Adıyaman University
Journal of Engineering Sciences
Adyü J Eng Sci 2026;13;e260130
Adıyaman Üniversitesi
Mühendislik Bilimleri Dergisi
Adyü Müh Bil Derg 2026;13;e260130
https://doi.org/10.54365/adyumbd.1787260

Research Article / Araştırma Makalesi

Enhanced Wastewater Treatment by Combination of Bioelectrochemical and Electrochemical Oxidation Processes

Biyoelektrokimyasal ve Elektrokimyasal Oksidasyon Proseslerinin Kombinasyonuyla Gelişmiş Atıksu Arıtımı

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Abstract

The electrochemical oxidation process (EOP) has been successfully applied to a wide range of wastewaters; however, its high energy consumption limits practical application. To overcome this drawback, EOP can be integrated with other treatment technologies. Despite extensive studies on EOP and bioelectrochemical systems (BES) individually, their integrated application for domestic wastewater treatment remains limited, particularly regarding simultaneous evaluation of treatment performance and energy consumption. This study investigates a combined BES–EOP system for real domestic wastewater, focusing on chemical oxygen demand (COD) and nitrogen removal efficiencies as well as energy consumption. A BES was used as a pre-treatment step prior to EOP. The effects of reaction time and pH on pollutant removal and energy consumption were evaluated, and the mechanisms of COD and nitrogen removal were also examined. The results showed that the combined BES–EOP system achieved a COD removal efficiency of 68.2±3.4% after 5 hours of BES treatment followed by 1 hour of EOP treatment, which was higher than that of standalone BES (42.5±0.9%) and EOP (58.6±2.8%) systems after 6 hours of operation. Similarly, NH_3–N removal reached 26.9±1.6%, compared with 10.0±0.9% and 10.3±0.2% for BES and EOP, respectively. While COD removal did not significantly improve beyond 6 hours, extended operation enhanced nitrogen removal. Notably, the integrated system reduced treatment cost by approximately 81.3% compared with standalone EOP. Overall, the combined BES–EOP system demonstrated higher treatment efficiency and lower energy demand, indicating strong potential for sustainable domestic wastewater treatment applications.

Keywords: *Electrochemical oxidation, bioelectrochemical systems, wastewater treatment.*

Öz

Elektrokimyasal oksidasyon prosesi (EOP), çok çeşitli atık suların arıtımında başarıyla uygulanmıştır; ancak yüksek enerji tüketimi, pratik uygulamalarını sınırlamaktadır. Bu dezavantajı gidermek için EOP, diğer arıtım teknolojileri ile entegre edilebilir. EOP ve biyoelektrokimyasal sistemler (BES) ayrı ayrı kapsamlı şekilde incelenmiş olmasına rağmen, özellikle evsel atık su arıtımında bu sistemlerin entegrasyonu ve arıtım performansı ile enerji tüketiminin eş zamanlı değerlendirilmesi sınırlı kalmıştır. Bu çalışma, gerçek evsel atık sular için kombine bir BES–EOP sistemini incelemekte; kimyasal oksijen ihtiyacı (KOİ) ve azot giderim verimliliği ile enerji tüketimine odaklanmaktadır. EOP öncesinde bir ön arıtım aşaması olarak BES kullanılmıştır. Kirletici giderimi ve enerji tüketimi üzerinde reaksiyon süresi ve pH'ın etkileri değerlendirilmiş, ayrıca KOİ ve azot giderim mekanizmaları incelenmiştir. Elde edilen sonuçlara göre, kombine BES–EOP sistemi, 5 saat BES ve ardından 1 saat EOP işletimi sonunda %68.2±3,4 KOİ giderim verimliliği sağlamış olup, bu değer tek başına BES (%42.5±0.9) ve EOP (%58,6±2,8) sistemlerinden daha yüksektir. Benzer şekilde, NH_3–N giderimi %26.9±1.6 olarak elde edilmiş; bu değer BES ve EOP için sırasıyla %10.0±0.9 ve %10.3±0.2 olarak belirlenmiştir. KOİ giderimi 6 saatten sonra anlamlı şekilde artış göstermemiş, ancak daha uzun işletme süreleri azot giderimini artırmıştır. Ayrıca, entegre sistemin tek başına EOP'a kıyasla arıtım maliyetini yaklaşık %81.3 oranında azalttığı belirlenmiştir. Genel olarak, kombine BES–EOP sistemi daha yüksek arıtım verimliliği ve daha düşük enerji gereksinimi sağlayarak sürdürülebilir evsel atıksu arıtımı için güçlü bir potansiyel sunmaktadır

Anahtar Kelimeler: *Elektrokimyasal oksidasyon, biyoelektrokimyasal sistemler, atık su arıtımı*

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Geliş Tarihi (Received):23.09.2025, **Kabul Tarihi (Accepted):** 19.06.2026

1. Introduction

Due to population growth, urbanization, and economic development, the volume of wastewater has increased significantly over time, particularly in developed countries, as a result of improved living standards [1]. Approximately 380 billion m^3 of municipal wastewater is generated globally each year, and this volume is expected to increase by 51% by 2050 [2]. In addition, current conventional wastewater treatment systems raise significant concerns in terms of both economic and environmental aspects [3]. Aerobic processes, which dominate domestic wastewater treatment systems worldwide, have high energy demands due to aeration requirements, as well as significant land use and sludge disposal volumes. On the other hand, anaerobic processes are associated with greenhouse gas emissions, odor and corrosive gas production, as well as higher capital costs and longer start-up times compared to aerobic systems [4]. Due to these challenges, it is essential to develop new approaches and technologies for wastewater treatment that overcome current limitations while improving treatment efficiency and enabling energy recovery [5]. As an alternative method, electrochemical oxidation processes (EOP) are wastewater treatment techniques that involve the application of electrical current through electrodes [6]. In EOPs, the electron is used as a clean reagent to degrade pollutants such as toxic and recalcitrant pollutants. Pollutants can undergo electrochemical oxidation (EO) through two different oxidation mechanisms as direct oxidation, and indirect oxidation. In the direct oxidation mechanism, the pollutants are absorbed and oxidized on the anode surface by the anodic electron transfer reactions. Indirect oxidation takes place in the bulk liquid, which is mediated by the strong oxidizing agents that are continuously produced electrochemically such as hydroxyl radicals, hydrogen peroxide, or/and chlorine [7]. The efficiency of EOP has been demonstrated for a wide range of wastewater types in the literature, including those from the chemical industry (e.g. petrochemical [8,9], detergent [10], pharmaceutical [11,12], pulp and paper [13]), textile industry [14], tannery industry [15], coking industry [16], landfill leachate [17]. It has also been reported that EOP can effectively remove chemical oxygen demand (COD), ammonia, and microorganisms under appropriate operating conditions [18]. EOPs have recently gained considerable attention for the removal of a wide range of pollutants from wastewater, due to their advantages such as compact reactor design with reduced space requirements, shorter treatment times with high efficiency, low chemical consumption, lower investment and operational costs, and ease of operation compared to conventional treatment systems [19]. Despite the advantages of EOPs, these systems are associated with high operating costs due to elevated energy consumption, which remains one of the main limiting factors for their large-scale field applications [19].

Hybrid processes are considered one of the potential solutions for reducing operating costs. In these systems, electrochemical oxidation processes (EOP) are combined with biological treatment processes to lower energy consumption, reduce treatment time, and improve overall process efficiency [20]. Hybrid processes can be combined through sequencing the EOP as pre-treatment, post-treatment, or integrated treatment [21]. When wastewater contains toxic substances and exhibits a low BOD_5/COD ratio, an indicator of biodegradability, EOP can be applied as a pre-treatment step to increase the biodegradability of organic pollutants and reduce toxicity, thereby improving the performance of subsequent biological treatment processes. Previous studies have demonstrated that EOP can be effectively applied as a pre-treatment step prior to membrane bioreactors [22], anaerobic digestion [23,24], waste activated sludge [25] and constructed wetlands [26] to enhance biodegradability and improve overall wastewater treatment efficiency. Moreover, a single pollutant, such as an organophosphorus pesticide of phosmet [27], a reactive dye of Procion Scarlet [28], phenol [29] can be treated can be effectively treated when EOP is applied as a pre-treatment step. On the other hand, several studies have reported the application of EOP as a post-treatment or polishing step in integrated wastewater treatment systems, particularly for the removal of residual refractory organic pollutants following biological or primary treatment stages. These applications include different types of wastewaters such as cooking wastewater [30], dairy manure [31], landfill leachates [32] and olive pomace leachate [33].

Bioelectrochemical systems (BESs) are a newly developed and environmentally friendly biotechnology that proposes an alternative wastewater treatment method to conventional systems [34]. Although BESs have diverse applications, they share the same principle, which is an oxidation half-reaction occurs at the anode, and a reduction half-reaction occurs at the cathode. Among all BESs, the microbial fuel cell (MFC) is the most studied type [35] in which organic matter in wastewaters is oxidized using electrochemically active bacteria (EAB) under anaerobic conditions. In previous studies, MFCs have been successfully applied to the treatment of various wastewaters, while simultaneously generating electricity [36–44]. To enhance wastewater treatment performance, MFCs have recently attracted increasing attention when integrated with other treatment technologies. Researchers have combined advanced oxidation processes (AOPs) such as Electro-Fenton (EF), Photoelectrocatalysis (PEC), and Electrocatalytic Oxidation (EO) with MFCs to develop an emerging AOPs-MFC treatment technology. AOPs are applied in integrated treatment systems as pre-treatment and post-treatment steps which involve the combination of two or more water treatment technologies within a single treatment train or unit. This integrated approach allows sequential or simultaneous processes to enhance overall treatment efficiency. Previous studies have primarily investigated hybrid EF–MFC systems mainly targeting persistent organic pollutants and antibiotics in various wastewater [45–49]. In contrast, direct EO–MFC systems remain relatively underexplored in the recent literature. Huang et al. [50] was conducted one of the earliest studies to evaluate a combined MFC and EOP for the treatment of coking wastewater. The MFC was first applied to degrade organic pollutants via electrochemically active bacteria while generating electricity, followed by EOP treatment of the effluent. The integrated system achieved total COD and nitrogen removal efficiencies of 82±1% and 68±1%, respectively. However, the process was limited by a long treatment duration (7 days for MFC and 7 hours for EOP) and the requirement for phosphate and chloride addition. Zou et al. [51] developed an electrolysis cell coupled with a microbial fuel cell (EC–MFC) system to treat methyl red-containing wastewater. The integrated system achieved complete decolorization and a COD removal efficiency of 89.3%, along with a voltage output of 0.56 V on the 5th day of operation. In another study, Mohanakrishna et al. [52] developed an integrated system combining an electrochemical cell (EC) and a MFC for the treatment of oil extraction wastewater. Following 4 hours of electrochemical degradation at a current density of 71 mA/cm², the treated effluent was subsequently introduced into the MFC for an additional 10 hours of treatment. Overall, the system achieved removal efficiencies of 89% for total petroleum hydrocarbons and 89.6% for COD, while generating a power density of 2.74 mW/m². Recently, Chen et al. [53] investigated an integrated system combining a MFC and an EOP to enhance the treatment efficiency of recalcitrant wastewater at where phenol was used as a refractory organic pollutant. In this configuration, the EOP was applied as a pre-treatment step, and the results demonstrate that compared with MFC treatment alone, this hybrid system shortens the treatment cycle and improves degradation efficiency. Even though significant improvements noted in the EOPs, the combined EOP technologies have not been fully developed yet and need extensive research to find more sustainable and cost-effective processes for efficiently treating wastewater. Also, based on previous studies, coupled EOPand BES offer improved treatment capacity relative to single treatment methods, enabling effective removal of pollutants from a wide range of wastewaters, including complex and refractory streams. Therefore, this study presents a novel integration of electrochemical oxidation process and a flow-through three-electrode bioelectrochemical system for the treatment of real domestic wastewater, enabling the simultaneous removal of COD and nitrogen. Unlike most previous studies that rely on model pollutants or synthetic wastewater, or that focus solely on organic matter removal, this work demonstrates enhanced treatment performance under realistic conditions while also addressing energy consumption, treatment time and mechanisms.

2.Materials and Methods

2.1. Bioelectrochemical System Configuration and Operation

The 3-electrode setup consisting of a Ag/AgCl reference electrode [54], a graphite felt working electrode (WE) and counter electrode (CE) (HP Materials Solutions, Inc., Woodland Hills, CA, USA) (Figure 1). The

anodes consisted of cylindrical coupons of graphite felt with a radius of 1.1 cm. We connected the graphite felt anode with a screwed titanium wire. 0 $V_{Ag/AgCl}$ potential was used to polarize the WE using a previously developed custom potentiostat [55] and current was recorded as a function of time in the BES. The working volume of the BES reactor is 120 ml and the feed stream (wastewater) flows perpendicular to the flow-through WE with 10 ml/min flow rate and the experiment carried out at room temperature (~25 °C) as shown in Figure 1. Municipal wastewater and mixed microbial culture as inoculum were obtained from the influent and anaerobic tank, respectively, at the Moscow, Idaho, Water Reclamation and Reuse Facility. The experiment starts with enrichment period during. In this stage, the biofilm was grown on the working electrode surface to enrich electrochemically-active biofilms were from mixed culture [34] with 20 mM acetate-supplemented wastewater. We used mixed culture as a rich inoculum because we successfully grew anodic electrochemically-active biofilms in preliminary experiments using this inoculum. The enrichment period lasted until the current values reached a pseudo-steady- state. After the enrichment process was completed, the system was operated in batch mode with fresh domestic wastewater. When we obtained the current at pseudo-steady-state, we lasted the enrichment period, and the system was started to operated in batch mode with real domestic wastewater.

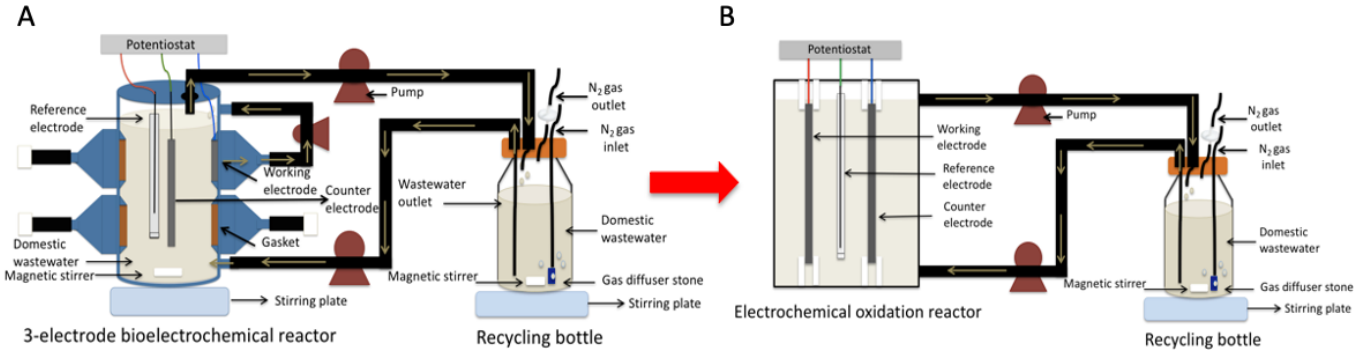


Figure 1. Schematic diagram of the BES reactor (A) and EO reactor (B), illustrating the working electrode (WE), reference electrode (RE), and counter electrode (CE) in a three-electrode configuration connected to a potentiostat (self-assembled setup). Dimensions and proportions are shown for illustrative purposes only.

The wastewater was chemically characterized in terms of COD, NH_3-N, conductivity, and pH to evaluate its suitability for biological and electrochemical treatment processes. The general characteristics of the wastewater supplied are summarized in Table 1.

Table 1. Domestic Wastewater Properties

Parameter	Value
COD (mg/L)	150±15
NH_3-N (mg/L)	52.5±4.5
Electrical conductivity (μΩ/cm)	450±50
pH	7.4±0.2

2.2. Electrochemical Oxidation Configuration and Operation

The working volume of the electrochemical oxidation reactor was 500 ml and working and counter electrodes were made of graphite felt (HP Materials Solutions, Inc., Woodland Hills, CA, USA) and a rectangular shape with a projected surface area of 56 cm^2 (7 cm × 8 cm). We also used a custom-made Ag/AgCl reference electrode (Figure 1). We set the gap between the working and the counter electrode to 1 cm. The partially treated domestic wastewater in BES reactor was treated electrochemically by polarizing working electrode at 10 $V_{Ag/AgCl}$ using a the Gamry Series G 1000™ potentiostat. The experiment carried out at room temperature (~25 °C).

2.3. Analytical Methods

We collected well-mixed wastewater samples and analyzed using standard protocols to test wastewater performance. First, we centrifuged the samples for 2 minutes at 10,000 rpm and then filtering through a 0.45 mm of pore size of Tisch Scientific membrane filters before measurements. The dissolved COD was determined using mercury-free potassium dichromate method (low range 0-150 mg/L COD and high range 20-1500 mg/L COD vials). The NH_3-N was determined using salicylate method (high range 2-47 mg/L NH_3-N vials). The NH_3-N were measured following HACH procedures with a spectrophotometer (DR 3900 HACH Company, CO). The total removal efficiencies of COD and NH_3-N were calculated by percent difference between the initial and final concentrations in the bulk solution in each sampling period [53]. The COD removal efficiency (%) was calculated using the following equation:

$$\text{COD removal efficiency (\%)} = \frac{COD_{in} - COD_{out}}{COD_{in}} \; x \; 100 \tag{1}$$

Where COD_{in} and COD_{out} represent the influent and effluent COD concentrations (mg/L), respectively.

Similar to the calculation of COD removal efficiency (%), the NH_3-N removal efficiency (%) was calculated using the following equation:

$$NH_3 - \text{N removal efficiency (\%)} = \frac{(NH_3 - \text{N})_{in} - (NH_3 - \text{N})_{out}}{(NH_3 - \text{N})_{in}} \; x \; 100 \tag{2}$$

Where NH_3-N_{in} and NH_3-N_{out} represent the influent and effluent NH_3-N concentrations (mg/L), respectively.

Hence, energy consumption is expressed as kWh/kg COD removed by normalizing the average energy consumption to the volume of treated wastewater. The energy consumption (EC) was calculated using Eq. 3 [56] The energy consumption (EC) was calculated using Eq. 3 [56].

$$\text{EC} = \frac{\text{V x A x t}}{\Delta\text{COD x } S_v} \tag{3}$$

Where V is the applied voltage (V); A is the current flowing through solution (A); t is the experimental time (h); ΔCOD is the removed COD (mg/L): and S_v is the sample volume (L).

At each sampling period (the beginning and end of each experiment), the pH of the medium was measured by collecting a 50 ml sample from the bulk medium using a calibrated pH meter (Denver Instrument Company, Arvada, Colorado USA) in a room temperature of 25 ^{o} C.

3.Results and Discussion

3.1 COD Removal

The removal of COD in different system configurations is illustrated in Figure 2. Initially, the BES was operated by controlling the electrode potential against the Ag/AgCl reference electrode. The reason of using 0 $V_{Ag/AgCl}$ as a polarized potential was that electrochemically active microorganisms exhibit higher selectivity at lower overpotentials, as confirmed in previous studies [34,56]. After the enrichment period, the average current density was determined to be 0.35 mA/cm² in the BES reactor. In the subsequent experimental setup, the EO reactor was operated at a constant applied voltage of 10 V. The initial current was 100 mA, corresponding to an initial current density of 1.79 mA/cm². The current gradually decreased during operation, indicating changes in solution resistance and electrode surface conditions during

electrochemical oxidation [57,58]. Previous studies by Díaz et al. [59] and Vijayaraghavan et al. [60] reported that increasing the applied voltage in the EOP enhances COD removal for explanations for aquaculture saline wastewater ([Cl^-]=26,167 mg/L) and distillery spent wash ([Cl^-]=8,000-8,500 mg/L), respectively due to improved generation of reactive oxygen species and accelerated anodic oxidation kinetics; however, excessive electrical input may reduce current efficiency due to competing oxygen evolution reactions efficiency. More recent studies have further confirmed a similar trend in real wastewater systems, in agreement with the results reported in [31,61,62]. Therefore, we used the maximum voltage that can be applied by the Gamry Series G 1000™ potentiostat. Finally, the combined BES–EO system was operated under the same conditions as those applied in the individual operations. Our experimental results show that the COD removal rates were 59.5±1.5 mg/L and 83.5±6.5 mg/L.h in the BES and EO reactors, respectively. Also, COD removal efficiencies were %42.5±0.9 and %58.6±2.8 within six hours of operation in the BES and EO reactors, respectively. On the other hand, the combined BES-EOP system showed the COD removal rate was 90.5±9.5 (mg/L) and COD removal efficiency (%) was 68.2±3.4 within 5 hours of operation in BES and then 1 hour of operation in the EO reactor (Figure 2). Moreover, COD removal efficiencies (%) for the BES and EOP systems were 50.3±1.8 and 60.2±1.3, respectively (data not shown) after 10 hours of operation, whereas the combined BES-EOP system achieved a higher COD removal efficiency of 69.3±1.2% over the same operational period. In summary, the combined BES–EO system resulted in higher COD removal efficiency compared to individual systems. However, no significant increase in COD removal efficiency was observed in any reactor after 6 hours of operation. All data are presented as means of two biological replicates, with standard deviations shown as error bars.

The treatment mechanism is a key factor in successful wastewater treatment. Therefore, the possible mechanisms responsible for COD removal were investigated in this study. In the EOP, organic pollutant oxidation occurs through two distinct pathways, namely direct anodic oxidation and indirect oxidation [63]. Direct anodic oxidation is known as electrochemical cycling (conversion), active oxygen species (AOSs) is chemically adsorbed on the electrode surface, and organic compounds are partially oxidized (Eq. 4-7) [64,65].

$$MO_x + H_2O \rightarrow MO_x(\bullet OH) + H^+ + e^- \quad (4)$$
$$MO_x(\bullet OH) \rightarrow MO_{x+1} + H^+ + e^- \quad (5)$$
$$R + MO_{x+1} \rightarrow RO + MO_x \quad (6)$$
$$R + MO_x(\bullet OH)_n \rightarrow MO_x + CO_2 + H^+ + e^- \quad (7)$$

Where MO_x is the oxide anode, $MO_x(\bullet OH)$ is physisorbed active oxygen, MO_{x+1} is chemisorbed active oxygen, R is general organic compounds.

Strong oxidants, including chlorine species, hypochlorite ions, and hydroxyl radicals, are electrochemically produced at the anode surface, where they subsequently oxidize contaminants in the bulk liquid phase [66]. Among them, chlorine is one of the most common electrochemical oxidants, which forms by in-situ electrochemical chloride oxidation, directly converting chloride ions to chlorine radicals without any chemical additions (Eq. 8) [65]. Chloride converted to strong oxidants such as chlorine/hypochlorite at anode (Eq. 9 and 10) [65].

$$2Cl^- \rightarrow Cl_2 + 2e^- \quad (8)$$
$$Cl_2 + H_2O \rightarrow HOCl + Cl^- + H^+ \quad (9)$$
$$HOCl \rightarrow H^+ + OCl^- \quad (10)$$

Domestic wastewater typically contains approximately 50–150 mg/L of chloride ions [67]; therefore, one possible mechanism for organic pollutant removal is the electrochemical oxidation of chloride (Cl^-) to chlorine (Cl_2), which is subsequently hydrolyzed to form hypochlorous acid (HOCl) (Eq. 9). Depending on the solution pH, HOCl is changed to the hypochlorite ion (OCl^-) (Eq. 10). The sum of Cl_2(g), HOCl(aq)

and OCl-(aq) is termed as free chlorine. Since the pH is one of the important influencing factors for the wastewater treatment performance, changes in pH were examined during the experiments. The initial pH of the raw domestic wastewater was around 7.4±0.2 and pH started to increase and finally reached around 8.2±0.1 at the end of the treatment of 6 hours of treatment (Figure 3).

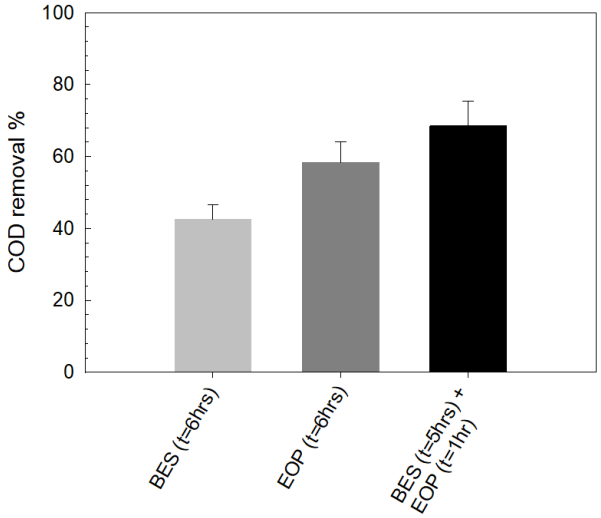


Figure 2. COD removal efficiency.

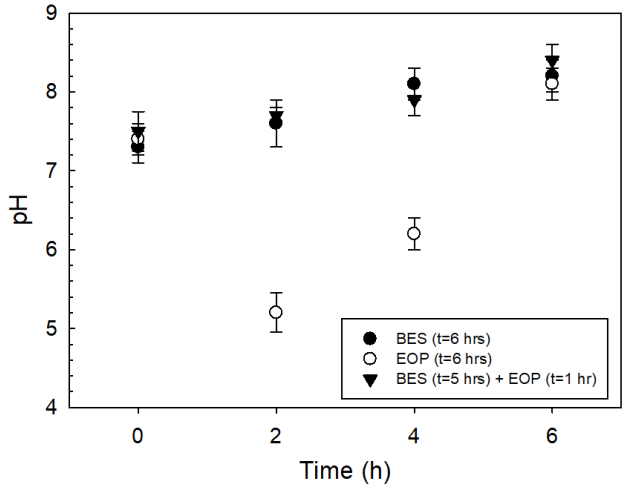


Figure 3. Changes in mean pH during the operation period.

The combined BES–EOP system exhibited a slight increase in pH, comparable to that observed in the BES reactors. Within the pH range of 7.4–8.2, free chlorine exists primarily as HOCl and OCl^-, with OCl^- predominating under these conditions. In contrast, the pH in the EOP did not exhibit a continuous increasing trend over the 6 hours operation period. It initially decreased to 5.2±0.2 after 2 hours of treatment and subsequently increased, although it remained below 8.15±0.5 at the end of the process. Unlike the BES and the combined BES–EOP systems, HOCl is the predominant species of free chlorine in the EOP, where it serves as a strong oxidizing agent. As reported previously, electrochemical oxidation of contaminants is accompanied by side reactions and pH was influenced by the anodic oxygen evolution reaction, cathodic hydrogen evolution reaction, and oxygen reduction reaction in electrochemical systems are described by the following half-reactions (Eq. 11-13) [68].These reactions may be responsible for the increase in pH during oxidation.

$$2H_2O \rightarrow O_2 + 4H^+ + 4e^- \quad (11)$$
$$2H_2O + 2e^- \rightarrow H_2 + 2OH^- \quad (12)$$
$$O_2 + 2H^+ + 2e^- \rightarrow H_2O_2 \quad (13)$$

When the experimental setup was designed, the EOP was initially applied as a pre-treatment step to enhance the biodegradability of organic pollutants in domestic wastewater. Therefore, operating the BES after the EOP was expected to improve overall wastewater treatment performance. However, the preliminary results indicated that applying the EOP prior to the BES did not lead to an increase in the COD removal rate. This phenomenon can be explained by the fact that by-products generated during the EOP may exert detrimental effects on specific bacterial phyla, such as *Bacteroidetes*, *Firmicutes*, and *Proteobacteria*, which are responsible for organic matter removal in BES systems. In addition to these phyla, *Epsilonbacteraeota* were also identified as a dominant phylum in the BES reactors in our previous study [34]. On the other hand, the dominant genera reported in the BES reactors were *Geobacter*, *Rikenellaceae*, *Paludibacter*, *Arcobacter*, *Lentimicrobiaceae*, *Clostridiales*, *Macellibacteroides*, *A7P-90m*, *Desulfovibrio*, and *Sulfurospirillum* [34]. Among the dominant genera in the BES reactors, *Geobacter* was identified as the predominant genus in the anodic biofilm. It is well known for its capability to oxidize organic matter and mediate extracellular electron transfer to the anode surface. The reactive oxygen species (ROS) generated include hydrogen peroxide (H_2O_2), hydroxyl radicals ($^{\bullet}OH$), and superoxide (O^{-}_2). Depending on their concentration and exposure time, these ROS can induce irreversible damage to biofilm cells. H_2O_2 and Cl_2 are both potent inhibitors of certain microbial metabolic activities and can also suppress the electrochemical activity and COD consumption of the anodic biofilm. In a study, An et al. [69] designed a stacked single chambered MFC for H_2O_2 generation to inhibit the anode-respiring bacteria by stimulating cathodic voltage reversal. Besides, Lu et al. [70] reported that the production of ROS inhibits the growth of *Geobacter sulfurreducens* PCA. However, contrary studies also reported in which catalase, superoxide dismutase and super-oxide reductase enzymes were encoded by *Geobacter* genomes for ROS detoxiation [71]. Therefore, ROS accumulation was prevented, which in turn allowed *Geobacter* cells to maintain viability after prolonged exposure to air [72]. In addition to ROS, routinely oxidizing Cl^- to Cl_2 (Eq.8) and OCl^- (Eq.10) at the anode is used to sterilize and kill microorganisms. To eliminate the production of these reactive chlorine species, different solutions can be applied such as using Cl- free electrolyte or an ion exchange membrane to physically separate the anode and bacteria.

3.2 NH_3-N Removal

The removal of NH_3-N in different system configurations is illustrated in Figure 4. Initially, the BES was operated by controlling the electrode potential against the Ag/AgCl reference electrode. In the following experimental set-up, we operated an EO reactor in which 10 V of an electric voltage was applied. Previous studies show that increasing applied voltage can cause an increase in the removal efficiency of nitrogen products [53]. As explained in the previous part, the higher current generation increases the reduction reaction of chloride ions which helps to reduce NH_3-N concentration as well as COD. The combined BES-EOP system was operated under the same conditions as individual operations. Our experimental results show that the NH_3-N removal rates were 4.0±1.0 and 4.25±0.25 (mg/L) and NH_3-N removal efficiencies (%) were 10.0±0.9 and 10.3±0.2 within 6 hours of operation in the BES and EO reactors, respectively. On the other hand, combined BES with EOP system showed the NH_3-N removal rate was 10.5±1.5 (mg/L) and NH_3-N removal efficiency (%) was 26.9±1.6 within 5 hours of operation in BES and then 1 hour of operation in the EO reactor (Figure 4). Moreover, while NH_3-N removal efficiencies (%) with BES and EOP were found 14.1±0.3, and 18.7±2.4 respectively (data not shown) within 10 hours of operation, combined BES with EOP systems achieved 49.2±1.6 % of NH_3-N removal efficiency within 10 hours of operation. Contrary to the COD removal efficiency results obtained within 10 h of operation, increased operation time significantly improved NH_3–N removal efficiency in the combined BES–EOP system. As a result, one of the critical conclusions of this study was that NH_3–N removal was enhanced with increasing operation time in the combined system. Nitrogen removal mechanisms in wastewater treatment systems can be highly complex, as numerous environmental and operational parameters may influence the removal efficiency of the system. In our previous work, we deeply investigated the mechanisms of nitrogen removal in our 3D printing bio-electrochemical reactor [34]. In this study, the same reactor configuration and experimental conditions as those used in our previous study were employed. Based on our previous findings, nitrogen removal mechanisms were initially considered to be dominated by anodic ammonia

oxidation (Eqs. 14–16), sulfidogenesis (Eq. 17), denitrification, and the reduction of NO_3^-–N to NO_2^-–N by *Geobacter* species [34].

$$2NH_4^+ \rightarrow N_2 + 8H^+ + 6e^- \quad (14)$$
$$2NH_4^+ + 2H_2O \rightarrow 2NO_2^- + 8H^+ + 4e^- \quad (15)$$
$$2NH_4^+ + 3H_2O \rightarrow 2NO_3^- + 14H^+ + 10e^- \quad (16)$$
$$SO_4^{-2} + 2NH_4^+ \rightarrow S + N_2 + 4H_2O \quad (17)$$

On the other hand, strong oxidants (OCl^- and HOCl) are responsible for the indirect oxidation of NH_3-N to N_2 in the EOP [73]. When energy (voltage) is applied to wastewater, water molecules break up to evolve O_2 and H_2 (Eqs. 18) [74].

$$2H_2O + \text{Energy} \rightarrow 2H_2 + O_2 \quad (18)$$

In an alkaline medium, the electrolysis in an aqueous chloride solution is the critical step of water dissociation and Cl_2 is generated in Cl^- solutions, the balanced equations are shown in Eqs. (19) [74] and (20) [75] .

$$2H_2O + 2e^- \rightarrow H_2 + 2OH^- \quad (19)$$
$$2Cl^- \rightarrow Cl_2 + 2e^- \quad (20)$$

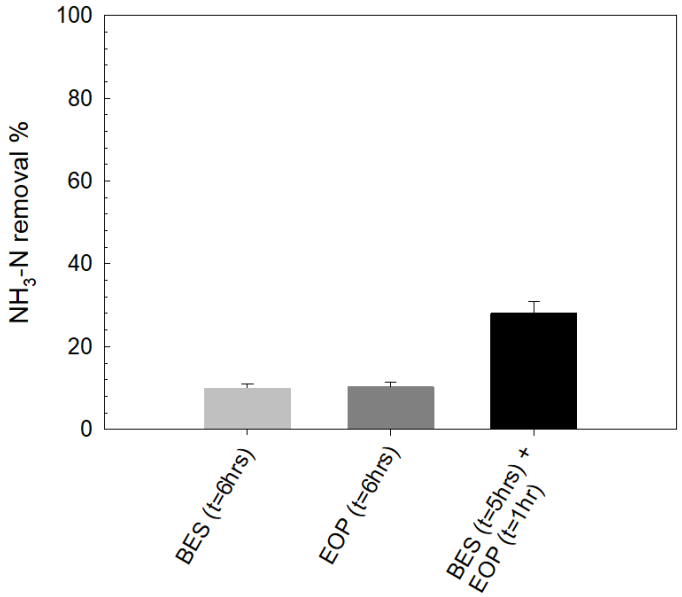


Figure 4. NH_3-N removal efficiency.

As mentioned in previous section, domestic wastewater contains approximately 50-150 mg/L of chloride [67]. Therefore, possible removal mechanisms of nitrogen could be Cl_2 is hydrolyzed to form HOCl (Eq. 9), which is subsequently changed to OCl^- (Eq. 10), depending on the pH of the solution (Eqs. 21 and 22)[76].

$$2NH_4^+ + 3HOCl \rightarrow N_2 + 3H_2O + 5H^+ + 3Cl^- \quad (21)$$
$$2NH_4^+ + 3OCl^- \rightarrow N_2 + 3H_2O + 2H^+ + 3Cl^- \quad (22)$$

NH_3-N removal in the 10 V electrochemical system was likely dominated by the indirect oxidation of NH_3-N to N_2 in the presence of HOCl and OCl^-, as described in Eqs. (21) and (22), due to the pH changes observed in the wastewater during the experiments. The initial pH of the raw domestic wastewater was approximately 7.4±0.2. Following 2 hours of EOP treatment, the pH decreased to 5.2±0.2. Thereafter, the

pH gradually increased, although it remained below 8.15±0.5 at the end of the treatment period. Therefore, the wastewater pH remained between 7.4 and 8.2 during the treatment process, which is within the range favorable for the formation of OCl^- and HOCl. Accordingly, the oxidation of NH_3-N to N_2 via OCl^-/HOCl-mediated reactions was considered the dominant nitrogen removal pathway. HOCl is considered a strong oxidizing agent for ammonium under alkaline conditions. As discussed previously, the EOP was initially applied as a pre-treatment step to enhance the biodegradability of organic pollutants in domestic wastewater during the experimental design. However, partial application of the EOP prior to the BES did not increase the NH_3-N removal rate. The observed behavior may be attributed to the inhibitory effects of EOP by-products on bacterial communities involved in nitrogen removal in the BES. In addition, our previous study reported that *Geobacter*, *Desulfuromonas*, and *Desulfovibrio* were the dominant species in the BES system [34]. These microorganisms have been previously identified as being responsible for chlorine removal from wastewater, which directly influences the NH_3-N removal rate [77]. Furthermore, a previous study demonstrated that hydrogen production at the cathode can suppress the denitrifying activity of the anodic biofilm in bioelectrical reactors [78].

3.3 Energy Consumption and Economic Aspects

The conventional activated sludge process is one of the most widely adopted wastewater treatment technologies. However, despite its extensive application, it is associated with several limitations, the most significant of which include high energy consumption and substantial land requirements [79]. Hence, energy consumption is a key parameter for evaluating the performance of wastewater treatment technologies. Using Eq. 3, the EC was calculated as 24 kWh/m^3 after 6 hours of operation in the EO reactor in this study. Since the estimated energy consumption is strongly influenced by substrate type, substrate concentration, conductivity, pH, applied current, and electrode material [21], a wide range of values has been reported in the literature for EOPs, ranging from 0.315 to 785 kWh/m^3 [6] and direct comparison of energy consumption values reported in the literature is challenging, as significant variations arise from differences in reactor design, electrode materials, wastewater composition, and operating conditions. In a study by Xing et al. [80], three different anodes (BDD, Ti/SnO_2-Sb/PbO_2 (PbO_2), and Ti/SnO_2-Sb (SnO_2)) were evaluated to investigate their effect on the EO of phenol. The BDD anode was found to be the most suitable material for phenol removal, exhibiting the lowest specific energy consumption per unit mass of COD at 17.6 kWh/kg COD, compared with PbO_2 (62.6 kWh/kg COD) and SnO_2 (170.2 kWh/kg COD). Another study by Luu et al. [81] evaluated the EO of biologically pre-treated landfill leachate using Ti/BDD and Ti/RuO_2 anodes in a continuous-flow sequencing batch reactor. The results showed that the Ti/BDD anode exhibited significantly lower energy consumption (30 kWh/kg COD) compared with the Ti/RuO_2 anode (190 kWh/kg COD). These studies indicate that anode material plays a key role in determining the energy consumption rate in EO systems. Consequently, both the cost of anode materials and the associated energy consumption represent the primary factors governing the economic feasibility of EOPs. On the other hand, typical conventional wastewater treatment systems, such as activated sludge processes, consume approximately 0.7–2 kWh/kg COD of energy [82]. BESs have been developed and operated as alternative treatment technologies, with the aim of reducing energy demand, particularly by minimizing the energy required for wastewater aeration. In practice, MFC consumes energy much less energy than conventional wastewater treatment systems [83]. It is straight forward to calculate specific energy consumption for MFCs. However, it is difficult to calculate for 3-electrode system in BES technology because even though the overall metabolic energy released at the anode is known, it does not consider other losses including ohmic losses. Therefore, we used an estimated energy consumption rate reported to provide estimated costs for all systems in this study as a reference level. Where EOP consumes 24 kWh/kg COD, BES can greatly reduce energy consumption (<0.1 kwh/ kg COD). Based on the results, the specific energy consumptions of combined BES–EOP system would be approximately maximum 4.5 kWh/kg COD. Therefore, using the BES reactors as a pre-treatment step in our combined BES–EOP system reduced energy consumption by 81% in comparison with alone EOP. According to Piagrid Energy Tariff 2026 Report, the electricity price for industrial/commercial users in Turkey was assumed to be approximately 0.10 US$/kWh, based on recent market and tariff data, with reported values ranging

between 0.08 and 0.11 US$/kWh [84] . Therefore, the energy consumption cost was calculated using the average electricity price in Turkey, which is 0.10 US$/kWh. In addition, we also evaluated the total cost in terms of U.S. dollars spent per kg COD removed from wastewater (US$/kg COD). Finally, wastewater treatment costs were estimated 2.4 $/kg COD using EOP and 0.45 $/kg COD using combined BES–EOP system.

4. Conclusions

Increasing environmental concerns regarding water pollution, along with greater public awareness of health risks linked to insufficiently treated wastewater, have strengthened the focus on wastewater treatment systems. One of the key challenges today is the simultaneous and efficient removal of COD and nitrogen compounds from real wastewater, while minimizing energy consumption and treatment time. EOPs have demonstrated successful application in wastewater treatment and compliance with effluent discharge limits; however, their primary limitation remains high energy consumption relative to other treatment technologies. Given the limitations of EOPs, significant efforts have been devoted to integrating them with BESs as a novel and enhanced treatment approach. Therefore, the combined BES–EOP system aims to take advantage of coupling biodegradation with a chemical process, reducing operational costs and shortening hydraulic retention time. The main conclusions of this study are as follows:

- The combined BES–EOP system is better at treating pollutants in a more environmental-friendly way than solitary EOP.
- The COD removal efficiency increased when the BES was combined with the EOP system; however, no significant additional COD removal was observed after 6 hours of treatment.
- The use of a combined BES–EOP system increased nitrogen removal efficiency during 6 hours of treatment. In addition, unlike COD removal, extending the treatment time to 10 hours improved NH_3–N removal efficiency, reaching 21%.
- Since the pH increased to the range of 7.4–8.2, HOCl and OCl^- were the dominant chlorine species, which are responsible for indirect oxidation in the bulk liquid.
- The combined BES–EOP system reduced energy consumption (24 to 4.5 kWh/m^3) and treatment cost (2.4 to 0.45 $/kg COD) compared to the standalone EO process.

In this study, it was demonstrated that the combined BES–EOP system enhanced nitrogen and COD removal efficiencies. However, no significant additional COD removal was observed after 6 hours of treatment, which can be considered a limitation of the present work. In this context, further optimization of key operational parameters, such as the number of electrodes and the hydraulic retention time, is recommended. For this purpose, the sequential simplex method could be applied to determine optimal operating conditions by simultaneously varying multiple parameters, including the treatment time of each process. In addition, phosphorus removal and its underlying mechanisms should be investigated in future studies, as the primary goal of wastewater treatment is the simultaneous and efficient removal of both organic matter and nutrients.

Acknowledgements

This research was supported by a grant from Office of Naval Research award number N000141512471 and US National Science Foundation award CBET #1706889. Seçil Tutar Öksüz is grateful for the help of her doctoral advisor, Haluk Beyenal. The author acknowledges the staff at the Moscow Water Reclamation and Reuse Facility (Moscow, ID) for providing wastewater and mixed culture inoculum. Seçil Tutar Öksüz also acknowledges support from Turkish Ministry of National Education Scholarships.

Conflict of Interest Statement

The authors of the article declare that they have no personal or financial conflicts of interest with any institution, organization, or individual.

References

[1] Obaideen K, Shehata N, Sayed ET, Abdelkareem MA, Mahmoud MS, Olabi AG. The role of wastewater treatment in achieving sustainable development goals (SDGs) and sustainability guideline. Energy Nexus 2022;7:100112. https://doi.org/10.1016/J.NEXUS.2022.100112

[2] Tariq A, Mushtaq A. Untreated Wastewater Reasons and Causes: A Review of Most Affected Areas and Cities. IJCBS 2023;23:2023

[3] Silva JA. Wastewater Treatment and Reuse for Sustainable Water Resources Management: A Systematic Literature Review. Sustainability (Switzerland) 2023;15:10940. https://doi.org/10.3390/SU151410940/S1

[4] Chan YJ, Chong MF, Law CL, Hassell DG. A review on anaerobic–aerobic treatment of industrial and municipal wastewater. Chemical Engineering Journal 2009;155:1–18. https://doi.org/10.1016/J.CEJ.2009.06.041

[5] McCarty PL, Bae J, Kim J. Domestic wastewater treatment as a net energy producer-can this be achieved? Environ Sci Technol 2011;45:7100–6. https://doi.org/10.1021/ES2014264

[6] Najafinejad MS, Chianese S, Fenti A, Iovino P, Musmarra D. Application of Electrochemical Oxidation for Water and Wastewater Treatment: An Overview. Molecules 2023, Vol 28, Page 4208 2023;28:4208. https://doi.org/10.3390/MOLECULES28104208

[7] Fu R, Zhang PS, Jiang YX, Sun L, Sun XH. Wastewater treatment by anodic oxidation in electrochemical advanced oxidation process: Advance in mechanism, direct and indirect oxidation detection methods. Chemosphere 2023;311:136993. https://doi.org/10.1016/J.CHEMOSPHERE.2022.136993

[8] Ganzoury MA, Ghasemian S, Zhang N, Yagar M, de Lannoy CF. Mixed metal oxide anodes used for the electrochemical degradation of a real mixed industrial wastewater. Chemosphere 2022;286. https://doi.org/10.1016/j.chemosphere.2021.131600

[9] Jiad MM, Abbar AH. Treatment of petroleum refinery wastewater by electrofenton process using a low cost porous graphite air-diffusion cathode with a novel design. Chemical Engineering Research and Design 2023;193:207–21. https://doi.org/10.1016/J.CHERD.2023.03.021

[10] dos Santos AJ, Shen H, Lanza MRV, Li Q, Garcia-Segura S. Electrochemical oxidation of surfactants as an essential step to enable greywater reuse. Environ Technol Innov 2024;34. https://doi.org/10.1016/j.eti.2024.103563

[11] Wu C, Ge J, Gu F, Bai L. Electrochemical oxidation technique to pharmaceutical pollutants removal. Chemosphere 2023;337. https://doi.org/10.1016/j.chemosphere.2023.139373

[12] Ganthavee V, Trzcinski AP. Removal of pharmaceutically active compounds from wastewater using adsorption coupled with electrochemical oxidation technology: A critical review. Journal of Industrial and Engineering Chemistry 2023;126:20–35. https://doi.org/10.1016/J.JIEC.2023.06.003

[13] Zayas T, Picazo M, and LS-J of WR, 2011 undefined. Removal of organic matter from paper mill effluent by electrochemical oxidation. AcademiaEduT Zayas, M Picazo, L SalgadoJournal of Water Resource and Protection, 2011•academiaEdu n.d

[14] Gümüş D. Electrochemical Treatment of a Real Textile Wastewater Using Cheap Electrodes and Improvement in Cod Removal. Water Air Soil Pollut 2023;234. https://doi.org/10.1007/S11270-023-06313-9

[15] Sundaramoorthy S, Singh N, Taube CR, Katiyar R, Muralidharan V, Palanivel S. Electro-oxidation of tannery wastewater to achieve zero discharge–a step towards sustainability. Environmental Technology (United Kingdom) 2023;44:2995–3003. https://doi.org/10.1080/09593330.2022.2049887

[16] Turan A, Keyikoglu R, Kobya M, Khataee A. Degradation of thiocyanate by electrochemical oxidation process in coke oven wastewater: Role of operative parameters and mechanistic study. Chemosphere 2020;255:127014. https://doi.org/10.1016/J.CHEMOSPHERE.2020.127014

[17] Deng Y, Englehardt JD. Electrochemical oxidation for landfill leachate treatment. Waste Management 2007;27:380–8. https://doi.org/10.1016/J.WASMAN.2006.02.004

[18] Yao J, Mei Y, Jiang J, Xia G, of JC-IJ, 2022 undefined. Process optimization of electrochemical treatment of COD and total nitrogen containing wastewater. MdpiComJ Yao, Y Mei, J Jiang, G Xia, J ChenInternational Journal of Environmental Research and Public Health, 2022•mdpiCom n.d.

[19] Garcia-Segura S, Ocon J, Environmental MC-PS and, 2018 undefined. Electrochemical oxidation remediation of real wastewater effluents—A review. Elsevier n.d.

[20] Song LJ, Zhu NW, Yuan HP, Hong Y, Ding J. Enhancement of waste activated sludge aerobic digestion by electrochemical pre-treatment. Water Res 2010;44:4371–8. https://doi.org/10.1016/J.WATRES.2010.05.052

[21] Garcia-Segura S, Ocon JD, Chong MN. Electrochemical oxidation remediation of real wastewater effluents — A review. Process Safety and Environmental Protection 2018;113:48–67. https://doi.org/10.1016/J.PSEP.2017.09.014

[22] Sinahroy A, Kim SH, Chung CM. Effect of electrooxidation pretreatment on treatment efficiency, membrane fouling and microbial community of a membrane bioreactor treating sludge dewatering wastewater. Process Safety and Environmental Protection 2024;191:466–77. https://doi.org/10.1016/J.PSEP.2024.08.112
[23] Arenas C, González R, González J, … JC-J of environmental, 2021 undefined. Assessment of electrooxidation as pre-and post-treatments for improving anaerobic digestion and stabilisation of waste activated sludge. ElsevierCB Arenas, R González, J González, J Cara, G Papaharalabos, X Gómez, EJ MartínezJournal of Environmental Management, 2021•Elsevier n.d.
[24] Barrios JA, Cano A, Rivera FF, Cisneros ME, Durán U. Efficiency of integrated electrooxidation and anaerobic digestion of waste activated sludge. Biotechnol Biofuels 2021;14. https://doi.org/10.1186/S13068-021-01929-7
[25] Arenas C, González R, González J, … JC-J of environmental, 2021 undefined. Assessment of electrooxidation as pre-and post-treatments for improving anaerobic digestion and stabilisation of waste activated sludge. ElsevierCB Arenas, R González, J González, J Cara, G Papaharalabos, X Gómez, EJ MartínezJournal of Environmental Management, 2021•Elsevier n.d.
[26] Mosquera-Romero S, Ntagia E, Rousseau DPL, Esteve-Núñez A, Prévoteau A. Water treatment and reclamation by implementing electrochemical systems with constructed wetlands. Environmental Science and Ecotechnology 2023;16:100265. https://doi.org/10.1016/J.ESE.2023.100265
[27] Roselló-Márquez G, Fernández-Domene RM, García-Antón J. Organophosphorus pesticides (chlorfenvinphos, phosmet and fenamiphos) photoelectrodegradation by using WO3 nanostructures as photoanode. Journal of Electroanalytical Chemistry 2021;894. https://doi.org/10.1016/j.jelechem.2021.115366
[28] Senthilkumar S, Basha CA, Perumalsamy M, Prabhu HJ. Electrochemical oxidation and aerobic biodegradation with isolated bacterial strains for dye wastewater: Combined and integrated approach. Electrochim Acta 2012;77:171–8. https://doi.org/10.1016/J.ELECTACTA.2012.05.084
[29] Ma J, Wei W, Qin G, Jiang L, Wong NH, Sunarso J, et al. Electrochemical oxidation of phenol in a PtRu/NbC membrane-based catalytic nanoreactor. J Environ Chem Eng 2023;11:111128. https://doi.org/10.1016/J.JECE.2023.111128
[30] Liang Z, Li S, Guo W, Fan C. The Kinetics for Electrochemical Removal of Ammonia in Coking Wastewater. Chin J Chem Eng 2011;19:570–4. https://doi.org/10.1016/S1004-9541(11)60023-2
[31] Das AK, Reza A, Chen L. Optimization of pollutants removal from anaerobically digested dairy wastewater by electro-oxidation process: a response surface methodology modeling and validation. J Appl Electrochem 2024;54:2415–36. https://doi.org/10.1007/S10800-024-02113-Z/FIGURES/5
[32] Tang J, Yao S, Xiao F, Xia J, Xing X. Electrochemical Oxidation of Landfill Leachate after Biological Treatment by Electro-Fenton System with Corroding Electrode of Iron. Int J Environ Res Public Health 2022;19:7745. https://doi.org/10.3390/IJERPH19137745/S1
[33] Katsoni A, Mantzavinos D, Diamadopoulos E. Sequential treatment of diluted olive pomace leachate by digestion in a pilot scale UASB reactor and BDD electrochemical oxidation. Water Res 2014;57:76–86. https://doi.org/10.1016/J.WATRES.2014.03.010
[34] Tutar Oksuz S, Beyenal H. Enhanced bioelectrochemical nitrogen removal in flow through electrodes. Sustainable Energy Technologies and Assessments 2021;47. https://doi.org/10.1016/j.seta.2021.101507
[35] Naseer MN, Zaidi AA, Khan H, Kumar S, Owais MT bin, Jaafar J, et al. Mapping the field of microbial fuel cell: A quantitative literature review (1970–2020). Energy Reports 2021;7:4126–38. https://doi.org/10.1016/J.EGYR.2021.06.082
[36] Sun F, Chen J, Sun Z, Zheng X, Tang M, Yang Y. Promoting bioremediation of brewery wastewater, production of bioelectricity and microbial community shift by sludge microbial fuel cells using biochar as anode. Science of The Total Environment 2024;929:172418. https://doi.org/10.1016/J.SCITOTENV.2024.172418
[37] Ren B, Wang T, Zhao Y. Two-stage hybrid constructed wetland-microbial fuel cells for swine wastewater treatment and bioenergy generation. Chemosphere 2021;268:128803. https://doi.org/10.1016/J.CHEMOSPHERE.2020.128803
[38] Elakkiya E, Niju S. Application of Microbial Fuel Cells for Treatment of Paper and Pulp Industry Wastewater: Opportunities and Challenges 2020:125–49. https://doi.org/10.1007/978-3-030-38196-7_6
[39] Kumar M, Pandit S, Patel V, Khanna N, Nag M, Lahiri D, et al. Maximization of Energy Recovery from Starch Processing Wastewater by Thermophilic Dark Fermentation Coupled with Microbial fuel Cell Technology. Geomicrobiol J 2024;41:392–403. https://doi.org/10.1080/01490451.2023.2209555
[40] Abu-Reesh IM, Kunju A, Sevda S. Performance of microbial fuel cells in treating petroleum refinery wastewater. Journal of Water Process Engineering 2022;49. https://doi.org/10.1016/j.jwpe.2022.103029
[41] Thapa B Sen, Pandit S, Patwardhan SB, Tripathi S, Mathuriya AS, Gupta PK, et al. Application of Microbial Fuel Cell (MFC) for Pharmaceutical Wastewater Treatment: An Overview and Future Perspectives. Sustainability 2022, Vol 14, Page 8379 2022;14:8379. https://doi.org/10.3390/SU14148379

[42] Ishaq A, Said MIM, Azman SB, Dandajeh AA, Lemar GS, Jagun ZT. Utilization of microbial fuel cells as a dual approach for landfill leachate treatment and power production: a review. Environmental Science and Pollution Research 2023 31:29 2023;31:41683–733. https://doi.org/10.1007/S11356-023-30841-W
[43] Roy H, Rahman TU, Tasnim N, Arju J, Rafid MM, Islam MR, et al. Microbial Fuel Cell Construction Features and Application for Sustainable Wastewater Treatment. Membranes (Basel) 2023;13:490. https://doi.org/10.3390/MEMBRANES13050490/S1
[44] Jain M, Sai Kiran P, Ghosal PS, Gupta AK. Development of microbial fuel cell integrated constructed wetland (CMFC) for removal of paracetamol and diclofenac in hospital wastewater. J Environ Manage 2023;344:118686. https://doi.org/10.1016/J.JENVMAN.2023.118686
[45] Wang W, Lu Y, Luo H, Liu G, Zhang R, Jin S. A microbial electro-fenton cell for removing carbamazepine in wastewater with electricity output. Water Res 2018;139:58–65. https://doi.org/10.1016/J.WATRES.2018.03.066
[46] Li S, Liu Y, Ge R, Yang S, Zhai Y, Hua T, et al. Microbial electro-Fenton: A promising system for antibiotics resistance genes degradation and energy generation. Sci Total Environ 2020;699. https://doi.org/10.1016/J.SCITOTENV.2019.134160
[47] Mkilima T, Saspugayeva G, Dakieva K, Tussupova Z, Zhaken A, Kumarbekuly S, et al. Enhancing slaughterhouse wastewater treatment through the integration of microbial fuel cell and Electro-Fenton systems: A comprehensive comparative analysis. Journal of Water Process Engineering 2024;57:104743. https://doi.org/10.1016/J.JWPE.2023.104743
[48] Ahmad A, Kumar P, Kanwar B, Singh SP. Microbial fuel cell powered Fenton degradation of ciprofloxacin from wastewater employing MIL-88B(Fe)-MOF-laser-induced graphene cathode. Environ Res 2026;293:123807. https://doi.org/10.1016/J.ENVRES.2026.123807
[49] Ahmad I, Basu D. Electrochemical Treatment of Reactive Orange 16 Dye Pollutant Using Microbial Fuel Cell as Renewable Power Source. Environ Eng Sci 2024;41:29–38. https://doi.org/10.1089/EES.2023.0136
[50] Huang L, Yang X, Quan X, Chen J, Yang F. A microbial fuel cell–electro-oxidation system for coking wastewater treatment and bioelectricity generation. Journal of Chemical Technology & Biotechnology 2010;85:621–7. https://doi.org/10.1002/JCTB.2320
[51] Zou H, Wang Y. Azo dyes wastewater treatment and simultaneous electricity generation in a novel process of electrolysis cell combined with microbial fuel cell. Bioresour Technol 2017;235:167–75. https://doi.org/10.1016/J.BIORTECH.2017.03.093
[52] Mohanakrishna G, Al-Raoush RI, Abu-Reesh IM. Integrating electrochemical and bioelectrochemical systems for energetically sustainable treatment of produced water. Fuel 2021;285:119104. https://doi.org/10.1016/J.FUEL.2020.119104
[53] Chen S, Wang X, Shi X, Li S, Yang L, Yan W, et al. Integrated system of electro-catalytic oxidation and microbial fuel cells for the treatment of recalcitrant wastewater. Chemosphere 2024;354:141754. https://doi.org/10.1016/J.CHEMOSPHERE.2024.141754
[54] Lewandowski Z, Beyenal H. Fundamentals of biofilm research. Fundamentals of Biofilm Research. CRC Press. 2007. https://doi.org/10.1201/b16291
[55] Renslow R, Donovan C, Shim M, Babauta J, Nannapaneni S, Schenk J, et al. Oxygen reduction kinetics on graphite cathodes in sediment microbial fuel cells. Physical Chemistry Chemical Physics 2011;13:21573–84. https://doi.org/10.1039/C1CP23200B
[56] Torres CI, Krajmalnik-Brown R, Parameswaran P, Marcus AK, Wanger G, Gorby YA, et al. Selecting anode-respiring bacteria based on anode potential: Phylogenetic, electrochemical, and microscopic characterization. Environ Sci Technol 2009;43:9519–24. https://doi.org/10.1021/ES902165Y
[57] Jang GG, Castillo C, Dutta S, Su YF, Jun J, Keum JK, et al. Performance Restoration of Iron Electrodes by Polarity Reversal after Long-Term Surface Fouling in Wastewater Electrocoagulation. ACS ES and T Water 2024;4:2390–402. https://doi.org/10.1021/ACSESTWATER.3C00741
[58] Mirshafiee A, Nourollahi M, Shahriary A. Application of electro oxidation process for treating wastewater from petrochemical with mixed metal oxide electrode. Scientific Reports 2024 14:1 2024;14:1760-. https://doi.org/10.1038/s41598-024-52201-5
[59] Díaz V, Ibáñez R, Gómez P, Urtiaga AM, Ortiz I. Kinetics of electro-oxidation of ammonia-N, nitrites and COD from a recirculating aquaculture saline water system using BDD anodes. Water Res 2011;45:125–34. https://doi.org/10.1016/J.WATRES.2010.08.020
[60] Vijayaraghavan K, Ramanujam TK, Balasubramanian N. In situ hypochlorous acid generation for the treatment of distillery spentwash. Ind Eng Chem Res 1999;38:2264–7. https://doi.org/10.1021/IE980166X
[61] Ghimire U, Jang M, Jung SP, Park D, Park SJ, Yu H, et al. Electrochemical Removal of Ammonium Nitrogen and COD of Domestic Wastewater using Platinum Coated Titanium as an Anode Electrode. Energies (Basel) 2019;12:1–13.

[62] Włodarczyk-Makuła M, Myszograj S, Włodarczyk M. Removal of Organic Micro-Pollutants from Wastewater in Electrochemical Processes—Review. Energies (Basel) 2023;16:1–19.
[63] Martínez-Huitle CA, Panizza M. Electrochemical oxidation of organic pollutants for wastewater treatment. Curr Opin Electrochem 2018;11:62–71. https://doi.org/10.1016/J.COELEC.2018.07.010
[64] Mohan N, Balasubramanian N, Basha CA. Electrochemical oxidation of textile wastewater and its reuse. J Hazard Mater 2007;147:644–51. https://doi.org/10.1016/J.JHAZMAT.2007.01.063
[65] Panizza M, Cerisola G. Direct and mediated anodic oxidation of organic pollutants. Chem Rev 2009;109:6541–69. https://doi.org/10.1021/CR9001319
[66] Ganiyu SO, Martínez-Huitle CA, Oturan MA. Electrochemical advanced oxidation processes for wastewater treatment: Advances in formation and detection of reactive species and mechanisms. Curr Opin Electrochem 2021;27:27. https://doi.org/10.1016/J.COELEC.2020.100678
[67] Gasser G, Rona M, Voloshenko A, Shelkov R, Tal N, Pankratov I, et al. Quantitative evaluation of tracers for quantification of wastewater contamination of potable water sources. Environ Sci Technol 2010;44:3919–25. https://doi.org/10.1021/ES100604C
[68] An J, Feng Y, Zhao Q, Wang X, Liu J, Li N. Electrosynthesis of H2O2 through a two-electron oxygen reduction reaction by carbon based catalysts: From mechanism, catalyst design to electrode fabrication. Environmental Science and Ecotechnology 2022;11:100170. https://doi.org/10.1016/J.ESE.2022.100170
[69] An J, Gao Y, Lee HS. Induction of cathodic voltage reversal and hydrogen peroxide synthesis in a serially stacked microbial fuel cell. J Environ Manage 2019;241:84–90. https://doi.org/10.1016/J.JENVMAN.2019.04.016
[70] Lu Y, Hu Y, Tang L, Xie Q, Liu Q, Zhong L, et al. Effects and mechanisms of modified biochars on microbial iron reduction of Geobacter sulfurreducens. Chemosphere 2021;283:130983. https://doi.org/10.1016/J.CHEMOSPHERE.2021.130983
[71] Imlay JA. Cellular defenses against superoxide and hydrogen peroxide. Annu Rev Biochem 2008;77:755–76. https://doi.org/10.1146/ANNUREV.BIOCHEM.77.061606.161055
[72] Lin WC, Coppi M V., Lovley DR. Geobacter sulfurreducens Can Grow with Oxygen as a Terminal Electron Acceptor. Appl Environ Microbiol 2004;70:2525. https://doi.org/10.1128/AEM.70.4.2525-2528.2004
[73] Kapałka A, Cally A, Neodo S, Comninellis C, Wächter M, Udert KM. Electrochemical behavior of ammonia at Ni/Ni(OH)2 electrode. Electrochem Commun 2010;12:18–21. https://doi.org/10.1016/J.ELECOM.2009.10.026
[74] Algburi S, Munther H, Al-Dulaimi O, Fakhruldeen HF, Sapaev IB, Seedi KFK Al, et al. Green Hydrogen Role in Sustainable Energy Transformations: A Review. Results in Engineering 2025;26:105109. https://doi.org/10.1016/J.RINENG.2025.105109
[75] Adnan RH, Ng YH. Clean production of chlorine (Cl2) and hypochlorous acid (HOCl) from photocatalytic and photoelectrochemical seawater splitting. Mater Horiz 2026. https://doi.org/10.1039/D5MH01556A
[76] Kapałka A, Katsaounis A, Michels NL, Leonidova A, Souentie S, Comninellis C, et al. Ammonia oxidation to nitrogen mediated by electrogenerated active chlorine on Ti/PtOx-IrO2. Electrochem Commun 2010;12:1203–5. https://doi.org/10.1016/J.ELECOM.2010.06.019
[77] Wen LL, Zhang Y, Chen JX, Zhang ZX, Yi YY, Tang Y, et al. The dechlorination of TCE by a perchlorate reducing consortium. Chemical Engineering Journal 2017;313:1215–21. https://doi.org/10.1016/J.CEJ.2016.11.021
[78] Yang C, Liu T, Chen N, Tong S, Deng Y, Xue L, et al. Performance and mechanism of a novel woodchip embedded biofilm electrochemical reactor (WBER) for nitrate-contaminated wastewater treatment. Chemosphere 2021;276:130250. https://doi.org/10.1016/J.CHEMOSPHERE.2021.130250
[79] Gonzalez-Martinez A, Rodriguez-Sanchez A, Lotti T, Garcia-Ruiz MJ, Osorio F, Gonzalez-Lopez J, et al. Comparison of bacterial communities of conventional and A-stage activated sludge systems. Scientific Reports 2016 6:1 2016;6:18786-. https://doi.org/10.1038/srep18786
[80] Xing X, Ni J, Zhu X, Jiang Y, Xia J. Maximization of current efficiency for organic pollutants oxidation at BDD, Ti/SnO_2-Sb/PbO_2, and Ti/SnO_2-Sb anodes. Chemosphere 2018;205:361–8. https://doi.org/10.1016/J.CHEMOSPHERE.2018.04.090
[81] Luu T Le. Post treatment of ICEAS-biologically landfill leachate using electrochemical oxidation with Ti/BDD and Ti/RuO2 anodes. Environ Technol Innov 2020;20:101099. https://doi.org/10.1016/J.ETI.2020.101099
[82] Metcalf & Eddy Inc, Tchobanoglous G, Burton FL, Tsuchihashi R, Stensel HD. Wastewater Emgomeeromg: Treatment and Resource Recovery. Wastewater Engineering 2013:2048
[83] Zhang F, Ge Z, Grimaud J, Hurst J, He Z. In situ investigation of tubular microbial fuel cells deployed in an aeration tank at a municipal wastewater treatment plant. Bioresour Technol 2013;136:316–21. https://doi.org/10.1016/J.BIORTECH.2013.02.107
[84] EPDK | Enerji Piyasası Düzenleme Kurumu n.d. https://www.epdk.gov.tr/Detay/Icerik/3-0-1/tarifeler (accessed May 20, 2026)