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Review

Electrocoagulation for the Removal of Antibiotics and Resistant Bacteria: Advances and Synergistic Technologies

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Abstract

The persistence of antibiotics and antibiotic-resistant bacteria (ARB) in aquatic environments poses a significant risk to both the environment and public health. Conventional wastewater treatment systems are often inefficient in completely removing these emerging contaminants, highlighting the need for advanced and integrative treatment approaches. Electrocoagulation (EC) has emerged as a promising electrochemical method due to its operational simplicity, low chemical demand, and versatility in treating a wide range of wastewater types. This review critically analyzes the efficiency of EC, both as a standalone process and in combination with complementary technologies such as electrooxidation, membrane filtration, advanced oxidation processes (AOPs), and biological treatments. Emphasis is placed on the removal mechanisms, influencing parameters (pH, current density, electrode material), and the synergistic effects that enhance the degradation of antibiotics and the inactivation of ARB. Additionally, the review discusses the limitations of EC, including electrode passivation and energy consumption. The integration of EC with other technologies demonstrates improved pollutant removal and process robustness, offering a viable alternative for treating complex wastewater streams. This work provides a perspective on the current state and future potential of EC-based hybrid systems in mitigating the environmental impact of antibiotic pollutants and antimicrobial resistance.

Keywords: antibiotic pollution; antibiotic-resistant bacteria; hybrid advanced technologies; electrocoagulation; process feasibility

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1. Introduction

The widespread use of antibiotics in human health, animal husbandry, and agriculture has led to their continuous release into aquatic environments, either as parent compounds or as biologically active metabolites [1–3]. Several studies have reported the presence of antibiotics such as ciprofloxacin, tetracycline, sulfamethoxazole, and amoxicillin in wastewater effluents, surface waters, and even drinking water sources, with concentrations ranging from ng/L to several μg/L [4–6]. The environmental persistence of these compounds, along with their sublethal effects on microbial communities, significantly contributes to the selection and dissemination of antibiotic-resistant bacteria (ARB) and resistance genes (ARGs), which are now considered emerging contaminants of global concern [7,8].

Traditional wastewater treatment processes, including activated sludge, chlorination, and UV disinfection, often fail to remove antibiotics or inactivate resistant microorganisms completely [9,10]. Moreover, some of these methods may promote the transformation of antibiotics into more persistent by-products or create conditions that enhance horizontal

gene transfer [11,12]. In response to these limitations, alternative technologies have gained attention for their ability to target both chemical and biological contaminants.

In several European countries, post-treatment technologies have already been implemented at wastewater treatment plants to remove micropollutants. For example, in Switzerland, the Neugut WWTP (Dübendorf) pioneered full-scale ozonation, achieving on average $\geq$ 80% removal of selected micropollutants, later complemented by biological post-treatment to mitigate potential ecotoxicological effects [13]. Similarly, in Germany, a pilot plant in Kaiserslautern successfully piloted a combination of ozonation followed by granular activated carbon filtration, achieving over 80% removal of organic micropollutants and microplastics [14]. Both cases serve as benchmarks for comparison with other emerging technologies, such as electrocoagulation. Electrocoagulation (EC) has emerged as a promising solution due to its operational simplicity, low reagent requirements, and versatility in treating complex matrices [15–17].

EC is increasingly regarded as a viable treatment for removing antibiotics and antibiotic-resistant bacteria (ARB) from contaminated water, supported by a convergence of physicochemical and engineering mechanisms. EC generates metal hydroxide coagulants (e.g., $Fe(OH)_3$, $Al(OH)_3$) that can adsorb, complex, or precipitate antibiotic molecules through interactions with functional groups such as carboxyl and amines [18]. Additionally, EC neutralizes surface charges and destabilizes colloidal species, promoting the aggregation and removal of both dissolved antibiotics and microbial contaminants [19]. EC may also contribute to the inactivation of ARB through oxidative stress, membrane destabilization, and physical entrapment of both cells and extracellular resistance genes in metal-based flocs [20,21]. From an engineering perspective, EC systems are modular, dynamically responsive to pollutant loads, and compatible with low-energy or off-grid configurations [22,23].

Despite its viability, challenges persist, including the formation of potentially more toxic or persistent antibiotic degradation by-products, the need for stringent optimization of operational parameters (e.g., current density, electrode material, pH) for varied water matrices, and the effective management of generated sludge containing concentrated contaminants. Furthermore, while EC effectively removes ARB, the elimination of extracellular antibiotic resistance genes (ARGs) often requires integration with advanced oxidation processes, such as electro-Fenton, due to their persistence in water even after bacterial lysis [20,24].

Although several reviews have explored the use of electrocoagulation (EC) for removing pharmaceutical compounds, antibiotics, or endocrine-disrupting chemicals [15,16,25–29], none to date have specifically addressed its dual capacity to eliminate both antibiotics and antibiotic-resistant bacteria (ARB) from contaminated water. This omission is particularly relevant given that these two classes of contaminants frequently co-occur in aquatic environments and are closely interrelated: the presence of residual antibiotics can promote the survival, selection, and genetic exchange of resistance traits among microbial populations. Therefore, evaluating EC technologies solely in terms of antibiotic removal overlooks a critical aspect of environmental antimicrobial contamination. This review addresses this gap by critically examining the capacity of EC to remove both antibiotics and ARB, analyzing key operational parameters and challenges. Emphasis is placed on its standalone and combined application with other treatment strategies, as well as its performance under realistic conditions relevant to wastewater from municipal, hospital, and agricultural sources.

2. Materials and Methods

For this purpose, a search was conducted in Scopus (https://www.elsevier.com/products/scopus accessed on 3 March 2025), where a period from 2015 to 2025 was defined,

and the following keywords were used: electrocoagulation, wastewater, greywater, antibiotics, bacteria, and resistance. With this, a total of 70 articles were selected. Additionally, a second search was conducted by replacing the word 'antibiotics' with each group of antimicrobial drugs. The search words were aminoglycosides, beta-lactams, penicillin, cephalosporins, monobactams, carbapenems, amphenicols, glycopeptides, lincosamide, macrolides, nitroimidazoles, oxazolidinones, quinolones, rifamycin, sulfonamides, and tetracyclines. Finally, the word 'bacteria' was replaced by the names of pathogenic microorganisms present in wastewater: *E. coli*, *Salmonella*, *Vibrio cholerae*, *Legionella pneumophila*, *Norovirus*, *Hepatitis A*, *Rotavirus*, *Giardia lamblia*, *Cryptosporidium*, and *Entamoeba histolytica*. These two changes resulted in 103 articles.

3. Results

3.1. General Analysis

The A keyword network map was also made using Vos Viewer 1.6.20 to visualize research trends on wastewater treatment for the removal of antibiotics, microorganisms, and resistant bacteria by electrocoagulation. Figure 1 identifies four thematic clusters: the blue node focused on electrocoagulation, its use in wastewater treatment and its connection with contaminants such as ciprofloxacin, tetracycline and ofloxacin; the yellow one associated with operational parameters such as current density, electrodes, pH, reaction time or energy consumption; green, which relates water treatment and post-treatment uses, as well as parameters associated with water quality improvement; and finally, red, which focuses on treatments that have been studied to create a synergy with electrocoagulation.

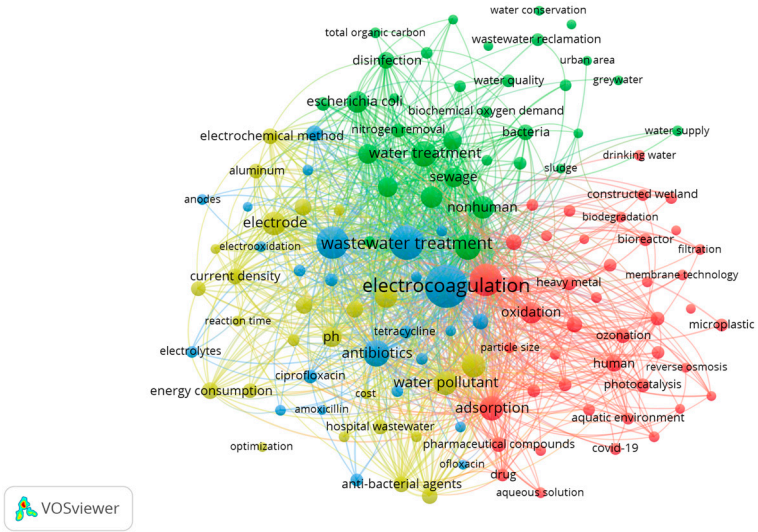


Figure 1. Network map of keywords made with Vosviewer 1.6.20.

This keyword network indicates that most publications in the defined period focused on studying wastewater treatment using electrocoagulation, with the adsorption of the electrogenerated coagulant as one of the main areas of research (134 links and 625 total link strength), alongside the oxidation process, water pollution, and electrodes. In addition, the network map enables the visualization of areas of opportunity, such as water conservation, water supply, microplastics, optimization, electrolytes, or anodes, to name a few.

Finally, between 2022 and 2023, electrocoagulation and wastewater treatment began to be related to the following issues:

1. The presence of drugs related to the COVID-19 pandemic (antibiotics such as tetracycline and sulfamethoxazole) or the removal of microplastics.

2. Coupling of electrochemically assisted coagulation with other treatments such as reverse osmosis, activated carbon, use of membranes, photocatalysis, or biodegradation, in addition to the analysis of generated sludge and coagulant by scanning electron microscopy.

Regarding antibiotic elimination, the most studied were fluoroquinolones (ciprofloxacin, levofloxacin and ofloxacin), cephalosporin (cephalexin, cephalothin, cefazolin and ceftriaxone), macrolides (tylosin and azithromycin), β-lactam antibiotics (amoxicillin and ampicillin), tetracyclines (tetracycline and doxycycline), amphenicols (chloramphenicol), dihydrofolate reductase inhibitors (trimethoprim) and sulfonamides (sulfamethoxazole) (Tables 1–4). The analysis of the bibliographic data revealed that, in addition to antibiotics, various drugs were investigated during the timeframe. Among these compounds are anti-inflammatory medications, analgesics, antiepileptics, antimicrobials, antiparasitics, and disinfectants (Tables 1–4).

Additionally, the disinfectant capabilities have been studied through the electrochemical generation of chlorine-derived radicals for the elimination of microorganisms, including *Escherichia coli*, other bacteria, viruses, and spores (Tables 1–4). Its effectiveness has also been investigated in removing antibiotic resistance genes, including the sulfonamide resistance gene type 1 (sul1), the sulfonamide resistance gene type 2 (sul2), the tetracycline ribosomal protection protein O (tetO), and the tetracycline-inactivating flavin-dependent monooxygenase X (tetX) (Table 6).

3.2. Basic Principles of Electrocoagulation

Electrocoagulation (EC) is an electrochemical treatment process in which an external electrical current is applied to water, triggering chemical and physical reactions. These reactions lead to the destabilization and removal of various contaminants, including suspended particles, dissolved substances, and emulsified compounds Liu [30].

The electrochemically assisted coagulation separation process is initiated by applying a potential difference across an electrochemical cell. This electrical input drives the anodic dissolution of electrode material, leading to the in situ generation of metal ions within the solution. Simultaneously, the applied potential difference facilitates water electrolysis, producing hydroxyl ions (OH^-). The synergistic action of these generated metal ions and hydroxyl ions results in the formation of insoluble metal hydroxides. These hydroxides serve as efficient coagulants and adsorbents, effectively sequestering contaminating compounds present in the fluid through mechanisms such as charge neutralization, sweep flocculation, and adsorption. According to some authors [31], cationic or anionic hydroxocomplexes are also formed, which destabilize the electrostatic repulsion forces present in the colloidal matter, either through charge neutralization or the formation of intraparticle bonds. This leads to the formation of agglomerations, which marks the beginning of the physical process of coagulation. These initial agglomerations then grow into larger solid particles, known as floccules, through a process of flocculation. Once formed, these floccules can be efficiently separated from the fluid using physical methods like filtration, clarification, or sedimentation (Figure 2).

During electrolysis, a series of reactions occur at the electrodes, providing positive and negative ions. The anode, or sacrificial electrode, dissolves as it donates metal ions, while the cathode remains undissolved. The electrodes typically employed are conductive or semiconductor materials chosen for their ability to generate the electromotive force essential for the electrochemical reactions. These reactions effectively destabilize the contaminants, causing them to form larger, less colloidal, and less emulsified solid aggregates compared to their initial equilibrium state [32].

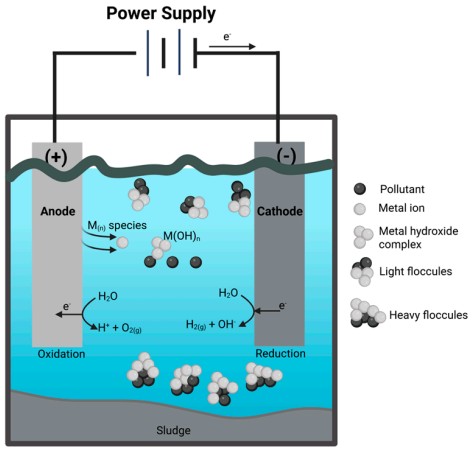


Figure 2. General mechanisms in a basic reactor of the electrocoagulation process (Created in BioRender. Sol, L. (2025) https://BioRender.com/o9eha7t).

The primary reactions during electrocoagulation include hydrolysis, electrolysis, ionization reactions, and free radical formation, which alter the compound-water properties and facilitate the elimination of the first one immersed in the fluid.

In general, EC is described in three phases [33]:

1. Formation of the electrocoagulant by electrolytic oxidation of the anode metal.
2. Destabilization of contaminants and blends.
3. Formation of flocs through the aggregation of contaminant particles or the adsorption of these particles by the coagulant.

More precisely, electrochemically assisted coagulation generates the processes shown in Figure 3 [34,35]:

- Electrodissociation occurs when the electrode dissociates due to the passage of electric charge in the system, generating the necessary force to release metal ions from the solid into the liquid. The dissolution rate is not significantly affected by salinity.
- Electrocoagulation: Starts when the electrocoagulant is formed.
- Electro flocculation: Flocculation of the material to be removed.
- Electro flotation: Generated by the bubbling of gases produced at the electrodes.
- Electro-sedimentation: This is due to the increase in solids that have been removed and their higher density.
- Disinfection: It can occur if sodium chloride is added as an electrolyte or if chlorinated compounds are present in the water.

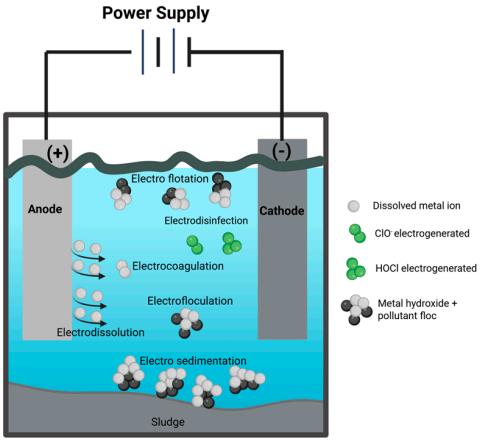


Figure 3. Processes involved in electrocoagulation (Created in BioRender. Sol, L. (2025) https://BioRender.com/1anke7f).

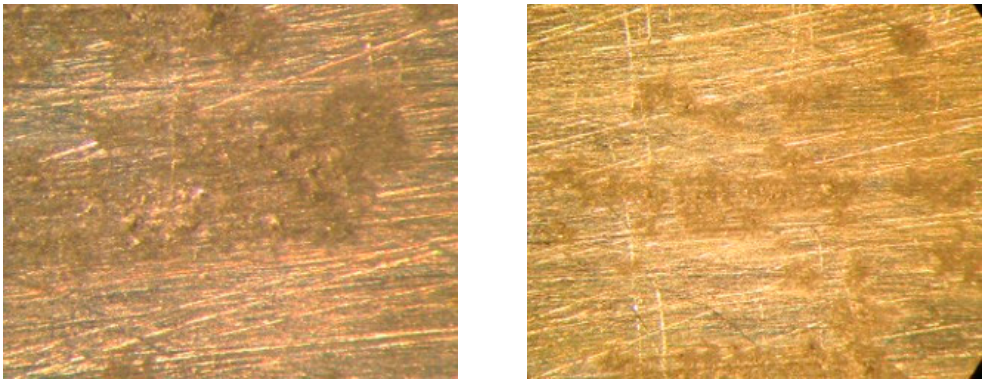
3.3. Considerations for Water Treatment Application

Several parameters must be considered when treating water with electrocoagulation for the elimination of antibiotics, microorganisms, and antibacterial resistance genes.

3.3.1. Material

The choice of material is crucial, since better efficiency in contaminant removal is partly due to the electrode generating the electromotive force (e.m.f.) necessary to provide metal ions that form the compounds described in Section 2, which promote the separation of the contaminant charge from the fluid. The most employed materials are aluminum and iron, due to their high electromotive force capabilities (e.m.f.), low cost, and ease of substitution.

Aluminum is favored for its ability to dissolve both chemically and electrochemically (Figure 4). This phenomenon is evident when an aluminum anode and cathode are used; the cathode exhibits slight corrosion due to chemical dissolution. This dissolution rate is further accelerated by an alkaline pH, which is promoted by the reduction of hydroxyl ions (OH^-) at the cathode [36].



(**a**) (**b**)

Figure 4. Aluminum electrodes are seen at the microscope. The corrosion is through: (**a**) Electrochemical dissolution at the anode; and (**b**) Chemical dissolution at the cathode.

Another combination frequently used in EC is that of iron/iron, either through pure iron electrodes or through materials such as carbon steel or stainless steel.

Table 1 summarizes recent applications of electrocoagulation, specifically highlighting the influence of electrode material on contaminant removal efficiency, with a particular focus on antibiotic compounds. Notably, both aluminum (Al) and iron (Fe) based electrodes demonstrate high efficacy in degrading complex organic pollutants, including pharmaceutical compounds and antibiotics.

For instance, the combination of Fe as an anode and Al as a cathode achieved a 92.3% reduction in chemical oxygen demand (COD) from real pharmaceutical wastewater, indicating its strong capability for treating complex organic matrices [37]. Similarly, electrocoagulation using Al or Fe electrodes has proven highly effective in removing tetracycline (TC) and its metal complexes (TC-Ni and TC-Cu) from synthetic spiked wastewater, achieving removal rates of nearly 100% in several cases [38]. The high efficacy of both Al and Fe electrodes in targeting tetracycline and its complexes indicates their potential in mitigating antibiotic pollution. Furthermore, a study utilizing Al/Fe electrodes successfully removed 94.6% of cefazolin from hospital effluents, demonstrating the applicability of these material combinations for specific antibiotic remediation in real-world scenarios [39]. While stainless steel (SS) and Al combinations have been explored for disinfection, the evidence presented suggests that iron hydroxide, generated from Fe electrodes, exhibits higher

inactivating properties against Escherichia coli compared to Al [40]. This suggests that the specific mechanism and efficiency can vary depending on the electrode material and target contaminant. The selection of electrode material also appears to be influenced by the initial pH, as indicated by studies on the disinfection of wastewater effluents using aluminum (Al) and commercial steel (CS) electrodes [41]. These findings indicate the key role of electrode material selection in optimizing electrocoagulation performance for diverse water treatment applications, particularly for antibiotic degradation.

Table 1. Influence of electrode material on the removal of antibiotics and microorganisms.

Electrode Material	Applicability	Type of Water	Target Pollutant	Pollutant Load	Removal	Findings	References
Al/Al Fe/Fe Fe/Al	Wastewater treatment	Real	COD of pharmaceutical compounds	7692 mgL^{-1}	92.3%	The best combination is Fe/Al	[37]
SS and Al	Disinfection	Synthetic WW	*E. coli*	5×10^5 UFC/100 mL	100%	Iron hydroxide was a better inactivant	[40]
Al CS	Disinfection of wastewater effluents	Spiked water	*E. coli*	5×10^5–4×10^7 UFC/100 mL	More than $5log_{10}$	Selection of material depends on the initial pH	[41]
Al/Fe	HWW	Real spiked water	CFZ	0.423 mgL^{-1}	94.6%	Able to remove cefazolin	[42]
Al Fe	Effluents with the formation of recalcitrant metal complexes	Synthetic spiked WW	TC TC:Ni	15 mgL^{-1}	TC: Fe: 99.3% Al: 99.8% TC-N: 1:1: 100% 1:2: 99.6%	The ratio TC:Ni influences the removal efficiency.	[38]
Carbon steel anode	Effluents with the formation of recalcitrant metal complexes	Synthetic spiked WW	TC TC:Cu	24.05 mgL^{-1}	TC-100% TOC-80.2% Cu^{2+}-8.1%	Promising to remove TC-Cu complexes	[42]
Al and SS 304 anodes	WW	Real spiked wáter	AMX TMP	10 mgL^{-1}	Al: AMX 21.52% AMX + TMP 7.84% SS 304: TMP 13.10%	Determination of corrosion velocities	[43]
Al Fe	GW	Real	COD COT *E. coli*	COD (460 mgL^{-1}) COT (185 mgL^{-1}) *E. coli*- ($2-2.8 \times 10^3$ CFU/100 mL)	EC + O_3: Al: COD 65% Fe: COD 85.8% TOC 71.14% *E. coli* 85.42% *EC (Fe) +O3 +* UV: COD 95.65% COT 87.35% *E. coli* 96.88%	Better removals with the iron electrode	[44]

Table 1. *Cont.*

Electrode Material	Applicability	Type of Water	Target Pollutant	Pollutant Load	Removal	Findings	References
Al And LCS Anodes	MWW	Spiked synthetic	AMP DOX STZ Tylosin	50 mgL^{-1} each	AMP 3.6 ± 3.2% DOX ~100% STZ 3.3 ± 0.4% Tylosin 3.1 ± 0.3%	DOX was the only antibiotic effectively removed	[17]
Fe Al	WW	Simulated	TC COT	0.05 $mmolL^{-1}$	Fe: TC-99.6% COT-79.8% Al: TC-97% COT-77%	The iron electrode showed higher performance.	[45]
For bath water: Al/Al For laundry water: Al/Fe (mild steel)	GW	Bath water (synthetic) Laundry water (synthetic and real spiked water)	*E. coli* BOD	Synthetic bath water: BOD 159 mgL^{-1} Synthetic laundry water: BOD 243 mgL^{-1} Spiked real laundry water: *E. Coli*: 105.6 $CFUmL^{-1}$	Bath water: BOD-51.8% Laundry water: *E. coli*: ≥6.1 log reduction	Al/Al and Al/Fe had the best results	[46]
Al/N doped porous carbon loaded with Co/Fe sites (N-Co/Fe-PC)	WW containing copper and antibiotics	Synthetic	Cu CIP COT	20 mgL^{-1}	Cu-99.69% CIP-96.40% COT-83.62%	The cathode material withstands up to 6 times of reuse. Loss of 15%.	[47]
Fe/carbon felt, Al, SS cathodes	DWW	Real	*E. coli*	392 ± 100 × 10^6 MPN mL^{-1}	99.99%	Optimal combination: an iron anode and a carbon felt cathode.	[48]
Al/Al (non-insulated and Insulated)	PWW	Synthetic	CFX	30.16 mgL^{-1} 34.26 mgL^{-1}	88.21% 81.73%	Insulated electrodes reduce the dispersion of electrical current,	[49]

Table 1. *Cont.*

Electrode Material	Applicability	Type of Water	Target Pollutant	Pollutant Load	Removal	Findings	References
Fe/Fe	RW	Simulated	*E. coli* Total coliforms Enterococci Phages	Total Coliforms 5 log(MNP/100 mL) *E. coli* 4.5 log(MNP/100 mL) Enterococci 3.5 log(MNP/100 mL) Somatic Coliphage: 3.5 log(MNP/100 mL)	*E. coli*: 1.7 log Total coliforms: 1.5 log Enterococci: 1.0 log Phages: 2.0 log	Fe/Fe combination was the best option.	[50]

3.3.2. Electrode Distance

The distance between electrodes is a critical design parameter for electrocoagulation reactors, as it directly influences system efficiency and operational costs. A greater inter-electrode distance can increase electrical resistance, consequently prolonging the time required for ion charge transmission and slowing the pollutant removal process. This, in turn, necessitates higher energy input, leading to increased energy consumption and operational expenses.

The optimal distance should be determined based on the specific characteristics of the water being treated, the electrode configuration, the applied electrical conditions, and the reactor design. Generally, a maximum inter-electrode distance of 7 cm and a minimum of 0.02 cm is proposed [51].

Regarding the treatment of waters containing antibiotics, microorganisms, and antibiotic-resistant bacteria (ARB), various studies have evaluated this parameter. As shown in Table 2, electrode spacing has a significant influence on contaminant removal efficiency in electrochemical treatment systems. Distances of 0.5 to 1.0 cm (e.g., steel/steel and aluminum-based systems) resulted in high removal rates, achieving up to 100% elimination of *E. coli*, ciprofloxacin, and cefazolin in both synthetic and real wastewater matrices. Intermediate spacings, such as 1.25 to 2.0 cm, were also effective, particularly for complexed tetracycline species and azithromycin, with removal efficiencies above 92%. Even at the widest spacing (3.0 cm, Fe/Fe electrodes), complete removal of metronidazole was achieved, likely due to the favorable reaction conditions and properties of the pollutant.

Table 2. Electrode spacing for removal of antibiotics and microorganisms in environmental matrices.

Electrode Material (Anode/Cathode)	Electrode Spacing (cm)	Applicability	Type of Water	Target Pollutant	Pollutant Load	Removal	References
Al/Al	1.0	WW	Synthetic	CPX	50 mgL^{-1}	98.48%	[52]
Al/Fe	2.0	WW with recalcitrant metal complexes	Synthetic	TC TC:Cu	15 mgL^{-1}	TC: >99% TC:Cu: 1:1-100% 1:2-99.6%	[38]

Table 2. *Cont.*

Electrode Material (Anode/Cathode)	Electrode Spacing (cm)	Applicability	Type of Water	Target Pollutant	Pollutant Load	Removal	References
Fe/Fe	3.0	PWW	Synthetic	MZN	21.6 mgL^{-1}	100%	[53]
Steel/Steel	0.5	WW	Real	*E. coli*	-	96%	[54]
Al anode	0.5	PW	Synthetic	*E. coli*	105 $UFCL^{-1}$	100%	[55]
Al (chitosan as adsorbent)	1.0	HWW	Synthetic	CFZ	60 mgL^{-1}	100%	[56]
Fe/Fe	1.58	HWW	Simulated	CIP	60 mgL^{-1}	100%	[57]
Fe/Fe	1.25	HWW	Real	AZM	-	92.3%	[58]
Al/Al	1.0	DCWW	Synthetic and real	CIP	Synthetic sample: 32.5 mg L^{-1} Real sample: 154 $\pm$ 6 µg L^{-1}	>88.00%	[59]
Fe/Fe	1.5	DCWW	Simulated	CAP	30 mgL^{-1}	PSPC-EC: 98.85% DC-EC: 98.28% APC-EC: 98.36%	[60]
Perforated SS sheets	0.5	WW	Synthetic	CIP LVX	25 mgL^{-1}	CIP: 93.47% LVX: 88% CIP:LVX: 1:1- 93.0:91.8% 1:4- 90.10:96.10% 4:1- 96.30%:92.97%	[61]
Al/Pt	4.0	SW	Simulated	*E. coli K-12* with plasmid RP4 carrying blaTEM, tetR and aphA	1×10^8 $CFUmL^{-1}$	ARB-3.04 log reduction	[62]

3.3.3. Number of Electrodes

This parameter mainly influences:

- The amount of metal ions to be supplied in the water matrix. A higher number of electrodes will promote a higher number of metal ions, resulting in better removal of the pollutant load. However, producing an excess of dissolved metals carries risks: it can significantly increase the volume of sludge produced, or, if the metals do not fully react, the treated liquid will retain a high concentration of unwanted metals.
- Electrical requirements: A greater presence of electrodes reduces the amount of energy supplied, since it no longer requires a large force to produce the metal ions. This primarily impacts the economic analysis, as lower energy consumption implies lower costs and makes the EC process more cost-effective.

Although none of the studies in this review explicitly assess the number of electrodes as an experimental variable, several report configurations that suggest its relevance. Some authors have employed different conditions, such as using six electrodes, which achieved a very low current density of 0.00000222 mA cm^{-2}, yet still obtained 98% ciprofloxacin and 87% total organic carbon removal [63]. Others used twelve electrodes to accommodate

the design of a continuous pilot-scale reactor [64]. Also used six aluminum electrodes, following the specifications of a 6 L electrocoagulation reactor [52].

3.3.4. Voltage Supplied

Electrical indicators, such as voltage and current intensity, are parameters associated with the cost of treatment, as they can be used to calculate the energy efficiency of the process. Electrical potential is considered when scaling is desired, since, although very efficient results in terms of pollutant load removal are achieved, i.e., 80% or more, it may turn out that the optimum is the one where the energy consumption parameter is lower. However, the chemical oxygen demand removal is not the highest. For example, a techno-economic optimization study was conducted for the removal of microplastics and benzyldimethyldodecylammonium chloride (DDBAC), a drug widely used during the COVID-19 pandemic. They evaluated the performance of electrocoagulation by applying voltages of 3, 6, 9, 12, and 15 volts in different electrode connection configurations: series monopolar, parallel monopolar, and series bipolar. Through ANOVA and RSM statistical analysis, they concluded that the optimum conditions were parallel monopolar at 6 V, with a removal efficiency of 60.3% [65].

In other research, a voltage range of 4.5–12 V was applied to remove tetracycline and its complexes with nickel ions in a synthetic aqueous matrix. They observed that removal efficiency increased progressively from 4.5 to 9.0 V; however, at 12 V, efficiency declined. This trend is attributed to the behavior of metal ion generation and transfer: at voltages between 4.5 and 9.0 V, the increase in electron flow enhances the release of metal ions into the solution, promoting contaminant removal. In contrast, beyond 9.0 V, the elevated concentration of metal ions favors side reactions, including the polymerization of aluminum hydroxide, which can hinder the adsorption of contaminants onto the resulting flocs [38].

Additionally, this decrease in efficiency may be linked to the behavior of charge carriers. As the electrical potential increases, a greater number of ions are consumed to sustain the process, leading to a reduction in the availability of electrons at the anode. This limits the electron transfer to the cathode, thus impairing the efficiency of reduction reactions. A similar trend was reported by [66], who tested 3.2 V and 6 V for the removal of total coliforms, fecal coliforms, and *Staphylococcus* spp., with the lower voltage resulting in better removal efficiency.

On the other hand, the increase in voltage does favor the removal of turbidity, suspended solids, or organic matter, since the polymeric forms of the metal hydroxide, having greater size and weight, drag the solids more easily. This phenomenon was observed when parameters affecting the degradation of cefazolin in hospital wastewater were investigated. In their study, they observed that by increasing the voltage from 15 V, 30 V, and 50 V, at an initial pH of 4, turbidity removal also increased, obtaining efficiencies of 74.69%, 84.03%, and 88.25%, respectively [39].

3.3.5. Current Intensity or Current Density—Which Parameter Is More Appropriate?

Due to the similarity of terms, the distinction between these parameters is often confusing, so it is necessary to define them clearly. Current intensity (I) is an electrical parameter that indicates the number of charges passing through a conductor over time, while current density (j) is the ratio of current intensity to the geometric area available at the electrodes.

In electrochemical studies, it is essential to define which electrical parameter will be controlled and which will be measured. When voltage is used as the control to perform the

electrocoagulation process, it is easier to measure the current intensity, as the devices used, such as multimeters, measure the former parameter rather than the current density.

However, if the control parameter will be the current intensity, in this case, it is important to express it in the form of current density since it indicates that the geometric area of the electrode available for the passage of current is also being controlled, thus facilitating experimental reproducibility and the influence of the available area of the electrode on the development of electrochemically assisted coagulation.

In electrochemical coagulation studies, it is more common to employ current density; therefore, the proposed range is from 10 to 150 Am^{-2} to carry out EC successfully [51]. It should be noted that the value of the current density depends primarily on the type of water to be treated, the material used, the number of electrodes in the reactor, and the ionic strength or conductivity of the solution being treated.

Table 3 provides an overview of the influence of applied current—reported either as intensity or density—on electrocoagulation performance. Alongside removal efficiencies, the table also details contaminant load, type of water, specific applications, and the optimal electrical conditions identified in each study. While a wide range of current values was reported, from as low as 0.3 mA for simulated wastewater containing tetracycline [45] to 1300 mA for domestic wetland effluent [67], removal efficiencies remained high across cases. Notably, low current intensity (0.3 mA) was sufficient to achieve 99.6% removal of tetracycline, indicating that under favorable synthetic conditions and low pollutant loads, minimal electrical input can yield effective results. Conversely, higher intensities (200–1300 mA) were employed in real and more complex matrices, particularly for microbiological contaminants such as total and fecal coliforms, where complete or near-complete removals were achieved. As is already known, the optimal current intensity depends not only on the type of pollutant but also on the complexity of the water matrix. While current density values were not always reported, the range of applied intensities reflects the need to balance sufficient generation of reactive species with energy efficiency and electrode stability.

Table 3. Current intensity and current density values applied to treat different types of antibiotics and microorganisms.

Target Pollutant	Removal	Applicability	Type of Water	Type of Current (I = Electrical Intensity; j = Current Density)	Optimal Value	References
COD of pharmaceutical compounds	92.3%	Effluents of PWW	Real	I (DC power supply)	40 mA	[37]
Total coliforms *E. coli*	2.3 $\log_{10}$ 2.35 $\log_{10}$	Vertical wetland effluent from domestic water	Real	I (DC power supply)	1300 mA	[67]
TC	99.6%	WW	Simulated	I (DC power supply)	0.3 mA	[45]
TC	95%	Livestock WW	Synthetic	I (Positive single pulse current, PSPC)	200 mA	[68]
CQ	95%	PWW	Spiked tap water	j (DC power supply)	66.89 $mAcm^{-2}$	[69]

Table 3. *Cont.*

Target Pollutant	Removal	Applicability	Type of Water	Type of Current (I = Electrical Intensity; j = Current Density)	Optimal Value	References
Fecal contamination (total coliforms, fecal coliforms and enterococci)	100%	WW reuse for agriculture	Real	I (AC power supply)	500 mA	[70]
COD (β-lactam antibiotic derivatives), and TC antibiotics	75.64%	PWW	Real	j (DC power supply)	46.83 $mAcm^{-2}$	[71]
CIP	98.48%	WW	Synthetic	j (DC power supply)	1.5 $mAcm^{-2}$	[52]
E. coli	GW and TW: 2.22–2.53 log_{10} units Tap water: 3.80 log_{10} units	GW TW Tap water	Real Real Spiked	j (DC power supply)	1.0 $mAcm^{-2}$	[72]
TC TOC TC-Cu	TC-100% TOC-80.2% Cu^{2+}-88.1%	Water with metal–organic complexes	Synthetic spiked	j (DC power supply)	4.17×10^{-7} $mAcm^{-2}$	[42]
E. coli Total coliforms	0% 26.09%	Poultry slaughterhouse WW	Real	j (DC power supply)	Raw water: 20 $mAcm^{-2}$ Polished water: 30 $mAcm^{-2}$	[73]
MNZ	57.30% and 41.70%	Effluent disinfection	Synthetic	j (DC power supply)	40 Am^{-2}	[74]
CIP TOC	98% 87%	WW	Synthetic	j (DC power supply)	22.2 Am^{-2}	[63]
E. coli	100%	Reclamation of urban TWW	Real	j (DC power supply)	5–7 Am^{-2}	[75]
AMX COD TOC	Synthetic water: AMX-90.56% COD-65.5% TOC-44.5% Real hospital wastewater: DQO-47.7% TOC-38%	HWW	Synthetic Real	j (DC power supply)	2.31 $mAcm^{-2}$	[76]

3.3.6. Initial pH

pH plays a crucial role in the development and efficiency of electrocoagulation (EC) processes, as its control and variation directly affect the amount of aluminum ions released into the solution. To understand how pH evolves during treatment, it is essential to consider the spatial distribution of pH in three distinct zones within the reactor [77]: (i) near the anode, where the pH tends to be acidic due to the oxidation of water and the release of H^+

ions; (ii) near the cathode, where the pH is more basic as a result of water reduction and OH^- generation; and (iii) in the bulk solution, where pH is governed by the interaction between electrochemically generated species and the physicochemical properties of the contaminants, leading predominantly to chemical reactions and adsorption phenomena rather than direct electrochemical transformations. Recognizing these spatial variations is key to understanding the buffering effects near the electrodes, as well as the overall behavior of pH in the system, which in turn depends on the nature and chemistry of the target compounds to be removed.

Although pH is a critical parameter in electrocoagulation (EC) processes, there is still no clear consensus on which electrical variable should be prioritized in degradation assays. A valuable tool to address this relationship is the Pourbaix thermodynamic corrosion diagram (Figure 5), which illustrates how pH and electrode potential (in volts) influence the formation of metallic species that can either promote or hinder EC performance. These diagrams provide a preliminary understanding of the conditions under which metal ions—such as Fe^{3+} or Al^{3+}—are released into solution, enabling the formation of coagulant salts that facilitate the aggregation and removal of pollutants. Additionally, they help identify pH and potential ranges that favor the formation of metal oxides, which should be avoided due to their tendency to form passivating layers on the electrodes. Such layers obstruct electron transfer, reduce contaminant removal efficiency, and increase energy consumption due to elevated electrical resistance.

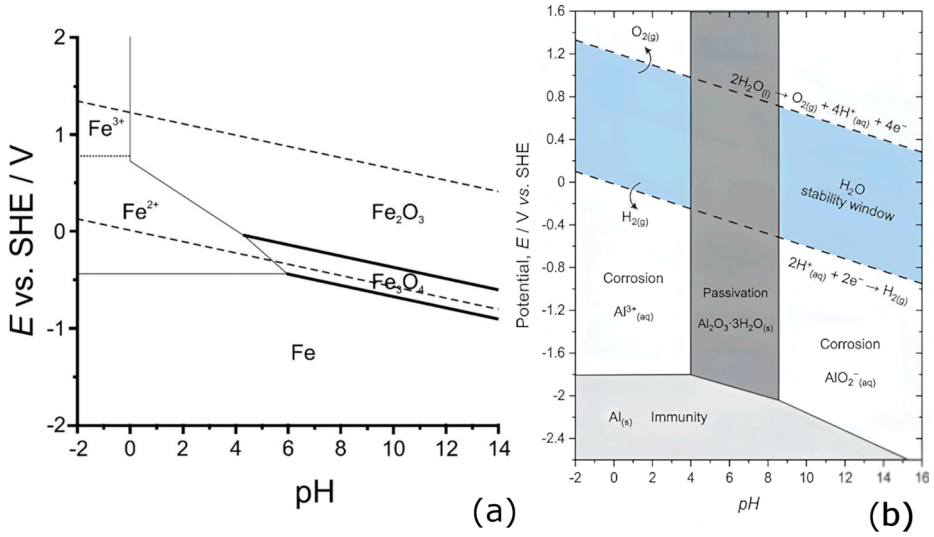


Figure 5. Pourbaix diagrams (**a**) For iron in water; (**b**) For aluminum in water at 25 °C. Reprinted from reference [78,79] with permission from IOPScience.

As illustrated in the Pourbaix diagrams, an initially acidic pH is favorable for electrocoagulation, as it promotes the electrochemical dissolution of metal electrodes and the subsequent release of metal ions, such as Al^{3+}, required for the formation of aluminum hydroxide complexes. Moreover, when the initial pH ranges between 3 and 6, a gradual increase in pH is commonly observed during the process. This shift can be attributed to three main factors: (i) the evolution of hydrogen gas at the cathode, which consumes protons and thereby reduces acidity; (ii) the degassing of dissolved CO_2, which under acidic conditions exists predominantly as carbonic acid (H_2CO_3), and whose release into the atmosphere reduces the concentration of acidic species; and (iii) the presence of anions such as Cl^- and SO_4^{2-}, which can compete with hydroxide ions for complexation with Al^{3+} or displace OH^- from the solid phase, thus contributing to an increase in the basicity of the solution.

Stainless steel has been used as a source of iron ions to generate a varied initial pH value, ranging from 6.2 to 3.8, for the removal of ciprofloxacin and levofloxacin [80]. It was observed that as the value decreased, the removal of both antibiotics increased. This is consistent with the Pourbaix diagrams, as at more acidic pH values, a greater amount of Fe^{3+} and Fe^{2+} ions are generated electrochemically, which favors the formation of the chemical species $Fe(OH)_2^{3-}$ y $Fe(OH)_4^-$. In addition, considering the chemistry of the quinolones, ciprofloxacin and levofloxacin are found in their cationic form, which facilitates the precipitation of these antibiotics.

On the other hand, aluminum electrodes have also demonstrated high performance in electrocoagulation. For example, the highest removal efficiency of chloroquine was achieved using aluminum electrodes at 60 min and under slightly acidic conditions (pH 6.5) [69].

Finally, other drugs, such as chloramphenicol, exhibit improved removal efficiency at pH values above 8 [60]. This enhancement is linked to electrochemical factors, particularly the overcoming of the passivation window of aluminum, which typically occurs between pH 4 and 8 due to the formation of a protective aluminum oxide layer. At an alkaline pH, this passive layer becomes unstable or dissolves, allowing for greater anodic dissolution and promoting the formation of soluble species, such as AlO_2^-. These species are chemically more effective in interacting with and removing certain antibiotics, such as chloramphenicol.

3.3.7. Electrical Conductivity and Molar Conductivity

In an electrolytic solution, ions move randomly, constantly, and unpredictably. But when a potential difference is applied, they start to move more directionally, depending on their charge. In that situation, the solution behaves like an electronic conductor, as described by Ohm's Law, which links current, voltage, and resistance.

In real electrochemical processes, measuring resistance directly is not very practical. Instead, conductivity (κ) or ionic strength is usually monitored. These parameters increase with the concentration of the electrolyte, which in turn facilitates the electrocoagulation process. With lower resistance to current flow, the metal ions generated at the anode can more easily migrate and react near the cathode, forming species that help drive coagulation and flocculation of pollutants.

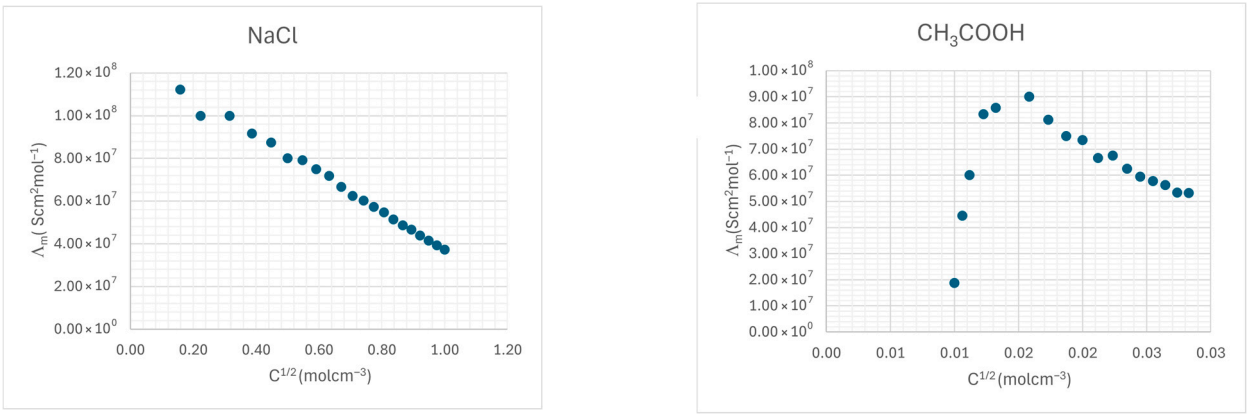
The value of κ varies with the nature and concentration of the solute. Therefore, if one solution of an electrolyte has a higher concentration than another, the more concentrated one will have higher conductivity because it has more ions. However, it has been observed that, on some occasions, when the ionic strength increases, the EC is not as favored as expected, as the removal efficiency may remain unchanged or decrease.

The influence of sodium chloride as a supporting electrolyte was analyzed at concentrations of 0.048, 0.096, and 0.192 M [38]. The findings demonstrated that at the lowest concentration, the removal efficiency of tetracycline with an iron electrode increased from 92.6% in 10 min to 95.5% in 60 min. At the medium value, the removal was 97.9% in 40 min, and at the highest level, there was no variation.

Others used a positive single pulse current (PSPC)-EC system to remove tetracycline [68]. They observed that when the conductivity increased from 0.5 to 2 mS cm^{-1}, the removal efficiency rose from 49.82% to 87.80%. However, when the conductivity increased to 3 $mScm^{-1}$, the removal rate dropped to 90.34%. Moreover, from this value, the removal of tetracycline increased and decreased.

The effect of supporting electrolyte concentration on the removal of ciprofloxacin was also investigated [52]. They used sodium chloride at concentrations ranging from 100 to 600 mgL^{-1}, increasing in 100 mgL^{-1} increments, and observed that the removal

efficiency rose from 61.25% to 98.48% when the NaCl concentration was increased from 100 to 500 mgL^{-1}, but then declined slightly to 95% at 600 mgL^{-1}. This behavior can be better understood through the concept of molar (or equivalent) conductivity, Λ_m, defined as the ratio of electrical conductivity κ to molar concentration c. While strong electrolytes like NaCl improve ionic strength and electrical conductivity up to a certain point, excess concentration promotes ion pairing and association, reducing the effective number of charge carriers and, thus, decreasing Λ_m as a result, despite higher κ, the current-carrying efficiency per mole of electrolyte drops, increasing resistance to current flow and ultimately limiting electrochemical performance (Figure 6).



(**a**) (**b**)

Figure 6. Relationship between molar concentration and molar conductivity. (**a**) For strong electrolytes such as sodium chloride, as concentration increases, molar conductivity decreases significantly due to increased ion–ion interactions and the formation of ion pairs, which hinder ionic mobility and increase electrical resistance; (**b**) For weak electrolytes like acetic acid, molar conductivity does not drop drastically with increasing concentration, due to its limited dissociation and different ionization behavior. (Graphs created based on data from [81,82]).

3.3.8. Characteristics of the EC Reactor

There are several designs for the electrocoagulation process. In general, the most used are batch and continuous flow. As for the geometry of the system, the most common are rectangular and cylindrical.

However, a reactor design that optimizes electrochemically assisted coagulation has not yet been determined. However, reactor design considerations have been defined and are as follows:

- Start with a small-sized reactor (laboratory or pilot scale) to perform the EC tests.
- Since, in most cases, foams are generated that contain contaminants, the use of mechanical paddles at the top of the reactor can be considered to remove the foam formed on the surface.
- A drain cock at the bottom of the reactor for the separation of sediment and treated wastewater.
- In continuous flow reactors, excessive turbulence must be avoided at both the inlet and outlet to prevent the floccules from breaking. In this case, the arrangement of the electrodes is a parameter to consider, as authors such as those have perforated the electrodes, which allows for adequate turbulence to favor the flocculation-coagulation process. In this case, the arrangement of electrodes is a parameter to consider, since, for example, in Figure 7, the use of perforated electrodes is illustrated, which allows adequate turbulence to improve the flocculation-coagulation process [54].

- Agitation speed: If it is too fast, it will break the flocs formed. If it is too slow, it may not favor the formation of agglomerates. In the case of antibiotic removal, Table 4 helps define the speed, depending on the reactor volume.
- The configuration and geometry of the electrodes: These are key factors that directly impact treatment performance. The electrode arrangement (e.g., monopolar or bipolar), spacing, orientation, and surface area must be carefully considered, as they significantly influence pollutant removal efficiency, energy consumption, and electrode durability. An optimized electrode setup can enhance current distribution, reduce internal resistance, and limit electrode passivation, contributing to a more efficient, cost-effective, and long-lasting reactor system (Figure 8).
- If possible, design a reactor with two cells (especially if the flow is continuous or a batch process with feedback), where electro-dissociation of the electrode is carried out in one cell and flocculation, coagulation, and sedimentation in the other. The above is to avoid interference in the analysis of the samples taken (Figure 9).
- If the electrocoagulation process is to be coupled with another treatment that is carried out simultaneously, the needs of both methods must be considered.

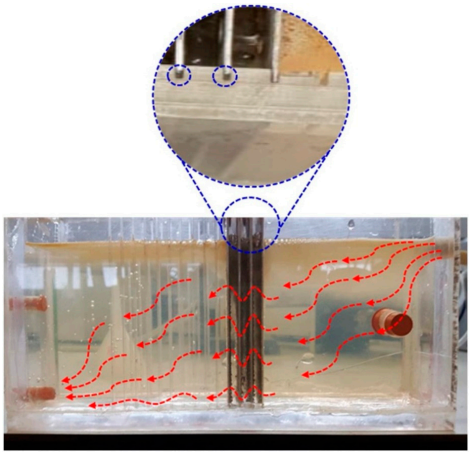


Figure 7. Perforated electrodes in a flow reactor help to promote agitation and improve the flocculation–coagulation process. Reprinted from reference [54] with permission from Elsevier.

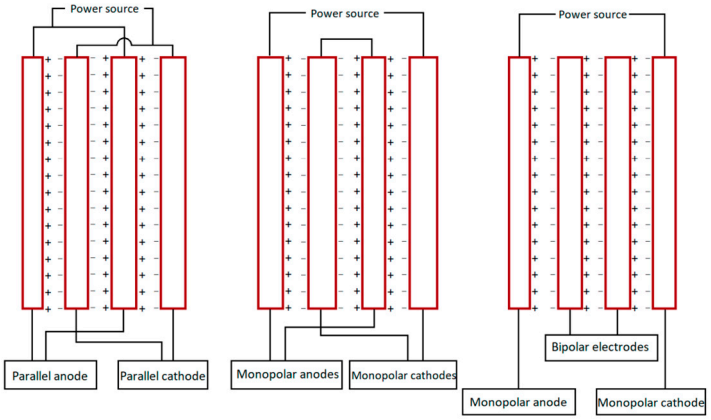


Figure 8. Different types of electrode connection, reprinted from reference [29] with permission from MDPI.

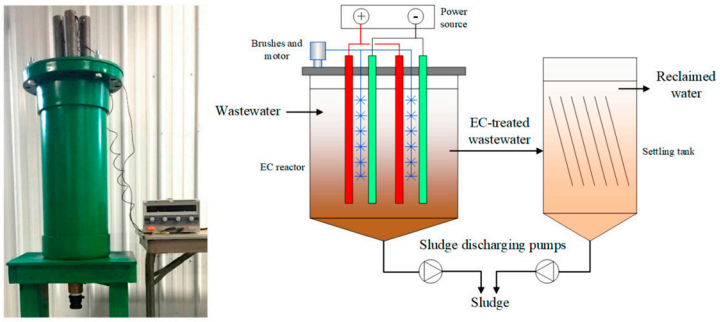


Figure 9. Diagram of EC reactor and settling tank with inclined baffles to facilitate sedimentation velocity reprinted from reference [50] with permission from MDPI.

Table 4. Range of speeds used for the treatment of different drugs, microorganisms and ARGs by electrocoagulation in batch reactors.

Target Pollutant	Volume Range (L)	Stirrer	Speed (rpm)
E. coli, TC, AMX, MZN, total coliforms, OFL, enterococci, clostridium perfringens spores, somatic coliphages and eukaryotes, CIP, ARGs, AMP, DOX, STZ, tylosin and CTX	0.100–0.250	Magnetic bar	70–400
E. coli, AMX, TMP, CAP, MZN, TC, DDBAC, ARGs, Enterococci and phages	0.250–0.500	Magnetic bar	100–600
E. coli, CIP, LVX, ARGs, OFL and CAP	0.500–1.00	Magnetic bar	120–1100
CFZ, TC, *E. coli*, coliphage ΦX174, CIP and RhB	1.50–2.00	ND	200–1000
CQ and CFX	2.00–3.00	ND	600

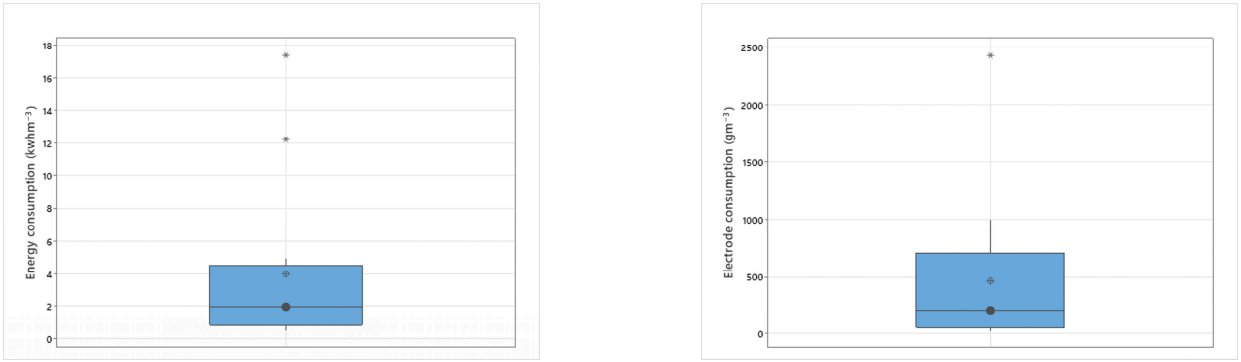
3.3.9. Energy and Electrode Consumption

One of the main limitations of electrocoagulation is its energy consumption, which can become a critical factor in large-scale or long-term applications. For this reason, calculating and monitoring energy usage is essential, as it allows for the assessment of process efficiency by linking the electrical power input to the pollutant removal performance. This relationship not only supports the optimization of operational parameters but also provides valuable insight into the economic and environmental feasibility of the treatment system.

The equations used to calculate this parameter are based on the figures of merit for the technical development [83]. Some authors consider the voltage parameter, while others prefer the current intensity. Some use the current density to relate the current intensity to the available electrode area, and others report the value determined by electrical power. Regardless of the electrical parameter, they all relate to the amount of energy applied to remove the target contaminant load.

Figure 10a illustrates the reported range of energy consumption by electrocoagulation for treating antibiotics and microorganisms [39,40,43,49,50,53,56,57,60,61,64,66,69,75,84,85]. It shows that the average power consumption is 4.014 kWh m^{-3} (σ = 5.095), with an interquartile range of 0.908 to 3.658 kWh m^{-3}, which more robustly represents typical process conditions. However, the presence of outliers such as 12, 18, and 75 kWhm^{-3} indicates the need to analyze operating conditions.

On the other hand, the consumption of the electrode enables the determination of the amount of aluminum electrolyzed into the solution, allowing for the monitoring of the metal hydroxide generated, the amount of aluminum in the sludge, and the concentration of aluminum remaining in the aqueous matrix after treatment.



(**a**) (**b**)

Figure 10. (**a**) Range of values for energy consumption; (**b**) Range of reported values for electrode consumption. Data compiled from multiple studies referenced in the text. * at the top indicates the maximum value of the data range.

The amount of metal dissolved over a given time can be estimated using Faraday's law (Equation (1)), which relates the mass of a substance altered at an electrode to the total electric charge passed through the system. This calculation considers the total charge transferred ($C \cdot s^{-1}$), the valence of the metal ion (z), and Faraday's constant (F), which represents the charge per mole of electrons.

$$n = \frac{I * t}{zF} \quad (1)$$

Based on this, Figure 10b shows the amounts of dissolved mass of the sacrificial electrode for the removal of antibiotics, microorganisms, and resistance genes are reported to range from 24 to 2435 g/m^3, with an average value of 466.87 (σ = 644.6) [37–39,43,56,57,59,61,64,69,80,85].

Monitoring the concentration of metal released from the anode is essential. On the one hand, it provides a basis for estimating when the anode should be replaced; on the other, it allows for quantifying the residual metal that may remain in the treated water. This latter aspect is particularly relevant, as the concentration of metals such as aluminum or iron can become a limiting factor in the treatment process due to regulatory limits for their presence in water.

Although these studies have demonstrated the effectiveness of EC for removing either antibiotics or ARB, its performance in addressing both contaminant types simultaneously—under realistic, mixed-effluent conditions—remains insufficiently explored. This is a relevant gap, particularly considering the frequent co-occurrence and mutual reinforcement of antibiotics and ARB in wastewater from hospitals, livestock operations, and pharmaceutical facilities.

Mechanistically, the operational conditions commonly used for antibiotic removal—such as current densities between 10 and 30 $mAcm^{-2}$, near-neutral pH, and treatment durations of 20–60 min—are also suitable for destabilizing and flocculating microbial cells. These conditions promote the in situ generation of coagulant species (e.g., $Fe(OH)_3$, $Al(OH)_3$) and reactive intermediates (e.g., hydroxyl radicals, Fe^{2+}/Fe^{3+} redox cycling), which enhance both molecular adsorption and microbial inactivation. As shown in Tables 3 and 4, many of the parameters used for either antibiotic or ARB removal overlap, suggesting that a unified EC system could, in theory, address both targets simultaneously under optimized conditions.

Nonetheless, in real wastewater matrices—where natural organic matter (NOM), surfactants, metals, nutrients, and other emerging contaminants are present—EC performance can be affected by competitive interactions. NOM and other dissolved organics may reduce

the availability of coagulant active sites or scavenge reactive species, ultimately diminishing removal efficiency. Additionally, while EC can partially degrade or transform some antibiotics, it generally does not achieve complete mineralization of persistent compounds, such as fluorinated pharmaceuticals, hormones, or polyaromatic structures. In this context, combined or sequential treatment strategies emerge as a practical and technically sound solution.

3.3.10. EC Operational Costs

The determination of operating costs is a crucial parameter for the application and scalability of treatment technologies. In the case of electrocoagulation (EC), costs include electrode consumption (ELC), energy consumption (EEC), and chemical consumption (CC) required to adjust pH (e.g., HCl, NaOH, or H_2SO_4) or electrical conductivity (e.g., NaCl, Na_2SO_4, KCl, or $NaNO_3$) [62]. These factors are typically considered at the laboratory scale. However, when moving to pilot or industrial scales, additional parameters such as labor (L), maintenance (M), sludge management (SM), and construction (C) must also be included [54].

The general expression for calculating operational costs is:

$$\text{Operational cost} = aELC + bEEC + cCC + dL + eM + gSM + hC \quad (2)$$

where the coefficients represent the relative contribution of each parameter.

Reported operational costs for EC vary widely, ranging from 0.011 to 4.13 US$m^{-3}. For microorganism inactivation, one study reported a cost of 0.0855 US$m^{-3}, slightly higher than nanofiltration but still competitive with desalination. Moreover, the treated water was suitable for agricultural reuse, specifically cucumber irrigation [66]. Another study reported 0.11 US$m^{-3} for *E. coli* inactivation using aluminum electrodes, while higher values (2.71 US$m^{-3}) were observed for total coliform and *E. coli* removal under similar conditions [54]. Nonetheless, an average operational cost of around 0.2 US$m^{-3} has been suggested as a representative value for EC [55].

Regarding antibiotic removal, chloroquine was eliminated with 95% efficiency using aluminum electrodes at an operational cost of 2.48 US$m^{-3} for a 3 L batch reactor and a 60 min treatment time [69]. In contrast, tetracycline removal achieved 100% efficiency with carbon steel electrodes in a 1 L batch reactor after only 15 min, at a much lower cost of 0.05 US$m^{-3} [42]. Similarly, 95% removal of tetracycline was reported at 0.011 US$m^{-3} using iron electrodes in a 0.5 L batch reactor with 30 min of treatment [68].

As in most electrochemical processes, energy consumption—directly linked to electrode dissolution—is the dominant factor influencing operational costs. Increased energy input enhances metal ion release into solution, but at the expense of higher costs [43]. Other key factors include treatment time [50] and electrode material, with aluminum electrodes often reported as more cost-effective than iron for the removal of total coliforms, *E. coli*, and COD [73].

To reduce energy-related expenses, alternative strategies have been proposed. For instance, hydrogen generated through water splitting has been explored as an energy source. In the case of tetracycline removal, complexation with nickel allowed cost reductions from 0.535 to 0.166 US$m^{-3} [71].

A comprehensive understanding of EC's potential requires not only evaluating removal efficiencies but also considering its economic and energetic implications. Table 5 presents a comparative overview of operational costs, energy demand, and removal efficiencies of EC in relation to conventional treatment technologies such as activated sludge [86], membrane bioreactors (MBR) [87], membrane-aerated biofilm reactors (MABR) [88,89], and

ozonation [90,91]. This comparison underscores both the strengths and limitations of EC in terms of cost-effectiveness, energy requirements, and pollutant removal performance.

Table 5. Efficiency of EC compared with some conventional treatment processes.

Technology	Cost (USD$m^{-3})	Energy Consumption (kWhm^{-3})	Antibiotic Removal Efficiency	Microorganism Removal Efficiency
Activated sludge	2.71	0.4	30–70%	89%
MBR	3.41	0.57	49.7 (SMX)	93.30%
MABR	NR	~0.8	~96% (TOC)	85%
Ozonation	~4.37–7.78	0.37	10–29% (TOC)	>99%
EC	0.011–4.13	0.52–17.2	80–98%	67–100%

NR–Not reported.

As shown in Table 5, EC exhibits a wide range of operational costs yet remains competitive with conventional treatment processes, particularly when considering its high efficiencies in removing antibiotics and microorganisms. While these data underscore EC's potential as a promising alternative, it is essential to note that cost comparisons are strongly influenced by the specific operational conditions and energy requirements reported, which vary considerably across studies. Moreover, the practical application of EC cannot be fully evaluated without addressing the challenges associated with sludge generation and management. Sludge production not only affects the overall operational costs but also has significant implications for environmental sustainability, highlighting the need for careful assessment of waste handling strategies prior to large-scale implementation.

3.3.11. Sludge Management

Electrocoagulation (EC) produces flocs that capture and remove contaminants primarily through adsorption. Depending on their density, these flocs can manifest as foams or as metallic hydroxide sludges in the form of $M(OH)_3$ or $M(OH)_2$. Kumari & Kumar [92] confirmed the presence of these compounds through FTIR analysis, where absorption peaks at 3623 cm^{-1} indicated O–H stretching vibrations associated with metallic hydroxides, while peaks at 845 cm^{-1} revealed the presence of oxides. XRD analysis further confirmed the formation of aluminum complexes, with characteristic peaks observed at 26.74°.

The volume of sludge generated in EC depends mainly on two factors: (i) the amount of coagulant released, which is directly linked to the applied electrical energy—greater energy input results in higher dissolution of metallic ions from the anode and, consequently, greater formation of polymeric hydroxide species; and (ii) the hydraulic retention time.

Although EC generally produces smaller sludge volumes than conventional chemical coagulation, sludge management can still account for 50–60% of operational costs due to treatment and disposal requirements. For this reason, several studies have explored sludge minimization strategies. For instance, Barshani et al. [93] proposed sludge dewatering using ferric chloride salts as coagulant agents. Applying the capillary suction time (CST) method, they reported that raw sludge exhibited a CST of 40.36 s, whereas at pH 11 with a coagulant dose of 20 mg/g sludge, the CST decreased to 16.08 s, yielding 3.32% sludge content.

Regarding microorganisms, while EC aims to inactivate them, the sludge generated can also be repurposed. Mbacké et al. [94] used sludge from industrial and domestic wastewater treatment in a two-compartment microbial fuel cell (MFC) separated by an ion-exchange membrane. They achieved power densities of 41.29 mW m^{-2} using acetate as a substrate and 27.57 mWm^{-2} with glucose.

Another study reported that EC sludge, after thermal treatment at 100 °C for dehydration, can be further utilized as catalysts, adsorbents, construction materials, or even in pollutant reduction processes [95].

Although promising strategies such as sludge dehydration, reuse in microbial fuel cells, and thermal valorization have been proposed, the management of sludge generated by electrocoagulation remains a significant challenge. Current studies are often fragmented and limited to laboratory-scale evaluations, failing to address long-term stability, scalability, or environmental trade-offs. Thus, sludge handling represents one of the least developed aspects of electrocoagulation research, requiring more systematic investigation to ensure the sustainability and competitiveness of this technology in real wastewater treatment scenarios.

3.3.12. Identification of By-Products and Toxicity Assessment

While sludge management is a key challenge in electrocoagulation (EC) systems, another critical aspect that requires attention is the potential formation of transformation by-products during the treatment process. In addition to the removal of pollutants via co-precipitation and adsorption onto electrochemically generated solids, hydroxyl radicals (–OH) produced by water reduction at the cathode can induce the degradation of contaminants into secondary compounds. These by-products may, in some cases, exhibit higher toxicity than the parent compounds. Despite its importance, the study of by-product formation and its toxicological implications in EC-treated water remains limited. Within the period considered in this review, only five publications directly addressed this issue.

Some studies employed Artemia salina as a bioindicator for toxicity assessment. For instance, M. C. Potrich et al. [73] reported that when iron electrodes were used, toxicity decreased within the first 24 h; however, at 48 h, 100% mortality was observed for both iron and aluminum electrodes under optimal electrolysis times of 30 and 10 min, respectively. Similarly, Lemna minor bioassays revealed that untreated water appeared less toxic than water subjected to EC, suggesting the transient formation of toxic intermediates. These findings align with those of F. R. Espinoza-Quiñones et al. [63], who observed that degradation products of ciprofloxacin induced high toxicity in A. salina nauplii at electrolysis times shorter than 30 min. Nonetheless, toxicity decreased as electrolysis progressed, with optimal conditions identified at 75 min, 18 A m^{-2}, and an initial pH of 9.

In addition to toxicity, EC has also been investigated for its role in mitigating microbial resistance to antibiotics. Bioassays with *Escherichia coli* and *Staphylococcus aureus* demonstrated that inhibition zones diminished as electrolysis time increased: from 21 mm at <20 min, to 15–21 mm between 20 and 40 min, and no inhibition beyond 40 min. This suggests that prolonged treatment reduces the risk of microbial adaptation and resistance. Comparable results were reported for ofloxacin and chloramphenicol treated waters, where bactericidal activity was fully eliminated after 40 and 50 min, respectively [96]. Moreover, the degradation of doxycycline products was associated with a reduction in toxicity, further supporting the potential of EC to mitigate both environmental risks and antibiotic resistance [17].

Beyond toxicity reduction, EC-treated water has shown promise for reuse when coupled with complementary processes. For example, S. Ibrahimi et al. [66] combined EC with nanofiltration and reported enhanced cucumber cultivation outcomes: 54.4% of plants exhibited increased stem height, 64.7% showed longer root growth, and overall healthier development was achieved.

In summary, although the identification of by-products and the evaluation of their toxicity remain underexplored, the available evidence indicates that EC holds significant promise. Its ability to degrade antibiotics, reduce toxicity, and lower the risk of antibiotic resistance positions EC as a competitive and environmentally sustainable treatment option for water remediation.

3.3.13. Combination of Electrocoagulation with Other Treatment Processes

Electrocoagulation has proven to be an efficient technology for removing emerging contaminants; however, its combination with other processes, such as ozonation, photolysis, adsorption, artificial wetlands, or ultrasound, allows for enhancing its performance and broadening its spectrum of action. These conjugated systems leverage physicochemical synergies to enhance the degradation of antibiotics, inactivation of microorganisms, and disruption of resistance genes, even in complex aqueous matrices. The integration of processes enables the optimization of treatment times, reduces the generation of toxic by-products, and increases energy efficiency.

Coupling EC with advanced oxidation processes (AOPs) such as ozonation, UV, photocatalysis, or Electro-Fenton significantly enhances removal efficiency (Table 6). This is mainly due to the in situ generation of reactive oxygen species (ROS), such as hydroxyl and chlorine radicals, which can degrade resistant molecules and disrupt microbial cell integrity.

For instance, the combination of EC with ozone and UV radiation achieved simultaneous removal of organic matter (87% TOC) and Escherichia coli (96%) in greywater [44]. The success of this integration lies in the synergy between the coagulation–flocculation capacity of EC and the strong oxidative power of hydroxyl radicals generated during ozonation and UV photolysis. This makes the system particularly effective for effluents with high microbial and organic loads.

Another relevant configuration involves EC coupled with the Electro-Fenton (EF) process, which exploits iron ions from EC as Fenton catalysts. In swine wastewater, this setup enabled the differential removal of ARGs depending on their location—EC inactivated intracellular ARGs (2.5–3.2 log reduction), whereas EF was more effective against extracellular ARGs (3.2–4.4 log reduction) [20]. This distinction is crucial because extracellular ARGs are often more mobile in aquatic environments, thus representing a greater risk for horizontal gene transfer.

Adsorption-based hybrid systems also show great promise, especially when incorporating green or engineered nanomaterials. The integration of EC with ultrasonic-assisted adsorption using chitosan–graphene oxide achieved 98% removal of ofloxacin in just 10 min [97]. Here, the ultrasonic waves enhance mass transfer, while EC generates metal hydroxides that improve particle destabilization and support adsorption kinetics. Such rapid treatment cycles can be especially beneficial for decentralized or emergency treatment applications.

The incorporation of constructed wetlands (CWs) as a pre- or post-treatment stage has also proven valuable. When EC was applied after a vertical flow constructed wetland (VFCW), significant additional removal of *E. coli* and total coliforms was observed (~2.35 $\log_{10}$), underscoring the value of EC as a polishing step in nature-based systems [67]. Similarly, the combination of EC with infiltration–percolation systems enabled complete elimination of fecal indicators and nutrients, while improving the removal of chemical oxygen demand (COD) and turbidity [70].

From an operational standpoint, the sequence and mode of integration matter significantly. In the system combining EC with electrooxidation, the sequential application of aluminum and stainless-steel electrodes, followed by anodic oxidation on IrO_2–Ta_2O_5/Ti, enabled the complete removal of amoxicillin in synthetic wastewater [84].

In terms of limitations, several hybrid systems rely on high current densities or specific pH ranges (e.g., 3–7), which may require additional chemical adjustments or energy input. Moreover, while many reported results are promising in controlled or synthetic matrices, the transferability to real, heterogeneous wastewater remains a challenge. Only a subset of the reviewed studies employed real wastewater samples, and even fewer analyzed the long-term stability or cost-effectiveness of the integrated systems.

Nonetheless, the growing number of studies employing real matrices—such as greywater, swine effluent, or hospital wastewater—suggests a maturing field moving toward full-scale applicability. A particularly illustrative example is the hybrid system tested, where EC, coupled with ozone and UV, was able not only to remove TOC and *E. coli* but also to significantly reduce turbidity and suspended solids, demonstrating a multi-barrier treatment approach suitable for reuse scenarios [44].

As can be seen, the strategic integration of EC with complementary processes enables the development of tailored, high-efficiency treatments that can address the complexity of antimicrobial pollution in wastewater. These systems, when properly optimized, can provide both environmental safety and regulatory compliance, particularly in contexts where conventional treatment methods fall short.

Table 6. Applications of integrated treatments with electrocoagulation in the removal of antibiotics, microorganisms and resistant genes.

Type of Integrated System	Technologies Involved	Key Features	Target Pollutant	Removal	Type of Water	References
Hybrid	EC + PHC	Enhances antibiotic removal and promotes the production of hydrogen gas.	CIP	92.5%	HWW (simulated)	[98]
Hybrid	EC + US	The EC provides iron ions that favor removal by adsorption to iron hydroxide and by Fenton reactions. Ultrasound-generated microbubbles attenuate the EC process.	AMX	>80%	PWW (synthetic)	[99]
Coupling	EC + EO	EC: Electrodes: Al, SS/titanium mesh Time: 60 min. EO: Electrodes: IrO_2-Ta_2O_5 | Ti/titanium mesh I: 10 mA Time: 120 min. Highlight: effectively removes the antibiotic and reduces entrained salts.	AMX	100%	WW (synthetic)	[84]
Coupling	EC + EF	EC is most effective in inactivating intracellular ARGs and bacteria. EF is more effective in inactivating extracellular ARGs. Coupling is more effective in removing both.	ARGs	intracellular ARGs-2.49–3.25 logs extracellular ARGs -3.23–4.38 log	Swine WW (real)	[20]
Coupling	VFCW + EC	The coupling enhances phosphorus removal and the effluent has better water quality.	Total coliforms *E. coli*	2.37 $\log_{10}$ 2.35 $\log_{10}$	MW effluent (real)	[67]

Table 6. *Cont.*

Type of Integrated System	Technologies Involved	Key Features	Target Pollutant	Removal	Type of Water	References
Coupling	EC + O_3	O_3 dose: 47.5 mgL^{-1} Electrodes: Al and Fe	TOC	85%	GW (real)	[44]
Coupling	EC + O_3 + UV	EC removes suspended solids, turbidity, and COD O_3 and UV disinfect	TOC *E. coli*	87% 96%	GW (real)	[44]
Hybrid	EC + UA	Time: 10 min. Adsorbent: chitosan + graphene oxide.	OFL	98%	WW (synthetic)	[97]
Hybrid	Ultrasonic + EC	Perforated electrodes that function as baffles minimize the need for stirrers and favor the inactivation of microorganisms.	*E. coli*	100%	PW (synthetic)	[55]
Coupling	EC + AD	Optimal conditions: pH: 7.8 j: 15.5 mA cm^{-2} $CFZ_{initial}$: 60 mg L^{-1} IED: 1.0 cm Chitosan dosage: 0.7 g L^{-1} Electrolyte support: NaCL Dose of electrolyte: of 0.07 M Time: 23 min	CFZ	100%	Hospital wastewater (synthetic)	[56]
Hybrid	EC + EF	The iron ion that promotes Fenton reactions, together with the absence of dissolved oxygen in the water, favors the inactivation of microorganisms in both the water and the sludge generated.	*E. coli*	Generated magnetite: 4.7 log cells GR: 3.2 log cells (both in the absence of DO)	WW (real)	[100]
Hybrid	EC + HEF	Enhances the removal of heavy metal-antibiotic complexes from wastewater.	CIP:Cu complexes	Cu-99.69% CIP-96.40% TOC-83.62%	WW with antibiotic and cupper (synthetic)	[47]
Coupling	UV disinfection + EC	J: 20 $mAcm^{-2}$ pHs between 3 and 7 Removal of both intracellular and extracellular ARGs.	sul1, sul2, tetO, and tetX	1.62 to 2.83 logs	Secondary clarifier effluent (real)	[20]
Coupling	IP + EC	Complete elimination of: fecal coliforms, total coliforms, and *enterococci*, turbidity, COD, $PO_4{}^{3-}$, NH_4+, and NO_3-.	Fecal coliforms Total coliforms Enterococci	100%	Wastewater (real)	[70]
Hybrid	UV + EC	pH: 7.4 j: $mAcm^{-2}$ Electrodes: Al and Fe Electrolyte dosis: 1 g Na_2SO_4/L Time: 40 min.	*E. coli*	100%	Grey water (real)	[64]

Table 6. *Cont.*

Type of Integrated System	Technologies Involved	Key Features	Target Pollutant	Removal	Type of Water	References
Hybrid	eMBR	Minimizes membrane fouling, and electrochemical processes favor the removal of recalcitrant organic contaminants.	DCF CMZ AMX	75.25 ± 8.79% 73.84 ± 9.24% 72.12 ± 10.11%	Municipal wastewater (simulated, spiked)	[101]
Simultaneous	MBBR combined with EC	Inicial suspended solids: <3000 mgL^{-1} Contact time: 24 h	AZM	92.3%	Hospital wastewater (real)	[58]
Coupling	EC + UF + chloration	Maximum permissible limits of: Turbidity: <1NTU *E. coli*: <50 $cfumL^{-1}$ pH: 6–9	*E. coli*, Norovirus *Salmonella*	*E. coli*: <50 cfu mL^{-1}	Dishwashing water (simulated)	[102]
Coupling	Ozone + EC	J: 33.2 Am^{-2} Time: 37.8 min pH: 8.4 Ozone dose: 0.7 gh^{-1}.	SMX	99.65%	Wastewater (modeling)	[103]
Hybrid	EC + Adsorption	It should be considered: initial concentration, pH, current density, retention time, and chitosan dosage.	CTX	100%	Polluted water (synthetic)	[85]
Coupling	EC + EO	EC: Removal: 97.9% Time: 20 min EO: improved by 2.09% Time: 30 min.	*E. coli*	99.9%	Domestic wastewater (synthetic)	[48]
Coupling	EC + EF	The adsorption of microorganisms on iron hydroxide, the disinfectant effects of electrogenerated chlorinated products, and the Fenton reaction with H_2O_2 favor the inactivation and disinfection of microorganisms.	*E. coli*, *enterococci*, *Clostridium perfringens* spores, somatic coliphages and eukaryotes (amoebae, flagellates, ciliates and metazoa)	Primary effluent: Heterotrophic bacteria: 3.5 log/mL *E. coli*: 0.5 log/mL *Enterococcus*: 1.5 log/mL *Clostridium perfringes* spores: 2.3 log/mL Coliphages and eukaryotes: 0 log/mL Secondary effluent: Heterotrophic bacteria: 3.5 log/mL *E. coli*: 0.2 log/mL *Enterococcus*: 1.7 log/mL *Clostridium perfringes* spores: 1.7 log/mL Coliphages and eukaryotes: 0 log/ml	Primary and secondary treatment effluent (real)	[104]

4. Conclusions and Future Perspectives

EC has emerged as a promising alternative for removing antibiotics and ARB in water treatment, due to its ability to combine coagulation, redox reactions, and localized destabilization phenomena in a single process. The available evidence indicates that EC can effectively reduce both molecular and biological contaminants by promoting interactions that favor aggregation, precipitation, and partial degradation, all without the need for external chemical reagents.

However, despite its potential, EC still faces critical challenges that must be addressed for its broader implementation. The efficiency of the process depends strongly on operational parameters such as current density, electrode material, interelectrode distance, and water matrix composition. In real wastewater, the presence of natural organic matter, surfactants, or chelating agents can interfere with floc formation, thereby reducing removal performance. Moreover, the management of the sludge generated—often enriched with retained antibiotics and resistance genes—poses additional environmental and regulatory considerations that cannot be overlooked.

Hybrid approaches that combine EC with advanced oxidation processes, filtration systems, or constructed wetlands offer a valuable opportunity to enhance performance and reduce energy consumption. Likewise, future studies should focus not only on removal efficiency but also on transformation products, life cycle impacts, and long-term stability under operational conditions. In contexts with limited infrastructure or variable water quality, the adaptability and low reagent demand of EC may represent a strategic advantage for decentralized applications.

Ultimately, electrocoagulation should be understood as part of an integrated portfolio of technologies aimed at containing the environmental dissemination of antimicrobial resistance. Its compatibility and potential for coupling with renewable energy sources reinforce its relevance in current efforts to improve water quality, particularly at critical interfaces between human health, agriculture, and the environment.

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Abbreviations

The following abbreviations are used in this manuscript:

AD	Adsorption
Al	Aluminum
AMP	Ampicilline
AMX	Amoxicillin
AOP	Advanced oxidation process
APC-EC	Alternating pulse current EC

ARB	antibiotic-resistant bacteria
ARG	antibiotic-resistance genes
AZM	Azithromycin
BOD	Biological Oxygen Demand
CMZ	Carbamazepin
CAP	Chloramphenicol
CFX	Cephalexin
CFZ	Cefazolin
CIP	Ciprofloxacin
COD	Chemical oxygen demand
COT	Carbon organic total
CQ	Chloroquine
CS	Commercial Steel
CTX	Ceftriaxone
CWs	Constructed wetlands
DCF	Diclofenac
DC-EC	Direct current electrocoagulation
DCWW	Drug-containing wastewater
DDBAC	benzyldimethyldodecylammonium chloride
DO	Dissolved oxygen
DOX	Doxycycline
DWW	Domestic Wastewaters
EF	Electro-Fenton
eMBR	Electro membrane bioreactor
EO	Electro oxidation
e.m.f.	Electromotive force (e.m.f.)
Fe	Iron
GR	Green Rust
GW	Greywater
HEF	Hetero Electro Fenton
HWW	Hospital wastewaters
IP	Infiltration-percolation
LVX	Levofloxacin
MABR	Moving-aerated biofilm reactor
MBBR	Moving bed biofilm reactor
MZN	Metronidazole
MW	Municipal water
MWW	Municipal wastewater
OFL	Ofloxacin
PHC	Photocatalysis
PCP-EC	Positive single pulse current Electrocoagulation
PW	Polluted waters
PWW	Pharmaceutical wastewater
RhB	Rhodamine B
RSM	Response Statistical Method
ROS	Reactive oxygen species
RW	Rainwater
SS	Stainless Steel
SMX	Sulfamethoxazole
STZ	Sulfathiazole
SW	Storm water
TC	Tetraclycline
TC:Cu	Tetracycline-Cupper complex
TC-Ni	tetracycline -nickel complexes

TOC	Total Organic Carbon
TMP	Trimethoprim
TW	Treated water
TWW	Treated wastewater
UA	Ultrasonic adsorption
UF	Ultrafiltration
US	Ultrasound
VFCW	Vertical Flow Constructed wetland
WW	Wastewater

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