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Review

Electrocoagulation as an effective all-in-one wastewater treatment: a critical review

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Abstract

Electrocoagulation has emerged as a promising alternative for the treatment of wastewater from a wide range of sources. Interest in this technology has grown due to its several advantages, including a small footprint, cost-effectiveness, and environmental sustainability. In contrast, conventional wastewater treatment methods that rely primarily on biological processes often exhibit limited removal of emerging contaminants, highlighting the need for more efficient and versatile treatment alternatives. This review article provides an in-depth discussion of the application of electrocoagulation technology for wastewater treatment, with a specific focus on its performance in real wastewater matrices across industrial, municipal, and agricultural case studies, including evidence from case studies demonstrating its use on a large scale. It examines key factors that influence process performance and presents the governing equations that determine the technology's efficiency and applicability. A key contribution of this review is the integrated analysis of reactor design, operational parameters, and emerging hybrid electrocoagulation (EC) systems, such as EC coupled with advanced oxidation processes (EC-AOP) and membrane processes (EC-membrane), highlighting recent technological advancements that improve treatment efficiency while reducing energy consumption. The review also addresses techno-economic considerations and synthesizes findings from life-cycle assessment studies, revealing critical trade-offs between environmental benefits, such as reduced eutrophication potential, and limitations including increased ecotoxicity. Finally, the article identifies key research gaps and highlights *the potential of replacing fossil fuel-based energy inputs with renewable energy sources as well as the importance of electrode optimization and sludge valorisation strategies to improve the overall sustainability and scalability of electrocoagulation systems.*

Keywords

Wastewater pollutants; electrochemical removal; reactor design; life-cycle assessment, sludge valorisation, case studies

Introduction

The rapid growth of population and unsustainable disposal practices have put a significant strain on the most important resource on Earth *i.e*. safe drinking water. According to a recent report by the WHO and UNICEF, 2.1 billion people worldwide still lack access to safely managed drinking water, with the problem more pronounced in developing countries [1]. The discharge of inadequately treated industrial wastewater (produced in billions of liters per day) into the environment significantly exacerbates water contamination [2,3]. Furthermore, wastewater from domestic, municipal and industrial sources is discharged at a rate that is unprecedented, with an increased contamination load that is detrimental to the environment [4-6].

Many wastewater treatment plants (WWTPs) rely on biological treatment methods, including the activated sludge process, for treating wastewater prior to its discharge to the receiving environment [7,8]. Biological processes are both cost-effective and performance-driven and can produce less sludge than other traditional treatment methods, such as chemical precipitation [9-11]. However, improvements in science have led to the proliferation of emerging contaminants in water, and human exposure to these contaminants has resulted in several toxicological impacts predominantly related to hormone-like, mutagenic and carcinogenic effects [12-14]. Emerging contaminants were not considered in the design of conventional treatment systems and are therefore often not removed during treatment. Other limitations of biological treatment methods include significant greenhouse gas emissions, the problematic characterization of oxygen quantity needed for organic matter degradation and the release of nitrogen or phosphorus above regulatory limits leading to eutrophication in the environment [15,16]. Therefore, it is imperative to develop a new generation of technologies that target and manage the diverse types of pollutants present in effluents from multiple wastewater streams. Recent application of advanced treatment technologies such as “*in situ* coagulation" at the demonstration level has indicated their potential for removing various contaminants, including several metals, which are suspected of being endocrine disruptors at minute concentrations.

Electrocoagulation (EC), a chemical treatment method, is an alternative for removing organic and/or inorganic pollutants from wastewater streams due to its simple yet efficient operation. It is based on the *in situ* generation of coagulants by the electrolytic dissolution of anodes made from specific metals such as iron and aluminium, leading to the formation of a precipitating cation hydroxide capable of removing both soluble and colloidal impurities from the aqueous media [17]. Hence, EC avoids the direct dosing of chemicals like aluminium sulphate ($Al_2(SO_4)_3$), ferric chloride ($FeCl_3$) and ferric sulphate ($Fe_2(SO_4)_3$), leading to less quantities of sludge being produced. Furthermore, when compared with chemical coagulation, EC was found to be more cost-effective [18]. Interest in electrocoagulation for wastewater treatment is growing, and the current decade has seen an increase in publications, which is reflected in search engines such as Google Scholar and the SCOPUS database.

Thus, this review was conducted to document findings from studies that used EC to treat real wastewaters and to evaluate the overall suitability of EC as a green strategy for wastewater detoxification and purification. A key specificity of this review is that it highlights findings from case studies that investigated the removal of pollutants such as heavy metals, organic pollutants and pathogens from real wastewater matrices. Furthermore, this work identifies suitable operating conditions for applying EC technology in wastewater treatment and provides a detailed techno-economic and environmental evaluation, including life-cycle assessment (LCA), in dedicated sections. Overall, this article provides an updated and comprehensive review of electrocoagulation technology as a suitable method for the treatment of wastewater from various sources.

Electrocoagulation technology

Electrocoagulation (EC) is an electrochemical water treatment process that applies an electrical current for in situ generation of coagulants. These coagulants decontaminate wastewaters by coagulation and adsorption. EC features include ease of operation, low sludge production, and high efficiency in short times compared with conventional coagulation. EC has been successfully applied to the treatment of wastewater from a wide range of sources, including hospital, tannery, domestic, and industrial effluents. It has proven effective in removing bacteria, oil emulsions, heavy metals, and a broad spectrum of organic and inorganic contaminants, including recalcitrant chemicals, persistent pigments, silica, and colloidal particles [19-24]. In addition, EC is an efficient, time-saving, economical, and eco-friendly water reclamation method with almost zero chemical addition [21,22,25]. EC utilizes metal electrodes, where anodes are positively charged poles that can electrochemically dissolve into ions in solution, while water molecules are reduced to hydroxide ions (OH-) near the cathode, the negative pole. Further, in solution, anodically leached ions react with OH- to form metal hydroxides, which act as coagulants and flocculants (Figure 1).

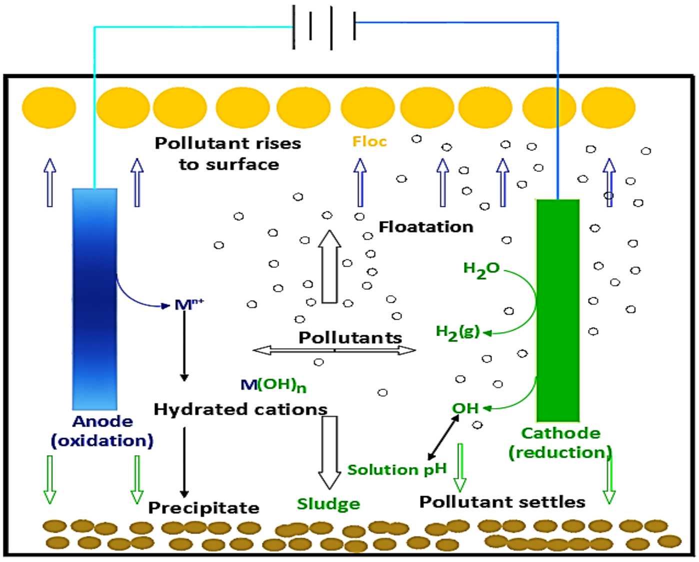

Figure 1. *Mechanism of electrocoagulation method, adapted and redrawn by the authors based on [26]*

Moreover, coagulant molecules possess a large surface area that promotes contaminant adsorption and destabilizes impurities through charge neutralization (Figure 2), reducing repulsive forces between suspended particles and facilitating their agglomeration into larger flocs. This leads to impurity destabilization and precipitation [19,27,28].

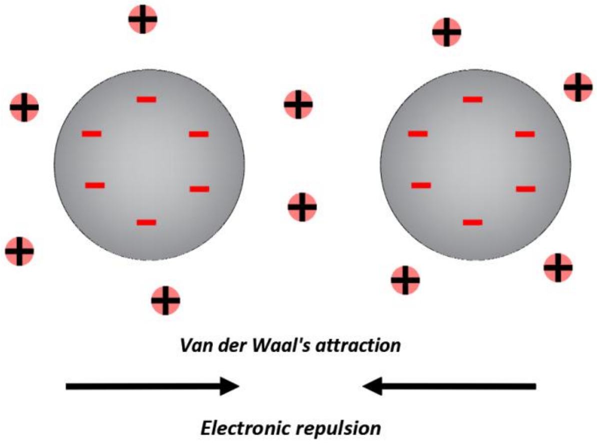

Figure 2*. Interaction between two colloid particles, adapted and redrawn by authors based on [26]*

EC coagulant metal ions were also reported to act as ligands, forming coordination complexes with water impurities [29,30]. Besides, gas bubbles formed by cathodic reduction act as a floating agent, carrying contaminant flocs to the surface and facilitating their separation [28,31]. The combination of several processes simultaneously taking place during EC, such as coagulation, flocculation, adsorption, chelation, and flotation, adds to the process importance, feasibility, applicability, and treatment efficiency [19,21,22]. In addition, compared with other wastewater treatment technologies, EC is particularly effective for comprehensive water decontamination, achieving high removal efficiencies for phosphorus [32], COD [33], dyes such as methylene blue, reactive black 5, and acid blue 29 [34], turbidity [35] and toxic heavy metals including chromium(VI) [36] with removal efficiencies reaching 100 %.

Hybrid and advanced approaches in electrocoagulation-based wastewater treatment

Recent advancements in electrode technology aim to address persistent problems, such as electrode passivation and energy extraction inefficiency. To address these challenges, research has focused on optimizing electrode design. These advancements include innovations in shape (plate, cylindrical, spiral, disc, mesh, ball), geometry (3D, perforated, helical), configuration (monopolar, bipolar, rotating, tubular), and material combinations (Al-Fe, Cu-Fe, Zn-Fe). The purpose of these upgrades is to maximize coagulant synthesis, reduce energy use, minimize sludge production, and improve treatment effectiveness across a wide range of wastewater types. Innovative topologies have demonstrated enhanced mass transfer, excellent current distribution, and effective mitigation of passivation through techniques such as electrode rotation, segmentation, or surface perforation [37,38]. A number of studies have demonstrated that a three-dimensional electrochemical reactor equipped with particle electrodes can greatly improve pollutant removal and reduce electrical energy consumption compared to conventional two-dimensional systems. The increased removal efficiency can be attributed to the larger response surface and improved mass transfer in the particle electrode construction. Enhanced efficiency is achieved using particle electrodes, which function as micro-electrolysis cells [39], combined with reduced electrical energy consumption. The effectiveness of COD removal in three-dimensional systems is much higher than that of two-dimensional systems [40-42]. Granular activated carbon, carbon aerogels, modified kaolin, metallic particles, and α-Fe_2O_4/ powdered activated carbon(PAC) are the types of particle electrodes that are typically analysed [40,43,44].

Furthermore, despite the success of electrocoagulation in the treatment of wastewater, its limitations when dealing with wastewater containing persistent dissolved organic pollutants has been noted [45]. As such, recent research has focused on hybrid electrocoagulation systems designed to overcome the inherent limitations of freestanding electrocoagulation. These systems incorporate advanced oxidation processes (AOPs) and nanotechnology. It has been established that hybrid EC-AOP systems, particularly those combining electrocoagulation with UV irradiation or ozonation, are exceptionally effective for treatment, achieving high COD removal efficiencies [46,47]. These systems use electro-generated flocs, which serve as both catalytic and adsorptive sites, thereby improving the effectiveness of oxidation and the mineralization of any pollutants present. Combining nanostructured membrane technology with electrocoagulation has significantly improved the process's sustainability. Compared with conventional membrane filtration systems, studies have shown that PVDF nanofiber membranes for filtering EC-generated flocs can reduce membrane fouling rates by up to 40 % [48,49]. Electrocoagulation-membrane hybrid reactors provide an efficient way to produce high-quality effluent while simultaneously reducing operational problems. Following the implementation of reverse osmosis, nanofiltration membrane bioreactors have demonstrated superior permeate quality, lower fouling rates, and comparable energy consumption at higher recovery rates than ultrafiltration membrane bioreactors combined with RO systems used for post-treatment of EC effluent [50]. These advances underscore the crucial importance of hybridization, nanotechnology, and electrode design in the development of electrocoagulation for effective, large-scale wastewater treatment.

Influencing factors

To ensure EC implementation achieves the highest efficiency, several factors influencing it need to be considered. For example, electrical conductivity and initial pH of the feed solution, selection of electrode material, types, geometry, spacing, and combinations, mixing speed, the process time and most importantly, the current density (j). First, the solution conductivity (σ) enables a medium for current passage through ions and electron movements in solution. Thus, low conductivity leads to poor current conduction in solution and fewer electrochemical reactions, and consequently, low process efficiency. Therefore, an electrolyte is usually added to the solution to enhance σ when the treatment solution has low ionic content. Examples of electrolytes are NaCl, Na_2SO_4, $NaNO_3$, NH_4Cl, KCl, and K_2SO_4 [51]. The choice of electrolyte type should not interfere with the contaminant removal. In addition, electrolyte concentration should be optimized to avoid excessive anodic dissolution [51]. This is because electrolytes, including chlorides, for example, can cause both anodic and cathodic corrosion, leading to super faradaic efficiency (SFE, %) [52,53] where the actual dissolution of electrodes surpasses the calculated metal dissolution from Faraday's law, Equation (1).

$$C_{\mathrm{th}} = \frac{\Delta m_{\mathrm{th}}}{V} = \frac{ItM_{\mathrm{w}}1000}{zFV} \tag{1}$$

$$C_{\mathrm{actual}} = \frac{\Delta m_{\mathrm{actual}}}{V} \tag{2}$$

$$\Delta m_{\mathrm{actual}} = m_{\mathrm{i}} - m_{\mathrm{f}} \tag{3}$$

where, C_{th} / mg L^{-1} is the theoretical concentration of leached ion in solution, Δm_{th} / mg is the electrode mass loss theoretically calculated, I / A is the current, t / s is the process time, M_{w} / g mol^{-1} is the molar mass of anodic metal, z is the valency of anodically leached ion, F is Faraday's constant (96485 C mol^{-1}), V / L is the effective reactor volume, $\Delta m_{\mathrm{actual}}$ / mg is the experimentally determined electrode mass loss, and m_i-m_f is the difference between the initial electrode mass before and after the EC, respectively.

Faradaic efficiency (FE, %) is given by Equation (4). If it exceeds 100 % due to specific ion pitting corrosion [54], it is then called SFE, %.

$$FE = \frac{C_{\text{actual}}}{C_{\text{th}}}100 = \frac{\Delta m_{\text{actual}}}{\Delta m_{\text{th}}}100 \tag{4}$$

Optimizing the feed water's initial pH (pH_i) can dramatically affect EC treatment efficiency by directing the electrochemical reaction toward specific mechanisms [54-56]. The initial pH also plays a role in reducing the EC electrical energy consumption (EEC), where using certain acidic pH_i can facilitate the anodic dissolution, leading to reduced EEC. Additionally, pH_i influences the formation of specific coagulant species of reactivity changing with the alteration of solution pH [54,57]. For example, at acidic pH_i, M^{n+} form in solution due to anodic oxidation, whereas at around neutral pH, coagulant molecules $M(OH)_3$ form, *i.e.* $Al(OH)_3$ and $Fe(OH)_3$. In aluminium electrocoagulation (Al-EC), Al metal protects its reactive surface by forming a passive oxide layer Al_2O_3, that reduces the metal dissolution [57]. In alkaline solutions (pH > 9), the dominant species change to $M(OH)_4$, *i.e.* $Al(OH)_4$ and $Fe(OH)_4$, which do not possess any coagulating activity [54]. In addition, pH_i governs the formation of polymeric aluminium oxyhydroxides, where at pH > 3, $Al(OH)^{2+}$ forms and turns into $Al(OH)_2^+$ with the increase in pH to 4, and around pH 5, Al_{13} polymeric species dominate. As pH increases > 5.8, hydrolysis of $[Al_{13}O_4(OH)_{24}(H_2O)_{12}]^{7+}$ into $Al(OH)_3$ takes place and the latter becomes the dominant species at pH 7 [54,58]. Table 1 summarizes the ionic species formed during EC at different pH.

Table 1. *Change of electrochemical reactions and dominant chemical species at different pH values*

Al-EC	pH range	Fe-EC
$Al + 6H_2O \text{ ® } [Al(H_2O)_6]^{3+} + 3e^-$ $Al^{3+} + 2H_2O \text{ ® } Al(OH)^{2+} + H_3O^+$	3	$2Fe_{(s)} + 4H_2O \rightarrow 2Fe(OH)_2 + 4H^+$ (anode) $Fe_{(s)} + 2H_2O \rightarrow Fe(OH)_2 + H_{2(g)}$ (cathode)
$Al(OH)^{2+} + H_2O \text{ ® } Al(OH)_2^+ + H_3O^+$ $Al^{3+} + H_2O \text{ ® } Al(OH)^{2+} + H^+$ $13 Al^{3+} + 40 H_2O \text{ ® } [Al_{13}O_4(OH)_{24}(H_2O)_{12}]^{7+} + 32 H^+$	5	$Fe^{2+} \rightarrow Fe^{3+} + e^-$ $Fe^{3+} + H_2O \rightarrow Fe(OH)^{2+} + H^+$ $Fe(OH)^{2+} + H_2O \rightarrow \alpha\text{-}FeOOH + 2H^+$ $2Fe(OH)^{2+} + H_2O \rightarrow \alpha\text{-}Fe_2O_3 + 4H^+$ $Fe^{3+} + 3H_2O \rightarrow Fe(OH)_3 + 3H^+$
$[Al_{13}O_4(OH)_{24}(H_2O)_{12}]^{7+} + 7OH^- + 4H_2O \text{ ® }$ $\text{® } 13Al(OH)_3 + 12H_2O$	6-7	$2Fe(OH)_2^+ \rightarrow \alpha\text{-}Fe_2O_3 + H_2O + 2H^+$
$Al(OH)^{2+} + 2H_2O \text{ « } Al(OH)_3 + 2H^+$	7	$Fe(OH)_2^+ + H_2O \rightarrow Fe(OH)_3^- + H^+$ (cathode) $Fe(OH)_3^- \rightarrow Fe(OH)_3 + e^-$ (anode)
$Al_2O_3 + 2OH^- + 3H_2O \text{ « } Al(OH)^-_4$	9	$2Fe(OH)_3 \rightarrow$ α- and γ-$Fe_2O_3 + 3H_2O$ $2Fe(OH)_2^+ + H_2O \leftrightarrow \alpha\text{-}Fe_2O_3 + 4H^+$
$Al(OH)_3 + OH^- \text{ ® } Al(OH)^-_4$	11	$2Fe(OH)_3 + Fe(OH)_2 \rightarrow Fe_3O_4 + 4H_2O$ $Fe_2O_3 + 2OH^- + 3H_2O \leftrightarrow 2Fe(OH)^-_4$ $2Fe(OH)^-_4 \rightarrow 2(\alpha\text{-}FeOOH) + 2OH^- + 2H_2O$
[54,58]		[55,56,59]

As shown in Table 1, pH_i plays a critical role in governing chemical reactions, coagulant formation, and the speciation of chemical species during the electrocoagulation process [54-56,58,59]. Other key influencing factors include the applied electrical parameters, such as current density (j / A cm^{-2}), which should be calculated using the effective electrode surface area. Instead of usually used electrode face area, the effective electrode area consists of its front and back, and two lateral and bottom surfaces.

Literature reported the use of many EC electrode materials [60,61], geometries [62-64], and combinations [65] in wastewater treatment. The purpose of these investigations was to achieve higher anodic dissolution and process efficiency with minimum energy consumption [64]. Among many electrode materials, aluminium and iron are the widely used types due to their abundance, cost-effectiveness, and efficiency [60,61,65]. Comparative studies reported the high efficiency of Fe electrodes in removing COD, dyes such as methylene blue, heavy metals, and pollutants in low concentrations compared to Al electrodes [66,67]. While Al electrodes were found to be more suitable for phosphate and turbidity removal [68,69].

Electrode spacing is another important factor to consider for achieving better EC performance. Larger anode-cathode gaps slow ion movement and the passage of electric current, and affect electrochemical reactions and pollutant removal [70,71]. Finally, determining the optimal operational time is equally important, as it yields the highest separation with lower electricity usage and process costs [72,73].

Kinetics

Designing an EC reactor requires an understanding of the type and content of the aqueous solution, which affects adsorption kinetics [74]. Understanding the adsorption mechanism can help to clarify the underlying mechanism that regulates the adsorption rate. In the work of *Dubsok et al.* [74], the highest coefficient of determination was provided by a pseudo-second-order (PSO) model ($R^2 = 0.99$) and the EC dynamics were described by a comprehensive model. As demonstrated by Equation (5), it was derived using material balances on the reactant species for a differential volume element of the liquid phase in the batch reactor. Their relevant supplementary information presented a complete set of ordinary differential equations, indicating that the system involves simultaneous chemical reactions.

$$\frac{\mathrm{d}C_i}{\mathrm{d}t} = \sum_{\mathrm{j}=1}^{\mathrm{n}} r_{i\mathrm{j}} \tag{5}$$

where C_i is the concentration of the i^{th} species, t is the reaction time, and r_{ij} is the individual molar production rate of the i^{th} species due to the j^{th} reaction, as highlighted by Dubsok *et al.* [74].

Hydrotechnical aspects and reactor design

Hydrotechnical aspects are key factors in the design and engineering of water treatment systems. This means the treatment design should consider controlling operational conditions such as pH, temperature (T), electrical conductivity, flow dynamics, and process time to optimize the treatment process in order to achieve the desired water quality and decontamination efficiency [75].

The hydrodynamic and electrochemical interactions that occur at solid/liquid and liquid/liquid interfaces during the electrolysis process are affected by stirring speed and mass transfer of ions. Therefore, the reactor design can enhance process hydrodynamics, leading to improved efficiency [76].

This section discusses different reactor designs for EC to optimize contaminant removal efficiency and effectiveness. The design and placement of electrodes are also reviewed, considering their critical role in optimizing operational performance within treatment systems. EC reactors are categorized according to their mode of operation, geometrical design, electrode connection and configuration [77-82].

First, depending on the mode of operation, wastewater (WW) is treated in batches [83] or in continuous-flow reactors (CFRs). In batch reactors, a fixed volume of WW is retained in a confined reactor for a specified period until a specified level of contaminant removal is achieved [83-85]. While in the CFRs, the WW is allowed to circulate in the reactor until it is sufficiently treated. CFRs have the advantage of treating larger WW volumes, but they take longer than batch reactors [86,87].

Second, electrode connections and configurations influence process efficiency and power consumption. The geometry of the electrodes and the electrolyte path should be taken into account, as they affect hydraulic resistance and, consequently, energy requirements and the cost of obtaining high-quality wastewater [88]. Therefore, EC studies used and compared different reactors, including various electrode configurations, such as plate *vs*. mesh electrodes and static *vs.* rotating disc (RDE)/cylindrical (RCE) electrodes. Moreover, mesh electrodes surpass plate electrodes in the surface area, reaction kinetics, and mass transfer [89] while RDE and RCE overperform the traditional static electrodes in the decontamination efficiency, enhanced mass transfer and less electrode passivation [90,91]. Also, the number of electrodes and connections impacts the EC efficiency and electrical requirements. For example, researchers found that the more electrodes are used, the higher the decontamination percentages obtained and the higher the energy required [92]. In addition, electrodes can be connected in monopolar parallel [93,94], where the power source is directly connected to each electrode, and in monopolar series, where the power source is connected to the first and last electrodes, with the inner electrodes connected to each other. Electrode connections can also be made in a bipolar parallel mode, in which the power source is directly connected to the first and last electrodes, while all inner electrodes are sacrificial [73,79]. Figure 3 illustrates the different electrode connections, adapted from [19].

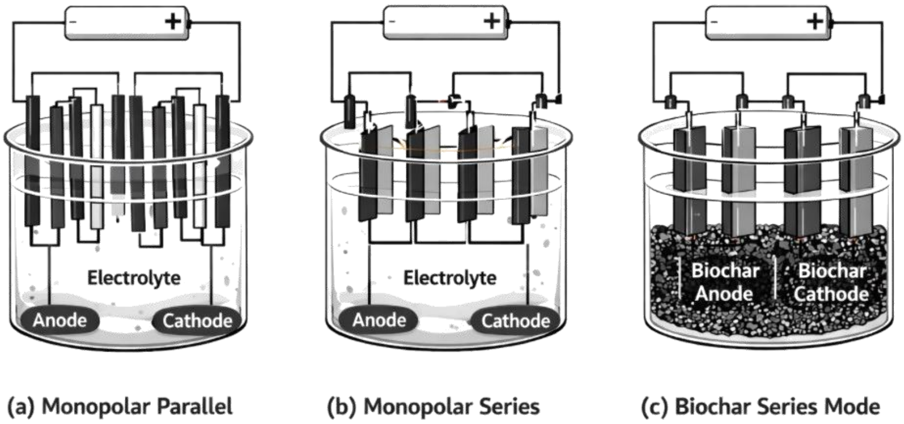

Figure 3. *Electrode connections within the EC*

The third categorization of EC reactors is by reactor geometry. For instance, cylindrical reactors are used in a tube-in-tube mode that provides a uniform electric field [95]. Also, EC reactors can be cylindrical baffled reactors, in which WW is forced to follow a zigzag path due to baffles, physical barriers [82]. This can improve mixing and an increased residence time, leading to better contact between coagulant flocs and coagulants, boosting efficiency for WW treatment like *E. coli* removal, dye decolourization, and heavy metal removal, often outperforming conventional designs by reducing energy costs and preventing issues like catalyst settling [81,82]. Another EC reactor configuration is the fluidized bed reactor (FBR), which uses small conductive metal beads instead of actual electrode plates. This prevents electrode passivation and improves the reaction kinetics, pollutant-metal contact and mass transfer [80].

Case studies and applications of electrocoagulation for wastewater treatment

The number of EC publications in the scientific literature has increased significantly over the past 25 years [38,96-100]. From the standpoint of preserving the environment and promoting community well-being, there is an increasing need for the creation of fresh, creative, and effective techniques for wastewater management and treatment [38,101]. In fact, EC is a highly advantageous water technology with wide-ranging applications and advantages that make it stand out as an efficient water recycling process or as a performance enhancement when integrated with other technologies. It is characterized by its low cost [102], ease of operation [103], hydrogen production [38,104], time savings [105], reduced chemical addition [105], less sludge produced [102], and broad applicability for treating a wide range of wastewater contaminants. It has been utilized in many treatment fields, such as surface water, stormwater, groundwater and WW. EC is utilized to treat WW such as hospital WW, municipal wastewater MWW, food processing effluents, industrial WW such as effluents of paper processing and tannery industry [84,104-107], oil and gas refinery effluents [26,95], domestic effluents such as greywater from residential or worship houses [65,72,83,108].

EC is widely used in various wastewater recycling applications. Literature reports many applications of EC in industrial, oil/gas refinery, municipal, domestic, hospital, and agricultural wastewater [19,23,100,104,109-111].

Hassoune *et al.* [98] studied the treatment of hospital wastewater (HWW) effluents using 60 min of EC at 334 A m-2, neutral pH, and 100 rpm. They were able to remove 96 % of turbidity, 94 % of COD, 82 % of total coliforms, 54 % of PO_4^-, 50 % of NO_3^-, 31 % of BOD, 22 % of NH_4^+ with 1 to 5 % error. Also, their EC-treated water had all microbial and physicochemical parameters within the WHO and Moroccan permissible limits. After that, they tested the treated water for radish germination and got an impressive increase in the plant mass, shoot and root growth.

HWW was also treated with EC by Veli *et al.* [112] in a batch reactor using Al, Fe, and stainless steel (SS) electrodes. They examined the effect of j between 2.03 and 4.87 mA cm^{-2}. They reached TOC reduction of 99.11 % using Fe electrodes, j 4.87 mA cm^{-2} and pH 7.56. In addition, 99.91 % TOC reduction was achieved with Al electrodes, j 4.46 mA cm^{-2} and pH 5.45. With SS electrodes, TOC removal was 99.89% at j = 2.68 mA cm^{-2} and pH 7.80. They used ANOVA analysis, and the results were significant with R^2 = 99.87, 96.02 and 94.15 % for Al, Fe and SS electrodes, respectively.

Moreover, Al-Shati *et al.* [113] investigated the treatment of HWW with 30 to 90 min of EC using different combinations of Al-Fe electrodes, j between 5 and 25 mA cm^{-2}, NaCl electrolyte concentration up to 4 g L^{-1}, and pH values (4 to 10). Consequently, 97.9 % COD was removed using Fe-Al electrodes with j = 24.7 mA cm^{-2}, 3.2 g L^{-1} NaCl, pH 7.4 at 81.7 min. While Al-Al electrodes removed 91.3 % of COD at 3.8 g L^{-1} NaCl, pH 7.7, j = 23.5 mA cm^{-2} at 86.3 min. Whereas the Fe-Fe electrodes usage resulted in 89.5 % of COD removal at j = 24.6 mA cm^{-2}, 2.3 g L^{-1} NaCl, pH 8.5 at 86.9 min of EC.

Researchers used EC to decontaminate domestic grey wastewater (GW) [65,99,114]. For instance, many researchers studied EC treatment for GW reuse in irrigation [115,116]. Other scientists implemented Al-EC for GW treatment, investigating various factors to predict optimal conditions for EC performance. For example, Patel *et al.* [117] studied the influence of initial pH (pH_i) 3-11, j =1-5 A m^{-2}, and EC time up to 90 min. They removed 87.5 % of BOD, 84.7 % of PO_4^-, 82.7 % of NO_3^- and 70 % of COD from GW at the best conditions they reached at pH 7, 60 min, and j = 3 A m^{-2} with electrical consumption of 0.153 kWh m^{-3} and process cost of 0.114 \$ m^{-3}.

Bani-Melhem and Smith [118] comparatively investigated the efficiency of combined EC--submerged membrane bioreactor (EC-SMBR) and SMBR alone for GW treatment. Their results indicated 13 % less membrane fouling and enhanced removal of COD, phosphate, colour, and turbidity from GW using EC-SMBR compared to SMBR alone. In addition, [119] studied three factors' optimization for Al-EC of GW; j and pH. Their response surface methodology (RSM) pointed at j as the most significant influencing parameter. Also, they emphasized the important role of the initial pH in EC efficiency. Furthermore, their sludge characterization with XRD, FTIR and SEM confirmed the amorphous structure of $Al(OH)_3$ coagulant, which is featured by its distinctive adsorption and flocculating characteristics. Therefore, they recommended reusing the EC for non-potable water conservation, such as irrigation. Also, Karichappan *et al.* [120] studied greywater decontamination using different functional parameters during EC with SS electrodes, including j between 10 and

30 mA cm-2, pH between 4 and 8, electrode spacing of 4 to 6 cm, and operation duration of 5 to 25 min. Their results showed significant EC efficiency, as confirmed by ANOVA. Their optimized conditions were pH 7, *j* = 20 mA cm^{-2}, 5 cm electrode spacing at 20 min of EC time. Under these conditions, 98.45 % removal of total solids (TS), 94.75 % removal of COD and 96.34 % of faecal contamination (FC) were achieved.

Moreover, EC is used to treat municipal wastewater (MWW). For instance, Al-Othman *et al.* [111] applied EC for GW reclamation and reuse in irrigation. They tested the effect of different electrical potential (22-30 V), pH_i 3, 5, 7, 7.5 and 9, and feed flow rate (FR) of 1, 3, and 5 L h^{-1}. Their optimal conditions were declared as 26 V, pH 7 to 7.4, and FR = 1 L/h, which resulted in removal of 99.9 % of coliform count, 98.3 % total suspended solids, 81 % of COD, 37.4 % of TDS, 30.9 % of NH_3-N, and 27.6 % of Cl^-. Also, Nguyen *et al.* [110] evaluated the use of a pilot EC system to remove phosphate from MWW under various operating conditions. They elucidated that the increase of NaCl electrolyte concentration led to higher energy consumption and phosphorous removal efficiency. Also, they were able to reach 0.15 mg L^{-1} through 95.5% P removal, which complies with the permissible limits for treated discharge water.

Another industrial application of the EC is the purification of effluents from paper and pulp processing. To optimize this EC utilization, Bassyouni *et al.* [121] studied the impact of factors like pH_i, electrolyte concentration (NaCl), number of electrodes, and applied *j* on the treatment of paper processing effluents, especially the consequential black-coloured liquor. They achieved 80 % colour removal efficiency with the use of 6 electrodes at 80 A m^{-2}, pH_i 6.5, and 2.5 g L^{-1} NaCl at 120 min of operation. Also, Izadi *et al.* [122] investigated the influence of the application of different pH_i in the range 3.5 to 11, voltages (4 to 13 V) and EC time from 10 to 60 min on the EC decontamination of industrial paper WW. Their process efficiency reached 98.5 % of colour removal, 85.3 % NH_3-N removal, 83.4 % TSS removal and 79.5 % COD removal with 60 min of EC at 10 V and pH_i 7.

Another key application of EC is the treatment of tannery toxic effluents. For example, Deghles and Kurt [104] studied the treatment of tannery WW with continuous EC using Al and Fe electrodes, different *j*, feed FR, and pH_i and the resultant process efficiency and hydrogen generation. They could achieve total Cr removal, 94 % colour removal, 73 % COD removal and 51 % NH_3-N removal with the Al-electrode and operation for 125 min at *j* = 14 mA cm^{-2} and pH 6. When they used Fe electrodes at 125 min of EC at pH 7 and *j* = 14 mA cm^{-2}, they achieved total Cr removal, 93 % colour removal, 67 % COD removal, and 46 % NH_3-N removal. Energy demand for H_2 generation, relative to the process total energy demand, was reported as 16 and 15 % for Al and Fe electrodes, respectively. The operational cost for the two Al-EC and Fe-EC WW treatments was found to be $ 0.675 m^{-3}. Other researchers studied the optimization of the process performance for tannery WW treatment using Cu-EC [123]. They studied the effect of 1 h of EC using 3 batch reactor volumes (0.5, 1 and 1.5 L) with 3 mixing speeds (60,780, and 1500 rpm), and *j* (4, 8, 12 and 16 mA cm^{-2}). Their best conditions were reactor volume of 1 L, *j* = 4 mA cm^{-2}, and the slowest mixing speed of 60 rpm, which led to removing 99 % of Cr, 96.5 % Cl^- and 92.3 % COD at 1 h of EC. Their estimated EEC was 1.659 kWh m^{-3}, electrode consumption was 0.341 kg m^{-3} and process cost was 2.622 $ m^{-3}.

On the other hand, EC can be used to treat effluents from restaurants and food processing facilities to reduce the high organic load and oily WW produced, and to prevent drain clogging [124]. Khanitchaidecha *et al.* [124] investigated the utilization of EC treatment of street food WW. Specifically, the WW originated from noodles & dumpling (ND), and Hainan chicken rice (HC). They reached optimized conditions of 10 min Al-EC at *j* 20 mA cm^{-2}, at which they achieved 95 % COD removal, 84 % oil and grease (O&G) removal from HC WW. While ND WW treatment resulted in removing 98 % of COD, 86 % of O&G at the same optimized conditions. Also, an Al concentration of 0.02 to 0.08 mg L^{-1} was detected in the treated effluents, emphasizing the advantages of rapidity, ease, and high efficiency of EC treatment. Their experiments used Al or Fe electrodes, each 150×70×1 mm, with a total area of 210 cm^2, spaced 2 cm apart. They applied 710 mA with Fe-EC, 940 mA for Al-EC, and 5V with both treatments. They also evaluated the scaling up of their reactor to generate 110 m^3 of WW per day, and estimated the EEC at 64.89 kWh, the electrode consumption at 1220 g m^{-3}, and the final solution temperature at 32.19 °C for Al-EC. While for Fe-EC, the EEC was 30.01 kWh, electrode consumption of 1483 g m^{-3} and final T = 25.64 °C.

EC can also efficiently treat oily WW as it can destabilize the oil-water emulsion balance and then easily separate oil by flotation [26]. Naser *et al.* [125] examined the use of EC for treating petroleum-water WW, demonstrating that increasing current density enhances removal efficiency, while careful control of pH and NaCl concentration is essential for process optimization. Under optimal conditions (15 mA cm^{-2} current density, 45 min treatment time, 1 g L^{-1} NaCl concentration and pH 7), the study achieved a COD removal efficiency of 95.3 % with an energy consumption of 27.78 kWh kg^{-1} COD.

Tomita and Friedler [126] comparatively studied EC for surface and greywater treatment using EC and a new oscillatory-mixing EC, where the normal EC resulted in good removal efficiency of 96 % of NTU (91 % of TP, 34 % of COD, 42 % of DOC and 2 % of TN from surface water. While their GW treatment at pH 8 resulted in removing 96 % of turbidity, 47 % of TP, 72 % of COD, 35 % of DOC and 11 % of TN. They indicated that gas flotation facilitated the separation of contaminants.

EC has also been applied for treating agricultural WW, such as effluents of coffee processing [127], farming [128] and manure [109] WW. For example, Marina *et al.* [127] investigated the study of EC for coffee processing WW using 5-parallel-electrode plates of Al or Fe electrodes, 1 cm electrode spacing, I = 1, 2.5 and 4 A, j = 42.74, 106.84 and 170.94 A m^{-2}, applied voltage of 1.6, 3.8, 5.1 V, and EC processing time t = 15, 30 and 45 min, where they achieved 96.82, 93.99 % of TSS, BOD_5, and along with significant COD removal, respectively, at their optimized conditions at 45 min with 4 A, and j = 170.94 A m^{-2}. Also, [128] studied the EC removal of tetracycline antibiotic from livestock effluents, using 1 cm spaced iron electrodes, solution conductivity = 2000 µS cm^{-1}, pH 4 and I = 0.2 A. Their EC application removed 95 % of TC with a total process cost of \$ m^{-3}.

To further improve the performance of EC systems in wastewater treatment, enhancements are made to the design of conventional EC systems. One such example involves injecting air into the system to promote oxidation and mixing. The study by Gören *et al.* [129] indicated the EC reactor fed with air at a flow rate of 6.0 L min^{-1} was effective in removing arsenic from contaminated water. The removal efficiency of arsenic (initial concentration = 200 µg L^{-1}) exceeded 99 % and the study indicated that the removal was facilitated by the oxidation of As(III) to As(V), followed by adsorption/ /complexation onto aluminium hydroxides generated during the EC process. In another study by Kobya *et al.*, arsenic was removed from groundwater in an air-fed EC system with the optimum conditions for the removal reported to be 0.30 A, pH 7.6, anode surface area of 210 cm^2 and an air-flow rate of 6 L min^{-1} [130]. These conditions resulted in a removal efficiency that exceeded 99 % and the concentration of arsenic was reduced to below 10 µg L^{-1} at the optimum operating conditions. Furthermore, another study investigated the effect of coexisting anions on arsenic removal by the EC process [131]. The presence of anions, including bicarbonate, fluoride, nitrate, phosphate, boron and silicate at relatively high concentrations, did not adversely affect the removal of arsenic during the air-fed electrocoagulation treatment. This study indicates the suitability of air-fed EC in the treatment of more complex matrices such as groundwater. Other studies, as [132-134], also indicate the suitability of air-fed EC for the treatment of arsenic in water.

To overcome electrode passivation challenges, EC reactors with rotating electrode systems have been used. Passive electrodes have drawbacks, including reduced efficiency due to dead zones and lower mass-transfer efficiency from inadequate mixing during operation [135]. The study by Villalobos-Lara *et al.* utilized a rotating cylinder electrode reactor for efficient removal of TDS, COD, turbidity and Cr(III) in industrial tannery wastewater [136]. Similarly, in the work done by Naje *et al.* [91], a rotated bed electrocoagulation reactor was utilized for the treatment of textile water, achieving removal efficiencies of 97, 95, 98, 96 and 98 % for COD, BOD, TSS, turbidity and colour, respectively. The optimum conditions for operating the reactor were documented as pH 4.57, $T = 25$ °C, $j = 4$ mA cm^{-2}, rotational speed = 150 rpm, the hydraulic residence time (RT) = 10 min and the inter-electrode distance (IED) = 1 cm. The system also achieved high phenol removal and showed significant treatment efficiency with low energy consumption and shortened reaction time. In two other studies [137,138], anions and TDS removal were enhanced by the use of a rotating reactor and the economic calculations revealed a suitable cost for the respective operating conditions. In one study, a total operating cost of 0.1766 $ *per* m^3 was reported [138]. Furthermore, the work by Verma and Kumar indicated that the use of a rotating electrode could enhance Cr(VI) removal along with COD removal [139]. At the optimum operating conditions, COD and Cr(VI) removal reached up to 91 and 95 %, respectively. Under these conditions, electrode consumption (ELC = 0.075 kg m^{-1}), stirrer energy consumption (SEC = 0.043 kWh m^{-1}), electrical energy consumption (EEC = 1.77 kWh m^{-3}) and operating cost (OC = 0.36 US $ m^{-3}) were computed.

Techno-economic analysis

It is undeniable that defining the expected cost-benefit aspects in terms of techno-economic evaluation, including energy consumption and cost analysis, and environmental impacts, is crucial for establishing long-term sustainable electrocoagulation processes for water treatment [70,140,141]. In the following lines, a review of the energetic, economical, and environmental considerations of the electrocoagulation process will be provided.

Economic impact

Up-scaling water reclamation applications such as electrocoagulation can be challenging in terms of energy consumption (EEC) [142]. Therefore, conducting a techno-economic evaluation is necessary to determine whether the application is of interest for water treatment. To pursue a costing analysis of the EC treatment, both electrodes and operational energy consumption should be considered (Equation (6)) [143-146]. Electrical energy consumption (EEC, kWh m^{-3}) during EC, can be calculated from Equation (7) in case a direct power course was used.

Process cost = electrodes cost + EEC cost (6)

$$\text{EEC} = \frac{UIt}{V} \tag{7}$$

where U / V is the cell voltage~~,~~ I / A is the electric current, t / h is the operational time and V / m^3 is the volume of wastewater treated [22,147]. Alternatively, if solar power were used, the process costs would differ. Generally, integrating a solar system can significantly reduce operating costs [148], even with the initial capital cost, making the process more sustainable and cost-effective over the long term. Moreover, integrated PV-EC could reduce energy costs by 15 to 17 % [149]. Electrode consumption time(h), t_{EC} / h given in Equation (8), is another crucial, impactful factor on the life cycle evaluation, where the initial electrode mass (M_0 / g) is divided by the electrochemical equivalent of the electrode metal (α) multiplied by the current intensity (I). α / g A^{-1} h^{-1} given by Equation (9) defines the amount of electrode dissolution at a given amount of charge passage.

$$t_{\mathrm{EC}} = \frac{M_{\mathrm{o}}}{\alpha I} \tag{8}$$

$$\alpha = \frac{M_{\mathrm{w}}}{nF} \tag{9}$$

Generally, t_{EC} determination can affect the sustainability judgment of the process and lead to more economically viable choices of electrode materials and configurations.

On the other hand, the reuse of EC by-products can lower the process costs. Notably, a significant by-product of the EC process is hydrogen, which is generated at the cathode. In industry, H_2 gas production is desirable because it is considered a clean fuel source [150]. Moreover, theoretical H_2 production is directly proportional to the current strength, process time and the number of H_2 molecules formed *per* redox electron (e^-) engaged. H represents the stoichiometric number of moles of H_2 produced *per* mole of electrons (e^-) transferred in the electrochemical reaction, which equals ½. The number of H_2 moles formed can be calculated using Equation (10)

$$n_{\mathrm{H_2}} = \frac{ItH}{F} \tag{10}$$

where $n_{\mathrm{H_2}}$ represents the number of moles of hydrogen gas produced during the electrochemical reaction. and F is Faraday's constant [104].

Moreover, electrical conditions, such as V and j, were found to directly affect the electrode consumption, EEC and the consequent EC process expenses [146,151,152]. Figure 4 shows the effect of EC time, V and j on the costing of the EC process from different studies.

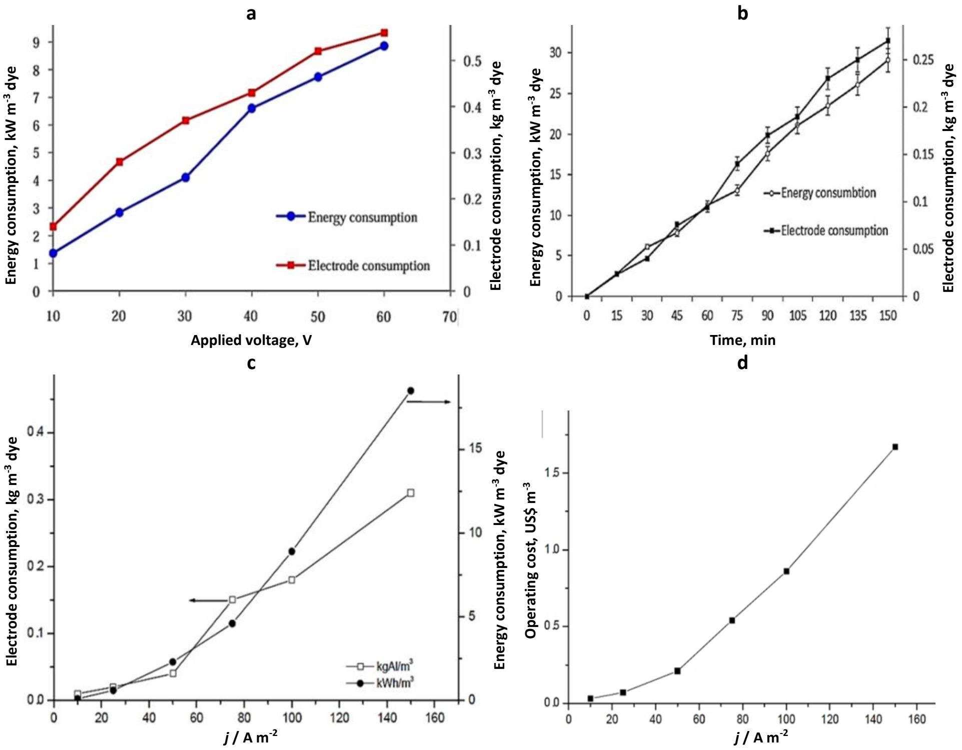

Figure 4. *Influence of a -* V*, b - EC time and c -* j *on the electrode consumption and EEC, and d - influence of* j *on process costs. Adapted and created from findings of [146,151]*

In addition, the widespread use of EC sludge provides an additional economic advantage [153]. For instance, EC sludge can be valorised as an adsorbent, fertilizer, catalyst, or a component in construction blocks and membrane fabrication [153-155].

The use of an external source as a power supply entails more energy requirements in case of electro-potential drop, *i.e.* when power is used in the interfacial reactions between the anode, cathode, and the liquid. Recently, attention to sustainability has focused on striking a proper balance between energy consumption and contaminant removal efficiency, as energy input can partly offset the operating costs of the process plant. The energy consumption for the electrocoagulation process, in batch or continuous mode, is strictly related to the reactor used and to the main operational parameters, such as current density, contaminant type and concentration, and coagulant characteristics. Many large differences are present in the data due to the variability of the aforementioned factors. Often, figures of merit, such as energy cost versus pollutant removal, are useful for obtaining meaningful comparisons of the energy intensity of the methods or for a preliminary idea of the different plant configurations. Some of them, however, are not useful for the comparison because, for instance, the energy requirements depend on the contaminants and flow rates. Zhang *et al.* [147] stated that comparisons with conventional methods are generally meaningless if the additional energy required to produce the agent or chemical agents is not included. Also, compared with desalination processes, the latter had an additional energy demand because the voltage was about twice as high. In addition, they discussed the energetic sustainability of electrocoagulation, with particular attention to which treatment options can be addressed and to improving the energy efficiency of the current state of the art, thereby offering an attractive technological option.

Tom *et al.* [156] found that the cost and economic value of the energy consumption discussed differ depending on the data provided. In other words, the faster the process of coagulation is, the less electrical energy is required. Other researchers have stated that the electrode material, electrode configuration, treatment duration, and the mixture ratio of raw water and effluent strongly influence energy consumption rates [157-159].

EC sludge valorisation

Sludge valorisation is essential to reduce waste disposal into the environment. Also, it is an advantageous benefit that adds to the EC economic and environmental value [153,155]. EC sludge (ECS) has been reused in many applications, such as fertilizers [160,161], adsorbents [162], catalysts [163], additives in construction [164] and membrane manufacturing [165,166] materials, and for valuable metal recovery [153,155].

Huang *et al.* [165] reused ECs in the fabrication of ceramic membranes for treating oil-water emulsions, while Agarwalla and Mohanty [166] developed low-cost sludge/kaolinite ceramic membranes as an eco-friendly water treatment solution. Sharma and Joshi [164] explored its application in construction block manufacturing, whereas Golder *et al.* [167] and Rumky *et al.* [162] demonstrated its effectiveness as an adsorbent for removing reactive dyes and recovering rare earth elements, respectively. In addition, Ghanbari *et al.* [163] investigated ECS as a catalyst for degrading emerging water contaminants, while Rajaniemi *et al.* [160] and Thomsen *et al.* [161] highlighted its potential as a fertilizer due to its high content of essential nutrients such as N and P.

Solid residues generated during electrocoagulation can offer certain advantages, but only when they undergo appropriate treatment and disposal that account for the physicochemical and toxicological properties of the electrolytes used. Given the potential operational consequences and associated electrical costs, thoroughly analysing the composition of the resulting sludge becomes essential. Robust policies are also needed to manage the impacts of these solid residues, particularly the disposal of contaminated sludge, which may require specialized landfill facilities. In addition, proper conditioning of the sludge is often necessary to prevent spontaneous oxidation processes. For these reasons, it is crucial to assess the environmental implications not only of reusing resuspended sludge in electrocoagulation systems but also of disposing of well-conditioned or stabilized sludge in landfills [74].

According to De Oliveira *at al.* [168], the spent electrolyte-containing metallic ions that precipitate as sludge require careful handling to prevent risks to both human health and the environment. The disposal of residues from industrial electrolytic processes, whether solid or liquid, represents one of the most significant challenges associated with the technique due to their hazardous nature and the potential for heavy metal contamination. Although saline solutions can be reused, the ability to regenerate them repeatedly should be viewed as a significant advantage. However, despite promising experimental studies, the greatest potential risk typically lies in the spent solution, which often contains toxic and hazardous heavy metals originating from the treated process [169].

Environmental aspects of electrocoagulation technology

Evaluation of the environmental footprint of the EC process, especially through life cycle assessment (LCA), is essential to enable greater holistic understanding and estimation of the environmental-human risk-to-benefit impact of the process [119,170,171]. The importance of LCA for WWT systems includes minimizing hazardous environmental impacts, optimizing resource use, and improving sustainable water treatment process design [172].

LCA is established through four main systematic phases: goal & scope, inventory data, impact assessment, and interpretation of results. In the first phase of LCA, the research problem, goal, and scope should be well-defined. This should be followed by addressing the system boundaries [173]. This means drawing lines or boundaries around the specific stage addressed by LCA [172]. For example, LCA of WWT can be done for one or more stages of the process, such as ore extraction, transportation, reactor construction, water treatment, mineral reuse, and sludge disposal [70]. The second step of the LCA is collecting LCA data through an inventory. Inventory data can be collected directly from experiments, treatment plants, literature reviews, LCA expert views, or LCA databases [70]. The third step of the LCA is called the impact assessment (LCIA), which includes the necessary classification of the possible environmental impacts, such as the impact of the water treatment system on climate change, greenhouse gases (GHG) emissions, toxicity for humans and ecology, eutrophication, acidification of soil and water, and possible resource use [173]. However, the literature reports only a few LCIA studies in water treatment that include normalization of environmental impacts at the same national and global scales and weighting them with a single unified indicator; for instance, Eco-indicator 99 [174]. The last step in LCA establishment is result interpretation, which includes applying sensitivity analyses and consistency checks, followed by identifying limitations of the treatment method, or stage, and concluding with recommendations to direct the water treatment process or stage towards better practices [172,173].

Studies have utilized the LCA approach to evaluate the sustainability of the EC process for the treatment of contaminated water and findings from these studies are documented and summarized in Table 2.

Table 2. *Summary of findings from LCA studies for EC treatment of wastewater*

Matrix	Contaminant	Functional unit	LCA tools	System conditions	Main contributors to impacts	No. indicators included	Most consequential indicators	Ref.
Wastewater	Manganese	0.5 L of treated water	OpenLCA 1.11 software was used. Inventory models were derived from ELCD and EF databases. Environmental impact potential of midpoints was evaluated using ReCiPe 2016 midpoint (H) method	Electrocoagulation units with titanium electrodes. The LCA was focused on the operation phase of the treatment while construction phase was neglected	Consumption of titanium electrode during manufacture and final water emissions	18	Carcinogenic toxicity, global warming and terrestrial ecotoxicity	[70]
Municipal wastewater	COD, BOD_5, NH^-_4, Total P, NO^-_3 and NO^-_2	1 m^3 of wastewater	Recipe 2016 Midpoint (H) and USEtox v.2 LCIA methods	Electrocoagulation units with aluminium electrodes. The study evaluated the environmental consequences of EC treatment as opposed to direct discharge of untreated water into the river	Energy consumption and disposal of untreated EC precipitated	3 for USEtox v.2 and 10 for ReCiPe	Positive impacts on water eutrophication and toxicity for freshwater and marine ecosystems. Negative impacts on acidification, human toxicity, global warming, terrestrial ecotoxicity, global warming & fossil fuel consumption	[175]
Textile industry wastewater	COD	1 kg of COD removal	ReCiPe 2016 Midpoint (H), CED analysis and End point. Inventory data was obtained from Ecoinvent v3.1 database and Simapro v9.2.0.2 software was used	EC setup with different electrodes. The work explored the impacts of using optimised condition for treating textile industry wastewater. System boundary included electrode manufacture up to wastewater discharge	Energy consumption	18	Global warming and terrestrial ecotoxicity	[141]
Wastewater	COD	1 kWh of energy	ReCiPe 2016 Midpoint (H) and SimaPro 9.3.0.2 software. Inventory was obtained from Ecoinvent database	Evaluation of different energy sources including biogas, hydro, solar, wind and natural gas for powering EC for wastewater treatment.	Emissions of GHG	18	Global warming	[176]
Mining effluent	Fluoride	1 m^3 of treated effluent	ReCiPe 2016 v 1.1 endpoint method	Electrocoagulation systems incorporated with different pH adjustment techniques including using biogenic CO_2 and HCl	Main contribution was from process related to EC and not pH adjustment systems	22	Toxicity (carcinogenic & non-carcinogenic), marine ecotoxicity, climate change potential and particulate matter formation	[177]
Groundwater	Arsenic and fluoride	720 L of contaminated groundwater	Tool for Reduction and Assessments of Chemicals and other Environmental Impacts (TRACI) and Centrum poor Milieukunde, Leiden 2001 (CML 2001). Gabi software was used	A comparison between electrocoagulation with aluminium electrode and adsorption for groundwater treatment	Electricity consumption and dissolution of aluminium electrode	10	Global warming potential, acidification potential, ozone depletion potential, abiotic depletion potential, freshwater aquatic eco-toxicity potential and human toxicity potential	[178]
Textile industry wastewater	COD	1 L of treated dye water	Simapro 9.1.0.8 software was used. Ecoinvent v3.3 database was used for inventory	A comparison of sequential and simultaneous EC with different electrodes and ozonation	Energy and chemical consumption from ozonation process	10	Global warming and human health indicators	[179]

CED = Cumulative energy demand, EC = electrocoagulation, LCA = life cycle assessment, GHGs = greenhouse gases, LCIA = life cycle impact assessment, ELCD = European Reference Life Cycle, EF - Environmental Footprints

Treating municipal water with EC as opposed to direct discharge to rivers (business-as-usual scenario) has the potential to reduce water eutrophication and toxicity for ecosystems dwelling in freshwater and marine environments when compared with direct discharge of untreated water to the environment, as revealed by the LCA study conducted by Leovac Maćerak *et al.* [175]. However, energy is needed to power the treatment process and a significant quantity of waste sludge that needs disposal is generated, leading to adverse consequences on other indicators, including acidification, human toxicity, global warming, terrestrial ecotoxicity, and fossil fuel consumption [175].

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Other studies have made attempts to compare the suitability of EC with other advanced treatment systems using LCA methodology. The comparison between EC and the adsorption system in the study by Goyal and Mondal, revealed that adsorption has up to 8 times more environmental impacts when compared to EC [178]. Coagulants generated in situ have higher adsorption capacity than other forms of dosed adsorbents, making EC more efficient for treating contaminated groundwater. The most affected indicators in the study included global warming potential, acidification potential, ozone depletion potential, abiotic depletion potential, freshwater aquatic eco-toxicity potential and human toxicity potential. In the study by Ahangarnokolaei *et al.* [179], EC exhibited lower environmental impacts than ozonation for treating textile wastewater. The EC process had lower energy consumption relative to the oxygen- and energy-intensive ozonation process. While using EC and ozonation in sequence significantly reduces the treatment time needed and enhances treatment performance, significant environmental impacts have resulted [179]. The findings from the discussed studies indicate that EC can be suitable and competitive with other processes for the treatment of wastewater; it is necessary to optimise EC process to limit its environmental impacts, especially in aspects related to global warming potential and human health indicators.

Another environmental aspect of EC treatment that warrants attention is the toxicological evaluation of electrocoagulated water. Studies that utilise EC for the treatment of wastewater often focus on the removal of specific contaminants, as evident in the studies discussed earlier. However, another important consideration is assessing the overall toxicity of treated water and sludge prior to their discharge into the receiving environment. From a toxicological assessment perspective, after the determination of the efficiency of electrocoagulation in the removal of specific toxic substances, the second important aspect is the detection of the generated by-products, including their concentration and mechanisms of elimination. The third aspect of detailed toxicological testing involves investigating the toxicity of the generated by-products mixture and the risk associated with their possible discharge. Nevertheless, studies that conduct toxicological assessments on electrocoagulated water do not go deep into analysing the specific by-products formed; rather, an overall assessment of toxicity is made through studying the growth inhibition of specific microbes (bioindicator) when exposed to the EC-treated water.

For instance, Lach *et al.* [180] conducted an acute toxicity test on *Daphnia magna* using synthetic textile effluent (contaminated with azo dye) and EC-treated effluent and found that the half maximal effective concentration (EC_{50}) after 48 hours was consistently higher when organisms were exposed to EC-treated water. Several factors may have resulted in the increased toxicity, including leaching of aluminium ions into treated solution, formation of recalcitrant and harmful compounds upon degradation of azo dye and the introduction of other variables such as electrolytes during the EC treatment process. However, in the work by Trigueros *et al.* [181]**,** electrocoagulation treatment was used to lower the acute toxicity of dairy wastewater treated by photochemical oxidation. Diary wastewater treated with photochemical oxidation exhibited extremely high toxicity for *Artemia salina,* with 100 % mortality over the study duration. However, post-treating the sample with electrocoagulation resulted in a much-improved lethal concentration (LC_{50}) value. After applying the optimised treatment conditions, the LC_{50} value ranged from 84 to 88 %, and this allows for the treated dairy effluent to be environmentally safe for disposal in accordance with the location where the study was conducted (Brazil). It is also worth noting that applying EC treatment alone showed medium acute toxicity, with LC_{50} values ranging from 62 to 67 %. Similarly, in the study by Yan *et al* [182]**,** coupling electro-Fenton and electrocoagulation processes for the treatment of tetracycline wastewater resulted in the formation of by-products that are less toxic than the parent compound. 75 % of the formed by-products had lower acute toxicity than tetracycline, as evident by the more favourable predicted oral rat lethal dose (LD_{50}) values for the intermediate compounds. The study also indicated the suitability of coupling electrocoagulation with other oxidation processes to produce less-toxic effluent streams. Furthermore, the study by Zhang *et al.* [128] also revealed that electrocoagulation treatment of livestock wastewater contaminated with tetracycline produces intermediate products that are less toxic than the parent compound (approximately 50 % of the intermediates).

Other studies also analysed the solid residue generated from electrocoagulation treatment to assess its toxic components. The study by Martins *et al.* [183] conducted XRF analysis on solid residue generated from electrocoagulation of textile effluent using stainless steel electrodes and found significant quantities of iron, chromium, nickel and chlorine in sludge. The findings were not surprising because the metallic elements are major components of stainless steel and therefore, leaching of electrode materials contributes significantly to the sludge composition. In a different study by Missinf [184] toxicity of sludge generated from the treatment of roxarsone (3-nitro-4-hydroxyphenylarsonic acid) was evaluated using the toxicity characteristics leaching procedure (TCLP). While the sludge contained arsenic, the leaching solution obtained from subjecting the sludge to TCLP had a concentration of arsenic that was far below the limits of the test, which was 5 mg L^{-1} [184]. The three obtained leachate samples contained arsenic at concentrations of 0.03, 0.113 and 0.45 mg L^{-1}. Based on the figures obtained from the study, it can be concluded that the generated sludge is not considered as a hazardous substance and therefore can be disposed of in a municipal solid waste landfill. The findings from the analysis of electrocoagulated water and the generated sludge are promising; however, there is a need to optimize the electrocoagulation process, especially to limit excessive leaching of the electrode material. As evident in the studies discussed, the leaching of electrode material has the potential to significantly affect the effluent streams generated from the electrocoagulation treatment process.

Conclusions and recommendations

Population growth, rapid industrialization and the emergence of new classes of contaminants have resulted in the need to upgrade conventional treatment methods and replace them with more efficient ones. Due to its numerous advantages, including low footprint, cost-effectiveness, and environmental sustainability, electrocoagulation has emerged as a suitable alternative. Studies conducted at both lab and pilot scales revealed that electrocoagulation is suitable for removing a range of contaminants, including heavy metals and COD, from a variety of wastewater sources, further increasing the appeal of large-scale application of the technology. However, crucial aspects of electrocoagulation need to be improved before the technology can be widely adopted. Findings from LCA studies revealed that the energy cost of electrocoagulation can be significantly reduced by utilizing renewable sources such as solar power to operate the treatment units. Furthermore, excessive leaching and consumption of the electrode material significantly contribute to the ecotoxicity of treated effluent. Hence, appropriate design and selection of electrode materials to meet environmental discharge regulations for treatment and water reclamation is necessary. Overall, there has been significant progress in the use of electrocoagulation technology for wastewater treatment, and the process would benefit from enhancements in reactor design, choice of electrode materials, optimization of operating conditions and new sludge valorisation alternatives.

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