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Removal of antibiotics from wastewater and its problematic effects on microbial communities by bioelectrochemical Technology: Current knowledge and future perspectives

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ABSTRACT

In this review, antibiotics are considered an emerging pollutant that has drawn worldwide attention in recent years. Therefore, the effective removal of antibiotic contaminants has become a hot issue in the field of environmental research. Most antibiotics applied to humans eventually enter municipal Wastewater Treatment Plants (WWTPs), because there are no appropriate commercially available pretreatment techniques. However, increasing anthropogenic activities, the high demand for animal-protein in developing countries as a nutritional alternative, and the extensive usage of antibiotics are mainly responsible for the persistence of antibiotic pollutants. One of the serious concerns regarding the presence of antibiotics in water and their potential role in exacerbating the emergence of antibiotics-resistance bacteria (ARB) and antibiotics-resistance genes (ARGs). In recent years, bioelectrochemical technologies are found promising for suppressing antibiotic contaminants through microbial metabolism and electrochemical redox reactions. Therefore, this review provides up-to-date insight research on bioelectrochemical systems (BESs), which improves the removal of the antibiotic in an efficient way. The focus of this review has been on the environmental sources of antibiotics, their health effects and possible degradation pathways, bacterial-antibiotics resistance mechanisms, and treatment of antibiotic-contained water using BES technology.

Keywords: Antibiotic resistance bacteria (ARB), Antibiotic resistance genes (ARGs), Bioelectrochemical System, Biodegradation, Microbial communities, Wastewater Treatment

1. Introduction

In the 20th century, various antibiotics were developed to combat bacterial infections [1]. As a result, bacteria respond to antibiotic resistance by adapting to an increase in antibiotic levels in the environment. This phenomenon leads to a high proportion of antibiotic residues in livestock, municipal [2], and other industrial wastewater [3-5]. Nowadays, humans face a rapidly cumulative number of antibiotic-resistant strains. However, increased human activities, the high demand for animal protein spreads in developing countries [6, 7], and the intensive use of antibiotics are the causes of the persistence of antibiotic pollutants in the environment [2, 8].

This contamination has raised more concerns because it could exacerbate the emergence of antibiotic resistant bacteria (ARB) and antibiotic resistant genes (ARGs) [9, 10]. However, this issue

raises serious concerns about public health [10, 11], as the ecological balance is threatened and continues to put pressure on the ARB and ARG. Also, ARB and ARG are moving globally on an unprecedented scale due to human activity, compared with the past billion years [2].

Moreover, the wastewater treatment plant only meets the existing standards; various emergent contaminants of pharmaceuticals and personal care products (PPCPs) [12] were released into the rivers and lakes without meeting satisfactory treatment [13, 14].

In recent years, different treatment methods have been carried out in terms of antibiotic residues, with significant removal efficiency. Advanced oxidation processes (AOPs) are of great concern due to their powerful removal efficiency [15, 16]. Some antibiotics were prohibitive and may produce sub-active toxic by-products [17]. The effectiveness of antibiotic removal in the adsorption



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process and membrane technology is satisfactory. But these techniques ultimately fail to degrade antibiotics and are significantly damaged by the presence of other organic pollutants. However, traditional biological methods were primarily noticed in terms of the high general performance of in-situ applications [18], but they are often time-consuming techniques [8, 19, 20].

Besides, bioelectrochemical technology combined with microbial fuel cells (MFCs) and microbial electrolytic cells (MECs), with bacterial metabolism and the electrochemical-redox reactions, were considered promising alternatives to degradation. Similarly, these systems have been used to improve the removal rate of antibiotics sulfamethoxazole (SMX) through redox reactions [21, 22]. Harnish *et al.* [23]. revealed the method of degradation of SMX 6.0 mgL^{-1} in a microbial electrolytic cell within seven days. Another study by Zhang *et al.* [24] observed that the removal rate of $200 \text{ }\mu\text{gL}^{-1}$ of sulfamethoxazole (SMX) in a three-dimensional biofilm electrode reactor (3D-BER). The system reached 85.5 - 90% removal with hydraulic retention time (HRT) of 40 h. While the multipurpose treatment of solid-phase mixtures (granular activated carbon), Song *et al.* [25] demonstrated that the removal performance of antibiotics in 3-Dimensional biofilm electrode reactors (3DBER), in solid phase environment and integration possibilities with other technologies, as shown in Table 1.

Furthermore, the bioelectrochemical technology has the following advantages; (a) eco-friendly: 3-amino-5-methylisoxazole, highly

bio-toxic sulfamethoxazole (SMX) intermediate that was further degraded by microbial fuel cells. In addition, the use of biocathodes also circumvents the production of highly toxic by-products of nitrofurazone (NFZ) and Chloramphenicol (CAP); (b) various antibiotic removal by constructed wetlands; (c) cost-effective: biocathodes and some external power inputs in BESs does not require additional reducing agents and therefore has high efficiency of antibiotic removals illustrated in Fig. 1. Bioelectrochemical technology shows both promising results [15, 26] on the excellent removal efficiency of a variety of other toxic-compounds in laboratories and full scales systems [27]. Municipal wastewater containing antibiotics had a critical problem due to improper wastewater treatment and a high level of toxicity in the environment [28, 29].

The aim of this review is to provide comprehensive details of the effects of bioelectrochemical technology on antibiotics pollutants removal with a broader perspective. Therefore, this systemic review primarily focuses on the mechanisms and degradation pathways of the various types of antibiotics and antibiotic-resistant genes. The special effects on the antibiotic-resistant bacteria and antibiotic-resistant genes and possible biodegradation process were also discussed. The present challenges and future scope of research on the latest technologies for the effective removal of antibiotics have also been discussed.

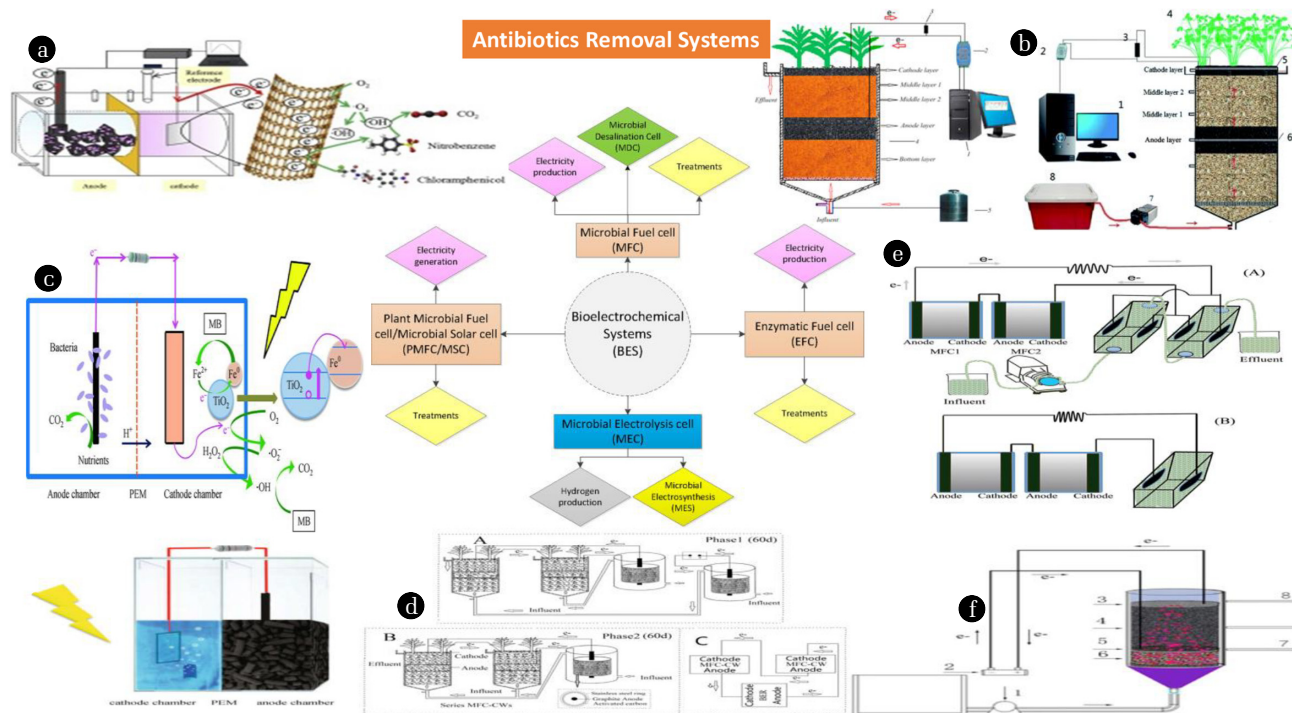


Fig. 1. Representation Illustrations of the different treatment processes of antibiotics removal and the main grouping of bioelectrochemical systems. (a) Advanced oxidation process coupled with BES. Reproduced with approval from references [30]. (b) Constructed wetlands coupled with a microbial fuel cell. Reproduced with approval from references [31]. (c) BES-AOP-coupled with advanced oxidation process with the cathode (FeO/TiO_2), and (B): design of the MFC-PEC System. Reproduced with approval from references [32]. (d) The coupled Constructed wetland with a biofilm electrode reactor system (CW-BES-MFCs). Reproduced with approval from references [33]. (e) A Configuration of microbial fuel cell (MFC-sorption) System. Reproduced with approval from references [34]. (f) 3D-BER: Three-dimensional biofilm electrode reactors. Reproduced with the approval from references [24].

Table 1. Previous Scientific Studies on the Removal of Antibiotics by Bioelectrochemical Systems/ Technologies (BESs)

Reactor Configuration	Antibiotics	Classes of Antibiotics	Initial Concentration	HRT	Feeding Source for microbial development	Removal Efficiency (%)	Ref
MFC coupled with constructed wetlands	Tetracycline	Tetracycline	800 µg/L	2.5 d	Anaerobic sludge	> 99	[52]
Microbial fuel cell (MFC-FR) with Fenton reaction (Two-chamber Process)	Paracetamol		10 mg L ⁻¹	9h	N/A	89 ± 2	[53]
Microbial fuel cell (MFC) coupled with anoxic / aerobic reactor (Two-chamber Process)	Acetaminophen		30 mg/L			100	[12]
	Sulfamethoxazole	Sulfonamides	2.0 mg/L		N/A	98.00	
	Ibuprofen		20.0 mg/L			99.00	
	Sulfamethoxazole	Sulfonamides	6.0 µg L ⁻¹			100	[23]
Single-chamber microbial bioelectrochemical systems (SMBES)	Sulfathiazole		6.0 µg L ⁻¹	7 d			
	Sulfadimidine	Sulfonamides	6.0 µg L ⁻¹		Anaerobic activated sludge		
	Sulfadiazine		6.0 µg L ⁻¹			50	
	Primidone		50 µg L ⁻¹				[54]
	Diphenhydramide		50 µg L ⁻¹			78% > 80	
	Sulfamethoxazole	Sulfonamides	50 µg L ⁻¹			100	
Single-chamber MFC- MEC	Caffeine		50 µg L ⁻¹	N.A		70	
	Atenolol		50 µg L ⁻¹		Anaerobic sludge	> 80	
	Trimethoprim		50 µg L ⁻¹			90	
	Atrazine		50 µg L ⁻¹			85	
	Carbamazepine		50 µg L ⁻¹	N.A		> 80	
Two-Chamber (MFC)	Sulfamethoxazole	Sulfonamides	20 ppm	12h	N/A	100	[55]
MEC-Bio-Pd)	Diclofenac		1.0 mg L ⁻¹		N/A	100	[56]
Single-chamber MFC	Ceftriaxone sodium		50 mg L ⁻¹	24h	N/A	91	[57]
Double -chamber MFC	Sulfamethoxazole	Sulfonamides	0.08 mmol/L	48 h	Sulfamethoxazole acclimatized culture	83.3	[58]
Two-chamber MFC	Tetracycline	Tetracycline	50 mg/L	7 d	Anaerobic activated sludge	79.1	[59]
Two-chamber MFC	Sulfamethoxazole	Sulfonamides	20 mg/L	48 h	Anaerobic sludge	> 99	[60]
Double-chamber MFC	Sulfanilamide	Sulfonamides	30 mg/L	96 h	Anaerobic sludge	90	[5]
3-DBERS with A.P of 0.8 v	Sulfamethoxazole	Sulfonamides	200 µg/L	40 h	Anaerobic sludge	88.9 to 93.5	[24]
Double -chamber MFC	Oxytetracycline	Tetracycline	10 mg/L	78 h	Pig manure	99	[61]
MFC coupled with membrane bioreactor	Tetracycline	Tetracycline	90 mg/L	17 h	Shewanella sp.	99.5	[62]
Two-chamber MFC	Chloramphenicol	Chloramphenicols	50 mg/L	12 h	Anaerobic sludge	84	[63]
Two-chamber MFC	Chloramphenicol	Chloramphenicol's	30 mg/L	48 h	Anaerobic sludge	83.7	[64]
MEC Two-chamber system A.P is 0.2 V (VS Ag/AgCl)	Sulfamethoxazole	Sulfonamides	6 µg/L	7 d	Municipal wastewater	100	[23]
Photo-electrochemical catalysis coupled with MFCS	Tetracycline	Tetracycline's	100 mg/L	2 h	N/A	70	[32]
MFC coupled with constructed wetlands	Sulfamethoxazole	Sulfonamides	800 µg/L	2.5 d	Anaerobic sludge	> 99	[52]
3-DBERS with A.P of 0.8 v	Sulfamethoxazole	Sulfonamides	200 µg/L	40 h	Anaerobic sludge	72.20 to 93.52	[25]
Air-cathode single-chambered MFC	Cefazolin sodium	Cephalosporins	50 mg/L	31 ± 3.7 h	Aerobic activated sludge	> 70	[65]
Bio-cathode with A.P of 0.8 V (VS SHE)	Nitrofurazone	Nitrofurans	50 mg/L	1 h	Nitrofurazone- reducing consortium	70.60 ± 4.21	[66]
Air-cathode single-chambered MFC	Penicillin	β-lactam	50 mg/L	24 h	Another MFC in group	98	[57]
Two-chamber MFC	Metronidazole	Nitroimidazoles	10 mg/L	24 h	Anaerobic sludge	85.4	[67]
Double-chamber MFC	Chloramphenicol	Chloramphenicol	30 mg/L	48 h	Anaerobic sludge	83.7	[64]
3D-BERS with A.P is 0.8V	Tetracycline	Tetracycline	200 µg/L	40 h	Anaerobic sludge	82.61 to 95.80	[25]
Two-chamber MFC	Chloramphenicol	chloramphenicol	50 mg/L	12 h	Anaerobic sludge	84	[63]

Note. Not available N.A; Standard hydrogen electrode (SHE); Hydraulic retention time (HRT); Saturated calomel electrode (SCE); Applied voltage (A.P)

2. Sources of Antibiotics-polluted Water

Several antibiotics and their metabolites were secreted continuously discharged into the natural environment. Moreover, from the veterinary, pharmaceutical plants, dairy plants, habitat, animal waste, animal husbandry, hospital waste, municipal waste, and poultry waste releases directly into the environment [35]. Antibiotics, known as monensin, promote animal growth and feed efficiency in dairy farms to achieve natural resources. Therefore, antibiotic residues indirectly contaminate soil and water resources [36, 37]. While the consumption of antibiotics in plants is deficient compared to animals. Antibiotics are commonly found in waterways in the countryside and are widely used in agricultural land. In addition to household and industrial wastewater, the pharmaceutical industry has a significant contribution to the increased total antibiotic concentration in wastewater treatment plants [37]. Some antibiotics have a low molecular weight (< 1,000 D) and therefore, dissolve rapidly in water, leading to persistence and reflection of antibiotics (i.e., sulfonamides and tetracycline's) and their isomers [38]. The overall characteristics and removal techniques of some imperative antibiotics by BES. On the worldwide scale, sewage treatment plants (STPs) could not remove pollutants properly, such as nitrates, organic, and antibiotics, in the effective ways, so they continue to flow into sediments and water bodies in our surroundings. Disposal of improper treatment of antibiotics might be considered an essential point of pollution, and these drugs were discharged directly into the sewer system or deposited in landfills, production wastewater, or unintentional wastewater during production or distribution [38].

2.1. Removal of Antibiotics Using Different Systems

It can efficiently degrade antibiotics [39], but in the process of BES degradation of antibiotics [40], it is necessary to understand the fate of ARB and ARGs [39, 41]. The main operational factors of BES, for instance, cathodic-potentials, have a potential influence on the occurrence and abundances of ARB and ARGs [42, 43]. For example, the applied electrical Stimulation can affect the physical properties of bacteria as a cross-membrane potential [44], and membrane permeability [45], Lead to changes in ARB and ARG abundance in bioelectrochemical systems [46]. Also, compared with WWTPs, less sludge would be produced in anaerobic bioelectrochemical systems (BES) [47, 48]. This avoids the proliferation of antibiotic genes during sludge disposal[30], however, most researchers are focusing on the spread of antibiotic-resistant bacteria and ARG in wastewater treatment plants [48], and very truncated information was available about the occurrence and abundance of ABR [49] and antibiotic resistance genes in the BES. Electrodes for innovation, a variety of application development and design of the membrane and catalyst of alternative materials to make BES become a promising technology [50, 51]. The usage of BESs for the elimination of toxic pollutants is not yet widespread. Whereas the previous studies are presented in Table 1.

3. Control Techniques of Antibiotics

Bioelectrochemical technologies have been used to degrade anti-

biotics by different substrates and mechanisms [68]. The purpose of removing pollutants from municipal wastewater treatment plants is to reduce the concentration of organic pollutants, antibiotics, and its different by-products [21, 68]. BES technologies effectively remove antibiotics with minimum time; however, aerobic and anaerobic conditions in the bioelectrochemical systems [8, 69], so catalysts may be used for the antibiotics elimination [42, 70]. Hence, removal methods can be divided into the following categories:

3.1. Microbial Electrolysis Cell with Bio-cathode

For the latest system of Bio-cathode for hydrogen (H_2) production in the microbial electrolytic cell (MEC) has a favorable substitute for precious metal catalysts. In this system, biological cathode usually made as, i.e., MEC was supplied through external energy since the reduction potential of antibiotics was always higher in the bioanode (Fig. 2(A)) [71, 72]. For minimizing the total power consumption, the anode of the system operates continuously as a combined microorganism [10]. Antibiotic control techniques were performed primarily associated with direct electrochemical reduction and biodegradation [40]. Though the antibiotics gained electrons from the cathode and were reduced directly by the electrochemical reduction [40, 41]. Since the biocatalysts rapidly up and then the decline of antibiotics by reducing the power consumption [40], microorganisms thrive on the cathode electrode [66]. It is another feature of the microbial communities in which carbon sources and substrates were used in bio-cathodic chambers, and bio-cathodes were further categorized with a mixture of different materials matrix. The relevant regulations under consideration as biocatalysts can be developed and activated in the following main three categories: (a) Full-biological two-chambered biocathodes microbial electrolytic cell (MEC); (b) MEC of Full-biological single-chambered biocatalysts MEC; (c) half-biological two-chambered biocathodes MEC. Further, to identify a variety of mixed cultures that produce hydrogen, and it has detected in pre-dominant microbial species [66, 73]. If the reduction potential of antibiotics is higher than the biological anode of the systems can also be worked as bio-cathode without using an external power source [41, 71, 73].

3.2. Microbial Fuel Cells (MFCs) with ORR Cathode Catalysts

MFC was mainly used for wastewater treatment, the characteristic of oxygen reduction reaction (ORR) was expected to have high catalytic activity, so the capital costs and maintenance costs should be comparable to conventional processing technologies. In this category, MFC consists mainly of biological anodes and abiotic cathodes. Two chamber-MFC or oxygen in abiotic cathodes (Air cathode with single-chamber MFC) was commonly used as an electron acceptor in abiotic cathodes illustrated in Fig. 2(B) [74].

In the biological anode, antibiotics were used as electron donors and played an active role in carbon sources. The exoelectrogenic microorganisms and degradable antibiotics bacteria observe to the anode, to form a biofilm in the extracellular polymer material, it plays a role to reduce the super-potential of biologically uncontrollable antibiotic and their bio-metabolites. The combination of anaerobic biodegradation and electrical stimulation is an important mechanism for supporting antibiotic mineralisation in microbial fuel cells. By continuous electrical stimulation (ES) directly stim-

ulates electron and microbial metabolism in the micro-environment by directly or indirectly transferring electrons to bacterial cells [75]. However, rapid stimulation of microorganisms to metabolize antibiotics by secreting enzymes, such as 3-amino-5-methylisoxazole (3A5MI), and their by-products [55].

3.3. Bioelectrochemical System with Modified Materials

It is considered to be one of the most effective techniques to remove uncontrollable organic contaminants. It depends on three different cathodes, i.e., nickel foam (NF), Copper foam (CuF), carbon rods (CR), for antibiotics chloramphenicol (CAP) removal in BES System, while NF and CuF were not used as cathodes for CAP degradation previously. While the modified materials, targeted antibiotics, are often attacked to produce free radical species decomposition for the BES cathode. This equation shows the following main reactions; (i) Air is used to produce hydrogen peroxide (H_2O_2) from a pump or Cathode material that absorbs electrons (e^-) to reduce dissolved oxygen, i.e., $O_2 + 2e^- + 2H^+ \rightarrow H_2O_2$; (ii) Electrons generated by the transfer of the bioanode to the surface of the cathode material transfer through an external circuit; and (iii) hydrogen peroxide (H_2O_2) below UV light or cathodic material can be produce radical species/microorganism, approximating, hydroxyl radicals ($\cdot OH$) which oxidise and rescind antibiotics paternities, i.e. $H_2O_2 + h\nu \rightarrow 2\cdot OH$ and antibiotics $\xrightarrow{\cdot OH} \text{degradation products}$.

Kin Wu et al. [30] revealed the rapid degradation of Chloramphenicol (CAP) in copper foam (CuF) cathode, in-situ the production of

hydroxyl radicals, direct electrochemical reduction affects the cathode (Fig. 2(C)). Additionally, the modified cathode of photochemical and electrochemical catalytic materials (FeO/TiO_2), efficiently produced hydroxyl radicals ($\cdot OH$), and rapid degradation of tetracycline (TC) and ARG by invisible electrons generated by bio-anodes has shown in (Fig. 2(D)) [32, 76].

3.4. Removal of Antibiotics by Single-double Microbial Fuel Cell

The detection of trace amounts of organic pollutants and different types of antibiotics, in Single chamber microbial fuel cell (SC-MFC) and double chamber microbial fuel cell (DC-MFC) with sodium acetate ($C_2H_3NaO_2$) as co-substrate were investigated. The contribution of MFC to the energy production of trace organic compounds is negligible. Similarly, microbial fuel cell operation has a smaller amount of removal quantity of contaminants, but longer Hydraulic retention time (HRT) enhanced biodegradability [77]. Approximately, various parameters that primarily affect individual contamination removal and efficiency were hydrophobic compounds that tend to gain to the electrodes. Due to the observation of positively charged toxins into interaction with negatively charged biofilms, dissolved oxygen (DO), and pH. For the best removal efficiency obtained by replacing the anode with a single chamber MFC and a cathode of double-chamber MFC. The biofilms can also successfully remove antibiotics (sulfonamides and N_4 -acetyl metabolites) from wastewater [78]. The wastewater was containing

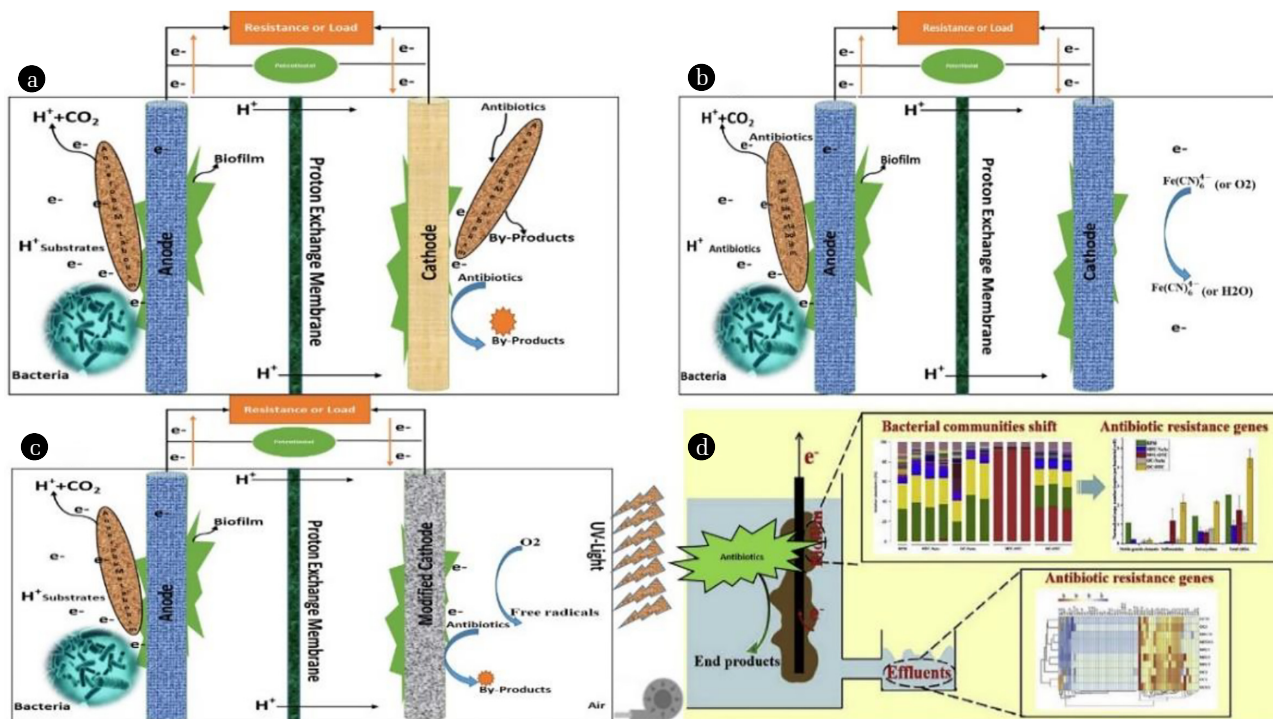


Fig. 2. Schematic of Antibiotics removal methodologies based on BES Technology (a) Schematic diagrams of MFC and MEC by standard potentials (vs NHE) of oxidation and reduction reactions in anodic and cathodic chambers, resulting in a spontaneous response in MFC and a nonspontaneous reaction in MEC. Reproduced with approval from references [71]. (b) Systemic illustrations of to reduce antibiotics by a biological anode coupled with the biological-cathode. Reproduced with approval from references [75]. (c-d) An anode connected to the cathode compartment of the biological system by removal of the antibiotic Chloramphenicol (CAP). Reproduced with approval from references [30, 32].

sulfamethoxazole, sulfamthiazole, sulfadiazine, sulfamethazine sulfonamide, and their N-4-acetyl analogues in a mixture treated with acetic acid added to the biofilm anode [23, 79]. The electrochemical performance of the biofilm was not exposed to sulfonamide influence: the trace contaminants poisoning Colomb efficiency remains unchanged, and a marginally increase in the current density [54]. Whereas the concentrations of sulfathiazole ($C_9H_9N_3O_2S_2$), and sulfadimidine ($C_{12}H_{14}N_4O_2S$) were almost constant, sulfamethoxazole was eliminated, and only sulfadiazine ($C_{10}H_{10}N_4O_2S$) was removed [23, 49, 79].

3.5. Power Density at Different Substrates

MFCs single-chamber with the air cathode shows the use of glucose-penicillin ($C_9H_{11}N_2O_4S$) as a successful fuel [80]. While the result indicates that penicillin can produce electricity at the same time as biodegradable, and the glucose-penicillin mixture plays an important role in the production of energy [53]. Glucose ($C_6H_{12}O_6$) 1.02 g/L and 50mg/L penicillin mixture, the power density was six times higher than 101.20 W/m³L and 1.0 g acquired glucose ($C_6H_{12}O_6$) 14.70 W/m³, and 50mgL⁻¹ penicillin of 2.10 W/m³ as a total of power density obtained an alternative fuel. Though, estimated the penicillin ($C_9H_{11}N_2O_4S$) can increase the permeability of electroactive bacteria cell membranes (EABCM), endorse direct electron transfer from microorganisms to the anode, and reduce the internal resistances of MFC [29]. By the mixed medium, the deterioration of penicillin has approximately 98% efficiency at the 24 h batch cycle. However, the presence of this mixture similarly increases the ability of the degradation of the organic substrate [57]. The biodegradable organic matter contained antibiotic wastewater might be the appropriate resources for the generation of electricity by the MFC system [57, 81].

3.6. Toxicity-reduction Processes

The applicability of MFCs to the disposal of antibiotics in wastewater [55]. Wang et al. [82] demonstrated his scientific research shows that MFCs apparatuses can successfully degrade the Sulfamethoxazole and its degradation products; 3-amino-5-methylisoxazole (3A5MI). Moreover, the adenosine triphosphate (ATP) level of MFC microorganisms is almost triple higher than open circuit control, although it may be related to the fast degradation of Sulfamethoxazole (SMX) in a microbial fuel cell (MFCs) [30, 53]. The mass spectrometry was detected the degradation products of SMX, and chemical standards confirmed the inference of three by-products. In accumulation, 3-amino-5-methylisoxazole (3A5MI) is a toxic chemical formed during the process of SMX degradation and can be more mineralized. In the process of degradation, the SMX, nitrogen (N₂) atoms are observed to be gradually eliminated, which should be related to the removal of SMX antibiotics and 3-amino-5-methylisoxazole (3A5MI) [82]. Antimicrobial activity tests have displayed that the bio-toxicity effect of SMX on "Escherichia coli and *Shewanella oneidensis*" was significantly reduced the treatment of microbial fuel cell (MFC) afterwards [55, 83, 84].

3.7. Biotransformation Systems

Microbial fuel cells (MFC) and microbial electrolysis cells (MEC) are two microbial electrochemical systems that convert chemical

energy into wastewater by microorganisms [54]. Worthwhile energy sources such as bio-electricity from a microbial fuel cell and hydrogen (H₂) gas from the microbial electrochemical cell. For the attenuation of organic trace compounds in synthetic wastewater, acetic acid used as a carbon source in a single-chamber reactor with standard hydrogen electrode since from -0.40V vs. SHE anode potential [57, 78, 82]. Adding fungicides of sodium azide (NaN₃), to the repeating system to distinguish the observed attenuation and the effect of biotransformation in the adsorption composition [78]. In MFCs and MECs, research carried out of eight compounds have similar attenuation effects, except Trimethoprim and caffeine, which has a slightly higher reduction [54]. The adsorption of the biofilm on the electrode is very large, but there is no consistent tendency to the physicochemical properties and adsorption degree of the adsorption compounds. The maximum current density/demand of the MFC/MEC remains unchanged before and after the number of trace contaminants of Carbamazepine 80% removal [54, 85]. The elimination efficiency was good, and the main pathway of eradication was determined in the anodic adsorption of biofilm [146]. As attention to the removal of trace organics, collective bio-electrochemical systems did not exhibit a particular advantage (or disadvantage) in conventional processes. These systems can be considered similar, thus maintaining the observed toxicity of the pollutants obtained in bioelectrochemical systems (BES). The preference of using the bioelectrochemical process for energy production in wastewater treatment does not affect the degradation of trace organic matter, but its applicability necessity considered in a more extended assessment [54, 86]. Several benefits of the specific capabilities of the bioelectrochemical system have also been improved, including technical integration or capacity expansion [54, 86, 87].

4. Main Systems for removal of antibiotics

4.1. Role of Bio-palladium (Bio-Pd) Catalyst Added with MECs

A novel bio-palladium (Bio-Pd) nanoparticles used as catalysts in the microbial electrolysis cell to remove iodinated contrast medium amidotrizoate (diatrizoate) from hospital wastewater [56]. The presence of Bio-Pd indicates that this is not only directly electrochemically reducing contaminants on the cathode, but it was also possible to enhance the catalytic reduction of hydrogen gas (H₂) production [88]. The system operated continuously synthesized and later discharged from the hospital wastewater in batch mode. The synthetic feed used to observe complete elimination with applied voltage from -0.60 V to -0.8 V. In the actual wastewater, the voltage applied at -0.8 V is 84.2%, the removal rate after 24 h is 84.90 ± 7.6%. In continuous mode, the voltage of -0.80 V (HRT = 4 to 8 h), while the synthetic feed is low (95% at HRT = 4 h and 96% at HRT = 8 h), with hospital wastewater. De Gussemme et al. [88], established the method of catalytic dechlorinating of diclofenac by biogenic palladium (bio-Pd) in MECs. At the application voltage of -0.8 V, the maximum batch effect was obtained and diclofenac removed after 3h. In continuous mode, the minimum HRT = 2 h was required to completely remove diclofenac in the

case of an applied voltages of -0.80 V. When applied voltages on continuous modes of hospital wastewater effluent, have the removal efficiency of $57 \pm 90\%$ at -0.8 V and $HRT = 8$ h were observed, mainly because of the low concentration of pollutants in the wastewater and the presence of disruptive substances [56, 88].

4.2. Dual Chamber MFC Coupled with Biological (A^2/O) System

A novel system with a dual-MFC chamber (MFC- A^2/O) process with a "proton exchange membrane" (PEM) and solid plain graphite plates. The electrodes, in combination with anoxic-aerobic (A^2/O) for the biological degradation of concentrations in wastewater, sulfonamide (sulfonamide), ibuprofen, and very high paracetamol, which is reduced simultaneously [18, 89]. Moreover, three pharmaceutically active compounds (PhACs) with high removal efficiencies ranging from 98.21% to 99.89% were obtained. The main purpose of using a single reactor within the anaerobic process for the development of facultative bacterial inhabitants in anodes and mixed liquor was better than aerobic reactions [90]. However, the maximum power density was 5.30 W/m^2 , and the maximum Coulomb efficiency or current efficiency of the anode was approximately 25.2%. The differences in the abundance of bacterial communities in different parts of the MFC- A^2/O system were observed [12, 91].

4.3. Paracetamol (PAM) Removal by Fenton System

Dual-chamber MFCs were developed to achieve biodegradation of paracetamol using microbial fuel cells by the Fenton process and did not use external feeding sources [92]. In the anode chamber, the degradable oxidation pollutants with low current intensity in domestic wastewater effluent and released electrons from the system. Additionally, in the cathode, the electron flux (Φ) of the anode facilitates the increase of free radicals to form hydroxyl radicals ($\cdot OH$) in the regeneration of the external iron sources [93]. The pH levels at different due to the adjustable concentration dose that allows a typical oxidation reaction during the Oxydrile radicals, which produces a base after the initial electrochemical reduction. The optimal condition of the process is 5.0 mg F/L, $pH = 2.0$. Under the best conditions, the performance of the system with the bio-electric power capacity of the MFC reactor is virtuous [53, 94]. The maximum degradation efficiency of PAM was 70.0% within 9 h and 25.0% to obtain complete mineralization, and most of the metabolites p-nitrophenol ($C_6H_5NO_3$) is converted, and a very fewer vulnerable dicarboxylic/carboxylic (carboxyl functional groups) acid [94, 95]. It shows that the results suggest that the MFC-Fenton process can be used as an energy-efficient method for the treatment of PAM-containing wastewater and the degradation of non-biocompatible drugs in the aquatic environment [53, 93].

5. Performance of Antibiotics in Bioelectrochemical Technology

5.1. Removals Performance of Antibiotics

Another significant factor is the biological factor, which acts as biocatalysts and then decomposes the substrates that affect the

electron transfer rate and the bioelectricity generation [21, 71]. The core part of MFC-biocatalysts is a variety of biochemical pathways, which can be used to obtain various electronic transmission mechanisms according to the parameters designed for MFC [21, 96, 97]. In the current review, the discussion of the key factors that regulate the performance of MFC, according to the bioelectronic mechanism. From the literature, some scientific studies on the removal of pollutants from BES indicate that MFC performance is affected by many factors [96]. Hence, apply the system in the future; it has necessary to discuss the influence of the parameters in detailed. However, other various parameters like immunization, additives, flow rates [71, 96], pH values, and buffer solutions have been determined [98, 99]. The electrochemical-properties of antibiotics and the many physico-chemical properties of the pollutants such as the initial concentrations, toxicity of salts [99], and the various applications of applied potential, electrodes, temperature, environmental factors, carbon sources, and salinity were also discussed. The effect on electrochemical performances, including energy production capacity, antibiotic removal efficiency, and conversion of degradation by-products [71, 98].

5.2. Substantial Selection for Antibiotics

The substantial selection is the basis for the overall performance of bioelectrochemical technology, as microorganisms flourish and form biofilm for electron transfer in electrodes as biocatalysts [100-102]. Many researchers use carbon felt as an electrode material [103]. However, there has been limited research into the properties of various electrode materials for wastewater treatment, including antibiotics. Only one main studied found by Wang et al [30], which the degradation effect of the cathode electrode in the antibiotic chloramphenicol (CAP) treatment of the three metal foams in BET. Copper foam (CuF), carbon rods, and nickel foams (NiF) are the best off-hat properties of cathode electrodes. However, the 32 mg/L chloramphenicol (CAP) were removed using copper foam electrodes [102] within 12 h, which was further based on the carbon rods after 24 h, and nickel foam more than 120 h [10, 77]. Antibiotics chloramphenicol is the main product, using carbon rods and foam nickel electrodes with copper foam electrodes, and completely mineralized with CO_2 and H_2O , each with nitrobenzene ($C_6H_5NO_2$), and 4-nitrobenzene alcohol. They revealed a reduction in the current of the material based on the Cyclic voltammetry (CV) results [30]. By comparing the carbon rods with the foam nickel electrodes, the highest reduction current is obtained by using the copper foam electrodes. The aim of sustainable development of a bio-electrochemical Technology can be able to develop electrode materials such as biochar and reforming materials to improve the redox reaction rate for low-cost environmental protection. The interaction between the electrodes and the microorganisms and the interaction between the electrodes and the antibiotics must be paid more attention. For example, the modified electrode leads to an increase in gene expression associated with electron transfer, whether to further improve the electron transfer rate involved in antibiotic degradation of the redox reaction [49, 77].

5.3. Importance of Carbon Source for Removal of Antibiotics

The vital contribution of carbon sources performing a significant

role in the balance between microbial communities and metabolic activities [104, 105]. All research of that area has focused on other carbon sources (antibiotics was the only source of carbon), and BES did not have the features of inorganic and organic carbon [104]. The feasibility and effectiveness of anodic-oxidation of sulfamethoxazole (SMX) [84] in MFC in an oligotrophic environment [55]. Deprived of additional carbon sources, SMX was the only source of nitrogen [60] and carbon that is highly extinguished to CH_4 as the final degradation product [55, 61]. This specifies that the metabolic capacity of the microbial community using antibiotics as a separate carbon source has improved significantly. Kong et al. [66] to investigate the effect of inorganic carbon source NaHCO_3 on the bio-catalytic metabolism of NFZ removal. However, it showed that the rate constants and NFZ reduction removal efficiencies in bioanode fed with NaHCO_3 were slightly lower than those with added glucose but significantly better than non-biological cathodes. This means that the biocathodes fed with NaHCO_3 still has significant bio-catalytic capacity reported [60, 106]. Furthermore, many scientific studies have been based on enthusiastically biodegradable carbon sources such as glucose and acetate as a common substrate to improve antibiotic treatment [55, 60]. The yield of MFC using glucose and penicillin as a co-substrate is nearly 48 times higher than that of MFC containing only penicillin as a substrate [57]. The observation suggests that the addition of common substrates provides enough carbon sources and more electron donors to improve the metabolism of the microbial communities.

5.4. Removal of Antibiotics by Applied Voltages

Changes in applied potentials voltages lead to electrochemically active biofilm with different degrees of electrical stimulation, and the number of electron donors [107], affecting on the mineralization of antibiotics is different [108]. Based on previous studies, additional negative application potentials significantly improved the elimination of antibiotics and significantly reduced the HRT of biocathodes [107]. Nitrofurazone reduction constant rate of 0.6770 ± 0.690 per hour and reduction efficiency of $42.250 \pm 1.351\%$ at 1.0 h, when the voltage was applying to -0.2V, then the coefficients increased from $1.202 \pm 0.124/\text{h}$ and $70.60 \pm 4.21\%$. Similarly, on a cathode potential of -0.8 V [66]. Additional, as the cathode voltage is high, the conversion of the degradation intermediate is achieved faster. In Sodium bicarbonate (NaHCO_3) fed biocathodes, 3-(5-Nitro-2-furyl) acrolein (NFF), and [(5-amino-2-furyl)-methylene]-hydrazinecarboxamide, and the products remarkably accumulated in 72 h at -0.2 Volts, equally support to degraded and disappeared within 48 h at -0.80 V [30, 66]. Increasing the applied voltage have escalations the elimination efficiency of the CAP [107, 109]. Guo et al. [110, 111] demonstrated that the CAP removal efficiency increased with the increase in cathode potential from -0.50V to -1.25 v; vs. SHE. The increase in cathode potential is closely related to the high removal rate of CAP antibiotics. However, if the applied potential of external is meagre, then the total energy consumption will be raised. Therefore, in the future, it is necessary to consider the possibility of optimizing the applied potentials to reduce energy consumption while maintaining the efficiency of the degradation of antibiotics.

5.5. Removal Efficiency of Antibiotics

The pollution capacity of the system is increased, and it is easy to see that the initial concentration of antibiotics increased progressively. So, it was tough to need more time and power to remove them [7, 112]. It has been pointed out that the degradation efficacy of the CAP in the biocathodes decreased with the increase of concentration [110]. Comparable as the SMX concentration increases from 20 mg/L to 200 mg/L, the half-life of SMX removal in MFC extension from 24 h to 72 h. The results show that the degradation efficiency of antibiotics depends to some extent on the initial concentration [60].

However, some studies have provided conflicting experimental results, that the higher concentration of antibiotic, the improved BES performance was reported. In single-chamber MFC, the removal rate of 50 mg/L ceftriaxone-sodium was much higher than 30 mg/L [81]. According to the electrochemical-impedance spectrum and anode terminal discharge performance, ceftriaxone-sodium incidentally affects the redox enzyme of microorganisms, resulting in an overall decrease in MFC resistance [57]. Though similar results were obtained in other MFC related studies [19, 113-115]. Therefore, as the concentration of penicillin increases, the power generated by MFC has also increased [116]. As for concerned that some researchers have reported that the performance of the BESs remains the same as the initial concentration of antibiotics increases. The anode biofilm of the cefazolin sodium was satisfactorily acceptable [65]. The anode biofilm has been exposed continuously to increased levels of cefazolin sodium from 50 mg/L to 450 mg/L and does not show a voltage drop of more than 1,200 h during MFC operation. The thickness of the extracellular polymer material and biofilm may be a toxin used to quench these antimicrobial compounds [65, 117]. One more example was that the voltage output of MFC had relatively high reported to be stable when the 500 mg/L to 1.0 mg/L. Finally, these scientific studies have shown considerable impact load capacity, providing scientific-reference for the treatment of antibiotic-containing pharmaceutical wastewater by bio-electrochemical systems [118].

6. Relationship between Temperature and Bacterial Growth

Very few scientific studies have presented that temperature can significantly affect the growth of bacteria [119], metabolic activities, and advance biochemical reactions associated with antibiotics treatment [120, 121]. Zhang et al. [63] envisioned single factor experimentation to explore the various effect of the temperatures (20°C , 30°C , and 40°C). They achieved a higher removal efficiency of CAP was 75.13% at 40°C , compared with 68.11% at 20°C . Guo et al. [111] examined the removal efficiency of CAP antibiotics in BESs at 10°C , 15°C , and 30°C , respectively. While the temperature was lowered from 30°C to 10°C , the removal rate of CAP decreased sharply from above 95% to below 10%. Interestingly, the temperature did not significantly affect the performance of BET at 2% salinity. When the reaction temperature was varied between 30°C and 10°C , the BET maintained a CAP removal efficiency of $> 75\%$.

The effect of temperature conversion on biocathodes removal of CAP by [122, 123]. Kong et al. [122] revealed as the temperature was lowered from 25°C to 10°C, the rate of CAP reduction, but at 10°C, contrasting the abiotic cathode, it still shows a higher percentage of decline at the biocathodes. The CAP reduction reaction at current through cyclic voltammetry (CV), based on the “Geo-Chip” and “Illumina sequencing” methods using the 16S *rRNA* gene, Liang et al. related microorganisms and functional genes were also resolute for changes in temperature from 10°C to 25°C. The major differences important genes that respond to elevated temperatures of 15°C include nitro-reductase and heat shock proteins, but no electron transfer genes. The excellent catalytic-stability of continuous low temperature at bio-cathodes exposed to 10°C was shown: (I) firstly, enrich two major taciturn adapted nitro-reductases that carry microorganisms, which were *Vagococcus* and *Aeromonas*; (II) maintains the relative abundance of critical electron-transferred genes [111]. Such as hydrogenated genes and cytochrome-specific genes. It was essential to evaluate the ability of large scale BESs to degrade antibiotics[2], at changing temperature. Thus, further experimental studies to understand the metabolic activity of the electrode are exposed to biofilms due to temperature and stress mechanisms [2, 10].

7. Major Problematic Effects on the Removal of Antibiotics

7.1. Effects of Initial Concentration

The effect of initial antibiotic concentrations on the efficiency of bioelectrochemical technology (BET) removal was quite complicated, as previous studies had contradictory conclusions. Some studies have publicized that the higher the concentration of antibiotics, the lower the BET removal ability [124]. Other researchers report that removal efficiency was increased with increasing antibiotic concentrations, while other studies have shown that initial antibiotic concentrations have no significant effect. It is easy to understand that the contamination load of the system increases as the initial concentration of antibiotics increases [125]. Therefore, more time is required to remove the antibiotic, and as the concentration increases, the degradation efficiency of antibiotics in the biocathodes decreases [10, 126].

7.2. pH Optimal Range and Its Effects on Microorganisms

The pH level was an important step in the anaerobic treatment process [16], and there was a need to distinguish between acidogenic and methanogenic microorganisms having an optimal pH [127]. If the pH ranged from 6.7 to 7.5 was not maintained in the anaerobic reactor, the anaerobic system may be devastated. When antibiotics are an occurrence in the anaerobic reactor, volatile fatty acids (VFA) are accumulated, and the pH drops. [128]. For example, it has been shown that the pH in the antibiotic reactor is much lower than the pH in the control reactor. For anaerobic digestion of swine slurry, the pH of chlortetracycline (CTC) containing the reactor is lower than that of the control reactor [128, 129]. Moreover, it was pointed out that the pH in the anaerobic reactor is sensitive

to high concentrations of antibiotics [130]. As stated by Miller et al., the pH continues to stabilize, which lasts between 7.4 and 7.6, adding 1.0 to 5.0 mg/L SMX to the reactor, then dropping the pH to 6.3 after 50 mg/L and added immediately to SMX in response to increased VFA [131]. It was also confirmed that the pH did not drop from just (6.8 – 7.2) to 5.9 until the concentrations of TC and SMX reached 20 mg/L and 1.5 mg/L, respectively [22, 132]. Therefore, the immobility of the anaerobic reactor can be influenced by the addition of high concentrations of antibiotics. However, a sharp drop in pH can only occur at the beginning of the anaerobic process, and the contact time between the antibiotic used and the anaerobic sludge is short and stable as the reaction proceeds [133].

8. Microbial Communities vs. Antibiotics Degradation

The microbial communities play an important role as a biocatalyst in the overall performance of the bioelectrochemical technology [134]. The rapid development of high quantity sequencing for the degradation of the antibiotic to explore the effective microorganisms in BET helps to understand the changes in the microbial community in response to environmental factors. Their influence on microorganisms, as well as initial antibiotic concentrations, temperature, applied potential, and salinity has been studied displayed in Table 2. These powerful factors stand out in altering the richness and diversity of the microbial bacterial communities. The maintenance of acclimation in the Chao I, Simpson, and Shannon indices of the bacterial communities of MFCs decrease sharply, and *Eubacterium* spp. dominant genus in MFCs [134]. The key factors accumulate corresponding specific functional microorganisms. Biocathodes were significantly enriched based on temperature changes of 10°C, for example, the cold-adapted ability of *Aeromonas* and *Vagococcus gene* [123]. Nevertheless, the effects of various factors vary from pharmacological and lead to significant transformations between the bacterial communities. The abundance of proteobacteria bacteria increases in biocathodes, and the potential adverse effects on the proper treatment of antibiotics CAP [124]. Whereas enriches of Firmicutes about 60.07% based on 0.8 V treating nitrofurazone (NFZ) group [66]. Table 2 showed the main microorganisms that can cause the degradation of antibiotics at the phylum level, three dominant organisms have its place to *Proteobacteria*, *Bacteroidetes*, and *Firmicutes* be present in most BET reactors. Intended for sulfamethoxazole, *Thauera* proficient in the degradation of aromatic hydrocarbons accompaniments in MFCs inoculating SMX acclimated cultures [58]. The growth of the abundance of *Methanobacterium*, *Methanosaeta*, and *Treponema* is responsible for the degradation of exceptionally concentrated SMX-antibiotics in MFCs reported [60, 135]. CAP-electrochemically active bacterium *pseudomonas* occurs in all biocathodes, having the ability to decrease nitroaromatics. For antibiotics NFZ, *Klebsiella* bacteria play a significant role as a dominant genus about 60.57% at 0.20 V and 56.96% at 0.5 V in biocathodes, respectively [66, 136]. However, the *Klebsiella* spp. did not only can decrease nitro-aromatics electrochemically active similarly, but *Oxytetracycline* spp. was the functional

Table 2. Diversity of Bacterial Communities for the Antibiotics Degradations at Various Conditions in the Bioelectrochemical Systems

Reactors Configuration	Antibiotic	Acronym	Operation Values	Conditions	Mechanism of Antibiotics degradation	Microbial community	Ref
Bioelectrochemical system	Chloramphenicol	CAP	-1.25 V -1 V -0.5 V	Applied potentials	CAP	Phylum: Proteobacteria Genus: Alkaliphilus, Halomonas, Methylophaga, and Acinetobacter Genus: Flavobacterium and Achromobacter	[124]
Bioelectrochemical system	Chloramphenicol	CAP	10°C 15°C 30°C	Temperature, Salts	ARGs	Leptolinea, Longilinea, and Acinetobacter Terrimonas and Erysipelothrix Methylophilus and Candidatus Methanogranum	[111]
Biocathodes	Chloramphenicol	CAP	10°C 10°C to 25°C	Temperature	AMCL, AMCL2	Aeromonas and Vagococcus Raoultella and Enterococcus	[139]
Bioelectrochemical system	Chloramphenicol	CAP	10 mg L ⁻¹ 20.2 mg L ⁻¹ 50 mg L ⁻¹	Concentration	CAP	Brevundimonas, Pseudomonas Saccharibacteria, and Methylobacillus Methylobacillus Pseudomonas and Methylophilus	[124]
Bioelectrochemical system	Chloramphenicol	CAP	0-0.5% 0.5% 2% 6%	Salts	CAP	Methylophilus, Chryseobacterium Proteiophilum, Comamonas Oribactera, Methyloversatilis, Candidatus Methanogranum Halangium-Methylophaga, Byssorox, Halomonas Pseudomonas, Lysinibacillus	[111]
Bioelectrochemical system	Chloramphenicol	CAP	-1.25v -1v -0.5v	Applied potentials		Proteobacteria (Phylum); Acinetobacter and Alkaliphilus (Genus) Proteobacteria (Phylum); Flavobacterium and Achromobacter (Genus) Proteobacteria (Phylum); Halomonas and Methylophaga (Genus)	[110]
Biocathodes	Nitrofurazone	NFZ	-0.20v -0.50v -0.80v	Applied potentials	Nitroaromatics reduction	Proteobacteria (Phylum); Klebsiella (Genus) Proteobacteria (Phylum); Klebsiella (Genus) Firmicutes (Phylum); Enterococcus (Genus)	[66]

bacteria as *Eubacterium spp.* was fertility approximately $91.69 \pm 0.27\%$ in MFCs reactor [135]. Here, some affiliates of *Eubacterium spp.* bacteria anaerobically change oxygen-containing hetero-cyclic-aromatic compounds or secrete enzymes to the catalyze-metabolic process of complex-compounds [137, 138]. For cefazolin-sodium, *Acinetobacter*, *Stenographmonas*, *Geobacter*, *Lysinibacillus* and *Dysgonomonas*, were detected and are liable for degradation in single-chamber MFCs through co-metabolism [65, 136]. Further considerate of efficient microbes would facilitate the rapid development of wastewater treatment containing antibiotics. Inaccessibility of virtuously functional bacteria, genome-level, metabolic modelling, and genetic engineering can provide beneficial species with superior degradation capabilities [58, 65, 136]. Immunisation of these functional species would contribute to the quick startup of biofilms and help the excellent treatment of antibiotics degradation in bioelectrochemical technology.

9. Bacterial Diversity at the Gene Level

Several studies have been investigated the fate of antibiotics removal from wastewater treatment plants [140], while many other studies have taken into account the environment, focusing on ARG's response to wastewater treatment [141, 142]. However, very few studies found from the literature that has examined the occurrence and the fate of AR bacteria in WWTPs [143]. Although a considerable amount of research supported out joining the investigation of ARGs and antibiotics [144] over and above ARB and ARGs [141, 145] along the wastewater treatment process, comprehensive studies evaluating ARB, ARGs and the fate of antibiotics, in WWTPs and its final influence on getting water bodies are still lacking [146]. Each gene is amplified using a specific primer set, PCR conditions include the initial denaturation of 3 min at 95°C, then 40 cycles at 95°C, then 20 s at specific annealing temperature, and finally two elongation steps at 72°C for 40 s, 78°C lasts 32 s. As well to investigate the extent of antibiotic contamination by WW, which is designed to fill prevalence and distinguish bacterial components from receiving rivers, faecal bacteria, *E. coli*, and the combination of nutrients that depend on nutrition and their effects of ARG, attached particles (PAB) and free Living (FLB) [9, 147]. *Escherichia coli* is selected as faecal to show bacteria that may be exposed to high antibiotic concentrations in the gastrointestinal tract of humans and animals, and gain resistance before being released into the sewer system and eventually reaching WWTP [146]. Therefore, high throughput quantitative PCR and functional metagenomics [148], including the abundance and diversity of ARGs, the mechanism of transmission of horizontal genes, the difference in ARGs in long-term operation [9], the expression of single and multiple antibiotics, BESs for the scientific aspects of bio-film are expected to unravel [149]. In a realistic environment, attention should be paid to the coexistence of antibiotics and other emerging substances [150], such as pharmaceuticals and personal care products (PPCPs) [151]. Hence, in addition to the feasibility of BESs joint removal of existing pollutants, the effects of substances coexisting in the electrically active biofilm antibiotic resistance genes must also be considered [151, 152].

10. Knowledge Gaps and Future Perspectives

As discussed in this review, the recent development and the performance of MFCs and MECs concerning the degradation of appropriate antibiotics were studied. In this, BES systems can remove many selected antibiotics from wastewater, which is a unique technology. However, the latest research has focused on wastewater effluents containing antibiotics and many other types of pollutants. Due to the physicochemical properties of wastewater containing antibiotics and their impact on microbial communities in anaerobic reactors, however, further investigation is needed to understand the mechanism by which BES removes and responds to other antibiotics. Most of the scientific studies on the anaerobic degradation of antibiotic conversion products and their Eco-toxicity or health risks are needed. For analysis of the effects of antibiotics on BES, microbes are required to understand their impact on various factors. However, there are several types of medicines in wastewater, and further research is needed to check the removal limits under different operating conditions. Although the significant progress in BESs antibiotic removal capacity is relatively low, given the future application of realistic pollution in the environment. In the future, it is necessary to investigate the influence of microbial communities on electron transfer and the comparison between electrodes. Moreover, it needs further study to improve the interaction between bio-electrodes and electron transfer medium by adding and removing the ability of electronic transfer medium and electrochemical active microorganism.

11. Concluding Remarks

This review on bioelectrochemical technologies should be developed for different environments, including the removal capacity of solid substrates containing antibiotics and the problematic effects on microbial communities. BES technology is almost still a relatively new technology in the experimental phase, but since the configuration of MFCs and MECs has so far achieved excellent performance, the removal of pollutants and the generation of electricity from the substrate. Therefore, the first and most important measure is to reduce the unnecessary use of antibiotics to address the problem at the source. Some BES systems show the possibility of complete mineralisation, including antibiotics with low energy costs. The hybrid MFC-coupled constructed wetlands and MFC-MEC systems have been proposed, and further research is needed to assess the removal performance of antibiotics.

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Authors Contributions

M.H. (Ph.D. Student) wrote novel review articles with the help

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