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Feasibility and application of membrane aerated biofilm reactors for industrial wastewater treatment

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ABSTRACT

Membrane aerated biofilm reactors (MABRs) have emerged as a promising technology for wastewater treatment, offering significant advantages over conventional activated sludge (CAS) systems. Over the past decades, membrane processes have revolutionized municipal water treatment with membrane bioreactors (MBRs) becoming a widely accepted process for municipal and then industrial wastewater (IW) treatment. By the same token, MABR technologies were initially applied to municipal wastewater; however, their application in industrial settings is still emerging. Despite the promise of MABRs due to the biofilm's tolerance to IW toxins, there is a lack of information on their industrial applications. Therefore, this paper critically reviews the feasibility and application of MABRs for IW treatment, including pharmaceutical, chemical, refinery, petrochemical, oilfield, landfill leachate and other complex industrial waters. Three existing technology vendors with full-scale experience were compared; however, additional providers with innovative designs may provide step-changes in performance. Key outcomes highlight the effectiveness of MABRs in reducing carbon, nitrogen, and xenobiotics from high-strength IWs at bench and pilot scales. Critical factors influencing MABR performance, such as biofilm thickness (BT) were correlated to organics and nitrogen removal efficiency in industrial applications. Review of advances in MABR modeling techniques showed that current models lack the needed resolution for large and dynamic industrial systems. Additionally, the review compares municipal and industrial applications of MABRs, emphasizing the unique challenges and innovations required for their adoption in IW treatment. Overall, the MABR process was found to be feasible for industrial applications with pilot and/or demonstration-scale testing being necessary to further optimize process performance.

1. Introduction

Industrial freshwater use in the US, spanning food, mining, paper, steel, chemical, and petroleum refining industries, is estimated at 14,800 million gallons per day (Dieter et al., 2018). Wastewater from these processes may contain carcinogenic and harmful pollutants (Jafarinejad and Jiang, 2019; Yasasve et al., 2022). Stricter environmental regulations, along with circular economy and climate change concerns, necessitate innovative extensive water treatment solutions while remaining intensive in application (Ramanathan and Feng, 2009; Gherghel et al., 2019; Smol et al., 2020; Dutta *et al.,* 2021; Al-Maas et al., 2022; Fox, 2022; Tóth et al., 2022). Biological treatment, a cost-effective technology, is widely used in industrial facilities for biological oxidation of dissolved chemicals, typically through CAS systems (Jafarinejad and Jiang, 2019; Olajire, 2020). CAS configuration include an aeration basin with suspended bacteria (activated sludge) and a gravity clarifier (Metcalf and Eddy, 2003). While effective in removing toxic pollutants, these systems are energy-intensive and may release nitrous oxide (N_2O) and other greenhouse gases due to excessive mixing and limited oxygen transfer efficiency (OTE) (~10%) (Conthe et al., 2019; Duan et al., 2021; He et al., 2021). Additionally, industrial CAS units are not designed for nitrogen removal, and upgrades to denitrifying configurations are costly (Faber, 2019; Ishak et al., 2012). Membrane aerated biofilm reactors (MABRs) afford a significant step-change over CAS. The key advantages of MABRs include (Côté et al., 2015; He et al., 2021; Houweling et al., 2017; Kinh et al., 2017a; Lu et al., 2021; Syron and Casey, 2012; Uri-Carreño et al., 2024; Veleva et al., 2022):

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- **Energy efficiency**: higher OTEs up to 80% reduce energy costs;
- **Lower emissions**: bubbleless systems reduce the release of air pollutants and greenhouse gases (GHGs);
- **Simultaneous processes**: enable nitrification, denitrification, and carbon removal in one bioreactor;
- **Compactness**: requires a smaller footprint compared to conventional biological systems; and
- **Retrofit capabilities**: can be quickly integrated into existing CAS systems for bioprocess intensification

MABRs use gas-permeable hollow fiber (HF) or spiral wound membranes to grow biofilm and supply air and O_2. Oxygen diffuses from the lumen or air contact side, while the substrate is supplied from the shell or water contact side, creating a counter-diffusional process (Nerenberg, 2016). This setup allows for efficient removal of dissolved carbon and ammonia, maintaining a bubbleless, anoxic environment conducive to nitrogen removal. Simultaneous nitrification and denitrification (SND) occurs as nitrifying bacteria convert ammonia to nitrite (NO_2^-) and nitrate (NO_3^-), and denitrifying bacteria convert these nitrogen oxides to nitrogen gas (N_2) (Metcalf and Eddy, 2003). Stratification in microbial community structure, with aerobic autotrophs and heterotrophs near the membrane and anoxic heterotrophs on the outer layer (Fig 1), enhances functional stability against shock loads and inhibitory chemicals typical of industrial wastewaters (IWs) (Martin and Nerenberg, 2012; Nerenberg, 2016; Rittmann et al., 2004; Syron and Casey, 2008a; Waheed et al., 2013; Wobus and Röske, 2000; Zhou et al., 2020). Over the past decades, membrane processes like ultrafiltration (UF), and reverse osmosis (RO) have revolutionized municipal water treatment (Jalab et al., 2019), and led to UF-based membrane bioreactors (MBR) for municipal wastewater treatment compliance and recycling projects (Adham et al., 2018; Hirani et al., 2010). Similarly, MABRs technologies have been initially applied to municipal wastewater (Corsino and Torregrossa, 2022; Guglielmi et al., 2020; Heffernan, 2024; Heffernan et al., 2017; Kunetz et al., 2016; Peeters and McMains, 2023; Uri-Carreño et al., 2021). However, industrial MABR applications are still emerging, despite showing significant promise due to the biofilm's tolerance to industrial toxins, salinity, and reduced formation of GHGs, including N_2O (He et al., 2021; Kinh et al., 2017b; Uri-Carreño et al., 2024). Currently, there is a gap in literature reviews on MABR IW applications, potentially hindering innovation and knowledge transfer from municipal to industrial settings. This paper aims to address this gap by examining:

- Current configurations and advancements in MABR technology for a wide range of IW applications;
- Key factors affecting MABR performance;
- Impact of BT on process performance;
- Advances in MABR process modeling;
- Comparison of municipal and industrial MABR applications; and finally
- MABR challenges, innovations, and emission considerations.

2. MABR configurations

Three major commercial players in MABRs technologies include Veolia (Water Technologies & Solutions) ZeeLung™, OxyMem™ (a Dupont brand) OxyFILM and OxyFAS, and Fluence Subre and Aspiral™. ZeeLung™ and OxyMem™ use HF dense membrane materials like polydimethylsiloxane (PDMS), while Fluence employs spiral wound gas-permeable membranes to support the biofilm-based treatment (Fig. 2). These configurations are typically used as pure MABR systems for greenfield applications or as Hybrid MABR/CAS or IFAS systems for process intensification.

Table 1 presents a comparison of key parameters of various commercially available MABRs. Zeelung™, OxyMem™, and Fluence all exhibit a high technology readiness level (TRL), aeration efficiency, low sludge production, and very good (>60–90%) municipal wastewater carbon and nitrogen removal performance. These HF membranes provide a significantly larger membrane surface area for biofilm growth compared to Fluence's spiral wound membranes, thus limiting Fluence's application to small and medium-sized decentralized treatment plants (Tirosh and Shechter, 2018). A distinguishing aspect of MABR performance is the control of BT to prevent membrane fouling and clogging. Zeelung™ controls BT by using exhaust air from the system, which is sent to a sensor that measures O_2 concentration. Additionally, based on O_2 results and proprietary models, scouring and substrate mixing are conducted at timed intervals using a coarse bubble aeration grid (Côté et al., 2015; Guglielmi et al., 2020). OxyMem™ employs a patented online automated BT measurement and scouring system based on an argon pressure decay diffusion curve, which correlates a biofilm thickness index (BTI) to biomass weight. The scouring system uses dedicated blowers to produce coarse bubbles, and an Airlift system utilizes process off-gas to enhance mixing and evenly distribute substrate and nutrients among the fibers (Heffernan, 2024; Heffernan et al., 2017). Fluence MABR manages BT by circulating mixed liquor through the water spacers that separate the spiral wound membrane module. Each MABR reactor is equipped with coarse bubble diffusers for periodic mixing. The air mixing typically operates 2.5%-5% of the time (Tirosh, 2018; Tirosh and Shechter, 2018).

3. MABR applications to IW treatment

MABRs like ZeeLung™ and OxyMem™ have been evaluated at pilot and bench scale for the biological removal of high-strength IWs including pharmaceutical, chemical, refinery, petrochemical, oilfield, landfill leachate, steel pickling, coal chemical RO concentrate,

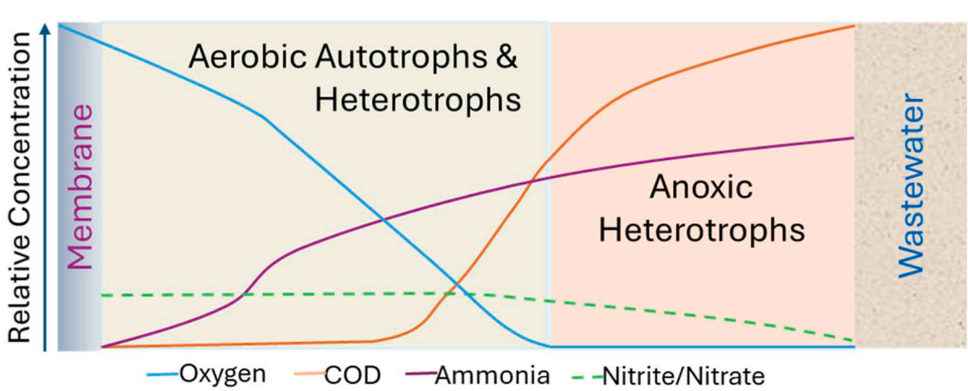

Fig 1. Membrane aerated biofilm concentration profile (adapted from Syron (2015)). Oxygen diffuses from the membrane outward, selecting for the various biofilm layers (Syron and Casey, 2008a).

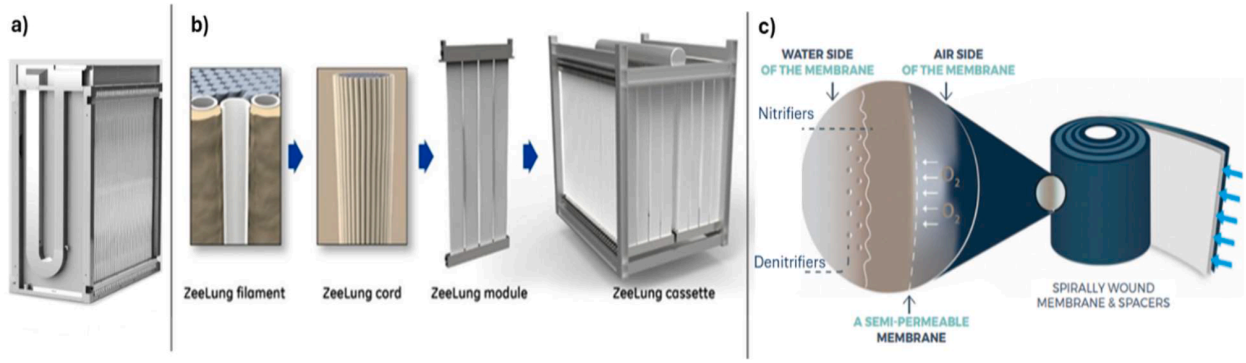

Fig. 2. Various MABR membrane configurations: a) OxyMem™ cassette (OxyMem, 2024); b) ZeeLung™ filament, cord, module and cassette (Veolia Technologies & Solutions, 2024); c) Fluence spiral wound membrane and spacers (Fluence, 2020).

Table 1
Comparison of commercially available MABR technologies.

Parameters	ZeeLung™[a]	OxyMem™[b]	Fluence[c]
TRL	9*	9*	9**
Aeration Efficiency	High	High	High
Carbon & Nitrogen Removal Performance	Very good	Very good	Very good
Nitrification/Denitrification	Yes	Yes	Yes
BT Control	Exhaust O_2	BTI	Mixed Liquor Recirculation
Specific Surface Area	High	High	Medium
Sludge Production	Low	Low	Low

[a] = Long et al., 2020; Guglielmi et al., 2020; Corsino and Torregrossa, 2022
[b] = Heffernan et al., 2017; Oxymem, 2024; Syron et al., 2014
[c] = Tirosh and Shechter, 2018; Fluence, 2020; Tirosh, 2018
* Municipal wastewater treatment applications
** Decentralized municipal wastewater treatment systems.

aquaculture, and livestock wastewaters. A summary of MABR performance with various membrane configurations and type, influent characteristics, hydraulic retention time (HRT), and carbon and nitrogen removal efficiencies (REs) are presented in Table 2.

3.1. Pharmaceutical and chemical wastewater

Several bench and pilot scale studies have demonstrated the feasibility of treating dissolved organic carbon, nitrogen compounds, and pharmaceutical and chemical waste using MABRs. These studies evaluated various configurations of MABR system including 1-stage and 2-stage Zeelung[TM] with dense HF membranes, PDMS and PVDF as pure MABR or as a pre-treatment before activated carbon or post-activated carbon and/or pre-treatment to ceramsite a microporous ceramic-based filter (Huang et al., 2023) (Table 2). The influent wastewaters were medium- to high strength with COD ranging from 300 to 3500 mg/L, NH_4^+ from ~5 to 2500 mg/L, and TN from ~9 to 300 mg/L. The COD, NH_4^+ and TN removals were up to 90–99% RE for all parameters. The HRTs varied widely from 8–50 hrs., with an average of ~25 hrs. which is in the range of an extended aeration systems employed in industrial CAS plants (HRT = 18–24 h) and municipal denitrifying facilities (HRT = 5–30 h) (Metcalf and Eddy, 2003). While IW generated by pharmaceutical operations may be mixed with municipal wastewater and not prevalent in industrial applications, it contains xenobiotics and organic micropollutants that are highly detrimental to the environment and human health (Singh et al., 2023). Fig. 3 shows the reported MABR REs of 24 pharmaceutical and chemical byproducts in IWs under specific conditions. The results indicated MABR capacity to remove 95–99.9% of acetaminophen, bisphenol-A, estrone (E1), ibuprofen, and triclosan which are nonpolar, hydrophobic and hydrophilic organic pollutants, respectively. However, lower REs (22–69%) were observed for gemfibrozil, ketoprofen, mefenamic acid and naproxen which are negatively charged and acidic substances. Lower REs were obtained for carbamazepine, diclofenac, primidone, gemfibrozil, and ketoprofen due to low biotransformation rates (Sanchez-Huerta et al., 2023, 2022). Phenolics, acetone, toluene and some fluorinated compounds were also readily removed above 90% (Heffernan et al., 2009; Misiak et al., 2011; Mei et al., 2019b; Tian et al., 2019, 2020; Wu et al., 2024).

3.2. Petroleum refinery and petrochemical wastewater

Typical wastewater from petroleum refinery and petrochemical processes contains a complex mix of pollutants, including hydrocarbons, sulfides, and heavy metals, which can impact bacterial activity in treatment systems (IPIECA, 2010; Jafarinejad and Jiang, 2019). The variability in molecular weight of these compounds, from low MW organics like acetate (≤59 g/mol) to particulate matter like asphaltenes (>40,000 g/mol), presents challenges for evaluating MABR performance due to the presence of particulate and non-diffusible substrates (National Center for Biotechnology Information, 2025; Barrera et al., 2013).

3.2.1. Petroleum refinery

A bench scale study investigated the degradation of petroleum refinery wastewater using an OxyMem™ MABR unit with PDMS HF membranes (Dicataldo, 2015). The unit operated for 169 days (Fig. 4) and reached a steady state at ~120 days, showing a consistent COD RE until the end of the experiment. Despite fluctuations in COD load and potential toxicity, the MABR system generally achieved on average a COD RE of 80% (Fig. 5). The TOC REs were also consistent with the COD during the same period, with an overall TOC RE reaching up to ~96% due to the removal of biodegradable organics. The NH_4^+ RE showed significant variability between day 27–120, with a downward trend from ~90% (day 29) to ~8% (day 71). The loss of nitrification was likely due to an inhibitory environment caused by occasional pH spikes and a significant increase in NH_4^+ load, which was not matched by the population density, metabolic capacity, and resilience of nitrifying bacteria to toxic organics. This behavior corroborates findings by Veleva et al. (2022) where shock loads of ammonia and toxic organics significantly inhibited nitrification. From day 71 to 120, the system showed an upward trend in NH_4^+ RE, indicating acclimation, recovery, and stabilization of nitrifiers. After day 120, the system reached a steady state with NH_4^+ RE up to 91%, reflecting a well-established nitrifying bacterial population. The MABR reached maximum TN RE of >90% corresponding to a TN in the effluent of the MABR unit (TNout) of 4 mg/L at day 164.

3.2.2. Petrochemical

Veleva et al. (2022) employed two OxyMem™ MABRs pilot units in series, with a volume of ~54 L and HRT of 10 h each to treat

Table 2
Summary of recent MABR applications to IW treatment.

Application	Membrane Configuration	Membrane Type	Influent Characteristics (mg/L)	HRT (hrs)	Carbon RE (%)	Nitrogen RE (%)	References
Pharma	Bench, 1-stage	Zeelung™, dense HF	COD: 300, NH_4^+-N: 25	20	COD: 80	NH_4^+-N: 95	(Sanchez-Huerta et al., 2022)
Pharma	Bench, 2-stage	Zeelung™, dense HF	COD: 450 ± 40, NH_4^+-N: 35±3	8	COD: 86 ±1.9	NH_4^+: 83 ± 4.6	(Sanchez-Huerta et al., 2023)
Pharma	Pilot, 1-stage hybrid (post-AC),	Custom propylene HF	COD: 2000–3500, $NH4^+$-N: 74–116, TN: 80–164	39–50	COD: >90	NH_4^+-N: 98, TN: >80	(Wei et al., 2012)
Chemical	Bench, 1-stage	Custom PDMS HF	COD: 161–805	19	COD: 89.8	TN: 94.8	(Mei et al., 2019b)
Chemical	Bench, 2-stage	Custom PVDF HF	COD: 367–2158, $NH4^+$-N: 21.4–74.3, TN: 25.4–80.7	14–32	COD: 91–97	TN: >90	(Tian et al., 2019)
Chemical	Bench, 1-stage, hybrid (ceramsite)	Custom polypropylene/ silicone HF	TOC: 200–500, TN: 100–300	18	TOC: ~80	TN: 65	(Mei et al., 2019a)
Refinery	Bench, 1-stage,	OxyMem™, PDMS HF	COD: 460±190, NH_4^+: 30±14, TN: 48±12	17.3	COD: 92	NH_4^+: 91 TN: 90	(Dicataldo, 2015)
Petrochem	Pilot, 2-stage, OxyMem™, PDMS HF	OxyMem™, PDMS HF	COD: 395 ± 122, NH_3: 4.7 ± 1.4, TN: 8.8 ± 0.9	20	TOC: 80–85	NH_3: 70–90	(Veleva et al., 2022)
Oilfield	Bench, 1-stage hybrid (O_3-BAC),	Custom dense HF	COD: 480, NH_4^+-N: 5.3, TN: 31	12–14	COD: 82.3	NH_4^+-N 32.1, TN: 71.9	(Li et al., 2015)
Landfill leachate	Pilot, 1-stage, OxyMem™ PDMS HF	OxyMem™, PDMS HF	COD: 1000–3000, NH_4^+: 500–2500	108–180	COD: 94.6	NH4+-N: 80–99, TN: 50	(Syron et al., 2015)
Steel pickling rinse	Bench, 2-stage	Hydroking* HF	COD: 110–120, NH_4^+-N: 80–90, TN: ~100	20	COD: 62.8	NH_4^+-N: 99.6, TN:51.7	(Sun et al., 2022)
Coal chemical RO concentrate	Bench, 3-stage	Hydroking* polymer composite HF	COD: 760–790, NH_4^+-N: 65.1, TN: 268–279	18–30	COD: 81	NH_4^+-N: 92.3, TN:70.7	(Lan et al., 2018)
Coal chemical RO concentrate	Bench, 3-stage	Hydroking* composite HF	COD: 280–320, NH_4^+-N: 2.1–2.9, TN: 147–165	12–30	COD: 69.4	NH_4^+-N: 81.0, TN:54.4	(Liu et al., 2020)
Aquaculture	Bench, 1-stage	Custom polypropylene HF	COD: 80, $NH4^+$-N: 8	12	COD: 94.6	NH_4^+-N: 73.9, TN: 50	(Xia et al., 2024)
Livestock	Bench, 1-stage	Custom PTFE	COD: 689–3444, NH_4^+-N: 38–188.5, TN: 42.5–212.5	24	COD: 85	NH_4^+-N: 90, TN: >90,	(Gong et al., 2020)

(-)= data not available; COD= chemical oxygen demand; NH_4^+= ammonium ion; NH_4^+-N= ammonium ion as nitrogen; TN= total nitrogen; TOC= total organic carbon; PVDF= polyvinylidene fluoride; PTFE= polytetrafluoroethylene; O_3= ozone; BAS= biological activated carbon; post-AC= post-MABR activated carbon treatment.
[*] Hydroking Sci & Tech, Ltd., Tianjin, China.

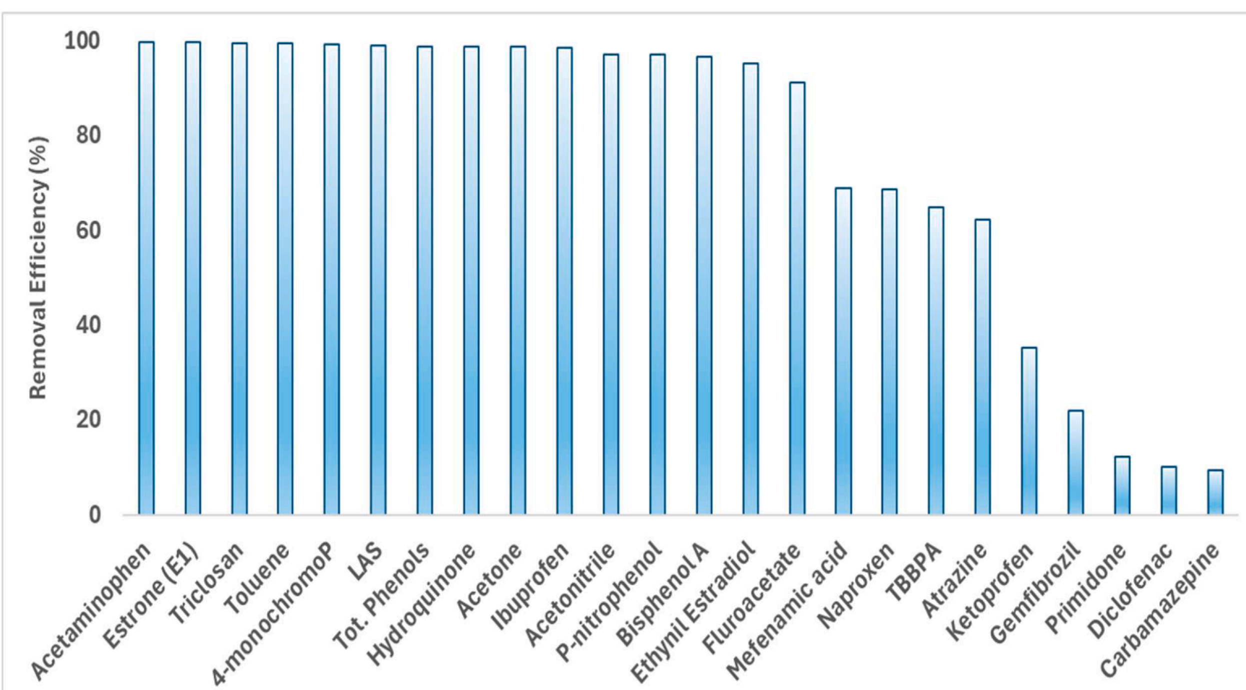

Fig 3. Reported REs of 24 xenobiotics and organic micropollutants from pharmaceutical and chemical industries under specific conditions. TBBPA = tetrabromobisphenol, LAS = linear alkylbenzene sulfonate (Heffernan et al., 2009; Misiak et al., 2011; Potvin et al., 2012; Li and Liu, 2019; Mei et al., 2019a,b; Tian et al., 2019, 2020; Kunlasubpreedee and Visvanathan, 2020; Sanchez-Huerta et al., 2022; Wu et al., 2024; Yang et al., 2024).

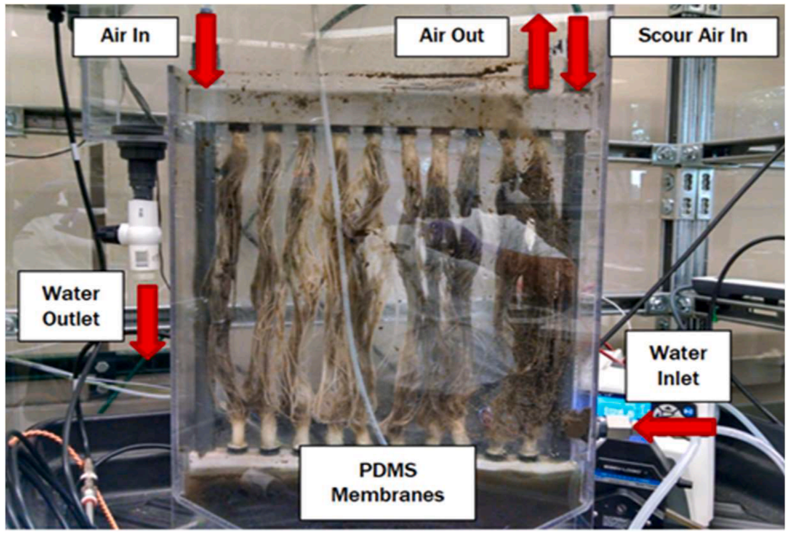

Fig. 4. OxyMem™ MABR bench scale unit.

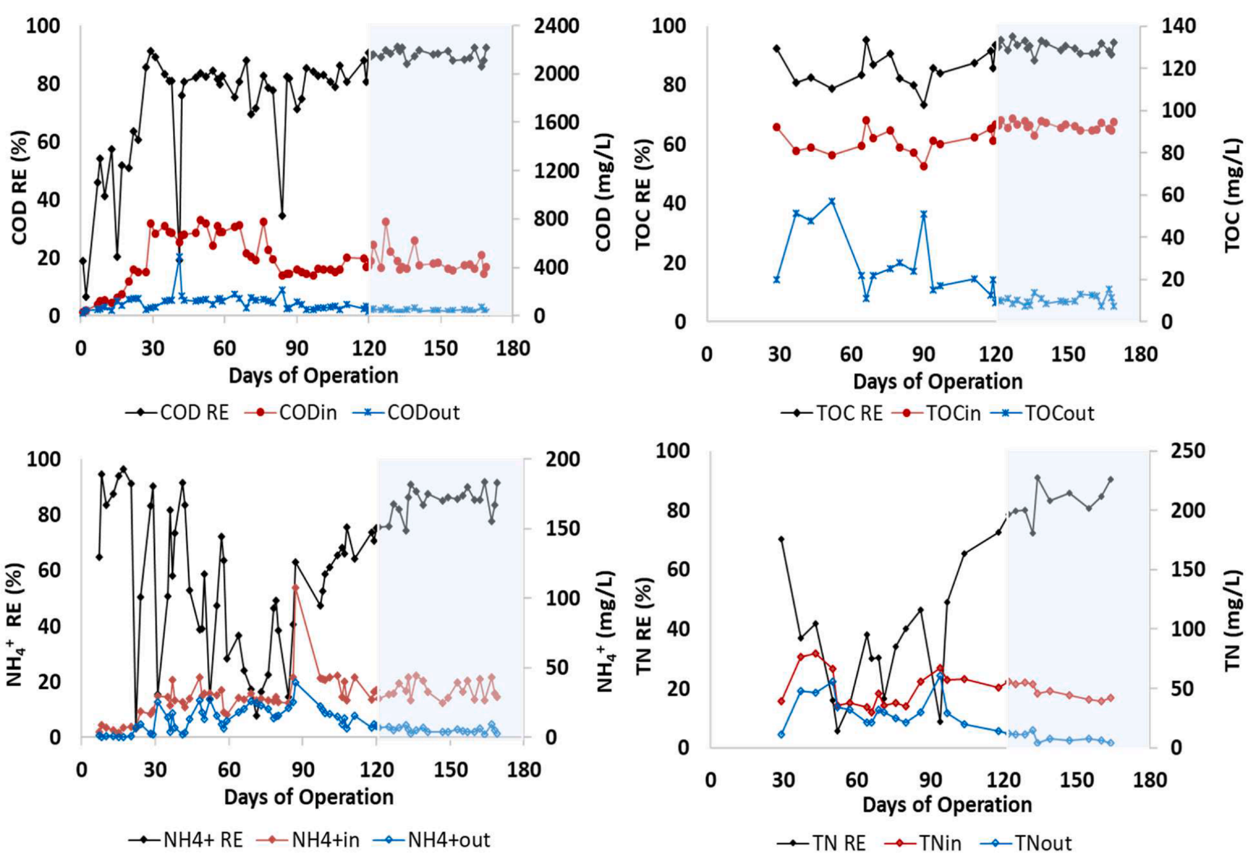

Fig. 5. COD, TOC, NH_4^+ and TN RE (%) and concentrations in and out of the MABR bench scale unit treating refinery wastewater. The shaded areas indicate steady state conditions.

petrochemical condensate. After start-up, the process was operated in continuous pilot operation on synthetic feed and transitioned to actual petrochemical feed. At steady state operation (> 100 d), the pilot achieved an overall RE of TOC, BOD_5, organic acids (acetate, propionate, and formate), phenol, and ammonia of 85%, 95%, 98%, 98%, and > 90%, respectively. The system was able to perform SND without traces of intermediate by-products, NO_2^- and NO_3^-. Higher OTE were observed of > 21%, compared to the CAS processes (~10%) (He et al., 2021).

3.3. *Oilfield wastewater*

Oilfield wastewater, typically known as produced water, is hard to biotreat due to high concentrations of organic and inorganic constituents and the presence of toxic compounds. The produced water typically contains various concentrations of TOC mainly hydrocarbons and field chemicals (500 - 2000 mg/L), total dissolved solids (2510 - 247,000 mg/L), chlorides (62 - 152,750 mg/L) and pH (4.3 – 8.8) (Adham et al., 2018). Li et al. (2015) employed a bench scale MABR coupled with ozone and biological activated carbon (hybrid MABR) to treat oilfield wastewater. The inoculum for the biofilm was composed by ADB350M engineering bacteria (Advance Biotechnologies of Canada) specifically formulated for high adaptability to petroleum pollutants. The influent characteristics and REs are presented in Table 2. A long-term study conducted for 60 days at 0.06 m/s feed flow rate and HRT of 12–14 h shows a good COD, oil, NH_4^+-N, and TN REs of 82.3%, 85.7%, 32.1%, and 71.9%, respectively. The study also examined dissolved oxygen concentrations gradients at different BTs under varying aeration pressures. At moderate lumen pressure, the biofilm exhibited better stratification of the community structure, with both anaerobic and aerobic layers suitable for microbes with different functions.

3.4. Landfill leachate wastewater

Landfill leachate is a type of industrial waste very high in ammonia which is harmful to fish and wildlife needing treatment before its release to the environment (Syron and Casey, 2012). Syron, et al. (2015) reported MABR ammonium RE of 80–99% with influent concentrations of 500–2500 mg/L, HRT of over 4 days as compared to 40 days for sequencing batch reactors. The ammonium loading rate reached up to 3.2 g NH_4^+-N m^{-2} d^{-1} with pure oxygen and observed data and multi-species AQUASIM model showed that the MABR performance was not limited by the O_2 delivery but by NH_4^+ transport to the biofilm attached to the HF membranes. The OTE was as high as 80% and the standard aeration efficiencies were up to 10 kg O_2 kWh^{-1} which are much higher than typical CAS (1–1.5 O_2 kWh^{-1}) (Table 3).

3.5. Steel pickling & coal chemical RO concentrate wastewater

Steel pickling wastewater is generated during the steel processing and manufacturing. Strong acid solutions (sulfuric, hydrochloric, and nitric acid) are used to rinse the steel to eliminate rust from its surfaces. This process generates wastewater containing high concentrations of refractory organics, nitrogen and dissolved solids. Conventional physico-chemical technologies like ion exchange, coagulation-flocculation, adsorption and catalytic reduction have been employed to treat these types of wastewaters, however they are costly and generate toxic byproducts (Sun et al., 2022). A two-stage MABR system was evaluated to treat steel pickling rinse wastewater. The best REs for COD, NH_4^+-N and TN were 62.8%, 99.6% and 51.7%, respectively. Aeration pressure was found to be more important than salinity in the control of shortcut nitrification. Moreover, the biofilm secreted ten times more extracellular polymeric substance (EPS) as aeration pressure and two times as much as salinity increased. Coal chemical RO concentrate wastewater is high in salinity, and refractory organics posing an environmental hazard if not properly treated. Advanced oxidation, adsorption, electro-oxidation and membrane distillation have been employed to treat these types of wastewaters; however, they are expensive and generate harmful bioproducts (Lan et al., 2018; Liu et al., 2020). A three-stage bench scale MABR achieved REs of COD, NH_4^+-N and TN of 81%, 92% and 71%, respectively (Table 2). SND as well as shortcut nitrogen removal were also achieved and a salinity of 3% did not caused a significant decrease in treatment efficiency and microbial diversity (Lan et al., 2018). A similar study employing a three-stage bench scale MABR on coal chemical RO concentrate with lower salinity (0.67%) showed REs for COD, NH_4^+–N, NO_3–N, and TN of 69%, 81%, 55%, and 54%, respectively (Liu et al., 2020).

3.6. Aquaculture and livestock wastewater

MABR technology was applied to IW laden with sulfamethoxazole (SMX) a commonly used antibiotic medication in aquaculture. The study showed that the MABR removed 77.2±2.6% of SMX (Xia et al., 2024). Livestock wastewater generated via anaerobic fermentation of cow manure containing pollutants such as nitrogen, phosphorus, and organic matter was treated using MABR. The REs of NH_4^+-N and COD were up to 90% and 85%, respectively, at optimal conditions. The authors concluded that because the inner biofilm had higher amounts of EPS compared to the outer biofilm, it reduced the toxicity impact of high cow manure waste concentrations on the aerobic bacteria (especially nitrifying bacteria) population located near the membrane surface (Gong et al., 2020).

4. Key factors in MABR performance

The MABR performance is significantly influenced by several key factors, including oxygen transfer rate (OTR) and OTE, the carbon to nitrogen ratio (C/N), biofilm development and microbial diversity, BT, and counter-diffusion mechanisms.

4.1. OTR and OTE

OTR and OTE are key performance indicators used to evaluate aeration capacity of MABRs (Corsino and Torregrossa, 2022; Côté et al., 2015; Guglielmi et al., 2020; He et al., 2021). The OTR (gO_2 d^{-1}) and OTE (%) are typically calculated by off-gas analysis of molar fractions of O_2 in the exhaust gas (O_{2ex} (%)), O_2 in the inlet air (Q_{2in}(%)), and air flow rate (Q_{in} & Q_{out} in Nm^3 m^{-2} h^{-1}) in flow-through MABR systems as shown in eq. (2)-(4) (Côté et al., 2015):

$$\text{OTR} = J_{O_2} \, . \, A_m \tag{2}$$

$$J_{O_2} = \frac{24\, M_O}{V_M}\left(Q_{in} \, . \, O_{2in} - Q_{out} \, . \, O_{2ex}\right) \tag{3}$$

$$\text{OTE} = \frac{J_{O2}}{24} \, . \, \frac{Vm}{Q_{in} M_{O2}\, O_{2in}} \tag{4}$$

where J_{O2} is the oxygen flux (gO_2 m^{-2} d^{-1}), A_m is the membrane surface area (m^2), M_O is the molecular weight of O_2 (32 gO_2 mol^{-1}), and V_m is the standard gas volume at STP (0.0224 m^3 mol^{-1}). A summary of aeration parameters for various industrial MABR applications is presented in Table 3. MABR aeration modes can be either flow-through or dead-end based on lumen gas flow. In the flow through mode, where the distal end

Table 3
Key aeration parameters in industrial MABR applications.

Application	System Type	Aeration Mode	Process Gas	Lumen Gas Pressure (kPa)	OTR (g O_2 m^{-2} d^{-1})	OTE (%)	AE (KgO_2 KWh^{-1})	References
Petrochem	Pilot	OE	Air	21	2±0.5	21±6	6.5–11	(Veleva et al., 2022)
Refinery	Bench	OE	Air	60	11.7±9	29±23	-	(Dicataldo, 2015)
Landfill Leachate	Pilot	OE	Pure O_2/ Air	23–120	8 (air), 25 (pure O_2)	20–75 (air) 50–80 (pure O_2)	4–10	(Syron, et al., 2015)
Industrial	Pilot	OE	Air	41	2.9–3	25–31	-	(Stricker et al., 2011)
CAS	Full scale	FBD	Air	-	-	10	1–1.5	(Rosso et al., 2008; He et al., 2021)

- = data not available; OE= open-end; FBD= fine bubble diffuser.

of the membrane is opened, gas (i.e., air or pure O_2) is passed through the membrane lumen directly to the biofilm and the exhaust gas (O_2 and/or CO_2) is monitored for treatment performance (Guglielmi et al., 2020; Veleva et al., 2022; Wei et al., 2012). In contrast, the dead-end mode, where the distal end is closed, delivers all supplied oxygen to the biofilm, achieving up to 100% OTE and reducing aeration energy (Tian et al., 2020). However, the dead-end system suffers from lower OTRs due to back-diffusion of gases including CO_2 and O_2. This causes a significant drop in O_2 partial pressure in the membrane lumen and consequent reduction in biological treatment capacity. The flow-through mode is preferred for commercial use due to its ability to achieve higher OTRs with respect to the dead-end systems, leading to better contaminant removal and treatment capacity (Casey et al., 2008; Guglielmi et al., 2020; Kunetz et al., 2016; Peeters and McMains, 2023). Selecting aeration modes involves balancing OTR vs. OTE and treatment capacity vs. aeration energy. Innovative strategies, such as alternating between flow-through and dead-end modes, can improve both OTE and OTR (Perez-Calleja et al., 2017). OTEs reported for industrial MABRs applications range from 21–75%, which is significantly higher than CAS fine bubble aeration systems. MABRs for industrial applications show superior aeration efficiencies (4–11 KgO_2 kWh^{-1}) compared to CAS systems (1.0–1.5 KgO_2 kWh^{-1}) (Table 3). These advantages make MABR systems more suitable for industrial applications requiring high-performance aeration, such as petrochemical, refinery, landfill leachate, and others. Given that aeration is a major energy cost in wastewater treatment plants (45–75%) (Rosso et al., 2008), MABRs' high efficiency can lead to significant energy savings (Casey et al., 2008). Using pure oxygen in MABRs enhances OTRs and its penetration into the biofilm, enabling high COD and ammonia REs (Brindle and Stephenson, 1996; Syron et al., 2015; Abdelfattah et al., 2024). This can reduce membrane area requirements and in turn capital investment (Syron et al., 2015). However, an enriched oxygen environment can lead to thicker aerobic biofilm, suppression of denitrification genes, increased mass transfer resistance, and the need for more vigorous biofilm scouring to maintain optimal MABR performance (Cole et al., 2004; Stricker et al., 2011; Syron et al., 2015). In addition, pure oxygen safety management constraints may hinder its use (Air Products and Chemicals, 2014).

4.2. C/N ratio

The C/N ratio is critical in the formation of both nitrifying and denitrifying bacterial populations within the MABR biofilm (Liu et al., 2010). Simultaneous nitrification and denitrification can be achieved in the stratified biofilm when proper conditions exist, such as adequate oxygen penetration and C/N ratio. Chang et al.(2022) reported optimal C/N as a COD/N ratio of 4.3 for maximum TN RE (78.9%). Liu et al., (2010) found that a COD/N ratio of 5 was optimal for nitrification and denitrification (93%, and 92%, respectively), with *Nitrosospira* and *Nitrospira* as the dominant nitrifiers. For a MABR system treating refinery wastewater the optimal COD/N ratio was found to be optimal between 7 and 13, with NH_4^+ and TN RE up to 96% and 91%, respectively, at steady state conditions (Dicataldo, 2015). Other studies have shown C/N ratios varying widely between 2 and 18, with stable SND results (Li and Zhang, 2018; Lin et al., 2016; Meng et al., 2008; Veleva et al., 2022). Lin et al. (2016) suggested that maintaining a filtered COD/N ratio above 5 is optimal for achieving NH_4^+ and TN RE above 80%. Using filtered or soluble COD (sCOD) can be limiting, especially in IWs with particulate COD from heavy compounds like asphaltenes found in crude oil laden wastewater (Barrera et al., 2013). However, in general, an adequate COD/N ratio coupled with sufficient oxygen flux, highly influence the microbial community structure in the MABR biofilm, particularly ammonia-oxidizing bacteria (AOB) and denitrifying bacteria, and in turn, nitrogen removal (Chang et al., 2022).

4.3. Biofilm development and microbial diversity

Biofilm development and microbial diversity are the foundation for the biological transformations occurring within the MABR system (He et al., 2021). Biofilm inoculums are usually sourced from local CAS, Anaerobic-Anoxic-Aerobic (A^2O) systems, or fit-for-purpose engineered or mixed bacterial cultures that are either well-adapted and/or have been exposed to the targeted wastewater streams (e.g., petrochemical, pharmaceutical, chemical etc.) (Mei et al., 2019a; Syron et al., 2015; Van Ginkel et al., 2008; Veleva et al., 2022; Wei et al., 2012). Typically, the startup procedures include 1–25 days inoculation in batch mode, where the activated sludge (AS) is recirculated and continuously mixed in the MABR system with various volumes of synthetic or real wastewater feeds (Dicataldo, 2015; Sanchez-Huerta et al., 2022; Syron et al., 2015; Tian et al., 2019; Veleva et al., 2022). For pure MABR systems, the bulk biomass is removed, leaving only sessile bacteria to establish a biofilm structure on the membranes. These types of bacteria are a mix of aerobic AOBs, nitrite-oxidizing bacteria (NOBs), anoxic denitrifiers and higher life forms (protozoa and metazoa). The timing for biofilm establishment, including attachment and growth to reach adequate microbial density and steady-state performance, is critical in implementing MABR technologies. Various reports indicate that bench and pilot-scale high-strength IW MABR applications reached steady-state conditions (i.e., stable COD, NH_4^+, and TN RE) after 60 – 120 days of operation (Dicataldo, 2015; Li et al., 2015; Veleva et al., 2022). A summary of the microbial communities and techniques used to identify and quantify composition and structure of biofilm in MABRs treating IWs is presented in Table 4. Genomic analyses were carried out using high throughput sequencing, 16 rDNA/16s rRNA gene sequencing and fluorescence in situ hybridization (FISH). The most prevalent genus in MABR biofilms was shown to be the *pseudomonas*, which is responsible for denitrification and carbon removal. Meanwhile, *nitrospira, nitrobacter* and *nitrosomonas* genera were reported to be the most abundant nitrifiers. In general, betaproteobacteria were found to be common among MABR biofilms.

4.4. Biofilm thickness

The thickness of a biofilm is a key characteristic in biofilm processes including MABRs. It has been observed that the rate of substrate diffusion, nitrification and denitrification, COD and TOC removal, as well as microbial ecology and interactions, are largely influenced by the thickness of the biofilm (Martin and Nerenberg, 2012; Torresi et al., 2016). Casey et al. (2000) discovered that the BT and intra-membrane oxygen pressure were the most important parameters affecting the MABR performance experimentally and via mathematical modelling. A thinner biofilm was shown to facilitate high substrate diffusion; however, MABR performance was hindered due to a lower concentration of biomass. On the other hand, a thicker biofilm increased pollutants removal but may lead to lower substrate diffusion (Casey et al., 1999, 2000; Martin and Nerenberg, 2012; Li and Zhang, 2018). Sanchez-Huerta et al. (2022) showed that an increase in BT and cell density from 0.10 mm to 1.02 mm and from 3.1 × 10^4 cells mL^{-1} to 2.2 × 10^6 cells mL^{-1}, respectively, enhanced the MABR performance. These results were confirmed by Sanchez-Huerta et al. (2023) in terms of NH_4^+ removal. However, there is an optimal BT where oxygen and substrates (NH_4^+ and COD) counter-diffusion and transport becomes limiting and an inactive layer of biofilm forms (Casey et al., 2000).

4.4.1. Correlation between BT, COD and NH_4^+ removal efficiencies

A comprehensive survey of available data on BT and MABR IW treatment performance was performed. Data trends revealed a good correlation between BT and COD RE (%) and NH_4^+ RE (%) ($R^2 = 0.65$ and 0.73) (Fig. 6). Also, COD and NH_4^+ RE showed a strong relationship between these two variables ($R^2 = 0.86$) (Fig. 6). Analysis of variance (ANOVA) showed a $p < 0.05$ indicating that there is a statistically

Table 4
Summary of relevant microbial communities in industrial MABR applications.

Removal Objective	Target Compound	Taxonomy	Genomic Analyses	References
VOC	Toluene/Acetone	Genus: *Pseudomonadota, Rhodanobacter*	High- throughput sequencing	(Wu et al., 2024)
OC	Phenol, PNP, p-DHB	Class: *Gammaproteobacteria, Actinobacteria, Betaproteobacteria, Alphaproteobacteria* *Genus: Peudomonas, Rhodococcus*	High- throughput sequencing	(Tian et al., 2020)
OC, SND	Cow manure from anaerobic fermentation	Phylum: *Proteobacteria* Class: *Gammaproteobacteria, Bataproteobacteria, Alphaproteobacteria, Deltaproteobacteria*	High- throughput sequencing	(Gong et al., 2020)
OC	Acid orange 7	Species: *Shewanella*	16 rDNA gene sequencing	(Wang et al., 2012)
OC, SND	O-aminophenol	Genus: *Pseudomonas Cupriavidus, Thauera*	PCR 16S rRNA and amoA genes and Illumina MiSeq sequencing	(Tian et al., 2019)
OC, NH_4^+	13 OMPs	Genus: *Zoogloea, Aqua- bacterium, Leucobacter, Runella, and Paludilbaculum*	Genomic sequencing	(Sanchez-Huerta et al., 2022)
SND	Steel pickling rinse	Genus: Nitrifiers (*Nitrosomonas, Nitrospira*), Denitrifiers (*Dechloromonas, Hyphomicrobium, Denitromonas, Denitratisoma, Candidatus Competibacter*) and Aerobic Denitrifiers (*Pseudomonas, Thauera*)	High- throughput sequencing	(Sun et al., 2022)
OC, SND	Coal chemical RO	Phylum: *Proteobacteria* Class: *Bacteroidetes*	High throughput sequencing	(Lan et al., 2018)
OC, SND	Oilfield	ADB350M (aerobic/anoxic)	-	(Li et al., 2015)
OC, SND	Refinery	Class: *Betaproteobacteria* Genus: *Nitrospira, Nitrobacter*	FISH	(Dicataldo, 2015)
-	4-fluorobenzoate, benzoate	Genus: *Pseudomonas knackmussii B13*	-	(Misiak et al., 2011)
-	Fluoroacetate	Genus: *Pseudomonas fluorescens*	-	(Heffernan et al., 2009)

VOC= volatile organic compound; OC= organic carbon; p-DHB= hydroquinone; PNP= p-nitrophenol; 13 OMPs= organic micropollutants including Acetaminophen, Bisphenol A, Estrone, Ethinyl Estradiol, Ibuprofen, Triclosan, Gemfibrozil, Ketoprofen, Mefenamic Acid, Naproxen, Carbamazepine, Diclofenac, and Primidone.

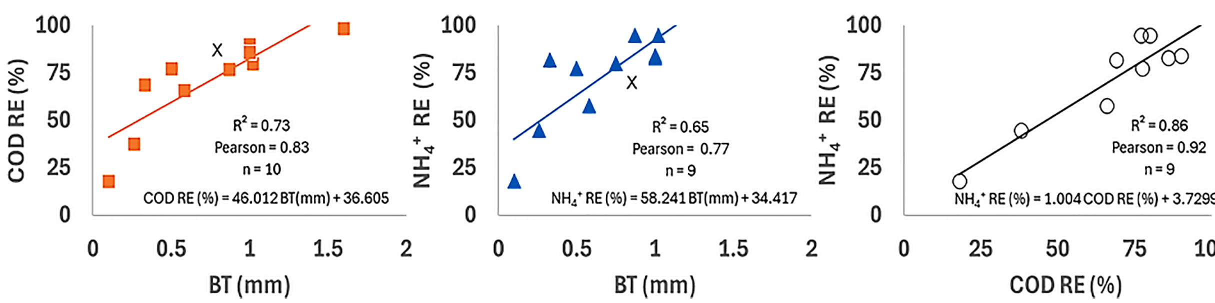

Fig. 6. Correlation between COD RE, NH_4^+ RE, and BT in IW; The one-way ANOVA test p-value was < 0.05. (Dicataldo, 2015; T. Li et al., 2008; Peng et al., 2015; Sanchez-Huerta et al., 2022, 2023; Wei et al., 2012). X indicates optimal BT for RE in municipal wastewater applications (Elsayed et al., 2021; Matsumoto et al., 2007).

significant difference between BT, COD and NH_4^+ REs. The highest COD RE of 98.6% was observed at a BT of 1.6 mm, while NH_4^+ RE reached a maximum of 95% at a BT of 0.87 mm. No improvement was evident in NH_4^+ RE for a thickness of 1.02 mm. Calibrated model simulations of MABR performance vs. BT applied in municipal wastewater application showed that for optimal SND a minimum BT of 0.6 mm is recommended. Elsayed et al. (2021) showed that COD RE >89% at BT of 0.8 mm and nitrogen RE (> 70%) was attained for BT of 0.6 to 1.2 mm (Matsumoto et al., 2007). Although biofilm thickness is challenging to measure due to its variable nature, it appears that optimal NH_4^+ and COD REs in IWs favor slightly higher BTs compared to municipal systems, likely due to the complexity of IWs.

4.5. *Biofilm counter-diffusion mechanisms*

Counter-diffusion of wastewater substrates and oxygen across the MABR biofilm is key in achieving optimal treatment. Studies have shown that the diffusion of organic substrate (e.g., xenobiotics), nutrients and other contaminants from the bulk liquid into the biofilm depends on several factors including molecular weight (MW), net charge of substrate in solution, heterogeneity of the biofilm (EPS, cell density and thickness), and sorption (Debus and Wanner, 1992; Horn and Morgenroth, 2006; Stewart, 1996). Also, seminal research work by Wanner et al. (1994) show the importance of unique stratification of specialist degrading microorganisms within the biofilm. Solutes with higher MWs and net negative charges showed lower diffusion due to matrix resistance and repulsion from negatively charged microorganisms. Positively charged compounds penetrated faster due to electrostatic interactions. No significant impact was observed to the diffusion coefficients for pH values varying between 4 and 9, and ionic strength between 0.1 to 100 mM. Furthermore, heterogeneity of the biofilm was linked to variations of ~1 of order of magnitude in diffusion coefficients (Zhang et al., 2011). In addition, microorganisms positioned deeper in the biofilm are subjected to lower substrate concentrations than those at the surface (Fig. 7). As a result, these deeper organisms either convert substrates slowly or remain inactive. Conversely, diffusion gradients lead to varying redox zones throughout the biofilm, allowing multiple biological reactions to occur within a single reactor (e.g., nitrification, denitrification, etc.). Overall, metabolic conversion rates could be managed by controlling the diffusion of soluble substrates like oxygen, NO_3^-, NH_4^+, and carbon sources (van den Berg, et al. 2021).. Although diffusion in different biofilms has been studied (Stewart, 1998), there is still lack of

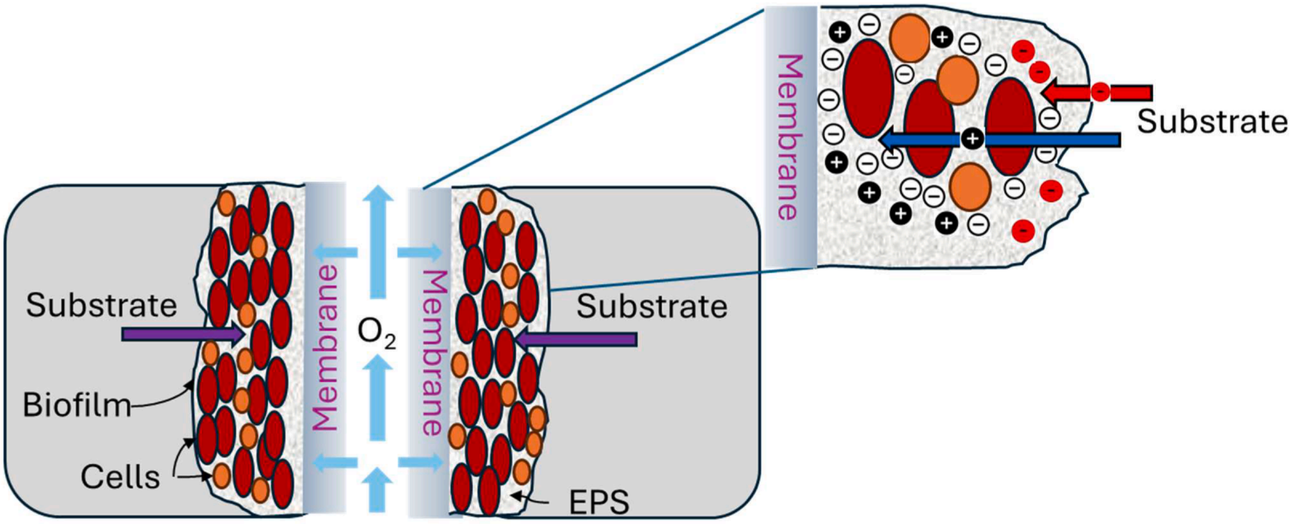

Fig. 7. Biofilm counter-diffusion mechanisms. The symbols ⊕ and ● represent the positive and negative net charges of the substrate, respectively, while ⊖ denotes the negative charge of the bacteria. Negatively charged high MW substrates (( )) diffuse slowly as compared to positively charged (( )) substrates.

knowledge on the impact of MWs of sCOD on MABR biofilm RE. Most studies have focused on small substrates like acetate and glucose with MWs < 350 g/mol (Stewart, 2003). Few studies have measured diffusion coefficients for higher MW compounds and particles (Peulen and Wilkinson, 2011; Takenaka et al., 2009; Zhang et al., 2011). Most IWs, particularly in oil and gas applications, contain high MWs compounds as

Table 5
Review of the latest MABR models.

Model Classification	Description	Modeled Substrate	ASM Implementation	Model Limitations	Refs.
Conventional Biofilm Model	Simulates biofilms with co-current substrate diffusion from the bulk liquid into the biofilm, creating an oxygen-rich outer layer and substrate gradients toward the biofilm interior. Useful for traditional biofilm reactor design.	O_2, OC, ammonia	ASM1	Oversimplifies biofilm heterogeneity; cannot model counter-diffusional systems like MABRs.	(Wanner et al., 2006)
MABR-Specific Model	Explicitly models the counter-diffusional mass transfer in MABRs, where oxygen diffuses inward through the membrane while substrates diffuse outward from the bulk liquid. It enables detailed analysis of oxygen-rich zones near the membrane for nitrification and heterotrophic processes.	O_2, ammonia, NO_2^-, NO_3^-	Modified ASM3	Limited to steady-state conditions; computationally intensive for large-scale systems.	(Syron and Casey, 2008b)
Pressure-Based Model	Focuses on oxygen transfer driven by partial pressure differences along the membrane length. Accounts for variations in oxygen flux across the membrane, critical for ammonia removal and optimizing aeration efficiency in MABRs.	O_2, ammonia	ASM2d for denitrification and nitrification	Requires accurate measurement of intramembrane pressure; ignores gas-phase interactions like nitrogen gas.	(Houweling and Daigger, 2019)
Exhaust Oxygen-Based Model	Models oxygen transfer efficiency (OTE) by tracking oxygen levels in the gas leaving the reactor. Useful for performance monitoring in large-scale MABRs where ammonia removal correlates with oxygen uptake.	O_2, ammonia, CO_2	-	Neglects nitrogen gas diffusion; less accurate for systems with high variability in substrate loads.	(Guglielmi et al., 2020)
1D MABR Model	Simulates substrate gradients along the biofilm depth at a single point on the membrane. Useful for understanding localized biofilm behavior but lacks ability to account for spatial variations along the reactor.	O_2, OC, ammonia, NO_2^-, NO_3^-	ASM1 with simplified substrate dynamics	Cannot account for longitudinal biofilm heterogeneity; lacks dynamic adaptation to operational changes.	(Carlson et al., 2021)
2D MABR Model	Expands on 1D models by incorporating spatial heterogeneity, such as variable biofilm thickness and density along the membrane. Particularly useful for modeling spiral-wound membranes and uneven biofilm distribution.	O_2, ammonia, NO_2^-, NO_3, OC	ASM2d with spatial adaptations	Computationally expensive; requires significant data for calibration and validation.	(Martin et al., 2013)
Dynamic MABR Model	Simulates time-dependent changes in biofilm growth, substrate consumption, and detachment under varying operational conditions. Useful for pilot-scale studies and evaluating system responses to dynamic wastewater loads.	Oxygen, ammonia, NO_2, NO_3^-, OC, phosphorus	ASM2d for dynamic nutrient removal processes	Complex implementation: difficulties in accurately predicting detachment and reattachment processes.	(Schraa et al., 2018)

compared to municipal wastewaters. Recent studies indicate that diffusion of a model substrate (polyethylene glycol (PEG)) with MW of 10,000 g/mol (10 kDa) is hindered by the presence of a biofilm in activated granular sludge (van den Berg et al., 2022). Therefore, diffusion of sCOD of high MWs represents an emerging area of study for MABR given the difference between industrial and municipal wastewater quality.

4.6. Advances in MABR modeling

Biofilm modeling has evolved to address the challenges of treating industrial wastewater, characterized by variable organic loads, toxic compounds, and high nutrient concentrations (Table 5). Conventional models, like those described by Wanner et al. (2006), focus on co-current oxygen diffusion and are less suited for the complexities of IW or advanced systems like MABRs. MABR-specific models, such as those by Syron and Casey (2008b), incorporate counter-diffusional oxygen transfer, improving predictions for systems with high oxygen demand or limited aeration capacity. Pressure-based and exhaust oxygen-based models (Guglielmi et al., 2020; Houweling and Daigger, 2019) optimize oxygen transfer rates, which are critical for industries with high COD, though they struggle with high solids content and fluctuating substrate loads, making them more scalable for large-scale industrial applications. Meanwhile, advanced models like 2D and dynamic frameworks (e.g., Carlson et al., 2021; Martin et al., 2013; Schraa et al., 2018) enable complex interactions between oxygen, nitrogen species, and organic carbon, essential for industrial applications involving nutrient removal. These models also provide deeper insights into spatial heterogeneity and temporal changes in biofilms, making them more suitable for fluctuating industrial wastewater. However, their application is limited by higher computational demands and difficulties in capturing rapid responses to toxic shocks, which are common in industrial discharges. Simpler models, like 1D frameworks, remain more practical for laboratory-scale studies but arguably do not consider the full-range of variables with scale-up to dynamic industrial systems. Existing frameworks fall short in representing biofilm detachment and regrowth dynamics, which are crucial for maintaining long-term MABR performance under the high variability of industrial systems. Future efforts must address these limitations by calibrating models with online measurement of biofilm dynamics such as stochastic events such as biomass detachment and incorporating dynamic feedback mechanisms for real-time optimization. These advancements will ensure biofilm models are robust, accurate, and practical for the complexities of industrial wastewater treatment.

5. Comparison of MABR treatment in industrial and municipal applications

In general, IWs are laden with xenobiotics, oil, metals, and other pollutants not typically found in municipal wastewater. Hydraulic retention times and sludge retention times for nitrification and denitrification in IW treatment systems are often much higher than those in municipal plants. This is due to the lower biodegradability of contaminants commonly found in petroleum refineries, petrochemical, pharmaceutical, chemical, and other difficult to treat IWs (Choi et al., 2017). As illustrated in Fig. 6, BT is directly correlated to COD/NH_4^+ RE, and a thicker biofilm is essential for removing xenobiotics from industrial wastewater. Consequently, the OTE and OTR are potentially impacted by the higher levels of organics and ammonium present in IWs (Syron et al., 2015). This, in turn, may result in higher lumen air pressure, increased air flows, and greater energy consumption compared to municipal applications (Syron and Casey, 2008a). Additionally, even though dense membrane materials like PDMS are more suitable to IWs than microporous materials (He et al., 2021) they can be affected by esters, ketones, acetone, chlorinated and aromatic solvents, and high chloride levels (OxyMem, 2024), which are more prevalent in IW than in municipal streams. As a result, MABRs are more widely applicable in municipal plants. The use of MABRs for treating oily petrochemical and petroleum wastewaters may be limited due to the potential impact of oils and organics on biofilm and membrane fouling (Wang et al., 2022). Future research direction should focus on creating new MABR membrane materials, such as those resistant to organic solvents used in other process industries (Ren et al., 2021). However, proper pre-treatment of free and emulsified oils using gravity separators or enhanced gravity separators (like dissolved gas flotation and hydrocyclones) is common practice in upstream and downstream oil and gas operations for any biological treatment (Adham et al., 2018; WEF, 2021). In recent years, the focus of MABR research has primarily been on municipal wastewater treatment, which has generated valuable scientific knowledge (Guglielmi et al., 2020). Nonetheless, MABRs have demonstrated high effectiveness at both bench and pilot scales in treating various types of IWs, either as standalone MABRs or hybrid MABR/CAS systems (Heffernan, 2024). In summary, the potential commercial application of MABRs at full scale is being evaluated for fit-for-purpose industrial applications, building on the knowledge acquired from municipal MABR systems (Adapa, 2024).

6. MABR challenges, innovations, and emission considerations

Although MABRs have shown promising performance in treating various types of IWs at bench and pilot scales, several challenges remain (Dicataldo, 2015; Li et al., 2015; Syron and Casey, 2008a; Veleva et al., 2022; Werkneh, 2022):

- **Biofilm growth control**: managing biofilm growth, especially at the ends of modules and in the center of HF membrane bundles, is difficult. Periodic scouring with large bubbles is used, but BT optimization for effective IW treatment and substrate distribution remains challenging.
- **Biofilm attachment and stability:** Biofilm treatment stability is typically achieved 60–120 days after startup. The complexity of IWs suggests that certain constituents may interfere with microbial attachment by altering surface charge, negatively impacting biofilm surface coverage. Evaluating factors that govern bacterial recruitment and proliferation to the membrane surface is vital to decrease reactor start-up time to steady state of nutrient removal.
- **Bacteria layering mechanisms**: fine-tuning the layering of bacterial groups controlled by oxygen diffusion through the membrane is often a trial-and-error process.
- **Mixing energy requirements**: adequate mixing energy is needed to ensure liquid distribution across all bundles. Outer edge HF membranes, especially those exposed to incoming influent flows may receive higher carbon and nitrogen loads, while those towards the center may not, leading to imbalances in BT and performance.
- **Impact of shock loads**: while steady-state performance for COD and NH_4^+ is less affected by shock loads, oils, high salinity, and toxic xenobiotics can alter biofilm structural integrity and biodiversity.
- **Scaling Up**: there is still a significant lack of clarity regarding guidelines for scaling up membrane modules to full-scale applications.

6.1. Novel hybrid and emerging MABRs

Novel combinations of MABRs coupled with other water treatment technologies have been investigated. Hybrid MABRs include bacterial-algae biofilms to treat wider ranges of COD/N ratios (Zhang et al., 2021), membrane bioreactors (Silveira et al., 2022), microbial electrolysis cells (De Paepe et al., 2020), activated carbon (Wei et al., 2012), ceramsite sand (Mei et al., 2019a), and ozone/biological activated carbon (Li et al., 2015). ZeeNAMMOX™ (Veolia) is an emerging partial nitrification/Anammox (PN/A) biofilm process combined with the

ZeeLung™ MABR. The PN/A is the most biologically efficient pathway for nitrogen removal (Wang et al., 2021). The process converts NH_4^+ to NO_2^- and N_2 gas using AOBs and anammox bacteria instead of a 2-step (AOB/NOB) nitrification and additional denitrification step. The PN/A process saves 57% in O_2 demand and 100% in sodium acetate as carbon source (Long, 2023). The ZeeNAMMOX™ promotes AOBs/Anammox and suppresses NOBs by controlling the OTR. Long et al. (2023) demonstrated that ammonium oxidation rate (AOR) and total inorganic nitrogen removal rate (TINRR) are a function of OTR. With OTR <16 $gO_2\ m^{-2}\ d^{-1}$, which is equivalent to AOR of around 7.5 $gN\ m^{-2}\ d^{-1}$ the NOBs were suppressed. In addition, the theoretical TINRR/AOR ratio to suppress NOBs was 0.89 (Long et al., 2023).

6.2. N_2O emission considerations

N_2O emissions from biological treatment systems are a significant concern due to their impact on climate change and the ozone layer (Kampschreur et al., 2009; Kinh et al., 2017a). Studies have shown that MABRs emit significantly less N_2O compared to conventional suspended biological systems and co-current biofilm systems (He and Daigger, 2023; Kinh et al., 2017a). For example, N_2O emissions from hybrid MABRs were found to be one-fifth of those from CAS units (He and Daigger, 2023). The main pathways for N_2O production include AOB activity, heterotrophic denitrification, and abiotic chemical reactions (Heil et al., 2014; Schreiber et al., 2012; Uri-Carreño et al., 2024). Real-time measurements from full-scale MABR plants confirmed that N_2O emissions are primarily due to nitrifier-nitrification and nitrifier-denitrification pathways (Uri-Carreño et al., 2024).

7. Conclusions

This paper critically reviews the feasibility of applying MABRs to more complex industrial wastewater treatment compared to municipal treatment. Key findings are included below.

- Key advantages of MABR include higher oxygen transfer to reduce energy costs, lower emissions of GHGs, SND and carbon removal in one bioreactor, a more compact process and retrofitting capabilities.
- Critical evaluation of vendors with full-scale experience was discussed highlighting the need for innovative designs to provide step changes in performance.
- Extensive literature reviews demonstrate that bench and pilot-scale MABR systems are effective in removing xenobiotics and treating high-strength IWs, including those from pharmaceutical, chemical, refinery, petrochemical, oilfield, landfill leachate, and other sources. However, the adoption of MABR in industrial applications has been slow.
- Key factors influencing MABR performance including BT has been shown to correlate well with COD and NH_4^+ removal in IW, indicating that a thicker biofilm seems to be more effective in carbon and nitrogen RE.
- Current MABR models lack the resolution for large, dynamic systems. Future advancements must focus on real-time optimization and robust biofilm dynamics representation for effective industrial application.
- Further research is needed to optimize MABR biofilm process control, substrate diffusion, modeling, and operational performance, confirming its feasibility for industrial applications with additional testing.

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CRediT authorship contribution statement

Gennaro Dicataldo: Writing – review & editing, Writing – original draft, Methodology, Conceptualization. **Peter Desmond:** Writing – review & editing, Validation. **Mashael Al-Maas:** Writing – review & editing, Validation. **Samer Adham:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

All relevant data are included in the paper.

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