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Waste to value: microbial electrochemical technologies for sustainable water, material, and energy cycles

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Abstract

Global wastewater production exceeds 359 billion m^3 annually, of which only 52% is treated, mostly in expensive and resource-consuming processes. Microbial electrochemical technologies (METs) offer a transformative approach to sustainable wastewater management by converting waste into valuable resources such as energy, clean water, and nutrients. They present a viable solution to the United Nations' Sustainable Development Goal 6 (to ensure access to water and sanitation for all) by enhancing both sanitation and resource recovery. METs, including microbial fuel cells (MFCs) and microbial electrolysis cells (MECs), harness electrogenic microorganisms to oxidize organic matter, generating electric energy or producing energy carriers like hydrogen and methane. METs also enable recovery of nutrients, such as ammonium and phosphates, which are essential for agriculture, thereby closing resource loops in a circular economy. Despite their potential, challenges remain in scaling up METs for widespread application. Pilot-scale MFCs and MECs have demonstrated feasibility, achieving up to 90% chemical oxygen demand removal and producing electric power, methane, or hydrogen from wastewater. However, high capital costs, material limitations, and energy efficiency barriers hinder commercialization. Innovations in electrode design, modular configurations, and integration with existing wastewater treatment processes (e.g., anaerobic digestion, membrane bioreactors, or constructed wetlands) are advancing METs toward higher technology readiness levels (TRLs 4–8). Field applications, like a system for urine-based electricity generation in underserved regions, highlight

METs adaptability and societal impact. The transition from laboratory to real-world implementation requires scaling, process integration, and further optimization to reduce costs and improve performance. By aligning with circular economy principles, METs can transform wastewater into a resource, contributing to energy security, environmental sustainability, and global sanitation goals. Future research should focus on scalable designs, economic viability, and interdisciplinary collaboration alongside understanding and optimizing the microbial "black box" to enable METs to transform previously unused wastewater streams into valuable resources with targeted applications.

KEYWORDS

microbial electrochemical technologies, wastewater treatment, resource recovery, microbial fuel cells, microbial electrolysis cells, energy recovery, nutrient recycling, circular economy

Key points

- Microbial electrochemical technologies (METs) present a transformative approach to wastewater treatment by simultaneously enabling energy recovery, clean water production, and nutrient recycling.
- Global wastewater could produce over 800,000 GWh of chemical energy annually, and this is currently largely untapped.
- The recovery of ammonia and phosphate using METs offers a compelling route to circular nutrient management and sustainable agriculture.
- Despite promising lab-scale results, METs face challenges in scalability, energy efficiency, and costs, with pilot-scale systems often underperforming due to increased resistance and complex wastewater matrices.
- Integrating METs with established treatment systems (e.g., anaerobic digestion or membrane bioreactors) and novel applications (e.g., hydroponics or remote sensors) extends their viability for both centralized and decentralized sanitation solutions.
- Field deployments in underserved regions demonstrate the potential of METs for low-cost, decentralized energy and sanitation infrastructure, addressing multiple UN Sustainable Development Goals.

Introduction

Access to clean and safe drinking water is a fundamental human right and is foundational to healthy environments and economic prosperity. This access is intrinsically connected with sanitation and, thus, to the treatment of used water (wastewater): jointly they form one of the United Nations' (UN) Sustainable Development Goals (SDGs) (1). However, despite intensive efforts, this goal remains unattained for a significant population on our planet, with approximately 3.4 billion people lacking access to safely managed sanitation (2) and 4% of the global population relying on open defecation. Providing safe and affordable sanitation is a highly complex matter that involves interconnected technological, cultural, economic, political, and social factors. Even estimating the extent of the gap to achieve SDG 6 (clean water and sanitation) is challenging, as the average UN Member State possesses data for only approximately two-thirds of the global indicators related to SDG 6 (3). Consequently, there are varying estimates of how much of the world's wastewater is treated. These range from 20% in 2017 (4) from combined domestic and industrial wastewater, based on the assumption that human activities that use water produce wastewater to 56–58% in 2024 (1, 2) from just domestic wastewater (household water and sanitation). Jones et al. (5) developed a multi-data source regression-based approach that estimated a global domestic and manufacturing wastewater production at 359×10^9 m^3yr^{-1}, of which 63% (226×10^9 m^3yr^{-1}) is collected, 52% (188×10^9 m^3yr^{-1}) is treated, and only 11% is intentionally reused.

In the past, due to the availability of cheap fossil resources for materials and energy, with limited options for enabling technologies, wastewater—like many other human waste streams—has been considered a form of waste rather than a resource. Consequently, there have been few economic incentives for recovery in value chains or even the establishment of closing cycles, leaving recovery of resources from wastewater largely unexploited. Moreover, with existing infrastructure, substantial amounts of energy are used for wastewater treatment, primarily for removing organic matter and nutrients, and for disinfection to make water suitable for discharge. Thus, wastewater treatment is an industry comparable even to aviation in its energy demand (6), in which the principal activity eliminates, but does not adequately make use of, energy-containing organic compounds and nutrients.

TABLE 1 Energy contained in the wastewater generated by each region of the world based on data from Jones et al. (5); values are in 10^9 m^3yr^{-1}, assuming an average concentration of 500 mg L^{-1} chemical oxygen demand.

Region	Wastewater Production (10^9 m^3yr^{-1})	Energy present in the form of organic matter (GWh)	Equivalent number of 1-GW power plants*
East Asia and the Pacific	117.6	262,967	32
North America	74.7	167,038	21
Latin America and the Caribbean	42.1	94,140	12
Western Europe	38.5	86,090	11
Eastern Europe and Central Asia	28.2	63,058	8
South Asia	25.6	57,244	7
Middle East and North Africa	21.9	48,971	6
Sub-Saharan Africa	11	24,597	3
Global	359.4	803,658	100

For simplicity reasons, we assume that a 1-GW nuclear power plant produces 8 TWh electric energy per year, corresponding to 11 months of full-capacity operation. The respective amounts of primary chemical energy are compared with the number of 1-GW power plants, a typical amount of power for a nuclear power plant, which would be required to produce the same amount of electric energy.

Wastewater resources in numbers

The main resource in wastewater that is usually considered from an energy perspective is the chemical energy contained in organic and inorganic compounds that are removed by oxidation processes. Since the concentration of this matter in wastewater—generally quantified in terms of chemical oxygen demand (COD)—is strongly dependent on the wastewater source, season, and time of day, the energy content (more precisely combustion enthalpies, $\Delta_c H$) can vary widely in values from −5.6 to −20.2 kJ L^{-1} and from −5.9 to −23.4 kJ g COD^{-1} (7–9). To assess the energy available in wastewater, we use an average energy content of 16.1 kJ g COD^{-1} (10) due to its good statistical basis of 61 composite municipal wastewater samples. We note that this value may still underestimate the energy content in wastewater due to the difficulty in capturing and quantifying volatile but easily biodegradable organics during the analysis of wastewater samples (11). Considering typical domestic wastewater with a COD concentration of 500 mg L^{-1}, which is equivalent to approximately 500 mg L^{-1} of sugar, this results in 8 kJ of chemical energy per liter of wastewater. On a $1m^3$ scale, wastewater contains 8 MJ or 2.2 kWh* of energy, which is enough to power a 2 kW electric water boiler for 1h, and is approximately the daily average food energy intake for a person (12). The comparison of the primary chemical energy contained in wastewater with a quantity of electric energy is, of course, not strictly valid, but it illustrates the magnitude of the resource.

Wastewater is a relatively dilute energy source, yet it is generated in vast volumes. Table 1 estimates the size of this resource in different regions of the world, based on collection and treatment estimates. An assumption is made that the global average COD concentration is 500 mg L^{-1}, even though we recognize that different regions can have large variations in COD concentrations. The amount of chemical energy available in wastewater is considerable. Thus, the energy content of the globally produced wastewater adds up to more than 800,000 GWh, which would compare to approximately 20% of the total annual electricity production of a country like the United States of 4.18×10^6 GWh in 2023 (13). In terms of energy content, wastewater can thus be considered a viable energy resource.

Wastewater not only has a large portion of untapped chemical energy, but current methods of wastewater treatment also consume vast amounts of energy. This contributes to the water sector's 4% share of the global energy consumption and positions water supply and wastewater treatment among the largest consumers of electricity globally (14). Depending on the technology, plant scheme, and quality of the treated water, wastewater treatment plants consume approximately 0.5–2.0 kWh m^{-3} of treated water (15). Using the wastewater production estimates in Jones et al. (5), Table 2 maps the energy currently used in wastewater treatment based on the lower end of this estimate (0.5 kWh m^{-3}). Considering that only 52% of global wastewater is being treated, a comprehensive wastewater treatment using the portfolio of existing technologies would further increase this energy demand (Table 2). It is clear that immense amounts of energy are currently used and more would be needed to achieve comprehensive global wastewater treatment if existing technologies were used. The energy bill for treating all of the world's wastewater would be around 180,000 GWh, equivalent to 22 1-GW power plants.

Apart from organic substances contained in wastewater, important nutrients like nitrogen [in the form of ammonium (NH_4^+) and nitrate (NO_3^-)] and phosphate (PO_4^{3-}) can also be harvested. The release of these nutrients to the environment creates environmental problems, e.g., eutrophication of waterways, thus their release is being increasingly strictly regulated in many countries around the world (16). Furthermore, recycling these nutrients is vital for maintaining our agricultural operations.

* From here on, for better visualization, we will switch from the SI unit Joule to kWh, as the technically more common unit.

TABLE 2 Energy costs for the treatment of the estimates of wastewater produced by each region based on the values in Jones et al. (5), assuming an average treatment energy demand of 0.5 kWh m^{-3}.

Region	Wastewater currently treated ($10^9 m^3$ yr^{-1})	Percentage currently treated	GWh energy estimate for current wastewater treatment and, in brackets, for treating 100% of currently produced wastewater	Equivalent number of 1-GW power plants required for treating 100% of currently produced wastewater*
East Asia and the Pacific	57	48%	28,200 (58,800)	7
North America	50.6	68%	25,400 (37,350)	5
Latin America and the Caribbean	15.4	37%	7,800 (21,050)	3
Western Europe	33	86%	16,550 (19,250)	2
Eastern Europe and Central Asia	14.9	53%	7,500 (14,100)	2
South Asia	4	16%	2,050 (12,800)	2
Middle East and North Africa	11.4	52%	5,700 (10,950)	1
Sub-Saharan Africa	1.8	16%	880 (5,500)	1
Global	188.1	52%	93,450 (179,700)	22

*For simplicity reasons, we assume that a 1-GW nuclear power plant produces 8 TWh electric energy per year, corresponding to 11 months of full-capacity operation. The energy equivalent is extrapolated to the equivalent number of 1-GW power plants.

Ammonia requires a significant amount of energy (corresponding to 3% of the world's natural gas production) for its production through the Haber–Bosch process, while phosphates are currently mined from finite, diminishing resources.

The average total nitrogen content in untreated municipal wastewater varies between 20 and 70 mg L^{-1} (17). For medium-strength wastewater, the total nitrogen concentration is 40 g m^{-3}, of which 25 g m^{-3} is in the form of ammonium nitrogen (NH_4–N) and 15 g m^{-3} is in the form of organic nitrogen. Since all organic nitrogen will eventually be converted to ammonium, we here assume 40 g m^{-3} of NH_4–N is found in municipal wastewater. The global resource within all wastewater (360×10^9 m^3 per year), if collected and fully recovered, would thus be approximately 14 million metric tonnes (14 Mt). In total, this represents 11% of the current annual global ammonia demand of 132 Mt (18), (See Table 3). Considering the energy consumption of the Haber–Bosch process of approximately 30 GJ $t_{NH_3}^{-1}$ or 8.3 MWh $t_{NH_3}^{-1}$ (18, 19), this amount of ammonia would account for an energy consumption of approximately 116 TWh (the annual energy output of 14.3 1-GW power plants).

A range of 4–15 g m^{-3} of phosphate is present in wastewater (17). Using the average of 9 g m^{-3}, this equates to 3.2 Mt in the wastewater produced globally. The global phosphate demand (calculated as P_2O_5) was 45.7 Mt in 2023, with a projected increase to 50 Mt in 2027 (20). The increasing consumption, combined with the uneven geographical distribution of this resource, has led to significantly increasing prices of phosphates on the global markets, calling for a transition toward a more circular phosphorous economy (21).

In many regions of the world, the above mentioned amounts of nitrogen and phosphorus contained in wastewater are close to the amounts of these elements utilized as mineral fertilizers in agriculture in those respective regions [see example from Germany: (22, 23)]. Closing the loop of treating wastewater to recover resources could thus have a significant impact not only on achieving SDG 6 but also more widely on sustainable agriculture and economic development.

The treated water itself is also an important resource. Removing the substances contained in wastewater prevents the pollution or eutrophication of surface water and groundwater and, thus, also the (indirect) impairment of drinking water resources. Further to that, it turns it into a resource as the treated water is made available for direct (re)use as service water or even potable water. Thus, the reclaimed water from most wastewater treatment plants could be used for different non-potable applications such as urban landscaping, agricultural irrigation, construction, industrial cooling, impoundments, and environmental purposes such as groundwater recharge (24, 25). For integration into the drinking water supply, a further polishing of the cleaned wastewater, e.g., by advanced membrane filtration technologies, is necessary (26). Treated water from decentralized wastewater treatment systems at the municipal and local levels can be used directly at the source in dry regions for gardening, horticulture, cleaning, toilet flushing, and recreational purposes (25). Producing fit-for-purpose water from wastewater and wider adoption of such reuse practices for reclaimed water can help lower the burden on freshwater resources and curtail groundwater overextraction across the globe (27).

Circularity in wastewater treatment is not new; there are technologies that treat wastewater and yield energy. The oldest and most widespread of these is anaerobic digestion (AD), where organics are digested to methane, which is in turn burnt to produce electric and thermal energy. McCarty et al. (28) provide a

TABLE 3 Estimated amounts of ammonia nitrogen and phosphate in wastewater produced by each region and the individual shares with respect to global ammonia and phosphate demands.

Region	Ammonia nitrogen		Phosphate	
	Thousand metric tons available	Fraction of the current world demand (%)	Thousand metric tons available	Fraction of the current world demand (%)
East Asia and the Pacific	4,704	3.6	1,058	2.4
North America	2,988	2.2	672	1.5
Latin America and the Caribbean	1,684	1.3	379	0.8
Western Europe	1,540	1.2	347	0.8
Eastern Europe and Central Asia	1,128	0.8	254	0.6
South Asia	1,024	0.8	230	0.5
Middle East and North Africa	876	0.7	197	0.4
Sub-Saharan Africa	440	0.3	99	0.2
Global	14,376	10.9	3,235	7.2

The data is based on an extrapolation of an average phosphate content of 9 g m^{-3} and an average nitrate concentration of 40 g m^{-3} of wastewater to the regional wastewater volumes. Data source: (17).

comprehensive review of the prospects and energy balances of AD versus METs. With AD, only approximately 81% of biodegradable matter can be converted to methane and, when combusted, only 35% of this can be converted to electricity (with 65% going to heat). Ultimately, only approximately 28% of the original chemical energy content can be converted to electricity—which acts as a helpful benchmark for MFCs to reach. Low temperatures and organic loads have been also cited as barriers; however, there are studies using AD reactors that overcome this with reasonable efficiencies (28), although METs also operate comfortably in these conditions (29). AD does not recover nitrogen or phosphorus directly, but add-on processes could perform this function. METs have the capacity to allow both. Achieving the recovery balances needed at a cost lower or equivalent to these more traditional methods remains a focus and challenge in this field of research.

Basic principles of METs

COD removal and energy recovery in METs

METs exploit the ability of electrochemically active bacteria to exchange electrons with their surroundings via extracellular electron transfer (EET) (30). In the context of wastewater treatment, the resulting ability to exchange electrons with electrodes allows coupling microbial degradation of organic and inorganic compounds with the generation of an electric current. Two primary METs are microbial fuel cells (MFCs) and microbial electrolysis cells (MECs). Both MFCs and MECs share the same anode process (Figure 1A), which is the driver of COD removal (31), but differ in the cathode reaction and thereby in the net energetics of these bioelectrochemical systems. At the anode, electrochemically active bacteria engage in EET and utilize the electrode as a terminal electron acceptor, while fully or partially oxidizing organic matter to generate electricity (32). These bacteria are often referred to as electrogenic bacteria or electrogens. From there, in MFCs and MECs, the electrogenic bacteria are typically utilized in the form of electrode-associated biofilms (often referred to as electroactive biofilms, EAB). Their performance, i.e., the magnitude of current they can produce and their ability to degrade organic matter in wastewater at sufficient rates, is intrinsically linked to the rate and efficiency of the EET. Here, the transfer of electrons across the cell membrane as well as through the biofilm and across multiple cell layers plays a central role. Due to this role, extensive research has been conducted on EET processes between electrogenic bacteria and electrodes, mostly concentrating on select model species (33–36). The molecular mechanisms behind EET involve transferring electrons across the cellular membrane and biofilm matrices. Substantial experimental evidence supports the involvement of key cell surface molecules such as multiheme c-type cytochromes (34, 35) and electrically conductive type IV pili, or e-pili (33). Multiheme cytochromes (MHC) facilitate electron transport through electron hopping between neighboring heme groups and from one adjacent MHC to another. In some cases, such as *Geobacter sulfurreducens*, highly conductive MHC nanowires can even be formed, transporting electrons far from the cell surface (37). On the other hand, e-pili has been shown to facilitate electron transport via aromatic amino acids located at periodic distances (38–41), although recent studies have questioned this role and pointed toward a secretory function (42). These functions, however, may not be mutually exclusive.

Non-electroactive microorganisms can also exist in MET systems, breaking down complex organic matter to molecules that can be used by the electrogenic microorganisms by, for example, hydrolysis and fermentation processes (43–45). Hence, the bioelectrochemical substrate degradation creates food webs and supports the coexistence of different microorganisms, which may

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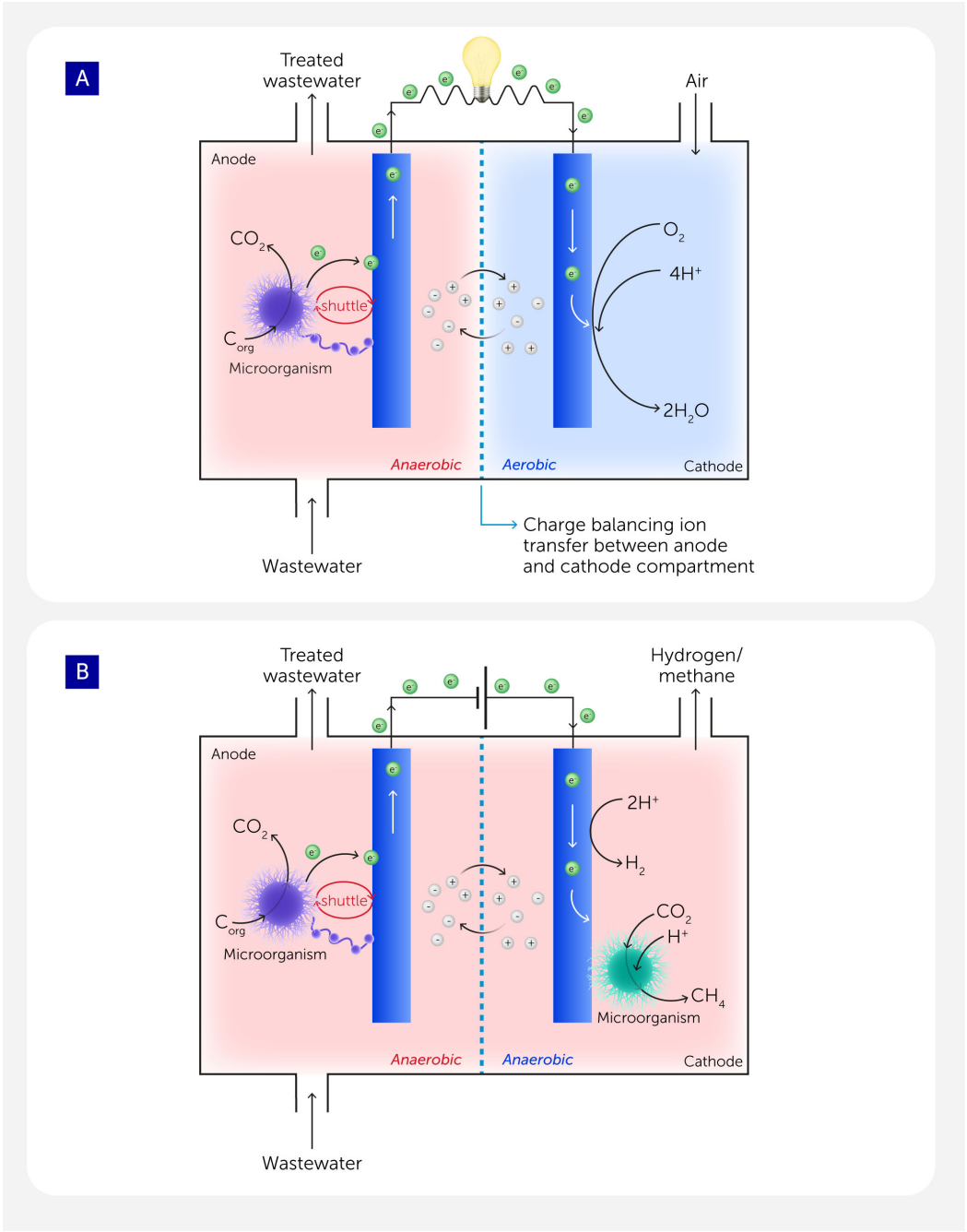

FIGURE 1
Simplified schematics of a **(A)** microbial fuel cell and **(B)** microbial electrolysis cell.

also include methanogens. Methanogens can exploit the by-products (e.g., H_2/CO_2, formate, acetate, and electrons/CO_2) of fermentative bacteria or incomplete oxidation by electrogens oxidizing complex organics (46, 47). Alternatively, they can directly utilize cathodic electrons, acting as electrotrophs (48–50). The exact nature of the symbiotic or potentially commensal relationships between the bacteria in an electroactive biofilm necessitates further empirical inquiry, as such interactions could inadvertently result in suboptimal MFC or MEC performance (51).

Electrogenic bacteria are part of virtually all anaerobic environmental compartments (52) and are, therefore, also present in wastewater (53). From there, they can be directly enriched as electroactive biofilms on MFC/MEC anodes. This means that there is no need to engineer specific organisms for operation, as the required microorganisms are already present in the target wastewater and are well adapted to a broad range of substrates. There are also numerous procedures for improving biofilm start-up and performance (54). A common and straightforward method is

the use of pre-cultivated electroactive bacteria, from the effluent of a running bioanode in an MFC/MEC for startup, the use of pre-grown electroactive biofilms as seed biomass for inoculation (55, 56), or the employment of a rigorous acclimation regime to accelerate metabolic/growth rates and biofilm formation (57–59).

Despite the practical progress in the cultivation of electroactive biofilms, the understanding of the complex dynamics in these mixed-species communities remains nascent. This opens a window into exploring the nuances of microbial survival strategies, co-adaptation of syntrophic partners to changing conditions, or possible decoupling during bioelectrochemical treatment. This information is necessary if we need to move forward from the "black box" concept that wastewater treatment METs currently are and improve conversion efficiency.

The fundamental difference between MFCs and MECs lies in the cathode reaction. MFCs (Figure 1A) exploit cathode reactions with a more positive potential than that of the anode reaction ($E_{cathode} > E_{anode}$). Therefore, the vast majority of MFCs utilize the oxygen reduction reaction (ORR). The combination with the anode reaction creates an exergonic net process and produces electric energy. In this scenario, the energy in the organic substrate is converted to electric energy. As illustrated schematically in Figure 2A, the amount of energy theoretically extractable in a microbial fuel cell, expressed as the Gibbs free energy of the reaction, ΔG, can be derived from the difference of the standard potentials of the cathode and anode reaction. Exemplarily, an eight-electron anodic oxidation of acetate as a key microbial metabolite to CO_2 ($E' = -290$ mV) combined with the cathodic oxygen reduction ($E' = 818$ mV) would yield $\Delta G_{total} = -855$ kJ mol^{-1} (at standard conditions) (60). Even theoretically, an MFC would not be able to extract the entire energy amount. As biocatalysts, the electrogenic microorganisms divert a part of the energy for their metabolism. This is evident in the redox potential at which the bacteria release the electrons via EET, which is more positive than that of the substrate. The redox potentials of the EET components are specific to the electrogenic bacteria. For *Geobacter sulfurreducens*, for example, it is about −140 mV vs normal hydrogen electrode (61). From the potential difference to the standard potential of the acetate oxidation, one can estimate that *Geobacter* would divert approximately 115 kJ mol^{-1} (ca 13%) of the total chemical energy, leaving 739 kJ mol^{-1} as extractable energy. It must be pointed out that this is a simplified consideration since redox potential in real systems may differ significantly from the standard redox potential values. In addition, this discussion does not include energy losses based on slow electrode kinetics and other losses, e.g., based on the ohmic resistance of the bioelectrochemical systems.

In contrast to MFCs, the cathode reaction of MECs—e.g., the reduction of protons to molecular hydrogen, the hydrogen evolution reaction (HER) (Figure 1B), or the reduction of CO_2 to methane (often referred to as electromethanogenesis)—takes place at potentials negative to the anode potential ($E_{cathode} < E_{anode}$), resulting in an endergonic net process and thus the necessity of adding electric energy to drive it. The result is energy storage in carriers such as hydrogen or methane. As Figures 2B, C illustrate, due to the energy-delivering anodic process, the hydrogen production in MECs requires considerably less addition of electric energy than conventional water electrolysis (62), which involves the energetically unfavorable oxygen evolution reaction as the anode reaction. Thus, as a simplified thermodynamic consideration based on the data depicted in Figure 2, the energy demand of a hydrogen-producing MEC would be 53 kJ mol^{-1} of hydrogen in comparison to 238 kJ mol^{-1} for the conventional water electrolysis.

The majority of METs rely on chemically catalyzed electrode reactions, although some cathodes based on the ORR (MFC) or the HER (MEC) can be catalyzed using microorganisms (63–65). For conversion of CO_2 into specific products such as methane via electromethanogenesis, cathodes relying on microbial catalysts (e.g., electrotrophic methanogens) can help in steering the overall reaction towards the desired product. Methanogens can directly uptake either electrons or hydrogen from the cathode, utilizing them as an electron donor (48, 49, 66). In MEC–AD, the presence of certain electrogens at the cathode led to the belief that interspecies interactions based on direct interspecies electron transfer (DIET) were important for effective cathodic conversions. Recent findings challenge this previously held notion that a DIET consortium benefits from a cathodic environment (67).

The performance of METs is evaluated by calculating overall energy efficiencies, current densities, power densities (in MFCs), or production rates and chemical recoveries in other types of systems (e.g., MECs). Understanding the factors that impact these performance metrics (and ultimately controlling these) is challenging due to the complexity of MET designs. Electrode-associated factors, such as the overpotential of anode and cathode reactions, depend on the nature of the underlying electrochemical or bioelectrochemical reactions. For the anode, it also depends on the extent of electron-scavenging side reactions (e.g., aerobic substrate degradation, nitrate or sulfate respiration, or methanogenesis) that lower the electron yield (also referred to as Coulombic or Faradaic efficiency). While these factors are not usually affected by a scale-up of the respective MET, physical factors like the internal (ohmic) resistance of the reactor are strongly dependent on the size and represent a major scale-up challenge. The internal resistance, which governs energy efficiency and electrochemical performance, is influenced by several factors: the solution chemistry, reactor scales (small vs. large electrodes), architecture (e.g., electrode spacing), electrode and separator materials, and dominant microbial communities (62, 68–70). In general, laboratory-based METs with optimal solution chemistry, consisting of well-buffered (and highly conductive) solutions and the substrates preferred by electrogens, such as acetate, achieve much higher performance than those fed with poorly buffered (and often poorly conductive) solutions and complex organic matter such as domestic and industrial wastewaters. Buffers used in laboratory-based systems have two effects: they limit the development of pH gradients at the electrodes, thus maintaining high electrode performance, and they produce comparatively high ionic concentrations and thus high ionic conductivities (70). As an example, MFCs have reached power densities of 8.8 ± 0.5 W m^{-2}

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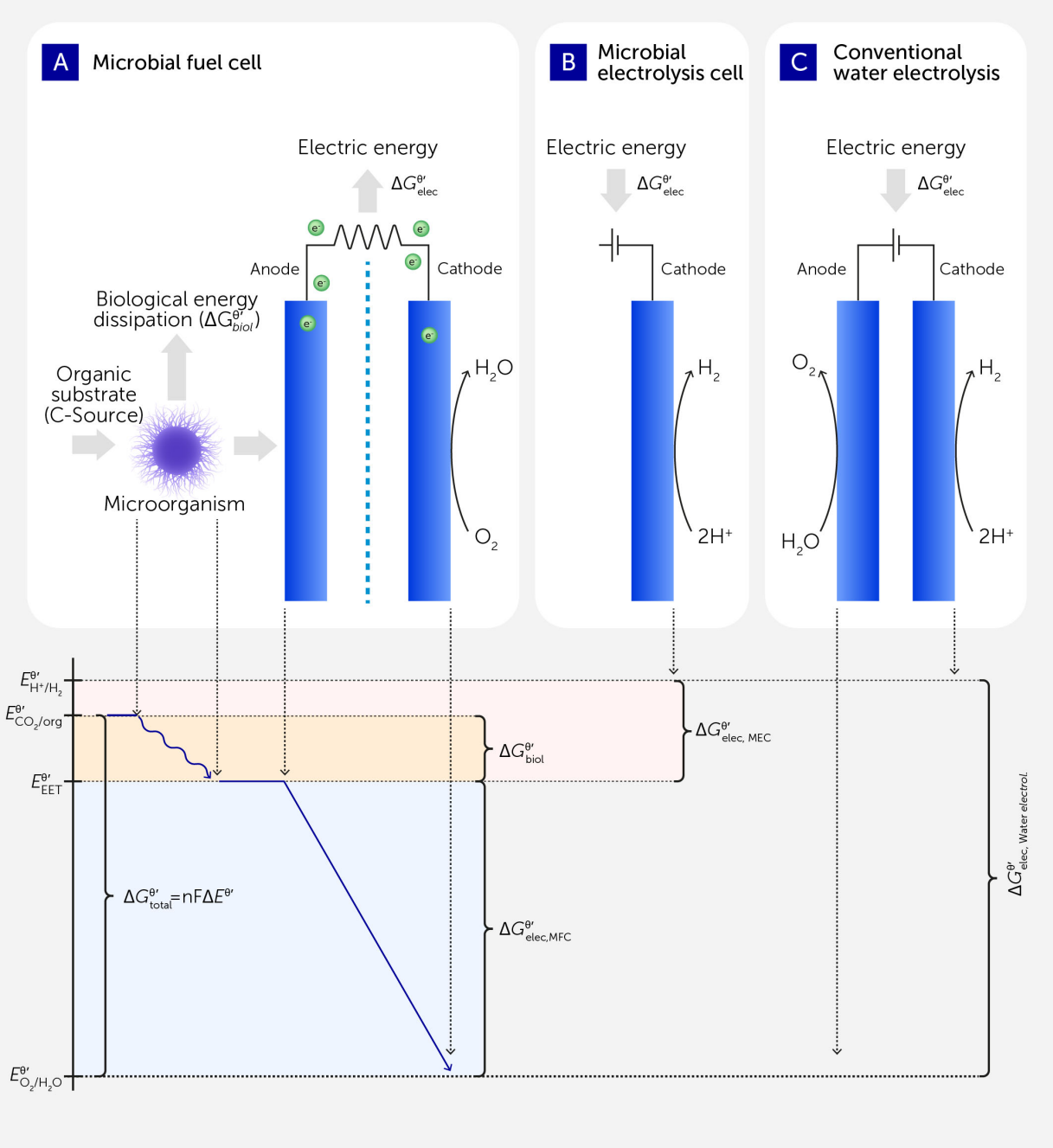

FIGURE 2
Simplified thermodynamic energy analysis of **(A)** a microbial fuel cell (MFC), **(B)** a microbial elekctrolysis cell (MEC), depicted here by the cathode only, which differs from MFC cathodes, and **(C)** conventional water electrolysis (for comparison with B). The total energy contents of a given organic substrate, ΔG_{total}, results from the difference between the biochemical standard potentials of substrate oxidation and oxygen reduction. During substrate oxidation, electrogenic microorganisms retain a part of this energy (ΔG_{biol}) and release the electrons at a lower energy level (more positive redox potential - E'_{EET}). The remaining energy can be extracted in an MFC **(A)** as electric energy, ΔG_{electr}. In an MEC **(B)**, the potential difference between EET and, e.g., hydrogen evolution reaction determines the additional energy required to drive the hydrogen evolution process. In contrast, the energy demand of conventional water electrolysis **(C)** is much higher, as it comprises the entire potential difference between the H_2/H^+ and the O_2/H_2O couple.

at 42 ± 1 A m^{-2} in a typical 50 mM phosphate buffer solution (71), whereas the same system produced only 1.34 W m^{-2} under otherwise identical conditions when the solution buffer strength was reduced to be similar to that of domestic water (alkalinity of 360 mg L^{-1}) (72).

Energy efficiencies, defined as the fraction of the chemical energy contained in the media that is effectively converted to electricity, ranged widely, from 35% under optimal conditions (73) to <1% for continuous reactors (74) using wastewater. Similar trends in performance due to solution chemistry and

reactor size have also been observed in MECs, with current densities reaching >60 A m^{-2} and hydrogen gas production rates of 72 L $L^{-1}d^{-1}$ using well-buffered synthetic media with plenty of acetate, much higher than those obtained with real media (75–78). Another challenge is low COD concentrations. When METs are operated with media containing low COD content (less than ~150 mg L^{-1}), electrochemical performance is sharply reduced (79). Thus, there is a trade-off in METs between maximizing electrochemical performance by operating at low COD removal rates (see also the Integration of METs into wastewater treatment and remediation section below) and maximizing organic matter removal, which will decrease current density and power production.

The performance of laboratory-scale METs is also not currently maintained during scale-up for larger reactors even with the same architecture. In appropriately scaled deployments, the main rate-limiting step is often related to constraints around the range of available materials and manufacturing processes, impacting on the system design and configuration. This results in performance levels lower than anticipated; clearly this is not a failure of the technology itself but of factors resulting in e.g., hidden short-circuits (electric or hydraulic), longer electron travel distances, or detrimental material deficiencies, giving increased internal resistances. Best engineering practices can and have improved systems on site but only to the extent that a rigid design and limited material availability would allow.

For example, pilot-scale MFCs, with a volume of 500 L or more, have not produced power densities above 0.1 W m^{-2} even though the same electrodes and reactor architecture were used for both the pilot and lab-scale systems. The lower performance of pilot MFCs was likely due to the increase of the electrode resistance (mΩ m^2, normalized by the electrode area) as the electrode dimension was expanded (80). The increase in electrode resistance was smaller for the anode than that of the cathode, increasing from 71 mΩ m^2 (7 cm^2) to 238 mΩ m^2 (6200 cm^2) and 66 mΩ m^2 to 555 mΩ m^2, respectively (81).

Recent advances in pilot- and full-scale METs have clarified the rate-limiting factors that constrain performance and have informed new design strategies. Full-scale systems such as METlands® have demonstrated effective wastewater treatment under real conditions, where performance was primarily limited by mass transfer and redox stratification within the conductive media, which are issues now mitigated through modular architectures and optimized hydraulic design (82). Similarly, passively aerated, membraneless MFCs employing granular activated carbon (GAC) electrodes have shown long-term stability in real wastewater treatment, overcoming ohmic and cathodic limitations through reduced electrode spacing and enhanced ionic pathways (83, 84). A meta-analysis of pilot-scale MFCs identified internal resistance, oxygen mass transfer at air cathodes, and shunt currents (based on crossover processes between anode and cathode compartment, among others) as dominant bottlenecks, guiding design improvements in stack configuration and electrolyte conductivity (85). Further progress has been achieved in cathode development, where non-precious metal catalysts such as Fe–N–C and activated carbon have lowered oxygen reduction overpotentials while improving durability and cost effectiveness (86). The adoption of 3D electrode architectures, including granular and fluidized beds, has also increased effective surface area and reduced diffusion limitations, thereby enhancing power output and scalability (87).

Nitrogen and phosphorus recovery through METs

In addition to energy harvesting based on COD removal, METs can play a role in the removal and the recovery of nutrients, most importantly nitrogen and phosphorus. Under anaerobic conditions, organic nitrogen compounds contained in wastewater are typically converted into ammonium nitrogen. Ammonium (NH_4^+) recovery becomes especially interesting when its concentration in the inflow is high. A typical ammonium-rich stream that has been tested in METs for ammonium recovery is source-separated urine with nitrogen concentrations up to 9 g NH_4^+ L^{-1} (88), a concentration more than two orders of magnitude higher than that found in municipal wastewater (30–50 mg L^{-1}). In addition to high ammonium levels, source-separated urine also has the advantage of containing high concentrations of degradable organic matter. Ammonium recovery in METs utilizes the fact that every electron flowing between the anode and the cathode of an electrochemical cell has to be counterbalanced by a stoichiometric ion flux (anionic or cationic) between both electrode compartments to maintain charge neutrality. When using a cation exchange membrane (CEM) as a separator between the anode and cathode compartments, the bioelectrochemical oxidation of organic matter at a MET anode is thus accompanied by the transport of cations—including ammonium—from the anode to the cathode compartment (89). By leveraging this mechanism, 92% ammonium transfer efficiency from the anode to the cathode compartment has been achieved in a scaled-up system treating urine (90). This transfer across the separating membrane results in the removal of ammonium from wastewater and its accumulation at the cathode (Figure 3A). During operation, the pH of the cathode compartment shifts towards strongly alkaline values due to low rates of transport of protons from the anodic to cathodic chamber and production of hydroxide ions in the cathode chamber (91). Due to the high cathode chamber pH, the ammonium ions are rapidly converted to volatile ammonia, which can then be removed by gas stripping or using hydrophobic membranes that trap the ammonia in an acid solution (92). The latter study demonstrated an ammonia recovery rate of approximately 36 g NH_3–$Nm^{-2}d^{-1}$. The resulting product is ammonium salt, e.g., ammonium sulfate or ammonium nitrate.

An important advantage of ammonium recovery compared to conventional removal is that the transport across a membrane keeps the nitrogen in its most reduced form. There is no biological conversion, eliminating the risk of nitrous oxide (N_2O) production, a greenhouse gas (GHG) more potent than CO_2. For wastewater with low ammonium concentrations, like sewage, the proportion of transferred ammonium compared to alternative cations, such as K^+, Na^+, and protons, is much lower compared to urine, so an economic recovery of NH_4–N may appear unfavorable.

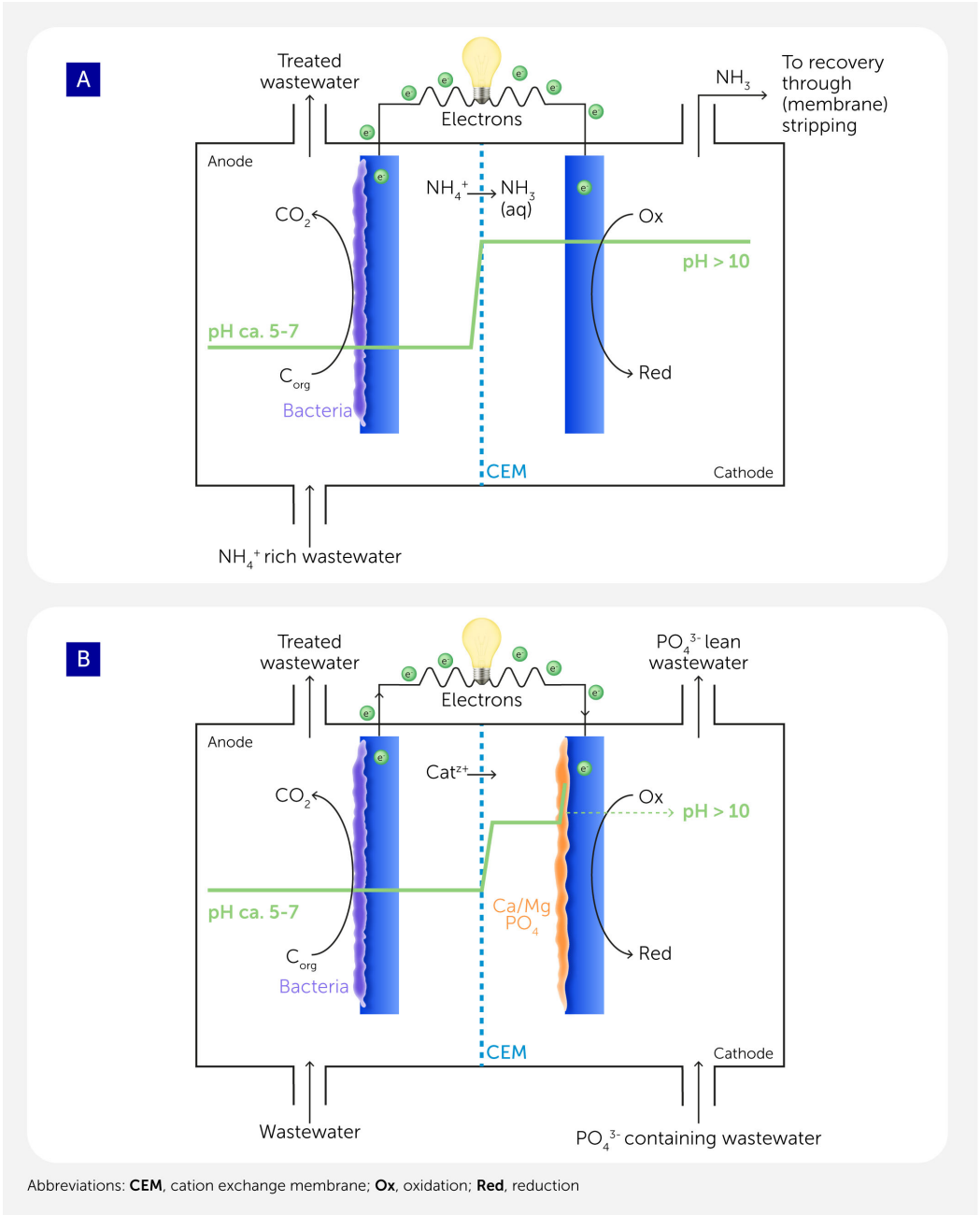

FIGURE 3
Schematic illustration of the mechanisms of **(A)** ammonia and **(B)** phosphate recovery via microbial electrochemical technologies.

For wastewater streams with low concentrations of ammonium, the sole removal of ammonium (as opposed to removal and recovery) is required with oxidation at bioanodes being an interesting route (93–95). So, rather than transporting ammonium from anode to cathode where it is being concentrated, ammonium is converted into N_2 gas at the bioanode. Indeed, several studies have shown that ammonium oxidation to inert N_2 gas at bioanodes is feasible, especially when wastewater with no organic matter (a competing electron donor at the anode) is used. However, this oxidation process results in the loss of a valuable ammonia product.

The recovery of phosphate in METs is based on the occurrence of pH gradients close to the cathode surface of an MFC or MEC (Figure 3B). Due to the consumption of protons for the HER (or ORR) at the cathode, the pH in the catholyte will increase, with the local pH closer to the electrode surface increasing even further. In the presence of Ca^{2+} or Mg^{2+}, a calcium/magnesium phosphate precipitate will form on the cathode, whereas in the presence of ammonium ions struvite (magnesium ammonium phosphate), precipitates are also formed (96). Since both Mg^{2+}/Ca^{2+} and PO_4^{3-} are typically present in domestic wastewater, there is no requirement for metal ion dosing (97). The precipitation of calcium

phosphates at cathodes is a purely abiotic chemical reaction driven by the operation of the respective MFC or MEC. This coupling lowers the overall energy demand more than conventional electrochemical treatments (88).

In comparison to the COD removal and energy recovery described in the previous section, significantly less attention and development work have been dedicated to the MET-based recovery of chemical resources. It is to be expected and hoped that this aspect will receive more attention in the coming years to reach a stage of development that allows for implementation in wastewater treatment strategies.

Integration of METs into wastewater treatment and remediation

Given the ability of METs to recover the energy contained in organic matter as electricity or valuable energy carriers, their integration in wastewater treatment facilities can enhance circularity, thereby improving the overall economics of the water treatment sector. The majority of the early studies on METs focused on small-scale (i.e., laboratory-scale) systems in optimized laboratory conditions and environments. Only recently have more studies been conducted on the design and operation of technical-scale and even pilot-scale reactors, primarily MFCs and MECs, and their integration in wastewater treatment facilities, especially for breweries/distilleries (98, 99) and domestic/municipal wastewater (100). Some of these efforts were pursued exclusively by private companies (101) or by universities in collaboration with different companies (80, 102), indicating that there is interest and potential revenue in bringing METs to market. However, the number of studies on pilot-scale systems remains scarce, and fully commercial demonstrations are lacking. The scarcity of studies is in part due to a limited understanding of the best approach for scaling up these systems and integrating them into conventional treatment trains but also due to the reluctance by companies and authorities to change process trains, since that carries additional economic burden.

MECs and MFCs could be integrated early in the treatment train, taking advantage of a high concentration of nutrients to maximize electricity (MFCs) or hydrogen and methane (MECs) production. Most wastewater cannot be treated to levels suitable for discharge using only an MET, as the further organic matter oxidation rate abruptly decreases around a COD concentration of ~150 mg L^{-1} (79). Therefore, additional treatment technologies are needed downstream of METs to further polish the effluent and ensure that its composition complies with local regulatory requirements. This additional treatment has been accomplished by coupling METs with different treatment systems, including ultrafiltration and nanofiltration units (103), membrane bioreactors (104), and biofilters (80) (Figures 4A, B). The combined bioelectrochemical reactor–polish unit produced effluents with a biochemical oxygen demand (BOD) typically lower than 15 mg L^{-1}, and thus it was acceptable for most treatment facilities in North America, Europe, and Asia.

An alternative approach is to integrate METs into upstream stages for treating high-concentration liquids resulting from the dewatering processes associated with AD and sludge conditioning. This approach leverages the high COD and nutrient concentrations of these systems and does not require discharge standards, as this water is typically recirculated back into the treatment works (105, 106). For example, a pilot MEC system was integrated with hydrothermal liquefaction (HTL) to treat HTL wastewater while producing high-purity H_2 that could be directly utilized for biocrude upgrading to jet fuel (107). Onsite parallel and serial operation of multiple MEC units demonstrated flexible performance, enabling either enhanced H_2 generation under high-current demand or advanced removal of organics and nitrogen under treatment-focused conditions.

While the importance of wastewater influent concentration for performance is known, the optimal design and operational parameters of the actual bioelectrochemical reactor are still debated. An initial challenge is that no treatment plant is alike, and average and peak flow rates and total hydraulic retention times can vary across different plants. This challenge in process train variability could be overcome by modularizing individual reactors once an optimized architecture is identified (101) (Figures 5A, B). Modular METs can then be connected in series (the effluent of one reactor is fed into the next one) or in parallel (the influent is split across a range of reactors). Assuming a similar retention time of each reactor for the two configurations, the parallel connection will achieve a low level of treatment, but the constantly high nutrient concentration will enhance current density and hydrogen or methane production rates (76). When connected in series, higher treatment levels can likely be achieved, but the bioelectrochemical performance of subsequent reactors will decrease as the concentration of nutrients diminishes.

No matter how individual reactors are connected, reactor volume and electrode sizes of current laboratory scale MFCs and MECs will need to be increased for successful integration in actual water treatment plants. However, increasing the electrode size increases the electrode resistance, which has been reported to significantly diminish current density and maximum power density in pilot MFCs as well as hydrogen production rates in pilot MECs (76, 108). This clearly emphasizes the development need for an optimal electrode design for large-scale MET systems.

Maintaining a high-current density is key to accelerating nutrient removal and boosting the electrochemical performance of the reactor. Previously published analysis on scaled up MFCs and MECs indicated that reactors with large electrode packing densities (area of electrode per volume of reactor) achieved the highest volumetric performance, due not only to the large electrode surface area but also the reduced spacing between the electrodes, which minimizes electric resistance due to the poor conductivity of conventional wastewater. In terms of electrode designs, carbon brushes in pilot MFCs have been shown to outperform flat electrodes, while air cathodes surpass the performance of liquid catholyte configurations (108). For MECs, more studies at the pilot scale will be needed to ascertain the optimal electrode configuration.

The integration of METs in water reclamation facilities goes beyond the oxidation of organic matter in the main process train.

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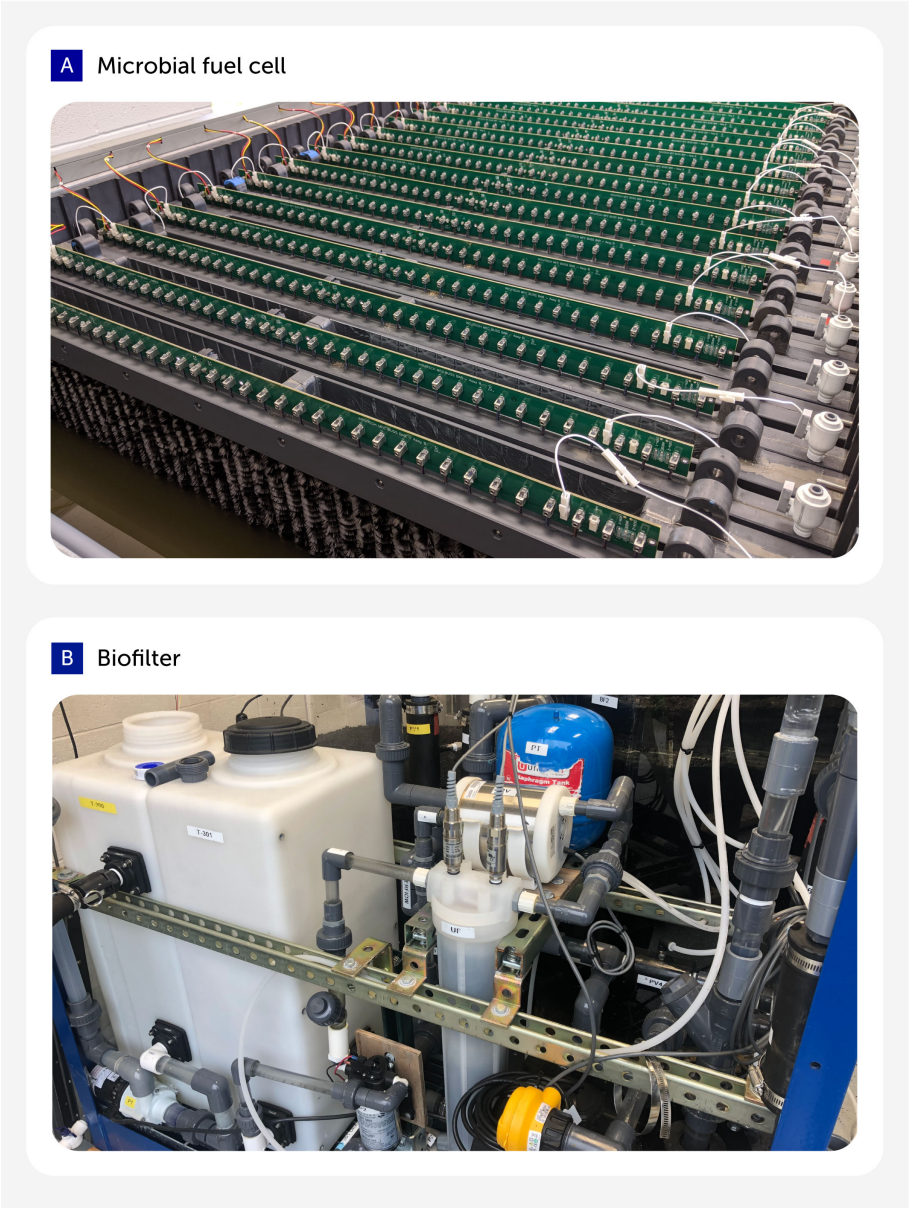


FIGURE 4
Integration example of microbial electrochemical technologies into water reclamation facilities: pilot-scale (1400 L) microbial fuel cell–biofilter system **(A, B)** for energy-efficient treatment of low chemical oxygen demand effluent.

For example, MECs can be coupled with AD to accelerate methanogenesis and increase methane yield (109, 110). In an integrated MEC–AD, the hydrolysis of organic matter is accelerated at the anode while the reducing equivalents produced are consumed at the cathode to increase the methane titer. This approach has already been commercialized by at least two companies: Cambrian Innovation in the United States with their EcoVolt reactor and WASE in the United Kingdom.

Another emerging application of MFC/MEC in wastewater is the remote sensing of pollutants for environmental monitoring. By leveraging the correlation between microbial degradation of pollutants and electric current responses, MET-based sensors have been used to detect various environmental parameters such as BOD and dissolved oxygen and pollutants such as toxic substances and heavy metals (111–113). The electric output of an MET sensor correlates with the concentration of these biodegradable or toxic compounds (analytes) when operated as an MFC, thereby allowing self-powered and real-time monitoring. Researchers have focused on the development of new selective electrode materials with improved conductivity and biocompatibility. Further, the engineering of microbial communities or genetically modified microorganisms to target specific analytes, leading to improved electron transfer efficiency and more accurate detection of pollutants like phenols and BOD, has been discussed (114, 115), but how these can be operated long-term in open environments remains to be solved. The integration of

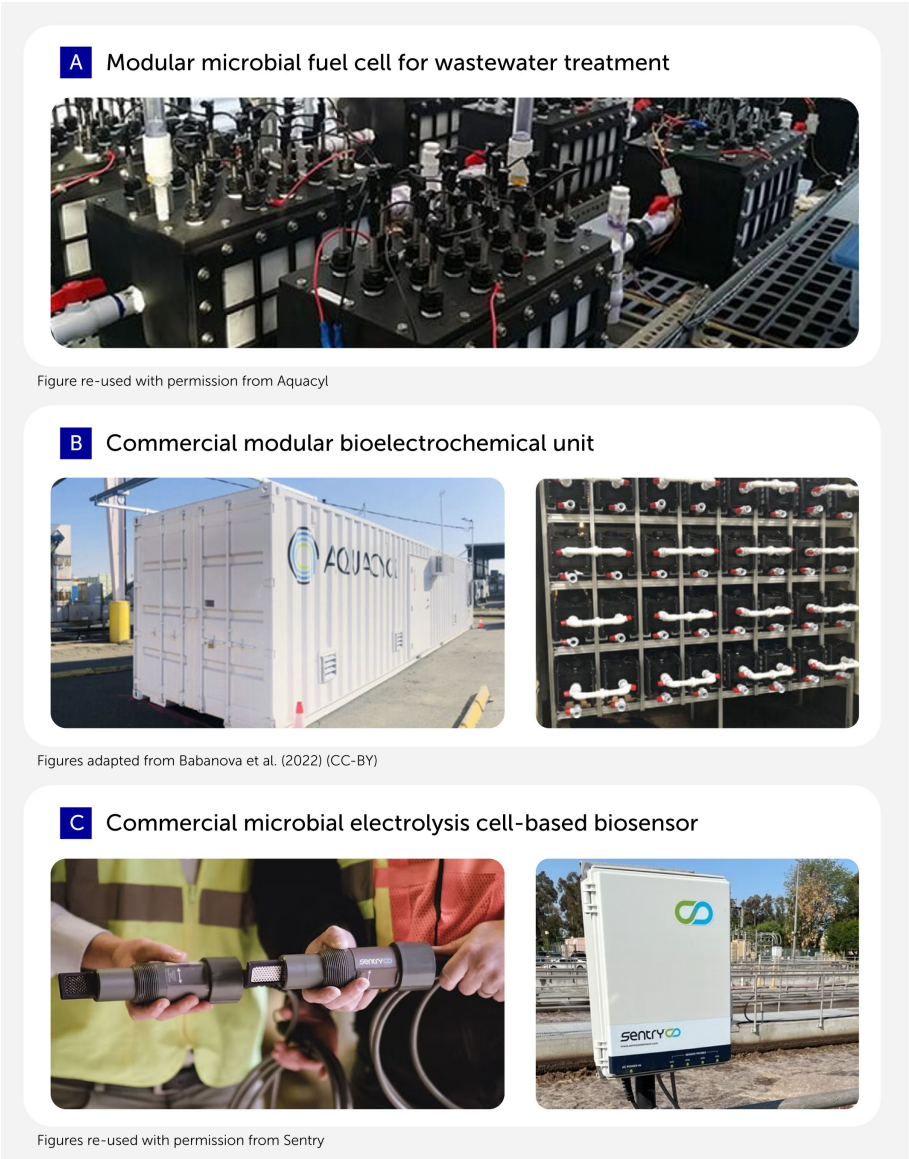
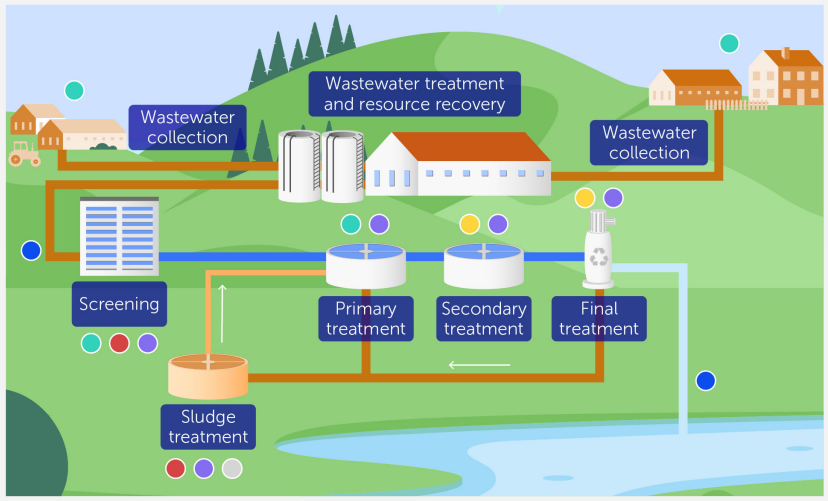


FIGURE 5
Examples of integration of microbial electrochemical technologies into water reclamation facilities. **(A)** Modular microbial fuel cells connected in series for swine wastewater treatment. This pilot demonstration has been carried out by Aquacycl. Re-used with permission from Aquacyl (116). **(B)** Aquacycl's modular bioelectrochemical treatment units Adapted from Babanova et al. (117). **(C)** microbial electrolysis cell-based biosensor for *in-situ* biological activity and water quality monitoring commercialized by Sentry. Re-used with permission from Sentry (118).

MFC-based biosensors with wireless data transmission systems enables remote and real-time monitoring of environmental conditions, with additional advantages of portability and low maintenance. Companies such as Sentry have developed MET-based sensors with electrodes pre-loaded with bacteria that measure electron generation from microbial activity and, by tracking changes in electron transfer rates, these sensors offer immediate insights into biological treatment efficiency, enabling optimization of processes and prevention of issues in water and wastewater treatment facilities (Figure 5C).

Research and development have also been directed toward investigating other integration possibilities. For example, MECs have been integrated with the partial nitritation–anammox process for nitrogen removal (119), where ammonium is oxidized with an anode serving as the electron acceptor, eliminating the need for prior nitrification. MECs have also been integrated with photocatalysis, which enables self-sustaining H_2 production with high efficiency and durability using actual wastewater (120). MFCs have been coupled with microalgae bioreactors to enhance CO_2 sequestration, nutrient recovery, and biofuel production (121). The integration of MFC technology into constructed wetlands ("CW-MFC"), typically via embedding an anode made of, for instance, granular material, into the anaerobic root zone of plants where oxygen is limited and placing a cathode in the oxygen-rich zone, has been proposed for uses such as remediation and decentralized wastewater treatment (122, 123). It may also be utilized as a final polishing step in a wastewater treatment



Integration points for METs in the wastewater treatment train and recommended complementary technologies:

- MECs and MFCs for energy and chemical recovery, respectively
- Ultrafiltration, membrane bioreactors, and biofilters to further polish effluent (for regulatory compliance)
- Anaerobic digestion and/or sludge conditioning for treating high-concentration liquids
- Modular METs, connected to modularized reactors, to address variability across plants
- MEC integrated with anaerobic digestion to increase methane yield
- MET-based sensors for environmental monitoring

Abbreviations: **MEC**, microbial electrolysis cells; **MET**, microbial electrochemical technology; **MFC**, microbial fuel cells

FIGURE 6
Potential integration points for microbial electrochemical technologies within a conventional wastewater treatment train, including key complementary technologies required for effective operation.

plant. The opportunities for MET integration into centralized or decentralized wastewater treatment are summarized in Figure 6. This figure shows the wide range of possible applications for an MET for wastewater treatment—depending on the main task and general scenario—as centralized or decentralized and as stand-alone or integrated systems.

A MET integration system that specifically applies in arid regions where a circular bioeconomy needs local closing of water cycles is combining MFCs with hydroponics or aquaponics (124). This integration involves MFC-based COD removal and associated energy recovery and a recirculation of the effluent that is still rich in nutrients (nitrogen and phosphorus) for plant growth. The integration of hydroponics with an MFC can be an effective solution for managing domestic wastewater in a more decentralized way. The dual functionality demonstrates the importance of resource recovery, thus enhancing the value of MFCs and advancing urban agriculture as well as vertical farming. Different reactor designs and stack configurations have been proposed for integrated hydroponic-MFC systems ("iHydroMET" (125), "Mi-Hy") for treating raw wastewater or urine while growing plants (126, 127) (Figure 7).

While this review cannot summarize all the integration possibilities, it is clear that MET provides a versatile platform for technology development for a variety of applications. The above developments demonstrate that METs are approaching early demonstration stages for integration with existing wastewater treatment infrastructure, with techno-economic hurdles being addressed, enabling this technology to deliver on its promise.

METs in the Global South

A further field of development and integration of this technology is in regions where existing wastewater infrastructure is not present. As an example, in areas of the Global South (especially in rural areas but also in disaster zones and refugee camps) there is often a lack of basic electricity supply in addition to sanitary facilities and safe water. This results in serious hygiene problems and, in addition, the lack of electricity to provide lighting in and around the toilet environments leads to a high safety risk, especially for women and young girls (128). MFCs could address this problem—provided that they are inexpensive and have robust designs (129, 130). Unlike mixed wastewater, urine does not contain solids that can lead to system clogging and, therefore, allows a much simpler reactor design. The use of human urine as a fuel for electricity was first proposed in 2012 (131), subsequently leading to several demonstrations whereby devices could simply be powered by urine-fed MFCs (132). Similar to conventional wastewater-fed

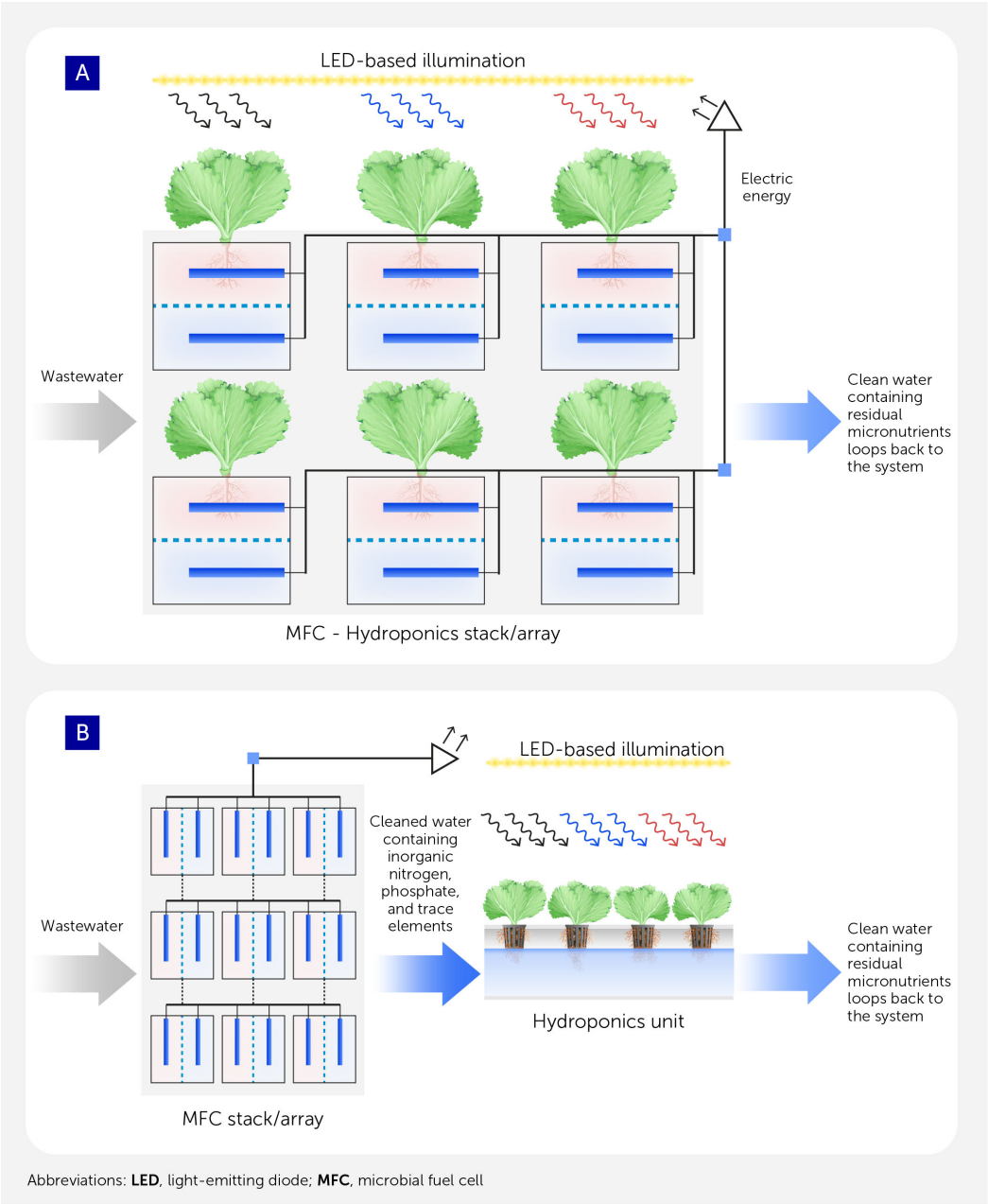


FIGURE 7
Simplified schematics of two integration approaches for microbial fuel cell (MFC)-hydroponic systems. **(A)** MFC integration into the hydroponic unit. **(B)** Use of a separate MFC stack in combination with a hydroponic unit. In both cases, the MFCs remove chemical oxygen demand and produce an electric current, which can be utilized to illuminate plants with an optimal wavelength composition. The cleaned water contains inorganic micro- and macronutrients to fertilize the hydroponics unit.

MFCs, urine-fed MFCs have also been found to produce highly alkaline catholytes (pH=11.0) with high levels of salinity. The catholyte may be utilized as a disinfectant or as a liquid fertilizer for plants with appropriate dilution factors due to the presence of concentrated nitrogen and phosphorus compounds and potassium ions (133–136).

Urine-fed MFCs were first trialed at the University of the West of England's Frenchay campus and at the Glastonbury Music Festival in June 2015 (137). The initial trials acted as a precursor to the longer-term field trials in real environments in Kisoro, Uganda (2017) (138), North Mathare, Kenya (2018) and Durban, South Africa (2019) (see Figure 8).

A 1.5 m^3 "bioelectric toilet" treating combined feces and urine has been demonstrated to produce usable electricity for LED lighting (130). Earlier work in Ghana ("Green Latrine") treated separated streams but achieved low electric energy yields (139).

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FIGURE 8
Pee Power® installation in Durban, South Africa. **(A)** Areal view of the installation site, **(B)** installed system in place receiving urine and greywater from the Community Ablution Block (CAB), **(C)** urinals, and toilet cubicles lit with electricity coming from the MFC stack.

Subsequent studies largely reported similarly low power outputs, with the notable exception of Pee Power®, which performed better because of the purpose-built urine-fed MFC stacks and tighter process control. In most low-yield systems, the mixed or poorly conditioned feed increased internal resistance and clogging; stacks were not optimally wired (series/parallel) to match load, electrode materials and geometries offered limited surface area, cathodes suffered biofouling and mineral scaling, and microbial communities on the anode were not actively managed. In contrast, Pee Power® used a urine-only operation, matched series–parallel stack configurations, high-surface-area electrodes, air-breathing cathodes with fouling/scale control, and controlled hydraulic residence times, which together reduced internal resistance and improved power density (140, 141).

State of METs development—technological readiness and techno-economics

Unlike several emerging technologies that follow the path of step-by-step extensive fundamental lab-based research followed by piloting and eventual commercial application, the METs research

community has been active in all these aspects right from the start. The norms and principles were still being established for the field in 2006 (95) but, by 2008, the first MFC pilot of 1 m^3 capacity was built and operated with brewery wastewater at Foster's Brewery in Yatala, Australia. Though the performance data of this pilot are not available in any scientific publication (85), this demonstration showed the intent of researchers to pursue upscaling METs and raised awareness of the challenges that arise when taking technologies from the lab to the field. Since this first study, the upscaling of METs has made significant progress, focusing on enhancing wastewater treatment performance and electric power output and field applications such as nutrient removal from wastewater (142), bioremediation of groundwater (143), and resource recovery from urine (144). In the previous decade, several groups reported pilot MFCs ranging from 10 L to >1000 L with variable performance (145). Among these, the ones that were operated for extended periods of more than a year include a 255 L MFC operated with real municipal wastewater in Germany (102) and an 850 L (1400 L total liquid volume) air-cathode MFC treating domestic-type wastewater at a centralized wastewater treatment facility in the United States (80).

One of the accepted approaches to measure the development of a technology is by tracking its evolution on a technology readiness level (TRL) scale. Initially designed by the National Aeronautics Space Administration in the 1970s, it has now been adopted globally to gauge the maturity of the technologies from lab to market (146). The development of different MET types in terms of TRLs is described here and has been the subject of more recent studies focusing on MET commercialization (147). A summary of the TRL of different technologies and applications is provided in Table 4.

Some examples of MET applications and their technoeconomic assessment include the 1.5 m^3 "bioelectric toilet" from Indian Institute of Technology Kharagpur that used 49 separator–electrode pairs with carbon-felt electrodes and a modified clayware separator. A preliminary techno-economic assessment estimated an initial investment of IN₹329,000 (≈ US$4,386) for a 10-year, five-person installation (130). In China, a 1.5 m^3 outdoor pilot with a biocathode (36 units) reported a stable power output of 406 ± 30 mW m^{-3} (effective liquid volume) and a CAPEX of US $1,702.1 (148), whereas in Switzerland, a dual-chamber MFC comprising 64 modules (16.25 L each) operated for one year in an underground wastewater treatment plant gallery, achieving 5.8–12.1% energy efficiency, 34–95% COD removal, and 5–15% coulombic efficiency (149). Stacked MFCs (series/parallel) continue to be explored to raise voltage/current and match loads (129, 150). Across pilots, real power densities remain modest and treatment performance varies, reflecting differences in stack configuration, electrode area, cathode management, and hydraulics.

Reported pilot costs for MFCs span roughly US$1,000–36,000 per m^3 of reactor volume, largely due to reactor construction, electrode and separator choices, and the degree of modular stacking versus single large reactors (145). For MECs, electrode materials and applied electricity dominate costs; improving energy efficiency and lowering the required voltage are thus central to viability.

MECs have undergone multiple pilots aimed at hydrogen recovery. Examples include a 1,000 L continuous-flow MEC on winery wastewater (98) and a sequence of pilots in the United Kingdom starting with 120 L units treating domestic wastewater for >12 months (105, 151, 152), to now progressing toward larger demonstrators of 0.5 m^3 per hour, and TRL 7-8 via METzero Technologies Ltd (153). An 150 L MEC in Spain (five electrode pairs in hydraulic/electric parallel) treating real wastewater for 63 days achieved ~80% total organic carbon and ~70% nitrogen removal (154). Commercial activity also includes methane-oriented, electrode-assisted processes (e.g., Cambrian Innovation; WASE), which blend conventional methanogenesis with enhanced rates from electrodes.

Given the slow march toward high-power standalone METs, several groups have combined MET components with established treatment platforms. Constructed wetlands-MFC ("electro-wetlands," "METland®/METfilter", "CW-MFC") have reached TRL 8 with pilots for urban and industrial effluents in multiple countries (155). Other integrations—hydroponics-MET, electrochemical MBRs, and bioelectrochemical anaerobic digesters (e.g., MEC-ADs)—are at TRL 4–6 (156). On the industrial side, Aquacycl reported an 110L, 12-unit stacked MFC treating swine wastewater continuously for >200 days with ~65% COD removal (101) and is expanding deployment in wastewater treatment markets (117).

TABLE 4 Estimated scales and technological readiness levels of different wastewater-associated microbial electrochemical technologies.

MET type	Main application	Real-world example(s)	Scale	~TRL (as of 2025)
Constructed wetlands (METland®)	Municipal/onsite wastewater treatment	Multiple full-scale METlands® (Spain/Denmark)	Full-scale beds; demo modules	7–8 (system prototype/demo in operational environment)
MFCs	COD removal; electricity output	Multiple pilot stacks reported	10–1000 L pilots	4–9 (component/bench to pilot/launched)
MFC-based biosensors	Online monitoring (BOD/toxicity/early-warning)	Field-deployable MFC sensors	Field trials; small pilot installations	6–9 (prototype in relevant operational env./launched)
MECs	Hydrogen generation; polishing wastewater	135 L, 6-month MEC pilots	100–1000 L pilots, months operation	5–6 (pilot/demo)
Urine treatment with MEC	NH_3 recovery (as $(NH_4)_2SO_4$/citrate) + H_2; P as struvite	Scaled-up MEC with NH_3/NH_4^+ recovery	Long-running pilots	5–6 (pilot/demo)

Abbreviations: **BOD**, biochemical oxygen demand; **COD**, chemical oxygen demand; **MEC**, microbial electrolysis cell; **MET**, microbial electrochemical technologies; **MFC**, microbial fuel cell; **TRL**, technological readiness level.

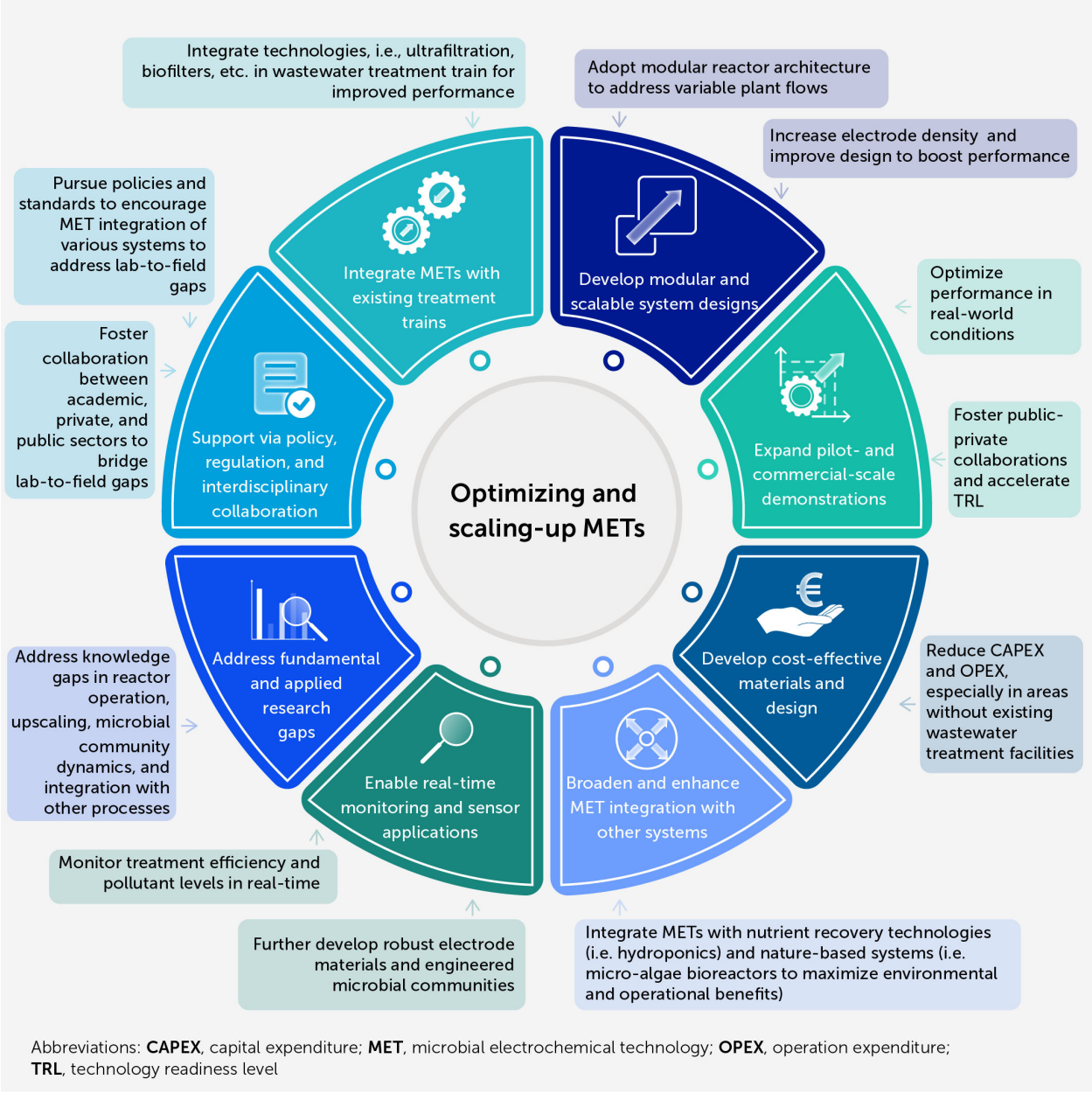

FIGURE 9
Recommendations for optimizing operations and scaling-up of microbial electrochemical technologies across the wastewater treatment train and in other relevant applications.

Across the MET family, most configurations sit between TRL 4 and TRL 8 (157). In wastewater applications, MFCs are best described as mid- to high-TRL in niche roles (e.g., environmental monitoring, with plant/sediment systems often around TRL 7–8), while large-scale treatment MFCs remain at pilot/demonstration, requiring further optimization before broad rollout. MECs are generally at TRL 6–7 with credible pilots for H_2 and CH_4-enhanced recovery, though economics hinge on materials and energy use.

The weight of field evidence indicates that MFCs are unlikely to serve as high-scale power generators. Their more realistic contribution is energy-efficient treatment and resource recovery in side-streams where loading, conductivity, and residence times can be controlled. For MECs, selective product recovery (H_2/CH_4/N-species) and coupling to existing assets offer the clearest path forward. Progress will depend on lower-cost, durable materials, standardized stack designs, fouling/scale control at the cathode, better power electronics/load matching, and, in MECs, higher energy efficiency at lower applied voltage.

Companies formed around METs include Trophos Energy, Lebone, Hy-SyEnce (MFCs), Cambrian Innovation and WASE (up-scaled MECs/biogas-enhanced), Plant-e (plant-based MFCs), Emefcy (MET-inspired systems), METzero Technologies Ltd., METFilter/IMDEA (METland®), and Aquacycl (MET-based treatment). Commercialization is occurring but at a deliberate

pace, with traction where METs complement rather than replace conventional treatment.

Conclusions

The authors of this article are convinced that METs could facilitate a paradigm shift by transforming waste into valuable resources such as energy, clean water, and nutrients. When considering all the potential and progress, one may ask why METs for wastewater management are not yet an integral component of the technosphere or, at least, occupy a set of application niches? One of the reasons is that wastewater treatment in industrialized countries is subject to very strict legal requirements regarding COD, nitrogen, and phosphorous removal. Implementation in regular operation requires (in addition to the necessary improvements in performance and costs discussed above) certified long-term performance data that meets legal requirements and convinces sewage treatment plant operators to use the technology.

In order to become successful, METs for wastewater management need to:

- achieve or even exceed required rates to allow roll-out for municipal wastewater. Most challenging is ensuring that METs replace or fit into the existing wastewater infrastructure of the northern hemisphere. Here, depreciation of >15 years of infrastructure as well as tight legal frameworks will make a market entry extraordinarily challenging. Given that land, especially in urban areas, is becoming increasingly precious, achieving or even exceeding the current rates (e.g., COD removal rates) of wastewater treatment that lead to a retention time of ca. 1 day could be a clear advantage.
- achieve efficient treatment of high-strength wastewater and recalcitrant substances. Acquacyl is a prime example: here a MET allows treating high-strength wastewater from the beverages industry using a modular MET with the harvest of electric energy being only a side benefit. Thus, other kinds of wastewater with different specificities, but the shared challenge of being conventionally only treated by tremendous energetic (and material) efforts, provide excellent niches for METs to come into play. Apart from treating these mostly industrial wastewaters with peculiar specificities, METs can also be used for widespread implementation. This can be achieved if METs meet demands that cannot be served by (low-cost) existing technologies; for instance, we foresee the removal of pharmaceuticals or chemicals like per- and polyfluoroalkyl substances from point sources as appealing entry points.
- provide added value when compared to existing technologies. As illustrated, METs can replace energy-demanding wastewater management processes, most prominently aerobic sludge treatment. Therefore, METs could decrease global energy consumption even when achieving energy self-sufficiency (see Table 1). Further to that, any validated use of wastewater as a resource, for both energy and material, that creates a real business case will lead to implementation.
- become included in (and be promoted by) the regulatory framework. The current legal framework, norms, and standards are tailored for a linear economy. Thus, they often do not suit a circular economy. This is also the case for using METs for wastewater management. For instance, the fertilizer derived from urine cannot, in most countries, be applied for aqua- or agriculture of food (and feed), and the cleaned wastewater is considered a free good. When re-adjusting our legal foundation and, thus, giving a (monetary) value to clean drinking water or promoting circularity via "waste" as resource, METs will certainly experience an upswing.

Even though great steps in the development of METs have taken place in the past 20 years, METs are not yet competitive in terms of technology and economy with other wastewater treatment and recovery technologies. Challenges remain in scaling METs for widespread adoption. High capital costs, material durability, and energy efficiency limitations must be addressed to enhance commercial viability. Pilot-scale implementations for decentralized sanitation and the integration of METs with anaerobic digestion, constructed wetlands, and membrane bioreactors demonstrate their potential in real-world settings.

Further research is needed to optimize reactor designs, reduce operational costs, and improve long-term stability. The future of METs lies in interdisciplinary collaboration, water industry interest, policy support, and technological innovation to bridge the gap between laboratory research and full-scale deployment (Figure 9). Although the challenges and tasks depicted in Figure 9 are located at very different levels and in different disciplines, they are equally important in terms of their significance and priority for successful MET development and implementation.

By harnessing the untapped potential of wastewater, METs can contribute significantly to sustainable development, energy security, and global sanitation goals and thus to the relevant SDGs. As the field progresses, METs may become a cornerstone of next-generation wastewater treatment, turning environmental challenges into economic and ecological opportunities.

Statements

Author contributions

Conceptualization: US, FH.

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Data availability statement

Publicly available datasets were analyzed in this study. This data can be found in the reference list. References are made to all used datasets.

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References

1. United Nations. *UN-Water SDG 6 Data Portal*. Available at: https://www.sdg6data.org/en

2. World Health Organization and the United Nations Children's Fund. *Progress on household drinking water, sanitation and hygiene 2000–2024: special focus on inequalities*. WHO; UNICEF (2025). Available at: https://data.unicef.org/resources/jmp-report-2025/

3. United Nations. *Blueprint for acceleration: sustainable development goal 6 synthesis report on water and sanitation 2023*. UN (2023). Available at: https://www.un-ilibrary.org/content/books/9789210026444

4. United Nations Educational, Scientific and Cultural Organization and UN-Water, World Water Assessment Programme. *The United Nations World Water Development Report 2017. Wastewater: the untapped resource*. Paris: UNESCO (2017). Available at: https://unesdoc.unesco.org/ark:/48223/pf0000247153

5. Jones ER, van Vliet MTH, Qadir M, Bierkens MFP. Country-level and gridded estimates of wastewater production, collection, treatment and reuse. *Earth Syst Sci Data* (2021) 13(2):237–54. doi: 10.5194/essd-13-237-2021

6. Dunn JB, Greene K, Vasquez-Arroyo E, Awais M, Gomez-Sanabria A, Kyle P, et al. Toward enhancing wastewater treatment with resource recovery in integrated assessment and computable general equilibrium models. *Environ Sci Technol Lett* (2024) 11(7):654–63. doi: 10.1021/acs.estlett.4c00280

7. Shizas I, Bagley DM. Experimental determination of energy content of unknown organics in municipal wastewater streams. *J Energy Eng* (2004) 130(2):45–53. doi: 10.1061/(ASCE)0733-9402(2004)130:2(45)

8. Heidrich ES, Curtis TP, Dolfing J. Determination of the internal chemical energy of wastewater. *Environ Sci Technol* (2011) 45(2):827–32. doi: 10.1021/es103058w

9. Korth B, Maskow T, Günther S, Harnisch F. Estimating the energy content of wastewater using combustion calorimetry and different drying processes. *Front Energy Res* (2017) 5:23. doi: 10.3389/fenrg.2017.00023

10. Dai ZY, Heidrich ES, Dolfing J, Jarvis AP. Determination of the relationship between the energy content of municipal wastewater and its chemical oxygen demand. *Environ Sci Technol Lett* (2019) 6:396–400. doi: 10.1021/acs.estlett.9b00253

11. Korth B, Heber C, Normant-Saremba M, Maskow T, Harnisch F. Precious data from tiny samples: revealing the correlation between energy content and the chemical oxygen demand of municipal wastewater by micro-bomb combustion calorimetry. *Front Energy Res* (2021) 9:705800. doi: 10.3389/fenrg.2021.705800

12. Logan BE. *Daily energy use and carbon emissions: fundamentals and applications for students and professionals*. Chichester: John Wiley & Sons Inc. (2022).

13. United States Energy Information Administration. *Short-term energy outlook Data Browser (electricity industry overview)*. Available at: https://www.eia.gov/outlooks/steo/data/browser//?v=19

14. International Energy Agency. *World Energy Outlook 2018*. Paris: IEA (2018). Available at: https://www.iea.org/reports/world-energy-outlook-2018

15. Hamawand I. Energy consumption in water/wastewater treatment industry—optimisation potentials. *Energies* (2023) 16(5):2433. doi: 10.3390/en16052433

16. Lamolinara B, Pérez-Martínez A, Guardado-Yordi E, Guillén Fiallos C, Diéguez-Santana K, Ruiz-Mercado GJ. Anaerobic digestate management, environmental impacts, and techno-economic challenges. *Waste Manag* (2022) 140:14–30. doi: 10.1016/j.wasman.2021.12.035

17. Metcalf & Eddy, Inc., Tchobanoglous G, Stensel H, Tsuchihashi R, Burton FL. *Wastewater engineering: treatment and resource recovery* (*5th edition*). New York: McGraw-Hill (2014).

18. Tonelli D, Rosa L, Gabrielli P, Parente A, Contino F. Cost-competitive decentralized ammonia fertilizer production can increase food security. *Nat Food* (2024) 5(6):469–79. doi: 10.1038/s43016-024-00979-y

19. Smith C, Hill AK, Torrente-Murciano L. Current and future role of Haber–Bosch ammonia in a carbon-free energy landscape. *Energy Environ Sci* (2020) 13 (2):331–44. doi: 10.1039/C9EE02873K

20. United States Geological Survey. *Mineral commodity summaries 2024*. Reston, VA: USGS (2024). Available at: https://pubs.usgs.gov/publication/mcs2024

21. Brownlie WJ, Sutton MA, Cordell D, Reay DS, Heal KV, Withers PJA, et al. Phosphorus price spikes: a wake-up call for phosphorus resilience. *Front Sustain Food Syst* (2023) 7:1088776. doi: 10.3389/fsufs.2023.1088776

22. STATISTA GmbH. Domestic sales of nitrogen fertilizers used in agriculture in Germany, 2010/11–2022/23 [online; German] (2024). Available at: https://de.statista.com/statistik/daten/studie/161843/umfrage/verbrauch-von-stickstoff-in-der-landwirtschaft/

23. STATISTA GmbH. Domestic sales of phosphate fertilizer in agriculture in Germany in the years 2010/11 to 2022/23 [online; German] (2024). Available at: https://de.statista.com/statistik/daten/studie/161844/umfrage/verbrauch-von-phosphat-in-der-landwirtschaft/

24. Tortajada C. Contributions of recycled wastewater to clean water and sanitation Sustainable Development Goals. *NPJ Clean Water* (2020) 3:22. doi: 10.1038/s41545-020-0069-3

25. United States Environmental Protection Agency, United States Agency for International Development. *Guidelines for water reuse*. Washington, D.C.: EPA (2012). Available at: https://www.epa.gov/sites/default/files/2019-08/documents/2012-guidelines-water-reuse.pdf

26. Public Utilities Board - Singapore's National Water Agency. *NEWater* [online] (2024). Available at: https://www.pub.gov.sg/Public/WaterLoop/OurWaterStory/NEWater/

27. Capodaglio AG. Fit-for-purpose urban wastewater reuse: analysis of issues and available technologies for sustainable multiple barrier approaches. *Crit Rev Environ Sci Technol* (2021) 51(15):1619–66. doi: 10.1080/10643389.2020.1763231

28. McCarty PL, Bae J, Kim J. Domestic wastewater treatment as a net energy producer–can this be achieved?*Environ Sci Technol* (2011) 45(17):7100–6. doi: 10.1021/es2014264

29. Heidrich ES, Edwards SR, Dolfing J, Cotterill SE, Curtis TP. Performance of a pilot scale microbial electrolysis cell fed on domestic wastewater at ambient temperatures for a 12 month period. *Bioresour Technol* (2014) 173:87–95. doi: 10.1016/j.biortech.2014.09.083

30. Schröder U, Harnisch F, Angenent LT. Microbial electrochemistry and technology: terminology and classification. *Energy Environ Sci* (2015) 8(2):513–9. doi: 10.1039/C4EE03359K

31. Pant D, Van Bogaert G, Diels L, Vanbroekhoven K. A review of the substrates used in microbial fuel cells (MFCs) for sustainable energy production. *Bioresour Technol* (2010) 101(6):1533–43. doi: 10.1016/j.biortech.2009.10.017

32. Lovley DR. Electromicrobiology. *Annu Rev Microbiol* (2012) 66:391–409. doi: 10.1146/annurev-micro-092611-150104

33. Walker DJ, Adhikari RY, Holmes DE, Ward JE, Woodard TL, Nevin KP, et al. Electrically conductive pili from pilin genes of phylogenetically diverse microorganisms. *ISME J* (2018) 12(1):48–58. doi: 10.1038/ismej.2017.141

34. Gupta D, Guzman MS, Bose A. Extracellular electron uptake by autotrophic microbes: physiological, ecological, and evolutionary implications. *J Ind Microbiol Biotechnol* (2020) 47(9–10):863–76. doi: 10.1007/s10295-020-02309-0

35. Paquete CM, Morgado L, Salgueiro CA, Louro RO. Molecular mechanisms of microbial extracellular electron transfer: the importance of multiheme cytochromes. *Front Biosci* (2022) 27(6):174. doi: 10.31083/j.fbl2706174

36. Gralnick JA, Bond DR. Electron transfer beyond the outer membrane: putting electrons to rest. *Annu Rev Microbiol* (2023) 77:517–39. doi: 10.1146/annurev-micro-032221-023725

37. Wang F, Gu Y, O'Brien JP, Yi SM, Yalcin SE, Srikanth V, et al. Structure of microbial nanowires reveals stacked hemes that transport electrons over micrometers. *Cell* (2019) 177(2):361–9.e10. doi: 10.1016/j.cell.2019.03.029

38. Reguera G, McCarthy KD, Mehta T, Nicoll JS, Tuominen MT, Lovley DR. Extracellular electron transfer via microbial nanowires. *Nature* (2005) 435(7045):1098–101. doi: 10.1038/nature03661

39. Giese B, Wang M, Gao J, Stoltz M, Müller P, Graber M. Electron relay race in peptides. *J Org Chem* (2009) 74(10):3621–5. doi: 10.1021/jo900375f

40. Vargas M, Malvankar NS, Tremblay PL, Leang C, Smith JA, Patel P, et al. Aromatic amino acids required for pili conductivity and long-range extracellular electron transport in Geobacter sulfurreducens. *MBio* (2013) 4(2):10–1128. doi: 10.1128/mBio.00105-13

41. Adhikari RY, Malvankar NS, Tuominen MT, Lovley DR. Conductivity of individual Geobacter pili. *RSC Adv* (2016) 6(10):8354–7. doi: 10.1039/C5RA28092C

42. Gu Y, Srikanth V, Salazar-Morales AI, Jain R, O'Brien JP, Yi SM, et al. Structure of Geobacter pili reveals secretory rather than nanowire behaviour. *Nature* (2021) 597 (7876):430–4. doi: 10.1038/s41586-021-03857-w

43. Koch C, Huber KJ, Bunk B, Overmann J, Harnisch F. Trophic networks improve the performance of microbial anodes treating wastewater. *npj Biofilms Microbiomes* (2019) 5(1):27. doi: 10.1038/s41522-019-0100-y

44. Xu S, Zhang Y, Luo L, Liu H. Startup performance of microbial electrolysis cell assisted anaerobic digester (MEC-AD) with pre-acclimated activated carbon. *Bioresour Technol Rep* (2019) 5:91–8. doi: 10.1016/j.biteb.2018.12.007

45. Cao M, Feng Y, Wang N, Li Y, Li N, Liu J, et al. Electrochemical regulation on the metabolism of anode biofilms under persistent exogenous bacteria interference. *Electrochim Acta* (2020) 340:135922. doi: 10.1016/j.electacta.2020.135922

46. Huang Q, Liu Y, Dhar BR. A critical review of microbial electrolysis cells coupled with anaerobic digester for enhanced biomethane recovery from high-strength feedstocks. *Crit Rev Environ Sci Technol* (2022) 52(1):50–89. doi: 10.1080/10643389.2020.1813065

47. Elsayed A, Laqa Kakar F, Mustafa Abdelrahman A, Ahmed N, AlSayed A, Sherif Zagloul M, et al. Enhancing anaerobic digestion efficiency: a comprehensive review on innovative intensification technologies. *Energy Convers Manage* (2024) 320:118979. doi: 10.1016/j.enconman.2024.118979

48. Cheng S, Xing D, Call DF, Logan BE. Direct biological conversion of electrical current into methane by electromethanogenesis. *Environ Sci Technol* (2009) 43(10):3953–8. doi: 10.1021/es803531g

49. Yee MO, Snoeyenbos-West OL, Thamdrup B, Ottosen LDM, Rotaru A-E. Extracellular electron uptake by two Methanosarcina species. *Front Energy Res* (2019) 7:29. doi: 10.3389/fenrg.2019.00029

50. Pandit S, Pandit C, Mathuriya AS, Chatterjee S, Jadhav DA, Yadav KK, et al. Wired for energy: electromethanogenesis redefining anaerobic digestion. *Process Saf Environ Prot* (2024) 185:588–601. doi: 10.1016/j.psep.2024.03.021

51. Dzofou Ngoumelah D, Heggeset TMB, Haugen T, Sulheim S, Wentzel A, Harnisch F, et al. Effect of model methanogens on the electrochemical activity, stability, and microbial community structure of Geobacter spp. dominated biofilm anodes. *npj Biofilms Microbiomes* (2024) 10(1):17. doi: 10.1038/s41522-024-00490-z

52. Jones LM, El Aidy S. Electroactive ecosystem insights from corrosion microbiomes inform gut microbiome modulation. *ISME J* (2025) 19(1):wraf112. doi: 10.1093/ismejo/wraf112

53. Santoro C, Babanova S, Cristiani P, Artyushkova K, Atanassov P, Bergel A, et al. How comparable are microbial electrochemical systems around the globe? An electrochemical and microbiological cross-laboratory study. *ChemSusChem* (2021) 14(11):2313–30. doi: 10.1002/cssc.202100294

54. Li Q, Wang Y, An C, Jia H, Wang J. Exploring novel approaches to enhance start-up process in microbial fuel cell: a comprehensive review. *J Water Process Eng* (2024) 63:105425. doi: 10.1016/j.jwpe.2024.105425

55. Liu Y, Harnisch F, Fricke K, Sietmann R, Schröder U. Improvement of the anodic bioelectrocatalytic activity of mixed culture biofilms by a simple consecutive electrochemical selection procedure. *Biosens Bioelectron* (2008) 24(4):1012–7. doi: 10.1016/j.bios.2008.08.001

56. Baudler A, Riedl S, Schröder U. Long-term performance of primary and secondary electroactive biofilms using layered corrugated carbon electrodes. *Front Energy Res* (2014) 2:1–6. doi: 10.3389/fenrg.2014.00030

57. Aelterman P, Versichele M, Marzorati M, Boon N, Verstraete W. Loading rate and external resistance control the electricity generation of microbial fuel cells with different three-dimensional anodes. *Bioresour Technol* (2008) 99(18):8895–902. doi: 10.1016/j.biortech.2008.04.061

58. Boghani HC, Kim JR, Dinsdale RM, Guwy AJ, Premier GC. Control of power sourced from a microbial fuel cell reduces its start-up time and increases bioelectrochemical activity. *Bioresour Technol* (2013) 140:277–85. doi: 10.1016/j.biortech.2013.04.087

59. Obata O, Salar-Garcia MJ, Greenman J, Kurt H, Chandran K, Ieropoulos I. Development of efficient electroactive biofilm in urine-fed microbial fuel cell cascades for bioelectricity generation. *J Environ Manage* (2020) 258:109992. doi: 10.1016/j.jenvman.2019.109992

60. Thauer RK, Jungermann K, Decker K. Energy conservation in chemotrophic anaerobic bacteria. *Bacteriol Rev* (1977) 41(1):100–80. doi: 10.1128/br.41.1.100-180.1977

61. Fricke K, Harnisch F, Schröder U. On the use of cyclic voltammetry for the study of anodic electron transfer in microbial fuel cells. *Energy Environ Sci* (2008) 1(1):144. doi: 10.1039/b802363h

62. Logan BE, Call D, Cheng S, Hamelers HVM, Sleutels THJA, Jeremiasse AW, et al. Microbial electrolysis cells for high yield hydrogen gas production from organic matter. *Environ Sci Technol* (2008) 42(23):8630–40. doi: 10.1021/es801553z

63. Harnisch F, Schröder U. From MFC to MXC: chemical and biological cathodes and their potential for microbial bioelectrochemical systems. *Chem Soc Rev* (2010) 39(11):4433–48. doi: 10.1039/c003068f

64. Anjum A, Ali Mazari S, Hashmi Z, Sattar Jatoi A, Abro R. A review of role of cathodes in the performance of microbial fuel cells. *J Electroanal Chem* (2021) 899:115673. doi: 10.1016/j.jelechem.2021.115673

65. Massazza D, Robledo AJ, Rodriguez Simón CN, Busalmen JP, Bonanni S. Energetics, electron uptake mechanisms and limitations of electroautotrophs growing on biocathodes – a review. *Bioresour Technol* (2021) 342:125893. doi: 10.1016/j.biortech.2021.125893

66. Deutzmann JS, Kracke F, Spormann AM. Microbial electromethanogenesis powered by curtailed renewable electricity. *Cell Rep Phys Sci* (2023) 4(8):101515. doi: 10.1016/j.xcrp.2023.101515

67. Yee MO, Ottosen LDM, Rotaru A-E. Electrical current disrupts the electron transfer in defined consortia. *Microb Biotechnol* (2024) 17(1):e14373. doi: 10.1111/1751-7915.14373

68. Logan BE, Hamelers B, Rozendal R, Schröder U, Keller J, Freguia S, et al. Microbial fuel cells: methodology and technology. *Environ Sci Technol* (2006) 40(17):5181–92. doi: 10.1021/es0605016

69. Liu B, Williams I, Li Y, Wang L, Bagtzoglou A, McCutcheon J, et al. Towards high power output of scaled-up benthic microbial fuel cells (BMFCs) using multiple electron collectors. *Biosens Bioelectron* (2016) 79:435–41. doi: 10.1016/j.bios.2015.12.077

70. Rossi R, Hall DM, Wang X, Regan JM, Logan BE. Quantifying the factors limiting performance and rates in microbial fuel cells using the electrode potential slope analysis combined with electrical impedance spectroscopy. *Electrochim Acta* (2020) 348:136330. doi: 10.1016/j.electacta.2020.136330

71. Rossi R, Logan BE. Using an anion exchange membrane for effective hydroxide ion transport enables high power densities in microbial fuel cells. *Chem Eng J* (2021) 422:130150. doi: 10.1016/j.cej.2021.130150

72. Rossi R, Baek G, Saikaly PE, Logan BE. Continuous flow microbial flow cell with an anion exchange membrane for treating low conductivity and poorly buffered wastewater. *ACS Sustain Chem Eng* (2021) 9(7):2946–54. doi: 10.1021/acssuschemeng.0c09144

73. Fan Y, Han S-K, Liu H. Improved performance of CEA microbial fuel cells with increased reactor size. *Energy Environ Sci* (2012) 5(8):8273–80. doi: 10.1039/C2EE21964F

74. Lee H-S, Parameswaran P, Kato-Marcus A, Torres CI, Rittmann BE. Evaluation of energy-conversion efficiencies in microbial fuel cells (MFCs) utilizing fermentable and non-fermentable substrates. *Water Res* (2008) 42(6–7):1501–10. doi: 10.1016/j.watres.2007.10.036

75. Rossi R, Baek G, Logan BE. Vapor-fed cathode microbial electrolysis cells with closely spaced electrodes enables greatly improved performance. *Environ Sci Technol* (2022) 56(2):1211–20. doi: 10.1021/acs.est.1c06769

76. Jiang J, Lopez-Ruiz JA, Bian Y, Sun D, Yan Y, Chen X, et al. Scale-up and techno-economic analysis of microbial electrolysis cells for hydrogen production from wastewater. *Water Res* (2023) 241:120139. doi: 10.1016/j.watres.2023.120139

77. Swaminathan P, Ghosh A, Sunantha G, Sivagami K, Mohanakrishna G, Aishwarya S, et al. A comprehensive review of microbial electrolysis cells: integrated for wastewater treatment and hydrogen generation. *Process Saf Environ Prot* (2024) 190:458–74. doi: 10.1016/j.psep.2024.08.032

78. Wang L, Du H, Elsyed AFN, Yun N, Wang X, Rossi R. Impact of reactor architecture and design parameters on the performance of microbial electrolysis cells revealed by the electrode potential slope analysis. *Electrochim Acta* (2024) 485:144072. doi: 10.1016/j.electacta.2024.144072

79. Zhang X, He W, Ren L, Stager J, Evans PJ, Logan BE. COD removal characteristics in air-cathode microbial fuel cells. *Bioresour Technol* (2015) 176:23–31. doi: 10.1016/j.biortech.2014.11.001

80. Rossi R, Hur AY, Page MA, Thomas AO, Butkiewicz JJ, Jones DW, et al. Pilot scale microbial fuel cells using air cathodes for producing electricity while treating wastewater. *Water Res* (2022) 215:118208. doi: 10.1016/j.watres.2022.118208

81. Rossi R, Cario BP, Santoro C, Yang W, Saikaly PE, Logan BE. Evaluation of electrode and solution area-based resistances enables quantitative comparisons of factors impacting microbial fuel cell performance. *Environ Sci Technol* (2019) 53(7):3977–86. doi: 10.1021/acs.est.8b06004

82. Aguirre-Sierra A, Bacchetti-De Gregoris T, Berná A, Salas JJ, Aragón C, Esteve-Núñez A. Microbial electrochemical systems outperform fixed-bed biofilters in cleaning up urban wastewater. *Environ Sci: Water Res Technol* (2016) 2(6):984–93. doi: 10.1039/C6EW00172F

83. Ge Z, Li J, Xiao L, Tong Y, He Z. Recovery of electrical energy in microbial fuel cells. *Environ Sci Technol Lett* (2014) 1(2):137–41. doi: 10.1021/ez4000324

84. Caizán-Juanarena L, Sleutels T, Kokko M, Berenguer R, ter Heijne A. Fluidized and fixed granular beds of activated carbon as electrodes in microbial electrochemical technologies. In: Yadav AK, Srivastava P, Noori MT, Zhang Y, editors. *Emerging trends and advances in microbial electrochemical technologies*. Amsterdam: Elsevier (2024). 425–78. doi: 10.1016/B978-0-443-15557-4.00024-1

85. Bird H, Heidrich ES, Leicester DD, Theodosiou P. Pilot-scale Microbial Fuel Cells (MFCs): a meta-analysis study to inform full-scale design principles for optimum wastewater treatment. *J Clean Prod* (2022) 346:131227. doi: 10.1016/j.jclepro.2022.131227

86. Santoro C, Arbizzani C, Erable B, Ieropoulos I. Microbial fuel cells: from fundamentals to applications. A review. *J Power Sources* (2017) 356:225–44. doi: 10.1016/j.jpowsour.2017.03.109

87. Matsena MT, Mabuse M, Tichapondwa SM, Chirwa EMN. Improved performance and cost efficiency by surface area optimization of granular activated carbon in air-cathode microbial fuel cell. *Chemosphere* (2021) 281:130941. doi: 10.1016/j.chemosphere.2021.130941

88. Lei Y, Du M, Kuntke P, Saakes M, van der Weijden R, Buisman CJN. Energy efficient phosphorus recovery by microbial electrolysis cell induced calcium phosphate precipitation. *ACS Sustain Chem Eng* (2019) 7(9):8860–7. doi: 10.1021/acssuschemeng.9b00867

89. Kuntke P, Sleutels THJA, Rodríguez Arredondo M, Georg S, Barbosa SG, ter Heijne A, et al. (Bio)electrochemical ammonia recovery: progress and perspectives. *Appl Microbiol Biotechnol* (2018) 102(9):3865–78. doi: 10.1007/s00253-018-8888-6

90. Zamora P, Georgieva T, Ter Heijne A, Sleutels THJA, Jeremiasse AW, Saakes M, et al. Ammonia recovery from urine in a scaled-up Microbial Electrolysis Cell. *J Power Sources* (2017) 356:491–9. doi: 10.1016/j.jpowsour.2017.02.089

91. Harnisch F, Schröder U, Scholz F. The suitability of monopolar and bipolar ion exchange membranes as separators for biological fuel cells. *Environ Sci Technol* (2008) 42(5):1740–6. doi: 10.1021/es702224a

92. Hou D, Iddya A, Chen X, Wang M, Zhang W, Ding Y, et al. Nickel-based membrane electrodes enable high-rate electrochemical ammonia recovery. *Environ Sci Technol* (2018) 52(15):8930–8. doi: 10.1021/acs.est.8b01349

93. Shaw DR, Ali M, Katuri KP, Gralnick JA, Reimann J, Mesman R, et al. Extracellular electron transfer-dependent anaerobic oxidation of ammonium by anammox bacteria. *Nat Commun* (2020) 11(1):2058. doi: 10.1038/s41467-020-16016-y

94. Pous N, Bañeras L, Corvini PF-X, Liu S-J, Puig S. Direct ammonium oxidation to nitrogen gas (Dirammox) in Alcaligenes strain HO-1: the electrode role. *Environ Sci Ecotechnol* (2023) 15:100253. doi: 10.1016/j.ese.2023.100253

95. Yan X, Liu D, Klok JBM, de Smit SM, Buisman CJN, ter Heijne A. Enhancement of ammonium oxidation at Microoxic bioanodes. *Environ Sci Technol* (2023) 57(31):11561–71. doi: 10.1021/acs.est.3c02227

96. Cusick RD, Logan BE. Phosphate recovery as struvite within a single chamber microbial electrolysis cell. *Bioresour Technol* (2012) 107:110–5. doi: 10.1016/j.biortech.2011.12.038

97. Wang Y, Kuntke P, Saakes M, van der Weijden RD, Buisman CJN, Lei Y. Electrochemically mediated precipitation of phosphate minerals for phosphorus removal and recovery: progress and perspective. *Water Res* (2022) 209:117891. doi: 10.1016/j.watres.2021.117891

98. Cusick RD, Bryan B, Parker DS, Merrill MD, Mehanna M, Kiely PD, et al. Performance of a pilot-scale continuous flow microbial electrolysis cell fed winery wastewater. *Appl Microbiol Biotechnol* (2011) 89(6):2053–63. doi: 10.1007/s00253-011-3130-9

99. Bowman K, Elaiuy M, Fudge G, Rutland H, Gambier W, Hembury T, et al. Performance characteristics of a pilot-scale electromethanogenic reactor treating brewery wastewater. *Energies* (2025) 18(11):2939. doi: 10.3390/en18112939

100. Guerrero-Sodric O, Baeza JA, Guisasola A. Scaling up microbial electrolysis cells (MECs) for hydrogen production: design, construction and operation of a 1 m3 pilot plant in an urban wastewater treatment plant. *Chem Eng J* (2025) 523:168035. doi: 10.1016/j.cej.2025.168035

101. Babanova S, Jones J, Phadke S, Lu MQ, Angulo C, Garcia J, et al. Continuous flow, large-scale, microbial fuel cell system for the sustained treatment of swine waste. *Water Environ Res* (2020) 92(1):60–72. doi: 10.1002/wer.1183

102. Hiegemann H, Littfinski T, Krimmler S, Lübken M, Klein D, Schmelz K-G, et al. Performance and inorganic fouling of a submergible 255 L prototype microbial fuel cell module during continuous long-term operation with real municipal wastewater under practical conditions. *Bioresour Technol* (2019) 294:122227. doi: 10.1016/j.biortech.2019.122227

103. Zinadini S, Zinatizadeh AA, Rahimi M, Vatanpour V, Bahrami K. Energy recovery and hygienic water production from wastewater using an innovative integrated microbial fuel cell–membrane separation process. *Energy* (2017) 141:1350–62. doi: 10.1016/j.energy.2017.11.057

104. Ren L, Ahn Y, Logan BE. A two-stage microbial fuel cell and anaerobic fluidized bed membrane bioreactor (MFC-AFMBR) system for effective domestic wastewater treatment. *Environ Sci Technol* (2014) 48(7):4199–206. doi: 10.1021/es500737m

105. Leicester DD, Amezaga JM, Moore A, Heidrich ES. Optimising the hydraulic retention time in a pilot-scale microbial electrolysis cell to achieve high volumetric treatment rates using concentrated domestic wastewater. *Molecules* (2020) 25(12):2945. doi: 10.3390/molecules25122945

106. Koskue V, Pyrhönen V-P, Freguia S, Ledezma P, Kokko M. Modelling and techno-economic assessment of (bio)electrochemical nitrogen removal and recovery from reject water at full WWTP scale. *J Environ Manage* (2022) 319:115747. doi: 10.1016/j.jenvman.2022.115747

107. Jiang J, Du L, Si B, Kawale HD, Wang Z, Summers S, et al. Pilot microbial electrolysis cell closes the hydrogen loop for hydrothermal wet waste conversion to jet fuel. *Water Res* (2025) 268(A):122644. doi: 10.1016/j.watres.2024.122644

108. Rossi R, Logan BE. Impact of reactor configuration on pilot-scale microbial fuel cell performance. *Water Res* (2022) 225:119179. doi: 10.1016/J.WATRES.2022.119179

109. Liu W, Cai W, Guo Z, Wang L, Yang C, Varrone C, et al. Microbial electrolysis contribution to anaerobic digestion of waste activated sludge, leading to accelerated methane production. *Renew Energy* (2016) 91:334–9. doi: 10.1016/j.renene.2016.01.082

110. Zhao L, Wang X-T, Chen K-Y, Wang Z-H, Xu X-J, Zhou X, et al. The underlying mechanism of enhanced methane production using microbial electrolysis cell assisted anaerobic digestion (MEC-AD) of proteins. *Water Res* (2021) 201:117325. doi: 10.1016/j.watres.2021.117325

111. Li T, Wang X, Zhou Q, Liao C, Zhou L, Wan L, et al. Swift acid rain sensing by synergistic rhizospheric bioelectrochemical responses. *ACS Sens* (2018) 3(7):1424–30. doi: 10.1021/acssensors.8b00401

112. Do MH, Ngo HH, Guo W, Chang SW, Nguyen DD, Liu Y, et al. Microbial fuel cell-based biosensor for online monitoring wastewater quality: a critical review. *Sci Total Environ* (2020) 712:135612. doi: 10.1016/j.scitotenv.2019.135612

113. Chu N, Liang Q, Hao W, Jiang Y, Liang P, Zeng RJ. Microbial electrochemical sensor for water biotoxicity monitoring. *Chem Eng J* (2021) 404:127053. doi: 10.1016/j.cej.2020.127053

114. Sonawane JM, Ezugwu CI, Ghosh PC. Microbial fuel cell-based biological oxygen demand sensors for monitoring wastewater: state-of-the-art and practical applications. *ACS Sens* (2020) 5(8):2297–316. doi: 10.1021/acssensors.0c01299

115. Han Y, Liao C, Meng X, Zhao Q, Yan X, Tian L, et al. Switchover of electrotrophic and heterotrophic respirations enables the biomonitoring of low concentration of BOD in oxygen-rich environment. *Water Res* (2023) 235:119897. doi: 10.1016/j.watres.2023.119897

116. Aquacyl [website]. BETT swine waste treatment (2024). Available online at: https://aquacycl.com/resources/case-studies/bett-swine-waste-treatment/.

117. Babanova S, Jones J, Wiseman K, Soles J, Garcia J, Huerta P, et al. Bioelectrochemical treatment technology—the new practical approach for wastewater management and GHG emissions reduction. *Front Chem Eng* (2022) 4:832505. doi: 10.3389/fceng.2022.832505

118. Sentry [website]. Available from: https://www.sentrywatertech.com/.

119. Vilajeliu-Pons A, Koch C, Balaguer MD, Colprim J, Harnisch F, Puig S. Microbial electricity driven anoxic ammonium removal. *Water Res* (2018) 130:168–75. doi: 10.1016/j.watres.2017.11.059

120. Lu L, Vakki W, Aguiar JA, Xiao C, Hurst K, Fairchild M, et al. Unbiased solar H^2 production with current density up to 23 mA cm^{-2} by Swiss-cheese black Si coupled with wastewater bioanode. *Energy Environ Sci* (2019) 12(3):1088–99. doi: 10.1039/C8EE03673J

121. Arun S, Sinharoy A, Pakshirajan K, Lens PNL. Algae based microbial fuel cells for wastewater treatment and recovery of value-added products. *Renew Sustain Energy Rev* (2020) 132:110041. doi: 10.1016/j.rser.2020.110041

122. Gupta S, Srivastava P, Patil SA, Yadav AK. A comprehensive review on emerging constructed wetland coupled microbial fuel cell technology: potential applications and challenges. *Bioresour Technol* (2021) 320(Part B):124376. doi: 10.1016/j.biortech.2020.124376

123. Jacobs DG, Kachienga LO, Rikhotso MC, Abia ALK, Traoré AN, Potgieter N. Assessing the current situation of constructed wetland-microbial fuel cells as an alternative power generation and wastewater treatment in developing countries. *Front Energy Res* (2024) 12:1448730. doi: 10.3389/fenrg.2024.1448730

124. Pous N, Korth B, Osset-Álvarez M, Balaguer MD, Harnisch F, Puig S. Electrifying biotrickling filters for the treatment of aquaponics wastewater. *Bioresour Technol* (2021) 319:124221. doi: 10.1016/j.biortech.2020.124221

125. Yadav RK, Sahoo S, Patil SA. Performance evaluation of the integrated hydroponics-microbial electrochemical technology (iHydroMET) for decentralized domestic wastewater treatment. *Chemosphere* (2022) 288(2):132514. doi: 10.1016/j.chemosphere.2021.132514

126. Yadav RK, Patil SA. Integrated Hydroponics-Microbial Electrochemical Technology (iHydroMET) is promising for Olericulture along with domestic wastewater management. *Bioresour Technol Rep* (2023) 22:101428. doi: 10.1016/j.biteb.2023.101428

127. Gajda I, Obata O, Greenman J, Ieropoulos IA. Electroosmotically generated disinfectant from urine as a by-product of electricity in microbial fuel cell for the inactivation of pathogenic species. *Sci Rep* (2020) 10(1):5533. doi: 10.1038/s41598-020-60626-x

128. O'Reilly K. From toilet insecurity to toilet security: creating safe sanitation for women and girls. *WIREs Water* (2016) 3(1):19–24. doi: 10.1002/wat2.1122

129. Das I, Ghangrekar MM, Satyakam R, Srivastava P, Khan S, Pandey HN. On-site sanitary wastewater treatment system using 720-L stacked microbial fuel cell: case study. *J Hazard Toxic Radioact Waste* (2020) 24(3). doi: 10.1061/(ASCE)HZ.2153-5515.0000518

130. Jadhav DA, Das I, Ghangrekar MM, Pant D. Moving towards practical applications of microbial fuel cells for sanitation and resource recovery. *J Water Process Eng* (2020) 38:101566. doi: 10.1016/j.jwpe.2020.101566

131. Ieropoulos I, Greenman J, Melhuish C. Urine utilisation by microbial fuel cells; Energy fuel for the future. *Phys Chem Chem Phys* (2012) 14(1):94–8. doi: 10.1039/c1cp23213d

132. Santoro C, Garcia MJS, Walter XA, You J, Theodosiou P, Gajda I, et al. Urine in bioelectrochemical systems: an overall review. *ChemElectroChem* (2020) 7(6):1312–31. doi: 10.1002/CELC.201901995

133. Ledezma P, Kuntke P, Buisman CJN, Keller J, Freguia S. Source-separated urine opens golden opportunities for microbial electrochemical technologies. *Trends Biotechnol* (2015) 33(4):214–20. doi: 10.1016/j.tibtech.2015.01.007

134. Ieropoulos I, Pasternak G, Greenman J. Urine disinfection and in *situ* pathogen killing using a microbial fuel cell cascade system. *PloS One* (2017) 12(5):e0176475. doi: 10.1371/journal.pone.0176475

135. Jermakka J, Freguia S, Kokko M, Ledezma P. Electrochemical system for selective oxidation of organics over ammonia in urine. *Environ Sci: Water Res Technol* (2021) 7(5):942–55. doi: 10.1039/D0EW01057J

136. Koskue V, Freguia S. Optimised start-up strategy for bioelectrochemical systems operating on hydrolysed human urine. *Bioelectrochemistry* (2024) 158:108706. doi: 10.1016/j.bioelechem.2024.108706

137. Ieropoulos IA, Stinchcombe A, Gajda I, Forbes S, Merino-Jimenez I, Pasternak G, et al. Pee power urinal–microbial fuel cell technology field trials in the context of sanitation. *Environ Sci: Water Res Technol* (2016) 2(2):336–43. doi: 10.1039/c5ew00270b

138. You J, Staddon C, Cook A, Walker J, Boulton J, Powell W, et al. Multidimensional benefits of improved sanitation: evaluating 'PEE POWER®' in Kisoro, Uganda. *Int J Environ Res Public Health* (2020) 17(7):2175. doi: 10.3390/ijerph17072175

139. Castro CJ, Goodwill JE, Rogers B, Henderson M, Butler CS. Deployment of the microbial fuel cell latrine in Ghana for decentralized sanitation. *J Water Sanit Hyg Dev* (2014) 4(4):663–71. doi: 10.2166/washdev.2014.020

140. Kretzschmar J, Riedl S, Brown RK, Schröder U, Harnisch F. ELatrine: lessons learned from the development of a low-tech MFC based on cardboard electrodes for the treatment of human feces. *J Electrochem Soc* (2017) 164(3):H3065–72. doi: 10.1149/2.0121703jes

141. AlSayed A, Soliman M, Eldyasti A. Microbial fuel cells for municipal wastewater treatment: from technology fundamentals to full-scale development. *Renew Sustain Energy Rev* (2020) 134:110367. doi: 10.1016/j.rser.2020.110367

142. Goglio A, Marzorati S, Rago L, Pant D, Cristiani P, Schievano A. Microbial recycling cells: first steps into a new type of microbial electrochemical technologies, aimed at recovering nutrients from wastewater. *Bioresour Technol* (2019) 277:117–27. doi: 10.1016/j.biortech.2019.01.039

143. Ceballos-Escalera A, Pous N, Bañeras L, Balaguer MD, Puig S. Advancing towards electro-bioremediation scaling-up: on-site pilot plant for successful nitrate-contaminated groundwater treatment. *Water Res* (2024) 256:121618. doi: 10.1016/j.watres.2024.121618

144. Cid CA, Stinchcombe A, Ieropoulos I, Hoffmann MR. Urine microbial fuel cells in a semi-controlled environment for onsite urine pre-treatment and electricity production. *J Power Sources* (2018) 400:441–8. doi: 10.1016/j.jpowsour.2018.08.051

145. Li C, Liang D, Tian Y, Liu S, He W, Li Z, et al. Sorting out the latest advances in separators and pilot-scale microbial electrochemical systems for wastewater treatment: concomitant development, practical application, and future perspective. *Environ Sci Technol* (2024) 58(22):9471–86. doi: 10.1021/acs.est.4c03169

146. Héder M. From NASA to EU: the evolution of the TRL scale in public sector innovation. *Innov J* (2017) 22(2):1–23. Available at: https://innovation.cc/document/2017-22-2-3-from-nasa-to-eu-the-evolution-of-the-trl-scale-in-public-sector-innovation/.

147. Ieropoulos IA, Singh A, Zertuche Moreno D, Greenman J. Bioelectrochemical systems and their readiness for commercialisation. *Curr Opin Electrochem* (2024) 46:101540. doi: 10.1016/j.coelec.2024.101540

148. He W, Dong Y, Li C, Han X, Liu G, Liu J, et al. Field tests of cubic-meter scale microbial electrochemical system in a municipal wastewater treatment plant. *Water Res* (2019) 155:372–80. doi: 10.1016/j.watres.2019.01.062

149. Blatter M, Delabays L, Furrer C, Huguenin G, Cachelin CP, Fischer F. Stretched 1000-L microbial fuel cell. *J Power Sources* (2021) 483:229130. doi: 10.1016/j.jpowsour.2020.229130

150. Mukherjee A, Patel V, Shah MT, Jadhav DA, Munshi NS, Chendake AD, et al. Effective power management system in stacked microbial fuel cells for onsite applications. *J Power Sources* (2022) 517:230684. doi: 10.1016/j.jpowsour.2021.230684

151. Heidrich ES, Dolfing J, Scott K, Edwards SR, Jones C, Curtis TP. Production of hydrogen from domestic wastewater in a pilot-scale microbial electrolysis cell. *Appl Microbiol Biotechnol* (2012) 97(15):6979–89. doi: 10.1007/s00253-012-4456-7

152. Cotterill SE, Dolfing J, Jones C, Curtis TP, Heidrich ES. Low temperature domestic wastewater treatment in a microbial electrolysis cell with 1 m2 anodes: towards system scale-up. *Fuel Cells* (2017) 17(5):584–92. doi: 10.1002/fuce.201700034

153. METzero – Waste to Resource (2025). Available at: https://metzero.co.uk/

154. Isabel San-Martín M, Mateos R, Carracedo B, Escapa A, Morán A. Pilot-scale bioelectrochemical system for simultaneous nitrogen and carbon removal in urban wastewater treatment plants. *J Biosci Bioeng* (2018) 126(6):758–63. doi: 10.1016/j.jbiosc.2018.06.008

155. Prado de Nicolás A, Berenguer R, Esteve-Núñez A. Evaluating bioelectrochemically-assisted constructed wetland (METland®) for treating wastewater: analysis of materials, performance and electroactive communities. *Chem Eng J* (2022) 440:135748. doi: 10.1016/j.cej.2022.135748

156. Yadav RK, Das S, Patil SA. Are integrated bioelectrochemical technologies feasible for wastewater management? *Trends Biotechnol* (2023) 41(4):484–96. doi: 10.1016/j.tibtech.2022.09.001

157. Roy M, Aryal N, Zhang Y, Patil SA, Pant D. Technological progress and readiness level of microbial electrosynthesis and electrofermentation for carbon dioxide and organic wastes valorization. *Curr Opin Green Sustain Chem* (2022) 35:100605. doi: 10.1016/j.cogsc.2022.100605