Recent Developments in Municipal Wastewater Treatment Using Anaerobic Membrane Bioreactor: A Review

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ABSTRACT

This review summarizes progresses in research on municipal wastewater treatment using anaerobic membrane bioreactors (AnMBR). At first, the AnMBR process options including membrane configuration and mean pore size or type are introduced. Next, many factors affect AnMBR performance in treating municipal wastewater, so this paper mainly focuses on the membrane characteristics, temperature, hydraulic retention time and solids retention time. The application of anaerobic membrane bioreactor relative to conventional anaerobic treatment processes is also evaluated. In recent years, pilot-scale research on AnMBR has been reported. However, there are still some problems which need to be overcome for future development. Finally, this paper summarizes the challenges to improve the AnMBR process, including the control of membrane fouling, the recovery of dissolved methane from the AnMBR system.

Keywords: Anaerobic membrane bioreactor; municipal wastewater treatment; treatment performance; dissolved methane; low temperature

1. INTRODUCTION

Current municipal wastewater treatment schemes are energy intensive, produce large quantities of residuals, and fail to recover the potential resources available in wastewater. Aerobic membrane bioreactor (MBR) technology was widely introduced for municipal wastewater treatment in the early 1990s (Smith et al., 2012). It is characterized by numerous advantages compared to the conventional activated sludge process: fast start-ups of the reactors (Ferraris et al., 2009), small footprint, high removal efficiency, high organic loading rates without any biomass losses, control over solids retention times (SRTs) and hydraulic retention times (HRTs), and maintenance of high mixed-liquor suspended solids (MLSS) concentrations (Akram and Stuckey, 2008a; Aquino et al., 2006; Van Nieuwenhuijzen et al., 2008; Yuan et al., 2008).

Anaerobic digestion is proposed as an energy saving process because there is no need for aeration and due to lower sludge production. In addition, since generated biogas
can be used as an energy source, the beneficial side-effect of producing energy is also expected. It is reported that in cases where anaerobic digestion is applied to effluents of low strength and high volumes such as municipal wastewater, a stable processing performance cannot be achieved and post-treatment processing may also be required (Khan et al., 2011).

The combination of membrane separation technology and an anaerobic bioreactor may allow for more sustainable municipal wastewater treatment with complete biomass retention, together with the added benefits of lower sludge production, enhanced high quality effluent, net energy production, and without the extra costs for aeration associated with the conventional activated sludge process (Ferraris et al., 2009). So many researchers have recently studied the municipal wastewater treatment using anaerobic membrane bioreactor (AnMBR).

There have been many reviews of AnMBR published elsewhere (Bérubé et al., 2006; Dereli et al., 2012; Skouteris et al., 2012; Smith et al., 2012; Stuckey, 2012). This article thus pays particular attention to the following new points; (1) This article reviews recent progress in AnMBR treatment of municipal wastewater, (2) The applications of AnMBR are introduced and summarized focusing on membrane characterization (type, pore size, surface area), membrane fouling, influences of different parameters like solid retention time (SRT), hydraulic retention time (HRT), temperature, methane formation and recovery, and compared with research on the conventional anaerobic treatment process.

2. THE PROCESS OF AnMBR

2.1 Membrane configuration

In the filtration method, there are two types of filtration; dead-end filtration and cross-flow filtration. Dead-end filtration has the following advantages: the collection rate is high (almost 100%); miniaturization is possible; the cost is low; and backwashing and chemical cleaning are not required. However, the membrane exchange frequency is high. On the other hand, with cross-flow filtration, the filter membrane does not have to be changed often and it can also be reused after backwashing and chemical cleaning. There are however disadvantages such as the low collection rate and high cost. Most AnMBR are using cross-flow filtration. The combination of anaerobic reactors and membranes can be engineered in primarily three different ways, as shown in Fig. 1 (Liao et al., 2006). In the first variation the membrane is situated outside the anaerobic reactor (Fig. 1a) which makes membrane cleaning and replacement simple, but sludge adheres to the membrane surface easily. Thereby it is necessary to scourge the membrane surface to reduce membrane fouling. This also provides quite a high pressure to force the liquid through the membrane. It results in high fluxes, the power and energy costs are high; it is also clear that some pump types lead to floc and cell shear with a decrease in overall particle size, and an increase in soluble organics in anaerobic systems (Kim et al., 2001).

Another way of operating these reactors is to use a vacuum to draw the effluent out through the membrane. The membrane can either be immersed in the anaerobic digester (Fig. 1b) where the advantage is that the energy required for pumping is eliminated, although biogas needs to be recycled from the headspace to keep the membranes clear from fouling, or in a separate reactor (Fig. 1c). The separation type requires a pump, but it is easier to clean the module. So the separation type is a common method in aerobic treatment. Recently, more and more literature using the submerged configuration has been published in research.
on AnMBR (Akram and Stuckey, 2008b; Bohdziewicz et al., 2008; Hu and Stuckey, 2006; Van Zyl et al., 2008; Walker et al., 2009).

2.2 Membrane pore size and type

The membrane mean pore size (pore size) and process configuration are important parameters in the system design. There are four membrane types i.e. microfiltration (MF; 0.05~10 µm), ultrafiltration (UF; 0.002~0.01 µm), nanofiltration (NF; 0.001~0.002 µm) and reverse osmosis (RO; ~0.002 µm) categorized in terms of membrane pore sizes. MF and UF are most commonly used in MBR and AnMBR. In addition, flat-sheet, tubular, hollow fiber membranes have been studied for AnMBR municipal wastewater treatment. Hollow fibers can be used for submerged reactors as well (Chu et al., 2005; Kim et al., 2010; Lew et al., 2009; Liu et al., 2013; Yoo et al., 2012). However in research institutions, flat-sheets have been focused on since they have high stability and are comparatively easy to clean and exchange (Kanai et al., 2010; Kim et al., 2007; Kocadagistan and Topcu, 2007; Lin et al., 2011). Tubular membranes have the advantage of a decrease in membrane fouling (An et al., 2009a; Calderón et al., 2011; Herrera-Robledo et al., 2011; Ho and Sung, 2007; Salazar-Pelaez et al., 2011). On the other hand, since it requires high pressure for filtration the energy input of the process operation is high.

3. MUNICIPAL WASTEWATER TREATMENT USING ANMBR

In municipal wastewater treatment by AnMBR, some researchers use synthetic and others real municipal wastewater. In addition, research using pilot scale plant for treating municipal wastewater has been reported. Research results using synthetic municipal wastewater, real municipal wastewater and treatment at pilot scale are summarized in Table 1, Table 2 and Table 3, respectively. Real municipal wastewater used in the experiments was after primary sediment or suspended solids removal, such as secondary treatment wastewater. Many studies have used submerged AnMBR. Moreover, the membranes used were mainly hollow fiber or flat-sheet, and there have been few studies using tubular configurations. The pore size used in the studies was mainly between 0.1-1.0 µm, but membranes with large pore sizes, such as 12 µm or 61 µm, have also been used (Ho et al., 2007; Zhang et al., 2010, 2011). In experiments at pilot scale, membranes with pore sizes of below 0.1 µm were mostly used.

![Figure 1](image-url) A schematic of AnMBR configurations; (a) cross-flow AnMBR, (b) Submerged AnMBR (immersed), (c) Submerged AnMBR (separated)
<table>
<thead>
<tr>
<th>Volume (L)</th>
<th>Conf.</th>
<th>Membrane (type, pore size, area)</th>
<th>Temp. (°C)</th>
<th>OLR (kg COD/m²/day)</th>
<th>HRT (h)</th>
<th>Inf. COD (mg/L)</th>
<th>Eff. COD (mg/L)</th>
<th>COD Removal (%)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>submerged</td>
<td>hollow fiber, 0.1 µm, 1 m²</td>
<td>27-30</td>
<td>1 or 6</td>
<td>2 or 12</td>
<td>500 ±10</td>
<td>-</td>
<td>TOC 90</td>
<td>Liu et al. (2013)</td>
</tr>
<tr>
<td>7</td>
<td>-</td>
<td>flat sheet, 0.2 µm, 0.0387 m²</td>
<td>15</td>
<td>0.44-0.66</td>
<td>16-24</td>
<td>440 ±68</td>
<td>36 ± 21</td>
<td>92</td>
<td>Smith et al. (2013)</td>
</tr>
<tr>
<td>4.33</td>
<td>submerged</td>
<td>tubular, 0.0085 m²</td>
<td>21-24</td>
<td>1.2</td>
<td>8</td>
<td>452 ±35</td>
<td>8-18</td>
<td>98</td>
<td>Cerón-Vivas et al. (2012)</td>
</tr>
<tr>
<td>12.5</td>
<td>cross-flow</td>
<td>tubular, 100kDa</td>
<td>-</td>
<td>-</td>
<td>4-12</td>
<td>350 ±10</td>
<td>45-65</td>
<td>80</td>
<td>Salazar-Pelaez et al. (2011)</td>
</tr>
<tr>
<td>5.93</td>
<td>submerged</td>
<td>hollow fiber, 0.1 µm, 0.091 m²</td>
<td>35</td>
<td>4.4-6.2</td>
<td>4.2-5</td>
<td>513</td>
<td>7 ± 4</td>
<td>99</td>
<td>Kim et al. (2010)</td>
</tr>
<tr>
<td>6</td>
<td>submerged</td>
<td>0.45 µm, 0.118 m²</td>
<td>25-30</td>
<td>1.1-1.65</td>
<td>8-12</td>
<td>550</td>
<td>-</td>
<td>99</td>
<td>Huang et al. (2011)</td>
</tr>
<tr>
<td>10</td>
<td>side-stream</td>
<td>flat sheet, UF</td>
<td>30</td>
<td>5.1</td>
<td>24</td>
<td>500</td>
<td>&lt; 20</td>
<td>&gt; 96</td>
<td>Gao et al. (2010)</td>
</tr>
<tr>
<td>4</td>
<td>cross-flow</td>
<td>tubular, 1 µm, 0.09 m²</td>
<td>25</td>
<td>1</td>
<td>12</td>
<td>500</td>
<td>20</td>
<td>95</td>
<td>Ho and Sung (2007)</td>
</tr>
<tr>
<td>-</td>
<td>cross-flow</td>
<td>10 or 12 µm, 0.03 m²</td>
<td>25</td>
<td>-</td>
<td>18</td>
<td>500</td>
<td>&lt; 30</td>
<td>98</td>
<td>Ho and Sung (2009)</td>
</tr>
<tr>
<td>-</td>
<td>submerged</td>
<td>flat sheet, 0.22 µm, 0.05 m²</td>
<td>35</td>
<td>0.3</td>
<td>6</td>
<td>150</td>
<td>5-10</td>
<td>-</td>
<td>Wu et al. (2009)</td>
</tr>
<tr>
<td>4</td>
<td>-</td>
<td>1 µm</td>
<td>25</td>
<td>1-2</td>
<td>6-12</td>
<td>500</td>
<td>&lt; 30</td>
<td>94</td>
<td>Ho and Sung (2009)</td>
</tr>
<tr>
<td>3</td>
<td>submerged</td>
<td>0.4 µm</td>
<td>35</td>
<td>-</td>
<td>3-24</td>
<td>460 ±20</td>
<td>35</td>
<td>95</td>
<td>Hu and Stuckey (2007)</td>
</tr>
<tr>
<td>3</td>
<td>submerged</td>
<td>flat sheet, 0.4 µm, 0.1 m²</td>
<td>35</td>
<td>-</td>
<td>6</td>
<td>450 ±20</td>
<td>18</td>
<td>96</td>
<td>Aquino et al. (2006)</td>
</tr>
<tr>
<td>3</td>
<td>submerged</td>
<td>flat sheet and hollow fiber, 0.4 µm, 0.1 m²</td>
<td>35</td>
<td>-</td>
<td>3-48</td>
<td>460 ±20</td>
<td>&lt; 45</td>
<td>95</td>
<td>Hu and Stuckey (2006)</td>
</tr>
<tr>
<td>4.7</td>
<td>submerged</td>
<td>hollow fiber, 0.1 µm, 0.1 m²</td>
<td>11-25</td>
<td>1.6-4.5</td>
<td>3.5-5.7</td>
<td>383-849</td>
<td>-</td>
<td>96</td>
<td>Chu et al. (2005)</td>
</tr>
<tr>
<td>6</td>
<td>submerged</td>
<td>flat sheet, 0.2 µm, 0.118 m²</td>
<td>25</td>
<td>0.22-2.12</td>
<td>6-48</td>
<td>470 ±90</td>
<td>21-70</td>
<td>87-94</td>
<td>Sunaba et al. (2012)</td>
</tr>
<tr>
<td>6</td>
<td>submerged</td>
<td>flat sheet, 0.2 µm, 0.118 m²</td>
<td>25</td>
<td>0.16-1.52</td>
<td>6-48</td>
<td>360 ±110</td>
<td>16-31</td>
<td>92-94</td>
<td>Watanabe et al. (2013)</td>
</tr>
</tbody>
</table>

Note: Conf. = Configuration; Temp. = Temperature.
Table 2  Performance of previous AnMBR studies for real sewage

<table>
<thead>
<tr>
<th>Volume (L)</th>
<th>Conf.</th>
<th>Membrane (type, pore size, area)</th>
<th>Temp. (˚C)</th>
<th>OLR (kg COD/m³/day)</th>
<th>HRT (h)</th>
<th>Inf. COD (mg/L)</th>
<th>Eff. COD (mg/L)</th>
<th>COD Removal (%)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>5.8</td>
<td>submerged</td>
<td>hollow fiber, 0.4 µm, 0.19 m²</td>
<td>15-35</td>
<td>0.75-0.95</td>
<td>6</td>
<td>247±44</td>
<td>-</td>
<td>51-74</td>
<td>Gao et al. (2014a)</td>
</tr>
<tr>
<td>5</td>
<td>submerged</td>
<td>flat sheet, 0.45 µm, 0.118 m²^2</td>
<td>25-30</td>
<td>1.02 ± 0.14</td>
<td>10</td>
<td>427±59</td>
<td>60</td>
<td>86</td>
<td>Huang et al. (2013)</td>
</tr>
<tr>
<td>0.442</td>
<td>submerged</td>
<td>hollow fiber, 0.1 µm, 0.0215 m²^2</td>
<td>25</td>
<td>3.9-4.7</td>
<td>1.75-</td>
<td>152±59</td>
<td>25±8</td>
<td>84</td>
<td>Yoo et al. (2012)</td>
</tr>
<tr>
<td>60</td>
<td>submerged</td>
<td>flat sheet, 0.6 m²^2</td>
<td>30</td>
<td>-1</td>
<td>10</td>
<td>342±52</td>
<td>40</td>
<td>90</td>
<td>Lin et al. (2011)</td>
</tr>
<tr>
<td>45</td>
<td>submerged</td>
<td>flat sheet, 61 µm</td>
<td>10-30</td>
<td>-</td>
<td>8</td>
<td>298±105</td>
<td>76</td>
<td>32</td>
<td>Zhang et al. (2011)</td>
</tr>
<tr>
<td>10</td>
<td>side-stream</td>
<td>0.1 µm, 0.1 m²^2</td>
<td>-</td>
<td>0.03-0.16</td>
<td>12-4</td>
<td>38-131</td>
<td>18-37</td>
<td>72</td>
<td>Baek et al. (2010)</td>
</tr>
<tr>
<td>45</td>
<td>-</td>
<td>flat sheet, 61 µm</td>
<td>10-15</td>
<td>3.9-4.7</td>
<td>8</td>
<td>302±121</td>
<td>121±34</td>
<td>63</td>
<td>Zhang et al. (2010)</td>
</tr>
<tr>
<td>12.9</td>
<td>submerged</td>
<td>tubular, 0.64 µm, 0.98 m²^2</td>
<td>15-20</td>
<td>2.36</td>
<td>2.6</td>
<td>260±3</td>
<td>76±8</td>
<td>84</td>
<td>An et al. (2009a)</td>
</tr>
<tr>
<td>50</td>
<td>cross-flow</td>
<td>UF, 100 kDa, 1 m²</td>
<td>37</td>
<td>0.23-2</td>
<td>15-6</td>
<td>685±8</td>
<td>88±6</td>
<td>88</td>
<td>Saddoud et al. (2007)</td>
</tr>
<tr>
<td>50</td>
<td>cross-flow</td>
<td>flat sheet, 0.2 µm, 0.003 m²^2</td>
<td>35</td>
<td>-</td>
<td>16</td>
<td>350-50</td>
<td>&lt;30</td>
<td>98</td>
<td>Ko-cadagistan and Topcu (2007)</td>
</tr>
</tbody>
</table>

Note: Conf. = Configuration; Temp. = Temperature.
### Table 3  Performance of pilot scale AnMBR studies

<table>
<thead>
<tr>
<th>Volume (L)</th>
<th>Conf.</th>
<th>Membrane (type, pore size, area)</th>
<th>Temp. (˚C)</th>
<th>OLR (kg COD/m³/day)</th>
<th>HRT (h)</th>
<th>Inf. COD (mg/L)</th>
<th>Eff. COD (mg/L)</th>
<th>COD Removal (%)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>2100</td>
<td>submerged</td>
<td>hollow fiber, 0.05 µm, 30 m²</td>
<td>15-33</td>
<td>-</td>
<td>5-24</td>
<td>388 ± 95</td>
<td>-</td>
<td>-</td>
<td>Robles et al. (2013)</td>
</tr>
<tr>
<td>350</td>
<td>submerged</td>
<td>flat sheet, 0.038 µm, 3.5 m²</td>
<td>20</td>
<td>0.52-0.81</td>
<td>-</td>
<td>612 ± 97</td>
<td>64 ± 14</td>
<td>94</td>
<td>Martinez-Sosa et al. (2012)</td>
</tr>
<tr>
<td>2100</td>
<td>submerged</td>
<td>hollow fiber, 0.05 µm, 30 m²</td>
<td>33</td>
<td>0.92 ± 0.14</td>
<td>-</td>
<td>386-431</td>
<td>-</td>
<td>-</td>
<td>Giménez et al. (2012)</td>
</tr>
<tr>
<td>2100</td>
<td>submerged</td>
<td>hollow fiber, 0.05 µm, 30 m²</td>
<td>33</td>
<td>-</td>
<td>6-21</td>
<td>445 ± 95</td>
<td>77 ± 33</td>
<td>90</td>
<td>Giménez et al. (2011)</td>
</tr>
<tr>
<td>350</td>
<td>submerged</td>
<td>flat sheet, 0.038 µm, 3.5 m²</td>
<td>35</td>
<td>0.6-1.1</td>
<td>19.2</td>
<td>630 ± 82</td>
<td>&lt; 80</td>
<td>90</td>
<td>Martinez-Sosa et al. (2011a)</td>
</tr>
<tr>
<td>350</td>
<td>submerged</td>
<td>flat sheet, 0.038 µm, 3.5 m²</td>
<td>35</td>
<td>0.2-0.5</td>
<td>19.2-33.6</td>
<td>750 ± 90</td>
<td>&lt; 70</td>
<td>90</td>
<td>Martinez-Sosa et al. (2011b)</td>
</tr>
<tr>
<td>849</td>
<td>cross-flow</td>
<td>tubular, 100 kDa, 5.02 m²</td>
<td>-</td>
<td>-</td>
<td>6</td>
<td>425±138</td>
<td>33 ± 8</td>
<td>92</td>
<td>Calderón et al. (2011)</td>
</tr>
<tr>
<td>849</td>
<td>-</td>
<td>tubular, 100 kDa, 5.1 m²</td>
<td>22 ± 3</td>
<td>-</td>
<td>6</td>
<td>445±138</td>
<td>33 ± 8</td>
<td>93</td>
<td>Herrera-Robledo et al. (2011)</td>
</tr>
<tr>
<td>1500</td>
<td>dead-end</td>
<td>flat sheet, 0.038 µm, 3.5 m²</td>
<td>ambient</td>
<td>-</td>
<td>16</td>
<td>197-553</td>
<td>-</td>
<td>86</td>
<td>Martin-Garcia et al. (2011)</td>
</tr>
<tr>
<td>-</td>
<td>side-stream</td>
<td>tubular</td>
<td>ambient</td>
<td>0.3-0.9</td>
<td>5.5-10</td>
<td>58-348</td>
<td>-</td>
<td>81</td>
<td>An et al. (2009b)</td>
</tr>
<tr>
<td>180</td>
<td>cross-flow</td>
<td>hollow fiber, 0.2 µm, 4 m²</td>
<td>25</td>
<td>1.08-4.32</td>
<td>4.5-12</td>
<td>540</td>
<td>65</td>
<td>88</td>
<td>Lew et al. (2009)</td>
</tr>
</tbody>
</table>

Note: Conf. = Configuration; Temp. = Temperature.
3.1 The membrane type and performance

Concerning the type of membrane, Hu and Stuckey (2006) studied treatment performances using synthetic municipal wastewater for about 100 days, using hollow fiber and flat-sheet membranes made of polyethylene with a pore size of 0.4 µm. HRT was started at 48 h and gradually shortened to 3.0 h. It was found that both membranes resulted in similar COD removals (> 90% soluble COD at HRT of 3.0 h), but that the trans-membrane pressure (TMP) across the hollow fiber membrane was higher than the flat sheet under similar conditions. When the flux was set at 15 L m⁻²/h, the average actual flux in hollow fiber and flat-sheet were 9.74, 11.74 L m⁻²/h respectively. This shows that under the same sparging rate and reactor configuration, the flat-sheet had an advantage in maintaining higher fluxes and lower TMPs relative to the hollow fiber configuration. Giménez et al. (2011) also reported that the average COD removal was 90% in real municipal wastewater treatment for 140 days using a 2100 L reactor with a hollow fiber membrane (pore size; 0.05 µm, surface area; 30 m², volume of reactor/surface area = 0.07). In addition, Martinez-Sosa et al. (2011b) reported that the average COD removal was 90% in real municipal wastewater treatment for 100 days using a 350 L reactor with a flat-sheet membrane (pore size; 0.038 µm, surface area; 3.5 m², volume of reactor/surface area = 0.1). In addition to these membranes, Chen et al. (2014) operated a submerged anaerobic membrane bioreactor with a forward osmosis membrane (FO-AnMBR) at 25°C for the treatment of synthetic wastewater. The FO-AnMBR process exhibited greater than 96% COD removal, nearly 100% of total phosphorus and 62% of ammonia-nitrogen, respectively. These results indicate that for short operating times, the membrane type did not have a significant influence on treatment efficiency. However, it is suggested that the treatment performance might be different for different types of membrane over long operating times.

3.2 Influence of temperature

The temperature of real municipal wastewater is in low mesophilic range, so it is important to understand the treatment performance of municipal wastewater at low temperatures. In previous reports, there have been many studies using synthetic and real municipal wastewater at mesophilic (30°C) or room temperature (20-30°C) conditions. Also in experiments at pilot scale, most conditions were more than 20°C, and research at low-temperature conditions was rare. It was observed that treatment of municipal wastewater under mesophilic conditions was mostly at high organic loading rates (OLRs) and high treatment capacities. Kim et al. (2010) reported that the COD removal was 99% at 35°C, with OLR of 4.4-6.2 kg COD/m³/d. In the work of Gao et al. (2010), the COD removal was more than 96% at 30°C with OLR of 5.1 kg COD m⁻³/d. However, it should be mentioned that the influent in their studies using synthetic wastewater composed of easy degradable organic compounds. In contrast, Lin et al. (2011) using real municipal wastewater at 30°C, the COD removal was 90% at an OLR of 1.0 kg COD m⁻³/d. Kocadagistan and Topcu (2007) also reported removal efficiency of 88% at 37°C with OLR of 0.2-2.0 kg COD m⁻³/d, and the treatment performance of real municipal wastewater was low even under mesophilic condition.

In research at room temperature, there was a tendency to operate the reactor at long HRT and low OLR (Cerón-Vivas et al., 2012; Ho et al., 2007; Ho and Sung, 2009). Therefore, high treatment performance could be obtained. Yoo et al. (2012) reported that COD removal was 84% at 25°C with OLR of 3.9-4.7 kg COD m⁻³/d. Based on the results of a COD mass
balance, the recovery of methane gas was only 22%, and 38% of the methane was dissolved into the water and lost with the discharged effluent. This is because the solubility of methane depends on the temperature according to Henry’s law. It indicated that 60% of the energy from the removed COD was collected as methane gas. In a report by Ho and Sung (2009), the COD removal was 94% with HRT up to 12, 8.0 and 6.0 h (OLR was 1.0, 1.5, 2.0 kg COD m$^{-3}$/d respectively) at 25°C. But with HRT decreased, soluble COD accumulated in the reactor. It has been suggested that both biological degradation, and the physical separation of organics by the membrane, contribute to the high COD removal efficiency of AnMBR. In the COD mass balance, the recovery of methane gas decreased from 48% to 35% as HRT became shorter. In the study by Sunaba et al. (2012), HRT was selected of 48 h, 24 h, 12 h and 6.0 h (OLR was 0.2-2.1 kg COD m$^{-3}$/d), and gradually shortened while the temperature was maintained at 25°C with synthetic wastewater. COD removals at HRTs of 12 h and 6.0 h were 94% and 87% respectively. The treatment performance tended to become worse at 6.0 h HRT. From the COD mass balance at HRT of 12 h, methane gas recovery was 72%. In addition, Watanabe et al. (2013) studied operations at HRT of 48 h, 24 h, 12 h and 6.0 h (OLR was 0.2-1.5 kg COD m$^{-3}$/d), gradually shortened at 25°C using synthetic wastewater. COD removal and methane gas recovery were 92-94% and 65% at HRT 12 h, respectively.

In research at low temperatures, Gao et al. (2014a) investigated the performance of a novel integrated anaerobic fluidized-bed membrane bioreactor (IAFMBR) for treating practical domestic wastewater as the temperature was reduced in steps from 35, 25, to 15°C. The COD removal rate was 74%, 67%, 51% at 35, 25, 15°C, respectively. And about 53%, 55%, 39% of COD$_{influent}$ was removed as methane respectively. As compared with a mesophilic temperature, a low temperature can accelerate membrane biofouling. Proteins were the dominant cause of membrane fouling at low temperature, and membrane fouling can be mitigated by granular active carbon (GAC) through protein absorption. Smith et al. (2013) operated an AnMBR treating synthetic municipal wastewater at 15°C and HRT of 24-16 h (OLR was 0.4-0.7 kg COD m$^{-3}$/d), and obtained a COD removal of 92%. It was suggested that because of the low OLR, the COD removal was high. In the work of Zhang et al. (2010) using real municipal wastewater, operated at 10-15°C and HRT 8.0 h (OLR was 3.9-4.7 kg COD m$^{-3}$/d), the COD removal was 63%. An et al. (2009a) reported a study operated at 15-20°C and HRT of 2.6 h (OLR was 2.4 kg COD m$^{-3}$/d) using real municipal wastewater and the COD removal was 84%.

Some studies have been reported on operations at ambient temperature. Zhang et al. (2011) treated real municipal wastewater at 10-30°C and HRT of 8.0 h. In this study, COD removal was 75% and gas production and COD removal decreased when the temperature became low.

Research on effects of temperature thus indicates that good water quality can be obtained using synthetic wastewater operated at HRT of more than 16 h in low temperature conditions (lower than 15°C). On the other hand, it appears that high COD removal efficiency can be achieved at HRT of 6.0 h at above 25°C temperature condition seven using real municipal wastewater. The reason why treatment performance varies with temperature may be related to the influence of microorganisms. It is suggested that the species of microorganisms which have high activity at low temperature are limited. If these psychrophilic bacteria were to increase, stable treatment performance might be obtained at low temperature or short HRT condition.
3.3 The influence of HRT and SRT

It has been reported that HRT and SRT could greatly influence the treatment efficiency. In the context of municipal wastewater treatment, a short HRT is desirable to reduce AnMBR volume size and the overall footprint of the system, whereas a long SRT may be required to achieve favorable treatment performance at low temperature (O’Flaherty et al., 2006). However, when increasing the SRT, while keeping the HRT constant, the suspended biomass concentration increases and potentially leads to a decrease in the permeate flux (Bérubé et al., 2006; Huang et al., 2011; Liao et al., 2006). Furthermore, increasing the SRT may result in higher soluble microbial production (SMP) and extracellular polysaccharide (EPS) production, which play a role in membrane fouling (Huang et al., 2011). Therefore, a tradeoff could exist between controlling HRT and SRT for membrane fouling mitigation and obtaining the necessary treatment performance. The influence of AnMBR treatment performance on HRT has been evaluated in various studies. Hu and Stuckey (2006) observed a marginal decrease in COD removal when they shortened the HRT from 48 h to 3.0 h in stages at 35°C to treat dilute wastewaters. And Chu et al. (2005) did not observe a correlation between treatment performance and HRT at HRT of 3.5 h, 4.6 h, 5.7 h with synthetic wastewater (Chu et al., 2005). Several other studies also concluded that HRT had little effect on permeate quality with synthetic and real wastewater (Baek et al., 2010; Ho and Sung, 2009; Lew et al., 2009). However, another study in which HRT was decreased from 12 h to 8.0 h and then 4.0 h observed an increase in permeate COD at the shortest HRT, while there was no significant difference between the permeate COD values obtained at HRTs of 12 h and 8.0 h with synthetic wastewater (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. Huang et al. (2011) noted that combining a short HRT with a long SRT leads to increases in suspended biomass concentrations, which would accelerate membrane fouling rates with synthetic wastewater.

Membrane separation enables complete retention of biomass and control of SRT. Because of this, SRT is an easily controllable operational parameter affecting both treatment performance and membrane fouling. In the work of Baek et al. (2010), the SRT was shortened from 217 to 40 days through biomass discharge using real municipal wastewater. 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.

3.4 Pilot scale studies

Recently, the performance of pilot-scale AnMBR in treating municipal wastewater has been assessed by Calderón et al. (2011), Gimenez et al. (2011, 2012), Herrera-Robledo et al. (2011) and Martinez-Sosa et al. (2011a, 2011b, 2012). Martinez-Sosa et al. (2011a) operated an anaerobic membrane bioreactor (AnSMBR) at pilot-scale treating a mixture composed of municipal wastewater and glucose under mesophilic temperature conditions. The performance of the AnSMBR was evaluated against the gas sparging velocities (GSV) which were used to control fouling. The GSV does not play a central role in controlling fouling under critical conditions. Nevertheless, GSV becomes more important as
soon as the critical flux is exceeded. By increasing the GSV to 62 m/h the fouling rate dropped from 14 to 8.4 m bar/day. And COD removal efficiencies were always close to 90%. Martinez-Sosa et al. (2011b) also operated a pilot-scale AnMBR with a total volume of 350 L for municipal wastewater treatment. The pilot reactor was operated for 100 days over which the temperature was reduced from 35 to 28°C and then to 20°C. Membrane fouling was controlled using biogas sparging, membrane relaxation, and periodic backwashing. The reactor was operated at HRT of 19.2 h and SRT of 680 days. Regardless of variations in temperature, COD removals remained approximately 90%. In addition, Martinez-Sosa et al. (2011b) operated the reactor under the same conditions at 20°C and obtained a COD removal of 94%. Nevertheless, the methane yield was always lower than the theoretical value, which indicates that the organic compounds were not completely degraded, but physically retained by the membrane in the reactor. Gimenez et al. (2011) operated a pilot scale facility fed with pre-treated municipal wastewater at a SRT of 70 days, an HRT ranging from 21 h to 6.0 h, and a temperature of 33°C. The total liquid volume of the system was 2,500 L. The COD concentration in the influent averaged 445 ± 95 mg/L. Sulfate concentrations were particularly high, averaging 297 ± 54 mg/L, an order of magnitude higher than average sulfate concentrations reported for municipal wastewater. The COD removal averaged 90% during stable operations. The high levels of sulfate in the influent greatly affected biogas production as methanogens and sulfate reducers compete for substrates, and about 45% of the influent COD was consumed for sulfate reduction rather than for methanogenesis. Calderón et al. (2011) operated a pilot-scale membrane-coupled upflow anaerobic sludge blanket bioreactor (UASB) at HRT of 6.0 h. The volume of reactor was 849 L, and the COD removal was 92%. Calderón et al. (2011) also investigated the biofilm formed by membrane fouling. Microorganisms were detected on the membrane even after chemical cleaning and backflushing, indicating that the formed biofilm was not completely removed by chemical cleaning and backflushing.

4. COMPARISON OF THE TRADITIONAL ANAEROBIC PROCESS WITH AnMBR

As mentioned above, the development of the UASB process treating municipal wastewater started from the 1980s. Subsequently, the anaerobic buffer reactor (ABR) and expanded granular sludge bed (EGSB) reactors have been developed. Table 4 summarizes the history of municipal wastewater treatment using anaerobic digestion technology. Fig. 2 provides the COD removal efficiency of the conventional anaerobic process and AnMBR under different temperatures. For a conventional anaerobic process, the reported COD removal efficiency is around 70% at temperatures from 10-15°C. A high removal efficiency of 90% could be obtained at 35°C. However, COD removal decreases to 70-85% at 20°C and to lower than 70% at temperature from 10-15°C. These results indicate that temperature significantly affects the COD removal efficiency. For AnMBR processes operated at temperatures higher than 20°C, COD removal is not significantly influenced by temperature. In the AnMBR process, the organic pollutants are degraded by microorganisms and partially separated by the membrane. More importantly, microorganisms are retained inside the reactor. These properties provide the advantage of maintaining microorganism activity at low temperatures.
Table 4  Performance of previous studies of sewage treatment in conventional anaerobic treatment

<table>
<thead>
<tr>
<th>Reactor type</th>
<th>Temp. (°C)</th>
<th>OLR (kg COD/m³/day)</th>
<th>HRT (h)</th>
<th>Inf. COD (mg/L)</th>
<th>COD Removal (%)</th>
<th>Methane Conversion (%)</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>EGSB</td>
<td>20</td>
<td>3.1-11</td>
<td>1.5-6</td>
<td>600-800</td>
<td>82</td>
<td>-</td>
<td>Tsushima et al. (2010)</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>4-5.6</td>
<td>2.9-4.2</td>
<td>73</td>
<td>73</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td></td>
<td>10</td>
<td>3-6</td>
<td>2.9-5.9</td>
<td>69</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>2.8-4.3</td>
<td>3.9-6</td>
<td>64</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>UASB</td>
<td>25</td>
<td>0.7-3</td>
<td>4.7</td>
<td>115-595</td>
<td>64-70</td>
<td>60</td>
<td>Uemura and Harada (2000)</td>
</tr>
<tr>
<td></td>
<td>19</td>
<td></td>
<td></td>
<td></td>
<td>72</td>
<td>52</td>
<td></td>
</tr>
<tr>
<td></td>
<td>13</td>
<td></td>
<td></td>
<td></td>
<td>64</td>
<td>35</td>
<td></td>
</tr>
<tr>
<td>ABR</td>
<td>23-31</td>
<td>1.2</td>
<td>10</td>
<td>500</td>
<td>&gt; 90</td>
<td>61</td>
<td>Gopala Krishna et al. (2009)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1.5</td>
<td>8</td>
<td></td>
<td></td>
<td>64</td>
<td></td>
</tr>
<tr>
<td>ASBR</td>
<td>15-25</td>
<td>0.8-1.2</td>
<td>12</td>
<td>500</td>
<td>80-85</td>
<td>-</td>
<td>Ndon and Dague (1997b)</td>
</tr>
<tr>
<td></td>
<td>0.6-0.9</td>
<td>16</td>
<td></td>
<td></td>
<td>81-85</td>
<td></td>
<td></td>
</tr>
<tr>
<td>ASBR</td>
<td>30</td>
<td>-</td>
<td>8</td>
<td>500</td>
<td>80-88</td>
<td>-</td>
<td>Rodrigues et al. (2003)</td>
</tr>
<tr>
<td>HUSB + UASB</td>
<td>14-21</td>
<td>0.5-3.2</td>
<td>2.8-5.7</td>
<td>118-401</td>
<td>64</td>
<td>36</td>
<td>Alvarez et al. (2008)</td>
</tr>
<tr>
<td>UASB</td>
<td>0.2-0.7</td>
<td>6.5-13.9</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MFBR</td>
<td>30-35</td>
<td>-</td>
<td>8-48</td>
<td>941</td>
<td>60</td>
<td>-</td>
<td>Reyes et al. (1999)</td>
</tr>
<tr>
<td>ASBR</td>
<td>25</td>
<td>0.2-1.2</td>
<td>12-48</td>
<td>500</td>
<td>84-92</td>
<td>-</td>
<td>Ndon et al. (1997a)</td>
</tr>
<tr>
<td></td>
<td>20</td>
<td></td>
<td></td>
<td></td>
<td>81-91</td>
<td></td>
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</tr>
<tr>
<td></td>
<td>15</td>
<td></td>
<td></td>
<td></td>
<td>80-90</td>
<td></td>
<td></td>
</tr>
<tr>
<td>UASB</td>
<td>20-35</td>
<td>1.2-4</td>
<td>3-6</td>
<td>500</td>
<td>83-88</td>
<td>-</td>
<td>Singh et al. (1996)</td>
</tr>
<tr>
<td>ABR</td>
<td>35</td>
<td>-</td>
<td>10</td>
<td>500</td>
<td>95</td>
<td>-</td>
<td>Langenhoff and Stuckey (2000)</td>
</tr>
<tr>
<td></td>
<td>20</td>
<td></td>
<td></td>
<td></td>
<td>70</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>10</td>
<td></td>
<td></td>
<td></td>
<td>60</td>
<td></td>
<td></td>
</tr>
<tr>
<td>UASB</td>
<td>12-20</td>
<td>0.6-3.5</td>
<td>7-8</td>
<td>190-1180</td>
<td>30-75</td>
<td>-</td>
<td>De Man et al. (1986)</td>
</tr>
<tr>
<td>UASB</td>
<td>12-18</td>
<td>0.62</td>
<td>18</td>
<td>465</td>
<td>73</td>
<td>-</td>
<td>Monroy et al. (1988)</td>
</tr>
<tr>
<td>AF + AH</td>
<td>13</td>
<td>0.9-1.1</td>
<td>12</td>
<td>460-530</td>
<td>70</td>
<td>-</td>
<td>Gopala Krishna et al. (2008)</td>
</tr>
<tr>
<td>UASB</td>
<td>25</td>
<td>1-3</td>
<td>-</td>
<td>500</td>
<td>-</td>
<td>57</td>
<td>Harada et al. (1994)</td>
</tr>
<tr>
<td>UASB</td>
<td>20</td>
<td>0.15-1.2</td>
<td>10-48</td>
<td>350-500</td>
<td>60-75</td>
<td>35-45</td>
<td>Singh and Viraraghavan (1998)</td>
</tr>
</tbody>
</table>

The effects of temperature on UASB performance have been widely investigated (Alvarez et al., 2008; De Man et al., 1986; Harada et al., 1994; Langenhoff and Stuckey, 2000; Monroy et al., 1988; Singh et al., 1996; Singh and Viraraghavan, 1998; Uemura and Harada, 2000). Uemura and Harada (2000) reported that under 25°C, 19°C and 13°C (OLR of 0.7-3.0 kg COD m$^{-3}$/day), the COD removal efficiency was 70%, 72% and 64% respectively, and the treatment performance maintained at a similar level at 25°C and 20°C. De Man et al. (1986) reported that at a temperature of 12-20°C and HRT of 7.0-8.0 h, COD removal varied between 30% and 75%. Low temperatures result in sharply decreased reactor performance. Monroy et al. (1988) obtained a COD removal of 73% at 12-18°C, HRT of 18 hours and OLR of 0.6 kg COD m$^{-3}$/day. The effects of low temperature on EGSB reactors have also been investigated. Tsushima et al. (2010) carried out a series of experiments at 20°C (OLR 3.1-11 kg COD m$^{-3}$/day), 15°C (OLR 4.0-5.6 kg COD m$^{-3}$/day), 10°C (OLR 3.0-6.0 kg COD m$^{-3}$/day) and 5°C (OLR 2.8-4.3 kg COD m$^{-3}$/day). The COD removal efficiency in order of decreasing temperature was 82%, 73%, 69% and 64% respectively.

Ndon and Dague (1997a) evaluated the performance of an anaerobic sequence batch reactor (ASBR) treating low strength municipal wastewater at temperatures of 15-25°C and OLR of 0.2-1.2 kg COD m$^{-3}$/day. From these
experiments, a COD removal efficiency of 80-90% was obtained at 15°C. Langenhoff and Stuckey (2000) evaluated the effects of temperature using AnABR. At 35, 20 and 10°C, the COD removal rate was 95%, 70% and 60% respectively. The process performance can be evaluated on the methane conversion ratio as well. Uemura and Harada (2000) reported that at 25, 19 and 13°C, the methane conversion ratio was 60%, 52% and 35% at HRT of 4.7 hours using ABR. Harada et al. (1994) obtained a methane conversion efficiency of 57% at 25°C and OLR of 1.0-3.0 kg COD m³/day. A similar result (35-45%, at 20°C and OLR of 0.2-1.2 kg COD m³/day) was reported by Singh and Viraraghava (1998). Gopala Krishna et al. (2009) evaluated the ABR performance at 23-31°C and HRT of 10 and 8.0 h. The COD removal efficiency was around 90% and methane recovery ratio was 61% and 64%. For the AnMBR process, Sunaba et al. (2012) obtained a methane conversion ratio of 72% at 25°C and OLR of 0.2-2.1 kg COD m³/day. Watanabe et al. (2013) reported a 65% methane recovery ratio at 25°C and OLR of 0.2-1.5 kg COD m³/day. Fig. 3 provides the COD mass balance of previous studies including Gopala Krishna et al. (2009) (A. 23-31°C, OLR 1.2 kg COD m³/day; B. 23-31°C, OLR 1.5 kg COD m³/day), Sunaba et al. (2012) (C. 25°C, OLR 1.1 kg COD m³/day) and Watanabe et al. (2013) (D. 25°C, OLR 0.8 kg COD m³/day) with synthetic wastewater. In this figure, the COD removal efficiency and methane conversion ratio was at high levels in the AnMBR process. The recirculation of biogas from the headspace created mixing in the reactor, cleaned the membrane and facilitated the release of dissolved methane through stripping. The production of dissolved methane in the AnMBR process was less than that in conventional anaerobic processes. Therefore, AnMBR can not only provide high quality treated water but also achieve a high efficiency for the recovery of methane.

5. CHALLENGES IN AnMBR DEVELOPMENT

The AnMBR can be more efficient than the conventional process; however, some problems in the AnMBR process have been reported recently. This paper summarizes the challenges to improve the AnMBR process, including the control of membrane fouling, the recovery of dissolved methane and the removal of nitrogen and phosphorus from the AnMBR system.

5.1 Control of membrane fouling

Membrane cleaning and exchange may increase the cost of an AnMBR facility and limit the application of this technology. Consequently, much research about AnMBR has focused on the control of membrane fouling. The absorption and attachment of organic and inorganic pollutants onto the membrane surface and deep membrane structure results in a decline in flux and an increase in TMP. Consequently, membrane fouling may shorten the lifetime of a membrane. Fig. 4 described the factors contributing to membrane fouling. Within the anaerobic system, the mechanism of membrane fouling is more complex and difficult to elucidate than aerobic system. Membrane fouling can greatly increase the energy consumption of reactor operation. It is thus important to prevent and delay the formation of membrane fouling in order to minimize energy consumption.
Hu and Stuckey (2007) tested the effects of adding powdered activated carbon (PAC) and granular activated carbon (GAC) to a membrane reactor on the prevention of membrane fouling with gas sparging. Compared with the control reactor, the TMP of the PAC and GAC in the AnMBR was small and flux increased. Kim et al. (2010) added GAC into an AFBR and controlled membrane fouling using recycling liquid instead of biogas sparging. Gao et al. (2014b) also operated an integrated anaerobic fluidized-bed membrane bioreactor (IAFMBR) system with GAC as carrier, which was developed to treat synthetic wastewater with energy recovery. GAC supplementation helped in the control of the membrane fouling and long-term operational period prior to back flushing and clean-up. As the higher dosage of GAC was applied, more protein in cake layer was absorbed by the GAC, and the reduction of protein in cake layer improved the membrane filtration performance. Positive results in terms of reducing energy consumption and preventing membrane fouling were obtained.

Liquid recirculation and/or gas sparging are common ways to provide shear over the membrane surface in order to remove foulants or to restrict their interaction with the membrane. Xie et al. (2010) reported that the membrane flux of a submerged AnMBR
increased and the fouling rate decreased when the biogas sparging rate was increased from 0.3 to 0.75 L/min in Kraft evaporator condensate treatment. Zhang et al. (2007) suggested that maintaining high cross-flow velocities was critical to limit the fouling rate in AnMBRs (this is not for treatment of municipal wastewater). However, even at high cross-flow velocities, the membrane resistance continues to increase. High shear rates may stimulate the break-down of microbial flocs and increase the cake layer resistance due to the deposition of small particles on the membrane surface or inside the membrane pores.

Cerón-Vivas et al. (2012) reported the effect of intermittent filtration combined with gas bubbling on membrane fouling in a submerged anaerobic membrane bioreactor (SAAnMBR) treating synthetic municipal wastewater. When the gas bubbling was introduced, filtration runs were more than ten times longer than without it. And an early blockage in intermittent filtration tests without gas bubbling was related to the presence of the carbohydrate fraction of EPS and SMP. The better performance obtained with nitrogen bubbling was caused by the increased internal mixing, which induced a greater removal of organic matter in the reactor. In Kola et al. (2014) research, transverse vibration of submerged hollow fibers is explored for enhancing the filtration of anaerobic bioreactor effluents where gas sparging is often undesirable. The critical flux value increased significantly with the aid of membrane vibration. Even at high mixed liquid suspended solid concentrations, the vibratory system was still able to significantly reduce fouling. By appropriately coupling periodical backwash/relaxation with vibrational filtration, the membrane performance was further improved.

5.2 Recovery of dissolved methane

In the anaerobic process, energy recovery can be obtained in the form of methane from municipal wastewater. Capturing methane is also important to mitigate direct greenhouse gas emissions, as methane has a global warming potential 25 times that of carbon dioxide (Solomon et al., 2007). Therefore, emission of methane into the atmosphere from an AnMBR effluent should be avoided. However, methane can dissolve in water as a natural property and it can be difficult to collect dissolved methane. Fig. 5 represents the saturated dissolved methane in liquor expressed as COD concentration at different temperatures for different methane concentrations. The dissolved methane concentration can be calculated using Henry’s law, shown in equation (1).

\[ X_{CH4} = \frac{P_{CH4}}{K(T)} \]  

\[ X_{CH4} \]: Molar fraction of methane in pure water  

\[ P_{CH4} \]: Partial pressure of methane (atm)  

\[ K(T) \]: Henry’s constant (atm)

The Henry constant (K) at different temperatures can be calculated using equation (2).

\[ K(T) = 10^{\frac{(-675.74/K(T))+6.88}{6.88}} \]  

1 liter of water contains 55.6 mole \[ \frac{1000 \text{ g}}{18 \text{ g/mol}} \], the mole faction of \( CH_4 \) is equal to

\[ X_{CH4} = \frac{\text{mole gas (methane)}}{\text{mole gas (methane)} + \text{mole water}} \]  

\[ = \frac{Y_{CH4}}{Y_{CH4} + 55.6 \text{ mol L}^{-1}} \]  

\[ Y_{CH4} \]: methane mole concentration (mol/L)

The ratio of methane in the gas is \( A \), temperature as \( T \) (K), the mole concentration of dissolved methane as \( Y_{CH4} \). By equation (1) and (3),

\[ \frac{1 \text{ atm}*A}{K(T)} = X_{CH4} = \frac{Y_{CH4}}{Y_{CH4} + 55.6} \]
\[ A(Y_{CH4} + 55.6) = K(T) \cdot Y_{CH4} \]

\[(K(T) - A)Y_{CH4} = 55.6 \cdot A\]

\[ Y_{CH4} = \frac{55.6 \cdot A}{K(T) - A} \]

In the above equations, the number of moles of dissolved methane (A) in a liter of water is much less than the number of moles of water. So A can be ignored as a small value. The equation can be simplified:

\[ Y_{CH4} = \frac{55.6 \cdot A}{K(T)} \text{(mol/L)} \]

The mass concentration of methane \( Z_{CH4} \) (mg/L) can be calculated using equation (4).

\[ Z_{CH4} = 16 \cdot 10^3 \cdot Y_{CH4} = \frac{8.896 \cdot 10^5 \cdot A}{K(T)} \text{(mg/L)} \] (4)

16 g/mol: Molecular weight of methane.

In addition, methane density is 0.717 kg/m\(^3\). Thus when 1.0 g COD is degraded, 0.35 L methane is produced based on stoichiometry. From these values, 1.0 g of methane produced can be calculated as equivalent to a reduction of 4.0 g of COD. From this figure, the dissolved methane concentration at 10\(^\circ\)C is 1.5 times lower than that at 35\(^\circ\)C. Low temperatures significantly increase the methane loss in discharged water. Since the treatment of municipal wastewater using AnMBR is required to be operated at low temperatures, it is important to improve the recovery of dissolved methane, not only to recover energy but also to prevent greenhouse gas emissions.

Many researchers have investigated methane dissolution and determined this quantitatively in AnMBR processes (Bandara et al., 2011; Hu and Stuckey, 2006; Kim et al., 2010; Smith et al., 2011). Kim et al. (2010) reported that about 30% of the methane from degradation of organic materials was dissolved in the discharged water at 35\(^\circ\)C. Low temperatures may increase this percentage. Smith et al. (2011) reported that at 15\(^\circ\)C, 50% of methane dissolved in water. Bandara et al. (2011) investigated the recovery and quantity of dissolved methane from AnMBR permeate by degassing the membrane. At 35, 25 and 15\(^\circ\)C, untreated dissolved methane concentration was 63, 82 and 104 mg COD/L respectively. Low temperature facilitates the dissolution of methane in water. These concentrations were higher than the calculated values using Henry’s law and indicate that the permeate was in a super-saturated state. This phenomenon was identified in other anaerobic processes treating low strength municipal wastewater as well (Hartley and Lant, 2006; Singh et al., 1996).

Some methods have been proposed to remove or recover dissolved methane from AnMBR processes including degassing the membrane and a combined process of aerobic stripping and down-flow hanging sponge (DHS) reactor (Bandara et al., 2011; Hatamoto et al., 2010; McCarty et al., 2011). Among these methods, a degassing membrane can recover methane easily. Bandara et al. (2011) used a degassing membrane to recover methane and obtained favorable results at low temperature. However, the operation of degassing a membrane needs a large amount of energy, which is nearly 300 times of that contained in the recovered methane. Aerobic stripping has been used as a general method to reduce the release of dissolved methane in leachate (McCarty et al., 2011). It is said that this technology is of low energy consumption, but it is unstable in removing dissolved methane from the discharged effluent. And so far, no real application has been reported for the recovery of evaporated methane. Hatamoto et al. (2010) used the DHS process to oxidize the dissolved methane and obtained a removal efficiency of 95%. Since methane was biologically oxidized into CO\(_2\) in the DHS process, it could not be used as energy. In addition, there are concerns over the energy consumption in the operation of DHS reactors and the energy loss of biological methane oxidation.
Figure 5 Saturation dissolved methane concentration (COD based) at each temperature for different methane concentrations

CONCLUSIONS

The AnMBR process can produce clean water to the same quality as that of aerobic processes. The AnMBR has been attracting more and more attention due to the potential advantages of facilitating methane gas utilization, reducing greenhouse gas emissions and generating bio-energy. The progress of AnMBR technology makes this technology attractive. In recent years, pilot-scale research of AnMBR has been reported. However, there are still some problems which need to be overcome in the future, including (1) management of steady-state anaerobic systems at low temperature; (2) mechanisms of membrane fouling and control; and (3) recovery of dissolved methane from low temperature AnMBR systems.

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