from studies that couple AnMBR treatment with downstream anammox treatment. Regardless, the anammox process has been proposed for mainstream DWW nitrogen removal (Kartal et al., 2010; O'Shaughnessy et al., 2011). A downstream anammox system that utilizes a biofilm process would enable nitrogen removal while maintaining the high quality of AnMBR effluent with regards to suspended solids. With additional research and process optimization, it may represent a viable nitrogen removal option for use in combination with AnMBR DWW treatment.

Physical/chemical nutrient removal processes are also promising, but can be significantly more energy intensive than biological treatment. Several studies have examined physical/chemical treatments coupled with anaerobic treatment to achieve nutrient removal. Aiyuk et al. (2004) proposed pretreatment to remove suspended solids and phosphorus through flocculation/coagulation, and ammonia removal through post-treatment zeolite adsorption in applications of anaerobic DWW treatment. Overall, 94% phosphorus and 99% nitrogen removals were achieved in their study, indicating that this approach may be applicable for achieving sufficient nutrient removal in AnMBR DWW treatment. Struvite (magnesium ammonium phosphate) precipitation represents another means of nutrient removal with potential use in AnMBR DWW treatment processes (de-Bashan and Bashan, 2004). A benefit of this approach is that recovered struvite is saleable as a fertilizer and struvite recovery limits potential pipe scaling and membrane fouling. A disadvantage of this approach is that magnesium must be added to encourage struvite formation because its levels are usually limited in DWW. Furthermore, although struvite precipitation typically removes all the phosphorus, the stoichiometry of the process means that for medium-strength US DWW only 12.5% of ammonium will be removed through struvite precipitation and that residual ammonium will remain (Tchobanoglous et al., 2003). Johir et al. (2011) proposed using an ion-exchange/adsorption process downstream of an AeMBR for nutrient recovery. The study found that an ion-exchange resin (Purolite, Bala Cynwyd, PA) achieved phosphate and nitrate removal efficiencies of 85% and 95%, respectively. Ion-exchange membrane bioreactors, similar to those demonstrated by Matos et al. (2009) to remove nitrate in marine systems, may also be coupled with AnMBRs to achieve nutrient removal. Alternatively, ion-exchange resins that are specific for ammonium and phosphate removal could be used. In either ion exchange process, the nutrients can be recovered during regeneration, but drawbacks include large capital and chemical regeneration costs (Miladinovic and Weatherley, 2008).

The nutrients in AnMBR effluent can be harnessed and recycled if the effluent is used for irrigation purposes (McCarty et al., 2011). Offsetting the environmental impacts associated with artificial fertilizer use could thus be an added benefit to AnMBR DWW treatment. Challenges regarding the transport of AnMBR effluent for irrigation or locating treatment facilities in close proximity to agricultural areas certainly exist; however, this solution maximizes resource recovery from DWW.

9. Trace contaminant fate considerations

As AnMBRs move towards full-scale implementation for the treatment of DWW, the fate of trace contaminants, such as pharmaceuticals and personal care products (PPCPs), during AnMBR treatment requires further attention. Trace contaminants are widely detected in aquatic environments (Kolpin et al., 2002) and many are present in DWW treatment plant effluents (Rosal et al., 2010). Despite the fact that DWW treatment plants represent an important first line of defense against the proliferation of these emerging contaminants in the environment, the DWW treatment plant design process typically does not consider trace contaminant removal. Nevertheless, partial to complete removal does occur in traditional DWW treatment systems for some compounds (Metcalfe et al., 2003), but levels of removal for a given compound can vary widely depending upon the process configuration (Fent et al., 2006). Few studies have focused on the fate of trace contaminants found in DWW during anaerobic treatment and, to the best of our knowledge, none have studied pharmaceutical and PPCP removal during AnMBR treatment. Carballa et al. (2007) studied the fate of PPCPs during anaerobic digestion of sewage sludge, but not during anaerobic DWW treatment, and found varying degrees of removal depending on the specific compound. Microbial aerobic (or oxic) degradation pathways for substituted and unsubstituted aromatic rings, chemical structures common to many PPCPs, have been studied extensively (e.g., Harayama et al., 1992). However, reductive pathways utilized during anaerobic treatment are not well understood and are an area in need of further research.

As we consider the potential role of AnMBR processes on trace contaminant fate in resource recovery systems, the applicability of existing knowledge about the anaerobic fate of xenobiotic compounds to trace contaminant fate needs to be evaluated. Most studies on microbial degradation in anaerobic environments work with high concentrations of the target compounds (mg/L–μg/L range), concentrations that are far from environmentally relevant (typically in the μg/L–ng/L range for DWW). This choice may dramatically impact values obtained for degradation kinetics. A variety of anaerobic microbes, including denitrifiers, iron-reducers, sulfate-reducers, methanogens, and anoxygenic phototrophs, are capable of degrading aromatic compounds (Tierney et al., 2010). A better understanding of the microbes involved and enzymes used in degrading trace contaminants in anaerobic environments will help inform the development of fate pathways in AnMBRs. In addition, transformation products of anaerobically degraded trace contaminants and their ecotoxicity or public health risks are largely unknown and uncharacterized. As pharmaceuticals and PPCPs evolve and take new forms, such as with the emergence of nanoparticles in medicine (Wagner et al., 2006), their behavior and biodegradability may also change. Understanding the fate of trace contaminants in AnMBRs and post-treatment processes, which utilize different redox environments than conventional treatment systems, is necessary to ensure a safe effluent that limits trace contaminant pollution.

10. Pilot-scale studies

The performance of AnMBRs treating DWW has been assessed in three recent pilot-scale studies by Gimenez et al. (2011), Dagnew et al. (2011) and Martinez-Sosa et al. (2011) (Table 2). Each one of these studies indicates that treatment performance similar to that observed during bench-scale research may be obtained at a larger scale. Furthermore, they report that membrane fouling may be avoided in the long-term. Gimenez et al. (2011), however, highlighted that high sulfate concentrations in DWW severely reduce the potential methane generation and energy recovery of AnMBR systems. Considering additional complications with sulfide corrosion and the need for biogas scrubbing, AnMBR treatment of sulfate-rich DWW should be avoided.

Gimenez et al. (2011) operated a pilot-scale facility fed with pre-treated DWW at a 70 day SRT, an HRT ranging from 20 to 6 h, and a temperature of 33 °C. The pilot consisted of an anaerobic reactor connected to two membrane tanks with 0.05 μm hollow fiber membranes. The total liquid volume of the system was 2500 L. The pilot also included a rotofilter for pre-treatment screening, an equalization tank, and a degasification vessel installed between the membrane tanks and permeate pump. Biogas sparging, relaxation, and backflushing were employed for membrane fouling control. Biogas sparging was also utilized in the anaerobic reactor to enhance mixing. The total and soluble COD concentrations in the influent averaged 445 ± 95 and 73 ± 25 mg/L, respectively. Sulfate concentrations were particularly high, averaging 297 ± 54 mg/L, an order of magnitude higher than average sulfate concentrations reported for DWW (Tchobanoglous et al., 2003). During the study, COD removal averaged 87% during stable operation resulting in a permeate COD of 77 mg/L. The high levels of sulfate in the influent greatly impacted biogas production as methanogens and sulfate reducers compete for substrates. Theoretically, 0.67 mg/L of COD is consumed per 1 mg/L of sulfate reduced, therefore, assuming complete sulfate reduction occurred, approximately 45% of the influent COD was consumed for sulfate reduction rather than for methanogenesis. The authors noted that sulfate removal was below 50% during startup but quickly increased to near complete removal. This increase in sulfate removal also correlated with an increase in the relative abundance of sulfate reducing bacteria. Despite substantial production of sulfides during operation, the methane content in the biogas averaged 55%. Gimenez et al. (2011) discussed methane solubility, but did not quantify dissolved methane. In addition, the effectiveness of the degasification vessel was not discussed. No irreversible fouling was observed during the study indicating that the combination of relaxation, backflushing, and biogas sparging was effective at preventing fouling while operating at a sub-critical flux of 10 L/m^2 h (LMH). According to the authors, the pilot is currently being operated at 20 °C to assess the impact of lower temperature on treatment performance.

Dagnew et al. (2011) operated a 630 L pilot-scale AnMBR for DWW treatment with a configuration similar to the one described by Gimenez et al. (2011). The system was operated at an HRT of 8.5 h, an SRT of 80–100 days, a temperature of 22 °C, and was fed screened DWW. Membrane relaxation and biogas sparging were used to control fouling, while operating at a sub-critical flux of 17 LMH. In addition, the membranes were chemically cleaned on a weekly basis. During the study, 79% and 85% COD and BOD_5 removals were observed, respectively. Permeate COD and BOD_5 concentrations averaged 47 and 14 mg/L, respectively, indicating good treatment performance. Essentially no membrane fouling was detected based on TMP at the flux used (17 LMH) even though the flux was relatively high in comparison to other studies. However, this performance may have resulted from the unnecessarily aggressive chemical cleaning schedule. Membrane cleaning was done weekly rather than based on feedback from membrane performance.

Martinez-Sosa et al. (2011) operated a pilot-scale AnMBR with a total volume of 350 L for DWW treatment. Consistent with the other pilot studies, their system consisted of an external submerged reactor configuration with hollow fiber membranes. The pilot was operated for 100 days over which the temperature was reduced from 35 to 28 °C on day 69, and then to 20 °C on day 79. Membrane fouling was controlled using biogas sparging, membrane relaxation, and periodic backwashing. The reactor was operated at a sub-critical flux of 7 LMH and an HRT of 19.2 h during the entire operational period. Suspended biomass was only removed from the AnMBR for sampling purposes and therefore the system had an SRT of approximately 680 days. DWW was used as the influent, however, it was substantially supplemented with glucose to increase the average total COD from 398 to 630 mg/L. Regardless of temperature, COD removals remained approximately 90% except for some brief performance perturbations when the reactor temperature was reduced. The high COD removal may be misleading, however, since glucose, an easily biodegradable substrate, was added to the influent. An analysis of VFAs in the reactor and permeate suggested VFA degradation by the membrane biofilm, supporting the results of bench-scale studies discussed above.

Table 2
Operational parameters and treatment performance results obtained in published pilot-scale AnMBR studies for DWW treatment.

Study	Average influent strength (mg/L TCOD[a])	Temp. (°C)	Bioreactor configuration	Membrane information	Fouling control	SRT (days)	HRT (h)	Average effluent (mg/L TCOD[a]/% removal)
Gimenez et al. (2011)	445[b]	33	Completely mixed anaerobic reactor (pilot-scale)	0.05 μm hollow fiber	Biogas sparging	70	6–21	77/83
Dagnew et al. (2011)	224[b]	22	Completely mixed anaerobic reactor (pilot-scale)	ZeeWeed™ hollow fiber	Biogas sparging; relaxation; weekly chemical cleaning	80–100	8.5	47/79
Martinez-Sosa et al. (2011)	630[c]	35 28 20	Completely mixed anaerobic reactor (pilot-scale)	38 nm PES[d] flat sheet	Biogas sparging; relaxation; backflushing	680	19.2	<80/90

™General Electric Compan.
[a] TCOD = total COD.
[b] Actual DWW.
[c] Actual DWW supplemented with glucose.
[d] PES = polyethersulfone.

11. Future research needs

This review paper has shown that many of the inherent benefits of anaerobic treatment may be obtained through AnMBR DWW treatment, while generating an effluent that is comparable in quality to effluent obtained through aerobic treatment. The majority of studies indicated adequate DWW treatment performance at a wide range of operational parameters including low temperatures and HRTs comparable to aerobic treatment. In addition, advancements in fouling control offer the potential to reduce energy requirements. The potential of AnMBR treatment of DWW has been assessed in several recent pilot-scale studies. However, additional fundamental research, pilot-scale investigations, as well as quantitative environmental and economic evaluations are needed before widespread full-scale AnMBR implementation will take place. For instance, more membrane fouling research is needed to enable operation at higher fluxes under the constraints of low energy requirements for fouling control. There is also limited research on the effects of different membrane materials and larger pore sizes (e.g., dynamic membranes) which may enable operation at higher fluxes and could greatly increase the likelihood of full-scale implementation. Furthermore, the lower limits of AnMBR treatment in terms of temperature have not yet been fully established. Implications of low temperatures on microbial pathways, microbial community structure, and the appropriate inoculum in AnMBR DWW treatment also requires further research. Additionally, the relationships among HRT, SRT, treatment performance, and membrane fouling in AnMBRs are complex and poorly defined in the current literature. The role of the membrane biofilm in treatment also warrants more research along with efforts to evaluate and characterize the fate of trace contaminants in AnMBR treatment. Moreover, nutrient recovery/removal processes such as struvite precipitation and anammox should be evaluated in conjunction with AnMBRs. Ultimately, the recovery or handling of dissolved methane represents the most challenging barrier to AnMBR implementation. Advancements must be made to sustainably recover or oxidize discharged dissolved methane before AnMBR technology can be implemented on a larger scale. In general, future research on AnMBR DWW treatment must be performed at low temperatures considering DWW in cold and temperate climates is relatively cold and it is not feasible to heat DWW to mesophilic temperatures. We recommend that future research efforts specifically focus on advancements in membrane fouling that reduce energy demands, efficient methods for dissolved methane handling, and establishment of a quantitative environmental and economic evaluation of the technology based on controllable design variables.

12. Conclusions

AnMBRs have the ability to produce effluents similar in quality to those generated during aerobic treatment, while recovering energy and producing substantially less residuals. The majority of studies at the bench-scale and pilot-scale indicated adequate treatment performance at HRTs comparable to those used in aerobic treatment and at low temperatures. However, a number of operational concerns exist that require further research before AnMBR DWW treatment can reach full-scale implementation. Specifically, future research efforts should focus on advancements in membrane fouling that reduce energy demands, efficient methods for dissolved methane handling, and establishment of a quantitative environmental and economic evaluation framework.

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