energies

Review

New Advances in Bioelectrochemical Systems in the Degradation of Polycyclic Aromatic Hydrocarbons: Source, Degradation Pathway, and Microbial Community

Yimeng Feng, Xuya Zhu, Xiulin Huang and Fengxiang Li *

Key Laboratory of Pollution Process and Environmental Criteria at Ministry of Education, China Tianjin Engineering Center of Environmental Diagnosis and Contamination Remediation, College of Environmental Science and Engineering, Nankai University, 38 Tongyan Road, Jinnan District, Tianjin 300350, China; 2120220769@mail.nankai.edu.cn (Y.F.); m15815251159@163.com (X.Z.); 2120220870@mail.nankai.edu.cn (X.H.)
* Correspondence: lifx@nankai.edu.cn

Abstract: Because of their high persistence, polycyclic aromatic hydrocarbons (PAHs) are found in a wide range of settings and pose a health risk to both humans and other organisms. Degradation of PAHs is an essential part of environmental management. By combining biological metabolism and electrochemical processes, bioelectrochemical systems (BESs) can degrade PAHs and provide important applications by converting the chemical energy of pollutants into electrical energy for energy conversion and recovery. This review provides a comprehensive introduction to PAH degradation by BESs, including PAH sources, degradation effects of BESs, performance enhancement methods, degradation pathways, and dominant microorganisms. By focusing on the relevant research in recent years, the main innovative research focuses on the optimization of the configuration, the electrode preparation, and the media additions to improve the removal performance of PAHs. It demonstrates the potential of BESs in the field of environmental remediation, especially their effectiveness in treating difficult-to-degrade pollutants such as PAHs, by concentrating on the application and mechanism of BESs in PAH degradation. This review is intended to provide the inexperienced reader with an insight into this research area and to point out directions for future research, especially in the design optimization of BESs and microbial community analysis.

Keywords: bioelectrochemical system; polycyclic aromatic hydrocarbons; pollution degradation; degradation pathway; microbial community

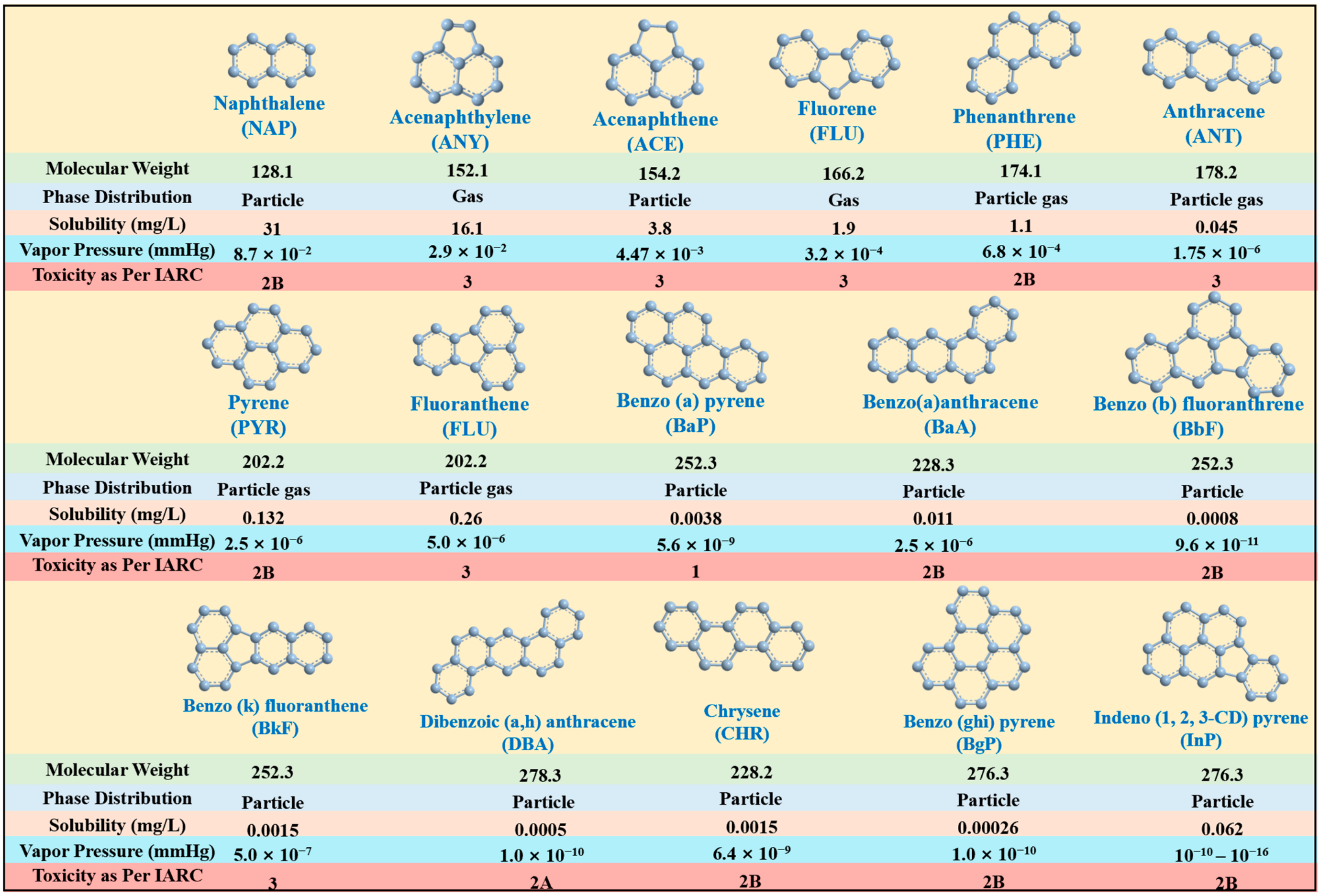
Academic Editor: Peter Foot

Received: 11 December 2024
Revised: 29 December 2024
Accepted: 2 January 2025
Published: 6 January 2025

Citation: Feng, Y.; Zhu, X.; Huang, X.; Li, F. New Advances in Bioelectrochemical Systems in the Degradation of Polycyclic Aromatic Hydrocarbons: Source, Degradation Pathway, and Microbial Community. *Energies* **2025**, *18*, 209. https://doi.org/10.3390/en18010209

1. Introduction

Polycyclic aromatic hydrocarbons (PAHs) are organic compounds composed of two or more benzene rings arranged in linear, angular, or cluster-like patterns [1]. To date, more than 200 PAH species have been found, some of these are highly toxic substances that can cause considerable damage to ecosystems and human health [2]. Based on frequency and toxicity, the USEPA identified 16 PAH species as priority control pollutants in 1999 (Figure 1) [3,4]. They are ubiquitous and harmful environmental pollutants produced mainly by industrial activities such as the burning of coal and hydrocarbon fuels and oil spills [5]. As the number of benzene rings increases, the solubility of PAHs decreases, hydrophobicity increases, and volatility decreases [6]. These compounds are chemically stable and persist in the environment due to their low water solubility and chelation of particulate matter deposited in soils and sediments [4,7]. The stable structure and properties of PAHs make them difficult to metabolize and degrade in the environment. The structure of PAHs also has carcinogenic teratogenic mutagenic effects on humans, and PAHs and their derivatives account for more than 30% of all carcinogens [7–9]. Moreover, PAHs are lipophilic, easy to accumulate in organisms through the food chain, and enter the human body through diet, skin, and the respiratory tract, increasing the risk of human ingestion due to bioaccumulation and biomagnification effects [10]. Foods that have been heated can also produce PAHs, which can then go through a number of chemical changes and become more dangerous [11]. Humans are often exposed to PAHs through breathing and food [12,13]. Recent studies have shown that human exposure to PAHs and metals can cause changes in human rDNA, resulting in systemic damage and inducing disturbances in cell signaling pathways, regulatory mechanisms, and cell functions [14,15]. In particular, the emitted metals and PAHs in the air easily mix to promote the development of lung inflammation, immune disorders, and cancer, as well as mosaic deletion of the Y chromosome [10]. Since sources of PAHs exist in various environments and slowly accumulate without interference to harm the environment and human beings, it is now necessary to study the degradation of PAHs.



	Naphthalene (NAP)	Acenaphthylene (ANY)	Acenaphthene (ACE)	Fluorene (FLU)	Phenanthrene (PHE)	Anthracene (ANT)
Molecular Weight	128.1	152.1	154.2	166.2	174.1	178.2
Phase Distribution	Particle	Gas	Particle	Gas	Particle gas	Particle gas
Solubility (mg/L)	31	16.1	3.8	1.9	1.1	0.045
Vapor Pressure (mmHg)	8.7×10^{-2}	2.9×10^{-2}	4.47×10^{-3}	3.2×10^{-4}	6.8×10^{-4}	1.75×10^{-6}
Toxicity as Per IARC	2B	3	3	3	2B	3

	Pyrene (PYR)	Fluoranthene (FLU)	Benzo (a) pyrene (BaP)	Benzo(a)anthracene (BaA)	Benzo (b) fluoranthrene (BbF)
Molecular Weight	202.2	202.2	252.3	228.3	252.3
Phase Distribution	Particle gas	Particle gas	Particle	Particle	Particle
Solubility (mg/L)	0.132	0.26	0.0038	0.011	0.0008
Vapor Pressure (mmHg)	2.5×10^{-6}	5.0×10^{-6}	5.6×10^{-9}	2.5×10^{-6}	9.6×10^{-11}
Toxicity as Per IARC	2B	3	1	2B	2B

	Benzo (k) fluoranthene (BkF)	Dibenzoic (a,h) anthracene (DBA)	Chrysene (CHR)	Benzo (ghi) pyrene (BgP)	Indeno (1, 2, 3-CD) pyrene (InP)
Molecular Weight	252.3	278.3	228.2	276.3	276.3
Phase Distribution	Particle	Particle	Particle	Particle	Particle
Solubility (mg/L)	0.0015	0.0005	0.0015	0.00026	0.062
Vapor Pressure (mmHg)	5.0×10^{-7}	1.0×10^{-10}	6.4×10^{-9}	1.0×10^{-10}	$10^{-10} – 10^{-16}$
Toxicity as Per IARC	3	2A	2B	2B	2B

Figure 1. Diagram of the main common PAHs and their structure, physical properties, and toxicity.

A comparison of common methods for PAH removal is shown in Table 1. PAHs are mainly removed by phytoremediation, adsorption, chemical oxidation, and microbial degradation, among which microbial degradation has the most applications and has become the main research direction for pollution control [8]. PAHs can be removed by adsorption through adsorbents such as biochar, organic minerals, modified clay, sand, sediment, cellulose, and membranes, but adsorbents are selective, capacity-limited, environmentally sensitive, and need to be replaced and regenerated when adsorption is saturated, which brings additional treatment costs [16]. Chemical oxidative removal has high efficiency and multi-selectivity of oxidizing agents but also has some limitations. The low availability of PAHs requires pre-treatment of the contaminated site, and the choice of oxidizing agent in different environments is critical and may require complex operational procedures and high costs, especially when multiple oxidizing agents and pre-treatment steps are involved [6,17,18]. Photodegradation is triggered by the direct absorption of light energy by PAHs, which excites electrons to produce unstable structures. For low molecular weight PAHs, photodegradation is more effective due to high bioavailability and long exposure time. However, the efficiency of photodegradation depends on the PAH structure, and some angular PAHs (e.g., PHE) are slow to degrade. The degradation capacity also decreases when PAHs are adsorbed by particles [19,20]. The former technologies have the defects of complicated technology, high cost, and serious secondary pollution, whereas biological remediation is safe and less polluting, and the cost is lower than other chemical remediation methods [21]. Biological remediation includes in situ (land tillage, biostimulation, biofortification, composting, and phytoremediation) and ex situ (bioreactor) approaches [22,23]. However, it is highly affected by weather conditions, has nutrient limitations, and requires pre-treatment to improve efficiency, and based on the complexity and diversity of the contamination on site, natural attenuation may outweigh the effects of biological enhancement [24,25]. The bioelectrochemical system (BES) has become a promising renewable energy process in recent years, which uses the electronic supply and recovery of microorganisms to achieve the degradation of pollutants and generate electricity at the same time, thus avoiding the traditional bioremediation defects [26,27]. These systems work through a microbial anode and a catalyst-coated cathode, combining biological treatment with electrochemical oxidation/reduction [28,29]. Anode microorganisms provide electrons to the anode by oxidizing organic material and transferring them to the cathode through an external circuit, forming a completely closed circuit [30]. It has been demonstrated that electroactive bacteria in the anode of the BES may produce electricity when exposed to PAH organic pollutants, and the capacity to produce electricity controls the production of bioelectricity [26]. Without adding additional chemicals or creating secondary pollutants, the BES offers an infinite supply of electron acceptors for the anaerobic breakdown of PAHs [25,31]. The electrons created by this process also speed up the metabolic activity of anaerobic bacteria [32,33]. The application of the BES has the following advantages: (1) it is applicable to a wide range of environments, relying on indigenous microorganisms enriched in the anode for degradation, which is more suitable for the local ecological niche and has a higher metabolic activity [31]; (2) the cathode is generally in contact with the air, with sufficient electron acceptors to ensure that the redox reaction continues to be carried out [34]; and (3) it does not require the incorporation of other energies, and it can also generate electrical energy for energy recovery through the degradation of pollutants [25,35]. Currently, the BES is widely studied for the removal of pollutants in water, soil, and sediment and has various applications in environmental management.

Table 1. Comparison of PAH degradation methods.

Methods	Efficiency	Costs	Ease of Operation	Environmental Impact	References
Adsorption	Middle	Middle	Low	Low (depending on adsorbent disposal)	[16,36]
Chemical oxidation	High	High	High	Possible byproducts	[37,38]

Table 1. *Cont.*

Methods	Efficiency	Costs	Ease of Operation	Environmental Impact	References
Photodegradation	High (Depending on the structure)	Low	Low	Possible byproducts	[19,20]
Biodegradation	Low	Low	Low	Low	[39,40]
Phytoremediation	Low	Low	Low	Low	[41,42]
BES degradation methods	High	Middle	Middle	Low	[25,43]

Many studies have used microbial fuel cells (MFCs) to degrade PAHs, and this method has achieved good degradation results in experiments [25,43,44]. In recent years, many studies have made improvements on the basis of traditional BESs, such as changing the anode, cathode, and configuration, in order to obtain better degradation or power generation effects [45–47]. In order to promote the further development of BES research and to explore the enhancement of the effect of BESs in treating PAH pollution, this review summarizes the recent progress and degradation effects of BESs in degrading PAHs. In this review, we analyze the PAH degradation systems of BESs in recent years; summarize the newly proposed configurations, electrode materials, and media additions to improve the degradation efficiency and power production performance; investigate the degradation pathways of common PAHs; and analyze the main dominant groups of anodes and their roles in the power production process.

2. The Fate of PAHs in the Environment

2.1. Common PAHs and Classifications

PAHs are structures containing multiple benzene rings composed of C and H. PAHs can be categorized into biphenyls, polyphenyl-substituted aliphatic hydrocarbons and thickened ring aromatic hydrocarbons based on the mode of connection. According to the molecular mass, they are categorized into low molecular weight PAHs (LMW-PAHs), medium molecular weight PAHs (MMW-PAHs), and high molecular weight (HMW-PAHs). The LMW-PAHs group contains 2- and 3-ring PAHs, the MMW-PAHs group contains 4-ring PAHs, and the HMW-PAHs group contains 5- and 6-ring PAHs [48,49].

In the linkage mode, the compounds formed by linking benzene rings with σ-bonds are called biphenyls. Biphenyl compounds are mainly used as solvents, heat transfer agents, and fruit mold inhibitors, and can also be used in organic synthesis, such as in the pesticide industry where it is widely used in the synthesis of various types of pesticides [50,51]. Compounds formed by several benzene rings instead of hydrogen atoms in aliphatic hydrocarbons are polybenzoic aliphatic hydrocarbons. Two or more benzene rings sharing two neighboring carbon atoms are thickened to form the thickened ring aromatic hydrocarbons, which are generally the most studied. Depending on their structure, thickened aromatic hydrocarbons can be divided into linearly arranged thickened aromatic hydrocarbons, angularly arranged thickened aromatic hydrocarbons, and thickened aromatic hydrocarbons with more complex structures [21]. Linear-thickened aromatic hydrocarbons can be used as organic semiconductor materials with important applications in molecular electronics and spintronics [52].

As the molecular mass of PAHs changes, their vapor pressure, water solubility, and degradability in the environment change. LMW-PAHs have a higher vapor pressure, are more volatile, mostly in the gas phase, and can travel long distances [53,54]. Although less toxic on their own, they can react with other pollutants to produce more toxic secondary pollutants [12]. NAP, ANT, and PHE are widely distributed throughout the environment, and their core structural backbones are present in many carcinogenic PAHs, so they are commonly used as signature compounds for detecting PAH contamination [54,55]. LMW-PAHs are more readily exposed to organisms, such as those detected in mollusks [56]. Through the detection of aquatic animals in the water polluted by PAHs, the aquatic animal organs were mainly LMW-PAHs [57]. The high assimilation efficiency and low lipophilicity of these LMW-PAHs suggest that they are degraded in biota and can be readily adsorbed into the lower respiratory system and cleared from the body, thus this type of PAH may cause acute toxicity but have low carcinogenicity [13,58,59]. Compared with LMW-PAHs, MMW-PAHs and HMW-PAHs are more difficult to degrade due to their complex molecular structure and are hard to oxidize with poor water solubility [60,61]. HMW-PAHs are non-volatile, mainly in the particulate phase, highly lipophilic, can stay in the body for a longer period of time, and their toxicity is relatively low, but they have carcinogenic, mutagenic, or teratogenic properties [12,54,62,63]. In addition, the prominence of HMW-PAHs may be related to emissions from high-temperature activities [58].

2.2. The Sources of PAHs in the Environment

PAHs are ubiquitous in the environment and originate from either natural or anthropogenic sources (Figure 2a) [64]. PAHs originate from the pyrolysis of carbonaceous materials; biochemical processes, natural fires, and volcanic eruptions are natural sources of PAHs [65]. Natural sources constitute a small environmental background value for PAHs and do not pose a major threat to the environment [66]. Various areas of human life, such as biomass fuels, coal, cooking, smoking, household waste, and gasoline and diesel vehicle exhaust, can be sources of combustion emissions of PAHs [49,67]. With the increase in human industrial activities, transportation emissions, and the burning of biomass, liquefied petroleum and natural gas are major sources of PAH pollution in these substrates [58,68]. Factory wastewater discharge also contains PAHs, such as textile printing and dyeing wastewater treatment sludge, with the dyes used in the production process being discharged PAHs [3]. The pollution caused by automobile emissions is mainly in the city, and the biomass combustion is mainly in the suburbs [69,70]. Outdoor PAH contamination is transmitted indoors and is the main source of indoor PAH contamination compared to cooking and smoking [71,72].

The main PAH pollutants generated differ by fuel type and combustion conditions [73]. PAHs associated with oil pollution are dominated by alkylated PAHs, while PAHs derived from incomplete combustion of fossil fuels are dominated by parent PAHs [74]. LMW-PAHs are products of incomplete biomass, for example, ACY and FLU are labeled compounds of wood combustion and fossil fuels. Fossil fuel combustion and coking processes are the main sources of HMW-PAHs, coal combustion emissions are associated with 4-ring PAHs, automobile tailpipe emissions are 5-ring and 6-ring PAHs, and the largest share of total PAHs emitted from diesel combustion is 3-ring PAHs [49,54,55,59,75]. The generation of PAHs increases and then decreases with increasing temperature, with 4-ring PAHs dominating at 600 °C, 2,3-ring PAHs dominating at 800 °C, and 5,6-ring PAHs being the highest at high temperatures [55,76]. As a result, PAHs can be identified from biomass, coal, fuel, and other pollutants using molecular diagnostic ratios. The different combustion sources of PAHs determine their obvious seasonal fluctuations. In the cold season, pollutant emissions such as CHR, BkF, BaP, IcP, and BgP are mainly generated due to the combustion of coal and biomass and the low temperatures cause PAHs to adsorb on particulate matter [77,78]. Solar radiation and high temperatures in the warm season, on the other hand, accelerate the reaction rate and enhance the evaporation and degradation of LMW-PAHs [59]. PAH pollution sources are diverse, originating from the inadequate combustion of substances, involving all aspects of clothing, food, housing, and transport, and have obvious seasonal fluctuations. The identification of the main sources of different types of PAH pollution is also an important means of preventing PAH emissions.

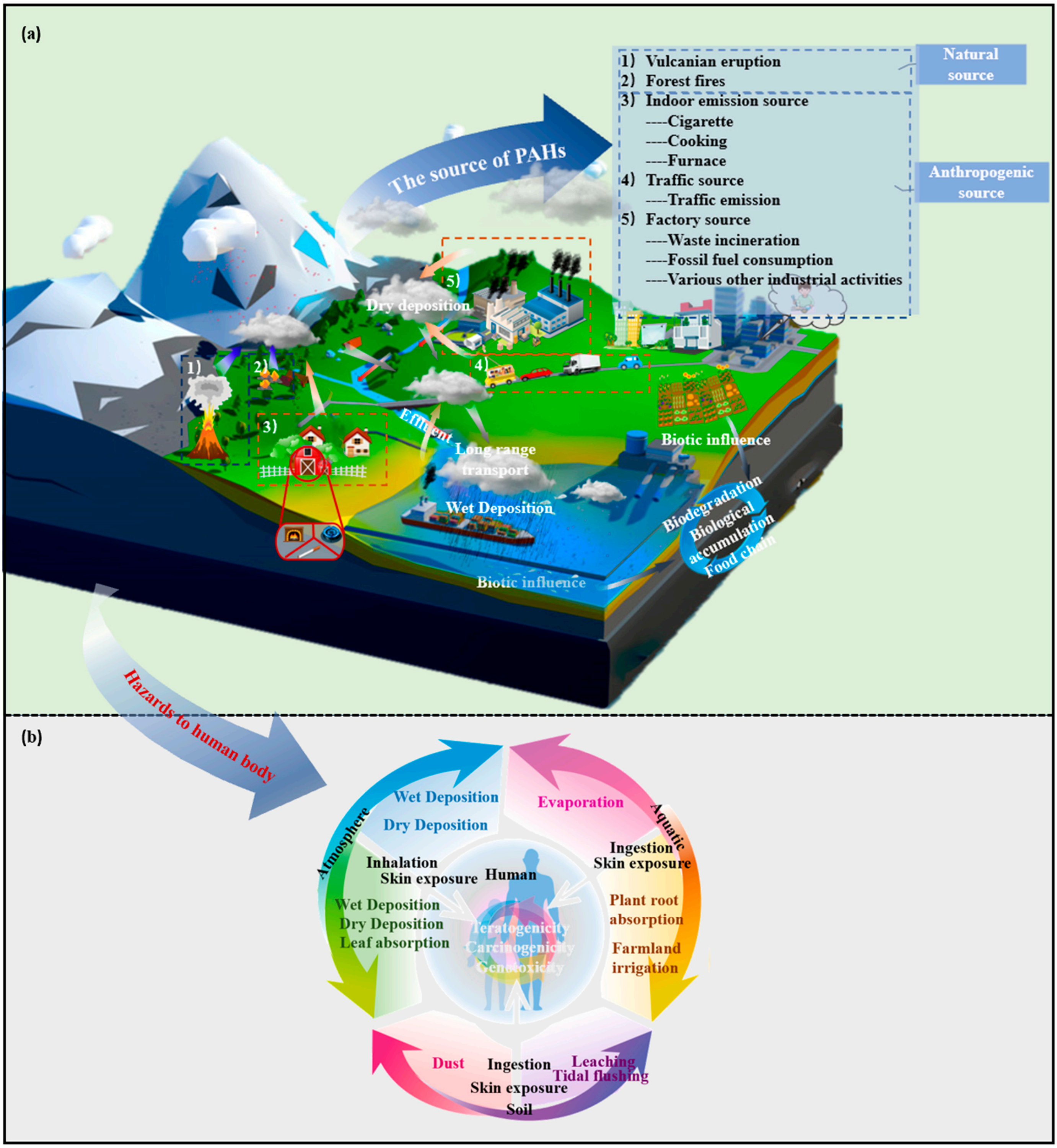


Figure 2. Schematic distribution of PAHs in the environment. (**a**) Sources of PAHs in the environment; (**b**) hazards of PAHs to humans.

2.3. PAHs in the Environment

The spatial distributions of different PAHs in the environment are different. Because of the different steam pressure and air–surface exchange capacities, LMW-PAHs are mainly volatilized, and HMW-PAHs are mainly deposited in all seasons. MMW-PAHs are more affected by soil–air transfer, showing a tendency to accumulate in soils in cold regions/seasons and to be re-volatilized into the atmosphere in warm regions/seasons [79]. Influenced by the physicochemical properties of water bodies, meteorological changes, and soil and sediment properties, PAHs can undergo closed-loop transport transformations in different environmental media (Figure 2b) [64]. The clarification of the source profile and distribution of PAHs in the environment is essential for the next step of ecological risk assessment and the implementation of degradation control technologies.

The source of PAHs in aquatic environments can be direct inflow from surface water runoff or atmospheric deposition. Due to changes in temperature and seasons, PAHs are exchanged at the air–water interface [80]. 4,5-PAHs dominate in rivers and HMW-PAHs are poorly water-soluble; these sediments become sinks for PAHs in aquatic ecosystems, and when they enter river systems they are adsorbed onto particulate matter and immobilized in surface sediments and are affected by the physicochemical properties and nutrient concentrations of the media in the sediment, which may recontaminate the water body [62,68,81,82]. PAH concentrations in sediments are 2 to 4 orders of magnitude higher than in water [37,81]. PAHs in aquatic environments can also be transported over long distances and are ingested and enriched by organisms, entering the food chain and further jeopardizing human health. Numerous processes, such as wave action, tides, and bioturbation, that alter the mixing of water exchange during changes in external factors can affect the bioavailability as well as the transport of PAHs [83]. In addition, PAHs accumulate in aquatic organisms regardless of their location in the water and feeding habits [84].

PAHs that are ubiquitous in the atmosphere mainly come from various human activities, with petroleum volatilization and coal combustion being the main sources of PAHs in particulate matter during non-heating and heating periods, respectively [85]. Indoor PAHs in the air may come from smoking, cooking, incense burning, etc. [58]. PAHs generated by combustion are first released into the atmosphere with the rise of the hot airflow from the flame and adsorbed on fine particulate matter, which then enters the water body or soil under the effect of rainfall washout and atmospheric deposition [63,86]. Due to the semi-volatile nature of PAHs, another source in the atmosphere is the evaporation of PAHs from soil and water into the air [10,69]. A significant correlation was also found between the amount of dust in the air and the concentration of PAHs in the particulate phase [12]. PAHs in the air can attach to dust and other particles; PAHs carried by fine particles undergo a long-range transport to remote areas while those adsorbed on coarse particles have a shorter lifetime in ambient air [58,87]. These small particle sizes can be exposed to the human body through nasal inhalation, skin contact, and accidental oral ingestion [4,88]. PAHs with higher molecular weights and generated from a mobile source tend to be bound with finer particles and can deeply enter the lungs, posing severe health risks to humans [49,55]. In addition, the combustion of PAHs produces symbiotic pollutants such as environmentally persistent free radicals, which produce reactive oxygen species (ROS) oxidation that can increase cardiovascular and respiratory hazards, and inhalation of PAHs into the human body can cause cases of lung cancer, which is closely related to human health [67,85].

Soil is considered to be the ultimate sink for PAHs due to their hydrophobic nature, which makes it easier to bind to soil particles. The origins of PAH soil contamination include natural and anthropogenic sources, such as combustion, atmospheric sedimentation, and industrial wastewater discharge [58]. PAHs can be retained in soil for long periods of time due to the occlusion of soil particle aggregates and the ease of adsorption of PAHs [89]. Soil organic matter and total nitrogen content have significant effects on PAHs, and the distribution of 2- to 3-ring as well as some 4-ring PAHs is more likely to be affected by soil organic matter, while 4- to 6-ring PAHs are more likely to be affected by environmental factors, such as oxygen reduction potential and heavy metals [90,91]. Biomass, coal, gasoline, and diesel combustion are the main sources of soil PAHs, with the 4- and 5-ring groups as the main components [92]. In the cultivation of vegetables and fruits, transfer of PAHs can occur from air and soil. PAHs from food are potentially toxic compounds via ingestion. In addition, the presence of heavy metals in soil can create a compound contamination with PAHs, increasing the risk of human carcinogenicity and posing an ongoing public health hazard [93]. PAHs have complex environmental behavior as they can migrate in water, air, and soil media, and have lipophilic hydrophobicity, enabling the particles to easily adsorb into the soil or sediment. Due to their persistence in the environment, PAHs eventually enter the food chain through a variety of pathways, causing cumulative harm to the health of organisms.

3. BES Degradation of PAH Applications

3.1. Mechanism of Pollutant Degradation by BESs

The BES, a process in which microorganisms act as catalysts to convert chemical energy into electrical energy, has advantages over microbial degradation methods [94]. The BES creates anaerobic conditions at the anode and utilizes electricity-producing microorganisms to decompose organic matter [95,96]. Firstly, large organic molecules are decomposed into small molecules under the action of degrading bacteria, and then further utilized by exoelectrogenic bacteria on the surface of the anode, which completely degrade the organic matter into harmless substances, such as CO_2 and H_2O, etc. [97,98]. The oxidation of the organic matter at the anode produces electrons, which are transferred to the electrodes directly and indirectly through the direct and indirect transmission of electrons, and H^+ and the electrons are transferred to the cathode [33,94,99,100]. Thus, the problem of insufficient electron acceptors for microbial degradation is solved [43]. According to the presence or absence of applied voltage, the BES can be categorized into a microbial fuel cell (MFC) or microbial electrolytic cell (MEC) system (Figure 3). When there is no applied voltage, the electrons transferred to the cathode react with O_2 to form H_2O under the action of the catalyst [101]. And an applied voltage is applied between the cathode and anode as an MEC; the applied current gives the cathode a more negative potential, which can recover energy in the form of H_2 or CH_4 at the cathode [28,102–104].

In conclusion, the MFC uses O_2 to produce electricity for the electron acceptors in the BES cathode chamber, where electrons from the anode combine with electron acceptors in the presence of a catalyst, while the MEC is powered by an external voltage to increase the cathode's spontaneity and produce chemical products like H_2 [105]. The mutual cooperation of degradative and electro-producing bacteria in the reactor's anode, as well as the efficient electron transfer process, allow the BES to degrade pollutants while recovering energy to achieve the goal of green degradation and increase the system's degradation capacity [106–108]. Furthermore, the BES can be applied to different environments, with low requirements for temperature and pH, and can degrade pollutants in soil, sediment, and sewage (Table 2) [109]. How to increase the power generation and degradation performance in the degradation process is what has been investigated in this study. In the whole electrochemical system, the electrode materials, configuration, electrolyte solution, and loaded microorganisms are extremely important to the system's performance. The following discussion focuses on the innovations in these aspects.

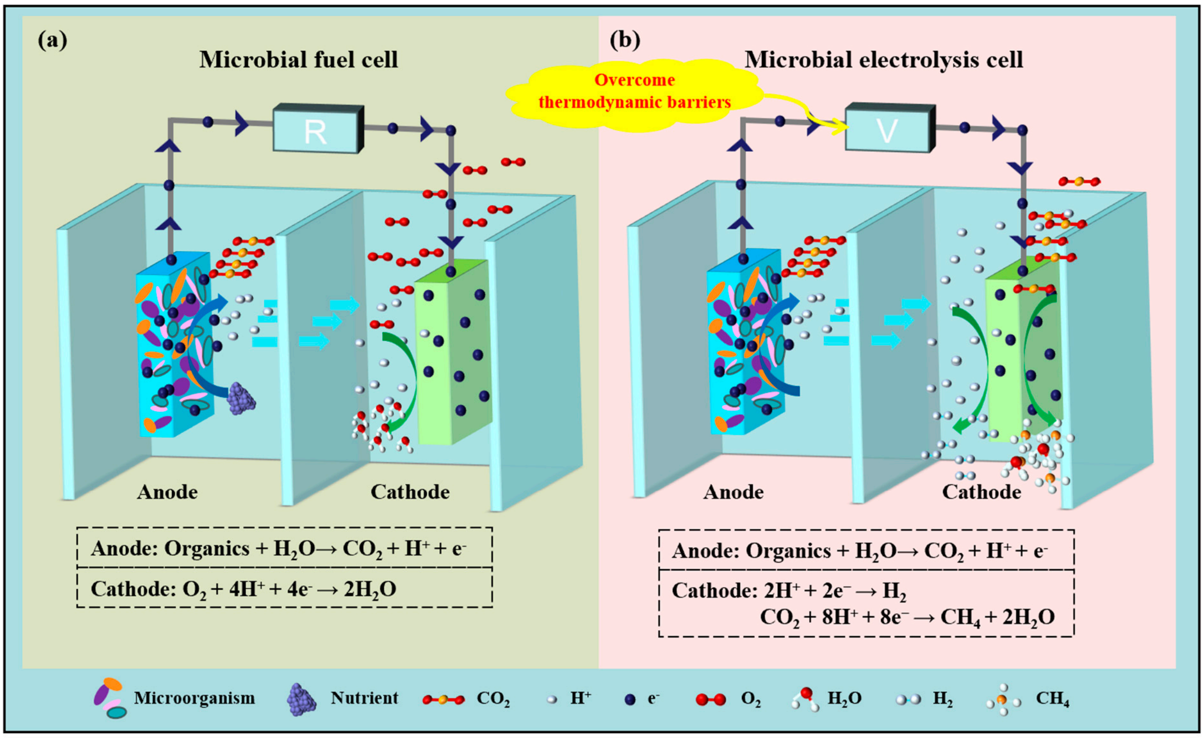


Figure 3. Schematic diagram of the BES power generation mechanism and pollutant degradation process. (**a**) Schematic diagram of the MFC system; (**b**) schematic diagram of the MEC system.

Table 2. Summary of PAH degradation by BESs in recent years.

BES	Anode	Cathode	PAHs	Electrical Performance	PAH Degradation Effect	References
Water/double chamber MFC	Graphite	Graphite *P. candolleana*	NAP, ACE, FLU, ANT	Current density: 525 ± 20 mA/m^2 Power density: 147 ± 10 mW/m^2	NAP: $25 \pm 3.0\%$ ACE: $28 \pm 2.5\%$ FLU: $31 \pm 3.6\%$ ANT: $62 \pm 1.1\%$	[45]
Water/double chamber MFC	Carbon cloth (PHE)	Carbon cloth loaded with Pt/C (Platinum/Carbon)	PHE	Current density: 304 mA/m^2 Power density: 37 mW/m^2	Fully degradable in 50 days	[110]
Water MFC	Carbon cloth	Carbon cloth loaded with Pt/C	NAP PHE PYR NAP, PHE NAP, PYR PHE, PYR	-	NAP: 97.6% PHE: 42.9% PYR:22.0% PHE: 89.2% PHE: 50.0% PHE: 87.0% PYR: 45.4%	[111]
Water MFC	Carbon fiber	Pt/C-coated carbon cloth	PHE (5 mg/L) PHE (10 mg/L) PHE (20 mg/L)	Power density: 737 mW/m^2 Power density: 4567 mW/m^2 Power density: 357 mW/m^2	PHE: 97.1% PHE: 94.9% PHE: 98.4%	[112]

Table 2. *Cont.*

BES	Anode	Cathode	PAHs	Electrical Performance	PAH Degradation Effect	References
Water MFC	Carbon cloth	Carbon cloth	PYR (5 mg/L)	Power density: 175 mW/m^2	44.9%	[113]
			PYR (20 mg/L)	Power density: 149 mW/m^2	72.5%	
			PYR (30 mg/L)	Power density: 96 mW/m^2	88.1%	
Water MFC	Nickel-plated sponge (*S. yanoikuyae* XLDN2-5)	Carbon paper	Carbazole	Power density: 497 mW/m^2	Degradation of 64.9% in 24 h, 100% in 144 h	[46]
Water MFC	Carbon fiber brush	Carbon cloth loaded with CuCo@NC	PHE	Power density: 3249 mW/m^2	PHE: 98.8%	[114]
Water MFC	Carbon cloth	Carbon cloth loaded with Pt/C	PYR	Power density: 96 mW/m^2	PYR: 88.1%	[113]
Up-flow MFC	Carbon brush	Activated carbon, carbon black, and polyvinylidene fluoride	Produced water	Power density: 227 mW/m^2	TPH: 89.0%	[47]
Soil MFC	Graphite felt	Graphite felt loaded with Pt/C	BaP	Voltage: 121.5 mV	BaP: 52.5%	[115]
Soil MFC	Activated carbon fiber	Activated carbon fiber	ANT, PHE	Power density: 12 mW/m^2	ANT: 54.2% PHE: 42.6%	[116]
Soil MFC	Carbon felt	Stainless steel mesh	PHE (defective hematite)	Power density: 325 mW/m^2	PHE: 61.4%	[117]
			PHE (hematite)	Power density: 306 mW/m^2	PHE: 53.94%	
Sediment MFC (aerobic)	Graphite plate	Graphite plate	NAP, ACE, PHE	Power density: 6 mW/m^2	NAP: 41.7% ACE: 31.4% PHE: 36.2%	[65]
Sediment MFC (anaerobic)	Graphite plate	Graphite plate		Power density: 3 mW/m^2	NAP: 76.9% ACE: 52.5% PHE: 36.8%	
Sediment MFC	Graphite nitrocellulose	Graphite	PHE, FLT, PYR, BaA, CHR, BaP	Maximum voltage: 254 mV	PAHs: 71.6%,	[118]
Sediment MFC	Carbon felt	Carbon felt	NAP, PHE, PYR (2 g/kg^{-1} humus)	Current: 0.675–0.870 mA	NAP: 53.0% PHE: 36.1% PYR: 34.4%	[119]
			NAP, PHE, PYR	Current: 0.625–0.695 mA	NAP: 46.9% PHE: 26.5% PYR: 22.0%	
Water/double chamber BES	Iron sheet	Carbon felt	PAHs	-	Anode: $\sum$16PAHs 64.7% Cathode: $\sum$16PAHs 16.1%	[3]
Water MEC	Carbon brush (10 mg/L NaCl)	Carbon brush	NAP	-	Fully degradable in 5 days	[120]
Water MEC	Titanium mesh-wrapped carbon cloth	Titanium mesh-wrapped carbon cloth	PHE	-	Total PHE degradation in 18 h	[121]

Table 2. *Cont.*

BES	Anode	Cathode	PAHs	Electrical Performance	PAH Degradation Effect	References
Soil MEC	Graphite Acetic acid	Graphite	TPH	-	TPH: 69.2%	[122]
	Graphite Sewage	Graphite	TPH	-	TPH: 73.3%	
Soil MEC	Carbon cloth (0% BC)	Carbon cloth	PHE, BP	-	PHE: 31.5% BP: 27.6%	[123]
	Carbon cloth (1% BC)	Carbon cloth			PHE: 44.9% BP: 42.7%	
	Carbon cloth (3% BC)	Carbon cloth			PHE: 53.4% BP: 54.1%	
	Carbon cloth (6% BC)	Carbon cloth			PHE: 61.6% BP: 71.7%	
	Carbon cloth (6% BC)	Carbon cloth			PHE: 73.9% BP: 75.7%	

3.2. Innovations in BES Configuration

Common BESs for degrading PAHs are single-compartment, with the cathode located in the upper layers of water or soil in contact with oxygen and the anode located in anoxic sediments or deep soils, with the use of wires and resistors between the electrodes (Figure 4a). Moreover, within a certain distance range, the ohmic loss of MFCs with similar distances is also smaller, and the power generation of MFCs and the removal rate of PAHs increase with decreasing electrode spacing [116]. For PAH removal from sediments, Cao et al. used a single-chamber membrane-free glass unit for PAH removal. The maximum removal of NAP, PHE, and PYR was 45.2%, 25.9%, and 21.9%, which were 40.37%, 56.97%, and 50% higher than that of the open-circuit group, respectively, over 56 days, suggesting that electrical stimulation increased the anodic electrical activity microbial activity and more effectively improved the removal efficiency of PAHs in sediments (Figure 4c) [119]. To further encourage the decomposition of PAHs, a voltage is applied between the cathode and anode in the BES, and water electrolysis encourages local oxidation of the anode. Yan et al. used horizontally placed graphite plates as electrodes with an external 2 V power supply; the addition of the power supply increased the redox potential around the anode, and an increase in PAH degradation genes in the vicinity of the anode was examined to enhance PHE degradation. In the sediment below the anode (0–1 cm range), the concentration of PHE was reduced by 50% within 70 days, while the level of phenanthrene remained unchanged in the system designed for the addition of the BES, which enhanced PHE degradation at a low cost (Figure 4b) [124]. In addition to this study, total TPHs in the extracted water could also be degraded up to 64.5% in one cycle with a 2 V addition [122].

Further incorporation of other processes in the BES can enhance the degradation of PAHs. Cabrera et al. connected a peristaltic pump into an MFC system to design a novel continuous up-flow MFC to treat extracted water, and the peristaltic pump was set to influence the internal resistance by controlling the wastewater flow rate. It was demonstrated that the flow direction could promote the proton flow at a suitable flow rate. At a flow rate of 0.4 mL/s, the resistance was reduced to 20 Ω and 89% of total petroleum hydrocarbons (TPHs) could be removed in three days, but at too high a flow rate (0.6 mL/s), the internal resistance increased due to the disruption of biofilm (Figure 4d) [47]. By introducing plants into the BES, plant inter-root secretions at the anode can serve as electron donors, and the addition of plants can optimize the structure of microbial organic metabolism at the anode and promote the transformation of organic pollutants. A plant

MFC adds plants to the reactor compared to an ordinary MFC, and the addition of a plant MFC increases the crude oil degradation rate at the anode by 58.6% relative to natural degradation (Figure 4e) [125]. Also, peroxymonosulfate (PMS) is currently frequently utilized in advanced oxidation processes (AOPs) because of its broad pH range of reaction. AOPs also have a high effectiveness in removing organic pollutants. Bai et al. combined PMS activation with MFCs for PAH degradation by sacrificing ions released from the electrolysis of the Fe electrode to activate the anode's PMS. The generated Fe^{3+} diffuses to the cathode under the action of the electric field force and receives circuit electrons at the cathode to regenerate Fe^{2+} to continue to be used for PMS activation and enhancement for the removal of PAHs from the anode and cathode (Figure 4f). The Fe^{3+} reduction and oxidation are recycled and used for PMS activation many times, which abbreviates the steps and cost. And in the context of this study, the optimal conditions for the operation of this MFC were explored by varying the applied voltage, PMS dose, and pH at an applied voltage of 7 V and pH 3. At this time, the activation of the free radicals produced by the PMS reaction, especially the ·OH, had the highest concentration and the best degradation effect on PAHs, along with the removal of PAHs at both the anode and the cathode. Both the anode and cathode can reach a degradation rate of 64.7% and 16.1% [3]. By increasing the flow rate to lower the internal resistance, adding plants to encourage degradation through inter-root secretion, and activating PMS to improve pollutant removal efficiency, the BES works in combination with other processes to effectively remove PAHs. Through creative integration of the BES with other technologies, PAHs can be more efficiently degraded.

3.3. Innovations in BES Electrodes

The cathode controls the rate of the reduction process, whereas the MFC anode controls the microbial loading. There is a need to create low-cost cathodic catalytic materials with strong and consistent catalytic activity because the cathode frequently requires an expensive Pt/C electrode. Huang et al. prepared Cu, Co bimetallic embedded N microporous carbon (CuCo@NC) by a simple in situ growth method and pyrolysis (Figure 5a) [114]. Bimetallic doping with N effectively increases the electrically active sites of the cathode catalyst, and the presence of Cu increases the N content in the framework due to Cu-N binding [126,127]. When the ratio of Cu to Co was 1.5, as can be seen from the CV curves, the CuCo@NC-MFC anode had greater microbial activity (Figure 5b), the power density of the modified cathode increased significantly to 3248.68 ± 28.21 mW/m^2 compared with the control (cathode of Pt/C electrode) of 1220.71 ± 21.15 mW/m^2. The degradation rate of the modified cathode was as high as 98.37% for 10 mg/L PHE, which was not much different from that of the control at 95.30%, but after PHE degradation the anode biofilm activity was changed to 95.25% of the previous performance compared with 73.33% of the control. However, the anodic biofilm activity changed after PHE degradation, and when the MFC performance was re-measured, the power density of the CuCo@NC MFC was 95.25% of that of the previous one, while that of the control group was only 73.33% [114]. In addition, biological enzymes can act as electron acceptors at the reducing cathode [128]. Since ligninolytic enzymes such as lignin peroxidase (LiP), manganese peroxidase (MnP), and laccase are naturally secreted by stramenophora fungi, they can be applied in the cathode, where MnP also plays an important role in the oxidative degradation of PAHs, and can degrade some pollutants while acting as a catalyst (Figure 5c) [129–131]. Fungal incorporation improved the power production of MFCs (Figure 5d). Through experiments, the strain *Psathyrella candolleana* was screened for the highest enzyme activity among several fungi. Wastewater plant effluent was added directly to the anode to cultivate the natural strain, which was added to the cathode and evaluated at the cathode for biodegradation of NAP, ACE, FIU, and ANT. Ligninolytic enzymes produced by *Psathyrella candolleana* can effectively degrade

PAHs. The fungal enzyme activity can be increased by adding PAHs to the cathode, which helps to enhance bioelectric power generation in the cathode chamber [45]. In addition to the production of enzymes at the biocathode, the fungal mycelium at the electrode also acts as an oxygen barrier, further ensuring anaerobic conditions at the anode [129].

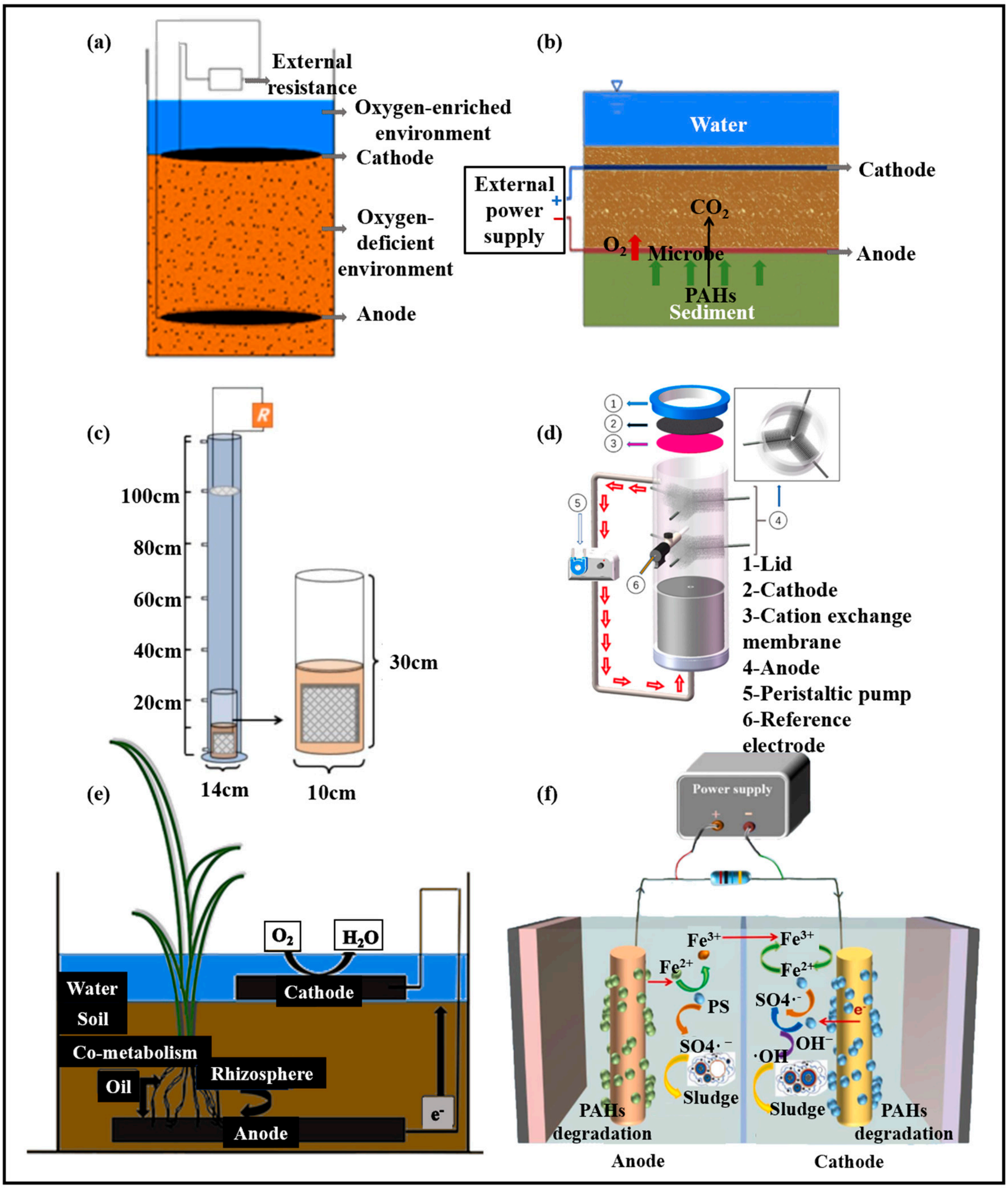


Figure 4. Schematic diagram of the BES application construction. The top structure of (**a**) MFC [116] and (**b**) MEC [124]; the cathode is located in the upper layer in contact with air and the anode is located in an anaerobic environment such as soil or sediment. (**c**) Single-compartment filmless glass MFC structure, with a cathode and anode distance up to about 1 m [119]; (**d**) new continuous up-flow MFC structure, with the MFC flow rate controlled by the addition of a peristaltic pump [47]; (**e**) plant MFC structure with plant roots at the anode [125]; and (**f**) PMS-MFC structure to promote cathodic and anodic PAH degradation via Fe^{2+} and Fe^{3+} redox [3].

Anode modification is used to boost power generation and pollution removal by increasing the electrode's biomass content and electrochemically active surface area. It also improves the extracellular electron transfer process. To increase the anode's surface area and electrical conductivity, anode modification can be accomplished by adding excessive amounts of metal, nanomaterials, etc. Graphene oxide (GO) has a larger electrochemically active surface area and biomass content than graphene (GR), carbon nanotubes (CNTs), and graphite felts (GFs) during the anode modification probe. During stable operation (day 50 to day 110), the GO-SMFCs produced electrical energy of 0.98 ± 0.14 kJ, which was 12.6%, 71.9%, and 98.0% higher than that of GR, CNT, and GF, respectively. However, the removal of pollutants near the anode did not correlate with electrical energy generation, and the removal of PHE in the region close to the anode was best with CNT-SMFCs at 78.1% and pyrene at 69.6% with GO-SMFCs. The total amount of electroactive bacteria was the key factor affecting the amount of power production when biofilm covered the anode, and carbon nanomaterials usually promoted thicker biofilm formation and stronger bacterial interactions due to their nanostructures and high surface roughness. GO anodes had the highest voltage output during the steady state due to hydrophilic groups promoting biofilm development and strong interactions, with more proteins compared to the GR and CNT anodes [132]. In an SMFC with activated carbon fiber mats as the anode, the reported removal rates of PHE and PYR were 42.6 ± 1.9% and 27.0 ± 2.1%, respectively, after 175 days, which were lower than those in this study [116]. The anode material also has certain adsorption properties, which reduce mass transfer resistance and promote microbial degradation through strong adsorption of PAHs from the surrounding environment [133]. Sharma et al. enriched the microbial population density by directly loading PHE on the anode to promote the microbial community to make direct contact with the degradants and achieved a power production capacity of 37 $\mathrm{mW/m^2}$ at low PHE loading (2 $\mathrm{mg/cm^2}$), and the absorption peaks of PHE at the anode after the reaction were significantly reduced after observational tests (Figure 5e,f) [110]. In addition to this, the enriched biofilm anode was more stable than the freshly inoculated anode and the anode showed better pollutant removal in the MFC after one year of incubation. This is mainly due to the interactions between microorganisms in the anode biofilm and the interspecies electron transfer mechanism, which can provide long-term selective enrichment of the anode electrode to improve the degradation of hydrocarbon pollutants [134]. Pollutant degradation, microbial enrichment, and system cost are all improved when electrode conductivity is increased through electrode modification. In particular, by using microorganisms rather than costly cathode metal catalysts and addressing the issue of catalyst poisoning, the suggested biocathode technique expands the research possibilities of BES electrodes.

3.4. Innovations in Media Additions

3.4.1. Carbon Source Addition

The challenges in applying the BES degradation of PAHs are low mass transfer, low bacterial activity, and low utilization of pollutants because the SMFC cathode is inserted on the surface of anaerobic sediment or soil layer and the anode is located deep in the sediment or soil, causing electrons or protons to receive a large mass transfer resistance during the transfer process. Additionally, the range of extracellular electron transfer is limited to the pollutant degradation area in the vicinity of the anode of a few tens of micrometers. In this situation, the addition of a mediator is required to raise the effective range of electron transfer, lower the mass transfer resistance, and enhance the degradation effect of PAHs (Figure 6a) [135]. Numerous studies have been conducted to enhance PAH degradation by adding organic substances to the environment, and when organic acids are added as co-substrates, higher levels of dissolved organic matter can promote the release of PAHs from the environment and increase PAH bioavailability [136]. Dissolved organic matter interacts with PAHs through hydrogen bonding, charge transfer, covalent bonding, and van der Waals forces to increase solubility. The addition of low molecular weight organic acids significantly promotes bacterial degradation of HMW PAHs BaP and PYR and enhances bioremediation [137]. In the BES, the organic substances in acetic acid and sewage can be utilized by electrochemically active bacteria, which decompose the organic substances near the anode and release electrons, which can be transferred to the anode, thus increasing the current and power density. The use of acetic acid and sewage as bioelectrical stimulants resulted in a power density nearly 15% higher than that of the no-stimulant control group, and the removal of total TPHs was 67.4% and71.8%, which is 64.5% higher than the control group [122]. Starch was added to the MFC and organic acids were produced from the starch decomposition; as the starch concentration was increased to 10 mg/g, the concentration of released organic acids increased, and the degradation rates of NAP, ACE, and PYR all increased continuously, reaching 69.9%, 55.6%, and 46.8%. However, the substrate could not present a good effect when it was added in excess when the starch concentration was too much, the environment was acidic, the microbial community changed, the *Firmicutes* increased, the *Bacteroides* and *Proteobacteria* decreased, and the power generation and the degradation rate decreased [138]. When glucose is used as a substrate for co-metabolism, 500 mg/kg of glucose is the optimal amount to add, and 1500 and 1000 mg/kg of glucose results in a decrease in the degradation rate; moreover, excessive co-substrate content leads to microbial degradation of glucose to satisfy their daily metabolic needs and reduces the utilization of PAHs [125]. In addition to this, organic pollutants such as PAHs can also be used directly as a carbon source. Zafar et al. added diesel fuel as a carbon source in an MFC, which promotes the enrichment of electrochemically active bacteria, enhances electron transfer, and promotes power generation. An amount of 50 mg/L of diesel fuel was added to maximize the efficiency of power generation and the rate of degradation, and it was observed that at this time, the anode had a thick and dense microbial film, but the high diesel fuel concentration rather inhibited bacterial activity [139]. Aleman-Gama et al. added 20.8 mg/cm^3 of kerosene to a sediment MFC and found that the internal resistance of the battery could be reduced by 28 times, and the power output increased by 47 times. Although gasoline is more difficult to biodegrade than kerosene, the optimal conditions (0.8 M Na_2SO_4, pH = 2) do not require the addition of additional growth media, and the biodegradation activity and power output of the system will increase [140]. By strengthening the anode microbial population and improving the conductive ability of the medium, the addition of carbon sources (such as acetic acid, starch, glucose, etc.) can increase the power production efficiency of the BES and the degradation efficiency of PAHs (Figure 6b). However, it is important to note that the process requires the right amount of these organics, and an excessive addition will decrease the bioavailability of PAHs.

3.4.2. Surfactant Addition

Electrokinetic remediation is not very effective for the degradation of hydrophobic PAHs, which are frequently adsorbed in soil and sediment [124]. The use of surfactants can desorb PAHs from soil and increase bioavailability [141]. Chemical surfactants are usually toxic to bacteria and prevent biofilm formation, but the addition of biosurfactants can be a good solution to this problem [125,142]. The biosurfactant rhamnolipid contains both hydrophobic and hydrophilic groups, which can enhance the bioavailability of PAHs and change the hydrophilicity of the anode, making it easier for microorganisms to gather at the anode for PAH degradation and, at the same time, act as an electron shuttling molecule to enhance the electron transfer and improve the microbial repair ability (Figure 6d) [143,144]. β-cyclodextrin and exogenous biomass humic acid (HA) addition all reduced the charge transfer resistance and improved the electron transfer efficiency in MFCs. The addition of HA electronic media increased the removal efficiencies of NAP, PHE, and PYR by 13.0%, 36.2%, and 56.4%, respectively. Compared with the control, the removal efficiencies of NAP, PHE, and PYR were higher than the control by 45.9%, 97.8%, and 85.6% in the middle and distant regions (≥3 cm). The increase in removal efficiency was significantly higher in the central and distant regions (≥3 cm) than in the proximal region of the anode (1 cm). This was attributed to the enhancement of bacterial electron transfer capacity by the addition of HAs [145]. HAs enhance electron transfer through the redox activity of its oxygen-containing functional groups, especially quinone groups, which cycle between hydroquinone and quinone to form an electron transfer network, and the multiple redox functional groups in HAs act synergistically to cover a wide range of redox potentials, further mediating electron transfer [119]. Although β-cyclodextrin is not a biosurfactant, as a biochemical product, it is not toxic to bacteria. When 5.0 mmol/kg β-cyclodextrin was added, PHE and PYR removal efficiencies of 54.2% and 48.4% were achieved, which were improved by 96.4% and 101.7% compared to the blank [125]. The bioavailability of PAHs is greatly increased by the addition of surfactants, which also serve as an electron shuttle to improve electron transfer and bioremediation, hence increasing the BES's efficiency in degrading PAHs.

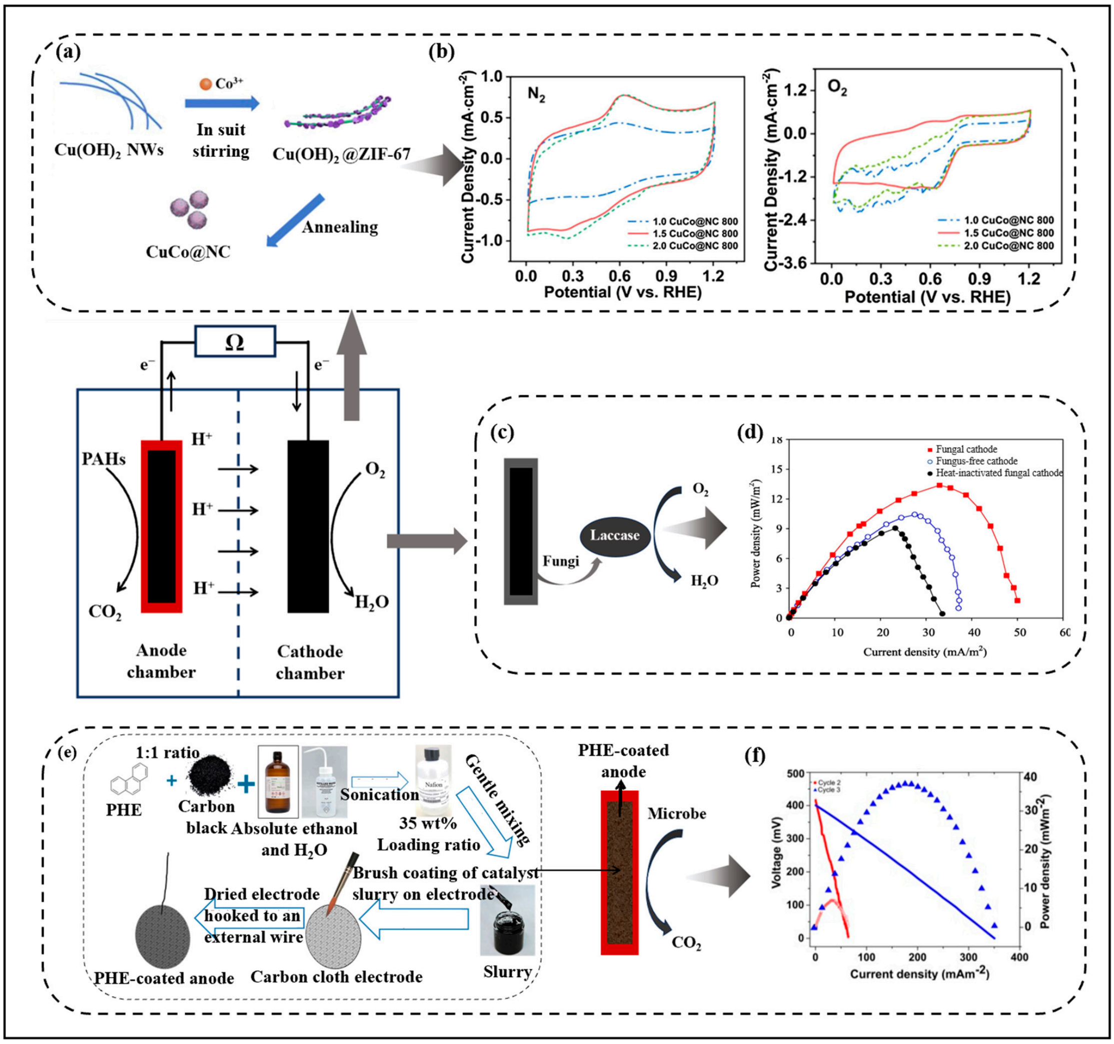


Figure 5. MFC innovative electrodes and their power production effects. (**a**) The synthesis step of CuCo@NC cathode [114]; (**b**) CV curves in O_2- or N_2-saturated PBS solution for CuCo@NC in different ratios [114]; (**c**) fungal cathode oxygen reduction by release of enzymes; (**d**) the power density of fungal cathode MFCs [129]; (**e**) PHE-loaded anode preparation process; and (**f**) the power density of the 2 mg/cm^2 PHE-loaded anode MFC [68].

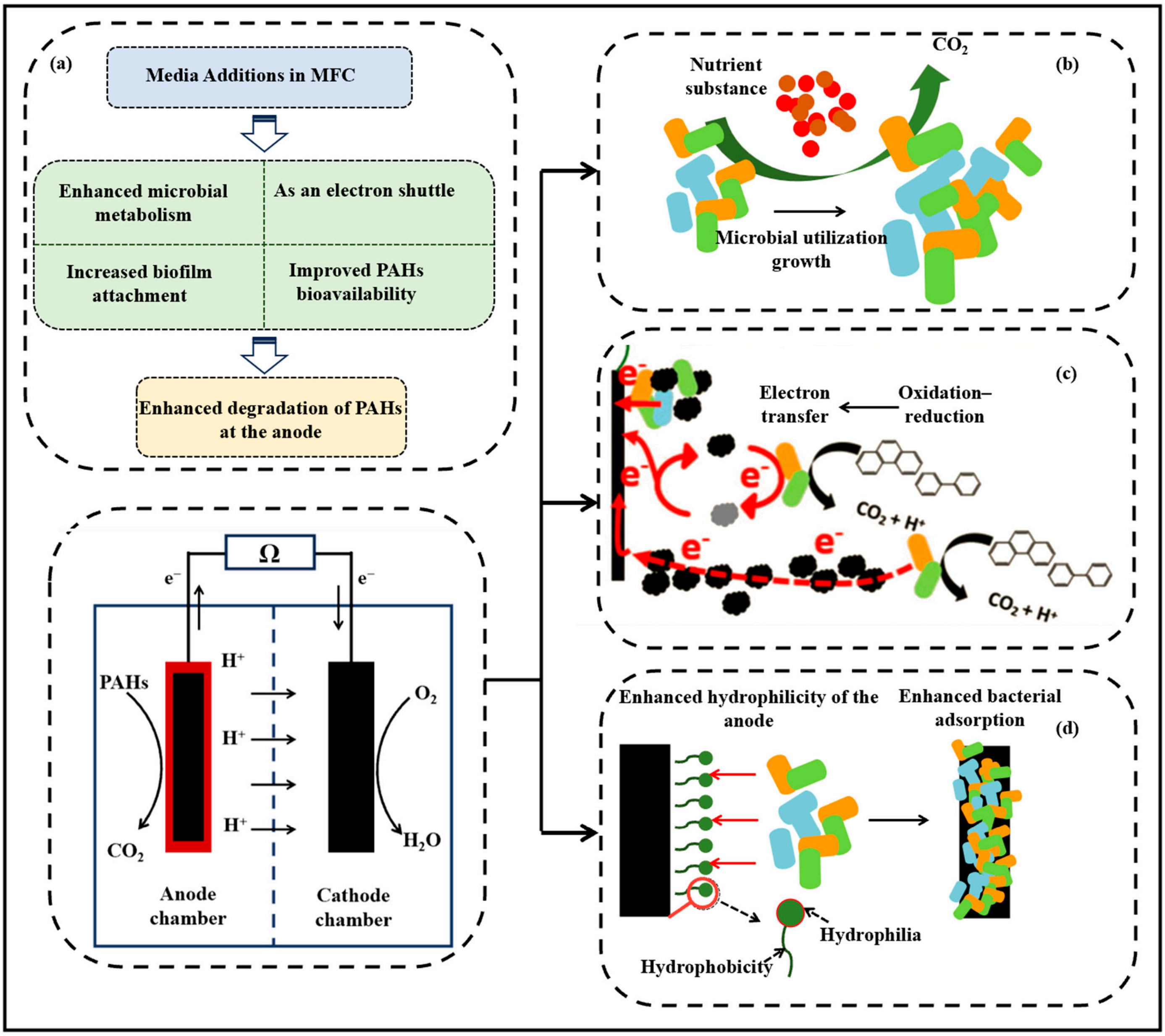


Figure 6. Schematic illustration of the principle of mediator addition to enhance the electrical production and degradation properties of the BES. (**a**) Reasons for the enhancement of PAH degradation by mediator addition; (**b**) enhancement of microbial growth by carbon source mediator addition; (**c**) electron transfer by redox mediator biochar addition [123]; and (**d**) enhancement of anodic microbial enrichment and adsorption by surfactant addition.

3.4.3. Addition of Other Redox Media

Redox mediators can improve BES conductivity by acting as electron shuttles [146]. The surface of biochar contains various oxidized functional groups, and the incorporation of biochar promotes long-range electron transport and enhances the electron transfer efficiency at the anode by improving the electrical conductivity of the soil. Additionally, biochar's potent adsorption ability can increase the bioavailability of contaminants by encouraging electron transport from the surrounding environment to the biochar. The degradation efficiency of PHE in MFCs treated with biochar pyrolyzed at 900 °C reached 86.4–95.1% within 27 days, and the effective range of pollutant degradation increased from 30–42 cm to 154–271 cm (Figure 6c) [123]. Electron acceptors such as nitrate can enhance biostimulation to promote the oxidation and biodegradation of contaminants in sediments and promote the performance of BESs. To address the migration of nitrate from the sediment to the upper water column, nitrate was continuously released using nitrocellulose (NC) to maintain a high concentration of nitrate at the anode at all times, and the expression of nitrogen cycling genes was enhanced by the combination of NC and MFCs to improve the removal efficiency of PAHs and acid-volatile sulfides (AVSs) at the anode. The voltage of the NC-MFC treatment on day 70 day was maintained at 146.1 mV, which was 5.37 times higher than that of the $Ca(NO_3)_2$-MFC treatment. This technique not only improved the removal efficiency of PAHs but also reduced the migration of nitrate from the sediment to the upper water column, reducing the risk to the aquatic ecosystem [118]. The incorporation of different electron mediators has been extensively researched, and the addition of redox mediators has been shown to be highly effective in improving the conductivity of BESs. Expanding the use of low-cost, safe electron mediators and learning more about the mechanics underlying electron transfer should be the main goals of the following phase.

4. Degradation Pathways of PAHs

4.1. Biodegradation of PAHs

PAH-degrading bacteria can be isolated where PAH contamination exists, such as in contaminated soils or sediments, as well as in petrochemical wastes discharged over time [147]. During microbial degradation of PAHs, PAHs are first degraded to small molecules by chain breaking, and then further metabolized to CH_4 and CO_2 [148]. In the presence of relevant degrading bacteria in the environment, PAHs are degraded mainly through ring cleavage, side-chain metabolism, and central metabolism processes. PAHs also affect other metabolic pathways in bacteria, such as significantly enhancing carbohydrate metabolism pathways and negatively correlating with amino acid metabolism pathways and membrane transport pathways [147]. Under aerobic conditions, oxygenases are the first to degrade, and their oxidative action is responsible for the key degradation steps of initial oxidation and benzene ring cleavage. Under anoxic conditions, the degradation is mainly carried out by carboxylation, methylation, or the addition of fumarate [149]. Enzymatic reactions occur with each other under the action of various enzymes, including mainly dioxygenases, dihydrodiol dehydrogenases, ring-breaking dioxygenases, epoxide hydrolases, ethanol dehydrogenases, acetaldehyde dehydrogenases, and decarboxylases [23,149]. As the number of rings in the molecule increases, the difficulty of degradation also increases, as does the number of potential hydroxylation sites for ring hydroxylases [149]. Zhou et al. explored the regularity in the degradation of pure and mixed PAHs and found that LMW-PAHs could be used as substrates to further promote the degradation of HMW-PAHs. In mixed PAHs, increased microbial community abundance and diversity were demonstrated by Chao1, Shannon, Simpson, and other indicators. The degradative enzymes produced by microorganisms during the mixing process of PAHs could undergo cross-domestication, adapt to the presence of multiple PAHs, and increase the utilization of PAHs, and thus the degradation rate of mixed PAHs was higher than that of single PAHs [111]. In the following sections, the degradation pathways of five common PAHs (NAP, ANT, PHE, PYR, and FLU) are summarized.

4.2. The Degradation Pathway of NAP

Degradation mechanisms vary throughout bacteria in the environment. The first step of NAP’s degradation mainly includes carboxylation and methylation [111]. Conversion of NAP to 2,3-dihydroxynaphthalene and 1,2-dihydroxynaphthalene by thermophilic bacteria such as *Thermophile Bacillus* under the action of dioxygenases and aromatization, and further degradation of cinnamic acid as an intermediate, results in phthalic or salicylic acids [150]. *Cycloclasticus*, through ring hydroxylation dioxygenase, can efficiently convert NAP into the corresponding hydroxylated product, which further undergoes ring breaking, oxidation, and decarboxylation [151]. In the absence of oxygenase, the first step can be a direct carboxylation reaction. *Desulfovibrio* plays a major NAP degradation role in the process of sulfate reduction, first by carboxylation, in which the generated naphthalenemethyl further generates the central metabolite 2-naphthoyl coenzyme A under the action of reductase, followed by reductive hydrogenation, dehydration, ring breakage, carbon skeletal rearrangement, and oxidation, directly degrading NAP into acetyl coenzyme A [152]. In the degradation of NAP, salicylic acid, benzoic acid, and catechol (catechol) are generally the end products, which then enter the tricarboxylic acid cycle through the degradation of these small molecules such as pyruvate (Figure 7a) [151].

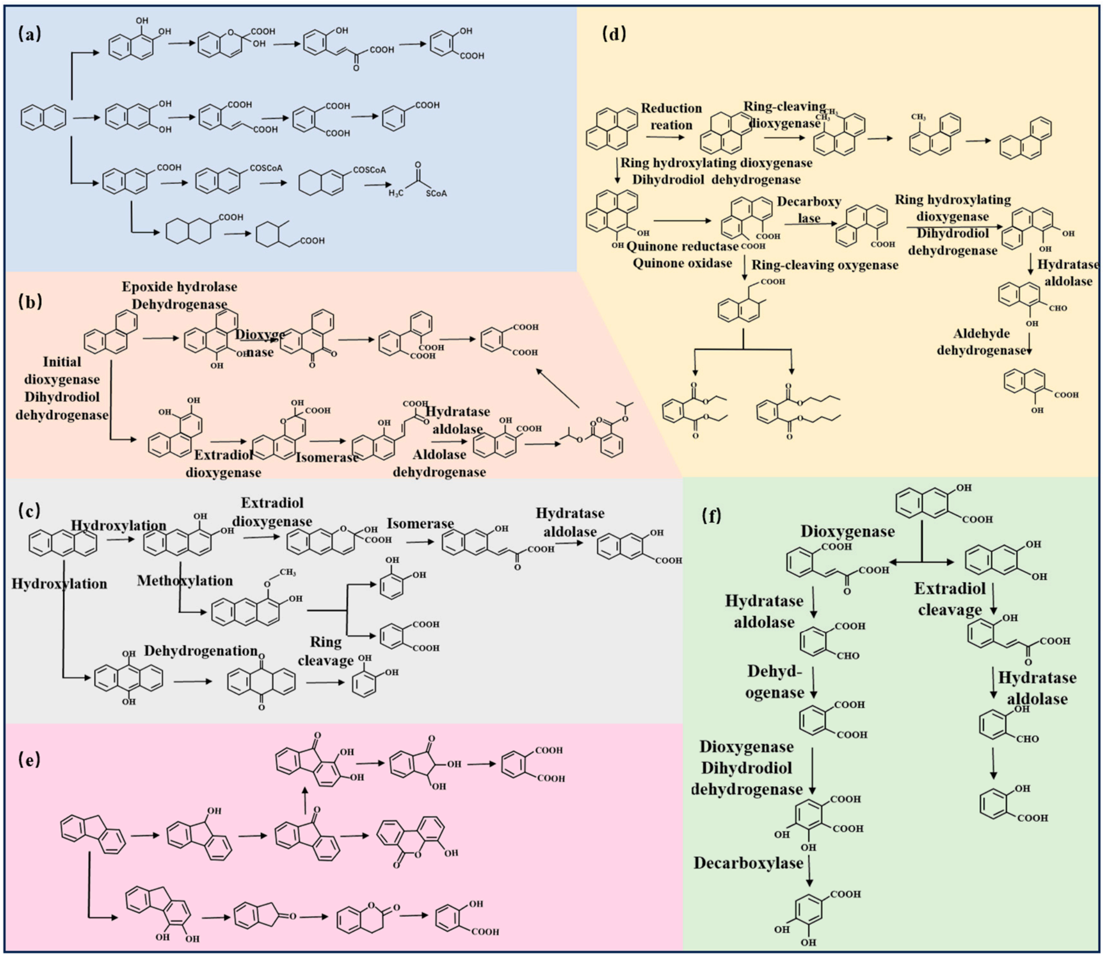


Figure 7. Diagram of common PAH degradation pathways. (**a**) NAP degradation pathway; (**b**) ANT degradation pathway; (**c**) PHE degradation pathway; (**d**) PYR degradation pathway; (**e**) FLU degradation pathway; and (**f**) 2-hydroxy-3-naphthoformic acid degradation pathway.

4.3. The Degradation Pathway of ANT

Under the influence of microorganisms, the initial degradation of ANT typically involves hydroxylation, followed by dehydrogenation and ring cleavage reactions, or oxidation under the action of dioxygenases. Intermediate products that can be detected include dihydroxyanthraquinone, anthraquinone, naphthalene-9,10-dicarboxylic acid, trans-2-(3′-hydroxynaphtho-4-yl)-oxo-but-2-enoic acid, and naphthalene dicarboxylic acid, etc. [153]. In the presence of free radicals and singlet oxygen, the main byproduct of ANT degradation is anthraquinone, which further degrades over time into dibenzofuran and 2,3,4-trimethylpentane, and is ultimately oxidized into CO_2 and H_2O. Other possible products include 2,3-dimethylnaphthalene, which continues to generate 1,3-di-tert-butylbenzene, and 2,3,4-trimethylpentane under the action of free radicals [154]. During the metabolic process of *L. fermentum* and *V. parvula*, ANT is converted into anthraquinone under the action of hydrolases and dehydrogenases and then generates salicylic acid through ring cleavage, which is further decarboxylated to form catechol. The anthracene molecule may be degraded in two different ways. *S. anginosus* degrades ANT by first generating anthracene-1,2-diol under the action of hydrolases, then methoxylates to form 1-methoxy-2-hydroxyanthraquinone, and finally produces catechol and phthalic acid derivatives through ring cleavage, which can be further metabolized into fatty acids after ring cleavage and polymerization (Figure 7b) [155].

4.4. The Degradation Pathway of PHE

In the presence of microbial hydroxylation, carboxylation, and methylation metabolism in the environment, PHE can generate 1-hydroxy-2-naphthoate intermediates, which in turn react to generate phthalic acid [111,118]. The metabolism of PHE initially involves the action of ring-hydroxylating dioxygenases to produce alcohols, either cis-3,4-dihydrodiols or cis-9,10-dihydrodiols. These are further oxidized by enzymes such as extradiol dioxygenases, isomerases, and hydratase-aldolases to yield 1-hydroxy-2-naphthoic aldehyde. The process continues under the influence of dehydrogenases to produce 2-carboxybenzaldehyde, which then cleaves to form 2-carboxyphenylaldehyde, ultimately leading to the formation of phthalic acid and catechol. Subsequently, under the influence of ring cleavage and redox reactions, the final products are generated and enter the tricarboxylic acid cycle [156]. Efficient degradation of PHE by cyclic hydroxylated dioxygenases also occurs [151]. Under anaerobic conditions, sulfate-reducing bacteria can biotransform phenanthrene in a series of hydration and hydrolysis reactions using sulfate as an electron acceptor, followed by decarboxylation to form p-cresol and phenol, and further biotransformation of the end product acetic acid (Figure 7c,f) [157].

4.5. The Degradation Pathway of PYR

Under anaerobic conditions, the degradation of PYR can first proceed through hydroxylation and methylation reactions to form phenanthrene, which then undergoes further degradation. Under aerobic conditions, PYR is initially converted into hydroxylated products under the action of monooxygenases and dioxygenases. Under the influence of dioxygenases, hydroxylation leads to the formation of cis-dihydrodiols, which are then re-aromatized into diol intermediates through dehydrogenation reactions. These are subsequently cleaved by angular or ortho cleavage by ring-cleaving dioxygenases, producing small molecules such as catechol, which enter the tricarboxylic acid cycle. Under the action of monooxygenases, arene epoxide intermediates are formed, which are then converted into trans-dihydrodiols [149]. Alternatively, under the action of monooxygenases and dioxygenases, 1-hydroxypyrene is first generated. Then, 1-hydroxy-pyrene is converted to 1-hydroxy-2-naphthenic acid. In addition, 1-hydroxy-2-naphthenic acid can be removed by different pathways. It can subsequently generate phenylacetic acid, which is further converted into catechol, or it can produce terephthalic acid, eventually entering the tricarboxylic acid cycle (Figure 7d) [113].

4.6. The Degradation Pathway of FLU

Degradation of FLU is usually first hydroxylated under the action of monooxygenase to 9-fluorenol and further dehydrogenated to 9-fluorenone, which is synthesized by dioxygenation to 1,2-dihydroxy-9-fluorenone and decomposed to 2,3-dihydroxyindanone by ring cleavage, and then further converted to phthalic acid by ring-opening oxidation and other series of reactions [158,159]. 9-Fluorenone can also be converted to biphenyl via a five-membered ring break, which is further esterified to produce phthalic acid or 8-hydroxy-3,4-benzocoumarin [160]. Alternatively, FLU degradation starts at the 3,4 site and begins with the formation of fluorene-3,4-diol under the action of dioxygenase. Fluorene-3,4-diol is metabolized to 1-indanone by ring-breaking, decarboxylation, and other reactions, which is metabolized to 3,4-dihydrocoumarin and then further metabolized to salicylic acid (Figure 7e) [161]. These eventually generate small molecule products that enter the cellular metabolic cycle.

5. Anode-Dominant Microorganisms in BES Degradation of PAHs

5.1. Dominant Microorganisms on Anodes

In BESs, anode microorganisms carry out the decomposition of organic matter to produce electricity and play a key role in electron generation and transfer [162]. The cathodic reaction, on the other hand, is relatively homogeneous, being mainly an oxygen reduction reaction that occurs in the presence of an abiotic catalyst with low microbial requirements [163]. In this regard, existing studies have usually explored anode-dominant microorganisms to explore means to further improve BES power generation performance and treatment effectiveness. The degradation of PAHs by the BES is mainly focused on the degradation ability of the anode microbial community, and exploring the dominant microbes at the BES anode and the changes to different environments by using high-throughput sequencing is essential to further elucidating the principle of the BES [164,165]. The addition of the BES changes the microbial community distribution through electrical stimulation. The anode microbial community richness and evenness increased and diversity decreased in the closed-circuit reactor compared to the open-circuit environment [166]. The BES promoted the growth of bacteria associated with bioelectrogenesis and acted as a microbial selector and enricher, increasing the abundance of microbial populations in the vicinity of the anode by two orders of magnitude under current stimulation [116]. Electroactive microorganisms attached near the anode have power generation and remediation effects, generate electrons by degrading pollutants in the environment, and transfer electrons efficiently, facilitating direct and indirect electron transfer mechanisms [167].

Exoelectrogens allow short-range electron transport through the outer membrane or c-type cytochromes as well as long-range electron transport through hyphae [113]. *Proteobacteria*, *Bacteroidota*, *Actinobacteriota*, *Desulfobacterota*, *Firmicutes*, and *Chloroflexi* are commonly found in anodes with external electrodes. *Proteobacteria* are the main anode-dominant microorganisms, to which *Pseudomonas*, *Alphaproteobacteria*, *Betaproteobacteria*, *Deltaproteobacteria*, *Gammaproteobacteria*, and *Thauera* belong; at the anode, these microorganisms were detected in high abundance and all were electroactive bacteria. *Firmicutes* and *Proteobacteria* can degrade organic matter and perform extracellular electron transfer [138]. *Pseudomonas*, one of the dominant species in anodic biofilms, produces phenazines and pyocyanin electron shuttling compounds and PAH-degrading enzymes, which directly enhance the electrode reaction to promote PAH removal, as well as biosurfactants, which alter cell surface hydrophobicity [168]. Adelaja et al. confirmed that *P. aeruginosa* of this species secretes biosurfactants to dissolve suspended PHE, enhancing the bioavailability of PAHs [133]. *Alphaproteobacteria*, *Betaproteobacteria*, *Deltaproteobacteria*, and *Gammaproteobacteria* are also the main bacterial categories on the anode in PHE wastewater, which are capable of degrading kerosene and gasoline, and their abundance is related to the power density of the BES [112,140]. *Thauera* is also believed to participate in the degradation process of PAHs by promoting the expression of hydrocarbon metabolic enzymes. Especially in the later stages of BES treatment of PHE wastewater operation, its relative abundance increases in the anodic biofilm, indicating that it may also play a role in the degradation of PHE [110]. *Diaphorobacter*, *Lysobacter*, and *Dechlorosoma* all play important roles in degrading PAHs in the BES, enriching the anode with PAHs and electrical effects [132,145]. A higher relative abundance of *Actinobacteria* is associated with good power generation and pollutant removal performance in MFCs, which can convert PAHs such as NAP and PYR into CO_2 using them as carbon sources [113,138]. *Coriobacteriia* under this bacterium is found on the anode during the degradation of kerosene and gasoline and is associated with high power density [140]. *Chitinophagaceaea* is also considered an important organism for the degradation of PAHs and other organic pollutants [111]. *Rhodococcus*, belonging to *Actinomycetes*, is known for its ability to degrade PAHs and is detected with high abundance on the anode [110]. *Desulfobacterota* includes *Geobacter*, *Desulfobulbus*, and *Desulfuromonas* [119]. *Geobacter*, as the most common electrogens, has a high abundance and strong competitiveness in the degradation of PAHs, and its abundance does not change significantly with the addition of electron mediators [116,119]. As a typical electroactive anaerobic bacterium, *Geobacter* plays a particularly important role in extracellular electron transfer, and its relative abundance increases with the addition of PAH pollutants, allowing it to rapidly adapt to its environment and use PAHs as a carbon source for degradation and electricity production [112]. *Desulfobulbus* is related to electron transfer [132]. *Thiobacillus*, belonging to *Anaerolineaceae* of *Chloroflexi*, is an anaerobic bacterium that degrades benzene and some hydrocarbons. In anaerobic environments, it promotes the degradation of PHE and PYR and shortens the half-life of PAHs [111]. *Pseudomonas*, *Thauera*, *Thiobacillus*, *Diaphorobacter*, *Tumebacillus*, and *Lysobacter* are closely related to the metabolism of PHE and PYR as well as electron transfer at the anode and play a key role in many studies of degraded PAHs [132,145]. In 20 mg/L PHE anode cloth, *Pseudomonas*, *Rhodococcus*, *Thauera*, *Ralstonia*, *Geobacter*, *Petrimona*, and *Dechlorosoma* are associated with the degradation of phenanthrene [110,112].

There are also some non-dominant bacteria that participate in the degradation of PAHs, such as *Lentisphaerae* and *Verrucomicrobia* that promote anaerobic digestion and hydrolysis. *Acidobacteria* also degrades metabolic carbon compounds [138] and *Cyanobacteria* play an important role in degrading PAHs and other organic pollutants [51]. Wang et al. first identified *Sphingobium yanoikuyae* XLDN2-5 as an efficient electrochemically active bacterium by electron transfer via self-generated conductive nanowires and applied it to MFCs for carbazole degradation, reaching a maximum power density of 496.8 mW/m^2 and a 100% carbazole degradation rate [46]. Some filamentous fungi, which can perform long-distance electron transport (LDET) (such as fiber optic cable bacteria), can be directly coupled over centimeter distances, with LDET connecting the oxygen layer to the sulfur-reducing layer. Replacement of sulfate by LDET uses remote oxygen as an electron acceptor for hydrocarbon oxidation and also contributes to improved microbial degradation of hydrocarbons [169]. In degrading PAH contamination, a variety of electroactive microorganisms are enriched in the BES anode to promote contaminant degradation through the production of electron shuttles and surfactants. The cooperation of electro-producing and degradative bacteria highlighted the importance of microbial communities in enhancing the removal efficiency of PAHs.

5.2. Effects of Environmental Changes on Microorganisms

It is evident that the dominating anode microbial populations varied in response to environmental changes (Table 3). Seawater and freshwater exhibit considerable differences, primarily due to the higher salinity levels found in seawater. *Desulfobacterota* is the dominant bacterium in treatments of marine sediments, and the presence of sulfate in the environment increases the abundance of *Desulfobacterota*. Utilizing S and Fe as electron acceptors, *Desulfobacterota* is in charge of removing hydrocarbons by utilizing sulfate and its derivatives as transfer electron acceptors, producing electricity for MFCs, and degrading sulfate and other derived elements [47]. Hamdan et al. explored the removal of PAHs from marine sediments with sulfur-reducing bacteria inhibition at the anode, by which the sulfate as an electron acceptor was reduced, at which time the anode as the sole electron acceptor reduced other redox processes, obtaining a higher removal of PAHs compared to an open-circuit SMFC. *Geoalkalibacter* and *Desulfuromonas* were enriched on the anodes of closed-circuit SMFCs. *Desulfuromonas* were mostly enriched on the anodes of marine SMFCs, *Geobacteraceae* were present in anoxic sediments, and all of these microorganisms oxidized multicarbon organic substrates [170]. The presence of iron minerals in the environment increases the relative abundance of *Firmicutes* and promotes their attachment to the anode, where electroactive and heterogeneous iron-reducing bacteria are enriched. *Pseudomonas* and *Dehalococcoides* are important electrogenic microorganisms, while *Pseudomonas* and *Thauera* are also important for PAH degradation. The heterogeneous iron-reducing bacteria *Petrimonas* and *Thermincola* are involved in the oxidation and reduction of Fe, promoting the redox cycle of Fe in the environment, and thereby accelerating electron transfer [117]. The oxygen environment also determines the power generation performance at the anode. Under aerobic conditions, the BES has a lower internal resistance but exhibits higher degradation efficiency in an anaerobic environment. The degradation rates for NAP and PHE were 41.7% and 36.2%, respectively, in the aerobic environment, and 76.9% and 36.8% in the anaerobic environment. This may be due to the higher redox potential provided by aerobic respiration, which is higher and improves the electron transfer efficiency; but in anaerobic environments, the solubility and mobility of PAHs may be increased, and some specific microorganisms may utilize PAHs more efficiently as an energy and carbon source [65].

Table 3. Anode-dominant bacteria in different BESs.

BES	PAHs	Phylum	Genus	Reference
Water MFC Anode: Carbon cloth Cathode: Pt/C	NAP	*Chloroflexi*, *Firmicutes*	*Anaerolineaceae*, *Roseiflexus*, *Roseiflexus*	[111]
	PHE	*Firmicutes*, *Chloroflexi*, *Cyanobacteria*	*Clostridium*, *Cyanobacteria*, *JG30-KF-CM45*	
	PYR	*Firmicutes*, *Acidobacteria*	*Anaerolineaceae*, *Blastocatellaceae*, *JG30-KF-CM45*	
	NAP, PHE	*Chloroflexi*, *Proteobacteria*, *Chlorobi*	*Chlorobi*, *Methylophilus*, *OPB56*	
	NAP, PYR	*Chloroflexi*, *Proteobacteria*	*Anaerolineaceae*, *Methylophilus*, *PHOS-HE36*	
	PHE, PYR	*Ignavibacteriae*, *Bacteroidete*	*PHOS-HE36*, *Chitinophagaceae*	
Water MFC Anode: Carbon brush Cathode: Stainless steel	PW (TPH)	*Proteobacteria*, *Desulfobacterota*, *Actinobacteriota*	*Desulfuromonas*, *Desulfovibrio*, *Marinobacter*	[47]
Water MFC Anode: Carbon felt Cathode: Carbon felt	NAP, PYR, ACE	*Firmicutes*, *Bacteroidetes*, *Proteobacteria*,	*Clostridium*, *Macellibacteroides*, *Acinetobacter*	[138]

Table 3. *Cont.*

BES	PAHs	Phylum	Genus	Reference
Water MFC Anode: Carbon cloth Cathode: Carbon cloth	PHE	*Proteobacteria*, *Bacteroidetes*	*Geobacter*, *Acidovorax*, *Petrimonas*	[112]
Water MFC Anode: Carbon cloth Cathode: Carbon cloth	PYR	*Proteobacteria*, *Bacteroidetes*	*Rhizobiaceae*, *Dysgonmonadaceae*, *Burkholderiaceae*	[113]
Soil MFC Anode: Carbon fiber felt Cathode: Carbon fiber felt	ANT, PHE, PYR	*Proteobacteria*, *Bacteroidetes*, *Actinobacteria*	*Geobacter*, *Candidate*, *BVA59*	[116]
Soil MFC Anode: Carbon cloth Cathode: Stainless steel	PHE	*Proteobacteria*, *Desulfobacterota*, *Firmicutes*	*Petrimonas*, *Thermincola*, *Soehngenia*	[117]
Sediment MFC Anode: GR, GO, CNT, GF Cathode: GF	PHE, PYR	*Proteobacteria*, *Chloroflexi*, *Acidobacteria*	*Thauera*, *Thiobacillus*, *Pseudomonas*	[132]
Sediment MFC Anode: Graphite Cathode: Graphite	PHE, PYR, BaP	*Pseudomonadota*, *Chloroflexi*	*Azospirillum*, *Ancylobacter*, *Longilinea*	[168]
Sediment MFC Anode: Graphite Cathode: Graphite Nitrate	PHE, PYR, BaP	*Pseudomonadota*, *Proteobacteria*, *Desulfobacterota*	*Thiobacillus*, *Lysobacter*, *Rhodanobacter*	
Sediment MFC Anode: Carbon felt Cathode: Carbon felt	NAP, PHE, PYR	*Desulfobacterota*, *Proteobacteria*, *Firmicutes*	*Sedimentibacter*, *Erysipelothrix*, *Geobacter*	[119]
Sediment MFC Anode: Carbon fiber brush Cathode: Carbon fiber brush	NAP, PHE	*Proteobacteria*, *Firmicutes*	*Thalassospira*, *Desulfuromusa*, *Halomonas*	[170]

To improve the degradation of PAHs by altering the microbial community, additional media were introduced. Zhang et al. investigated microbial community changes in BES anodes under the co-contamination of PAHs and acid-volatile sulfides, and the addition of nitrate acted as an electron acceptor for the oxidative sulfate reaction and increased the population density of *Pseudomonas*. At this time, *Thiobacillus* was a dominant population in the environment, and electrochemical activity and nitrate limited the circumstances, with the anode as the electron acceptor, while driving the generation of electrical energy [168]. The highest amount of anodic *Bacteroides* and *Proteobacteria* was detected when 10 mg/L of starch was introduced. *Proteobacteria* became less abundant and *Firmicutes* became more abundant when some acid was added. This is because *Firmicutes* are acid-resistant, but *Proteobacteria* are sensitive to acidity. Therefore, when a high concentration of acidic medium is added, the degradation of PAHs will decline [138]. Biochar addition promoted the growth of *Proteobacteria* and POP-degrading bacteria. Biochar improves pollutant degradation efficiency by increasing the abundance of functional genes associated with PAH degradation and establishing a metabolic network of interactions [123]. The anode microbial community structure in BESs is greatly influenced by environmental factors including oxygen levels and salinity, which have an impact on power generation and the efficiency of PAH decomposition. The adaptability and metabolic activity of specific microbial communities are key to achieving efficient environmental remediation and energy recovery.

6. Challenges and Prospects

1. The complexity of BES configurations and operating conditions makes it cumbersome to operate when applied to real environments, limiting its application to real environmental remediation. By innovating BES configurations (e.g., plug-in and portable) to adapt to the needs of complex environments and optimizing the BES design, including electrode material, configuration, and media addition, the energy conversion efficiency and pollution removal performance of the system can be improved. Existing BESs often use expensive Pt/C electrodes, which restricts their large-scale application. Although there are some cathode materials to prepare modified BESs, they have not been applied on a large scale. Subsequently, the development of cheap, efficient, and stable electrode materials is still needed, and their popularization and application, such as the development of metal oxides, carbon nanomaterials, natural conductive materials, and other low-cost and high-efficiency materials, are needed to further enhance the performance and economic feasibility of BESs. A balance between degradation efficiency and control of operating costs can be achieved through the development of simpler and more cost-effective equipment to facilitate the commercialization and scale-up of BES technology.
2. The stability of the microbial community and its synergy with the electrode are crucial for degradation efficiency but are affected by changes in environmental conditions, which may lead to fluctuations in degradation efficiency. Through an in-depth study of the degradation mechanism of microorganisms at the anode, more efficient strains can be screened for use in the BES, and the electrochemically active microbial community can be optimized through genetic engineering or metabolic modulation to enhance its degradation efficiency and stability. In particular, the microorganisms at the anode have a limited degradation range for organic pollutants, and more strains capable of long-distance electron transport are considered to be screened. This will improve electron utilization, help the anode microorganisms to adapt to the complex environment, and improve the removal efficiency of PAHs.
3. Although the BES can reduce secondary pollution, intermediate products such as phthalate derivatives and 9-fluorenyl formaldehyde derivatives may be produced during degradation. These intermediate products still have ecological risks and human health exposure risks, which need to be further studied and controlled. The BES can be combined with other processes to enhance the synergistic degradation ability of the system and reduce the leakage of intermediate products. The advantages of various technologies can be fully utilized to improve the degradation effect of PAHs and broaden the application scope of BESs. For example, AOPs can provide strong oxidizing radicals to participate in pollutant degradation, artificial wetlands can realize natural purification through plants and microorganisms, and membrane technology can improve the separation efficiency of water treatment.
4. As a green and sustainable core technology to solve PAH pollution, the above investigation of the BES is only in the experimental stage and has not been practically applied. Scale application of the BES still faces technical difficulties, including electrode life, system integration, and cost control. How to apply the BES in different environments and ensure its longevity and reliability are key challenges for practical application. The next step should be to ensure maximum resource recovery and minimum environmental burden in system operation, systematically assess the environmental impact and economic benefits of the BES, formulate standardized operating procedures, and promote its transition from laboratory to practical application.
5. PAHs are oxidized by combustion, light, or other means to produce a range of derivatives containing heteroatoms in the environment. Although these compounds are less

toxic, they are more carcinogenic and mutagenic, and their accumulation in the food chain makes them more hazardous to humans. It is therefore also crucial to investigate the sources, distribution, remediation techniques, and degradation pathways of these substances, as well as their possible effects on humans and the environment. Since the characterization of PAHs is altered by externally added groups, the next step in BES studies of these derivatives not only contributes to a better understanding of the environmental behavior and health risks of PAHs but also provides a scientific basis for the development of effective remediation strategies.

7. Conclusions

An overall overview of BES degradation of PAHs is provided by exploring the source distribution of PAHs and outlining the principles of the BES and its application in PAH removal. And special attention has been paid to the relevant studies in recent years, mainly through changing the configuration, electrode modification, and media additions to improve the BES efficiency. PAH degradation is further analyzed to elucidate some PAH degradation pathways and BES anode-dominant microorganisms. Finally, the challenges of the BES technology, such as the impracticality of the configuration, the instability of the microorganisms, the limited range of degradation, and the toxicity of the intermediates, are emphasized, and the directions for future research are pointed out, including accelerating the degradation efficiency in conjunction with other processes, screening of the anodic strains through further studies, and more studies on derived PAHs. Through the summary analysis, this review provides a comprehensive perspective on the research field of PAH degradation by the BES that is easy for non-professional readers to understand. The basic principles and applications of the BES for PAH degradation are presented in a comprehensive format in order to lay a foundation for further research and application.

Funding: This work was supported by the National Natural Science Foundation of China (32171615) and the National Key Research and Development Program of China (2019YFC1804102).

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

Abbreviations

NAP	Naphthalene
ANY	Acenaphthylene
ACE	Acenaphthene
FLU	Fluorene
PHE	Phenanthrene
ANT	Anthracene
PYR	Pyrene
FLU	Fluoranthene
BaP	Benzo (a) pyrene
BaA	Benzo(a)anthracene
BbF	Benzo (b) fluoranthene
BkF	Benzo (k) fluoranthene
DBA	Dibenzoic (a,h) anthracene
CHR	Chrysene
BgP	Benzo (ghi) pyrene
InP	Indeno (1, 2, 3-CD) pyrene
BES	Bioelectrochemical system

PAHs	Polycyclic aromatic hydrocarbons
LMW	Low molecular weight
MMW	Medium molecular weight
HMW	High molecular weight
TPHs	Total petroleum hydrocarbons
MFC	Microbial fuel cell
MEC	Microbial electrolysis cell
AOPs	Advanced oxidation processes
PMS	Peroxymonosulfate
GO	Graphene oxide
Pt/C	Platinum/Carbon
CNTs	Carbon nanotubes
GFs	Graphite felts
HAs	Humic acids
NC	Nitrocellulose
AVSs	Acid-volatile sulfides
LDET	Long-distance electron transport

References

1. Abdel-Shafy, H.I.; Mansour, M.S.M. A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation. *Egypt. J. Pet.* **2016**, *25*, 107–123. [CrossRef]
2. Domingo, J.L.; Nadal, M. Human dietary exposure to polycyclic aromatic hydrocarbons: A review of the scientific literature. *Food Chem. Toxicol.* **2015**, *86*, 144–153. [CrossRef] [PubMed]
3. Bai, X.; Sun, H.; Sun, J.; Zhu, Z. Efficient removal of sixteen priority polycyclic aromatic hydrocarbons from textile dyeing sludge using electrochemical Fe^{2+}-activated peroxymonosulfate oxidation-A green pretreatment strategy for textile dyeing sludge toxicity reduction. *J. Hazard. Mater.* **2022**, *435*, 129087. [CrossRef] [PubMed]
4. Vijayanand, M.; Ramakrishnan, A.; Subramanian, R.; Issac, P.K.; Nasr, M.; Khoo, K.S.; Rajagopal, R.; Greff, B.; Azelee, N.I.W.; Jeon, B.-H.; et al. Polyaromatic hydrocarbons (PAHs) in the water environment: A review on toxicity, microbial biodegradation, systematic biological advancements, and environmental fate. *Environ. Res.* **2023**, *227*, 115716. [CrossRef] [PubMed]
5. Singh, S.; Ashesh, A.; Devi, N.L.; Yadav, I.C. A comprehensive review on occurrence, source, effect, and measurement techniques of polycyclic aromatic hydrocarbons in India. *Microchem. J.* **2022**, *183*, 108005. [CrossRef]
6. Rubio-Clemente, A.; Torres-Palma, R.A.; Peñuela, G.A. Removal of polycyclic aromatic hydrocarbons in aqueous environment by chemical treatments: A review. *Sci. Total Environ.* **2014**, *478*, 201–225. [CrossRef] [PubMed]
7. Rengarajan, T.; Rajendran, P.; Nandakumar, N.; Lokeshkumar, B.; Rajendran, P.; Nishigaki, I. Exposure to polycyclic aromatic hydrocarbons with special focus on cancer. *Asian Pac. J. Trop. Biomed.* **2015**, *5*, 182–189. [CrossRef]
8. Zhang, Y.; Yuan, L.; He, S.; Tao, H.; Xie, W.; Zhang, X.; Ren, X.; Jiang, T.; Li, L.; Zhu, Z. Contemporary research progress on the detection of polycyclic aromatic hydrocarbons. *Int. J. Environ. Res. Public Health* **2022**, *19*, 2790. [CrossRef] [PubMed]
9. Perera, F.P.; Tang, D.; Wang, S.; Vishnevetsky, J.; Zhang, B.; Diaz, D.; Camann, D.; Rauh, V. Prenatal polycyclic aromatic hydrocarbon (PAH) exposure and child behavior at age 6–7 years. *Environ. Health Perspect.* **2012**, *120*, 921–926. [CrossRef] [PubMed]
10. Sun, K.; Song, Y.; He, F.; Jing, M.; Tang, J.; Liu, R. A review of human and animals exposure to polycyclic aromatic hydrocarbons: Health risk and adverse effects, photo-induced toxicity and regulating effect of microplastics. *Sci. Total Environ.* **2021**, *773*, 145403. [CrossRef]
11. Xu, X.; Liu, X.; Zhang, J.; Liang, L.; Wen, C.; Li, Y.; Shen, M.; Wu, Y.; He, X.; Liu, G.; et al. Formation, migration, derivation, and generation mechanism of polycyclic aromatic hydrocarbons during frying. *Food Chem.* **2023**, *425*, 136485. [CrossRef]
12. Kim, K.H.; Jahan, S.A.; Kabir, E.; Brown, R.J.C. A review of airborne polycyclic aromatic hydrocarbons (PAHs) and their human health effects. *Environ. Int.* **2013**, *60*, 71–80. [CrossRef] [PubMed]
13. Qin, N.; He, W.; Liu, W.; Kong, X.; Xu, F.; Giesy, J.P. Tissue distribution, bioaccumulation, and carcinogenic risk of polycyclic aromatic hydrocarbons in aquatic organisms from Lake Chaohu, China. *Sci. Total Environ.* **2020**, *749*, 141577. [CrossRef]
14. Zhang, Y.; Dong, S.; Wang, H.; Tao, S.; Kiyama, R. Biological impact of environmental polycyclic aromatic hydrocarbons (ePAHs) as endocrine disruptors. *Environ. Pollut.* **2016**, *213*, 809–824. [CrossRef]
15. Tufail, M.A.; Iltaf, J.; Zaheer, T.; Tariq, L.; Amir, M.B.; Fatima, R.; Asbat, A.; Kabeer, T.; Fahad, M.; Naeem, H.; et al. Recent advances in bioremediation of heavy metals and persistent organic pollutants: A review. *Sci. Total Environ.* **2022**, *850*, 157961. [CrossRef] [PubMed]

16. Lamichhane, S.; Krishna, K.C.B.; Sarukkalige, R. Polycyclic aromatic hydrocarbons (PAHs) removal by sorption: A review. *Chemosphere* **2016**, *148*, 336–353. [CrossRef] [PubMed]
17. Lemaire, J.; Buès, M.; Kabeche, T.; Hanna, K.; Simonnot, M.O. Oxidant selection to treat an aged PAH contaminated soil by in situ chemical oxidation. *J. Environ. Chem. Eng.* **2013**, *1*, 1261–1268. [CrossRef]
18. Usman, M.; Faure, P.; Ruby, C.; Hanna, K. Application of magnetite-activated persulfate oxidation for the degradation of PAHs in contaminated soils. *Chemosphere* **2012**, *87*, 234–240. [CrossRef]
19. Korfmacher, W.A.; Wehry, E.L.; Mamantov, G.; Natusch, D.F.S. Resistance to photochemical decomposition of polycyclic aromatic hydrocarbons vapor-adsorbed on coal fly ash. *Environ. Sci. Technol.* **1980**, *14*, 1094–1099. [CrossRef]
20. Niu, J.; Sun, P.; Schramm, K.W. Photolysis of polycyclic aromatic hydrocarbons associated with fly ash particles under simulated sunlight irradiation. *J. Photochem. Photobiol. A Chem.* **2007**, *186*, 93–98. [CrossRef]
21. Liu, S.; Zeng, G.; Niu, Q.; Liu, Y.; Zhou, L.; Jiang, L.; Tan, X.; Xu, P.; Zhang, C.; Cheng, M. Bioremediation mechanisms of combined pollution of PAHs and heavy metals by bacteria and fungi: A mini review. *Bioresour. Technol.* **2017**, *224*, 25–33. [CrossRef] [PubMed]
22. Mohan, S.V.; Kisa, T.; Ohkuma, T.; Kanaly, R.A.; Shimizu, Y. Bioremediation technologies for treatment of PAH-contaminated soil and strategies to enhance process efficiency. *Rev. Environ. Sci. Bio/Technol.* **2006**, *5*, 347–374. [CrossRef]
23. Haritash, A.K.; Kaushik, C.P. Biodegradation aspects of polycyclic aromatic hydrocarbons (PAHs): A review. *J. Hazard. Mater.* **2009**, *169*, 1–15. [CrossRef] [PubMed]
24. Kuppusamy, S.; Thavamani, P.; Megharaj, M.; Naidu, R. Biodegradation of polycyclic aromatic hydrocarbons (PAHs) by novel bacterial consortia tolerant to diverse physical settings—Assessments in liquid- and slurry-phase systems. *Int. Biodeterior. Biodegrad.* **2016**, *108*, 149–157. [CrossRef]
25. Hao, D.; Li, X.; Xiao, P.; Wang, L. The utility of electrochemical systems in microbial degradation of polycyclic aromatic hydrocarbons: Discourse, diversity and design. *Front. Microbiol.* **2020**, *11*, 557400. [CrossRef] [PubMed]
26. Bajracharya, S.; Sharma, M.; Mohanakrishna, G.; Benneton, X.D.; Strik, D.P.B.T.B.; Sarma, P.M.; Pant, D. An overview on emerging bioelectrochemical systems (BESs): Technology for sustainable electricity, waste remediation, resource recovery, chemical production and beyond. *Renew. Energy* **2016**, *98*, 153–170. [CrossRef]
27. Lovley, D.R.; Nevin, K.P. A shift in the current: New applications and concepts for microbe-electrode electron exchange. *Curr. Opin. Biotechnol.* **2011**, *22*, 441–448. [CrossRef] [PubMed]
28. Sleutels, T.H.J.A.; Heijne, A.T.; Buisman, C.J.N.; Hamelers, H.V.M. Bioelectrochemical systems: An outlook for practical applications. *ChemSusChem* **2012**, *5*, 1012–1019. [CrossRef]
29. Rabaey, K.; Girguis, P.; Nielsen, L.K. Metabolic and practical considerations on microbial electrosynthesis. *Curr. Opin. Biotechnol.* **2011**, *22*, 371–377. [CrossRef]
30. Ojha, R.; Pradhan, D. The potential of microbial fuel cell for converting waste to energy: An overview. *Sustain. Chem. Environ.* **2025**, *9*, 100196. [CrossRef]
31. Kirmizakis, P.; Doherty, R.; Mendonça, C.A.; Costeira, R.; Allen, C.C.R.; Ofterdinger, U.S.; Kulakov, L. Enhancement of gasworks groundwater remediation by coupling a bio-electrochemical and activated carbon system. *Environ. Sci. Pollut. Res.* **2019**, *26*, 9981–9991. [CrossRef]
32. Li, X.; Wang, X.; Zhao, Q.; Wan, L.; Li, Y.; Zhou, Q. Carbon fiber enhanced bioelectricity generation in soil microbial fuel cells. *Biosens. Bioelectron.* **2016**, *85*, 135–141. [CrossRef] [PubMed]
33. Long, X.; Cao, X.; Song, H.; Nishimura, O.; Li, X. Characterization of electricity generation and microbial community structure over long-term operation of a microbial fuel cell. *Bioresour. Technol.* **2019**, *285*, 121395. [CrossRef] [PubMed]
34. Liu, B.; Zhai, H.; Liang, Y.; Ji, M.; Wang, R. Increased power production and removal efficiency of polycyclic aromatic hydrocarbons by plant pumps in sediment microbial electrochemical systems: A preliminary study. *J. Hazard. Mater.* **2019**, *380*, 120896. [CrossRef]
35. Zhou, L.; Deng, D.; Zhang, D.; Chen, Q.; Kang, J.; Fan, N.; Liu, Y. Microbial electricity generation and isolation of exoelectrogenic bacteria based on petroleum hydrocarbon-contaminated soil. *Electroanalysis* **2016**, *28*, 1510–1516. [CrossRef]
36. Balati, A.; Shahbazi, A.; Amini, M.M.; Hashemi, S.H. Adsorption of polycyclic aromatic hydrocarbons from wastewater by using silica-based organic–inorganic nanohybrid material. *J. Water Reuse Desalination* **2014**, *5*, 50–63. [CrossRef]
37. Mojiri, A.; Zhou, J.L.; Ohashi, A.; Ozaki, N.; Kindaichi, T. Comprehensive review of polycyclic aromatic hydrocarbons in water sources, their effects and treatments. *Sci. Total Environ.* **2019**, *696*, 133971. [CrossRef]
38. Dai, C.; Han, Y.; Duan, Y.; Lai, X.; Fu, R.; Liu, S.; Leong, K.H.; Tu, Y.; Zhou, L. Review on the contamination and remediation of polycyclic aromatic hydrocarbons (PAHs) in coastal soil and sediments. *Environ. Res.* **2022**, *205*, 112423. [CrossRef]
39. Gan, S.; Lau, E.V.; Ng, H.K. Remediation of soils contaminated with polycyclic aromatic hydrocarbons (PAHs). *J. Hazard. Mater.* **2009**, *172*, 532–549. [CrossRef]

40. Ambaye, T.G.; Chebbi, A.; Formicola, F.; Rosatelli, A.; Prasad, S.; Gomez, F.H.; Sbaffoni, S.; Franzetti, A.; Vaccari, M. Ex-situ bioremediation of petroleum hydrocarbon contaminated soil using mixed stimulants: Response and dynamics of bacterial community and phytotoxicity. *J. Environ. Chem. Eng.* **2022**, *10*, 108814. [CrossRef]
41. Tian, L.; Yin, S.; Ma, Y.; Kang, H.; Zhang, X.; Tan, H.; Meng, H.; Liu, C. Impact factor assessment of the uptake and accumulation of polycyclic aromatic hydrocarbons by plant leaves: Morphological characteristics have the greatest impact. *Sci. Total Environ.* **2019**, *652*, 1149–1155. [CrossRef]
42. Huang, H.; Tang, J.; Niu, Z.; Giesy, J.P. Interactions between electrokinetics and rhizoremediation on the remediation of crude oil-contaminated soil. *Chemosphere* **2019**, *229*, 418–425. [CrossRef]
43. Cecconet, D.; Sabba, F.; Devecseri, M.; Callegari, A.; Capodaglio, A.G. In situ groundwater remediation with bioelectrochemical systems: A critical review and future perspectives. *Environ. Int.* **2020**, *137*, 105550. [CrossRef] [PubMed]
44. Li, X.; Li, Y.; Zhang, X.; Zhao, X.; Sun, Y.; Weng, L.; Li, Y. Long-term effect of biochar amendment on the biodegradation of petroleum hydrocarbons in soil microbial fuel cells. *Sci. Total Environ.* **2019**, *651*, 796–806. [CrossRef] [PubMed]
45. Thulasinathan, B.; Jayabalan, T.; Sethupathi, M.; Kim, W.; Muniyasamy, S.; Sengottuvelan, N.; Nainamohamed, S.; Ponnuchamy, K.; Alagarsamy, A. Bioelectricity generation by natural microflora of septic tank wastewater (STWW) and biodegradation of persistent petrogenic pollutants by basidiomycetes fungi: An integrated microbial fuel cell system. *J. Hazard. Mater.* **2021**, *412*, 125228. [CrossRef]
46. Wang, H.; Qi, X.; Zhang, L.; Zhang, X.; Xu, P.; Wang, X. Efficient bioelectricity generation and carbazole biodegradation using an electrochemically active bacterium Sphingobium yanoikuyae XLDN2-5. *Chemosphere* **2022**, *307*, 135986. [CrossRef]
47. Cabrera, J.; Dai, Y.; Irfan, M.; Li, Y.; Gallo, F.; Zhang, P.; Zong, Y.; Liu, X. Novel continuous up-flow MFC for treatment of produced water: Flow rate effect, microbial community, and flow simulation. *Chemosphere* **2022**, *289*, 133186. [CrossRef]
48. Patel, A.B.; Shaikh, S.; Jain, K.R.; Desai, C.; Madamwar, D. Polycyclic aromatic hydrocarbons: Sources, toxicity, and remediation approaches. *Front. Microbiol.* **2020**, *11*, 562813. [CrossRef]
49. Zhang, L.; Yang, L.; Zhou, Q.; Zhang, X.; Xing, W.; Wei, Y.; Hu, M.; Zhao, L.; Toriba, A.; Hayakawa, K.; et al. Size distribution of particulate polycyclic aromatic hydrocarbons in fresh combustion smoke and ambient air: A review. *J. Environ. Sci.* **2020**, *88*, 370–384. [CrossRef] [PubMed]
50. Landeros Rivera, B.; Hernández Trujillo, J. Control of molecular conformation and crystal packing of biphenyl derivatives. *ChemPlusChem* **2022**, *87*, e202100492. [CrossRef] [PubMed]
51. Jain, Z.J.; Gide, P.S.; Kankate, R.S. Biphenyls and their derivatives as synthetically and pharmacologically important aromatic structural moieties. *Arab. J. Chem.* **2017**, *10*, S2051–S2066. [CrossRef]
52. Jiang, W.; Li, Y.; Wang, Z. Heteroarenes as high performance organic semiconductors. *Chem. Soc. Rev.* **2013**, *42*, 6113–6127. [CrossRef] [PubMed]
53. Liu, L.; Liu, Y.; Lin, J.; Tang, N.; Hayakawa, K.; Maeda, T. Development of analytical methods for polycyclic aromatic hydrocarbons (PAHs) in airborne particulates: A review. *J. Environ. Sci.* **2007**, *19*, 1–11. [CrossRef]
54. Dat, N.D.; Chang, M.B. Review on characteristics of PAHs in atmosphere, anthropogenic sources and control technologies. *Sci. Total Environ.* **2017**, *609*, 682–693. [CrossRef]
55. Chen, Y.; Zeng, Y.; Guan, Y.; Huang, Y.; Liu, Z.; Xiang, K.; Sun, Y.; Chen, S. Particle size-resolved emission characteristics of complex polycyclic aromatic hydrocarbon (PAH) mixtures from various combustion sources. *Environ. Res.* **2022**, *214*, 113840. [CrossRef]
56. Aarab, N.; Godal, B.F.; Bechmann, R.K. Seasonal variation of histopathological and histochemical markers of PAH exposure in blue mussel (*Mytilus edulis* L.). *Mar. Environ. Res.* **2011**, *71*, 213–217. [CrossRef] [PubMed]
57. Utvik, T.I.R.; Johnsen, S. Bioavailability of polycyclic aromatic hydrocarbons in the North Sea. *Environ. Sci. Technol.* **1999**, *33*, 1963–1969. [CrossRef]
58. Ossai, C.J.; Iwegbue, C.M.A.; Tesi, G.O.; Olisah, C.; Egobueze, F.E.; Nwajei, G.E.; Martincigh, B.S. Distribution, sources and exposure risk of polycyclic aromatic hydrocarbons in soils, and indoor and outdoor dust from Port Harcourt city, Nigeria. *Environ. Sci. Process. Impacts* **2021**, *23*, 1328–1350. [CrossRef] [PubMed]
59. Famiyeh, L.; Xu, H.; Chen, K.; Tang, Y.; Ji, D.; Xiao, H.; Tong, L.; Jia, C.; Guo, Q.; He, J. Breathing in danger: Unveiling the link between human exposure to outdoor $PM_{2.5}$-bound polycyclic aromatic hydrocarbons and lung cancer risk in an urban residential area of China. *Sci. Total Environ.* **2024**, *907*, 167762. [CrossRef]
60. Zhao, L.; Lyu, C.; Li, Y. Analysis of factors influencing plant–microbe combined remediation of soil contaminated by polycyclic aromatic hydrocarbons. *Sustainability* **2021**, *13*, 10695. [CrossRef]
61. Zhao, Q.; Ji, M.; Li, R.; Ren, Z.J. Long-term performance of sediment microbial fuel cells with multiple anodes. *Bioresour. Technol.* **2017**, *237*, 178–185. [CrossRef]
62. Kumar, M.; Bolan, N.S.; Hoang, S.A.; Sawarkar, A.D.; Jasemizad, T.; Gao, B.; Keerthanan, S.; Padhye, L.P.; Singh, L.; Kumar, S.; et al. Remediation of soils and sediments polluted with polycyclic aromatic hydrocarbons: To immobilize, mobilize, or degrade? *J. Hazard. Mater.* **2021**, *420*, 126534. [CrossRef]

63. Ravindra, K.; Sokhi, R.; Van Grieken, R. Atmospheric polycyclic aromatic hydrocarbons: Source attribution, emission factors and regulation. *Atmos. Environ.* **2008**, *42*, 2895–2921. [CrossRef]
64. Tiwari, M.; Sahu, S.K.; Pandit, G.G. Distribution of PAHs in different compartment of creek ecosystem: Ecotoxicological concern and human health risk. *Environ. Toxicol. Pharmacol.* **2017**, *50*, 58–66. [CrossRef]
65. Sherafatmand, M.; Ng, H.Y. Using sediment microbial fuel cells (SMFCs) for bioremediation of polycyclic aromatic hydrocarbons (PAHs). *Bioresour. Technol.* **2015**, *195*, 122–130. [CrossRef]
66. Ghosal, D.; Ghosh, S.; Dutta, T.K.; Ahn, Y. Current state of knowledge in microbial degradation of polycyclic aromatic hydrocarbons (PAHs): A review. *Front. Microbiol.* **2016**, *7*, 1369. [CrossRef]
67. Wang, J.; Xu, H.; Guinot, B.; Li, L.; Ho, S.S.H.; Liu, S.; Li, X.; Cao, J. Concentrations, sources and health effects of parent, oxygenated- and nitrated- polycyclic aromatic hydrocarbons (PAHs) in middle-school air in Xi'an, China. *Atmos. Res.* **2017**, *192*, 1–10. [CrossRef]
68. Lin, W.; Fan, F.; Xu, G.; Gong, K.; Cheng, X.; Yuan, X.; Zhang, C.; Gao, Y.; Wang, S.; Ng, H.Y.; et al. Microbial community assembly responses to polycyclic aromatic hydrocarbon contamination across water and sediment habitats in the Pearl River Estuary. *J. Hazard. Mater.* **2023**, *457*, 131762. [CrossRef] [PubMed]
69. Liu, Y.; Yan, C.; Ding, X.; Wang, X.; Fu, Q.; Zhao, Q.; Zhang, Y.; Duan, Y.; Qiu, X.; Zheng, M. Sources and spatial distribution of particulate polycyclic aromatic hydrocarbons in Shanghai, China. *Sci. Total Environ.* **2017**, *584*, 307–317. [CrossRef]
70. Kumari, A.; Upadhyay, V.; Kumar, S. A critical insight into occurrence and fate of polycyclic aromatic hydrocarbons and their green remediation approaches. *Chemosphere* **2023**, *329*, 138579. [CrossRef]
71. Bai, L.; Li, C. Investigation of indoor polycyclic aromatic hydrocarbons (PAHs) in rural northeast China: Pollution characteristics, source analysis, and health assessment. *Buildings* **2022**, *12*, 153. [CrossRef]
72. White, A.J.; Bradshaw, P.T.; Herring, A.H.; Teitelbaum, S.L.; Beyea, J.; Stellman, S.D.; Steck, S.E.; Mordukhovich, I.; Eng, S.M.; Engel, L.S.; et al. Exposure to multiple sources of polycyclic aromatic hydrocarbons and breast cancer incidence. *Environ. Int.* **2016**, *89–90*, 185–192. [CrossRef] [PubMed]
73. Phoungthong, K.; Tekasakul, S.; Tekasakul, P.; Furuuchi, M. Comparison of particulate matter and polycyclic aromatic hydrocarbons in emissions from IDI-turbo diesel engine fueled by palm oil–diesel blends during long-term usage. *Atmos. Pollut. Res.* **2017**, *8*, 344–350. [CrossRef]
74. Seo, J.S.; Keum, Y.S.; Li, Q.X. Bacterial degradation of aromatic compounds. *Int. J. Environ. Res. Public Health* **2009**, *6*, 278–309. [CrossRef] [PubMed]
75. Khalili, N.R.; Scheff, P.A.; Holsen, T.M. PAH source fingerprints for coke ovens, diesel and, gasoline engines, highway tunnels, and wood combustion emissions. *Atmos. Environ.* **1995**, *29*, 533–542. [CrossRef]
76. Peng, N.; Li, Y.; Liu, Z.; Liu, T.; Gai, C. Emission, distribution and toxicity of polycyclic aromatic hydrocarbons (PAHs) during municipal solid waste (MSW) and coal co-combustion. *Sci. Total Environ.* **2016**, *565*, 1201–1207. [CrossRef]
77. Miura, K.; Shimada, K.; Sugiyama, T.; Sato, K.; Takami, A.; Chan, C.K.; Kim, I.S.; Kim, Y.P.; Lin, N.-H.; Hatakeyama, S. Seasonal and annual changes in PAH concentrations in a remote site in the Pacific Ocean. *Sci. Rep.* **2019**, *9*, 12591. [CrossRef] [PubMed]
78. Besis, A.; Tsolakidou, A.; Balla, D.; Samara, C.; Voutsa, D.; Pantazaki, A.; Choli-Papadopoulou, T.; Lialiaris, T.S. Toxic organic substances and marker compounds in size-segregated urban particulate matter—Implications for involvement in the in vitro bioactivity of the extractable organic matter. *Environ. Pollut.* **2017**, *230*, 758–774. [CrossRef] [PubMed]
79. Hong, W.-J.; Li, Y.-F.; Li, W.-L.; Jia, H.; Minh, N.H.; Sinha, R.K.; Moon, H.-B.; Nakata, H.; Chi, K.H.; Kannan, K.; et al. Soil concentrations and soil-air exchange of polycyclic aromatic hydrocarbons in five Asian countries. *Sci. Total Environ.* **2020**, *711*, 135223. [CrossRef]
80. Wu, X.; Wang, Y.; Zhang, Q.; Zhao, H.; Yang, Y.; Zhang, Y.; Xie, Q.; Chen, J. Seasonal variation, air-water exchange, and multivariate source apportionment of polycyclic aromatic hydrocarbons in the coastal area of Dalian, China. *Environ. Pollut.* **2019**, *244*, 405–413. [CrossRef]
81. Jesus, F.; Pereira, J.L.; Campos, I.; Santos, M.; Ré, A.; Keizer, J.; Nogueira, A.; Gonçalves, F.J.M.; Abrantes, N.; Serpa, D. A review on polycyclic aromatic hydrocarbons distribution in freshwater ecosystems and their toxicity to benthic fauna. *Sci. Total Environ.* **2022**, *820*, 153282. [CrossRef]
82. Pannu, J.K.; Singh, A.; Ward, O.P. Influence of peanut oil on microbial degradation of polycyclic aromatic hydrocarbons. *Can. J. Microbiol.* **2003**, *49*, 508–513. [CrossRef]
83. Zhang, L.; Wang, Y.; Tan, F.; Yang, Y.; Wu, X.; Wang, W.; Liu, D. Tidal variability of polycyclic aromatic hydrocarbons and organophosphate esters in the coastal seawater of Dalian, China. *Sci. Total Environ.* **2020**, *708*, 134441. [CrossRef] [PubMed]
84. Damir, N.A.; Coatu, V.; Pantea, E.D.; Galaichi, M.; Botez, E.; Birghila, S. Assessment of polycyclic aromatic hydrocarbons content in marine organisms of commercial interest from the Romanian Black Sea Coast. *Polycycl. Aromat. Compd.* **2022**, *42*, 7595–7606. [CrossRef]
85. Sun, Y.; Chen, J.; Qin, W.; Yu, Q.; Xin, K.; Ai, J.; Huang, H.; Liu, X. Gas-PM2.5 partitioning, health risks, and sources of atmospheric PAHs in a northern China city: Impact of domestic heating. *Environ. Pollut.* **2022**, *313*, 120156. [CrossRef] [PubMed]

86. Wang, Z.; Ren, P.; Sun, Y.; Ma, X.; Liu, X.; Na, G.; Yao, Z. Gas/particle partitioning of polycyclic aromatic hydrocarbons in coastal atmosphere of the north Yellow Sea, China. *Environ. Sci. Pollut. Res.* **2013**, *20*, 5753–5763. [CrossRef]
87. Shen, G.; Wang, W.; Yang, Y.; Ding, J.; Xue, M.; Min, Y.; Zhu, C.; Shen, H.; Li, W.; Wang, B.; et al. Emissions of PAHs from indoor crop residue burning in a typical rural stove: Emission factors, size distributions, and gas−particle partitioning. *Environ. Sci. Technol.* **2011**, *45*, 1206–1212. [CrossRef]
88. Samanta, S.K.; Singh, O.V.; Jain, R.K. Polycyclic aromatic hydrocarbons: Environmental pollution and bioremediation. *Trends Biotechnol.* **2002**, *20*, 243–248. [CrossRef] [PubMed]
89. Humel, S.; Führer, B.; Svetitsch, M.; Mayer, P.; Loibner, A.P. Targeting sorbed PAHs in historically contaminated soil—Can laccase mediator systems or Fenton's reagent remove inaccessible PAHs? *J. Hazard. Mater.* **2023**, *443*, 130286. [CrossRef]
90. Yu, L.; Duan, L.; Naidu, R.; Semple, K.T. Abiotic factors controlling bioavailability and bioaccessibility of polycyclic aromatic hydrocarbons in soil: Putting together a bigger picture. *Sci. Total Environ.* **2018**, *613–614*, 1140–1153. [CrossRef] [PubMed]
91. Gholami, S.; Behnami, A.; Hesami Arani, M.; Rezaei Kalantary, R. Impact of humic substances on the bioremediation of polycyclic aromatic hydrocarbons in contaminated soils and sediments: A review. *Environ. Chem. Lett.* **2024**, *22*, 889–918. [CrossRef]
92. Li, B.; Zhao, L.; Zhong, S.; An, R.; Ma, R.; Xu, X.; Chen, Q. Occurrence, distribution and risk assessment of polycyclic aromatic hydrocarbons in soils around main water source areas of Beijing, China. *Environ. Geochem. Health* **2023**, *45*, 7569–7584. [CrossRef]
93. Wu, J.; Yang, G.; Chen, H.; Zhai, Y.; Teng, Y.; Li, J.; Chen, R. Source apportionment and source specific health risk assessment of HMs and PAHs in soils with an integrated framework in a typical cold agricultural region in China. *Sci. Total Environ.* **2023**, *904*, 167337. [CrossRef] [PubMed]
94. Madondo, N.I.; Rathilal, S.; Bakare, B.F.; Tetteh, E.K. Application of bioelectrochemical systems and anaerobic additives in wastewater treatment: A conceptual review. *Int. J. Mol. Sci.* **2023**, *24*, 4753. [CrossRef]
95. Addagada, L.; Goel, M.; Shahid, M.K.; Prabhu, S.V.; Chand, S.; Sahoo, N.K.; Rout, P.R. Tricks and tracks in resource recovery from wastewater using bio-electrochemical systems (BES): A systematic review on recent advancements and future directions. *J. Water Process Eng.* **2023**, *56*, 104580. [CrossRef]
96. Gebregiorgis Ambaye, T.; Vaccari, M.; Franzetti, A.; Prasad, S.; Formicola, F.; Rosatelli, A.; Hassani, A.; Aminabhavi, T.M.; Rtimi, S. Microbial electrochemical bioremediation of petroleum hydrocarbons (PHCs) pollution: Recent advances and outlook. *Chem. Eng. J.* **2023**, *452*, 139372. [CrossRef]
97. Venkata Mohan, S.; Velvizhi, G.; Vamshi Krishna, K.; Lenin Babu, M. Microbial catalyzed electrochemical systems: A bio-factory with multi-facet applications. *Bioresour. Technol.* **2014**, *165*, 355–364. [CrossRef] [PubMed]
98. Rozendal, R.A.; Hamelers, H.V.M.; Euverink, G.J.W.; Metz, S.J.; Buisman, C.J.N. Principle and perspectives of hydrogen production through biocatalyzed electrolysis. *Int. J. Hydrogen Energy* **2006**, *31*, 1632–1640. [CrossRef]
99. Zhang, X.; Li, X.; Zhao, X.; Li, Y. Factors affecting the efficiency of a bioelectrochemical system: A review. *RSC Adv.* **2019**, *9*, 19748–19761. [CrossRef] [PubMed]
100. Lin, R.; Xie, L.; Zheng, X.; Patience, D.; Duan, X. Advances and challenges in biocathode microbial electrolysis cells for chlorinated organic compounds degradation from electroactive perspectives. *Sci. Total Environ.* **2023**, *905*, 167141. [CrossRef]
101. Arun, J.; SundarRajan, P.; Grace Pavithra, K.; Priyadharsini, P.; Shyam, S.; Goutham, R.; Hoang Le, Q.; Pugazhendhi, A. New insights into microbial electrolysis cells (MEC) and microbial fuel cells (MFC) for simultaneous wastewater treatment and green fuel (hydrogen) generation. *Fuel* **2024**, *355*, 129530. [CrossRef]
102. Guo, Z.; Liu, W.; Yang, C.; Gao, L.; Thangavel, S.; Wang, L.; He, Z.; Cai, W.; Wang, A. Computational and experimental analysis of organic degradation positively regulated by bioelectrochemistry in an anaerobic bioreactor system. *Water Res.* **2017**, *125*, 170–179. [CrossRef]
103. Wang, H.; Zeng, S.; Du, H.; Xie, H.; Igarashi, Y.; Luo, F. The changes of microbial abundance and functional genes in bioelectrochemistry at 15 °C. *J. Environ. Chem. Eng.* **2022**, *10*, 106996. [CrossRef]
104. Fernández Verdejo, D.; Cortés, P.; Guisasola, A.; Blánquez, P.; Marco Urrea, E. Bioelectrochemically-assisted degradation of chloroform by a co-culture of Dehalobacter and Dehalobacterium. *Environ. Sci. Ecotechnology* **2022**, *12*, 100199. [CrossRef] [PubMed]
105. Sharma, A.; Hussain Mehdi, S.E.; Pandit, S.; Eun-Oh, S.; Natarajan, V. Factors affecting hydrogen production in microbial electrolysis cell (MEC): A review. *Int. J. Hydrogen Energy* **2024**, *61*, 1473–1484. [CrossRef]
106. Ramírez Vargas, C.A.; Prado, A.; Arias, C.A.; Carvalho, P.N.; Esteve Núñez, A.; Brix, H. Microbial electrochemical technologies for wastewater treatment: Principles and evolution from microbial fuel cells to bioelectrochemical-based constructed wetlands. *Water* **2018**, *10*, 1128. [CrossRef]
107. Wang, H.; Xing, L.; Zhang, H.; Gui, C.; Jin, S.; Lin, H.; Li, Q.; Cheng, C. Key factors to enhance soil remediation by bioelectrochemical systems (BESs): A review. *Chem. Eng. J.* **2021**, *419*, 129600. [CrossRef]
108. Li, T.; Li, R.; Zhou, Q. The application and progress of bioelectrochemical systems (BESs) in soil remediation: A review. *Green Energy Environ.* **2021**, *6*, 50–65. [CrossRef]

109. Kronenberg, M.; Trably, E.; Bernet, N.; Patureau, D. Biodegradation of polycyclic aromatic hydrocarbons: Using microbial bioelectrochemical systems to overcome an impasse. *Environ. Pollut.* **2017**, *231*, 509–523. [CrossRef]
110. Sharma, M.; Nandy, A.; Taylor, N.; Venkatesan, S.V.; Kollath, V.O.; Karan, K.; Thangadurai, V.; Tsesmetzis, N.; Gieg, L.M. Bioelectrochemical remediation of phenanthrene in a microbial fuel cell using an anaerobic consortium enriched from a hydrocarbon-contaminated site. *J. Hazard. Mater.* **2020**, *389*, 121845. [CrossRef]
111. Zhou, Y.; Zou, Q.; Fan, M.; Xu, Y.; Chen, Y. Highly efficient anaerobic co-degradation of complex persistent polycyclic aromatic hydrocarbons by a bioelectrochemical system. *J. Hazard. Mater.* **2020**, *381*, 120945. [CrossRef]
112. Hua, T.; Wang, H.; Li, S.; Chen, P.; Li, F.; Wang, W. Electrochemical performance and response of bacterial community during phenanthrene degradation in single-chamber air-cathode microbial fuel cells. *Environ. Sci. Pollut. Res.* **2021**, *28*, 22705–22715. [CrossRef]
113. Wang, H.; Chen, P.; Zhang, S.; Jiang, J.; Hua, T.; Li, F. Degradation of pyrene using single-chamber air-cathode microbial fuel cells: Electrochemical parameters and bacterial community changes. *Sci. Total Environ.* **2022**, *804*, 150153. [CrossRef]
114. Huang, S.; Xia, J.; Chen, D.; Li, N.; Xu, Q.; Li, H.; He, J.; Lu, J. Cu, Co embedded N-enriched mesoporous carbon cathode catalyst for the efficient bioelectrochemical removal of phenanthrene in microbial fuel cell. *Appl. Surf. Sci.* **2022**, *599*, 153759. [CrossRef]
115. Liang, Y.; Ji, M.; Zhai, H.; Wang, R. Removal of benzo[a]pyrene from soil in a novel permeable electroactive well system: Optimal integration of filtration, adsorption and bioelectrochemical degradation. *Sep. Purif. Technol.* **2020**, *252*, 117458. [CrossRef]
116. Yu, B.; Tian, J.; Feng, L. Remediation of PAH polluted soils using a soil microbial fuel cell: Influence of electrode interval and role of microbial community. *J. Hazard. Mater.* **2017**, *336*, 110–118. [CrossRef] [PubMed]
117. Jiang, X.; Gao, X.; Yang, K.; Hu, J.; Cao, X.; Sakamaki, T.; Li, X. Promotion and mechanism of defective hematite on the power generation and phenanthrene degradation of soil microbial fuel cells. *Environ. Eng. Res.* **2025**, *30*, 240135. [CrossRef]
118. Chen, L.; Zheng, X.; Zhang, K.; Wu, B.; Pei, X.; Chen, W.; Wei, X.; Luo, Z.; Li, Y.; Zhang, Z. Sustained-release nitrate combined with microbial fuel cell: A novel strategy for PAHs and odor removal from sediment. *J. Hazard. Mater.* **2023**, *455*, 131610. [CrossRef] [PubMed]
119. Cao, X.; Li, X.; Wang, H.; Zhang, S.; Zhang, H.; Sakamaki, T.; Li, X. The promotion of the polycyclic aromatic hydrocarbons degradation mechanism by humic acid as electron mediator in a sediment microbial electrochemical system. *Bioresour. Technol.* **2024**, *404*, 130909. [CrossRef]
120. Ding, P.; Wu, P.; Jie, Z.; Cui, M.-H.; Liu, H. Damage of anodic biofilms by high salinity deteriorates PAHs degradation in single-chamber microbial electrolysis cell reactor. *Sci. Total Environ.* **2021**, *777*, 145752. [CrossRef] [PubMed]
121. Naderi, A.; Rezaee, A.; Pasalari, H.; Esrafili, A.; Kalantary, R.R. Bio-electrical stimulation process on degradation of Phenanthrene from aqueous solution using a novel anode modified with carbon cloth: Operational performance, microbial activity and energy. *Ecotoxicol. Environ. Saf.* **2022**, *239*, 113654. [CrossRef] [PubMed]
122. Mohanakrishna, G.; Al Raoush, R.I.; Abu Reesh, I.M. Sewage enhanced bioelectrochemical degradation of petroleum hydrocarbons in soil environment through bioelectro-stimulation. *Biotechnol. Rep.* **2020**, *27*, e00478. [CrossRef] [PubMed]
123. Cai, X.; Luo, X.; Yuan, Y.; Li, J.; Yu, Z.; Zhou, S. Stimulation of phenanthrene and biphenyl degradation by biochar-conducted long distance electron transfer in soil bioelectrochemical systems. *Sci. Total Environ.* **2021**, *797*, 149124. [CrossRef] [PubMed]
124. Yan, F.; Reible, D. Electro-bioremediation of contaminated sediment by electrode enhanced capping. *J. Environ. Manag.* **2015**, *155*, 154–161. [CrossRef]
125. Zhao, L.; Deng, J.; Hou, H.; Li, J.; Yang, Y. Investigation of PAH and oil degradation along with electricity generation in soil using an enhanced plant-microbial fuel cell. *J. Clean. Prod.* **2019**, *221*, 678–683. [CrossRef]
126. Huo, M.; Wang, B.; Zhang, C.; Ding, S.; Yuan, H.; Liang, Z.; Qi, J.; Chen, M.; Xu, Y.; Zhang, W.; et al. 2D metal–organic framework derived CuCo alloy nanoparticles encapsulated by nitrogen-doped carbonaceous nanoleaves for efficient bifunctional oxygen electrocatalyst and Zinc–air batteries. *Chem. –A Eur. J.* **2019**, *25*, 12780–12788. [CrossRef] [PubMed]
127. Wang, Y.; Zhong, K.; Li, H.; Dai, Y.; Zhang, H.; Zuo, J.; Yan, J.; Xiao, T.; Liu, X.; Lu, Y.; et al. Bimetallic hybrids modified with carbon nanotubes as cathode catalysts for microbial fuel cell: Effective oxygen reduction catalysis and inhibition of biofilm formation. *J. Power Sources* **2021**, *485*, 229273. [CrossRef]
128. Luo, H.; Jin, S.; Fallgren, P.H.; Park, H.J.; Johnson, P.A. A novel laccase-catalyzed cathode for microbial fuel cells. *Chem. Eng. J.* **2010**, *165*, 524–528. [CrossRef]
129. Lai, C.; Wu, C.; Meng, C.; Lin, C. Decolorization of azo dye and generation of electricity by microbial fuel cell with laccase-producing white-rot fungus on cathode. *Appl. Energy* **2017**, *188*, 392–398. [CrossRef]
130. Wu, C.; Liu, X.; Li, W.; Sheng, G.; Zang, G.; Cheng, Y.; Shen, N.; Yang, Y.; Yu, H. A white-rot fungus is used as a biocathode to improve electricity production of a microbial fuel cell. *Appl. Energy* **2012**, *98*, 594–596. [CrossRef]
131. Chupungars, K.; Rerngsamran, P.; Thaniyavarn, S. Polycyclic aromatic hydrocarbons degradation by Agrocybe sp. CU-43 and its fluorene transformation. *Int. Biodeterior. Biodegrad.* **2009**, *63*, 93–99. [CrossRef]

132. Liang, Y.; Zhai, H.; Liu, B.; Ji, M.; Li, J. Carbon nanomaterial-modified graphite felt as an anode enhanced the power production and polycyclic aromatic hydrocarbon removal in sediment microbial fuel cells. *Sci. Total Environ.* **2020**, *713*, 136483. [CrossRef] [PubMed]
133. Adelaja, O.; Keshavarz, T.; Kyazze, G. Enhanced biodegradation of phenanthrene using different inoculum types in a microbial fuel cell. *Eng. Life Sci.* **2014**, *14*, 218–228. [CrossRef]
134. Venkidusamy, K.; Megharaj, M.; Marzorati, M.; Lockington, R.; Naidu, R. Enhanced removal of petroleum hydrocarbons using a bioelectrochemical remediation system with pre-cultured anodes. *Sci. Total Environ.* **2016**, *539*, 61–69. [CrossRef]
135. Ambaye, T.G.; Formicola, F.; Sbaffoni, S.; Franzetti, A.; Vaccari, M. Insights into rhamnolipid amendment towards enhancing microbial electrochemical treatment of petroleum hydrocarbon contaminated soil. *Chemosphere* **2022**, *307*, 136126. [CrossRef] [PubMed]
136. Gao, Y.; Yuan, X.; Lin, X.; Sun, B.; Zhao, Z. Low-molecular-weight organic acids enhance the release of bound PAH residues in soils. *Soil Tillage Res.* **2015**, *145*, 103–110. [CrossRef]
137. Sivaram, A.K.; Logeshwaran, P.; Lockington, R.; Naidu, R.; Megharaj, M. Low molecular weight organic acids enhance the high molecular weight polycyclic aromatic hydrocarbons degradation by bacteria. *Chemosphere* **2019**, *222*, 132–140. [CrossRef] [PubMed]
138. Zhang, H.; Chao, B.; Gao, X.; Cao, X.; Li, X. Effect of starch-derived organic acids on the removal of polycyclic aromatic hydrocarbons in an aquaculture-sediment microbial fuel cell. *J. Environ. Manag.* **2022**, *311*, 114783. [CrossRef]
139. Zafar, Z.; Naz, S.; Malik, N.H.; Ahmed, F.; Ali, N. Bioelectrogeneic performance of air-cathode microbial fuel cells with diesel contaminants. *Fuel* **2024**, *355*, 129407. [CrossRef]
140. Aleman Gama, E.; Cornejo Martell, A.J.; Kamaraj, S.K.; Juárez, K.; Silva Martínez, S.; Alvarez Gallegos, A. Boosting power generation by sediment microbial fuel cell in oil-contaminated sediment amended with gasoline/kerosene. *J. Electrochem. Sci. Technol.* **2022**, *13*, 308–320. [CrossRef]
141. Bezza, F.A.; Nkhalambayausi Chirwa, E.M. Biosurfactant-enhanced bioremediation of aged polycyclic aromatic hydrocarbons (PAHs) in creosote contaminated soil. *Chemosphere* **2016**, *144*, 635–644. [CrossRef]
142. Gomaa, O.M.; Selim, N.; Fathy, R.; Maghrawy, H.H.; Gamal, M.; El Kareem, H.A.; Kyazze, G.; Keshavarz, T. Characterization of a biosurfactant producing electroactive Bacillus sp. for enhanced Microbial Fuel Cell dye decolourisation. *Enzym. Microb. Technol.* **2021**, *147*, 109767. [CrossRef] [PubMed]
143. Phulpoto, I.A.; Khan, S.; Qazi, M.A. Insights into rhamnolipid-assisted bioelectrochemical system for remediating soil pollution: A promising green approach towards the sustainable environment. *Int. Biodeterior. Biodegrad.* **2024**, *191*, 105808. [CrossRef]
144. Curiel Alegre, S.; de la Fuente Vivas, D.; Khan, A.H.A.; García Tojal, J.; Velasco Arroyo, B.; Rumbo, C.; Soja, G.; Rad, C.; Barros, R. Unveiling the capacity of bioaugmentation application, in comparison with biochar and rhamnolipid for TPHs degradation in aged hydrocarbons polluted soil. *Environ. Res.* **2024**, *252*, 118880. [CrossRef] [PubMed]
145. Stern, N.; Mejia, J.; He, S.; Yang, Y.; Ginder-Vogel, M.; Roden, E.E. Dual role of humic substances as electron donor and shuttle for dissimilatory iron reduction. *Environ. Sci. Technol.* **2018**, *52*, 5691–5699. [CrossRef]
146. Adelaja, O.; Keshavarz, T.; Kyazze, G. The effect of salinity, redox mediators and temperature on anaerobic biodegradation of petroleum hydrocarbons in microbial fuel cells. *J. Hazard. Mater.* **2015**, *283*, 211–217. [CrossRef]
147. Lu, J.; Li, M.; Tan, J.; He, M.; Wu, H.; Kang, Y.; Hu, Z.; Zhang, J.; Guo, Z. Distribution, sources, ecological risk and microbial response of polycyclic aromatic hydrocarbons in Qingdao bays, China. *Environ. Pollut.* **2023**, *338*, 122687. [CrossRef]
148. Logeshwaran, P.; Megharaj, M.; Chadalavada, S.; Bowman, M.; Naidu, R. Petroleum hydrocarbons (PH) in groundwater aquifers: An overview of environmental fate, toxicity, microbial degradation and risk-based remediation approaches. *Environ. Technol. Innov.* **2018**, *10*, 175–193. [CrossRef]
149. Elyamine, A.M.; Kan, J.; Meng, S.; Tao, P.; Wang, H.; Hu, Z. Aerobic and anaerobic bacterial and fungal degradation of pyrene: Mechanism pathway including biochemical reaction and catabolic genes. *Int. J. Mol. Sci.* **2021**, *22*, 8202. [CrossRef]
150. Alvarez Benítez, L.; Silva Martínez, S.; Hernandez Perez, A.; Kamaraj, S.K.; Abbas, S.Z.; Alvarez Gallegos, A. Quantification of internal resistance contributions of sediment microbial fuel cells using petroleum-contaminated sediment enriched with kerosene. *Catalysts* **2022**, *12*, 871. [CrossRef]
151. Wang, W.; Wang, L.; Shao, Z. Polycyclic aromatic hydrocarbon (PAH) degradation pathways of the obligate marine PAH degrader cycloclasticus sp. Strain P1. *Appl. Environ. Microbiol.* **2018**, *84*, e01261-18. [CrossRef] [PubMed]
152. Kümmel, S.; Herbst, F.-A.; Bahr, A.; Duarte, M.; Pieper, D.H.; Jehmlich, N.; Seifert, J.; von Bergen, M.; Bombach, P.; Richnow, H.H.; et al. Anaerobic naphthalene degradation by sulfate-reducing Desulfobacteraceae from various anoxic aquifers. *FEMS Microbiol. Ecol.* **2015**, *91*, fiv006. [CrossRef]
153. Zhang, L.; Qiu, X.; Huang, L.; Xu, J.; Wang, W.; Li, Z.; Xu, P.; Tang, H. Microbial degradation of multiple PAHs by a microbial consortium and its application on contaminated wastewater. *J. Hazard. Mater.* **2021**, *419*, 126524. [CrossRef]
154. Sun, Y.; Wang, K.; Chen, D.; Xu, Q.; Li, N.; Li, H.; Lu, J. Activation of persulfate by highly dispersed FeCo bimetallic alloy for in-situ remediation of polycyclic aromatic hydrocarbon-contaminated soil. *Sep. Purif. Technol.* **2023**, *317*, 123781. [CrossRef]

155. Wasfi, R.; Moussa, H.A.; Bakr, R.O.; Abdeltawab, N.F.; Megahed, S.A. Anaerobic biodegradation of anthracene by oral Firmicutes isolates from smokers and its potential pathway. *Int. Biodeterior. Biodegrad.* **2023**, *180*, 105598. [CrossRef]
156. Wen, L.; Huang, Y.; Wang, W.; Zhang, L.; Xu, J.; Li, Z.; Xu, P.; Tang, H. A novel Diaphorobacter sp. strain isolated from saponification wastewater shows highly efficient phenanthrene degradation. *Environ. Res.* **2022**, *214*, 114047. [CrossRef]
157. Tsai, J.-C.; Kumar, M.; Lin, J.-G. Anaerobic biotransformation of fluorene and phenanthrene by sulfate-reducing bacteria and identification of biotransformation pathway. *J. Hazard. Mater.* **2009**, *164*, 847–855. [CrossRef]
158. Hadibarata, T.; Kristanti, R.A. Potential of a white-rot fungus Pleurotus eryngii F032 for degradation and transformation of fluorene. *Fungal Biol.* **2014**, *118*, 222–227. [CrossRef]
159. Reddy, P.V.; Karegoudar, T.B.; Monisha, T.R.; Nayak, A.S. Catabolism of fluorene through 2,3-dihydroxy indanone in Paenibacillus sp. PRNK-6. *Int. Biodeterior. Biodegrad.* **2017**, *123*, 156–163. [CrossRef]
160. Desta, M.; Zhang, L.; Wang, W.; Xu, P.; Tang, H. Molecular mechanisms and biochemical analysis of fluorene degradation by the Pseudomonas sp. SMT-1 strain. *3 Biotech* **2021**, *11*, 416. [CrossRef] [PubMed]
161. Goveas, L.C.; Selvaraj, R.; Kumar, P.S.; Vinayagam, R.; Sajankila, S.P. Biodegradation kinetics and metabolism of Benzo(a)fluorene by Pseudomonas strains isolated from refinery effluent. *Chemosphere* **2022**, *307*, 136041. [CrossRef]
162. Dvořák, P.; Nikel, P.I.; Damborský, J.; de Lorenzo, V. Bioremediation 3.0: Engineering pollutant-removing bacteria in the times of systemic biology. *Biotechnol. Adv.* **2017**, *35*, 845–866. [CrossRef] [PubMed]
163. Zheng, T.; Li, J.; Ji, Y.; Zhang, W.; Fang, Y.; Xin, F.; Dong, W.; Wei, P.; Ma, J.; Jiang, M. Progress and prospects of bioelectrochemical systems: Electron transfer and its applications in the microbial metabolism. *Front. Bioeng. Biotechnol.* **2020**, *8*, 10. [CrossRef] [PubMed]
164. Afsharian, Y.P.; Rahimnejad, M. Functional dynamics of microbial communities in bioelectrochemical systems: The importance of eco-electrogenic treatment of complex substrates. *Curr. Opin. Electrochem.* **2022**, *31*, 100816. [CrossRef]
165. Martinez, C.M.; Alvarez, L.H. Application of redox mediators in bioelectrochemical systems. *Biotechnol. Adv.* **2018**, *36*, 1412–1423. [CrossRef] [PubMed]
166. Xue, W.; Zhou, Q.; Li, F. Bacterial community changes and antibiotic resistance gene quantification in microbial electrolysis cells during long-term sulfamethoxazole treatment. *Bioresour. Technol.* **2019**, *294*, 122170. [CrossRef]
167. Kokko, M.; Epple, S.; Gescher, J.; Kerzenmacher, S. Effects of wastewater constituents and operational conditions on the composition and dynamics of anodic microbial communities in bioelectrochemical systems. *Bioresour. Technol.* **2018**, *258*, 376–389. [CrossRef] [PubMed]
168. Zhang, Z.; Zhang, K.; Ouyang, H.; Li, M.K.K.; Luo, Z.; Li, Y.; Chen, C.; Yang, X.; Shao, Z.; Yan, D.Y.S. Simultaneous PAHs degradation, odour mitigation and energy harvesting by sediment microbial fuel cell coupled with nitrate-induced biostimulation. *J. Environ. Manag.* **2021**, *284*, 112045. [CrossRef]
169. Yuan, Y.; Zhou, L.; Hou, R.; Wang, Y.; Zhou, S. Centimeter-long microbial electron transport for bioremediation applications. *Trends Biotechnol.* **2021**, *39*, 181–193. [CrossRef] [PubMed]
170. Hamdan, H.Z.; Salam, D.A.; Hari, A.R.; Semerjian, L.; Saikaly, P. Assessment of the performance of SMFCs in the bioremediation of PAHs in contaminated marine sediments under different redox conditions and analysis of the associated microbial communities. *Sci. Total Environ.* **2017**, *575*, 1453–1461. [CrossRef]