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Impact of reactor configurations on the performance of a granular anaerobic membrane bioreactor for municipal wastewater treatment

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This study compared overall performance of an external granular anaerobic membrane bioreactor and a submerged granular anaerobic membrane bioreactor (EG-AnMBR and SG-AnMBR, respectively), to determine which type of G-AnMBRs is more preferred for municipal wastewater treatment. Both systems presented similar COD removal efficiencies (over 91%) and methane yield of 160 mL $CH_{4\ (STP)}$ (g COD $_{removed}$)$^{-1}$ although volatile fatty acids (VFA) accumulation was found in the SG-AnMBR. Membrane direct incorporation into the SG-AnMBR significantly affected the concentration and properties of microbial products (e.g. soluble microbial products (SMP) and extracellular polymeric substances (EPS)) in the cake layer, mixed liquor and granular sludge, as well as granular sludge size and settleability. The EG-AnMBR demonstrated less SMP and EPS in the mixed liquor and cake layer, which might reduce the cake layer resistance and lower the fouling rate. Liquid chromatography-organic carbon detection (LC-OCD) analysis of foulant revealed that biopolymers along with low molecular weight neutrals and acids and building blocks were responsible for higher fouling propensity in the SG-AnMBR. It is evident that compared to the SG-AnMBR, the EG-AnMBR serves as a better G-AnMBR configuration for municipal wastewater treatment due to less fouling propensity and superior granule quality.

1. Introduction

Anaerobic membrane bioreactors (AnMBRs) have gained particular interest for municipal wastewater treatment in recent years due to its competitive advantages (i.e., bioenergy production, quality effluent, low sludge disposal, high loading capacity, nutrient recovery, footprint efficiency, lower energy requirements, and decentralized operation) over the conventional anaerobic systems and aerobic MBRs (Mnif et al., 2012; Galib et al., 2016; Pretel et al., 2016). However, membrane fouling has remained as one of the most challenging issues impeding the progress of AnMBRs (Sanguanpak et al., 2015; Huang et al., 2012; Saleem et al., 2016), especially with high biomass concentration in widely used conventional AnMBRs (C-AnMBRs).

In view of this concern, many researchers have devoted their efforts into developing various AnMBR configurations such as vibrating AnMBRs (V-AnMBRs) (Kola et al., 2014) Gas-lifting AnMBRs (Gl-AnMBRs) (Gimenez et al., 2012), anaerobic bio-entrapped membrane bioreactors (AnBEMRs) (Ng et al., 2014), anaerobic dynamic membrane Bioreactor (AnDMBRs) (Saleem et al., 2016) and anaerobic membrane sponge bioreactors (AnMSBRs) (Kim et al., 2014) for sustainable fouling mitigation strategies. Granular anaerobic membrane bioreactor (G-AnMBR), a hybrid anaerobic biotechnology that incorporates the granular technology with membrane based separation, has offered a promising approach to the C-AnMBR in terms of fouling mitigation (Chen et al., 2016a). Unlike C-AnMBRs predominantly in the form of completely stirred tank reactor (CSTR) configuration, biomass retention is achieved by the spontaneous formation of granular sludge in G-AnMBRs without the need for mechanical mixing. The anaerobic granular bed is usually featured with total biomass concentrations ranging from 20 to 40 g L^{-1}. All the biological

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reactions occurred within the dense sludge bed at the bottom of the upflow anaerobic granular bioreactor (UAGB). When combining UAGB with membrane filtration, the entrapment of most particulate organics by adsorption and biodegradation in the granular sludge bed allowed membrane module only being challenged by the supernatant of the granular sludge bed, thus reducing the organic loading to the membrane (Martin-Garcia et al., 2011; Ozgun et al., 2015). Hence, less apparent formation of dense cake layer and its consolidation occurred as compared to C-AnMBR (Ozgun et al., 2015). Martin-Garcia et al. (2013) confirmed the lower fouling potential in the G-AnMBR as compared to the C-AnMBR, due to the reduced solid and colloidal load (by a factor of 10 and 3) to the membrane. Furthermore, the critical flux test also revealed the G-AnMBR required much lower gas sparging intensity, resulting in lower energy demand for fouling control. The filtration performance of three MBRs (i.e. C-AnMBR, G-AnMBR and conventional aerobic MBR) for domestic wastewater treatment was also investigated (Martin-Garcia et al., 2011). Comparing to the C-AnMBR, it was found that the G-AnMBR was characterized with 50% lower mixed liquor suspended solids (MLSS) concentration and soluble microbial products (SMP), contributing to lower fouling rate than that of the C-AnMBR.

The predominated configuration of current G-AnMBR operation for municipal wastewater treatment was found as the external G-AnMBR (EG-AnMBR) where membrane filtration was applied as a polishing stage for UAGB effluent (Herrera-Robledo et al., 2010, 2011; Salazar-Pelaez et al., 2011a; Salazar-Pelaez et al., 2011b). In this case, membrane tank is usually situated after the main biological treatment process (i.e. UAGB) and the concentrate streams are not recycled back to the main bioreactor. The main advantages include undisturbed hydraulics in the UAGB, and the ease of operation and membrane cleaning. Nevertheless, Ozgun et al. (2015) elucidated that the EG-AnMBR may be encountered with the progressive increase in the SS loading on the membrane unit. Very few researchers employed submerged membrane in the SG-AnMBR to provide nearly absolute biomass retention and allow for operation at nearly infinite SRTs (Chu et al., 2005). Membrane, in this case, not only acts as a physical barrier for active biomass retention, but also promotes a general cultural adaptation to the prevailing organic loading conditions in the SG-AnMBR (Liu et al., 2013). On the other hand, Liu et al. (2012) pointed out membrane filtration could exacerbate sludge bioflocculation in the SG-AnMBR and induced greater cake resistance, resulting in more serious fouling. To date, no references have been found to compare the two mainstream G-AnMBRs for the treatment of municipal wastewater.

The objective of this study is, therefore, to determine which type of G-AnMBR configurations is favourable for municipal wastewater treatment. To this aim, a direct comparison of external and submerged membrane operation in G-AnMBR (namely EG-AnMBR and SG-AnMBR) was conducted. The comprehensive evaluation of the two G-AnMBRs included the investigation of treatment efficiencies, granules properties (e.g. particle size, settling velocity, extracellular polymeric substances (EPS), etc.), membrane fouling behaviour (transmembrane pressure (TMP), potential foulants, fouling resistance analysis), and renewable energy recovery (methane yield).

2. Materials and methods

2.1. Synthetic wastewater

Both EG-AnMBR and SG-AnMBR were fed with synthetic wastewater simulating the domestic wastewater just after primary treatment. The synthetic wastewater is comprised of organics and macronutrients, and trace nutrients. The synthetic wastewater was characterized by dissolved organic carbon (DOC) of 100–120 mg L^{-1}, chemical oxygen demand (COD) of 320–360 mg L^{-1}, ammonia nitrogen (NH_4^+-N) of 5.2–6.5 mg L^{-1}, nitrite nitrogen (NO_2^--N) of 0–0.03 mg L^{-1}, nitrate nitrogen (NO_3^--N) of 0.2–0.7 mg L^{-1} and orthophosphate (PO_4^{3-}-P) of 3.0–3.5 mg L^{-1} (COD: N: P = 100: 2: 1). NaOH or $NaHCO_3$ was used to adjust pH to 7.

2.2. Experimental setup and operating conditions

Two G-AnMBRs with equal working volume of 4 L, namely EG-AnMBR and SG-AnMBR were operated in parallel at 20 ± 0.5 °C in the Environmental Engineering lab at the University of Technology, Sydney. Both G-AnMBRs were fed with identical inoculated anaerobic sludge with similar initial sludge concentration (21.48 ± 0.98 g L^{-1} for the EG-AnMBR, 21.41 ± 1.12 g L^{-1} for the SG-AnMBR) at the beginning of the experiments. For the EG-AnMBR, a polyvinylidence (PVDF) hollow fiber membrane with a pore size of 0.22 μm and surface area of 0.06 m^2 was immersed in the subsequent membrane tank located after the UAGB. Membrane tank was fed with the UAGB effluent and a suction pump was operated with an intermittent suction cycle of 8 min on and 2 min off to acquire permeate from the membrane module. While in the SG-AnMBR, an identical membrane module was directly immersed into the mixed liquor at the settling zone of the UAGB. Both systems were operated at a constant filtration rate of 7 L m^{-2} h^{-1}, hydraulic retention time of 12 h, and upflow velocity of 0.7 m h^{-1}. The membrane fouling was indicated by development of the normalized TMP, which was recorded by a pressure transmitter. When TMP reached 30 kPa, G-AnMBR operation was terminated.

2.3. Analytical methods

DOC of the influent and effluent was measured using a DOC analyzer (Analytikjena Multi N/C 2000). The analysis of COD was carried out according to Standard Methods (APHA, 1999). NH_4^+-N, NO_2^--N, NO_3^--N and PO_4^{3-}-P were measured by spectrophotometric method using Spectroquant Cell Test (NOVA 60, Merck). The pH and temperature of the reactor were measured everyday using pH meter (Hach Company, model no. HQ40d).

The granular sludge was collected at 3 sampling port at different heights of the UAGB (Port 1: 20 cm, Port 2: 40 cm and Port 3: 60 cm height from the bottom). Mixed liquor suspended solids (MLSS), mixed liquor volatile suspended solids (MLVSS), sludge volume index (SVI), settling velocity and zeta potential were conducted based on the methods described in Standard Methods (APHA, 1999). Particle size distribution (PSD) of granule sludge samples was determined using the laser particle size analysis system (Mastersizer Series 2000 supplied by Malvern Instruments Ltd., UK) with a detection range of 0.02–2000 mm. The scattered light was detected by means of a detector that converted the signal to a size distribution based on volume. Each sample was measured three times with a standard deviation of 0.1–4.5%. D (0.1) (i.e. 10% of the volume distribution was below this value) was used to describe the colloidal and fine particle fractions. The sludge granules were examined by Olympus System Microscope Model BX41 (Olympus, Japan) and the images were captured and analyzed using Image-Pro Plus software.

Based on the resistance-in-series model, fouling resistance of the G-AnMBR was determined after G-AnMBR experiments by using measurement protocol proposed by Deng et al. (2015) and applying Eqs. (1) and (2) (Choo and Lee, 1996):

$$J = \Delta P/\mu R_T \quad (1)$$