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► To cite this version:

Olga El Kik, Geoffroy Lesage, François Zaviska, Andres Sauvêtre, Marc Heran, et al.. Synergistic approach for enhanced wastewater treatment: Harnessing the potential of bioelectrochemical systems in integration with anaerobic membrane bioreactors. *Journal of Environmental Chemical Engineering*, 2024, 12 (4), pp.113162. 10.1016/j.jece.2024.113162 . hal-04594014

HAL Id: hal-04594014

<https://imt-mines-ales.hal.science/hal-04594014v1>

Submitted on 30 May 2024

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Synergistic approach for enhanced wastewater treatment: Harnessing the potential of bioelectrochemical systems in integration with anaerobic membrane bioreactors

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ARTICLE INFO

Keywords:

Anaerobic membrane bioreactor
Bioelectrochemical systems
Fouling mitigation
Sustainable energy recovery
Micropollutant removal

ABSTRACT

Anaerobic membrane bioreactor (AnMBR) technology emerges as a groundbreaking solution for domestic wastewater treatment, offering high effluent quality and substantial potential for neutral or positive energy balances. This is achieved through the production of renewable methane biogas, concurrently reducing aeration and sludge handling costs. Despite these strides, fouling mitigation persists as a pivotal challenge, constituting a significant portion of membrane-based bioreactors energy requirements. In response, AnMBR technology has evolved through seamless integration with bioelectrochemical systems (BES), specifically microbial fuel and electrolysis cells. This integration aims to address fouling challenges while enhancing resource recovery from wastewater. In this context, the paper centers on key performance parameters, including removal efficiency, fouling mitigation strategies, and biogas generation, shedding light on the latest advancements in these integrated technologies. By scrutinizing recent progress, it identifies research gaps and needs, emphasizing potential optimizations such as integrating granular sludge or catalytic electrodes for micropollutant degradation. Overall, this review contributes to a deeper understanding and advocates for the broader application of the integrated AnMBR-BES approach, playing a pivotal role in advancing sustainable wastewater treatment.

1. Introduction

The surging global population and its increasing demand for fresh, safe water necessitate a substantial shift toward reclaimed and reused water. Simultaneously, the rising burden on wastewater treatment plants emphasizes the urgent need to safeguard the environment from the effects of urban and industrial wastewater discharges. Traditional methods like activated sludge and anaerobic digestion (AD) have long

been employed to address these challenges, focusing on AD's energy recovery opportunities. However, they encounter limitations in energy consumption, sludge management, and recalcitrant pollutant removal. Globally, countries strive to align policies with circular economy principles to enhance urban wastewater sector integration. As part of the United Nations Sustainable Development Goals, goal 6 emphasizes universal access to clean water and sanitation. European directives, notably the European Water Framework Directive (EU WFD), set more

Abbreviations: AD, Anaerobic Digestion; AFMBR, Anaerobic Fluidized Bed Membrane Bioreactor; AnGS, Anaerobic Granular Sludge; AnMBR, Anaerobic Membrane Bioreactor; AnOMBR, Anaerobic Osmotic Membrane Bioreactor; BES, Bio Electrochemical System; CE, Coulombic Efficiency; CEM, Cation Exchange Membrane; CFD, Computational Fluid Dynamics; COD, Chemical Oxygen Demand; CTC, Chlorotetracycline; DIET, Direct Interspecies Electron Transfer; EAB, Electrochemically Active Bacteria; EABR, Electrochemical Anaerobic Biofilm Reactor; EET, Extracellular Electron Transfer; EPS, Extracellular Polymeric Substances; EU, European Union; FO, Forward Osmosis; GAC, Granular Activated Carbon; HRT, Hydraulic Retention Time; LMH, Liters per Square Meter per Hour; MEC, Microbial Electrolysis Cell; MFC, Microbial Fuel Cell; MLSS, Mixed Liquor Suspended Solids; NCC, Net Cathodic Chamber; OLR, Organic Loading Rate; OMP, Organic Micropollutant; PAC, Powdered Activated Carbon; PVDF, Polyvinylidene Fluoride; SCOD, Soluble Chemical Oxygen Demand; SMP, Soluble Microbial Product; SRT, Solid Retention Time; TCOD, total Chemical Oxygen Demand; TOC, Total Organic Carbon; TP, Total Phosphorous; TMP, Transmembrane Pressure; TS, Total Solids; TSS, Total Suspended Solids; UASB, Upflow Anaerobic Sludge Blanket; UN, United Nations; VFA, Volatile Fatty Acid; VS, Volatile Solids; WFD, Water Framework Directive; ZVI, Zero-Valent Iron.

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<https://doi.org/10.1016/j.jece.2024.113162>

Received 5 February 2024; Received in revised form 2 May 2024; Accepted 23 May 2024

Available online 28 May 2024

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stringent nutrient and micropollutant removal requirements, reinforcing the need for advanced treatment facilities. This underscores the importance of research, technological innovation, and efficient organics and micropollutant removal to achieve energy neutrality, fostering a sustainable and resilient wastewater management future.

The anaerobic membrane bioreactor (AnMBR) process has gained widespread attention as a sustainable wastewater treatment technique, capturing interest from researchers and the commercial sector [1]. By seamlessly integrating membrane technology with anaerobic processes, AnMBR demonstrates distinct advantages over traditional methods, making it a promising solution for global wastewater treatment challenges. Its key strengths include reduced energy demand without the need for aeration, energy recovery through methane production, superior solid-liquid separation, minimized sludge production, high organic removal rates, and adaptability to diverse wastewaters [2]. Furthermore, it exhibits exceptional ammonia tolerance [3,4] and high chemical oxygen demand (COD) removal surpassing physico-chemical methods that are often plagued by exorbitant costs and secondary pollutant generation [5]. Despite its remarkable advantages, AnMBR faces a critical limitation in membrane fouling [6], primarily attributed to rapid fouling in anaerobic systems, necessitating effective fouling mitigation strategies for long-term viability in wastewater treatment applications [7].

Addressing the membrane fouling challenge in AnMBR has become a prominent research focus, with the aim of ensuring the technology's continued applicability and efficiency in wastewater treatment. Researchers have been exploring various innovative approaches for fouling mitigation and membrane lifespan extension [8]. Physical and chemical cleaning methods, while effective, raise concerns about potential membrane damage and secondary pollution, prompting the search for sustainable alternatives [7]. Recently, combining bioelectrochemistry with AnMBR has shown promise in preventing fouling through low-intensity electric fields [9]. Moreover, bioelectrochemical systems (BES), specifically microbial electrolysis cell (MEC) and microbial fuel cell (MFC), have demonstrated their exceptional capability in degrading micropollutants [10]. Consequently, integrating these advanced systems with AnMBR offers a cutting-edge advantage, providing enhanced robustness, reduced fouling, improved degradation rates for organic load and micropollutants, and significantly lower energy consumption, a substantial step towards achieving a net positive energy system.

While a recent comprehensive review on the topic of AnMBR coupled with electrochemical systems has already been published, with a primary focus on electro-assisted integration [11], the present review aims to complement and expand upon the existing work. A broader perspective will be provided by including all relevant studies involving integration with both MFC and MEC. Importantly, this paper not only complements existing literature but also facilitates a more thorough and holistic examination of the subject matter. By summarizing operational parameters and performance indicators, it unlocks potential needs for optimizing the AnMBR-BES's performance such as incorporating granular sludge or catalytic electrodes. The review also considers future regulations and requirements, such as micropollutant removal, making it a valuable resource for researchers and practitioners alike.

2. Overview of AnMBR and BES systems

2.1. AnMBR: system fundamentals and applications in wastewater treatment

The recognition of wastewater as a valuable resource for fresh water, compounds of interest, and energy has gained momentum, with the AnMBR standing out as a transformative technology [12]. Similar to aerobic membrane bioreactors, AnMBRs integrate membrane separation with biodegradation processes to eliminate pollutants from wastewater, but with a key distinction - the absence of oxygen. Within the system, organic waste undergoes biodegradation in the bioreactor mixed liquor

and along the membrane biofilm, facilitating the separation of treated water through physical mechanisms at the membrane level [13]. The true essence of AnMBRs lies in their fundamental reliance on the anaerobic process driven by specific microorganisms, unfolding four stages: hydrolysis, acidogenesis, acetogenesis, and methanogenesis [14]. At the latter stage, two main equations occur due to two groups of microorganisms: acetoclastic methanogens that use acetate to produce methane ($\text{CH}_3\text{COOH} \rightarrow \text{CH}_4 + \text{CO}_2$), and hydrogenotrophic methanogens that convert hydrogen to methane by using carbon dioxide as an electron acceptor ($\text{CO}_2 + 4 \text{H}_2 \rightarrow \text{CH}_4 + 2 \text{H}_2\text{O}$) [15]. Those methanogenic archaea, known for their slow growth rate ranging from 12 h to 1 week [3], result in high solid retention time (SRT: 19–300d [16]) beneficial for high methane yield and achieving positive energy wastewater treatment plants [17]. In AnMBR treating domestic effluents, methane yield ranges from 0.1 to 0.25 L- CH_4 /gCOD removed. Despite the need for higher temperatures to optimize methanogenic bacteria activity, resulting in increased energy costs, the net energy balance for AnMBR systems (-0.1–0.1 kWh/ m^3) remains notably advantageous compared to conventional aerobic systems (-0.2 to -0.8 kWh/ m^3) [18].

In general, two main configurations exist depending on the membrane location: side stream or external configuration (Fig. S1b), and submerged or immersed configuration (Fig. S1a). Due to its lower energy consumption and smaller space requirement, the submerged configuration gained popularity for domestic wastewater treatment [3, 13,16]. It can also serve as a two-stage system, isolating the membrane tank from the main bioreactor for cleaning without disrupting the microbial community. Contaminants in the retentate can be recirculated for further biodegradation and to maintain desired mixed liquor suspended solids (MLSS) levels, making it suitable for full-scale applications [19].

AnMBRs are versatile in wastewater treatment (Fig. S1), initially demonstrating remarkable COD removal rates exceeding 95% in synthetic wastewater tests [16]. Their application extends to municipal wastewater (COD removals > 85%, total suspended solids (TSS) removals > 99%) and industrial wastewater, particularly in treating food-processing wastewater, achieving COD removal efficiencies above 90% with organic loading rates (OLRs) ranging from 2 to 15 kg COD/ m^3 /d [16]. AnMBRs further demonstrated efficiency in removing pharmaceutical compounds and organic micropollutants (OMPs) [20, 21], as well as treating high-solid-content streams and leachate, achieving COD removal rates of approximately 90% under specific conditions [16].

Beyond achieving high effluent quality across diverse wastewater types, another notable advantage of the AnMBR process is biogas recovery. Nevertheless, economic and technical viability requires careful consideration due to challenges such as high membrane costs, loss of dissolved methane, and increased membrane fouling [3,4]. Fouling, a major limitation of MBR technology in all forms [3], poses additional costs for AnMBRs due to higher biomass concentration, leading to increased transmembrane pressure (TMP), impaired economic efficiency, and requiring more frequent cleaning and additional nitrogen removal treatments [22]. To overcome these challenges, combining AnMBRs with BES based on microbial redox reactions has recently emerged as a promising strategy, to reduce membrane fouling, enhance energy recovery, and reduce environmental impact.

2.2. BES: advancing wastewater treatment and resource recovery

BES represent a sustainable approach, converting chemical energy from wastewaters and lignocellulosic biomass into valuable resources like electrical energy, hydrogen, and biochemicals through microbial electrochemical technology [23]. These systems use ion-exchange membranes to separate the anodic and cathodic compartments, where electroactive microorganisms oxidize electron donors on the anode while reducing electron acceptors, allowing for direct or shuttle-mediated electron transfer to electrodes, thus enabling

electricity generation.

Within the realm of BES, various types such as MFCs, MECs, microbial electrosynthesis cells, microbial solar cells, and microbial desalination cells are distinguished by their configurations and applications [24]. Notably, MFCs and MECs stand out as the most extensively researched and well-established BES to date, with successful pilot-scale systems developed [25,26]. MFCs demonstrate remarkable dual functionality in efficiently decomposing diverse organic compounds and

aiding resource recovery [27], while MECs produce hydrogen instead of water at the cathode by combining electrons and protons through external voltage under anaerobic conditions [23,28]. By supplying external energy, MECs enable bacteria to electrochemically convert inorganic carbon into valuable biofuels and chemicals, including biohydrogen, biomethane, acetic acid, and alcohols [29]. Both MFCs and MECs operate within two principal configurations (Fig. S2): a single-chamber arrangement encompassing a cathode and an anode, and

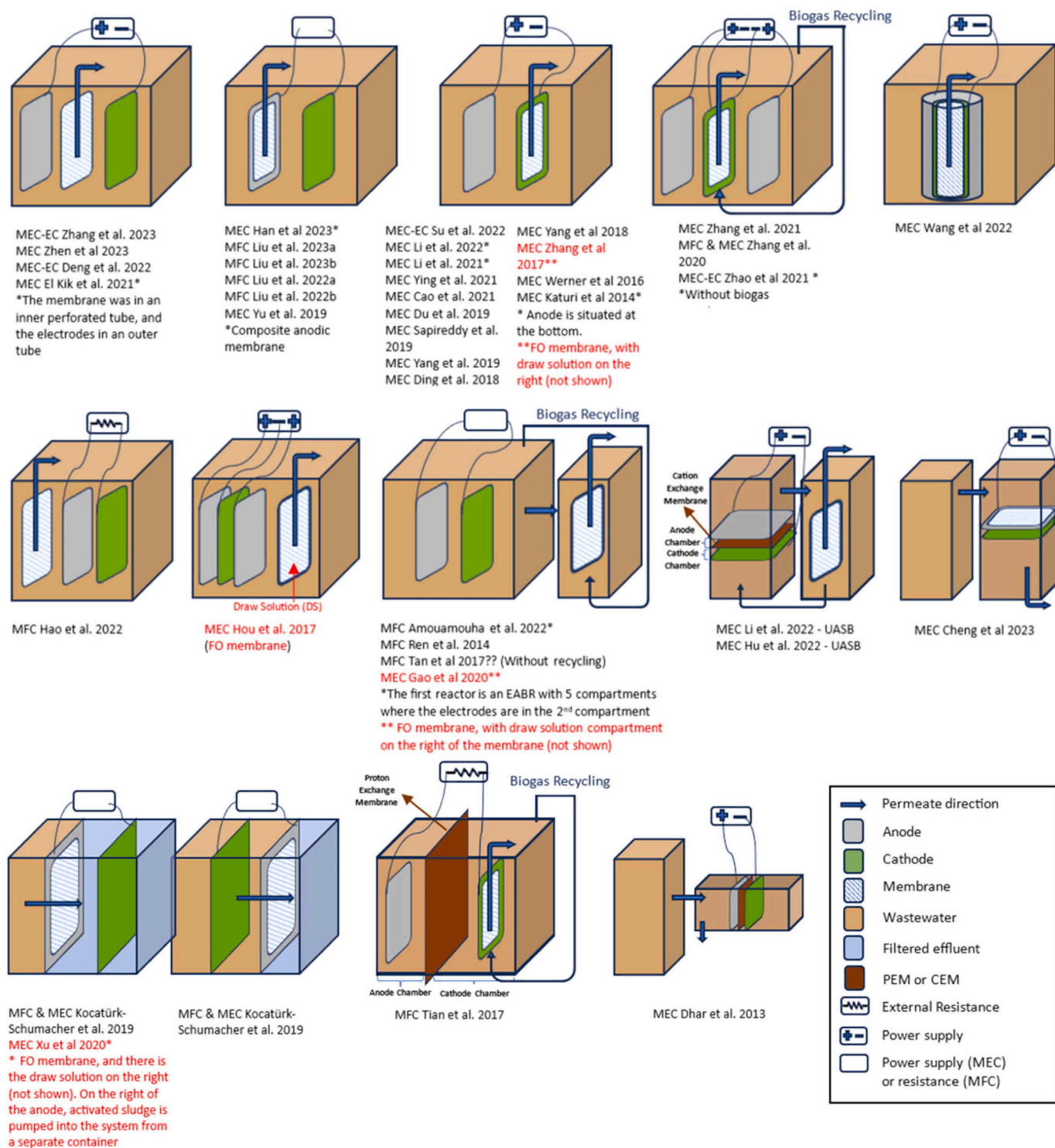


Fig. 1. An extensive illustration providing a panoramic view of the diverse configurations explored in all conducted studies. The depiction of reactor forms, electrodes, membranes, or inter-module distances is intentionally generalized, offering a comprehensive overview of the range of experimental setups employed in the exploration of AnMBR-BES systems for wastewater treatment.

a two-chamber setup integrating a proton exchange membrane that acts as a Cation Exchange Membrane (CEM) [30]. While the CEM in MECs ensures hydrogen purity, the single-chamber configuration mitigates membrane-related issues like fouling and maintenance costs, presenting a trade-off between hydrogen quality and operational simplicity.

Recently, there has been significant attention on coupling MEC systems with AD to optimize biogas production and treatment efficiency [31]. Electrochemically Active Bacteria (EAB) or exoelectrogenic bacteria in MECs such as *Geobacter* species (*G. sulfurreducens* specifically) and *Desulfuromonas acetexigens* showcase a distinctive respiratory pathway known as the extracellular electron transfer (EET) pathway, allowing them to transfer metabolically generated electrons from organic waste oxidation to insoluble extracellular electron acceptors at the anode, producing high current densities [32]. However, this hydrogen is vulnerable to conversion into methane by hydrogenotrophic methanogens [33], which can decrease the system's Coulomb efficiency (CE) [34].

While MFCs and MECs partially reduce energy consumption, achieving the required water quality for reuse autonomously remains challenging [35], necessitating integration with membrane filtration processes [32]. In this context, recent studies extensively explored combining MFCs with MBRs, and later integrating MFCs and MECs with AnMBRs. In cases of seamless integration, this coupling not only manages fouling but also significantly enhances methane and hydrogen yields, particularly in MEC systems, while alleviating acclimation period constraints in larger-scale anaerobic processes.

3. Coupling AnMBR and BES

Several recent studies have explored AnMBR technology coupled with BES, specifically MFC and MEC, under varied operating conditions (OLR, hydraulic retention time (HRT), temperature (T), pH, flux) and electrochemical characteristics (electrode types, material, specific surface area, external voltage, external resistance). Fig. 1 provides an overview of diverse AnMBR-BES configurations, featuring generalized reactor forms, electrodes, membranes, and inter-module distances, summarizing experimental setups. Despite a thorough examination, the intricate nature of AnMBR-BES systems and multifaceted interactions among variables hindered clear correlations between operating parameters and performance outcomes. Design parameters of reported AnMBR-BES applications are in Table 1, while Tables 2 and 3 detail AnMBR-MFC and AnMBR-MEC performances, respectively, focusing on COD removal under various OLRs (0.21–18 kgCOD/L.day), HRTs (0.33–10 days), SRTs (10–50 days), and temperatures (15–55°C), reflecting operational variations due to wastewater characteristics and system configurations. Subsequently, a comprehensive examination will delve into the various configurations and the impact of key parameters for each AnMBR-MFC and AnMBR-MEC systems.

3.1. Integration of MFCs with microfiltration-based AnMBRs

Studies have investigated the integration of MFCs with AnMBRs to enhance effluent quality. Initially, these systems were not fully integrated; instead, the two processes were separated, either in two compartments [50,69,71] (with the MFC serving as a pretreatment step before the AnMBR [69,71], or by implementing an electrochemical anaerobic biofilm reactor (EABR) with an external membrane cell [50]) or in a single-reactor system with split anodic and cathodic chambers [74]. In general, these systems demonstrated high COD removal efficiencies, mainly associated to the participation of electroactive bacteria and the MFC's ability to utilize inhibitory substances, like simple aromatic proteins and tryptophan protein-like substances found in the loosely bound extracellular polymeric substances (EPS) of the sludge [75]. Additionally, they demonstrated reduced membrane fouling rates with lower TMP values compared to the control AnMBR reactors. However, more recent studies have presented truly integrated hybrid

AnMBR-MFC systems employing a conductive membrane for both the bio-electrochemical and membrane functions [9,41,40,48,49,59,64]. The summarized performance data in Table 2 demonstrates the high removal efficiencies ranging from 75% to 99.8%, apart from one study demonstrating the applicability of the system on high strength brewery wastewater which had COD removals between 18.6% and 69.1%. These removal rates were observed across various system configurations, HRTs spanning from 0.375 to 4 days, OLRs ranging from 0.299 to 16.485 kgCOD/m³.day, and temperatures varying between 23 and 45°C.

3.1.1. COD removal in AnMBR-MFC systems

3.1.1.1. Two-stage system. Both small-scale laboratory and pilot-scale reactors demonstrated favorable efficiencies when treating synthetic and real wastewater. The MFC unit used as a pretreatment for the AnMBR achieved high removal efficiencies above 90%, indicating an increase of over 8% [71,74]. Notably, COD removal in the coupled system treating high-strength palm oil mill effluent showed a significant dependence on temperature [69]. An increase in COD removal from $81.36 \pm 0.76\%$ to $97.66 \pm 0.41\%$ was observed as the temperature rose from 25 to 45°C, with conditions at 45°C promoting increased biomass growth and superior COD removal performance. However, a further increase in temperature to 55°C was not favorable for COD removal. Similar results were observed in the AnMBR alone, with significant variability in COD removal suggesting potential factors such as propionate accumulation, reduced microbial community diversity, and increased toxicity of intermediates contributing to process instability. Compared to a control reactor devoid of electrochemistry, the AnMBR-MFC displayed higher COD removal efficiencies ($81.36 \pm 0.76\%$ to $97.66 \pm 0.41\%$), versus the control reactor ($75.50 \pm 0.42\%$ to $95.60 \pm 0.30\%$), highlighting the pronounced positive impact of coupling MFC with AnMBR. In a parallel context, a pilot-scale reactor with a 72 L capacity demonstrated a COD removal efficiency of 90% for the AnMBR-MFC system, surpassing the electrochemistry-absent system that achieved an 80% removal rate [50].

3.1.1.2. Single-stage system. Similar to the two-stage system, the performance evaluation of the coupled AnMBR-MFC system primarily focused on assessing COD removal efficiency, consistently achieving high removal rates ranging from 80% to 99.81% (Table 2), and often surpassing the control AnMBR [49]. This is attributed to the pivotal role of the bioelectric field in the AnMBR-MFC, which augmented methanogenic activity and facilitated sludge adaptation to changing conditions, resulting in greater tolerance to influent load variations [49]. Notably, studies investigating the impact of varying the external resistance from 0 to 1200 Ω revealed that the highest COD removal efficiency (> 95%) was consistently reported at an external resistance of 10 Ω [9, 48]. Furthermore, in a comprehensive study that explored both AnMBR-MFC and AnMBR-MEC systems, COD removal performance remained consistently high, whether in the presence or absence of an electric field (0.5 V), with mean COD values recorded at $95 \pm 0.4\%$ and $94 \pm 1\%$, respectively [59]. Recent studies also indicated that while the integration of MFCs in AnMBRs demonstrated significant COD removal, self-generated micro-electric fields and microcurrents within the system did not significantly influence COD removal performance or the functionality of the microbial communities responsible for organic matter degradation [40,76]. These findings indicate that, irrespective of micro-electric fields, the incorporation of electrodes provides certain enhancements to the system, likely by serving as physical surfaces for microbial attachment or facilitating a conductive interface for electroactive microorganisms.

3.1.2. Fouling propensity in AnMBR-MFC systems

3.1.2.1. Two-stage system. In terms of fouling mitigation, the TMP rate

Table 1
Summary of AnMBR-BES Configurations.

Ref.	Configuration/ Process	Feeding Regime	Filtration & Membrane cleaning Mode	Volume (L)	Wastewater Type	Cathode Type ^a	Anode Type	Membrane type	Membrane Pore Size (μm)
[36]	Two-stage system (UASB - Membrane & MEC module)	Continuous	-	2.25	Synthetic (glucose & Starch)	C cloth	CNT	CNT	0.38
[37]	Single-chamber AnMBR-MEC	Continuous	119-min off and 1- min on mode	10	Food Waste & Sewage Sludge	CF & Ti/RuO ₂ mesh plate	PVDF covered with SS and CF within Ti/ RuO ₂ mesh plate	PVDF flat sheet	0.4
[38]	Single-chamber AnMBR-MEC	Continuous	Biogas circulation	10	Food Waste & Sewage Sludge	Ti mesh	CF	PVDF flat sheet	0.2
[39]	Single-chamber EC-AnMBR	Continuous	Replacement at 60 kPa	-	Synthetic	Gr plate	Mg plate & Gr Plate	PVDF HFM	0.1
[40]	Two- communicating- chamber AnMBR-MFC	Continuous	-	3.6	Synthetic (glucose)	CF-incorporating SS mesh	PPy/AQDS/ PTFE	PPy/AQDS/ PTFE	0.45
[41]	Single chamber AnMBR-MFC	Continuous	-	3.6	Synthetic (glucose)	SS mesh	PPy/AQDS/ PTFE	PPy/AQDS/ PTFE	0.45
[42]	Two-stage system (UASB-MEC & membrane)	continuous	Membrane leaning cycles	1500	Pesticide	SS	SS mixed with C	HF Curtain	0.05
[43]	Two-stage system (UASB-MEC & membrane)	continuous	Three membrane cleaning operations	1500	Pesticide	Corrugated sandwich SS mesh	Sandwich SS mesh mixed with CF	HF flat plate	0.05
[44]	Single-chamber AnMBR-MEC	Continuous	-	2.4	Synthetic (glucose) + Chlortetracycline	SS mesh	C fiber brush	SS mesh	10
[45]	Single-chamber AnMBR-MEC	Continuous	Several membrane cleanings	2.4	Synthetic (glucose)	SS (1) & Conductive HFM (2)	Ti mesh	Conductive HFM	-
[46]	Single-chamber EC-AnMBR – Continuous & intermittent fields	Continuous	Filtration/ relaxation ratio of 8:2 min	1.6	50% Domestic 50% Synthetic	Modified Conductive Al ₂ O ₃ flat Ceramic	Fe sheet	Modified Conductive Al ₂ O ₃ flat Ceramic	-
[47]	Two-stage EC- AnMBR: AD - Membrane & Electrodes	Continuous	Increasing membrane area	3	Synthetic kitchen WW	Conductive Ti membrane	Fe	Conductive Ti	0.22
[9]	Single chamber AnMBR-MFC	Continuous	-	21.66	Synthetic (glucose)	SS mesh with 2 CF layers	PPy/AQDS/ PTFE	PPy/AQDS/ PTFE	0.45
[48]	Single chamber AnMBR-MFC	Continuous	-	21.66	Synthetic (glucose)	CF	PPy/AQDS/ PTFE	PPy/AQDS/ PTFE	0.45
[49]	Single-chamber AnMBR-MFC	Intermittent / continuous	Chemical cleaning	2	Synthetic (glucose)	Air Cathode: non-woven C cloth, PTFE emulsion, C black, Pt C	CF	PVDF	0.1
[50]	Two-stage EABR & Membrane Cell	Continuous	Biogas backwash	72	Synthetic (molasses)	Fe plate	Fe plate	PVDF	0.22
[51]	Single-chamber AnMBR-MEC	Continuous	Filtration/ relaxation ratio of 8:2 min & gas recycling	10	Synthetic (glucose & Starch)	Reticular Ti	Double flat Ti	Modified PVDF	0.1
[52]	Single-chamber AnMBR-MEC	Continuous	-	2.4	Synthetic (glucose)	SS mesh	C fiber brush	SS mesh	10
[53]	Single-chamber AnMBR-MEC	Continuous	Chemical cleaning	0.5	Synthetic (glucose)	Conductive SiC ceramic	Gr	Conductive SiC ceramic	0.7
[54]	Single-chamber AnMBR-MEC	Continuous	-	3.2	Synthetic (glucose)	HF-PVDF-CNT	Cylindrical C Gr felt with a Pt core	HF-PVDF-CNT	0.4
[55]	Single-chamber AnMBR-MEC	Continuous	GAC fluidization	1.43	Synthetic (acetate)	Ni mesh	C Cloth	PVDF HFM	0.1
[56]	Two-chamber MEC & External membrane	-	Physical cleaning	2	Synthetic (glucose)	SnO ₂ Nps electrocatalytic	Microbial CF	CTA-ES FO	-
[57]	Single-chamber EC-AnMBR	Continuous	Physical cleaning	1	Synthetic	SS microfiltration	Fe plate & Gr plates	-	1–5
[58]	Single-chamber AnMBR-MEC	Continuous	-	-	Synthetic	SS wire mesh	TFN FO membrane	TFN FO	-

(continued on next page)

Table 1 (continued)

Ref.	Configuration/ Process	Feeding Regime	Filtration & Membrane cleaning Mode	Volume (L)	Wastewater Type	Cathode Type ^a	Anode Type	Membrane type	Membrane Pore Size (μm)
[59]	Single-chamber AnMBR-MFC	Continuous	Filtration/ relaxation ratio of 8:2 min & Gas recycling	10	Synthetic (glucose & Starch)	Double Ti in membrane module	double flat ZVI	hydrophilic modified PVDF	0.1
[60]	Single-chamber AnMBR-MEC	Continuous	-	2.7	Synthetic (acetate)	Ti mesh (built-in HF)	CF	External pressure HF PVDF	0.01
[61]	Single-chamber AnMBR-MEC	Batch	Chemical cleaning	0.28	Synthetic (acetate)	Ni-HFMs	Gr fiber brush with Ti core	Ni-HFMs	1
[62]	Single-chamber AnMBR-MEC	Continuous	Hydraulic cleaning	1	Synthetic (glucose)	CNTs-HFMs	Ti mesh	CNTs-HFMs	1.43–7.43
[63]	Single-chamber AnMBR-MEC	Continuous	-	1	Synthetic (glucose & complex sewage)	Six Gr rods	CNT	CNT	0.55
[64]	Single-chamber AnMBR-MFC	Batch & continuous	Electrochemical cleaning & Turbulence promotor	5	Synthetic (brewery)	Pt mesh	SS filter plate & PVDF With SS mesh	SS filter plate & PVDF With SS mesh	0.1, 0.5 & 0.45
[65]	Single-chamber AnMBR-MEC	Continuous	-	30	Synthetic (acetate)	Ti mesh (membrane built-in)	CF	PVDF HFM	0.01
[66]	Single-chamber AnMBR-MEC	Continuous	-	1	Synthetic (glucose)	Electro-enhanced CNTs-HFMs	Ti mesh	CNTs-HFMs	0.45
[67]	Single-chamber AnMBR-MEC	Batch	Physical cleaning	4.5	Synthetic (glucose)	NiFe layered double hydroxide Ni foam	Two CF	FO CTA	-
[68]	Single-chamber AnMBR-MEC	Continuous	Physical & chemical cleaning	8.6	Synthetic	SS mesh	C brush	CTA-NW	0.00037
[69]	Two-stage AnMBR-MFC	-	PAC	1 (MFC) 1 (AnMBR)	Palm oil mill effluent (POME)	Air Cathode	-	Conventional	-
[70]	Single-chamber AnMBR-MEC	Batch	Chemical cleaning	0.35 (Tubular) 0.165 (Rectangular)	Synthetic (acetate)	Gr-HFM	Gr fiber brush with Ti core	Gr-HFM	1
[35]	Single-chamber AnMBR-MEC	Continuous	-	0.35	Synthetic (acetate)	Ni-HFMs	Gr fiber brush with Ti core	Ni-HFMs	1
[71]	Two-stage AnMBR-MFC	Continuous	Fluidization	0.135 (MFC) 0.065 (AFMBR)	Domestic	Wet-proofed C cloth	Gr fiber brushes with a Ti wire core	PVDF HFM	0.1
[72]	AnMBR followed by dual-chamber MEC	Batch (MEC) Continuous (AnMBR)	-	0.4 (MEC) 5 (AnMBR)	Synthetic (glucose)	SS mesh	High density C fibers connected with a SS frame	HFM	0.04

^a C: Carbon; CNT: Carbon Nano-Tubes; CF: Carbon Fiber; Ti: Titanium; RuO₂: Ruthenium Dioxide; PVDF: Polyvinylidene Fluoride; SS: Stainless Steel; HFM: Hollow Fiber Membrane; SiC: Silicon Carbide; Gr: Graphite; Pt: Platinum; Ni: Nickel; SnO₂ Nps: Tin Dioxide Nanoparticles; TFN FO: Thin-Film Nanocomposite Forward Osmosis; Ni: Nickel; Fe: Iron; Mg: Magnesium; Al₂O₃: Aluminum Oxide; PPy/AQDS/PTFE: Polypyrrole/Anthraquinone-2,6-disulfonic Acid/ Polytetrafluoroethylene; ZVI: Zero Valent Iron.

in systems where MFC was used as a pretreatment was small of approximately 0.07 kPa/d at a permeate flux of 16 L/m².hr (LMH) in an anaerobic fluidized bed membrane bioreactor (AFMBR) [71] and around 0.57 kPa/d at a higher flux of 22 LMH and a notably high feed COD of 85858 ± 9623 mg/L [69]. It's worth noting that in these studies, fouling mitigation strategies were employed, such as using fluidization of granular activated carbon (GAC) at 170 mL/min rate with a 70–80% bed expansion [71] or the addition of 5 g/L extra pure charcoal powdered activated carbon (PAC) from the initial stage of biomass cultivation [69]. Similarly, a pilot scale reactor demonstrated effective fouling control, with an average TMP rate of 0.42 kPa/d [50]. This performance is mainly attributed to the use of biogas purging as a cleaning mechanism and the application of electric fields, which altered the properties of the mixed liquor and improved its filterability [51]. When the membrane was combined with the cathode in a separate chamber from the anode, also a significant reduction in membrane fouling was observed in the system, especially during the first period from 0 to 500 h (~ 0.3 kPa/d), primarily due to the decreased zeta potential of particles and reduced concentration of soluble microbial

products (SMP) in the cathodic mixed liquor. However, a subtle increase occurred after this period and only within 50 h to reach 31.9 kPa, known as membrane filtration TMP jump [77].

3.1.2.2. Single-stage system. In most of the reviewed studies, the impacts of the system configuration and operating conditions on membrane fouling were evaluated by investigating the TMP evolution, which characterized the membrane fouling behavior under constant-flux mode. In general, the results showed that the coupled system exhibited enhanced mitigation of fouling compared to the conventional AnMBR. Higher external resistance effectively controlled membrane fouling when treating wastewater with an influent COD of 1000 mg/L and Ca²⁺ concentration of 350 mg/L to study organic and inorganic fouling [40,41]. At low external resistances, the micro-current played a significant role in reducing membrane fouling, but inorganic fouling eventually caused blockage. Conversely, at high external resistances, the internal field strength increased, resulting in a substantial reduction in inorganic fouling but an increase in organic fouling leading to blockage. The micro-electric field was found to be more important than the current

Table 2
AnMBR-MFCs Operating Conditions and major performance parameters.

Ref.	Configuration / Differentiated Parameter	Operation period (d)	COD influent (mg/L)	T (°C)	HRT (d)	OLR (kgCOD/m ³ .d)	Permeate Flux (LMH)	Resistance (Ω)	COD Removal Efficiency (%)	TMP kPa/d	Voltage Generated (mV)	Internal Resistance (Ω)	Current or power density
[40]	Two-communicating chamber	9 25 38 48 62	1000	-	2.5	0.4	15.49	0 50 500 1000 1500	~ 90 ~ 90 ~ 90 ~ 90 ~ 90	3.42 1.2 0.79 0.62 0.49	- 13.7 ± 4.2 56.2 ± 5.0 102.6 ± 15.4 146.1 ± 19.3	- 750.1 626.9 606.2 573.3	- 46.13 mW/m ² 23.17 mW/m ² 21.21 mW/m ² 20.83 mW/m ²
[41]	Single-chamber	11	1000	-	2.5	0.4	15.5	0 50 500 1000 1500	86–90 86–90 86–90 86–90 86–90	3.3 1.3 0.9 0.8 0.6	- 16.7 ± 2.4 63.6 ± 4.1 108.7 ± 4.3 140.7 ± 4.7	- 1181.1 980.7 956.1 894.4	- 38.1 mW/m ² 46.9 mW/m ² 48.2 mW/m ² 50.5 mW/m ²
[9]	Single-chamber	12 101 70 62 53 29	1000	23–30	3.34	0.299	15.49	0 10 100 400 800 1200	75–90 75–90 75–90 75–90 75–90 75–90	2.53 0.30 0.43 0.49 0.58 1.05	- - - - - -	- 83.2 - 712.5 1406.9 9133.3	- 0.392 A/m ² 0.237 A/m ² 0.206 A/m ² 0.186 A/m ² 0.094 A/m ²
[48]	Single-chamber	21	-	-	-	-	15.49	0 10 400 1200	> 80 > 95 90 85	1.52 0.10 0.57 0.87	- - - -	- 5.9 23.2 31.7	- 0.454 A/m ² 0.278 A/m ² 0.113 A/m ²
[49]	Intermittent feeding Continuous feeding	35 40	1587 ± 9 3177 ± 11	35	4	-	0.5	1000	90–98.6 92.8–99.8	~ 0 0.46	107 ± 1 94 ± 4	- -	- -
[50]	Two-Stage EABR & Membrane	159	50–2400	35 ± 2	1.208	-	104–91	-	90	0.42	-	-	-
[59] ^a	Single-chamber	95	483 ± 16	35	0.417	1.15944	15	-	95 ± 0.4	~0.0364 (day 0–55) ~0.45 (day 56–95)	-	-	-
[64] ²	0.5 μm membrane 0.1 μm membrane*	14	25144 – 25720	30	-	1	~85–0 (8 d) ~80–~0 (14 d) ~51–~2 (14 d) ~27–~2 (9 d) ~48–~21 (8 d) ~42–~22 (8 d) ~60–~20 (10 d)	-	- 61.8 18.6 67.7 43.1 37.3 51.5	held at 100 kPa (flux behaviors analyzed with a fixed TMP in this study)	- - - - - -	- - - - - -	- -
[69]	Two-stage	0.087	85858 ± 9623	25 35 45 55	0.521	16.485	22	-	81.4 ± 0.8 84.3 ± 0.8 97.7 ± 0.4 85.2 ± 0.6	0.57 0.57 0.57 0.57	- - - -	- - - -	- - - -
[35]	Dual-chamber	25	-	-	0.5 (anode) 0.208 (cathode)	-	6.7	100	91.6	~ 0.3 (0–500 h) ~10.512 (550–600 h)	132	-	1.16 W/m ³ NCC
[71]	Two-stage	50	210 ± 11 (tCOD)	25	0.375	-	16	1000	92.5	0.07	590 ± 30	-	89 ± 6 mW/m ² (max) ³

^a Studies that involved the MEC system by applying an external voltage to the electrodes: This study observed nearly identical results with a voltage of 0.5 V; however, fouling increased significantly when employing ZVI with an electric field. Improved outcomes were demonstrated through the application of electrochemical cleaning (MEC system) with a voltage of 0.241 V, leading to heightened permeate fluxes over an extended operational period

Table 3
AnMBR-MECs Operating Conditions and major performance parameters.

Ref.	Configuration / Differentiated Parameter	Operation period (d)	COD influent (mg/L)	T (°C)	HRT (d)	OLR (kgCOD/m ³ .d)	Permeate Flux (LMH)	Voltage (V)	COD Removal Efficiency (%)	Methane yield (L-CH ₄ /gCOD _{removed})	TMP (kPa/d)	Net Energy (Kwh/m ³)
[36]	Two-stage system (UASB-Membrane & MEC)	34	500	35 ± 2	0.5	1.0	17	0.2	~ 96	0.2	0.6	-0.758
[37]	Single-chamber	158	1.2 ± 0.2 gCOD/d	35 ± 1	50	1.19 ± 0.16–2.49 ± 0.19	7	0.8	98.3 ± 0.5	0.0762 ± 0.0491 to 0.144 ± 0.0318	~ TMP stable at 1–1.2 kPa	-
[38]	Single-chamber	209	289.3 ± 78.8	35	10	0.77 ± 0.11	60–40	0–1.4	> 96	0.04 ± 0.01–0.12 ± 0.02 L/day (at 0.8 V)	~ stable at 25 kPa	+31.82 kJ/day (at 0.8 V)
[42]	Two-stage system (UASB-MEC & membrane)	23 26 32 22	10350 ± 145 10378 ± 159 10391 ± 127 10326 ± 126	20 ± 3	4 3 2 1	2.588 3.459 5.196 10.326	15	0.4 V/cm	~ 79 ~ 81 ~ 79 ~ 65	- - - -	0.99 1.09 1.19 1.44	- - - -
[43]	Two-stage system (UASB-MEC & membrane)	23 26 32 22	10365±142	20 ± 3	4 3 2 1	2.59±0.04 3.46±0.05 5.2±0.06 10.33±0.13	15	0.4	> 70	0.27 ± 0.01 0.29 ± 0.01 0.26 ± 0.01 0.22 ± 0.02	0.99 1.09 1.19 1.44	- - - -
[44]	Single-chamber	52 58 35 16 61	~500 + 0 CTC ~500 + 2.5 CTC ~500 + 15 CTC ~500 + 45 CTC ~500 + 45 CTC	35 ± 1	0.583	0.8571	15.9	0 0 0 0 0.7	≥ 96 ≥ 96 ≥ 96 ~ 80 - < 68 > 94–72	~ 450 mL/d ~ 450 mL/d ~ 450 mL/d ~ 380 mL/d ~ 380 mL/d	0.21 0.19 0.11 0.75 0.16	- - - - -
[45]	Stainless steel Cathode Conductive HFM	100	1200 (TOC)	33 ± 1.0	0.375	-	7	1.2	93.33 (TOC) 95.76 (TOC)	260.4–564 mL/d 250.88–922.88 mL/d	3.57 3.38	- -
[51]	Single chamber	99	483 ± 16	35	0.417	1.15944	15	0.5	93 ± 1	0.32	0.20	-
[52]	Single-chamber	20 28 36 53 23	~500	35 ± 1	0.583	0.857	15.9	0 0.5 0.7 0.9 0	> 94% > 94% > 94% > 94% > 94%	~ 0.18 ~ 0.19 ~ 0.22 ~ 0.22 ~ 0.2	0.12 0.12 0.12 0.36 0.48	- - - - -
[53]	Single-chamber	25	2000	-	1	2	12.5	0.5 1 2	~ 80 ~ 83 ~ 90.3	~ 0.28–0.36 m ³ /m ³ /d ~ 0.31–0.38 m ³ /m ³ /d ~ 0.40–0.47 m ³ /m ³ /d	1.81 ± 0.52 kPa/h ~ 0.7 kPa/h 0.39 ± 0.10 kPa/h	-0.02406 -0.02382 -0.01545
[54]	Single-chamber	204	300–330	20–34	1.5	0.21	-	-	~ 60–90	51.85 mL/d	0.26	-
[55]	Single-chamber	95	463	20–30	1.50	0.31	6.5	0.7	85–96	0.2	0.07	-0.131
[56]	Two-chamber MEC & External FO membrane	30	4000	35 ± 1	-	-	5.01–0.4 in 15 days	1.2	96.4 ± 2.7 - 95.4 ± 2.3	0.27–0.28	-	-
[58]	Single-chamber (FO membrane)	10	500 (TOC)	25	-	-	0.063 LMH/h	0.5	> 95% (TOC)	-	-	-
[60]	Single-chamber	35	10000	28–30	2	4.8	0.56	0.8	tCOD ~ 92	2.3–6.2 L/d	Stages 1&2: 0.208 Stage 3: 1.3	-
[61]	2 m ² /m ³ cathode surface area 4 m ² /m ³ cathode surface area	75	-	-	-	-	16	0.7	89.17 ± 0.15 (acetate) 89.46 ± 0.09 (acetate)	0.0466 ± 0.0100 m ³ /m ³ /d 0.0280 ± 0.0100 m ³ /m ³ /d	~ 3.93 (over 30 d) ~ 2.67 (over 30 d)	-

(continued on next page)

Table 3 (continued)

Ref.	Configuration / Differentiated Parameter	Operation period (d)	COD influent (mg/L)	T (°C)	HRT (d)	OLR (kgCOD/m ³ .d)	Permeate Flux (LMH)	Voltage (V)	COD Removal Efficiency (%)	Methane yield (L-CH ₄ /gCOD _{removed})	TMP (kPa/d)	Net Energy (Kwh/m ³)
	8 m ² /m ³ cathode surface area								93.76 ± 0.63 (acetate)	0.0175 ± 0.0088 m ³ /m ³ /d	~ 1.76 (over 21 d)	
[62]	1.43 μm membrane	15	500	15–20	1.333	0.375	-	-1.2	> 95	1.13–111.12 mL/gVSS.d	0.67	-2.55 to 51.46 kJ/d
	2.43 μm membrane		1000			0.75					0.67	
	3.43 μm membrane		2000			1.5					0.67	
	4.43 μm membrane		3000			2.25					0.87	
	5.43 μm membrane		6000			4.5					1.33	
	6.43 μm membrane	10	6000		0.667	9					2.30	
	7.43 μm membrane	12	6000		0.333	18					2.25	
[73]	Single-chamber	38	2000	38	0.667	3	-	0.2	< 50	-	26.49 ± 6.41 (kPa)	-
								0.3			5.52 ± 1.15 (kPa)	
[65]	Single-chamber	60	7500	30	1.5	5	-	0	52.6	-	-	-
		60						0.4	~ 62	-	-	-
		72						0.6	70.6	-	-	-
		92						0.8	63	-	-	-
		98						1	~ 60	-	-	-
		-						1.2	49.4	-	-	-
[66]	Single-chamber	30	503	25	0.333	1.509	-	1.2	> 98	-	1.17	-
[67]	Single-chamber (FO membrane)	70	275 ± 10	25.0 ± 0.5	-	-	8.7 ± 0.3 to 4.0 ± 0.2	0.8–1	> 93	0.19	-	-
[68]	Single-chamber (FO membrane)	42	2000	35 ± 1	-	-	3.772–1.2	0.5	80.61	0.281	-	+0.000149 kWh/d
[70]	Single-chamber (tubular)	52	320	25	-	-	6.9	0.7	99.7 ± 0.5 (acetate)	0.02 ± 0.004 m ³ /m ³ /d	0.88	-0.16 ± 0.09
	Single-chamber (rectangular)	72						0.7	85.0 ± 11.3 (acetate)	0.04 ± 0.03 m ³ /m ³ /d	0.68	-0.11 ± 0.42
								0.9	83.3 ± 3.7 (acetate)	0.09 ± 0.033 m ³ /m ³ /d	0.32	-0.48 ± 0.37
[35]	Single-chamber	5	320	25	-	-	6.9	0.5	> 95 (acetate)	0.011 ± 0.002 m ³ /m ³ /d	0.81	-0.16 ± 0.07
		56						0.7		0.028 ± 0.006 m ³ /m ³ /d	0.81	-0.24 ± 0.1
		9						0.9		0.03 ± 0.005 m ³ /m ³ /d	-	-0.03 ± 0.04
[72]	AnMBR	-	5300	24 ± 2	10	0.530	1	-	53 ± 4	198 ± 36 mL/d	(low flux)	-
	Dual-chamber MEC	-	2500 ± 200	25 ± 1	1	2.500	-	1.25 ± 0.05	15 ± 4	-	-	-
					4	0.625			42 ± 10	-	-	-
					8	0.313			47 ± 3	-	-	-
[39]	Mg anode	120	2500–3000	36 ± 1	2	1.375	-	0.6	> 95	-	~ 35 after 6 d	-
	Gr anode								> 95	-	~ 45 after 6 d	-
[46]	Continuous electric field	30	250–350	~ 28	-	-	41.2	5	76.4 ± 2.7	0.0627	1.76	-1.042
	1-on/9-off electric field								~ 87	0.0544	2.50	-0.104
	2-on/8-off electric field								91.3 ± 4.6	0.0584	1.88	-0.208
	4-on/6-off electric field								~ 80	0.0593	2.00	-0.417
[47] ^a	Two-stage: AD - Membrane & Electrodes	30	5000	37 ± 1	7.6	0.66	-	0.6	> 96	-	~ 40 kPa	-
		70	12000			1.58			> 96			
		80	20000			2.63			> 95			
[57]	Fe anode	60	480–570	37 ± 1	1.042	0.504	40 mL/h	0.6	80.38	0.33	80 kPa in 6.5 d	-
	Gr anode								78.5	0.33	80 kPa in 4.5 d	-

^a Note: A control reactor (AnMBR-Ti) without external voltage (MFC-like configuration) was employed for comparison. The electro-AnMBR consistently exhibited superior performance, maintaining a lower TMP (reduction between 20% and 42%) compared to the AnMBR-Ti. Additionally, the electro-AnMBR featured larger sludge floc size, contributing to enhanced fouling mitigation compared to the AnMBR-Ti reactor.

in controlling membrane fouling, with a higher impact on fouling reduction (95.34% versus 67.19%) [40]. In similar studies, however without the addition of Ca^{2+} , it was also observed that increasing the external resistance from 0 to 10 Ω resulted in a decrease in the TMP fouling rate (as shown in Table 2) with low TMP rates of 0.1 and 0.304 kPa/d in the studies under a permeate flux of 15.49 LMH [9,48]. However, a further increase in the external resistance from 10 to 1200 Ω caused fouling rates to start increasing slightly again until reaching around 0.87 and 1 kPa/d.

In another study, the TMP of the coupled system experienced slow growth throughout the operational cycle, with a significantly lower TMP value (18.513 kPa on day 75) compared to the control reactor which required frequent chemical cleaning (exceeding 35 kPa on days 38, 56, and 73) [49]. It was found that the bioelectric field enhanced electrostatic repulsion between microorganisms, stripping pollutants from the membrane surface, and reducing fouling. Comparison of SMP and EPS between the systems showed significantly lower levels in the coupled system than in the conventional AnMBR (65.3% and 43.1%, respectively). Despite a higher protein-to-carbohydrate ratio, the integrated system experienced lower membrane fouling compared to the control. Additionally, the bioelectric field influenced particle size, sludge charge, and microbial community structure, collectively contributing to minimizing membrane fouling [49]. Furthermore, the application of zero valent iron as anode proved to suppress membrane fouling by 20%, mainly due to the improved filterability of mixed liquor, reaching 20 kPa after a long period of 95 days of operation [59]. However, the fouling rate increased 17 times faster with a 0.5 V external voltage due to severe organic and inorganic foulants, forming a thick Fe-enriched fouling layer [59].

A singular study investigated the variations in permeate fluxes under a constant TMP of 100 kPa/d [64]. It found that fouling rates were influenced by membrane pore size, with smaller filter grade of 0.1 μm exhibiting superior flux compared to the 0.5 μm pore size. The study employed two fouling mitigation techniques: electrochemical cleaning, achieved through a conductive filtration membrane and on-site hydrogen production, and the use of a turbulence promoter, a potential strategy for minimizing fouling by modifying the flow dynamics to induce increased turbulence. Both methods had a positive effect on fouling reduction, with electrochemical cleaning improving permeate flux more than two-fold when using the 0.1 μm membrane [64].

3.1.3. Electrochemical properties AnMBR-MFC systems

3.1.3.1. Two-stage system. The primary objective of MFCs is to leverage microbial metabolism for efficient wastewater treatment while concurrently generating electric current as a valuable by-product. Initial studies investigating the coupling of MFCs with AnMBRs focused on measuring key parameters such as voltage production, power density, and the system's CE. Research exploring the use of a separator electrode assembly MFC as a pretreatment for AnMBR demonstrated consistent voltage output of 590 ± 30 mV with a 1000- Ω external resistor. The overall CE of the system was 18%, while the maximum power density achieved reached 89 ± 6 mW/m² [71]. It is worth noting that this power density was lower than those observed in previous electrochemical studies [78], probably due to the use of a weaker feed with a total COD (tCOD) of 217 mg/L. In the more integrated system, where the membrane and cathode were combined in the catholyte, the generated voltage was lower at 120 mV when using a 100- Ω external resistor [74]. The system achieved a maximum power density of 1.16 W/m³ net cathodic chamber (NCC) and a CE of approximately 10.3% for acetate oxidation. Likewise, these results were considered moderate compared to other studies, primarily due to two factors. First, the stainless-steel cathode used was found to be less efficient compared to carbon-based materials, which provide a higher surface area for the microbial attachment and electron transfer [79]. Second, the low nitrate loading

rate of 0.12 kg/m³NCC/d limited the availability of nitrate for microbial respiration, thereby impeding efficient electron transfer within the system [80].

3.1.3.2. Single-stage system. With increasing the external resistance, the voltage generated across the anode and cathode increased [40,41], while current densities decreased [9,48]. Adding 50 mg/L Ca^{2+} concentration weekly until reaching 350 mg/L resulted in higher maximum power densities in the coupled system than previously reported [81,82], likely due to improved electron conduction from higher Ca^{2+} concentrations [40,83]. Polarization curve analysis revealed that the internal resistance of the system decreased with increasing external resistance [40,41]. In contrast, the charge transfer resistance (R_{ct}), which is solely dependent on external resistance, increased and was further influenced by the cathodic reduction reaction rate and electroactive bacteria activity on the anodic membrane surface [48]. Specifically, at R-1200 Ω , the R_{ct} value was 11.79 Ω , which was 11.79 times higher than at R-400 Ω and 74.93 times higher than at R-10 Ω [48]. However, the best electrochemical performance, characterized by current densities of 0.392 and 0.4536 A/m² and a CE of 5.12%, was achieved at a low external resistance of 10 Ω , surpassing results from similar systems [84]. Remarkably, under identical operating conditions, significant correlations were observed between the reactions on the membrane's surface and the electrochemical performance of the anode, with improvements in current efficiency primarily attributed to the consumption of organic matter during bacterial growth and the electricity generated at the anode [85].

During a continuous 1800-h monitoring cycle, the coupled system was successfully initiated at 414 h, maintaining a stable voltage output of 107 ± 14 mV and a maximum power density of 342.23 mW/m² until a significant increase in influent COD concentration at Stage III. This change led to a voltage stabilization at 95 ± 4 mV and a reduction in power density to 225.63 mW/m², with the bioelectric field remaining stable throughout the cycle despite the low abundance of exoelectrogens due to complex flora. However, following the influent load increase, voltage output and power density decreased by 11.2% and 34.1%, respectively, as exoelectrogens required time to adapt to the higher organic load. The CE, initially below 7.79%, fell to 1.9% later in the experiment, likely because glucose was consumed by various non-exoelectrogens. This efficiency decrease may be attributed to the peak reproduction and metabolism of exoelectrogens, coupled with an increase in non-exoelectrogens, collectively leading to the reduced CE [49].

Investigations into the effects of membrane pore size (ranging from 0.1 to 0.5 μm) [64,86] and cathode placement (either on the retentate or permeate side) [64] showed no significant impact on current densities, which are generally influenced by anode material, operational parameters, and feed composition. Using real wastewater, current densities usually vary between 1 and 10 A/m² [87]. Different anode types, including stainless steel filtering anodes and hybrid PVDF-stainless steel anodes, were tested, demonstrating good electrochemical performances with current densities reaching 5.4 A/m² and around 2.5 A/m², respectively, facilitated by turbulence promoters [64].

3.1.4. Biogas generation and energy efficiency in AnMBR-MFC systems

3.1.4.1. Two-stage system. When considering methane generation, it is crucial to acknowledge that not all studies included explicit reporting of methane produced, as it may not be the primary goal of the MFC system. Furthermore, the units of measurement used across different studies often varied, presenting a challenge when attempting direct comparisons between results. The use of MFC as a pretreatment for AnMBR systems was found to be unfavorable for biogas generation, with methane production reaching only approximately one-tenth of the potential achieved through methanogenesis alone [71]. However, the

exact causes remained unclear [71].

Only one study provided a calculation of the energy balance for the system [71], finding that the energy required for GAC fluidization and permeate pumping was 0.0186 kWh/m³, which is lower than previously reported values in AFMBRs [88]. On the other hand, the energy produced solely by the MFC was 0.0197 kWh/m³ (as the methane production was negligible), sufficient to offset the operational requirements. However, it should be noted that scaling up may alter this energy balance, necessitating further system optimization [71].

Single-stage system

Only two studies have reported on the biogas generation performance of the single-stage AnMBR-MFC system. Initial observations revealed that methane production in the coupled systems significantly exceeded that of the conventional AnMBR, with increases of 137.4% and 161.9%, driven by direct electron transfer between species [89]. As the operation progressed, methane production accelerated, concluding at 1.2 times that of the conventional AnMBR. This enhancement was largely attributed to the bioelectric field, which boosted methanogen activity and overall methanogenic performance. Additionally, the coupled system maintained lower levels of volatile fatty acids (VFAs) compared to the conventional system, which indicated reduced acidification and contributed to a shorter start-up period. The bioelectric field also improved VFA utilization, speeding up COD removal in the early stages of the experiment [49]. Moreover, a separate study employing zero-valent iron (ZVI) as the anode achieved high methane yields of around 0.32–0.33 L/(gCOD_{removed}), approaching the theoretical maximum of 0.35 L/(gCOD_{removed}) [90]. Further analysis revealed that only 3% of the methane was produced via hydrogenotrophic methanogenesis, suggesting that the high yield was primarily due to favorable conditions such as suitable OLR and consistent neutral pH, rather than the influence of the electric field [59].

3.1.5. Microbial community dynamics in AnMBR-MFC systems

Several studies have highlighted the importance of microbial community composition and dynamics in AnMBR-MFC systems, particularly for membrane fouling control in single-stage systems. Significant enrichment of electroactive microorganisms, such as *Trichococcus*, *Acidovorax*, and *Desulfovibrio*, on the anodic membrane surface indicated their involvement in electron transfer and membrane fouling reduction [9,48,91,92]. Other key taxa including *Firmicutes*, *Patescibacteria*, *Bacteroidota*, *Actinobacteria*, *Proteobacteria*, *Chloroflexi*, *Verrucomicrobiota*, *Desulfobacterota*, and *Myxococcota* also contributed to fouling degradation through their diverse functionalities [40,41]. While *Firmicutes* are known for their ability to degrade EPSs and capability of EET [93], anaerobic fermentation bacteria, such as *Patescibacteria* and *Bacteroidota*, demonstrated their capability to degrade high-molecular-weight compounds [94,95]. *Actinobacteria* contributed to metabolites and toxic compound degradation [96], whereas *Proteobacteria*, as electrochemically active bacteria, facilitated EET and influenced membrane fouling [97]. Moreover, the presence of filamentous bacteria, including *Chloroflexi* and *Verrucomicrobiota*, *Desulfobacterota*, and *Myxococcota* was observed in the filter cake layer, positively impacting fouling degradation by their complex synergistic effects [98]. Notably, electrochemically active bacteria such as *Anaeromusa*, *Lactococcus*, and *Lactobacillus* were predominant on the membrane surface, showing strong fouling resistance, whereas *Clostridium* and *Christensenellaceae_R - 7_group* showed differential abundances and highlighted their role in fouling development [40,41].

In general, the bioelectric field influenced these interactions by promoting *Actinobacteria* growth and reducing *Chloroflexi* and *Megaspheera* abundances, thereby mitigating fouling [99]. It was found that the abundance of *Desulfobacterota* ensured stable operation of the bioelectric field due to its high electron transfer efficiency [100]. Genus-level abundance analysis revealed higher abundance of *Brooklawnia* and *Trichococcus*, with *Brooklawnia* contributing to hydrolysis and acid production [101] and *Trichococcus* decomposing complex

organic matter [89]. Additionally, the presence of the acidophilic bacterium *Desulfovibrio* in AnMBR-MFC improved hydrolytic acidification, enhancing methanogenic performance. It exhibited higher methanogenic and exoelectrogenic abundance, with dominant *Methanosaeta* contributing to improved acid resistance. Furthermore, syntrophic relationships were evident, as bacteria such as *Syntrophobacter* and *Smithella*, were positively correlated with methanogens, enhancing the overall methanogenic activity [102]. An exceptional performance resulted from the synergistic interaction of *Desulfovibrio*, *Methanosaeta*, and *Methanosarcina*, highlighting the significance of the bioelectric field environment in facilitating these complex microbial dynamics [49]. Therefore, understanding microbial dynamics in AnMBR-MFC systems highlights key bacterial taxa for fouling control and promises improved strategies through focused manipulation and further investigation of their interactions with fouling processes.

3.2. Integration of MECs with microfiltration-based AnMBRs

3.2.1. Biodegradation and removal of soluble substances in AnMBR-MEC systems

High removal efficiencies of COD were consistently achieved in both AnMBR-MEC and AnMBR systems (> 90%) even with long operation periods, and in most cases the electrochemical-AnMBR outperformed the conventional system reaching removal rates with COD > 95% [36–38,44,52,62,66]. Even in pilot-scale studies with high-strength pesticides wastewater influents, AnMBR-MEC maintained stable average COD removal rates ranging from over 70% to around 81% [42, 43]. In terms of nutrient removal, effective membrane separation was demonstrated by high removal rates of particulate nitrogen and particulate sulfur, reaching 98.1 ± 0.9% and 98.0 ± 1.4%, respectively [37]. Moreover, high removal efficiencies of total solids (TS) (> 80%), and volatile solids (VS) (> 90%) were achieved [38], mainly attributed to the excellent interception capabilities of microfiltration membranes [103]. Additionally, studies employing Fe sacrificial anodes in electrochemical AnMBRs demonstrated significant enhancements in phosphate removal, achieving a remarkable 55% increase in kitchen wastewater treatment [47], and approximately 43% higher TP removal rates when treating synthetic wastewater (AnMBR-MEC with Fe anode: 83.16%, AnMBR-MEC: 63.02%, AnMBR: 58.19%) [57]. Substantial improvements in TP removal (170–370%) were also observed under various electrical exposure modes in mixed domestic and synthetic wastewater treatment [46], highlighting the effectiveness of electrochemical AnMBRs in achieving high phosphate removal efficiencies and overall contaminant removal across diverse wastewater scenarios.

In an attempt to test the coupled system performance in the degradation of chlortetracycline (CTC), the system achieved > 96% COD removal at open circuit with low influent COD concentrations (< 30 mg/L in the supernatant and < 20 mg/L in the effluent) and 0–15 mg/L CTC (day 0–182). When influent CTC concentration increased to 45 mg/L, COD removal decreased to < 68% after 15-day exposure at open circuit (day 183–198). However, discontinuing CTC addition in the influent (day 199–219) led to a gradual recovery of COD removal to 98%. Notably, subsequent dosing of 45 mg/L CTC at an external voltage of 0.7 V (day 222–280) maintained COD removal above 94% for the initial 15-day exposure, but gradually decreased to 72% after 58-day exposure, demonstrating favorable performance with voltage application [44].

Testing the effect of voltage on performance revealed that the application of gradient voltages in AnMBR-MEC systems had no impact on COD removal up to 0.7 V but resulted in an 81% increase in supernatant COD concentration at 0.9 V [52]. Promising results were also obtained with high removal efficiencies demonstrated for acetate and COD. Moreover, soluble COD (sCOD) removal rates exceeded 95% at various applied voltages, resulting in low permeate sCOD concentrations [35]. Rectangular AnMBR-MEC achieved acetate removal rates above 83%, while tubular systems achieved rates exceeding 99% [70]. Furthermore, lower TSS concentrations were observed in rectangular

AnMBR-MEC compared to tubular configurations, attributed to higher current densities at the anode [70]. These findings emphasize the potential of the integrated system to achieve high removal efficiencies of COD and acetate, with applied voltages playing a significant role.

3.2.2. Fouling propensity in AnMBR-MEC systems

In general, the application of an electric field was observed to effectively suppress membrane fouling development, particularly at low flux. Notably, at a constant flux of 7.1 LMH, the electrochemical AnMBR maintained a stable and low TMP of 1–1.2 kPa over 158 days, effectively controlling fouling while preserving membrane permeability despite high TS concentration and sludge viscosity. The carbon felt-PVDF composite anodic membrane in the AnMBR-MEC demonstrated superior fouling control compared to the single membrane in AnMBR [37]. While operating at a low and constant flux is a well-established approach to prevent significant biofouling during the initial treatment phase [104], the application of an electric field proved effective in suppressing membrane fouling development even at high flux rates. Despite initial aggravated membrane fouling at 0 V attributed to the increased TS in the bioreactor, the electrostatic repulsion force and electrochemical oxidation reduced membrane fouling, resulting in a stable TMP of around 25 kPa for over 120 days [38].

It's worth noting that as the HRT decreased, both suspended and attached sludge exhibited a noticeable decrease in EPS content. Compared to the conventional AnMBR system, the AnMBR-MEC showed a significant reduction (1.71 ± 0.01 times, $p < 0.01$) in EPS concentration in suspended sludge. This reduction suggested that electrochemical stimulation could effectively reduce EPS accumulation and mitigate membrane fouling [43,51,105]. In many cases, the AnMBR-MEC has demonstrated a considerably slower fouling rate when compared to the conventional AnMBR. Studies have reported reductions of approximately 55% [45] and 23% with TMP values decreasing from 0.31 kPa/d to 0.24 kPa/d after the application of an electric field on Day 20 [51]. Additionally, the analysis of extracted organic foulants showed a decrease in total organic carbon (TOC) (10%), polysaccharide (7%), and protein (52%) deposition with the application of an electric field [51]. The induced electrophoresis effectively removed charged polysaccharides and proteins, the main components of membrane foulants, from the mixed liquor [106], further supporting the efficacy of the electrochemical-AnMBR in fouling control.

Furthermore, studies demonstrated the potential of electrochemical AnMBRs with Mg or Fe sacrificial anodes in reducing membrane fouling and enhancing performance. An electrochemical AnMBR with a Mg anode achieved up to a 30% reduction in membrane fouling attributed to alterations in sludge characteristics induced by electric fields and the presence of struvite precipitates, hindering adhesion to the membrane surface [39]. Similarly, in kitchen wastewater treatment, an electrochemical AnMBR with an Fe anode achieved a notable 50% reduction in TMP through coagulation from $\text{Fe}^{2+}/\text{Fe}^{3+}$ ions, increasing floc size and porosity [47]. Lower EPS levels, achieved through increased *Methanosarcina* abundance in systems with electrocoagulation and electrochemistry (100–150% increase compared to systems without) [47], were also facilitated by long-term Fe^{3+} dosage and electric fields, which are known to stimulate *Methanosarcina* growth and reduce EPS production, thereby diminishing membrane fouling and enhancing system resilience [107,108]. Additionally, intermittent electric fields effectively reduced irreversible fouling resistance and enhanced anti-fouling capabilities, leveraging electrostatic repulsion and electrocoagulation during different fouling stages [46]. Integration of electrocoagulation with AnMBRs extended membrane service cycles, with average duty cycles increasing significantly compared to conventional AnMBR systems (109.68% in AnMBR-MEC with the integration of electrocoagulation and 45.16% in AnMBR-MEC compared to AnMBR) [57]. Electric fields also facilitated gas scouring on the cathodic membrane, and a positive correlation between applied voltage and electrostatic repulsion between EPS and the cathodic membrane was noted [47,109].

As noted from the previous discussion, two distinct strategies were employed in mitigating membrane fouling within the MEC configuration: utilizing the membrane either as the anode or the cathode. When the membrane acted as the anode, electron transfer with microbes enhanced microbial activity and accelerated anodic oxidation, maintaining a dynamic balance between foulant decomposition and fouling growth. This process potentially upregulated adenosine triphosphate generation and metabolic rates through redox regulation. Conversely, when the membrane acted as the cathode, electrostatic repulsion and in-situ biogas production were key factors in restraining biofouling. Although a recent study demonstrated very low TMP with an anodic membrane for over 158 days of operation [37], most studies coupling MEC with AnMBR employed a cathodic membrane, investigating fouling mechanisms. It was observed that increasing hydrogen production rates by elevating the applied voltage from 0.5 to 0.9 V resulted in a noticeable decrease in TMP and facilitated self-cleaning of the cathodic membrane surface through enhanced hydrogen bubble formation [35]. The low cathode potential and localized high pH at the cathode electrode surface further contributed to fouling reduction in the AnMBR-MEC system compared to the control reactor [35]. Similarly, investigations into reactor configuration and operating parameters revealed that rectangular AnMBR-MEC operated at higher voltage (0.9 V) exhibited lower TMP than those at lower voltage (0.7 V), highlighting a correlation between current density, hydrogen production rates, and TMP. Modifying the configuration by increasing the specific cathode surface area and reducing electrode spacing may also lead to reduced fouling and lower TMP [70]. In addition, a comparison of different AnMBR-MEC configurations with varying cathode specific surface areas (2, 4, and $8 \text{ m}^2/\text{m}^3$) revealed significantly lower TMP in systems with larger membrane areas, attributed to factors such as bubble size, distribution, nucleation sites, and rates of hydrogen production [61]. The study further identified an inverse relationship between TMP and hydrogen production rates, indicating that higher rates correlated with lower TMP. During pH disturbance periods, biofilm thickness differences influenced TMP behavior, with thicker biofilms being more susceptible to detachment due to the in-situ hydrogen evolution reaction [35,61,70]. Notably, a very low fouling rate (0.0659 kPa/d) was reported in a study that partially separated the membrane from the electrodes by placing it in a perforated concentric cylinder and positioning the electrodes externally [55]. The effective fouling mitigation was speculated to be attributed to two primary factors: 1) the repelling effect of electrical fields on foulants from the membrane surface, hindering their attachment [81], and 2) a configuration that provided distinct environmental niches (GAC, anode, cathode) for microbial colonization and growth, which resulted in lower suspended solids and mitigated membrane fouling, as indicated by electron balance analysis [55].

3.2.3. Electrochemical properties of AnMBR-MEC systems

Investigating electrochemical properties in AnMBR-MEC coupling is imperative for optimizing energy generation, system performance, and efficient configurations. This optimization involves enhancing electron transfer kinetics, minimizing energy losses, and informing decisions regarding electrode size, material, and spacing. Numerous studies have examined the influence of electrochemical factors on AnMBR-MEC performance and methane production, consistently highlighting the impact of applied voltage on system behavior.

The application of voltage has been found to enhance electrical conductivity, electron transport activity, and biofilm formation on the cathode, thereby improving electron transfer in the AnMBR-MEC [37]. This enhancement promoted biofilm stability, influenced microbial metabolic pathways, and affected the redox potential of the AD system [110–112]. For instance, voltage increase from 0.5 to 0.9 V has been associated with the enrichment of electroactive bacteria and higher current density, with a significant increase of up to 300% [52]. Moreover, increasing the applied voltage from 0.7 V to 0.9 V led to higher current density ($11.1 \text{ A}/\text{m}^3$ to $15 \text{ A}/\text{m}^3$) and maximum hydrogen

production rate ($0.20 \text{ m}^3\text{H}_2/\text{m}^3/\text{d}$) [35]. However, prolonged operation at higher voltages resulted in a decline in current density and hydrogen production due to methanogen adaptation [52]. The externally applied voltage also influenced microbial community composition, electrochemical conversion of CO_2 and H_2 to methane, and the growth and metabolic activity of microorganisms [45], resulting in a decrease in total resistance and an increase in potential difference [54]. Conversely, lowering the voltage to 0.5 V resulted in decreased current and biogas production rates but increased methane content (93%) due to extended batch duration and complete hydrogen conversion. In addition, the use of GAC was found to be disruptive to the electrochemical performance, resulting in CE values between 41% and 58% [55]. Notably, the removal of GAC initially reduced biogas yield but gradually restored system performance.

In terms of reactor design, rectangular AnMBR-MEC configurations exhibited higher current densities ($29 \pm 2 \text{ A}/\text{m}^2$ at 0.7 V and $49 \pm 5 \text{ A}/\text{m}^2$ at 0.9 V) compared to tubular configurations ($10 \pm 1 \text{ A}/\text{m}^2$ at 0.7 V), attributed to reduced electrode spacing and increased specific cathode surface area (SCSA) [70]. Initially, CE values in the rectangular system exceeded 100% due to hydrogen recycling by exoelectrogens [70]. However, over 50 days of operation, the CEs decreased to 83% (0.7 V) and 79% (0.9 V). Subsequent cathode cleaning significantly increased the hydrogen content, possibly due to microorganism-driven hydrogen recycling, contributing to improved CEs, reaching 87% (0.7 V) and 95% (0.9 V). In contrast, the tubular reactor consistently exhibited CEs below 60%, indicating lower electron transfer and utilization efficiency [70]. In a similar study involving Ni-HFM cathode, the effect of SCSA on electrochemical performance was investigated [61]. The reactor with the highest SCSA of $8 \text{ m}^3/\text{m}^2$ achieved the highest CE of $97.20 \pm 0.18\%$, compared to $85.50 \pm 0.17\%$ with a SCSA of $4 \text{ m}^3/\text{m}^2$ [61]. Notably, chemical cleaning of the cathode bundles did not affect reactor performance, which may be attributed to the suitability of the Nickel cathode for the hydrogen evolution reaction, in contrast to the Graphene cathode used in a previous study [70].

3.2.4. Biogas generation and energy efficiency in AnMBR-MEC systems

External voltage and electrical stimulation promoted methane production in AnMBR-MEC systems. However, while a significant increase in biogas production rate was reported at 0.8 V ($0.18 \pm 0.12 \text{ L}/\text{L}_{\text{reactor}}/\text{day}$ versus $0.12 \pm 0.01 \text{ L}/\text{L}_{\text{reactor}}/\text{day}$ without external voltage), excessively high voltages of 1.0 V and 1.4 V resulted in decreased CH_4 production [38], mainly due to electrode corrosion and microorganisms inhibition [113]. The maximum energy recovery was achieved at 0.8 V (31.82 kJ/day), significantly surpassing the baseline of 3.12 kJ/day at 0 V. Another study examining voltages from 0.5 to 2 V found that operating at 1 V demonstrated minimal energy consumption ($24.06 \text{ Wh}/\text{m}^3$ at 0.5 V, $23.82 \text{ Wh}/\text{m}^3$ at 1 V, and $15.45 \text{ Wh}/\text{m}^3$ at 2 V) and notable anti-fouling performance [53]. Moreover, shortening the HRT from 96 to 24 h had also a positive effect on the average biogas production which increased from 810 ± 42 – $3652 \pm 85 \text{ L}/(\text{d}\cdot\text{m}^3)$, respectively [43]. In addition, the application of external voltage preserved system efficiency, even when exposed to high micropollutant concentrations. It was observed that at open circuit conditions, low concentrations of CTC dosing (ranging from 2.5 to 15 mg/L) had no discernible impact on CH_4 production. However, when the concentration was increased to 45 mg/L, system inhibition occurred, requiring an external voltage of 0.7 V to safeguard the system's continued optimal functionality [44].

In a series of studies, a consistent transition from hydrogen to methane as the dominant gas component was observed. Following a two-month acclimation period with an applied voltage of 0.7 V, biogas composition shifted to over 80% methane production. This resulted in the generation of $3 \text{ mol CH}_4/\text{m}^3$, equivalent to $2673 \text{ kJ}/\text{m}^3$ or $0.74 \text{ kWh}/\text{m}^3$, effectively meeting 47% of the electrical energy demand [35]. While the system didn't achieve energy neutrality, operating the AnMBR-MEC at 0.7 and 0.9 V required significantly less energy than

aerobic MBRs [35,70]. Notably, a positive energy balance was achieved only with 2-bromoethansulfonate addition, inhibiting acetoclastic methanogens and increasing hydrogen yields. Furthermore, rectangular reactors achieved higher initial hydrogen production rates, peaking at $0.41 \pm 0.08 \text{ m}^3 \text{ H}_2/\text{m}^3/\text{d}$ at 0.9 V after 30 days, while tubular systems exhibited faster methane generation [89]. Electron balance analysis revealed higher electron loss at 0.7 V compared to 0.9 V [70]. Remarkably, hydrogen and methane production rates were proportional to cathode surface area, with total electron recovery below 100%, suggesting the existence of other electron sinks [61]. Furthermore, GAC particles used for fouling control enhanced methane production (initial yield of $0.31 \text{ L}/\text{gCOD}$ removed) through direct interspecies electron transfer (DIET) between *Geobacter* and methanogens, although their removal resulted in a temporary reduction in biogas yield ($0.2 \pm 0.047 \text{ L}/\text{gCOD}$ removed, methane rate of $0.031 \pm 0.014 \text{ m}^3/\text{m}^3/\text{d}$) [55]. The net energy requirement for operating the reactors was found to be $0.131 \text{ kWh}/\text{m}^3$, lower than reported values for anaerobic processes (0.25 – $1.00 \text{ kWh}/\text{m}^3$) [88], suggesting potential supplementation with renewable energy sources like solar power [55,114].

Surprisingly, certain configurations demonstrated that the conventional AnMBR either slightly outperformed the AnMBR-MEC in terms of biogas production [54], or yielded comparable results [51]. In the latter study, both reactor types exhibited remarkable average methane yields of $0.32 \text{ L}/(\text{g COD removed})$, approaching the theoretical value of $0.35 \text{ L}/(\text{g COD removed})$. Notably, the conversion of applied electric energy into hydrogen at the cathode did not increase methane production, primarily due to the complex biological transformation of hydrogen into methane, linked to the cathode's positioning inside the membrane module, distant from the anaerobic sludge [51]. Therefore, despite the promising improvements in methane generation and dissolved CH_4 recovery from AnMBR-MEC implementation, various parameters can significantly impact system efficiency and total energy requirements. Thus, careful consideration of the configuration and positioning of different compartments within the system is crucial due to their complex nature and potential influence on performance.

3.2.5. Microbial community dynamics in AnMBR-MEC systems

In terms of methane production, several studies have highlighted the importance of hydrogenotrophic methanogens, such as *Methanobacterium* [35,37,44]. Specifically, their presence on the cathode with *Methanobrevibacter*, along with the absence of acetolactic methanogens, indicated that the occurrence of methane generation was primarily through hydrogenotrophic methanogenesis [35,70]. In another study, acetolactic methanogens *Methanosaeta* and *Methanosarcina* were found to be more dominant on the cathode compared to *Methanobacterium*. In fact, methane generation at the biocathode can occur through two pathways: one involves hydrogen, which is converted to methane by carbon dioxide reduction mediated by *Methanosarcina* members [115], and the other pathway involves direct electron transfer from the cathode to *Methanosaeta*, an acetoclastic methanogen capable of reducing carbon dioxide into methane [116]. The presence of acetoclastic methanogens was accompanied by the prevalence of *Geobacter* and *Desulfuromonas* on the GAC, which suggested the potential for direct electron transfer between the two [55,117], as *Geobacter* species are known to participate in bioelectrochemical oxidation through EET, facilitated by cytochrome c and conductive pili (e-pili) [51]. Also, the AnMBR-MEC showed presence of *Desulfuivibrio*, a genus of sulfate-reducing bacteria, suggesting their potential involvement in electron transfer processes [54,55,70]. Particularly, *Desulfuivibrio* dominance on the cathodes of rectangular AnEMBRs showed its ability to catalyze hydrogen production at low cathode potentials when electron acceptors are limited. This hydrogen production was subsequently consumed by hydrogenotrophic methanogens, contributing to bioelectrochemical methane production in the system [70].

In general, the microbial community underwent significant changes between the conventional system and the AnMBR-MEC. Notably, there

was a discernible shift in the bacterial-to-archaeal ratio, with bacteria exhibiting a higher prevalence compared to archaea in the integrated system [36]. As observed in some studies, certain bacterial taxa such as *Chryseobacterium* (2.6%), *Thermomonas* (2.4%), and *Gordonia* (1.4%) were exclusively present in AnMBR-MEC, playing important roles in co-metabolic degradation, amino acid and protein treatments, and EET, respectively [112,118,119]. Enrichment of functional microorganisms and the secretion of various enzymes facilitated the formation of anodic biofilm and enhanced the electron transfer rate in the AnMBR-MEC system [120]. Additionally, the enrichment of Firmicutes at the cathode was found to promote VFAs degradation and contribute to system stability. Moreover, the carbon fiber cathode in the electrochemical system exhibited an increase in basophilic methanogens, aiding in VFAs oxidation and hydrogen generation [45]. Notably, the application of external voltage potentially enhanced the removal of the micropollutant CTC at high concentrations of around 45 mg/L [44]. This was attributed to the increased abundances of tetracycline resistant genes (e.g., *tet* (32), *tetD-01*, *tetE*, *tetQ*, and *tetX*) in the sludge and the presence of the tetracycline degradation gene *tetX* in the sludge, supernatant, and effluent [44]. Furthermore, some studies revealed that bacterial communities on the cathode primarily consisted of Firmicutes and Bacteroidetes, with the dominant genus *Acetobacterium*, which belongs to the group of homoacetogens capable of producing acetate from CO₂ and H₂ [61,121].

These findings shed light on the intricate interplay within the microbial community and underscore the significance of specific microbial species and functional genes in the AnMBR-MEC system. Notably, hydrogenotrophic methanogens, EAB, homoacetogens, and methane-consuming acetoclastic methanogens play crucial roles in shaping methane production pathways, electron transfer processes, and the overall performance of AnMBR-MEC systems, collectively driving desired outcomes in terms of degradation, methane generation, and system resilience.

3.3. Integration of MECs with nanofiltration-based AnMBRs

The conventional approach in AnMBR typically employs microfiltration or ultrafiltration membranes for wastewater treatment. However, there is growing interest in anaerobic osmotic membrane bioreactors (AnOMBR) that utilize nanofiltration, offering distinct advantages. The incorporation of forward osmosis (FO) in AnOMBR reduces energy consumption by eliminating the need for additional pressure in water transport, effectively reducing membrane fouling. This has prompted efforts to combine MEC with AnOMBRs, aiming to capitalize on the potential benefits and synergies provided by these integrated technologies.

A novel MEC-AnOMBR system demonstrated excellent removal of sCOD demand and phosphorus, achieving stable removal rates of over 95% and 100%, respectively. By integrating an electrocatalytic-assisted MEC with AnOMBR, carbon dioxide in biogas was effectively reduced to formate using a SnO₂ nanoparticle electrocatalytic cathode, reaching a remarkable faradic efficiency of formate up to 85% at 1.2 V. Moreover, a significant increase in methane content and yield was observed, with the methane content rising from 55% to 90% at the end of the operation, in comparison to traditional AnOMBR [56]. Likewise, a similar system achieved excellent organic removal (> 93%) and phosphorus removal (> 99%), accomplishing around 65% and 45% recovery of PO₄³⁻ and NH₄⁺ from the influent, respectively. The maximum methane yield observed was 0.19 L CH₄/g COD, and the effluent exhibited low levels of methane (< 2.5 mL CH₄/L) [67]. In addition to the remarkable removal rates and methane production, the electro-assisted AnOMBR has shown the ability to reduce soluble microbial products (SMP) content by 26% and proteins/polysaccharides content by 15% [58]. The absolute value of the Zeta potential of the sludge mixture in MEC-AnOMBR was 1.2 times higher than that in AnOMBR, indicating stronger electrostatic repulsion. Furthermore, the new system exhibited 35.7% lower flux

decline rate than the conventional AnOMBR, with an overall wastewater treatment capacity nearly 1.5 times that of AnOMBR. A lower conductivity increasing rate (0.08 ms/(cm.d)) was also observed compared to AnOMBR without an electric field (0.11 ms/(cm.d)), indicating lower propensity for fouling in the system. In fact, this was proven as the MEC-AnOMBR exhibited a longer membrane operation cycle, (~ 1.27 times) and higher methane yield (11.07% increase) compared to the AnOMBR [68].

Nevertheless, to further optimize reactor treatment performance, it is crucial to focus on key factors such as modeling membrane flux, solute accumulation, and solute recovery/desalination, as they play a pivotal role in determining system efficiency and effectiveness [67].

4. Recent advances in BES: unlocking potential for AnMBR-BES synergy

4.1. Coupling MECs with anaerobic granular sludge

Anaerobic granular sludge (AnGS) has emerged as an invaluable biocatalyst in MECs and MFCs, owing to its remarkable capacity to significantly enhance anaerobic performance. Characterized by its dense microbial consortium, AnGS forms a substantial and electroactive biofilm pivotal in the successful development of these systems [122]. In traditional AD, issues such as a slow electron transfer rate and an imbalance in the production and consumption of electron donors often lead to process instability and the accumulation of VFAs [123]. To address these challenges, the concept of the DIET pathway (CO₂ + 8 H⁺ + 8 e⁻ → CH₄ + 2 H₂O) has emerged as a promising strategy. DIET facilitates direct electron transfer between microbial cells via exoelectrogens and membrane-bound proteins, resulting in rapid transfer rates and enhanced energy efficiency [123,124].

Recent research has explored the integration of MECs with AD to bolster the DIET pathway, where methane generation utilizes electrons from the cathode, while substrates in the anode chamber follow the classical pathway for conversion into methane [123,125]. Furthermore, the inclusion of AnGS within MECs has been studied and found to foster DIET interactions among diverse microbial species, facilitating efficient electron transfer from microbial cells to external electron acceptors. This process, known as EET, contributes to improved system performance and overall efficiency (Organic compound (e.g., acetate) + *Geobacter* + Electron Acceptor (e.g., solid surface or other microorganisms) → CO₂ + *Geobacter* (oxidized) + Reduced Electron Acceptor). Table 4 provides a comparative overview of key aspects related to the utilization of AnGS in traditional anaerobic systems and anaerobic systems enhanced with electrochemical processes. The table highlights differences in electron transfer mechanisms, metabolic pathways, microbial community composition, biofilm formation, biogas composition, start-up procedures, treatment efficiency, energy generation potential, and applications between the two approaches.

Among recent studies investigating the addition of AnGS to BES, a notable approach involved employing a complex of magnetite nanoparticles and AnGS (Fe-AnGS) to enhance the DIET pathway. This innovative strategy has proven highly effective in refining biogas and has provided valuable insights into the realms of electrical energy storage and conversion [122]. In addition, the coupling of AnGS with MECs has demonstrated its capability to effectively treat high-organic-load saline wastewaters, including challenging sources such as undiluted real bilge water [126,127]. This approach also efficiently degraded 2,4,6-trichlorophenol, resulting in slightly increased biogas production compared to the control MEC reactor [128], indicating promise for recalcitrant micropollutant degradation alongside biogas production. Moreover, intact AnGS demonstrated its potential as a highly effective biocatalyst in a MFC, enabling simultaneous carbon removal and electricity generation, offering several distinct advantages over conventional biocatalysts [129]. Firstly, the AGS's large surface area facilitated significant electrogenic bacterial growth, resulting in

remarkably higher current generation compared to standard MFCs operating at the same substrate level. Secondly, the CE showed a substantial improvement from 14% to 34%, significantly reducing coulombic losses from the methanogenic process [129]. Although the improvement may appear modest, it reflects the considerable potential of AGS in enhancing MFC performance. The presence of a thin biofilm derived from AGS on the anode electrode surface proved crucial, serving as a conduit for efficient electron transfer between the bulk AGS and the electrode [129].

Despite efforts in single MEC or AD reactors, there's limited research on enhancing the DIET pathway for methane production in AD-MEC and AnMBR-MEC systems, calling for dedicated investigations, particularly in conjunction with AnGS incorporation. The formation of granular sludge in AnMBR-MECs is challenging due to the complex interactions between anaerobic microbial processes and electrochemical dynamics, yet it is essential for improving treatment efficiency and stability. Strategic optimization of operational parameters, such as hydraulic conditions, OLRs, and nutrient availability, plays a vital role in promoting microbial metabolism and EPS secretion conducive to granule formation. The use of Computational Fluid Dynamics (CFD) can significantly contribute to this optimization, allowing for precise modeling and enhancement of the dynamic fluid field within the system, which is crucial for the even distribution of substrates and the minimization of disruptive shear forces [130]. For instance, carefully managing OLR stimulates microbial activity and EPS production, facilitating the aggregation of microbial communities into stable granules. Simultaneously, balancing hydraulic conditions, through techniques like gentle agitation or tailored flow velocities assessed by CFD simulations, helps support the development of robust microbial aggregates. Furthermore, optimizing electrode designs and potentials enhances the selective enrichment of electroactive microbial communities, particularly exoelectrogens such as *Geobacter* and *Shewanella*. These exoelectrogens are crucial as they facilitate DIET and generate EPS that acts as a structural binder, fostering microbial adhesion and aggregation. Efficient electron transfer by exoelectrogens not only maintains a balanced redox environment within the granules but also minimizes toxic byproducts, promoting syntrophic relationships essential for granular stability. This comprehensive approach not only facilitates the initial formation of granular structures but also secures their long-term stability, optimizing the overall efficacy of AnMBR-MEC systems.

4.2. The potential of AnMBR-MEC in micropollutant removal

To date, information on AnMBR-BES for treating OMPs-containing wastewater remains limited. As shown in Table 1, only two studies specifically addressed pesticide streams [42,43], and one assessed the system's capability in eliminating the micropollutant CTC in synthetic wastewater [44]. These studies primarily focused on high concentrations in the mg/L range and lacked comprehensive identification of transformation products and pathways. Meanwhile, concentrations of OMPs in wastewater, encompassing pharmaceuticals, industrial chemicals, personal care products, pesticides, and polycyclic aromatic hydrocarbons, typically span from ng/L to µg/L [5]. Substantial efforts have been dedicated to comprehending the behavior and transport of these OMPs within water systems due to their significant implications for human health and eco-environmental security [131]. European regulations, aligning with WFD objectives since 2000, actively pursue emission reduction. This commitment results in periodic identification of substances for discharge reduction, updating discharge limits, and establishing new limit values for micro-pollutants requiring additional treatment: "Certain non-domestic pollutants will have to be regularly monitored in the inlets and outlets of the wastewater treatment plants... New limit values will be established for micro-pollutants that require additional treatment". Ongoing research also focuses on unraveling the complexities of 'emerging' micropollutants, given their uncertain effects on aquatic ecosystems and potential exposure levels [5].

Table 4

Comparison of AnGS in Conventional Anaerobic Systems vs. AnGS in Anaerobic Systems with Electrochemistry.

Aspect	AnGS in Anaerobic System	AnGS in Anaerobic System with Electrochemistry
Electron Transfer	Microbial community relies on natural electron donors and acceptors in wastewater.	Introduction of external electrodes allows for direct electron transfer to/from microorganisms.
Metabolic Pathways	Microbes follow typical AD pathways (fermentation, acetogenesis, methanogenesis).	Electroactive microbes can engage in electrogenesis and electromethanogenesis as alternative pathways.
Microbial Community Composition	Reflects a diverse range of anaerobic microorganisms.	Selects for electroactive microorganisms adapted to electron transfer to/from electrodes.
Biofilm Formation	Biofilm formation on granular sludge particles.	Biofilm forms on electrode surfaces, potentially distinct from granular sludge structure.
Biogas Composition	Biogas mainly consists of CH ₄ and CO ₂ .	Biogas composition may vary due to electrochemical reactions affecting metabolic pathways. Expected composition: H ₂ , increased CH ₄ , CO ₂ and trace gases
Start-up and System Stability	Traditional start-up and operational considerations for anaerobic systems.	Requires specific start-up procedures and optimization for stable electrochemical performance.
Treatment Efficiency	Relies on microbial degradation and conversion of organic matter.	Enhances treatment efficiency by utilizing electrochemical reactions to supplement microbial processes.
Energy Generation/Potential	Limited energy recovery through biogas combustion.	Electrochemical reactions offer the potential for additional energy recovery in the form of electricity or increased biogas recovery.

Moreover, MBR and AnMBR technologies, featuring porous micro-filtration or ultrafiltration membranes, have effectively managed high organic loads and emerging contaminants. Despite the porous membrane's constrained rejection efficiency for some OMPs due to size differences, it indirectly enhanced overall removal efficiency by retaining colloidal organic carbon, primarily through biodegradation and membrane-mediated rejection. AnMBRs exhibited comparable efficiency in removing specific OMPs (e.g., caffeine, clozapine, and trichloro-carban) to aerobic MBRs [132]. However, these systems face limitations in addressing hazardous micropollutants at trace concentrations, especially persistent ones like ketoprofen, bisphenol A, and ibuprofen, due to their limited rates of anaerobic biotransformation [133]. This puts them at a disadvantage compared to aerobic MBRs, prompting research into improvement strategies. Promising solutions involve incorporating additives (GAC, PAC, Biochar, Fe₃O₄) and harnessing bio-electrochemical potential [134], providing avenues for enhancing micropollutant removal in anaerobic treatment systems.

BES have also demonstrated efficacy in reducing low concentrations of micropollutants [135–137]. Table 5 provides a concise overview of recent studies on the degradation of key recalcitrant pollutants, including concentration levels and removal efficiencies across various electrochemical systems. While proven effective for mg/L concentrations, their applicability remains limited for lower concentrations (µg/L and ng/L) typically found in urban wastewater. Notably, recalcitrant pollutants like diuron and imidacloprid persist at quantification rates of approximately 70% and 80%, respectively, in both influent and effluent streams of sewage treatment plants in mainland France, according to research by the reduction action for the discharge of hazardous substances into water.

Given the demonstrated effectiveness of both AnMBR and MEC systems in removing micropollutants, their collaborative synergy holds

promise. Furthermore, research on standalone electrochemical processes also concentrated on non-trace concentrations, indicating room for improvement in treatment systems. This underscores the critical integration of AnMBR and MEC as a promising avenue for addressing trace-level micropollutants.

5. Research gaps and future needs

The integration of AnMBR systems with BES represents a promising avenue for advancing wastewater treatment capabilities, with significant environmental and operational benefits. However, the reported work on this system still presents many research gaps, both in refining laboratory techniques and in transitioning these technologies to practical applications (Table 6). These efforts are crucial for improving efficiency, sustainability, and the applicability of wastewater treatment technologies.

- (1) Within the current research landscape, the predominant emphasis rests on synthetic wastewater, often featuring glucose or acetate as substrates. While this approach allows meticulous control over variables, there is an imperative need to extend investigations to encompass real complex wastewater scenarios, ensuring a holistic understanding that aligns with actual treatment challenges. Furthermore, despite the growing body of work,

a considerable knowledge gap persists regarding the removal kinetics and degradation pathways of micropollutants. While a singular study addressed CTC removal, it overlooked the investigation of its transformation products, signaling the need for systematic optimizations to enhance the interaction between anaerobic processes in AnMBR and electrochemical systems. Future research should emphasize these optimizations, particularly focusing on the degradation of recalcitrant micropollutants to align with stringent regulations, as well as the identification of micropollutant transformation products and evaluating their toxicity in final effluents. As evident in Table 5, while electrochemical processes alone have been employed for micropollutant removal, they haven't adequately addressed environmental concentrations, highlighting the potential of AnMBR-BES in this context. Simultaneously, research endeavors should encompass a meticulous examination of the intricate relationships among various operational conditions, including the impact of psychrophilic and thermophilic temperatures, influent variations mirroring real-world scenarios, and external factors affecting system performance. Innovations might include the use of adaptive control systems that dynamically adjust operational parameters, such as electric field strength and HRT, based on real-time data from sensor networks to optimize treatment processes.

Table 5
Various electrochemical processes and techniques for common contaminant removal.

Ref.	System Type	Wastewater Type	Initial contaminant concentration (mg/l)	Removal performance
[138]	Batch Electrolytic cell with bismuth-doped 3D carbon felt/PbO ₂ (CF/Bi-PbO ₂) anode	Synthetic - Diuron	40	97.6% after 25 min
[139]	Batch Electrolytic cell with porous 3D carbon felt/PbO ₂ (CF/ β -PbO ₂) and conventional planar graphite /PbO ₂ (G/ β -PbO ₂)	Synthetic - Diuron & Atrazine & Simazine	40	98.8% after 50 min (CF/ β -PbO ₂) 75% after 50 min (G/ β -PbO ₂)
[140]	Porous CF/ β -PbO ₂ and planar titanium /PbO ₂ in a three-dimensional electrochemical reactor (3DER) equipped with GAC particle	Synthetic - Diuron	40	100% after 50 min (CF/PbO ₂) 75% after 50 min (titanium/PbO ₂)
[141]	Cylindrical glass reactor as electrolysis cell with iron-copper graphite felt (Fe-Cu/HGF) cathode and platinum sheet anode	Synthetic - Diuron	20	100% after 35 min
[142]	Photoelectrochemical oxidation (PECO), electrochemical oxidation (ECO) and photocatalytic oxidation (PCO) using carbon fiber anode and Pt counter cathode	Synthetic - Diuron & Atrazine	0.06	PECO & ECO: 100% within 1 h at 2 V PCO: 70% after 5 h
[143]	Electrolysis cell with Ti-porous/blue TiO ₂ NTA anode and Ti-plate cathode	Synthetic - Carbamazepine	20	99.75% at 8 mA.cm ⁻² after 60 min electrochemical oxidation
[144]	Electrochemical cell using laser-induced graphene (LIG) electrodes	Synthetic - Carbamazepine & MB	2	64% of carbamazepine after 6 h at 2.5 V and, together with adsorption to the electrode, 82% was removed from solution after 24 h.
[145]	Electrochemical reactor with graphite block anode and perforated stainless steel cathode	Synthetic - PFOS	0.0133	> 99% removal
[146]	Single-cell cylindrical quartz batch reactor under irradiation by a UV-C lamp (16 W, 254 nm, Phillips, Poland) inserted in a quartz tube using GOTiO ₂ film photoelectrodes (PEC: Photoelectrochemical and EC: electrochemical)	Synthetic - PFOS	0.0005	94.83% and 75% in the PEC and EC processes, respectively, within 240 min of reaction time
[147]	Dissolved black carbon (DBC) addition in anaerobic glass bottle	Synthetic - imidacloprid	10	Up to 96.4% within 60 days of incubation
[148]	Small beaker with addition of ferrate VI	Synthetic - imidacloprid	0.00782	As molar ratio (Fe/IMI) increased from 1.0 to 5.0, IMI removal increased from 25% to 84%
[149]	Beakers with Ni thin plates at the bottom (Nickel Oxide (NiO) and reduced stated Nickel Oxide (r-NiO))	Synthetic - imidacloprid alkaline solution	100	80.2% (r-NiO) & 66.3% (NiO)
[150]	Electrochemical cells: Pt anode/carbon felt cathode (EF-Pt), BDD anode/carbon felt cathode (EF-BDD), and anodic oxidation with BDD anode/carbon felt cathode (AOBDD)	Synthetic - imidacloprid	25.57	91, 96 and 93%, respectively
[151]	Erlenmeyer with a novel the nickel (Ni) doping layered double hydroxide (LDH), named Ni/CuMgFe LDH	Synthetic - Benzotriazole	10	88.9% after 6 cycling experiments
[152]	Plexiglas cylindrical electrolytic cell with a total volume of 1.0 containing a tubular Titanium suboxides (TiSO) anode (Electroactive Ceramic Membrane) and a SS cathode	Synthetic - Benzotriazole	10	98.1% removal at current density of 20 mA.cm ⁻² and permeate flux of 692 LHM under cathode-to-anode flow pattern (1 h).
[153]	Erlenmeyer flask with Cu (II)-doped V ₂ O ₅ (Vanadium pentoxide) mediated persulfate	Synthetic - Benzotriazole	20	96% for 10% Cu-V ₂ O ₅
[154]	Large-scale column experiment designed to mimic the passage of recycled water containing BTri and 5-MeBT	Secondary treated wastewater	0.2	> 97.5% after 140 days

- (2) Maximizing biogas production is essential for improving the sustainability and energy efficiency of AnMBR-BES systems. Future research should focus on advanced bioremediation strategies, such as the use of AnGS and the integration of co-culture strategies, which employ multiple complementary microbial species to enhance biodegradation rates and system resilience. Bioaugmentation with genetically engineered microbes capable of efficient electron transfer could further revolutionize bioremediation capabilities. While many studies have examined relative abundances, the lack of quantitative analyses hinders a discerning understanding of microbial interactions, population dynamics, and responses to changing conditions. Precise analysis of microbial communities through advanced techniques like qPCR or digital PCR (dPCR) is necessary to fine-tune the microbial consortia for optimal system performance. Additionally, operational tweaks such as the optimization of pH and alkalinity, thermal integration using low-grade waste heat, and the strategic management of HRTs can significantly boost methanogenic activity and biogas yields. Furthermore, despite progress in AnMBR-BES research, detailed COD mass balance analyses are still limited. Prioritizing these studies is crucial to comprehensively evaluate organic matter transformations, COD distribution, and the impacts of operational parameters.
- (3) In essence, the scalability of integrated AnMBR-BES systems, crucial for their widespread adoption, is unexplored due to a focus on laboratory-scale studies, highlighting the need for research on overcoming technical challenges for larger applications. Notably, while BES coupling with AnMBR has alleviated fouling issues, the use of conductive membrane materials often leads to secondary pollution due to the dissolution of modifiers, consequently reducing efficiency. Investigating more robust membrane materials, resistant to dissolution, and employing advanced surface modification techniques such as graphene oxide coatings or conductive polymer applications, would effectively address these durability challenges. Additionally, techniques like multi-layer deposition assembly could fine-tune membrane properties, enhancing their selectivity and resistance to fouling. Further research into interface reactions on membrane surfaces, through advanced imaging and molecular diagnostic techniques, is essential for providing critical insights into fouling mechanisms that impede system performance. Simultaneously, developing comprehensive monitoring and regulation strategies, including real-time fouling sensors and automated cleaning cycles, is crucial for optimizing the efficiency and scalability of AnMBR-BES systems. Moreover, the prevalent use of conventional electrodes results in suboptimal electric field distribution in engineering-scale applications, thus unnecessarily increasing energy consumption. Transitioning to three-dimensional electrode configurations and integrating flow-through electrode structures could significantly improve electric field uniformity and reduce energy demands. This enhancement can be achieved using novel materials such as carbon nanotubes or metallic foams, which provide higher conductivity and greater surface area, facilitating more efficient bio-electrochemical interactions. To ensure these technological improvements translate into viable large-scale applications, it is crucial to also consider economic and environmental impacts. Lifecycle assessments and cost-benefit analyses should be incorporated early in the design process to evaluate the sustainability of large-scale implementations. Potential strategies might include modular system designs that can be easily scaled up, or the integration of AnMBR-BES systems with existing wastewater treatment plants to enhance their energy efficiency and treatment capacity. Addressing these gaps holistically can significantly advance the practical implementation of AnMBR-BES systems, laying a strong foundation for integration into large-scale wastewater treatment infrastructures.

Table 6
Reported work on AnMBR-BES & Future Needs.

Indicator	Reported Work in AnMBR-BES	Gaps and Research Opportunities
Type of wastewater	Most of the studies targeted synthetic wastewater with mainly glucose or acetate as carbon source	Need to study the system's performance with complex and real wastewater compositions
Micro-pollutants	Two studies targeted the removal of ARGs [44,52] with one focusing on chlortetracycline removal [44]	Need to study the system's efficiency in removing and degrading micropollutants, especially at relevant concentration level (ug/L and ng/L), and the combined impact of micropollutant mixtures on removal efficiencies
Variability in operating conditions	Mostly the effects of HRT and OLR were studied (Table 2). Only one study evaluated the effect of temperature in a two-stage system [69], and all the conducted studies were mainly conducted in the mesophilic range. A series of studies was conducted on the effect of external resistance on the AnMBR-MFC [9,41,40,48] and the effect of different external power supply in AnFMBR-MECs. One study tested continuous vs. intermittent electric field [46]	Study the interactions and correlations between different operating conditions in detail. Study the effect of psychrophilic (as in anaerobic treatment, hydrolysis might be limited at low temperatures [155] and thermophilic temperatures (which can have several advantages including the elimination of pathogens and an increased destruction rate of organic solids [69]. Examine the effects of changing influent conditions (to mimic real influent) on the system performance
Bioremediation boosters	Only two studies inoculated their system with AnGS [45, 53]	Future studies shall explore the integration of AnGS, co-culture strategies, bioaugmentation of specific microbial strains, and optimizing the electrolyte composition in AnMBR-BES coupling
AnMBR-BES variations	A variety of electrodes and membranes were used, but limited studies compared the effect of different types of electrodes or system configurations under the same operating conditions to optimize material use and arrangement. Very few studies targeted both MFCs and MECs [59,64,47]	Conduct a comprehensive study that systematically compares different electrode and membrane arrangements within AnMBR-BES systems while maintaining identical operating conditions to allow for the isolation of the effects of specific configurations and direct comparison. Also, to compare the same reactor in different systems AnMBR-MFC with AnMBR-MEC
Microbial community dynamics	Microbial communities were identified but most studies reported relative abundances. Only four studies performed quantitative polymerase chain reaction (qPCR) [35, 52,70]	Future studies could focus on incorporating quantitative microbial analysis methods, such as qPCR or digital PCR. These techniques provide absolute quantification of specific microbial taxa, offering a more precise understanding of their abundance in the system
COD Mass balance.	Five studies have explored detailed analyses of COD mass balance [49,36,38,42, 67]	Need to prioritize detailed COD mass balance analysis for AnMBR-BES integration, encompassing organic matter transformations, COD distribution across process stages, and assessment of

(continued on next page)

Table 6 (continued)

Indicator	Reported Work in AnMBR-BES	Gaps and Research Opportunities
Scalability	Most of the studies were conducted in a laboratory-scale reactors and only two were pilot-scale [42,43]	operational parameter impact. Determining the contribution rates of multiple effects is essential for system design, optimization, and scale-up Need to assess the system's performance on a commercial/industrial scale. Need to develop strategies for scaling up the integrated system for practical applications
Modeling	-	Conduct modeling analyses to establish meaningful correlations and intricately examine the dynamic interplay between biological processes, membrane performance, electrochemical processes, and operational conditions
Economic and environmental benefit analysis	Only one study evaluated the economic and environmental benefit of the technology [42]	Conduct comprehensive economic and environmental assessments of the integrated system to provide insights into the overall costs associated with implementing and operating the AnMBR-electrochemistry system compared to the benefits it offers in terms of wastewater treatment and resource recovery

6. Conclusion

The integration of AnMBR systems with BES represents a transformative approach in wastewater treatment, promising high-effluent quality while addressing energy costs. This integration resolves various AnMBR limitations by reducing fouling, enhancing sludge filterability, and improving cleaning through merged BES-membrane configurations. Additionally, the bioelectric field enhances biogas recovery by altering microbial communities and converting dissolved methane into gas. Yet, ongoing research reveals gaps, necessitating focus on scalability, remediation boosters, comprehensive evaluations, micropollutant removal kinetics, transformation products, and environmental toxicity. Moreover, a lifecycle analysis is essential for large-scale implementation. Through holistic methods, AnMBR-BES integration establishes itself in real-world wastewater treatment, ensuring viability across diverse applications.

CRedit authorship contribution statement

Marc Heran: Writing – review & editing, Validation, Supervision. **Francois Lestremou:** Writing – review & editing, Validation, Supervision. **Andrés Sauvêtre:** Writing – review & editing, Validation, Supervision. **Francois Zaviska:** Writing – review & editing, Formal analysis. **Geoffroy Lesage:** Writing – review & editing, Supervision, Resources, Formal analysis. **Olga El Kik:** Writing – original draft, Resources, Formal analysis, Data curation, Conceptualization.

Declaration of Generative AI and AI-assisted technologies in the writing process

During the preparation of this work the authors used ChatGPT only to improve readability and language. After using this tool, the authors reviewed and edited the content as needed and take full responsibility

for the content of the publication.

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.

Data availability

No data was used for the research described in the article.

Acknowledgments

Special thanks are extended to the Occitanie Region, France, for funding this work.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.jece.2024.113162](https://doi.org/10.1016/j.jece.2024.113162).

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