# Progress in osmotic membrane bioreactors research: contaminant removal, microbial community and bioenergy production in wastewater

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# Abstract

Renewable energy, water conservation, and environmental protection are the most important challenges today. Osmotic membrane bioreactor (OMBR) is an innovative process showing superior performance in bioenergy production, eliminating contaminants, and low fouling tendency. However, salinity build-up is the main drawback of this process. Identifying the microbial community can improve the process in bioenergy production and contaminant treatment. This review aims to study the recent progress and challenges of OMBRs in contaminant removal, microbial communities and bioenergy production. OMBRs are widely reported to remove over 80% of total organic carbon, $PO_4^{3-}$, $NH_4^+$ and emerging contaminants from wastewater. The most important microbial phyla for both hydrogen and methane production in OMBR are *Firmicutes*, *Proteobacteria* and *Bacteroidetes*. *Firmicutes*' dominance in anaerobic processes is considerably increased from usually 20% at the beginning to 80% under stable condition. Overall, OMBR process has great potential to be applied for simultaneous bioenergy production and wastewater treatment.

*Keywords*: Bioenergy; Forward osmosis; MBR; Microbial community; OMBR

## 1. Introduction

The rapid industrialization, urbanization, and global population growth have led to considerable problems in the environmental and energy fields. Today, fossil fuel is the most widely used energy source in industry, agriculture, transport and household throughout the world. It has been reported that the global energy demand will be increasing considerably in the next few decades, as energy is the most fundamental driver of the global economy. More fossil fuel consumption has resulted in increasing emission of greenhouse gases and contaminants into the atmosphere and, consequently, global warming and deteriorating air quality (Huang et al., 2019; Hosseinzadeh et al., 2020a). Therefore, striking a harmony between the anthropogenic activities and sustainability of the environment is of great importance (Ali et al., 2016) resulting in more attention in renewable energy production (Sun et al., 2019b). It is expected that renewable energy sources will contribute to more than 50% of the total electricity generation by 2040, which more than doubles the value of 22% in 2016 (Alassi et al., 2019). In addition, water shortage is currently a serious problem worldwide, which is exacerbated by climate change. The annual water requirement is growing rapidly owing to world population increase and industrialization (Hosseinzadeh et al., 2020a; Hosseinzadeh et al., 2020b). There are different technologies to tackle each of these challenges individually, e.g. by applying advanced oxidation (Bao et al., 2020), adsorption (Alidadi et al., 2018) or membrane processes (Cheng et al., 2018; Kheirieh et al., 2018; Luo et al., 2018) for water treatment and reclamation, and by producing energy from renewable resources, e.g. geothermal, ocean, solar, hydro, wind and wave in lieu of fossil fuels (Tran and Smith, 2017; Wu et al., 2021). More interestingly, the development of processes which can simultaneously address all three challenges of renewable energy production, water resources conservation and environmental protection are extremely important for the society today, as part of our efforts to meet the UN sustainable development goals (Hosseinzadeh et al., 2020a).

Membrane bioreactor (MBR) technology, which integrates conventional activated sludge with physical processes of membrane separation like ultrafiltration (UF) and microfiltration (MF), has been extensively developed to treat and reclaim wastewater (Cheng et al., 2018). This technology is promising and reliable by easier maintenance and operation, smaller footprint, lower generation of sludge, and better effluent quality (Liu et al., 2014; Yurtsever et al., 2015). In addition, anaerobic MBR (AnMBR) is considered as a remarkable process for wastewater treatment and energy production (Liu et al., 2021), due to the high degradation capacity of anaerobic microorganisms, longer sludge retention time, and better effluent qualities (Cheng et al., 2018). Therefore, AnMBR has a great potential to produce energy, treat wastewater and consequently protect the environment in one process (Liu et al., 2014).

Conventional activated sludge process has recently been combined with forward osmosis (FO) to create a new process called osmotic membrane bioreactor (OMBR) (Achilli et al., 2010; Hosseinzadeh et al., 2020b). In the OMBR process, the difference of osmotic pressures between two sides of the membrane is the driving force for purified water from a low salinity feed solution into the draw solution (DS) through a FO semipermeable membrane. Subsequently, some other desalination processes such as distillation and reverse osmosis (RO) may be applied to regenerate clean water from DS for different usages like irrigation as fertilizers and potable water (Alturki et al., 2012; Cai, 2016; Hosseinzadeh et al., 2020b). Concerning the orientation of the membranes, two different modes, FO and pressure retarded osmosis (PRO) are proposed for this process. In a way that when DS runs against the selective thin and support layers, the process will be called FO and PRO, respectively (Ge et al., 2013). OMBR has some advantages over the conventional MBRs such as the low energy consumption, low fouling propensity and superior performance in the removal of contaminants particularly emerging contaminants e.g. endocrine disrupting chemicals, steroid hormones, pesticides and pharmaceutically active compounds, which are of the greatest concern currently (Luo et al., 2018). The results demonstrated that the emerging contaminants with a high molecular weight (> 266 Da) were removed by more than 80%, while the removal of low molecular weight compounds was sporadic, due to the fact that FO membrane can more effectively retain high molecular weight contaminants resulting in their longer retention time and more biological degradation (Alturki et al., 2012; Blandin et al., 2018). Luo et al. (2018) reported that by using a novel biomimetic aquaporin FO membrane, 30 trace organic contaminants (TrOC) were removed by over 85% regardless of their physicochemical properties. Despite these advantages, the salinity build-up is one of the most important disadvantages of this process, which occurs in the bioreactor by virtue of the DS reverse diffusion and salt rejection (Hosseinzadeh et al., 2020b). Several MBRs review articles have focused on various characteristics such as high strength wastewater treatment with MBRs (Mutamim et al., 2012), OMBR (Viet et al., 2019), OMBR salinity build-up (Song et al., 2018), and extracellular polymeric substances in MBRs (Lin et al., 2014). Furthermore, the capability of the OMBRs in energy-nutrient-water solute recovery was reviewed and concluded that the energy balance of either electrodialysis or bioelectrochemical based OMBR processes was negative. The anaerobic OMBRs were regarded as energy efficient systems; however, the salinity build-up of OMBRs is regarded as a considerable drawback hindering such capability (Yang et al., 2021). In a biological-driven process such as OMBR, the microorganisms play a crucial role in its overall energy and environmental performance. Yet, there is a lack of study concerning the microbial community in OMBRs to assess the capability of these systems for different applications especially energy recovery. Therefore, this study aims to address the recent advances in OMBR process, particularly the microbial community controlling the process efficiency. In addition, the potential of energy production by OMBR with a focus on microbial community and other components of OMBR will be discussed. Finally, the main challenges and potential solutions are addressed in future outlook.

## 2. MBR

MBR is a hybrid treatment system composed of both biological treatment and filtration by membrane process (Luo et al., 2018). It is reported that the performance of the biological process is higher than the filtration by membrane process. The biological process converts particles and dissolved organic matter (DOM) of wastewater to flocs, which are then separated from the effluent by membrane filtration (Mutamim et al., 2012). The strengths and biodegradability of the wastewater are two important factors affecting the appropriate selection process for wastewater treatment. For more biodegradable and high-strength wastewater, AnMBRs are regarded as a better option than the aerobic MBRs (AeMBRs) from the economic and technical aspects (Mutamim et al., 2012; Pretel et al., 2016). Each of these processes has its advantages and disadvantages, which are summarized in Table 1.

Generally, the biological reactor and membrane module of MBRs can be presented in two different configurations, i.e. submerged and side-stream MBRs (Aslam et al., 2018). In submerged MBR, a submerged membrane module is directly installed in a bioreactor containing mixed liquor suspended solids (MLSS) with a vacuum pump to generate permeate from the bioreactor. In a side-stream configuration, the bioreactor is separated from the membrane module and run under pressure to generate permeate.

**Table 1.**

A summary comparison of the anaerobic and aerobic MBRs.

| | **AnMBR** | **AeMBR** | **Reference** |
|---|---|---|---|
| Advantage | Degradation of organic matter without aeration<br>Energy recovery from biogas production<br>Lower sludge production<br>Working at both low and high temperatures<br>Lower energy consumption<br>Slower fouling rates | Less fouling<br>Working at low temperature<br>Higher efficiency in treatment<br>Less membrane area needed | Wang et al., 2018; Pretel et al., 2016; Xiong et al., 2016b; Skouteris et al., 2020 |
| Disadvantage | Higher membrane investment<br>Energy demand<br>Harder fouling control<br>More fouling<br>Need for more membrane area<br>More dissolved methane at warmer effluents and more emission into the atmosphere<br>Minimal nutrient removal<br>Lower operation fluxes and higher fouling tendency | Fouling<br>More sludge production<br>Need for aeration | Wang et al., 2018; Pretel et al., 2016; Yurtsever et al., 2017; Yurtsever et al., 2015; Skouteris et al., 2020 |
| Flux | 5-12 $Lm^{-2}h^{-1}$ | 20-30 $Lm^{-2}h^{-1}$ | Wang et al., 2018 |
| $H_2$ production | √ | - | Bakonyi et al., 2015 |
| Electricity production | √ | Aerobic-anaerobic-MBR/MFC process for bioelectricity production | Tian et al., 2014; Liu et al., 2014 |
| VFA recovery | √ | - | Khan et al., 2019 |
| Micro-plastic removal | √ In a combined anaerobic/aerobic and membrane system | In a combined anaerobic/aerobic and membrane system | Talvitie et al., 2017 |
| Pharmaceutical contaminant | √ | √ | Chen et al., 2018; Ng et al., 2016 |
| Pesticide removal | √ | √ | Bakonyi et al., 2015; Mukherjee et al., 2018 |
| Textile dye removal | Almost 100% dye removal | 30-50% dye removal | Yurtsever et al., 2015 |

## 3. OMBR units

As OMBR couples FO membranes for physiochemical separation and biological activated sludge process for organics and nutrients removal, it is composed of different components that can potentially affect the performance of the system from both aspects of energy production and treatment efficiency (Aslam et al., 2018; Luo et al., 2018). These include the feed solution, FO membrane, bioreactor and DS. Furthermore, important operating parameters such as the microbial communities, salinity build-up, and fouling all affect the system's performance, especially in energy production.

### 3.1. *FO membranes*

FO membranes are commonly configurated as the asymmetric hollow fiber and flat sheet (Widjojo et al., 2013). Commercial FO membranes made from cellulose triacetate (CTA) are widely used in different applications of FO process; however, low salt rejection and water flux, especially in seawater desalination, are considered two of the most important drawbacks of these membranes (Widjojo et al., 2013). Therefore, the attentions are being paid to producing the next generation of FO membranes such as thin-film composite (TFC) membranes (Dutta and Nath, 2018). FO membranes are composed of two layers, a very thin-film active layer on a much thicker porous layer called the support or the substrate layer. The support and active layers are responsible for providing the mechanical resistance for the modules and salt rejection, respectively (Dutta & Nath, 2018). Both TFC and CTA membranes are asymmetric with a substrate and active layers(Ismail et al., 2015), and have been used in FO processes (Hosseinzadeh et al., 2020b). The main characteristics of TFC membranes are the formation of a very thin active layer (< 0.2 μm) of polyamide (PA) on the porous substrate layer with < 0.5 nm interstitial pore size. Generally, interfacial polymerization is a procedure by which active thin layers are formed in a way that a monomer as a PA thin film is polymerized on the surface of a substrate (Ismail et al., 2015) Phase inversion is another procedure by which the substrate is generated (Akther et al., 2019).

Among these two membranes, TFC is regarded as the most effective one to desalinate water (Ismail et al., 2015). According to the results obtained by (Li et al., 2018), the water permeability of TFC is approximately three times higher than CTA. In addition, reverse salt fluxes as the main reason for salinity build-up are almost three times more in TFC than CTA. By contrast, CTA membranes have demonstrated great selectivity and superior chlorine resistance in comparison to TFC (Ong and Chung, 2012). Nevertheless, the high biodegradability which can decrease the membrane lifespan, more vulnerability to high temperature, and high sensitivity to pH, are considered as some of the restrictions of CTA (Cath et al., 2006; Ismail et al., 2015). Therefore, the majority of the recent FO research has focused on the fabrication of TFC membranes as well as their modifications.

Since 2012, inorganic nanomaterials have been applied in the enhancement of the membrane performances. These nanomaterials have been used in four states, including the nanomaterial-coated active layer, nanocomposite PA active layer, nanocomposite substrate and nanomaterial interlayer (Akther et al., 2019). The nanomaterials may be dispersed in either organic trimesoyl chloride (TMC) or aqueous m-phenylenediamine (MPD) phase according to their hydrophobicity and hydrophilicity features.

Despite all of the improvements in FO membranes, more studies are needed to promote the characteristics of such membranes including antifouling property, mechanical strength, permeability, reduction of the reverse solute flux, selectivity, salt rejection, concentration polarization and chemical stability (Cath et al., 2006; Xu et al., 2017). To improve the quality of the membranes, the current focus is towards substrates, additives and two fields of interfacial polymerization including the application of novel monomers, and the modification of membrane formation processes (Lau et al., 2012).

### 3.1.1. *Monomers and nanomaterials*

Chemical characteristics such as functional groups, bonds and crosslinking along with membrane structure properties like hydrophilicity, roughness, thickness and pore dimension demonstrate strong effects on membrane performances; therefore, considering the fundamental principles of the effectiveness of the different membrane components is essential (Asghari & Afsari, 2017; Ismail et al., 2015; Kardani et al., 2020). It is apparent that the membranes' performance depends on how to optimize the fabrication condition of the membranes substrates such as materials and effective factors; furthermore, the monomers and other additives applied also play an important role in determining membrane characteristics particularly membrane selectivity (Lalia et al., 2013). Some of the extensively used monomers are isophthaloyl chloride (IPC), 5-isocyanato-isophthaloyl chloride (ICIC) and TMC, as some of the acyle chloride monomers in the organic phases, along with MPD, piperazine (PIP) and p-phenylenediamine (PPD) as diamine monomers in the aqueous phases (Lau et al., 2012; Xiong et al., 2016a). Overall, TMC and PD are the two most applied monomers for the formation of PA layer (Ismail et al., 2015; Xiong et al., 2016a).

To improve membrane performance, carbon nanotubes (CNTs), silica, graphene oxides and zeolites are considered as the most widely used nanomaterials for FO membrane modifications (Akther et al., 2019). Table 2 compares the properties and performances of different FO membranes modified by the different nanomaterials. Considering the different merits of CNTs in FO membrane structures such as self-cleaning characteristic, chemical stability, mechanical resistance, low biofouling capability and superior separation, attention has been focused on applying CNTs in FO membrane modifications. The membrane selectivity can be promoted by salt ions rejection and allowing water molecules transport owing to the specific pore diameter of CNTs. In addition, with CNT's tubular structures, its application in membranes can facilitate frictionless water molecules transport and consequently improve the permeability of the membranes. It is worth highlighting that CNTs are regarded as a hydrophobic material; therefore, they can be functionalized with polar functional groups like amine or carboxylic ones through treating by either amines or acids to become more appropriate for water treatment and disperse properly in monomer solutions (Akther et al., 2019).

Graphene oxide can increase the membrane water permeability due to some of its exclusive properties such as amphiphilicity, a large surface to thickness ratio, and ample surface functional groups. For example, the amphiphilic property of the graphene oxide may result in channel creation in its interlayer and improve water permeability. The hydrophilic hydroxyl groups of graphene oxide can firstly adsorb water molecules, and then become quickly dispersed among the hydrophobic carbon core (Hung et al., 2014).

As a microporous aluminosilicate with a porous crystalline structure, zeolites can play a molecular sieve role. In addition, zeolites can not only be resistant to different thermal and chemical conditions, but also facilitate the shape and size-selective molecules separation, due to their uniform pore system (McLeary et al., 2006).

Silica, regarded as another porous material with uniform nanostructure, is useful for size selectivity in membranes. In contrast to zeolites, silica has a spherical morphology helping to scatter properly. Owing to some exclusive characteristics of silica like high surface hydrophilicity and large specific surface area, it has been utilized in various processes such as FO, nanofiltration (NF), RO and gas separation (Kresge et al., 1992).

**Table 2.**

The properties and performances of the modified TFC membranes by CNTs, silica, GO and zeolite

| Membrane | Substrate | Sublayer/monomers and nanomaterials | Draw solution (NaCl) | Membrane Orientation | Contact angle | Application | Water flux | Reference |
|---|---|---|---|---|---|---|---|---|
| TFC | PVDF | PA/CNTs | 2 M | FO mode | 48 ° | Flux | 32 $L/m^2.h$ | Zhao et al., 2017 |
| TFC | Polyetherimide (PEI) | PA/multi-walled CNTs | 1 M | FO mode | NP | Flux | 61 $L/m^2.h$ | Tian et al., 2015b |
| TFC | PEI | PA/Silica | 1 M | FO mode | 131° | Flux | 72 $L/m^2.h$ | Tian et al., 2017 |
| TFC | PSf/GO | PA | 0.5 M | FO mode | 62 ° | Flux | 40.5 $L/m^2.h$ | Park et al., 2015 |
| TFC | PSf | PA/Zeolite | 1 M | FO mode | 72 ° | Flux/desalination | 38 $L/m^2.h$ | Ma et al., 2012 |

### *3.1.2. Substrates*

Substrates play an important role in TFC membranes performances. Hydrophilic substrates have higher preferences because they can enhance water transport into the membrane and increase water permeability. In other words, hydrophilic substrates can decrease the internal concentration polarization (ICP), hence increasing water flux. However, higher substrate hydrophilicity can adversely affect the adhesion among the active and substrate or support layers. In addition, the pore size of the substrate materials should potentially be controlled with adjusting the fabrication process like polymer concentration, coagulation environment and temperature (Akther et al., 2019; Ismail et al., 2015). Furthermore, substrates should provide a better condition for the membranes' properties, e.g. antifouling, chemical, and mechanical stability (Akther et al., 2019). Polysulfone (PSf) is a relatively hydrophobic material and most widely used substrate in membrane fabrication, due to a wide range of advantages such as high chemical, mechanical, thermal and chlorine resistance along with wide pH tolerances and high flexibility in membrane fabrication (Ismail et al., 2015; Lalia et al., 2013). Polyethersulfon (PES) polymer which has similar characteristics to PSf, is another hydrophobic and conventional substrate with higher pore size and slightly less hydrophilicity, and is widely used in membrane fabrication as well (Akther et al., 2019). In contrast to conventional membranes, Bucky-paper is regarded as a new substrate for FO membrane fabrication (Dumée et al., 2013). One of the new procedures for membrane substrate modification is embedding some nanomaterials such as CNTs, silica, zeolites, graphene derivatives and zinc oxide in the polymer dope solution as a raw material improving the chemical, thermal and mechanical properties of the membranes (Akther et al., 2019).

## *3.2. DS and energy production*

The chemical potential of an isolated system tends to be spontaneously equilibrated based on the second law of thermodynamics. In FO process, the solvent molecules spontaneously pass across a membrane from a less concentrated solution (feed solution) to a more concentrated solution (DS) to equilibrate the general chemical potential of this isolated system (Hosseinzadeh et al., 2020b). Under the osmotic pressures between two sides of the membrane and the semipermeable membranes' exclusive properties, some of the components can move to the more concentrated solution. Thus, the difference in osmotic pressure acts as a driving force in the separation process in FO. DS plays a critical role in the FO process, and its quality and property are so important in FO performance. The main characteristics of a viable DS are high osmotic pressure production resulting in high water flux and low back DS permeability; inexpensive recovery of the diluted DS and reuse; non-toxicity particularly for the production of potable water; inherent characteristics like sensitivity to pH and aqueous affinity to maintain an appropriate interaction with water; no adverse effects on bacterial activity and sludge quality in OMBR, and the environment and human health (Bowden et al., 2012; Cai, 2016; Gwak et al., 2015); good degradability resistance unless in post-FO (Lutchmiah et al., 2014a); low viscosity even at high concentrations (Cai, 2016); and high solubility, diffusivity and viscosity (Pathak et al., 2018). Up to now, a wide range of DSs has been used in the FO process. The applied DS in FO process has been classified into two major categories: novel synthetic materials and conventional compounds, which are further divided into four subcategories comprising organic and inorganic solutes, volatile compounds and gases (Alejo et al., 2017; Ge et al., 2013). DS may also be classified into responsive and non-responsive groups. The responsive DS refers to the DS’s important water affinity changes in response to different incentives like light, electro-magnetic, pH, and temperature. In contrast, there is no significant change in water affinity of the non-responsive DSs upon exposure to the different incentives. The mentioned ability of the responsive DSs can result in their easy regeneration while maintaining their high quality (Cai, 2016). The majority of the non-responsive DSs focus on the enhancement of the osmotic pressure and the abatement of the reverse diffusion led salinity build-up. One of the advantages of these DSs is the fact that NF instead of RO can be applied to regenerate most of these DSs. In addition, it has been claimed that FO-RO or FO-NF processes need less energy than FO alone to desalinate seawater (Cai, 2016); however, this has been called into question by other studies from at least the theoretical aspects (Elimelech & Phillip, 2011; Phuntsho et al., 2014). Organic and inorganic salts along with a wide range of water-soluble polymers have been applied as some of the non-responsive DSs in the FO process. Since the 1970s, saccharides and sugars, as well as various inorganic salts such as $MgCl_2$, NaCl, $KHCO_3$, $Ca(NO_3)_2$, $(NH_4)_2SO_4$, and $Na_2SO_4$ have been used as DSs in FO process (Achilli et al., 2010). Since the monovalent salts such as NaCl, $NH_4Cl$ and $KNO_3$ can generate more water fluxes, these traditional DSs have been widely used in FO processes. However, these salts need to be separated by the RO process, which will increase energy consumption of the overall process. Besides, more back diffusion is regarded as another drawback of these DSs (Achilli et al., 2010; Lee et al., 2010). To tackle these problems, di- and trivalent salts like $CaCl_2$, $MgCl_2$ and $FeCl_3$ have been proposed as alternatives, although the back diffusion of 5.6 $g/m^3$ reported for 0.5 $MgCl_2$ was considered high (Nguyen et al., 2015; Trung et al., 2017). Also, cobaltous and ferric hydroacid compounds were applied as a DS. One of the most important properties of these compounds is high osmotic pressure due to high dissociation rate and solubility, resulting in a high flux of water ranging from 60 to 80 $L\ m^{-2} \cdot h^{-1}$. On the other hand, the reported back diffusion for these DSs is lower than the monovalent salts due to their larger size (Ge et al., 2014). Table 3 summarizes the commonly used DSs in OMBRs, detailed information of membranes, and procedures of controlling the salinity build-up.

### 3.2.1. *Organic DSs*

A wide spectrum of organic compounds has been applied as DS in FO processes such as sodium lignin sulfonate (NaLS), poly(sodium acrylate), sodium ethylenediaminetetraacetic acid (EDTA-Na) salts and the sodium salt of poly(aspartic acid) (Alejo et al., 2017). Reverse diffusion is one of the most important parameters affecting the efficiency of the FO process, as reverse diffusion directly depends on the molecular dimensions of the DS components. In general, the higher the molecular dimensions and weights of the components, the lower the reverse diffusion. However, high viscosity can heighten the problems like DS circulation and concentration polarization in both FO and RO processes (Tian et al., 2015a). Based on such conditions, MF and UF are regarded as the appropriate procedures to regenerate these DSs. Although proved to be perfectly efficient, their main drawback for field-scale implementation is toxicity (Alejo et al., 2017). Polyelectrolytes such as poly sodium4-styrenesulfanate (PSS) with various molecular weights (70,000, 200,000, 1,000,000 Da) have recently been applied as a DS in the FO process, and have demonstrated higher FO process performance from the aspect of water flux and the reverse diffusion due to their large molecular size. Nonetheless, the application of the recovered DS by UF process did not show satisfactory results with > 40% decrease in water flux, due to its loss during recovery (Tian et al., 2015a).

#### 3.2.2. *Fertilizers and volatile gases*

Another interesting approach is to use fertilizers as a DS in the FO process so as to avoid the regeneration of the diluted DS, which can be used directly for agricultural application. However, the most important drawback of this idea is finding an appropriate membrane for such a system. Some of the applied fertilizers as DS in the FO process were calcium nitrate, potassium sulphate, ammonium sulphate, potassium chloride and ammonium chloride (Alejo et al., 2017). Also, $CO_2$ and ammonia were simultaneously used to produce ammonium bicarbonate as a DS, which can be regarded as a fertilizer (Cai, 2016). Similarly, human urine was also used as a DS in the FO process (Volpin et al., 2019). In addition, different soluble gases may be applied as DS. For example, the application of $SO_2$ along with $CO_2$ or $NH_3$ has been proposed as potential DSs with desirable osmotic pressures. Their regeneration process can be accomplished by evaporation and air-striping under pressure and temperature controlled conditions (Cai, 2016).

**Table 3.**
The draw solutions applied in osmotic membrane bioreactors.

| DS | DS concentration | Water flux (L.m$^{-2}$.h$^{-1}$) | Osmotic pressure | Membrane | Membrane orientation | Effective mem. Area (m$^2$) | Salinity build-up control method | Removal (%) | Reference |
|---|---|---|---|---|---|---|---|---|---|
| $MgCl_2$,<br>NaCl | 48400 mg L$^{-1}$,<br>49000 mg L$^{-1}$ | 6.46 | 4.0 MPa at 23 °C | CTA-ES flat-sheet | FO mode | 2 × 0.018 | Daily discharge of 146 mL supernatant | > 97% TOC;<br>> 97% $NH_4^+$;<br>> 95% $PO_4^{3-}$ | Qiu & Ting, 2014 |
| Synthetic wastewater | 35000 mg L$^{-1}$ | 10±2 | | TFC | FO mode | 0.05 | MF | >90% TrOC | Blandin et al., 2018 |
| NaCl | 0.5 M | 6.06-9.67 in FO, 7.23-9.24 in MF | | TFC flat sheet<br>MF (PVDF) | FO mode | 0.056 | MF (PVDF): 0.20 µm | >95% TrOC | Zhu et al., 2018 |
| Industrial effluent containing ammonium sulphate | EC: 130 mS cm$^{-1}$<br>pH: 4 | 1.6-7.8 | | CTA-NW flat sheet | FO mode | 0.042 | | 80% COD | Luján-Facundo et al., 2018 |
| NaCl | 1 | FO: 7.7–9.5<br>MF: 11±1 | | TFC flat sheet<br>MF (PVDF) | FO mode | 0.056 | MF (PVDF): 0.20 µm | 98% TOC;<br>> 85% $NH_4^+$; | Wang et al., 2017b |
| Synthetic seawater with10 salts, e.g.<br>NaCl<br>$MgCl_2$<br>$Na_2SO_4$<br>$CaCl_2$ | <br><br>24530 mg L$^{-1}$<br>5200 mg L$^{-1}$<br>4090 mg L$^{-1}$<br>1160 mg L$^{-1}$ | ~3-7 | 26.45 bar | CTA | FO mode | 0.020 | | >90% TOC;<br>80.2% $NH_4^+$;<br>>90% $PO_4^{3-}$ | Sun et al., 2016 |
| NaCl | 0.5 M | 3 | | CTA-flat sheet | FO mode | 0.12 | | 98% TOC;<br>98% $PO_4^{3-}$-;<br>80% $NH_4^+$; | Aftab et al., 2017 |
| NaCl | 6 M | ~3 (stable flux) | | CTA-flat sheet | FO and PRO modes | 0.0162 | | >80% TrOC | Alturki et al., 2012 |
| NaCl | 1.1 M | 10.7-6.8 | 51.78 atm;<br>EC: 91.26 mS cm$^{-1}$ | flat-sheet TFC-PA<br>MF | FO mode | 0.0264 | MF: 0.33 µm | 92% TOC;<br>76% TN;<br>63% $PO_4^{3-}$ | Pathak et al., 2020 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| NaCl,<br>KCl,<br>NaOAc | 0.75 M<br>0.75 M<br>0.75 M | 7.95–6.5<br>7.46–6.0<br>7.1–2.6 | 34.08;<br>32.45;<br>30.38 atm | CTA-flat sheet | FO mode | 0.0264 | MF: 0.33 μm; 0.1 $m^2$ area | 98% TOC; 97% $PO_4^{3-}$; 85% TN | Pathak et al., 2018 |
| NaCl | 1.1 M | 11.54–6.98 | 51.78 atm; EC: 91.26 mS $cm^{-1}$ | TFC-flat sheet | FO mode | 0.0264 | MF: 0.33 μm; 0.1 $m^2$ area | 97% TOC; 87% $PO_4^{3-}$; 94% TN | Pathak et al., 2017 |
| NaCl | 49000 mg $L^{-1}$ | 8.64-5.15 | 4.0 MPa at 23.5 °C | CTA-flat sheet | NP | 0.036 | Discarding 150 mL clear supernatant daily | >98% TOC; >97% $NH_4^+$; > 75% antibiotics | Srinivasa Raghavan et al., 2018 |
| NaCl | 0.5 M | 4-8 | | TFC-flat sheet | FO mode | 0.0300 | - | 100% TOC | Luo et al., 2017 |
| NaCl | 0.5 M | 8.7-4 | | CTA-flat sheet | FO mode | 0.0272 | - | >93% COD; >99% phosphorus | Hou et al., 2017 |
| NaCl | 0.5 M | 9.22-2 | | TFC-flat sheet | FO mode | 0.025 | MF (PVDF): pore Size: 0.20 μm 0.025 $m^2$ area | 98 % TOC; 100 % TP; $NH_4^+ \approx 0\%$ | Wang et al., 2017a |
| NaCl | 0.25 M<br>0.5 M<br>1 M<br>2 M | ~4.5-1.2<br>~7.7-2.5<br>~13.8-5<br>~17-3 | 11.45 bar<br>22.95 bar<br>47.39 bar<br>102.04 bar | TFC | FO mode | 0.025 | MF (PVDF):<br>0.025 $m^2$ area | > 98% TOC; 99.7 TP; $NH_4^+$ = 35-53% | Wang et al., 2019 |

TP: total phosphorous

### *3.2.3. Nutrient compositions*

Nutrients such as glucose, glucose-fructose combination and sucrose (Su et al., 2012) are the other types of DSs used in the FO process. They were first applied to a water supply in lifeboats and then used in wine, food and wastewater treatment (Su et al., 2012). Currently, the application of such DSs in FO processes is attracting more attention. It was reported that the water flux produced from 1 M sucrose is comparable to that from 1 M $MgCl_2$. As the molecular size of sucrose is greater than that of $MgCl_2$, therefore, the reverse flux of sucrose DS is negligible (Su et al., 2012).

### *3.2.4. Advanced DSs*

#### *3.2.4.1. Nanoparticles and coated nanoparticles*

In general, nanoparticles in DSs are applied to provide various benefits including higher osmotic pressure, less reverse flux and higher DS regeneration ability. In some cases, various modifications on nanoparticles are performed, e.g. coating some polymers on nanoparticle to prevent irreversible agglomerations, dimension uniformity of the used nanoparticles, which are favorable for efficient regeneration (Cai, 2016). The theoretical support for providing higher osmotic pressure by these DSs is when a hydrophilic polymer like poly(ethylene glycol) diacid or polyacrylic acid is coated on magnetic nanoparticles, the synthesized hydrophilic nanocomposites are able to absorb more water molecules into DS. Therefore, there is a potential to provide more water flux, although a moderate water flux (approximately 18 LMH) was reported at the first stage. On the other hand, since the size of nanoparticles, ranging from 1 to 100 nm, is bigger than the pore sizes of TFC and CTA membranes as two common FO membranes, the reported reverse flux was lower than the traditional DSs. Regarding the regeneration, various procedures are used to recover the diluted DSs such as different membrane technology, and electric and magnetic fields based on the characteristics of the applied nanoparticles (Alejo et al., 2017; Cai, 2016). One of the important merits of magnetic nanoparticle-based DSs is their easy regeneration by the application of a magnetic field. However, the agglomeration under high-strength magnetic field is regarded as its main drawback. Ultrasonication is reported as a non-efficient procedure for the regeneration of such

DSs, although the application of an electric instead of magnetic field together with UF has demonstrated better outcomes. It is worth mentioning that the size of nanoparticles used is a crucial parameter for DS regeneration in FO processes, as nanoparticles less than 11 nm or larger than 20 nm are difficult to be regenerated. In addition, the application of uniform sized nanoparticles is recommended for maximizing regeneration efficiency (Alejo et al., 2017; Cai, 2016).

#### 3.2.4.2. *Thermal responsive DSs*

These DSs demonstrate different solubility under various temperature conditions. There is a paucity of information concerning this type of DSs performances. The scarcity of appropriate membranes for different temperature conditions can also be a limiting factor for applying such DSs. It was reported that a smart DS, thermal responsive magnetic nanoparticles, as a subcategory of thermal responsive DSs, was produced by coating composite nanoparticles with a polymer (Cai, 2016; MingáLing, 2011). However, the spontaneous agglomeration of these coated nanoparticles, which can occur above the critical temperature, is a reason to restrict the use of magnetic fields for their regeneration (MingáLing, 2011; Zhang et al., 2010). In addition, there is another category of DSs with lower critical solution temperature (LCST) in aqueous solutions, which undergo phase separation to form two phases, i.e. water-rich and DS-rich (Nakayama et al., 2014).

#### 3.2.4.3. *Polarity switchable DSs*

Another type of DS having many applications in different industries is switchable polarity solvents, which can change the water affinity of these DSs. For example, a hydrophobic amine reacts with $CO_2$ and is subsequently changed to become hydrophilic. Therefore, this hydrophilic DS draws water with more osmotic pressure. In order to regenerate this DS, $CO_2$ is eliminated from the structure of these molecules, and the hydrophilic DS produced is changed back to become hydrophobic, which facilitates both water and amine recoveries (Stone et al., 2013).

However, as shown in Table 3, the most widely used DS in OMBR remains NaCl. This salt, as well as KCl, command no toxicity and great solubility. Furthermore, NaCl and KCl as the most widely used DS of OMBR with great water flux have been introduced as ideal DSs to produce high osmotic pressure and minimize ICP (Pathak et al., 2018). However, it has been noted that the application of organic-based DSs like NaOAc demonstrates less reverse flux due to their biodegradability. In addition, it has been reported that the presence of different elements in bioreactors can positively increase the activity of the activated sludge and energy production (Zhang et al., 2015). Thus, applying appropriate DSs and the reverse diffusion of these DSs can demonstrate desired effects on the metabolisms of the bioenergy producing microorganisms leading to more energy production.

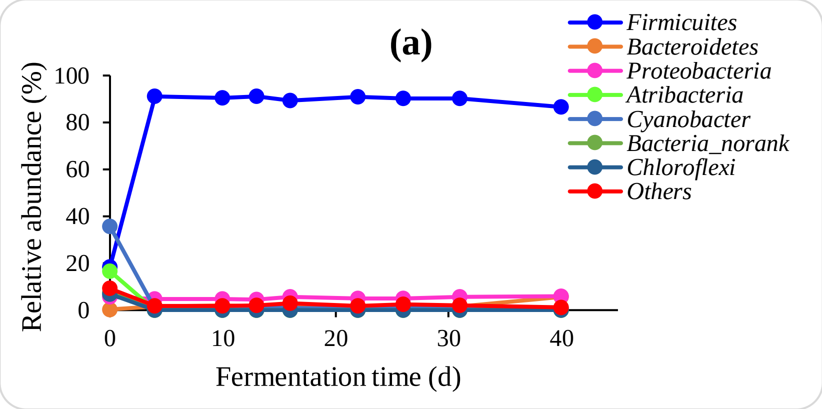
## 4. Microbial communities and energy production

In order to obtain the optimized outcomes from anaerobic biological processes for wastewater treatment and energy production, the collaboration between the microbial species in the reactor plays a critical role (Appels et al., 2011). It has been reported that bacteria and archaea are the primary microorganisms of the anaerobic systems. According to the results obtained, the degradation of the organic matter and formation of volatile fatty acids (VFAs) as intermediate products is accomplished by bacteria, and archaea carry out further degradation of VFAs to produce biogases (Nakasaki et al., 2015). In investigating the microbial communities of three reactors including upflow anaerobic sludge blanket (UASB), microbial fuel cell (MFC)-blast furnace dusting ash (BFDA)-UASB and BFDA-UASB, *Firmicutes* were observed to be the most dominant phylum, and *Actinobacteria, Proteobacteria,* and *Bacteroidetes* were the most abundant in all three reactors (Yang et al., 2019). In addition, (Nakasaki et al., 2015) studied the diversity and richness of the microorganisms in anaerobic biological reactors with and without salinity, and found that the salinity increased the diversity but decreased the richness of the microorganisms in the reactors. Salinity is regarded as one of the most important parameters affecting microbial communities in various biological systems. A wide range of halophilic microorganisms can be active in saline biological systems. The biological evolutionary process, ecological environment, temperature, and saline level can be considered as the most effective parameters in selecting the halophilic microorganisms in various systems (Tan et al., 2019). Halotolerant or halophilic microorganisms are classified into four categories i.e. salt-sensitive, low salinity, medium salinity and high salinity which are able to activate under < 10000 mg $L^{-1}$, 10000-30000 mg $L^{-1}$, 30000-150000 mg $L^{-1}$ and > 150000 mg $L^{-1}$ NaCl, respectively. Luo et al. (2016b) studied the structure and diversity of the microorganisms in a MBR, and indicated that the salinity reduced the performance of the biological process, but did not affect the diversity of the microbial community. Furthermore, they showed that *Proteobacteria*, *Bacteroidetes, Planctomycetes* and *Armatimonadetes* were the dominant phyla of the process without salinity; however, after augmentation of the salinity, *Actinobacteria* were identified as one of the most dominant phyla while *Armatimonadetes* could not tolerate the salinity. Table 4 shows the dominant microorganisms in various biological systems at phylum and genus levels.

**Table 4.**
Dominant microorganisms of the various saline biological systems at phylum and genus levels.

| Process | Dominant microorganism | | Type of salinity | Aerobic/ Anaerobic | Product | Reference |
|---|---|---|---|---|---|---|
| | phylum level | genus level | | | | |
| An-OMBR | *Firmicutes*<br>*Proteobacteria*<br>*Aminicenantes*<br>*Chloroflexi*<br>*Actinobacteria* | NP | $MgCl_2$ | Anaerobic | | Zhao et al., 2019 |
| An-MBR | *Proteobacteria*<br>*Bacteroidetes*<br>*Chloroflexi*<br>*Firmicutes*<br>*Actinobacteria* | Bacterial:<br>*Dechloromonas*<br>*Smithella*<br>*Ottowia*<br>Archaeal:<br>*Methanosaeta*<br>*Methanolinea*<br>*Methanobacterium* | | Anaerobic | methane | Niu et al., 2020 |
| An-MBR | *Firmicutes*<br>*Bacteroidetes*<br>*Bacteroidetes*<br>*Proteobacteria* | *Clostridium*<br>*Bacteroidetes*<br>*Cytophaga*<br>*Geobacter* | | Anaerobic | methane | Seib et al., 2016 |
| An-MBR | *Proteobacteria*<br>*Bacteroidetes*<br>*Firmicutes*<br>*Verrucomicrobia* | Archaeal:<br>*Methanosaeta*<br>(*Methanothrix*) | | Anaerobic | methane | Damodara Kannan et al., 2020 |
| An-MBR | *Firmicutes*<br>*Chloroflexi*<br>*Proteobacteria*<br>*Bacteroidetes* | Bacterial:<br>*Trichococcus,*<br>*Clostridium-sensu-stricto*<br>*Ornithobacterium*<br>Archaeal:<br>*Methanosaeta* | | Anaerobic | methane | Li et al., 201) |
| OMBR | *Proteobacteria*<br>*Bacteroidetes*<br>*Gemmatimonadetes*<br>*Nitrospirae* | NP | NaCl | Aerobic | - | Luo et al., 2017 |
| OMBR | *Proteobacteria*<br>*Bacteroidetes*<br>*Saccharibateria*<br>*Nitrospirae* | Bacterial:<br>*Ferribacterium*<br>*PHOS-HE51_norank*<br>Archaeal:<br>NP | Seawater | Aerobic | - | Sun et al., 2018 |
| MBR | *Proteobacteria*<br>*Bacteroidetes*<br>*Planctomycetes*<br>*Actinobacteria* | NP | | Aerobic | - | Luo et al., 2017 |

As shown in Table 4, *Firmicutes, Proteobacteria, Bacteroidetes* and *Chloroflexi* were the most dominant phyla in both saline and non-saline anaerobic systems in which treatment coupled with methane production is regarded as the main targets. However, it should be stressed that in a study, it was suggested that *Bacteroidetes* in OMBR with salinity build-up can be more active than in the MBR without salinity. *Proteobacteria* and *Bacteroidetes* are regarded as the most dominant phylum in both saline and non-saline aerobic systems (Luo et al., 2017). Concerning biological $H_2$ producing systems, Yang and Wang analyzed the changes of the microbial community of a dark fermentation process, and found that after 6 h fermentation, the relative abundance of *Firmicutes* phylum was more than 99.5% while the first inoculum structure was *Proteobacteria* (2%), *Synergistetes* (2.4%), *Chloroflexi* (5.1%), *Actinobacteria* (6.8%), *Atribacteria* (8.3%), *Bacteroidetes* (36.1%) and *Firmicutes* (38.1%) (Yang & Wang, 2019). In addition, (Rafieenia et al., 2019) detected *Proteobacteria* and *Firmicutes* with 48.2% and 34.8% as the most dominant phyla at the end of fermentation. Furthermore, in another study, Yin and Wang detected *Firmicutes* and *Proteobacteria* phyla with 40.7% and 35.9% dominance respectively (Yin & Wang, 2016). Therefore, for anaerobic $H_2$ production, *Firmicutes* and *Proteobacteria* phyla are widely reported as the most dominating species. As shown in Fig. 1, which is based on the literature results (Jia et al., 2019; Laothanachareon et al., 2014; Slezak et al., 2017; Yang & Wang, 2019), the augmentation of *Firmicutes* from the beginning to the final stable condition of the anaerobic process (dark fermentation) for $H_2$ production is considerable.



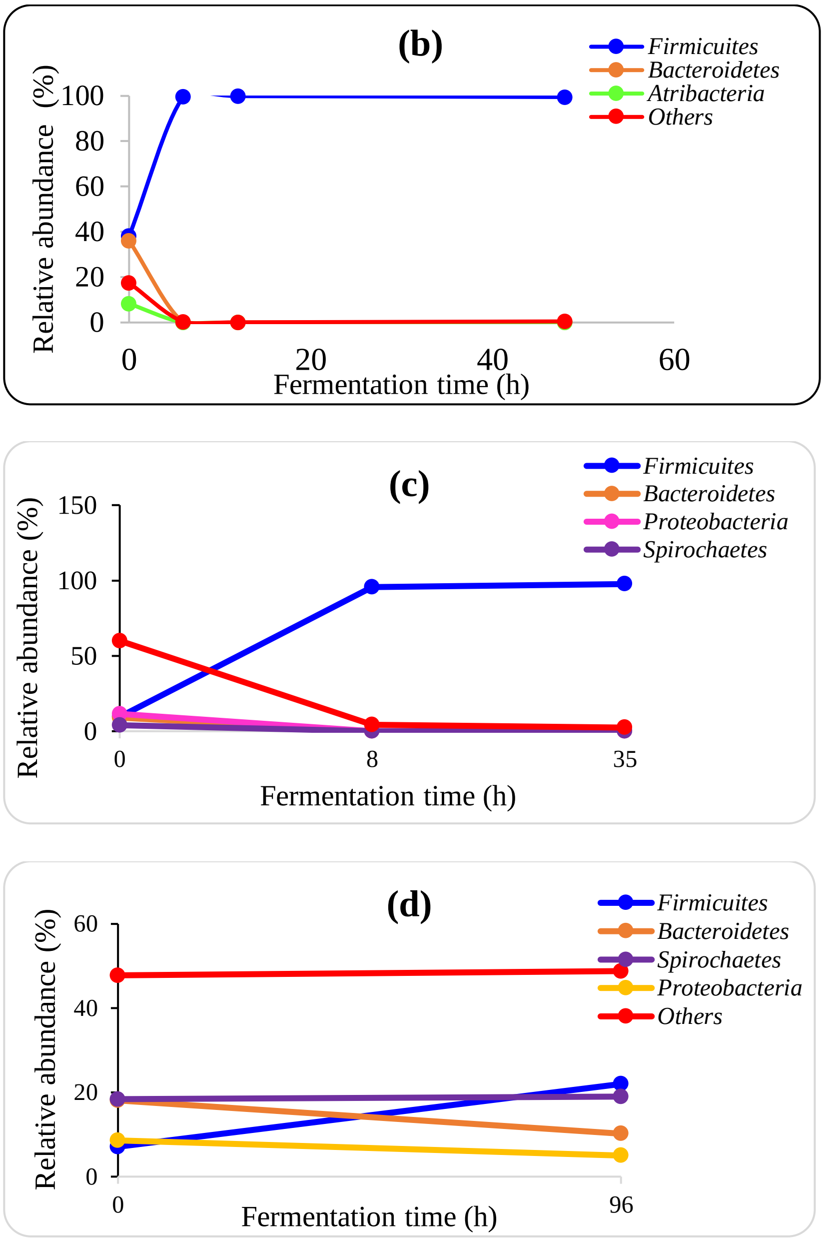


**Fig. 1.** Changes in the composition of the microbial communities during the anaerobic process for $H_2$ production in different studies. Data for (a) from Jia et al. (2019); (b) from Yang and Wang, (2019); (c) from Laothanachareon et al. (2014); (d) from Slezak et al. (2017).

## 5. Effective operating parameters and energy production

### *5.1. Temperature and pH*

There are three temperature conditions for bacteria growth, being < 15°C for psychrophilic, 25-40 °C for mesophilic, and 50-60 °C for thermophilic, respectively. The temperature has a significant effect on microbial activity and consequently, the performance of biological processes. For instance, in a microbial electrolysis cell, a reduction of process operating temperature from 25-30 °C to 4 and 9 °C reduced methane production as a process final product with changing microbial diversity (Lu et al., 2011; Lu et al., 2012). *Geobacter (Ge.)* was detected as a dominant genus in a temperature range of 4-30 °C in this system; however, temperature variation led to changes in the types of *Geobacter*. It has been reported that the domination of *Ge. chapelleii* changed to *Ge. psychrophilus,* when the temperature was reduced from 25 to 4 or 9 °C (Lu et al., 2011). The optimum temperatures for aerobic and anaerobic processes are different as shown in Tables 5 and 6, with 35 °C and 25 °C as the desirable temperature for anaerobic and aerobic processes, respectively.

Biogas production is regarded as one of the most important advantages of anaerobic processes. Methane, as well as hydrogen, are two of the most important biogases; however, each of them can be produced under some specific conditions. The reported optimum temperatures for both hydrogen and methane production are listed in Table 5. In addition, (Khan et al., 2016) suggested that 43–47 °C is the temperature range in which most methane has been produced. However, a noticeable increase in methane production from 30 to 40 °C is obvious. Therefore, the optimum temperature for OMBR operation depends on the defined purpose of the process.

**Table 5.**

The reported optimum pH and temperature for hydrogen and methane production.

| | Process | Optimum pH | Optimum temperature | Optimum ORP | $H_2$ production | Reference |
|---|---|---|---|---|---|---|
| Hydrogen production | Dark fermentation | 5.4 | 26 °C | | 1 mole $H_2$/mole glucose | Infantes et al., 2011 |
| | Dark fermentation | Optimum: 5.8<br>Initial: 7 | 30-55 °C (30–40 °C) | -507 mV<br>Initial -192 | 1.3 mol $H_2$/(mol-xylose) | Lin et al., 2008 |
| | Dark fermentation | 5.6 | At only 35 °C | -491 mv | 20 L $H_2$/L.d HPR;<br>1.53 ± 0.7 (mol $H_2$/mol hexose) HY | Lin et al., 2016 |
| | Dark fermentation | 3.7-6.5 | 35 °C | | 1090 mL cumulative $H_2$ production | Ma and Su, 2019 |
| | Dark fermentation | Inlet pH: 6.5<br>Influent pH: 4.49 ± 0.46 | 37 °C | | 2.35 mol $H_2$/mol substrate HY;<br>0.085 L/h.L HPR | Tomczak et al., 2018 |

|  |  |  |  |  |  |
|---|---|---|---|---|---|
|  | Effluent pH: 3.63 ± 0.51 |  |  |  |  |
| Methane production | 7.0 ± 0.2 | 35 ± 1.0 °C | - | Maximum methane accumulation 270.6 ± 13.4 to mL $g^{-1}$ VSS | Tao et al., 2020 |
|  | ≈ 7 | Thermophilic condition |  | Maximum cumulative methane production and yield $7386 \pm 134$ mL $d^{-1}$ and $310.4 \pm 9.2$ mL $g^{-1}$ VS | Sun et al., 2019a |
|  | 7-8 | 54 °C | - | Production rate and methane yield of 15.63 L $CH_4$ $d^{-1}$ and 0.803 L $CH_4$ $g^{-1}$ COD rem.$d^{-1}$ | Zainal et al., 2020 |

All of the biological processes like OMBR can potentially be operated in an anaerobic state and produce value-added products. In order to tackle the production of biohydrogen, biomethane and VFAs in anaerobic processes, pH is regarded as one of the most important factors (Khan et al., 2016). Reported desirable pH values for hydrogen and methane production through biological processes are approximately 5.5 and 7, respectively (Ruggeri et al., 2015; Sun et al., 2019a). The reported optimum pH for production of hydrogen and methane in different studies are presented in Table 5. To produce VFAs by different fermentation processes, approximately pH 10 has been reported as the optimum condition (Jie et al., 2014; Wu et al., 2010). However in other studies, a wide range of pH from 5.25 to 11 has been reported as the optimum for extracting different VFAs from different feedstocks depending on the type of the wastes (Lee et al., 2014). The pH has direct effects on the structure, morphology and metabolic processes of microorganisms (Lin et al., 2012), hence each enzyme can reach its highest activity at a specific pH. The pH can be so important in the selection of certain bacteria with specific abilities (Lay et al., 1997); therefore, pH is applied as a control to choose desired microbial communities for a specific purpose. For instance, by reducing pH to 5-6, Chung and Okabe suppressed the methanogens bacteria so that the hydrogen-producing bacteria became dominating in a biohydrogen production process (Chung & Okabe, 2009).

Besides, pH can be effective in influencing membrane function and its fouling propensity as well. Acid pH 4.9 was suggested as more suitable than alkaline pH 9 in controlling the fouling behavior of FO membrane (Viet et al., 2019). In addition, pH affects water flux and reverse salt flux in FO processes. It was reported that the increase of pH from 4.5 to 7 caused an increase in osmotic pressure in DS, and a slight increase in the proportions of water flux and reverse salt flux (Hau et al., 2014; Wang et al., 2014a).

### *5.2. Hydraulic retention time (HRT)*

HRT is a vital parameter in biological treatment processes affecting both the properties of the sludge and the process performance (Song et al., 2017), as well as membrane fouling (Zhen et al., 2019). Based on a study by (Kunacheva et al., 2017), the elimination of COD in an AnMBR experienced a stable condition under a wide range of HRT from 2 h to 12 h; however, more reduction of HRT down to 1 h resulted in decreasing elimination of COD and accumulation of the VFAs and consequently the poor quality of the effluent. However, when Ho and Sung (2009) reported that HRTs from 6 to 12 h did not considerably affect the elimination of COD and methane production. Nevertheless, these different studies (Ho & Sung, 2009; Kunacheva et al., 2017) recommended the use of HRTs ranging 2 to 10 h. In comparison, as presented in Table 6 and a study by (Viet et al., 2019), a wide range of HRTs up to 408 h have been applied in OMBRs indicating that the optimal HRT may be highly dependent on the system being studied.

**Table 6.**
Operating condition of the OMBRs and AnOMBRs.

| | HRT (h) | SRT (d) | DO (mg $L^{-1}$) | Temperature (°C) | Initial MLSS (g $L^{-1}$) | pH | Reference |
|---|---|---|---|---|---|---|---|
| OMBR | 25-158 | 30 | 5 | 26 | | | Yao et al., 2020 |
| OMBR | 5.7-6.8 | 10 | 3.5 | 25 | | | Wang et al., 2017b |
| OMBR | 30-80 | 50 | 5 | 22 | | 6-7 | Luo et al., 2016a |

|  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|
| AnOMBR | 408 | 40-50 | Anaerobic | 25 |  | Zhao et al., 2019 |
| AnOMBR | 15-40 | 60 | Anaerobic | 25 | 5 | Wang et al., 2018b |
| AnOMBR | 12.5-90 | 90 | Anaerobic | 25 | 3.8 | Wang et al., 2017a |

### *5.3. Solid retention time (SRT)*

SRT is another important factor of biological treatment processes affecting the elimination of COD and production of biogases from wastewater or sludge. In addition, there is a reverse relationship between the SRT and the proportion of sludge produced from biological treatment processes (Zhen et al., 2019). It has been reported that appropriate concentration of biomass and SRT are needed for the complete decomposition of COD. In a study by (Huang et al., 2011), it was reported that limited SRT increase could positively affect the proportion of the produced biogases, and optimum SRT is essential for maintaining the performance in biological treatment processes. It may be attributed to the more adsorption of substrate in biological process than degradation by microorganisms. Therefore, the SRT for MBRs can be different depending on various factors, e.g. the substrates (Zhen et al., 2019). According to Fig. 2 which drew results from literature (Aftab et al., 2017; Qiu and Ting, 2014; Srinivasa Raghavan et al., 2018; Sun et al., 2016; Wang et al., 2019; Wang et al., 2017b), OMBR demonstrates maximum performance in removing TOC and $NH_4^+$ from wastewater under different operating conditions and SRTs. As observed in Table 6 and Fig. 2, the value of the SRT in OMBR is highly variable ranging from 10 to 100 days. It has been reported that at longer SRTs, the MBRs face with more membrane fouling. As in such conditions, the biological system is operated in endogenous phase owing to less food per microorganism resulting in more death and degradation of the microorganisms and more accumulation of the soluble microbial products (SMPs) in the system. Since the SMPs are regarded as one of the most important causes of the membrane fouling, the higher membrane fouling can be caused due to this reason (Huang et al., 2011). Apart from the effects on the properties of sludge and membrane fouling, SRT can adjust the accumulation of salts in the OMBRs. Higher SRT can result in salt accumulation, and consequently, less water flux as well (Wang et al., 2014b; Zhu et al., 2018). It is worth highlighting that the relationship between HRT and SRT is thought-provoking in a way that at high HRTs (> 5 h), the performance of the AnMBR becomes SRT-independent, and the reverse is true when SRTs are longer than 30 d (Zhen et al., 2019).

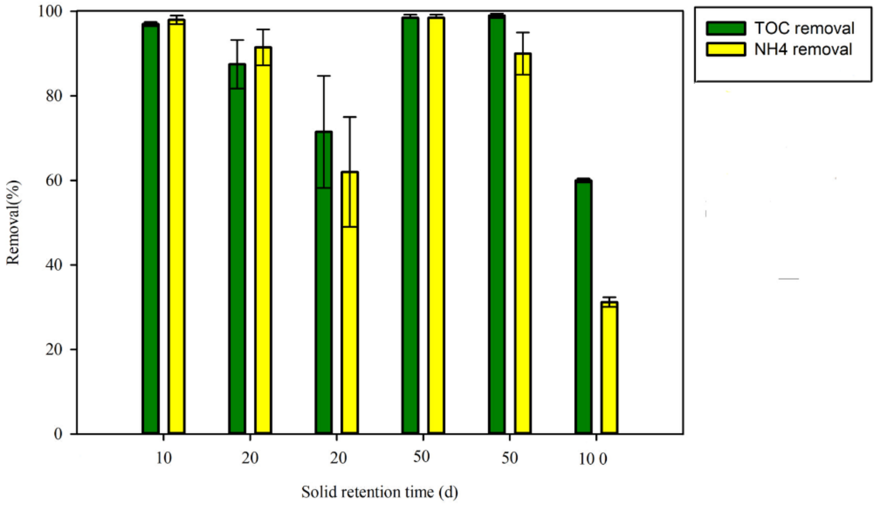


**Fig. 2.** Performance of OMBR in TOC and $NH_4^+$ removal from wastewater with various SRTs (Aftab et al., 2017; Qiu & Ting, 2014; Srinivasa Raghavan et al., 2018; Sun et al., 2016; Wang et al., 2019; Wang et al., 2017b).

## 6. Challenges

### *6.1. Salinity build-up*

In OMBR, the reverse diffusion of DS transfers solutes to the bioreactor and forms salinity build-up, usually indicated by mixed liquor conductivity. Salinity build-up can detrimentally influence the biological process, and cause more dilutive ICP in OMBRs resulting in water flux reduction (Hosseinzadeh et al., 2020b). The extent of the salinity build-up is highly dependent on the membrane selectivity determined by the water vs. salt permeability of the membrane (Song et al., 2018). In contrast to applying the highly selective FO membrane in OMBRs, the proportion of the reverse solute and consequently salinity build-up will be negligible in OMBR systems in case of using a less selective membrane (Hou et al., 2016). Moreover, the system operating parameters can be effective in controlling the proportion of the solute in OMBRs, as for example, the higher the HRT and the lower the SRT, the more appropriate for the reduction of salinity build-up in OMBRs. However, in general, the more suitable condition for better operation of the biological systems is longer SRT and shorter HRT. Therefore, the optimization of the operating parameters in OMBRs is essential which can be considered as the salinity build-up control in OMBRs (Xiao et al., 2011). In addition, installing MF membranes in OMBRs for salinity build-up reduction can be another option in this regard (Hosseinzadeh et al., 2020b). Different studies have reported the detrimental effects of the salinity build-up on biomass. (Wang et al., 2014b) demonstrated decreased dehydrogenase enzyme activity, which is considered an indicator of microbial activity, over time in OMBR. Furthermore, the reduction in the proportion of the biomass in reactor along with a reduction in the specific oxygen uptake rate as another microbial activity indicator suggested the adverse effects of the salinity build-up on the process (Luo et al., 2018). However, it has been reported that the adverse effects of salinity build-up on biomass proportion and biomass activity decreased over time, due to microbial adaptation (Luo et al., 2016b; Qiu and Ting, 2013). In addition, the influence of the salinity build-up on the production of extracellular polymeric substances (EPS) and SMP was detected. Also, the salinity increase in the OMBR process broke the flocks and reduced the size of particles (Wang et al., 2014b). It is worth highlighting that EPS and SMP can augment the fouling of membrane systems. Therefore, as observed by Qin and Tin, salinity increase in OMBR reduces the water flux and consequently the membrane performance because of the decline in osmosis force by the accumulation of the solute in the bioreactor (Qiu & Ting, 2013). Overall however, the OMBR process showed great capability for contaminant removal despite the salinity build-up (Song et al., 2018).

### *6.2. Salinity build-up and bioenergy*

Despite some reports regarding adverse effects of salinity on biogas production, the generation of the biogas from saline wastewater in OMBRs is usually widely carried out. (Picos-Benítez et al., 2019) studied the salinity effect on biogas production in the fishing industry wastewater, and observed that the production of the biogas decreased to 64% when the salinity was increased from 0 to 20000 mg $L^{-1}$. (Lefebvre & Moletta, 2006) reported that more than 10000 mg $L^{-1}$ sodium concentrations in wastewater caused a severe inhibitory effect on methanogenesis during a long period. Similarly in another study, the inhibitory effects of OMBR salinity build-up on the growth of methanogenic bacteria, especially acetoclastic and hydrogenotrophic ones, were demonstrated, as shown by the reduction in the methanogenic bacteria strength due to competition from the sulphate reducing bacteria (Wu et al., 2017). Despite these findings, (Gu et al., 2015) indicated the potentially high resistance of the methanogenic bacteria against salinity, as methane production in an OMBR maintained a relatively stable condition after a long operation period. Therefore, methanogenic bacteria can develop adaptation to saline water provided sufficient time of transition is allowed.

### *6.3. Membrane fouling*

Membrane fouling is a common issue for all of the membrane-based water and wastewater treatment technologies, and can be defined as the sedimentation of dissolved or suspended solids on or within the membrane pores, resulting in the performance deterioration of the membranes (Wang et al., 2016). Fouling usually shows adverse effects on membrane lifespan, water flux, rejection abilities of the membranes and consequently, maintenance and operational expenditures of the membrane-based systems (Hosseinzadeh et al., 2020b; Wang et al., 2016). The term of membrane-foulants refers to all substances causing membrane fouling that can be EPS, SMP, sludge flocks, colloids, biopolymer clusters, DOM, and other inorganic and organic matter. A wide spectrum of studies has been carried out to identify the nature of these foulants (Lin et al., 2014). By virtue of less water flux and the hydraulic pressure deficiency, FO processes are usually regarded as low fouling membrane technology compared to the pressure-driven processes such as RO (Lutchmiah et al., 2014b). Nonetheless, fouling is one of the most important challenges of the FO processes, which is considerably further complicated in OMBRs (Wang et al., 2016). All of the membrane-foulants are generally categorized in four classes of colloidal, inorganic, organic and biological, while the concentration polarization (CP) is proposed as the main cause of fouling in membrane processes including FO. There are four categories of CP, i.e. concentrative internal CP, concentrative external CP, dilutive external CP and dilutive internal CP. The last one is the most effective factor in reducing the water flux in the FO process. There is a direct relationship between the proportion of the driving force (osmotic pressure) and the concentration of DS (Hosseinzadeh et al., 2020b). Therefore, the interactions between the CP and MLSS coupled with ingredients of the feed solution can increase the complexity of causation of fouling in OMBRs.

## 7. Future outlook

Regarding the obstacles facing OMBRs, most current research has been conducted to improve the quality of the membranes to reduce the back diffusion and increase the water flux, and to improve the quality of the DSs to augment the osmotic pressure and simplify the regeneration of the DS. Such research should be able to enhance the operating condition and performance of this emerging process. In addition, with respect to the capabilities of the OMBR, two general procedures should be given more consideration for OMBR process in future. Firstly, the hybridization and combination of OMBR with other processes can potentially take advantage of the salinity build-up more appropriately to produce more value-added products. Secondly, more research is needed for avoiding DS regeneration involving additional energy consumption; as an example, appropriate liquid fertilizers can be applied as DS, which when diluted, can be used for spraying to agricultural land.

## 8. Conclusions

This study critically reviewed recent research progress in microbial community and bioenergy production by OMBR processes. OMBR membranes have been extensively studied in relation to membrane fouling and reverse diffusion. Different DSs are being studied to improve process performance, augmentation of the osmotic pressure and direct reclamation of diluted DS as a fertilizer. Furthermore, the microbial communities in OMBRs contain *Firmicutes*, *Proteobacteria* and *Bacteroidetes* phyla, which are the most important ones for biogas production. By optimizing the operating conditions and bioenergy production processes, there is a considerable scope in using OMBR for efficient bioenergy production and wastewater treatment simultaneously.

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## References

1. Achilli, A., Cath, T.Y., Childress, A.E. 2010. Selection of inorganic-based draw solutions for forward osmosis applications. *Journal of Membrane Science*, **364**(1), 233-241.
2. Aftab, B., Khan, S.J., Maqbool, T., Hankins, N.P. 2017. Heavy metals removal by osmotic membrane bioreactor (OMBR) and their effect on sludge properties. *Desalination*, **403**, 117-127.
3. Akther, N., Phuntsho, S., Chen, Y., Ghaffour, N., Shon, H.K. 2019. Recent advances in nanomaterial-modified polyamide thin-film composite membranes for forward osmosis processes. *Journal of Membrane Science*, **584**, 20-45.
4. Alassi, A., Bañales, S., Ellabban, O., Adam, G., MacIver, C. 2019. HVDC Transmission: Technology Review, Market Trends and Future Outlook. *Renewable and Sustainable Energy Reviews*, **112**, 530-554.
5. Alejo, T., Arruebo, M., Carcelen, V., Monsalvo, V.M., Sebastian, V. 2017. Advances in draw solutes for forward osmosis: Hybrid organic-inorganic nanoparticles and conventional solutes. *Chemical Engineering Journal*, **309**, 738-752.
6. Ali, G., Bashir, M.K., Ali, H., Bashir, M.H. 2016. Utilization of rice husk and poultry wastes for renewable energy potential in Pakistan: An economic perspective. *Renewable and Sustainable Energy Reviews*, **61**, 25-29.

7. Alidadi, H., Dolatabadi, M., Davoudi, M., Barjasteh-Askari, F., Jamali-Behnam, F., Hosseinzadeh, A. 2018. Enhanced removal of tetracycline using modified sawdust: Optimization, isotherm, kinetics, and regeneration studies. *Process Safety and Environmental Protection*, **117**, 51-60.
8. Alturki, A., McDonald, J., Khan, S.J., Hai, F.I., Price, W.E., Nghiem, L.D. 2012. Performance of a novel osmotic membrane bioreactor (OMBR) system: Flux stability and removal of trace organics. *Bioresource Technology*, **113**, 201-206.
9. Appels, L., Lauwers, J., Degrève, J., Helsen, L., Lievens, B., Willems, K., Van Impe, J., Dewil, R. 2011. Anaerobic digestion in global bio-energy production: Potential and research challenges. *Renewable and Sustainable Energy Reviews*, **15**(9), 4295-4301.
10. Asghari, M., Afsari, M. 2017. Effect of Ethylene Oxide Functional Groups in PEBA-CNT Membranes on CO2/CH4 Mixed Gas Separation. *Journal of Membrane Science and Research*, **4**(1), 34-40.
11. Aslam, M., Ahmad, R., Yasin, M., Khan, A.L., Shahid, M.K., Hossain, S., Khan, Z., Jamil, F., Rafiq, S., Bilad, M.R., Kim, J., Kumar, G. 2018. Anaerobic membrane bioreactors for biohydrogen production: Recent developments, challenges and perspectives. *Bioresource Technology*, **269**, 452-464.
12. Bakonyi, P., Nemestóthy, N., Lankó, J., Rivera, I., Buitrón, G., Bélafi-Bakó, K. 2015. Simultaneous biohydrogen production and purification in a double-membrane bioreactor system. *international journal of hydrogen energy*, **40**(4), 1690-1697.
13. Bao, T., Damtie, M.M., Hosseinzadeh, A., Frost, R.L., Yu, Z.M., Jin, J., Wu, K. 2020. Catalytic degradation of P-chlorophenol by muscovite-supported nano zero valent iron composite: Synthesis, characterization, and mechanism studies. *Applied Clay Science*, **195**, 105735.
14. Blandin, G., Gautier, C., Sauchelli Toran, M., Monclús, H., Rodriguez-Roda, I., Comas, J. 2018. Retrofitting membrane bioreactor (MBR) into osmotic membrane bioreactor (OMBR): A pilot scale study. *Chemical Engineering Journal*, **339**, 268-277.
15. Bowden, K.S., Achilli, A., Childress, A.E. 2012. Organic ionic salt draw solutions for osmotic membrane bioreactors. *Bioresource technology*, **122**, 207-216.
16. Cai, Y. 2016. A critical review on draw solutes development for forward osmosis. *Desalination*, **391**, 16-29.
17. Cath, T.Y., Childress, A.E., Elimelech, M. 2006. Forward osmosis: principles, applications, and recent developments. *Journal of membrane science*, **281**(1-2), 70-87.
**18.** Chen, Z., Li, X., Hu, D., Cui, Y., Gu, F., Jia, F., Xiao, T., Su, H., Xu, J., Wang, H. 2018. Performance and methane fermentation characteristics of a pilot scale anaerobic membrane

bioreactor (AnMBR) for treating pharmaceutical wastewater containing m-cresol (MC) and iso-propyl alcohol (IPA). *Chemosphere*, **206**, 750-758.

19. Cheng, D., Ngo, H.H., Guo, W., Liu, Y., Chang, S.W., Nguyen, D.D., Nghiem, L.D., Zhou, J., Ni, B. 2018. Anaerobic membrane bioreactors for antibiotic wastewater treatment: performance and membrane fouling issues. *Bioresource Technology* **267,** 714-724.
20. Chung, K., Okabe, S. 2009. Continuous power generation and microbial community structure of the anode biofilms in a three-stage microbial fuel cell system. *Applied microbiology and biotechnology*, **83**(5), 965-977.
21. Damodara Kannan, A., Evans, P., Parameswaran, P. 2020. Long-term microbial community dynamics in a pilot-scale gas sparged anaerobic membrane bioreactor treating municipal wastewater under seasonal variations. *Bioresource Technology*, **310**, 123425.
22. Dumée, L., Lee, J., Sears, K., Tardy, B., Duke, M., Gray, S. 2013. Fabrication of thin film composite poly (amide)-carbon-nanotube supported membranes for enhanced performance in osmotically driven desalination systems. *Journal of membrane science*, **427**, 422-430.
23. Dutta, S., Nath, K. 2018. Prospect of ionic liquids and deep eutectic solvents as new generation draw solution in forward osmosis process. *Journal of Water Process Engineering*, **21**, 163-176.
24. Elimelech, M., Phillip, W.A. 2011. The future of seawater desalination: energy, technology, and the environment. *science*, **333**(6043), 712-717.
25. Ge, Q., Fu, F., Chung, T.-S. 2014. Ferric and cobaltous hydroacid complexes for forward osmosis (FO) processes. *Water Research*, **58**, 230-238.
26. Ge, Q., Ling, M., Chung, T.-S. 2013. Draw solutions for forward osmosis processes: developments, challenges, and prospects for the future. *Journal of membrane science*, **442**, 225-237.
27. Gu, Y., Chen, L., Ng, J.-W., Lee, C., Chang, V.W.C., Tang, C.Y. 2015. Development of anaerobic osmotic membrane bioreactor for low-strength wastewater treatment at mesophilic condition. *Journal of Membrane Science*, **490**, 197-208.
28. Gwak, G., Jung, B., Han, S., Hong, S. 2015. Evaluation of poly (aspartic acid sodium salt) as a draw solute for forward osmosis. *Water Research*, **80**, 294-305.
29. Hau, N.T., Chen, S.-S., Nguyen, N.C., Huang, K.Z., Ngo, H.H., Guo, W. 2014. Exploration of EDTA sodium salt as novel draw solution in forward osmosis process for dewatering of high nutrient sludge. *Journal of Membrane Science*, **455**, 305-311.
30. Ho, J., Sung, S. 2009. Anaerobic membrane bioreactor treatment of synthetic municipal wastewater at ambient temperature. *Water environment research*, **81**(9), 922-928.

31. Hosseinzadeh, A., Zhou, J.L., Altaee, A., Baziar, M., Li, D. 2020a. Effective modelling of hydrogen and energy recovery in microbial electrolysis cell by artificial neural network and adaptive network-based fuzzy inference system. *Bioresource Technology*, **316**, 123967.
32. Hosseinzadeh, A., Zhou, J.L., Altaee, A., Baziar, M., Li, X. 2020b. Modeling water flux in osmotic membrane bioreactor by adaptive network-based fuzzy inference system and artificial neural network. *Bioresource Technology*, **310**, 123391.
33. Hou, D., Lu, L., Ren, Z.J. 2016. Microbial fuel cells and osmotic membrane bioreactors have mutual benefits for wastewater treatment and energy production. *Water Research*, **98**, 183-189.
34. Hou, D., Lu, L., Sun, D., Ge, Z., Huang, X., Cath, T.Y., Ren, Z.J. 2017. Microbial electrochemical nutrient recovery in anaerobic osmotic membrane bioreactors. *Water Research*, **114**, 181-188.
35. Huang, Y., Surawski, N.C., Organ, B., Zhou, J.L., Tang, O.H.H., Chan, E.F.C. 2019. Fuel consumption and emission performance under real driving: comparison between hybrid and conventional vehicles. *Science of the Total Environment* **659,** 275-282.
36. Huang, Z., Ong, S.L., Ng, H.Y. 2011. Submerged anaerobic membrane bioreactor for low-strength wastewater treatment: effect of HRT and SRT on treatment performance and membrane fouling. *Water research*, **45**(2), 705-713.
37. Hung, W.-S., An, Q.-F., De Guzman, M., Lin, H.-Y., Huang, S.-H., Liu, W.-R., Hu, C.-C., Lee, K.-R., Lai, J.-Y. 2014. Pressure-assisted self-assembly technique for fabricating composite membranes consisting of highly ordered selective laminate layers of amphiphilic graphene oxide. *Carbon*, **68**, 670-677.
38. Infantes, D., González del Campo, A., Villaseñor, J., Fernández, F.J. 2011. Influence of pH, temperature and volatile fatty acids on hydrogen production by acidogenic fermentation. *International Journal of Hydrogen Energy*, **36**(24), 15595-15601.
39. Ismail, A., Padaki, M., Hilal, N., Matsuura, T., Lau, W. 2015. Thin film composite membrane—Recent development and future potential. *Desalination*, **356**, 140-148.
40. Jia, X., Wang, Y., Ren, L., Li, M., Tang, R., Jiang, Y., Hou, J. 2019. Early warning indicators and microbial community dynamics during unstable stages of continuous hydrogen production from food wastes by thermophilic dark fermentation. *International Journal of Hydrogen Energy*, **44**(57), 30000-30013.
41. Jie, W., Peng, Y., Ren, N., Li, B. 2014. Volatile fatty acids (VFAs) accumulation and microbial community structure of excess sludge (ES) at different pHs. *Bioresource Technology*, **152**, 124-129.

42. Kardani, R., Asghari, M., Hamedani, N.F., Afsari, M. 2020. Mesoporous copper zinc bimetallic imidazolate MOF as nanofiller to improve gas separation performance of PEBA-based membranes. *Journal of Industrial and Engineering Chemistry*, **83**, 100-110.
43. Khan, M.A., Ngo, H.H., Guo, W., Chang, S.W., Nguyen, D.D., Varjani, S., Liu, Y., Deng, L., Cheng, C. 2019. Selective production of volatile fatty acids at different pH in an anaerobic membrane bioreactor. *Bioresource Technology*, **283**, 120-128.
44. Khan, M.A., Ngo, H.H., Guo, W.S., Liu, Y., Nghiem, L.D., Hai, F.I., Deng, L.J., Wang, J., Wu, Y. 2016. Optimization of process parameters for production of volatile fatty acid, biohydrogen and methane from anaerobic digestion. *Bioresource Technology*, **219**, 738-748.
45. Kheirieh, S., Asghari, M., Afsari, M. 2018. Application and modification of polysulfone membranes. *Reviews in Chemical Engineering*, **34**(5), 657-693.
46. Kresge, C., Leonowicz, M., Roth, W.J., Vartuli, J., Beck, J. 1992. Ordered mesoporous molecular sieves synthesized by a liquid-crystal template mechanism. *nature*, **359**(6397), 710.
47. Kunacheva, C., Soh, Y.N.A., Trzcinski, A.P., Stuckey, D.C. 2017. Soluble microbial products (SMPs) in the effluent from a submerged anaerobic membrane bioreactor (SAMBR) under different HRTs and transient loading conditions. *Chemical Engineering Journal*, **311**, 72-81.
48. Lalia, B.S., Kochkodan, V., Hashaikeh, R., Hilal, N. 2013. A review on membrane fabrication: Structure, properties and performance relationship. *Desalination*, **326**, 77-95.
49. Laothanachareon, T., Kanchanasuta, S., Mhuanthong, W., Phalakornkule, C., Pisutpaisal, N., Champreda, V. 2014. Analysis of microbial community adaptation in mesophilic hydrogen fermentation from food waste by tagged 16S rRNA gene pyrosequencing. *Journal of Environmental Management*, **144**, 143-151.
50. Lau, W., Ismail, A., Misdan, N., Kassim, M. 2012. A recent progress in thin film composite membrane: a review. *Desalination*, **287**, 190-199.
51. Lay, J.-J., Li, Y.-Y., Noike, T. 1997. Influences of pH and moisture content on the methane production in high-solids sludge digestion. *Water Research*, **31**(6), 1518-1524.
52. Lee, S., Boo, C., Elimelech, M., Hong, S. 2010. Comparison of fouling behavior in forward osmosis (FO) and reverse osmosis (RO). *Journal of Membrane Science*, **365**(1), 34-39.
53. Lee, W.S., Chua, A.S.M., Yeoh, H.K., Ngoh, G.C. 2014. A review of the production and applications of waste-derived volatile fatty acids. *Chemical Engineering Journal*, **235**, 83-99.
54. Lefebvre, O., Moletta, R. 2006. Treatment of organic pollution in industrial saline wastewater: A literature review. *Water Research*, **40**(20), 3671-3682.
55. Li, J.-Y., Ni, Z.-Y., Zhou, Z.-Y., Hu, Y.-X., Xu, X.-H., Cheng, L.-H. 2018. Membrane fouling of forward osmosis in dewatering of soluble algal products: Comparison of TFC and CTA membranes. *Journal of Membrane Science*, **552**, 213-221.

56. Li, Y., Hu, Q., Chen, C.-H., Wang, X.-L., Gao, D.-W. 2017. Performance and microbial community structure in an integrated anaerobic fluidized-bed membrane bioreactor treating synthetic benzothiazole contaminated wastewater. *Bioresource Technology*, **236**, 1-10.
57. Lin, C.-Y., Lay, C.-H., Sen, B., Chu, C.-Y., Kumar, G., Chen, C.-C., Chang, J.-S. 2012. Fermentative hydrogen production from wastewaters: A review and prognosis. *International Journal of Hydrogen Energy*, **37**(20), 15632-15642.
58. Lin, C.-Y., Leu, H.-J., Lee, K.-H. 2016. Hydrogen production from beverage wastewater via dark fermentation and room-temperature methane reforming. *International Journal of Hydrogen Energy*, **41**(46), 21736-21746.
59. Lin, C.-Y., Wu, C.-C., Wu, J.-H., Chang, F.-Y. 2008. Effect of cultivation temperature on fermentative hydrogen production from xylose by a mixed culture. *Biomass and Bioenergy*, **32**(12), 1109-1115.
60. Lin, H., Zhang, M., Wang, F., Meng, F., Liao, B.-Q., Hong, H., Chen, J., Gao, W. 2014. A critical review of extracellular polymeric substances (EPSs) in membrane bioreactors: Characteristics, roles in membrane fouling and control strategies. *Journal of Membrane Science*, **460**, 110-125.
61. Liu, C., Ren, L., Yan, B., Luo, L., Zhang, J., Awasthi, M.K. 2021. Electron transfer and mechanism of energy production among syntrophic bacteria during acidogenic fermentation: A review. *Bioresource Technology*, **323**, 124637.
62. Liu, J., Liu, L., Gao, B., Yang, F., Crittenden, J., Ren, N. 2014. Integration of microbial fuel cell with independent membrane cathode bioreactor for power generation, membrane fouling mitigation and wastewater treatment. *international journal of hydrogen energy*, **39**(31), 17865-17872.
63. Lu, L., Ren, N., Zhao, X., Wang, H., Wu, D., Xing, D. 2011. Hydrogen production, methanogen inhibition and microbial community structures in psychrophilic single-chamber microbial electrolysis cells. *Energy & Environmental Science*, **4**(4), 1329-1336.
64. Lu, L., Xing, D., Ren, N., Logan, B.E. 2012. Syntrophic interactions drive the hydrogen production from glucose at low temperature in microbial electrolysis cells. *Bioresource Technology*, **124**, 68-76.
65. Luján-Facundo, M.J., Fernández-Navarro, J., Alonso-Molina, J.L., Amorós-Muñoz, I., Moreno, Y., Mendoza-Roca, J.A., Pastor-Alcañiz, L. 2018. The role of salinity on the changes of the biomass characteristics and on the performance of an OMBR treating tannery wastewater. *Water Research*, **142**, 129-137.
66. Luo, W., Hai, F.I., Price, W.E., Guo, W., Ngo, H.H., Yamamoto, K., Nghiem, L.D. 2016a. Phosphorus and water recovery by a novel osmotic membrane bioreactor–reverse osmosis system. *Bioresource Technology*, **200**, 297-304.

67. Luo, W., Phan, H.V., Hai, F.I., Price, W.E., Guo, W., Ngo, H.H., Yamamoto, K., Nghiem, L.D. 2016b. Effects of salinity build-up on the performance and bacterial community structure of a membrane bioreactor. *Bioresource Technology*, **200**, 305-310.
68. Luo, W., Phan, H.V., Xie, M., Hai, F.I., Price, W.E., Elimelech, M., Nghiem, L.D. 2017. Osmotic versus conventional membrane bioreactors integrated with reverse osmosis for water reuse: Biological stability, membrane fouling, and contaminant removal. *Water Research,* **109**, 122-134.
69. Luo, W., Xie, M., Song, X., Guo, W., Ngo, H.H., Zhou, J.L., Nghiem, L.D. 2018. Biomimetic aquaporin membranes for osmotic membrane bioreactors: Membrane performance and contaminant removal. *Bioresource Technology* **249,** 62-68.
70. Lutchmiah, K., Lauber, L., Roest, K., Harmsen, D.J., Post, J.W., Rietveld, L.C., van Lier, J.B., Cornelissen, E.R. 2014a. Zwitterions as alternative draw solutions in forward osmosis for application in wastewater reclamation. *Journal of membrane science,* **460**, 82-90.
71. Lutchmiah, K., Verliefde, A.R.D., Roest, K., Rietveld, L.C., Cornelissen, E.R. 2014b. Forward osmosis for application in wastewater treatment: A review. *Water Research,* **58**, 179-197.
72. Ma, H., Su, H. 2019. Effect of temperature on the fermentation of starch by two high efficient H2 producers. *Renewable Energy*, **138**, 964-970.
73. Ma, N., Wei, J., Liao, R., Tang, C.Y. 2012. Zeolite-polyamide thin film nanocomposite membranes: Towards enhanced performance for forward osmosis. *Journal of Membrane Science*, **405-406**, 149-157.
74. McLeary, E., Jansen, J., Kapteijn, F. 2006. Zeolite based films, membranes and membrane reactors: Progress and prospects. *Microporous and mesoporous materials*, **90**(1-3), 198-220.
75. MingáLing, M. 2011. Facile synthesis of thermosensitive magnetic nanoparticles as “smart” draw solutes in forward osmosis. *Chemical communications*, **47**(38), 10788-10790.
76. Mukherjee, D., Bhattacharya, P., Jana, A., Bhattacharya, S., Sarkar, S., Ghosh, S., Majumdar, S., Swarnakar, S. 2018. Synthesis of ceramic ultrafiltration membrane and application in membrane bioreactor process for pesticide remediation from wastewater. *Process Safety and Environmental Protection*, **116**, 22-33.
77. Mutamim, N.S.A., Noor, Z.Z., Hassan, M.A.A., Olsson, G. 2012. Application of membrane bioreactor technology in treating high strength industrial wastewater: a performance review. *Desalination*, **305**, 1-11.
78. Nakasaki, K., Kwon, S.H., Takemoto, Y. 2015. An interesting correlation between methane production rates and archaea cell density during anaerobic digestion with increasing organic loading. *Biomass and Bioenergy*, **78**, 17-24.

79. Nakayama, D., Mok, Y., Noh, M., Park, J., Kang, S., Lee, Y. 2014. Lower critical solution temperature (LCST) phase separation of glycol ethers for forward osmotic control. *Physical Chemistry Chemical Physics*, **16**(11), 5319-5325.
80. Ng, K.K., Shi, X., Ong, S.L., Lin, C.-F., Ng, H.Y. 2016. An innovative of aerobic bio-entrapped salt marsh sediment membrane reactor for the treatment of high-saline pharmaceutical wastewater. *Chemical Engineering Journal*, **295**, 317-325.
81. Nguyen, H.T., Chen, S.-S., Nguyen, N.C., Ngo, H.H., Guo, W., Li, C.-W. 2015. Exploring an innovative surfactant and phosphate-based draw solution for forward osmosis desalination. *Journal of Membrane Science*, **489**, 212-219.
82. Niu, C., Pan, Y., Lu, X., Wang, S., Zhang, Z., Zheng, C., Tan, Y., Zhen, G., Zhao, Y., Li, Y.-Y. 2020. Mesophilic anaerobic digestion of thermally hydrolyzed sludge in anaerobic membrane bioreactor: Long-term performance, microbial community dynamics and membrane fouling mitigation. *Journal of Membrane Science*, **612**, 118264.
83. Ong, R.C., Chung, T.-S. 2012. Fabrication and positron annihilation spectroscopy (PAS) characterization of cellulose triacetate membranes for forward osmosis. *Journal of Membrane Science*, **394-395**, 230-240.
84. Park, M.J., Phuntsho, S., He, T., Nisola, G.M., Tijing, L.D., Li, X.-M., Chen, G., Chung, W.-J., Shon, H.K. 2015. Graphene oxide incorporated polysulfone substrate for the fabrication of flat-sheet thin-film composite forward osmosis membranes. *Journal of Membrane Science*, **493**, 496-507.
85. Pathak, N., Chekli, L., Wang, J., Kim, Y., Phuntsho, S., Li, S., Ghaffour, N., Leiknes, T., Shon, H. 2017. Performance of a novel baffled osmotic membrane bioreactor-microfiltration hybrid system under continuous operation for simultaneous nutrient removal and mitigation of brine discharge. *Bioresource Technology*, **240**, 50-58.
86. Pathak, N., Li, S., Kim, Y., Chekli, L., Phuntsho, S., Jang, A., Ghaffour, N., Leiknes, T., Shon, H.K. 2018. Assessing the removal of organic micropollutants by a novel baffled osmotic membrane bioreactor-microfiltration hybrid system. *Bioresource Technology*, **262**, 98-106.
87. Pathak, N., Phuntsho, S., Tran, V.H., Johir, M.A.H., Ghaffour, N., Leiknes, T., Fujioka, T., Shon, H.K. 2020. Simultaneous nitrification-denitrification using baffled osmotic membrane bioreactor-microfiltration hybrid system at different oxic-anoxic conditions for wastewater treatment. *Journal of Environmental Management*, **253**, 109685.
88. Phuntsho, S., Hong, S., Elimelech, M., Shon, H.K. 2014. Osmotic equilibrium in the forward osmosis process: Modelling, experiments and implications for process performance. *Journal of membrane science*, **453**, 240-252.

89. Picos-Benítez, A.R., Peralta-Hernández, J.M., López-Hincapié, J.D., Rodríguez-García, A. 2019. Biogas production from saline wastewater of the evisceration process of the fish processing industry. *Journal of Water Process Engineering*, **32**, 100933.
90. Pretel, R., Robles, A., Ruano, M., Seco, A., Ferrer, J. 2016. Economic and environmental sustainability of submerged anaerobic MBR-based (AnMBR-based) technology as compared to aerobic-based technologies for moderate-/high-loaded urban wastewater treatment. *Journal of Environmental Management*, **166**, 45-54.
91. Qiu, G., Ting, Y.-P. 2014. Direct phosphorus recovery from municipal wastewater via osmotic membrane bioreactor (OMBR) for wastewater treatment. *Bioresource Technology*, **170**, 221-229.
92. Qiu, G., Ting, Y.-P. 2013. Osmotic membrane bioreactor for wastewater treatment and the effect of salt accumulation on system performance and microbial community dynamics. *Bioresource Technology*, **150**, 287-297.
93. Rafieenia, R., Pivato, A., Campanaro, S., Treu, L., Schievano, A., Lavagnolo, M.C. 2019. Study of microbial dynamics during optimization of hydrogen production from food waste by using LCFA-rich agent. *Bioresource Technology Reports*, **5**, 157-163.
94. Ruggeri, B., Tommasi, T., Sanfilippo, S. 2015. *BioH2 & BioCH4 through anaerobic digestion: from research to full-scale applications*. Springer.
95. Seib, M.D., Berg, K.J., Zitomer, D.H. 2016. Influent wastewater microbiota and temperature influence anaerobic membrane bioreactor microbial community. *Bioresource Technology*, **216**, 446-452.
96. Skouteris, G., Rodriguez-Garcia, G., Reinecke, S.F., Hampel, U. 2020. The use of pure oxygen for aeration in aerobic wastewater treatment: A review of its potential and limitations. *Bioresource Technology*, **312**, 123595.
97. Slezak, R., Grzelak, J., Krzystek, L., Ledakowicz, S. 2017. The effect of initial organic load of the kitchen waste on the production of VFA and H2 in dark fermentation. *Waste Management*, **68**, 610-617.
98. Song, X., Xie, M., Li, Y., Li, G., Luo, W. 2018. Salinity build-up in osmotic membrane bioreactors: Causes, impacts, and potential cures. *Bioresource Technology*, **257**, 301-310.
99. Song, Y.-X., Liao, Q., Yu, C., Xiao, R., Tang, C.-J., Chai, L.-Y., Duan, C.-S. 2017. Physicochemical and microbial properties of settled and floating anammox granules in upflow reactor. *Biochemical Engineering Journal*, **123**, 75-85.
100. Srinivasa Raghavan, D.S., Qiu, G., Ting, Y.-P. 2018. Fate and removal of selected antibiotics in an osmotic membrane bioreactor. *Chemical Engineering Journal*, **334**, 198-205.

101. Stone, M.L., Rae, C., Stewart, F.F., Wilson, A.D. 2013. Switchable polarity solvents as draw solutes for forward osmosis. *Desalination*, **312**, 124-129.
102. Su, J., Chung, T.-S., Helmer, B.J., de Wit, J.S. 2012. Enhanced double-skinned FO membranes with inner dense layer for wastewater treatment and macromolecule recycle using Sucrose as draw solute. *Journal of Membrane Science*, **396**, 92-100.
103. Sun, C., Liu, F., Song, Z., Wang, J., Li, Y., Pan, Y., Sheng, T., Li, L. 2019a. Feasibility of dry anaerobic digestion of beer lees for methane production and biochar enhanced performance at mesophilic and thermophilic temperature. *Bioresource Technology*, **276**, 65-73.
104. Sun, C., Xia, A., Liao, Q., Fu, Q., Huang, Y., Zhu, X. 2019b. Life-cycle assessment of biohythane production via two-stage anaerobic fermentation from microalgae and food waste. *Renewable and Sustainable Energy Reviews*, **112**, 395-410.
105. Sun, Y., Tian, J., Song, L., Gao, S., Shi, W., Cui, F. 2018. Dynamic changes of the fouling layer in forward osmosis based membrane processes for municipal wastewater treatment. *Journal of Membrane Science*, **549**, 523-532.
106. Sun, Y., Tian, J., Zhao, Z., Shi, W., Liu, D., Cui, F. 2016. Membrane fouling of forward osmosis (FO) membrane for municipal wastewater treatment: A comparison between direct FO and OMBR. *Water Research*, **104**, 330-339.
107. Talvitie, J., Mikola, A., Koistinen, A., Setälä, O. 2017. Solutions to microplastic pollution–Removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. *Water research*, **123**, 401-407.
108. Tan, X., Acquah, I., Liu, H., Li, W., Tan, S. 2019. A critical review on saline wastewater treatment by membrane bioreactor (MBR) from a microbial perspective. *Chemosphere*, **220**, 1150-1162.
109. Tao, Z., Wang, D., Yao, F., Huang, X., Wu, Y., Du, M., Chen, Z., An, H., Li, X., Yang, Q. 2020. The effects of thiosulfinates on methane production from anaerobic co-digestion of waste activated sludge and food waste and mitigate method. *Journal of Hazardous Materials*, **384**, 121363.
110. Tian, E., Hu, C., Qin, Y., Ren, Y., Wang, X., Wang, X., Xiao, P., Yang, X. 2015a. A study of poly (sodium 4-styrenesulfonate) as draw solute in forward osmosis. *Desalination*, **360**, 130-137.
111. Tian, M., Wang, Y.-N., Wang, R. 2015b. Synthesis and characterization of novel high-performance thin film nanocomposite (TFN) FO membranes with nanofibrous substrate reinforced by functionalized carbon nanotubes. *Desalination*, **370**, 79-86.
112. Tian, M., Wang, Y.-N., Wang, R., Fane, A.G. 2017. Synthesis and characterization of thin film nanocomposite forward osmosis membranes supported by silica nanoparticle incorporated nanofibrous substrate. *Desalination*, **401**, 142-150.

113. Tian, Y., Ji, C., Wang, K., Le-Clech, P. 2014. Assessment of an anaerobic membrane bio-electrochemical reactor (AnMBER) for wastewater treatment and energy recovery. *Journal of membrane science*, **450**, 242-248.
114. Tomczak, W., Ferrasse, J.-H., Giudici-Orticoni, M.-T., Soric, A. 2018. Effect of hydraulic retention time on a continuous biohydrogen production in a packed bed biofilm reactor with recirculation flow of the liquid phase. *International Journal of Hydrogen Energy*, **43**(41), 18883-18895.
115. Tran, T.T.D., Smith, A.D. 2017. fEvaluation of renewable energy technologies and their potential for technical integration and cost-effective use within the U.S. energy sector. *Renewable and Sustainable Energy Reviews*, **80**, 1372-1388.
116. Trung, N.Q., Van Nhan, L., Thao, P.T.P., Giang, L.T. 2017. Novel draw solutes of iron complexes easier recovery in forward osmosis process. *Journal of water reuse and Desalination*, **8**(2), 244-250.
117. Viet, N.D., Cho, J., Yoon, Y., Jang, A. 2019. Enhancing the removal efficiency of osmotic membrane bioreactors: A comprehensive review of influencing parameters and hybrid configurations. *Chemosphere*, **236**, 124363.
118. Volpin, F., Yu, H., Cho, J., Lee, C., Phuntsho, S., Ghaffour, N., Vrouwenvelder, J.S., Shon, H.K. 2019. Human urine as a forward osmosis draw solution for the application of microalgae dewatering. *Journal of Hazardous Materials*, **378**, 120724.
119. Wang, H., Wang, X., Meng, F., Li, X., Ren, Y., She, Q. 2019. Effect of driving force on the performance of anaerobic osmotic membrane bioreactors: New insight into enhancing water flux of FO membrane via controlling driving force in a two-stage pattern. *Journal of Membrane Science*, **569**, 41-47.
120. Wang, K.M., Jefferson, B., Soares, A., McAdam, E.J. 2018a. Sustaining membrane permeability during unsteady-state operation of anaerobic membrane bioreactors for municipal wastewater treatment following peak-flow. *Journal of membrane science*, **564**, 289-297.
121. Wang, W., Zhang, Y., Esparra-Alvarado, M., Wang, X., Yang, H., Xie, Y. 2014a. Effects of pH and temperature on forward osmosis membrane flux using rainwater as the makeup for cooling water dilution. *Desalination*, **351**, 70-76.
122. Wang, X., Chang, V.W.C., Tang, C.Y. 2016. Osmotic membrane bioreactor (OMBR) technology for wastewater treatment and reclamation: Advances, challenges, and prospects for the future. *Journal of Membrane Science*, **504**, 113-132.
123. Wang, X., Chen, Y., Yuan, B., Li, X., Ren, Y. 2014b. Impacts of sludge retention time on sludge characteristics and membrane fouling in a submerged osmotic membrane bioreactor. *Bioresource Technology*, **161**, 340-347.

124. Wang, X., Hu, T., Wang, Z., Li, X., Ren, Y. 2017a. Permeability recovery of fouled forward osmosis membranes by chemical cleaning during a long-term operation of anaerobic osmotic membrane bioreactors treating low-strength wastewater. *Water Research*, **123**, 505-512.

125. Wang, X., Zhang, J., Chang, V.W.C., She, Q., Tang, C.Y. 2018. Removal of cytostatic drugs from wastewater by an anaerobic osmotic membrane bioreactor. *Chemical Engineering Journal*, **339**, 153-161.

126. Wang, X., Zhao, Y., Li, X., Ren, Y. 2017b. Performance evaluation of a microfiltration-osmotic membrane bioreactor (MF-OMBR) during removing silver nanoparticles from simulated wastewater. *Chemical Engineering Journal*, **313**, 171-178.

127. Widjojo, N., Chung, T.-S., Weber, M., Maletzko, C., Warzelhan, V. 2013. A sulfonated polyphenylenesulfone (sPPSU) as the supporting substrate in thin film composite (TFC) membranes with enhanced performance for forward osmosis (FO). *Chemical engineering journal*, **220**, 15-23.

128. Wu, H., Gao, J., Yang, D., Zhou, Q., Liu, W. 2010. Alkaline fermentation of primary sludge for short-chain fatty acids accumulation and mechanism. *Chemical Engineering Journal*, **160**(1), 1-7.

129. Wu, Y., Wang, X., Tay, M.Q.X., Oh, S., Yang, L., Tang, C., Cao, B. 2017. Metagenomic insights into the influence of salinity and cytostatic drugs on the composition and functional genes of microbial community in forward osmosis anaerobic membrane bioreactors. *Chemical Engineering Journal*, **326**, 462-469.

130. Wu, Y., Zhang, T., Gao, R., Wu, C. 2021. Portfolio planning of renewable energy with energy storage technologies for different applications from electricity grid. *Applied Energy*, **287**, 116562.

131. Xiao, D., Tang, C.Y., Zhang, J., Lay, W.C.L., Wang, R., Fane, A.G. 2011. Modeling salt accumulation in osmotic membrane bioreactors: Implications for FO membrane selection and system operation. *Journal of Membrane Science*, **366**(1), 314-324.

132. Xiong, S., Zuo, J., Ma, Y.G., Liu, L., Wu, H., Wang, Y. 2016a. Novel thin film composite forward osmosis membrane of enhanced water flux and anti-fouling property with N-[3-(trimethoxysilyl) propyl] ethylenediamine incorporated. *Journal of Membrane Science*, **520**, 400-414.

133. Xiong, Y., Harb, M., Hong, P.-Y. 2016b. Characterization of biofoulants illustrates different membrane fouling mechanisms for aerobic and anaerobic membrane bioreactors. *Separation and Purification Technology*, **157**, 192-202.

134. Xu, W., Chen, Q., Ge, Q. 2017. Recent advances in forward osmosis (FO) membrane: Chemical modifications on membranes for FO processes. *Desalination*, **419**, 101-116.

135. Yang, G., Wang, J. 2019. Changes in microbial community structure during dark fermentative hydrogen production. *International Journal of Hydrogen Energy*, **44**(47), 25542-25550.
136. Yang, G., Wang, J., Zhang, H., Jia, H., Zhang, Y., Gao, F. 2019. Applying bio-electric field of microbial fuel cell-upflow anaerobic sludge blanket reactor catalyzed blast furnace dusting ash for promoting anaerobic digestion. *Water Research*, **149**, 215-224.
137. Yang, Y.-L., Wu, Y., Lu, Y.-X., Cai, Y., He, Z., Yang, X.-L., Song, H.-L. 2021. A comprehensive review of nutrient-energy-water-solute recovery by hybrid osmotic membrane bioreactors. *Bioresource Technology*, **320**, 124300.
138. Yao, M., Duan, L., Wei, J., Qian, F., Hermanowicz, S.W. 2020. Carbamazepine removal from wastewater and the degradation mechanism in a submerged forward osmotic membrane bioreactor. *Bioresource Technology*, **314**, 123732.
139. Yin, Y., Wang, J. 2016. Changes in microbial community during biohydrogen production using gamma irradiated sludge as inoculum. *Bioresource Technology*, **200**, 217-222.
140. Yurtsever, A., Calimlioglu, B., Sahinkaya, E. 2017. Impact of SRT on the efficiency and microbial community of sequential anaerobic and aerobic membrane bioreactors for the treatment of textile industry wastewater. *Chemical Engineering Journal*, **314**, 378-387.
141. Yurtsever, A., Sahinkaya, E., Aktaş, Ö., Uçar, D., Çınar, Ö., Wang, Z. 2015. Performances of anaerobic and aerobic membrane bioreactors for the treatment of synthetic textile wastewater. *Bioresource Technology*, **192**, 564-573.
142. Zainal, B.S., Danaee, M., Mohd, N.S., Ibrahim, S. 2020. Effects of temperature and dark fermentation effluent on biomethane production in a two-stage up-flow anaerobic sludge fixed-film (UASFF) bioreactor. *Fuel*, **263**, 116729.
143. Zhang, S., Wang, K.Y., Chung, T.-S., Chen, H., Jean, Y., Amy, G. 2010. Well-constructed cellulose acetate membranes for forward osmosis: minimized internal concentration polarization with an ultra-thin selective layer. *Journal of Membrane Science*, **360**(1-2), 522-535.
144. Zhang, W., Zhang, L., Li, A. 2015. Enhanced anaerobic digestion of food waste by trace metal elements supplementation and reduced metals dosage by green chelating agent [S, S]-EDDS via improving metals bioavailability. *Water Research*, **84**, 266-277.
145. Zhao, J., Li, Y., Pan, S., Tu, Q., Zhu, H. 2019. Performance of a forward osmotic membrane bioreactor for anaerobic digestion of waste sludge with increasing solid concentration. *Journal of Environmental Management*, **246**, 239-246.
146. Zhao, X., Li, J., Liu, C. 2017. A novel TFC-type FO membrane with inserted sublayer of carbon nanotube networks exhibiting the improved separation performance. *Desalination*, **413**, 176-183.
147. Zhen, G., Pan, Y., Lu, X., Li, Y.-Y., Zhang, Z., Niu, C., Kumar, G., Kobayashi, T., Zhao, Y., Xu, K. 2019. Anaerobic membrane bioreactor towards biowaste biorefinery and chemical energy

harvest: Recent progress, membrane fouling and future perspectives. *Renewable and Sustainable Energy Reviews*, **115**, 109392.

148. Zhu, W., Wang, X., She, Q., Li, X., Ren, Y. 2018. Osmotic membrane bioreactors assisted with microfiltration membrane for salinity control (MF-OMBR) operating at high sludge concentrations: Performance and implications. *Chemical Engineering Journal*, **337**, 576-583.