Electrochemical Oxidation for the Removal of Pharmaceutically Active Compounds from Wastewater: Current Issues, Environmental Risks, and Future Perspectives

Vivek Sharma[a], Ritesh Patidar[b]*

[a] Department of Mechanical Engineering, M.L.V. Textile & Engineering College, Rajasthan Technical University, Bhilwara, Rajasthan 311001, India

[b] Department of Chemical Engineering, Rajasthan Technical University, Kota, Rajasthan 324010, India

*Corresponding author: riteshpatidar@gmail.com

Abstract

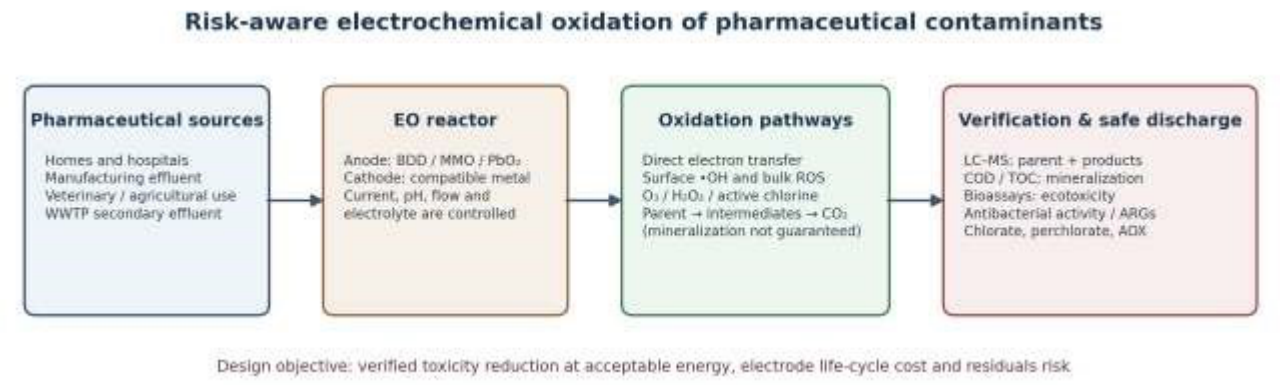
Pharmaceutically active compounds (PhACs) are continuously released to wastewater from domestic use, hospitals, pharmaceutical manufacturing and veterinary applications. Conventional biological treatment removes some compounds but leaves persistent parent molecules and transformation products in treated effluents. Electrochemical oxidation (EO) is attractive because oxidants can be generated in situ and operating conditions can be adjusted through electrode selection, current density, pH and electrolyte composition. Its performance, however, cannot be judged from parent-compound disappearance alone. This review critically examines EO of PhACs through five connected lenses: India-specific occurrence and treatment needs; comparison of experimental conditions and wastewater matrices; electrode-dependent oxidation and by-product formation; energy and process-scale considerations; and ecotoxicity and antimicrobial-resistance endpoints. Published studies show high parent removal for selected compounds under optimized laboratory conditions, but the extent of mineralization is often lower or not reported. Chloride-containing electrolytes can accelerate oxidation while also producing halogenated transformation products, chlorate and perchlorate. Antibiotic oxidation may reduce residual antibacterial activity without necessarily eliminating toxicity or antibiotic-resistance genes. Evidence from real hospital wastewater confirms that performance in complex matrices can differ from synthetic solutions. The review therefore distinguishes measured findings from transferable evidence and identifies a need for standardized energy reporting, longduration continuous-flow trials, transformation-product monitoring and ecotoxicological validation before EO is considered a sustainable full-scale option for pharmaceutical wastewater treatment.

Keywords: electrochemical oxidation; pharmaceutical residues; wastewater; boron-doped diamond; mixed-metal-oxide anodes; transformation products; mineralization; total organic carbon; specific energy

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consumption; electrical energy per order; hospital wastewater; antibiotic resistance; ecotoxicity; chlorate; perchlorate; hybrid treatment.

Graphical abstract



Graphical abstract. Pharmaceutical inputs, electrochemical oxidation pathways and the analytical endpoints required to demonstrate environmentally safe treatment.

1. Introduction

Pharmaceutically active compounds (PhACs) are contaminants of emerging concern because their continuous release can sustain exposure in receiving waters even when individual compounds are present at low concentrations. Domestic wastewater, hospital discharges, pharmaceutical manufacturing, veterinary use and improper disposal all contribute to the load. The fraction of a dose excreted unchanged is highly compound-dependent; published values span less than 10% to more than 90%, and a single generalized range should not be applied to all pharmaceuticals (Daughton & Ternes, 1999; Kümmerer, 2009). Conventional wastewater treatment plants (WWTPs) are primarily designed for bulk organic matter, nutrients and suspended solids, rather than trace micropollutants. Consequently, the removal of individual PhACs varies with biodegradability, sorption, operating conditions and the composition of the wastewater matrix (Bampos et al., 2021; Moreira et al., 2017). The issue has particular relevance in India, where monitoring evidence is geographically uneven but indicates pharmaceutical residues in wastewater-treatment-plant effluents, hospital discharges and selected rivers. A review of Indian water bodies found that hospital effluents generally contained higher pharmaceutical concentrations than domestic wastewater and that only limited groundwater data were available at the time of review (Balakrishna et al., 2017). Selected reported Indian WWTP-effluent values included ciprofloxacin at 17.7 $\mu g\ L^{-1}$, metoprolol at 4.0 $\mu g\ L^{-1}$ and ofloxacin at 2.47 $\mu g\ L^{-1}$; these are reported sample values, not national mean concentrations (Balakrishna et al., 2017; IIT Madras SAICM report, 2019). This evidence supports

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targeted monitoring and source-specific treatment, not an assumption that one treatment technology is suitable for all Indian wastewater streams.

Electrochemical oxidation (EO), including anodic oxidation and related electrochemical advanced oxidation processes (EAOPs), uses electrical current to drive direct electron transfer and/or generate reactive oxidants at the electrode surface. Boron-doped diamond (BDD), lead dioxide (PbO_2), tin oxide (SnO_2), mixed-metal oxides (MMOs) and modified composite electrodes have all been investigated. The performance of each is conditioned by its oxygen-evolution overpotential, surface chemistry, stability and tendency to generate secondary oxidants. In chloride-containing water, active chlorine can increase apparent removal but can also produce halogenated products and inorganic oxychlorine species (García-Espinoza & Mijaylova Nacheva, 2019; Zöllig et al., 2015).

Previous reviews have summarized pharmaceutical EO studies and broader electrochemical treatment of synthetic and real wastewaters (Bampos et al., 2021; Moreira et al., 2017). A recent broad review of electrocatalytic wastewater oxidation further emphasizes inconsistent reporting of electrode area, potential, current density, current efficiency, energy per volume or mass removed, and time on stream (Begildayeva et al., 2026). The present review does not claim to be the first review of EO. Its contribution is a focused, risk-aware synthesis that connects (i) reported pharmaceutical occurrence in India, (ii) quantitative PhAC studies and their operating conditions, (iii) real wastewater matrix effects, (iv) parent-compound removal versus mineralization and transformation-product toxicity, including antimicrobial-resistance endpoints, and (v) energy and technology-readiness comparisons with ozonation, activated carbon and UV/H_2O_2. These themes are carried through the discussion and conclusions to identify where EO may be competitive and which evidence is still missing.

2. Review methodology

A structured narrative literature search was conducted and updated. Searches covered PubMed/Medline and PubMed Central for indexed biomedical and environmental studies; publisher platforms including ScienceDirect, SpringerLink, ACS Publications, MDPI and Nature Portfolio for full-text article records; and DOI/Crossref metadata to verify publication identity, year, journal, volume, article number and DOI. The search was supplemented by backward citation screening of relevant reviews and primary studies. Scopus and Web of Science were not used as exhaustive primary screening databases in this revision; therefore, this article is described as a structured narrative review rather than a PRISMA systematic review, and no comprehensive database record count is claimed.

The principal search strings combined the following terms: ―electrochemical oxidation‖ OR ―anodic oxidation‖ OR ―electrochemical advanced oxidation‖ OR ―electro-Fenton‖ with ―pharmaceutical‖ OR

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—pharmaceutical active compounds‖ OR —PhAC‖ OR —antibiotic‖ OR —diclofenac‖ OR —ibuprofen‖ OR —carbamazepine‖ OR —sulfamethoxazole‖ OR —cephalexin‖ OR —oxytetracycline‖. Additional searches combined EO terms with —transformation products‖, —TOC‖, —COD‖, —mineralization‖, —electrical energy per order‖, —specific energy consumption‖, —antibacterial activity‖, —antibiotic resistance genes‖, —hospital wastewater‖, —real wastewater‖, —pilot plant‖, —flow reactor‖, —chlorate‖ and —perchlorate‖. Searches for Indian context combined —pharmaceuticals‖, —India‖, —wastewater‖, —hospital effluent‖, —river‖ and —WWTP‖. The principal publication window was 2000–6 November 2026, with earlier seminal electrochemical and contaminant-fate studies retained where needed to explain mechanisms or environmental risk.

Studies were included when they reported (a) EO/EAOP treatment of one or more PhACs, (b) electrochemical transformation of antibiotics or antibiotic-resistance endpoints, (c) real wastewater treatment or pilot/flow-reactor evidence relevant to the intended application, or (d) validated comparative data for a competing technology or an electrode/by-product mechanism. Peer-reviewed original studies, review articles used for context, and technical reports with identifiable methods and quantitative metrics were eligible. Studies dealing only with adsorption, biological treatment, photocatalysis or ozone without a relevant EO comparison were excluded from the EO performance table, although they could be cited in the technology comparison. Duplicates, bibliographic entries without a traceable publication record, and qualitative claims lacking a verifiable source were excluded. Titles and abstracts were screened first; records retained as potentially relevant were checked at fulltext or publisher-record level. For each treatment study, available data were extracted for matrix type, initial PhAC concentration, anode/cathode, current density or applied potential, electrolyte and concentration, pH, treatment time, parent-compound removal, COD/TOC and energy metrics, and reported transformation products/toxicity. —NR‖ in tables means that the cited source did not report the item in the accessible publication record or the value was not recoverable with confidence.

Because the studies differ substantially in target compounds, concentration, reactor geometry, current density, electrolyte, wastewater matrix and analytical endpoint, a pooled quantitative metaanalysis was not appropriate. The synthesis therefore compares studies conditionally and avoids treating percentages obtained under unlike conditions as directly comparable. Parent disappearance, degradation, mineralization and detoxification are treated as separate endpoints.

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3. Pharmaceutical contaminants in wastewater: occurrence and treatment context

3.1 Global occurrence and Indian evidence

PhACs are detected in municipal influents and effluents, hospital wastewater, surface water and groundwater. The reported concentrations are commonly in the ng L^{-1} to µg L^{-1} range for municipal effluents, whereas hospital or manufacturing streams can contain higher and more variable concentrations. The number of measured compounds and the sampling frequency differ widely among regions; apparent differences between countries therefore reflect both environmental conditions and monitoring coverage (Balakrishna et al., 2017; Kümmerer, 2009).

Figure 1 presents three selected pharmaceutical concentrations reported for Indian WWTP effluent: ciprofloxacin (17.7 µg L^{-1}), metoprolol (4.0 µg L^{-1}) and ofloxacin (2.47 µg L^{-1}). The values are reproduced from a comparison compiled from Indian occurrence literature (Balakrishna et al., 2017; IIT Madras SAICM report, 2019). They should not be interpreted as current nationwide averages, as they originate from selected monitoring reports and do not represent every Indian WWTP. The Indian evidence base also highlights comparatively high residues in hospital effluent and limited groundwater monitoring, supporting source-specific surveillance and improved characterization of hospital and pharmaceutical-industry discharges.

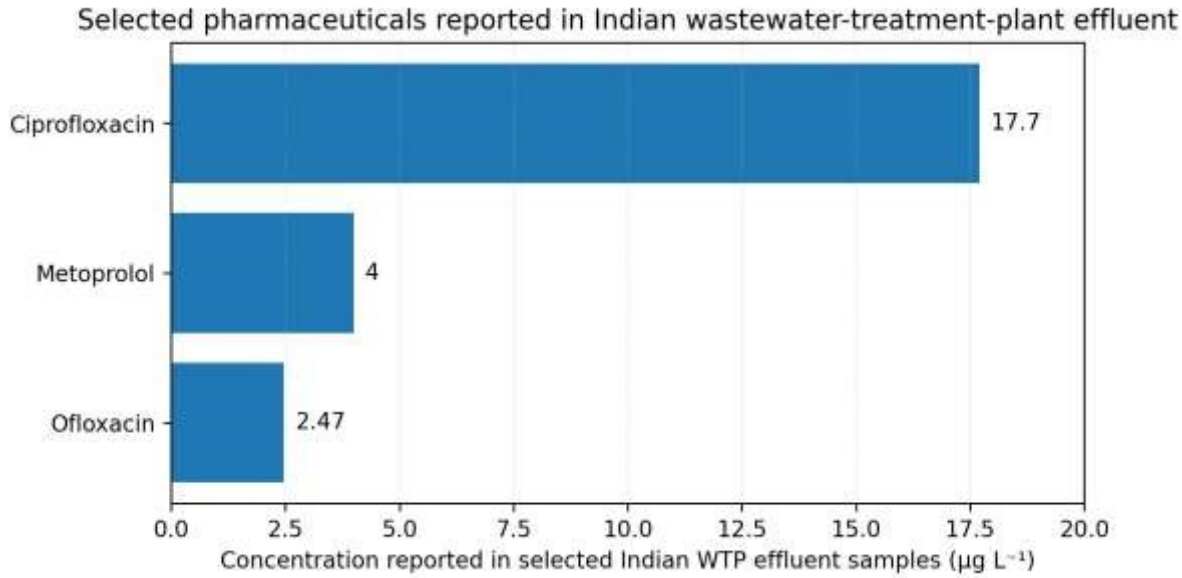


Figure 1. Selected pharmaceutical concentrations reported in Indian WWTP effluents. Values are ciprofloxacin 17.7 µg L^{-1}, metoprolol 4.0 µg L^{-1} and ofloxacin 2.47 µg L^{-1}; they are individual reported values, not national mean concentrations (Balakrishna et al., 2017; IIT Madras SAICM report, 2019).

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3.2 Synthetic solutions and real wastewater matrices

A synthetic single-compound solution provides controlled conditions for identifying reaction kinetics and operating-variable effects, but it generally under-represents the oxidant demand of real wastewater. Dissolved organic matter competes for hydroxyl radicals and other oxidants; bicarbonate and carbonate can convert highly reactive hydroxyl radicals to less reactive carbonate radicals; chloride can generate active chlorine; and suspended solids may shield pollutants, obstruct mass transfer or foul electrode surfaces. Sulfate can support conductivity and, under appropriate conditions, the formation of sulfatederived oxidants. These matrix effects mean that removal observed in deionized water cannot be assumed for secondary effluent, hospital wastewater or pharmaceutical manufacturing effluent (Moreira et al., 2017; Cibati et al., 2022).

An example of this matrix dependence is provided by UV/H_2O_2 treatment of real wastewater, a useful comparator rather than an EO study. Cibati et al. (2022) measured up to 30 pharmaceuticals and 13 transformation products in two urban secondary effluents, greywater, hospital wastewater and pharmaceutical-industry effluent. Overall pharmaceutical removal was 69–86% in urban secondary effluents, 59% in greywater, 36% in hospital wastewater and 17% in the industrial effluent. The authors attributed the reduction to matrix complexity, lower UV fluence and radical scavenging. These results illustrate why EO claims based solely on synthetic solutions require validation in the target wastewater matrix.

Direct pharmaceutical EO evidence in real wastewater remains more limited. Ouarda et al. (2019) applied anodic oxidation to real hospital wastewater and monitored 12 pharmaceutical residues using high-resolution mass spectrometry together with UV absorbance and fluorescence surrogates. A BDDbased anode configuration was operated at 4.42, 17.7 and 35.4 mA cm^{-2} for 120 min; reported abatement of targeted residues reached up to 100% at the highest current density. The study is important because it uses real wastewater and multiple analytes, but it is a batch/experimental demonstration rather than evidence of long-term full-scale operation. The accessible study record does not provide one comparable initial concentration for all target residues; therefore, no single initial concentration is assigned in Table 1.

3.3 Distinguishing removal, degradation, mineralization and detoxification

Four endpoints must not be conflated. (1) Parent-compound removal is the decrease in concentration of the original pharmaceutical, usually measured by HPLC or LC–MS. (2) Degradation or transformation indicates conversion of the parent molecule to one or more products; the products may remain organic

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and biologically active. (3) Mineralization is the conversion of organic carbon toward CO_2 and inorganic end products and is assessed using TOC or, less specifically, COD. (4) Detoxification requires evidence that biological or ecological effects decrease, for example through a suitable algal, bacterial, invertebrate or fish bioassay. A 99% decrease in the parent compound therefore does not establish 99% TOC removal or complete detoxification.

COD indicates the oxygen equivalent of oxidizable matter and can be influenced by inorganic reducing species; TOC measures organic carbon and is a more direct indicator of carbon mineralization. Mineralization current efficiency (MCE) estimates the fraction of charge theoretically used to oxidize organic carbon. Transformation-product identification by LC–HRMS or GC–MS, coupled with bioassays, is needed to determine whether oxidation has generated persistent, toxic or antimicrobial intermediates. The reported removal percentages in Table 1 are explicitly labelled as parent removal, COD removal or TOC removal to prevent false equivalence.

4. Electrochemical oxidation mechanisms relevant to PhACs

4.1 Direct anodic oxidation and active/non-active anodes

In direct anodic oxidation, a contaminant transfers electrons at or near the anode surface. The efficiency depends on adsorption, electron-transfer kinetics, electrode composition, mass transport and competing reactions. The commonly used active/non-active anode framework distinguishes the bonding character and oxidizing strength of surface oxygen species. Active anodes, including Pt and RuO_2/IrO_2-based surfaces, tend to form more strongly chemisorbed oxygen species and can favour selective or partial oxidation. Non-active anodes, especially BDD and some SnO_2 or PbO_2 materials, have higher oxygenevolution overpotentials and can support weakly adsorbed hydroxyl radicals, which may promote less selective oxidation and deeper mineralization (Panizza & Cerisola, 2009; Moreira et al., 2017). This is a mechanistic classification, not a guarantee that every compound will mineralize completely: applied current, substrate composition, electrolyte and wastewater matrix remain decisive.

The formation of surface hydroxyl radicals is commonly represented as: $M + H_2O \rightarrow M(\bullet OH) + H^+ + e^-$, where M denotes an anode surface site. A balanced generic anodic mineralization equation for an organic compound C_aH_bO is: $C_aH_bO + (2a - c)H_2O \rightarrow aCO_2 + (4a + b - 2c)H^+ + (4a + b - 2c)e^-$. The coefficient of electrons and protons is therefore balanced; the expression is a general completeoxidation stoichiometry and does not imply that each PhAC follows a single elementary reaction.

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4.2 Indirect oxidation and electro-generated oxidants

Indirect EO occurs when species generated electrochemically react with PhACs in the bulk solution. Depending on the cell and electrolyte, these species can include hydroxyl radicals (•OH), ozone (O_3), hydrogen peroxide (H_2O_2), sulfate-derived oxidants and reactive chlorine species. At the cathode, oxygen reduction can generate H_2O_2; in an electro-Fenton system, Fe^{2+} activates H_2O_2 to produce additional •OH. Active chlorine is formed when chloride is oxidized: $2Cl^- \rightarrow Cl_2 + 2e^-$, followed by $Cl_2 + H_2O \rightleftharpoons HOCl + H^+ + Cl^-$. HOCl/$OCl^-$ speciation is pH-dependent. Such mediated oxidation can improve apparent removal, but halogenation and subsequent formation of chlorinated products must be considered before chloride is selected as a supporting electrolyte (García-Espinoza & Mijaylova Nacheva, 2019; Zöllig et al., 2015).

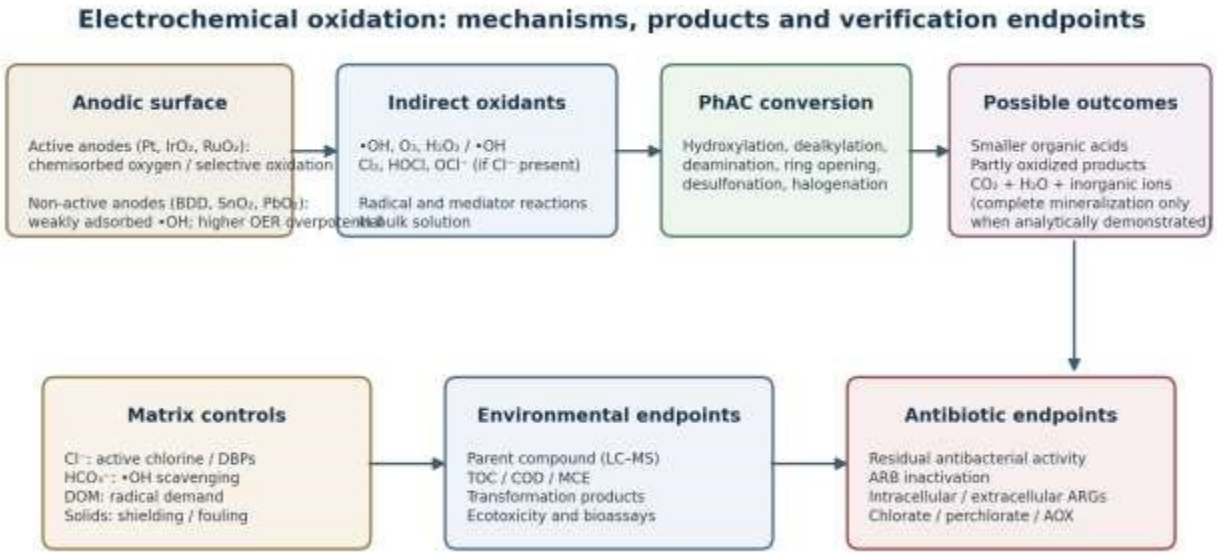


Figure 2. Conceptual schematic of EO mechanisms, matrix controls, transformation pathways and the analytical endpoints required to evaluate treatment safety. Prepared by the authors from the mechanistic and by-product literature (Panizza & Cerisola, 2009; García-Espinoza & Mijaylova Nacheva, 2019; Zöllig et al., 2015; Zhu et al., 2016).

5. Pharmaceutical EO performance, electrode effects and environmental risks

5.1 Quantitative comparison of pharmaceutical studies

In the oxytetracycline study discussed below, OTC denotes oxytetracycline and ECO denotes electrocatalytic oxidation, following the terminology used by Osman and Uğurlu (2025). The published data show that electrode material alone does not determine performance. Initial concentration influences the pollutant-to-oxidant ratio and can alter apparent percentage removal; current density affects the rate

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of oxidant production but can become inefficient when it exceeds the mass-transfer or limiting-current regime; electrolyte concentration affects conductivity and mediator production; and pH changes pharmaceutical ionization, radical pathways and chlorine speciation. Treatment duration also matters: a short experiment can show rapid parent disappearance while longer treatment is required for TOC reduction. A meaningful comparison must therefore report the full set of conditions rather than rank studies by percentage removal alone.

Table 1 compiles traceable studies for which experimental details could be verified. ―NR‖ means not reported in the source record used for extraction. The table preserves distinct analytical endpoints. For example, the paracetamol study reported >90% parent removal but only >65% TOC removal; the ibuprofen study reported COD and TOC ranges rather than a directly comparable parent-removal percentage. The cephalexin study reported a maximum COD removal under conditions distinct from the condition for maximum parent removal. These differences are retained instead of being combined into one efficiency value.

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236 Table 1. Experimental conditions and reported outcomes for selected pharmaceutical electrochemical oxidation studies. NR = not reported; model = 237 synthetic/simulated solution; real = actual wastewater matrix.

Study / matrix	**Target; initial concentration**	**Anode / cathode**	**Current density or potential**	**Electrolyte; pH**	**Duration / reactor**	**Parentcompound removal**	**Mineralization, products or energy**
Periyasamy &Muthukumar (2018); model water	Paracetamol; 20 mg L^{-1}	Graphite anode; Ptwire counter electrode in CV tests	5.1 mA cm^{-2} (optimized); tested 3.1–7.1 mA cm^{-2}	0.1 M Na_2SO_4; pH 4	240 min; batch aqueous electrolysis	>90%	COD >82%; TOC >65%; hydroquinone, benzoquinone and carboxylic acids identified by HPLC; energy not reported

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Ciríaco et al. (2009); model solution	Ibuprofen; 0.22–1.75 mM for Ti/Pt/PbO_2; 1.75 mM for BDD	Ti/Pt/PbO_2 and Si/BDD anodes; cathode NR	10, 20 and 30 mA cm^{-2}	0.035 M Na_2SO_4; pH NR	6 h; batch cell	Absorbance monitored; no single parentremoval percentage reported in abstract	COD 60–95%; TOC 48–92%; BDD gave higher values; general and mineralization current efficiencies measured; energy per m^3 NR
Domínguez et al. (2010); model solution	Carbamazepine; initial concentration NR in accessible abstract	BDD anode; cathode NR	190 mA cm^{-2} at optimum	0.48 M Na_2SO_4; pH 9	Flow rate 1.25 cm^3 min^{-1}; treatment duration NR	100% parent degradation at reported optimum	TOC/COD and energy values not available in abstract; very high current density and sulfate dose limit direct comparison

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Osman & Uğurlu (2025), EO; simulated OTC effluent	Oxytetracycline (OTC); 36 mg L^{-1} at EO optimum	Graphite anode and cathode	15 V; current density NR	0.5 g L^{-1} salt (NaCl); pH 7	45 min; 200 mL stirred cell, 130 rpm, 21 °C	79%	HRMS used to identify intermediates; COD, TOC and normalized energy not reported
Osman & Uğurlu (2025), ECO; simulated OTC effluent	OTC; 30 mg L^{-1} at ECO optimum	Graphite electrodes plus TiO_2/AC particle electrode	10 V; current density NR; catalyst 0.25 g L^{-1}	0.5 g L^{-1} NaCl; pH 7	30 min; threephase / 3D system	86%	HRMS used to identify intermediates; COD, TOC and normalized energy not reported

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Hadavand et al. (2025); model solution	Cephalexin; 50 mg L^{-1} for reported optimum (50, 100 and 300 mg L^{-1} tested)	SDS-modified Ti/TiO_2/β-PbO_2 anode; stainlesssteel cathode	20 mA cm^{-2} (10–30 tested)	0.20 M phosphate buffer at pH 7.01 for maximum parent removal; 0.20 M phosphate pH 3 and 0.20 M acetate pH 5 also tested	20 min for 97.60% parent removal; 30 min for separate COD optimum; undivided cell, 20 cm^2 anode	97.60% by LC–MS at pH 7.01	Maximum COD removal 70.27% under separate pH 3 optimum; LC–MS/MS transformation products; author-reported energy $E_c = 0.23 \times 10^{-3}$ kWh mg^{-1} (not directly comparable with volumetric SEC)
Chen et al. (2025); model aqueous solution	Ofloxacin; initial concentration NR in abstract	IrO_2–RuO_2–TiO_2/Ti anode; Ti cathode	3.9 mA cm^{-2}	5.65 mM electrolyte (salt identity NR in abstract); pH 7.12	3.7 min at model optimum	99.011%	Four degradation pathways proposed from GC–MS and DFT; hydroxyl and reactive chlorine radicals implicated; TOC and comparable EEO not reported in abstract

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Ouarda et al. (2019); real hospital wastewater	12 pharmaceutical residues at native concentrations; analyte-specific initial concentrations not tabulated in accessible abstract	Nb/BDD anode; Ti cathode	4.42, 17.7 and 35.4 mA cm^{-2}	Native wastewater; pH/electrolyte concentration NR in abstract	120 min; experimental hospitalwastewater reactor	Up to 100% abatement of selected residues at 35.4 mA cm^{-2}	HRMS and UV/fluorescence surrogates used; COD/TOC and kWh m^{-3} not stated in abstract; realmatrix study, not full-scale demonstration

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Interpretation of Table 1: The results are condition-specific and should not be used as a league table. In the paracetamol study, sulfate concentration enhanced oxidation and sulfate-derived oxidants, but TOC removal remained below the parent-compound removal. The ibuprofen comparison directly tested PbO_2 and BDD and showed greater COD/TOC reduction with BDD over six hours. Carbamazepine reached complete parent disappearance at a very high reported current density and electrolyte concentration; absence of corresponding TOC data prevents a claim of complete mineralization. The simulated OTC study showed a higher parent-removal percentage in a TiO_2/activated-carbon three-phase system, but its applied voltage, catalyst dose and reactor differ from the conventional EO experiment. The cephalexin and ofloxacin values are similarly optimized laboratory endpoints. The real hospital-wastewater study is more environmentally representative but still needs long-duration continuous operation and energy normalization.

5.2 Electrode material: oxygen-evolution overpotential, by-products, stability, cost and toxicity

The oxygen-evolution overpotential (OEP) is a useful descriptor of anode behaviour, but it is not a material constant independent of measurement conditions. It varies with substrate, coating, surface state, electrolyte, pH and current density. Table 2 therefore gives a qualitative relative comparison rather than unsupported universal numerical potentials. In general, BDD exhibits a very high OEP; PbO_2 and doped SnO_2 are also high-OEP anodes; and RuO_2/IrO_2-based MMOs generally have lower OEP and can favour oxygen evolution at lower potentials. The distinction affects radical availability and mineralization, but actual performance must be demonstrated under the same reactor and matrix conditions (Panizza & Cerisola, 2009; Moreira et al., 2017; Moradi et al., 2020).

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260 Table 2. Qualitative comparison of electrode classes used in electrochemical advanced oxidation. Relative oxygen-evolution behaviour is 261 condition-dependent; the table summarizes general material tendencies rather than universal numerical potentials (Panizza & Cerisola, 2009; 262 Moreira et al., 2017; Moradi et al., 2020).

Electrode class	**Relative OER overpotential**	**Typical oxidation behaviour**	**Potential byproducts / risks**	**Stability and lifetime**	**Relative cost / scale-up**	**Toxicity and safety**
BDD	Very high	Weakly adsorbed •OH; strong nonselective oxidation; often favourable for mineralization	In chloride media can form active chlorine, organohalogen products, chlorate and perchlorate; risk depends on charge and matrix	High chemical stability; coating/substrate integrity and longduration performance still require testing	Very high capital/material cost; diamond deposition and substrate preparation are cost drivers	Carbon/boron electrode is generally chemically inert; the main concern is oxidant/by-product toxicity rather than BDD metal leaching

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PbO_2 (including modified β-PbO_2)	High	Strong oxidative capacity; can favour •OH formation when OEP is raised by modification	Transformation products depend on electrolyte; Pb release from coating failure or corrosion is a material-specific concern	Good activity, but coating adhesion, passivation and lead release must be assessed during ageing tests	Lower material cost than BDD in many laboratory comparisons; replacement and hazardous-waste management add costs	Lead toxicity makes leaching control and post-use electrode management essential
SnO_2/ doped SnO_2	High to very high (compositiondependent)	Non-active behaviour; hydroxylradical-mediated oxidation possible	Chlorinated products if chloride is present; coating-specific metal/antimony release needs monitoring for doped forms	Potentially limited by coating stability and substrate corrosion; accelerated-life data are needed	Potentially lower than BDD, but fabrication and lifetime determine true cost	Assess leaching of dopants and substrate metals; materialspecific ecotoxicity testing needed

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MMO / DSA (RuO_2-, IrO_2based Ti coatings)	Low to moderate relative to BDD	Active-anode behaviour; selective/partial oxidation and mediator generation; often robust for industrial electrolysis	Active chlorine may dominate in chloriderich water; partial oxidation can leave organic intermediates	Generally good industrial dimensional stability; service life depends on coating formulation and operating potential	Moderate; established manufacture and industrial use, but noble-metal coatings increase cost	Potential Ru/Ir/Ti release should be checked; oxidation by-products can dominate overall risk
Carbon / graphite / carbon felt	Low to moderate	Direct electron transfer and adsorption; lower OEP generally limits •OH-driven mineralization	Surface oxidation, fouling and adsorption transfer; chloride-mediated products depend on electrolyte	Mechanical stability varies; fouling and consumption/regeneration may limit life	Generally low to moderate; attractive for lowcost and 3D systems	Usually lower metalleaching concern, but carbon fines, binder or catalyst components require assessment

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Modified / composite electrodes (e.g., TiO_2/AC; doped coatings)	Designdependent	Combines catalytic active sites, adsorption and improved mass transfer; may lower required applied voltage	Adsorption can mask incomplete destruction; leaching of modifiers and halogenated products remain possible	Performance is formulationspecific; cycling and acceleratedlife tests required	Variable; catalyst synthesis and electrode fabrication may offset material savings	Assess nanoparticle/catalyst release and transformationproduct toxicity; do not infer safety from high parent removal

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Electrode choice is application-specific. BDD supports deeper oxidation but is costly and can generate oxychlorine species in chloride-rich water. MMOs are robust but may favour oxygen evolution or mediated oxidation; PbO_2 requires lead-leaching controls. Modified and three-dimensional electrodes may improve parent removal, yet catalyst release, adsorption artefacts, TOC removal and cycling stability must be tested. High first-cycle removal alone does not establish durability or low cost (Moradi et al., 2020; Hadavand et al., 2025; Osman & Uğurlu, 2025).

5.3 Transformation products, chlorine-mediated oxidation and toxicity

EO can transform a parent pharmaceutical through hydroxylation, dealkylation, deamination, desulfonation, cleavage of side chains, aromatic-ring opening and, in halide-containing electrolytes, halogenation. The resulting products may undergo further oxidation, but the sequence may stop at persistent organic acids or other intermediates when the applied charge, contact time or radical exposure is insufficient. Transformation-product analysis is therefore required alongside parent-compound monitoring. A claim of detoxification should be reserved for studies that include appropriate bioassays or validated toxicity endpoints.

The supporting electrolyte is a major control on both degradation kinetics and environmental risk. García-Espinoza and MijaylovaNacheva (2019) studied sulfamethoxazole, propranolol and carbamazepine in a 2-L undivided batch reactor with a Nb/BDD mesh anode and Ti cathode at 2.5 A. The electrolytes were Na_2SO_4, NaCl and NaBr, each at 7 mM. Ten byproducts were identified with Na_2SO_4, compared with 19 in NaCl (including seven halogenated products) and 20 in NaBr (including ten halogenated products). The authors reported significant inhibition effects on Vibrio fischeri for mixtures formed with reactive halogen species, whereas acute toxicity was not observed in the Na_2SO_4 test at the tested pharmaceutical concentration of 5 µg L^{-1}. Thus, faster degradation in chloride/bromide media is not necessarily a safer outcome.

Oxychlorine formation is another concern. In chlorine-mediated electro-oxidation experiments with BDD and thermally decomposed iridium-oxide-film anodes, chlorate and perchlorate became dominant chlorine by-products at high specific charge; the product distribution also depended on anode type and operating conditions (Zöllig et al., 2015). These experiments were conducted with urine rather than pharmaceutical wastewater, so they provide mechanistic evidence of chloride-related risk, not a quantitative prediction for every PhAC reactor. In practical EO, chloride concentration, specific charge, pH, organic load and treatment time should be measured and controlled. Monitoring should include free/total chlorine, chlorate,

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perchlorate and, where relevant, adsorbable organic halogens (AOX) and volatile chlorinated organics.

The diclofenac photoproduct literature also illustrates why the parent concentration is an insufficient proxy for environmental safety. Schmitt-Jansen et al. (2007) observed increased phytotoxicity during sunlight exposure as diclofenac phototransformation products accumulated, while Schulze et al. (2010) identified a phytotoxic product with greater toxicity than the parent. These are phototransformation studies rather than EO experiments; they are cited here to demonstrate the broader risk of assuming that transformation automatically means detoxification. For EO specifically, Zhu et al. (2016) found that residual antibacterial activity of ciprofloxacin, norfloxacin and ofloxacin decreased after electrochemical oxidation, but toxicity to green algae remained or increased for some oxidation mixtures.

This evidence reinforces the need for multiple biological endpoints.

5.4 Antibiotics, residual antimicrobial activity and antimicrobial resistance

Antibiotic disappearance and inactivation of antimicrobial activity are not equivalent. Fluoroquinolone electro-oxidation can modify the piperazinyl side chain and form multiple transformation products; residual antibacterial activity generally decreased in the study by Zhu et al. (2016), but toxicity to green algae did not decline uniformly. A decrease in bacterial inhibition in one assay therefore cannot be taken as proof that the effluent is ecologically harmless or that resistance-related risks have been eliminated.

Electrochemical oxidation also needs to be distinguished from antibiotic-resistance-gene removal. Chen et al. (2020) found that EO inactivated selected antibiotic-resistant bacteria but did not effectively remove intracellular or extracellular antibiotic-resistance genes under the tested conditions. Their electro-Fenton process, by contrast, achieved reductions of 3.8 and 4.1 log for intracellular tetA and ampC, and 5.2 and 4.8 log for extracellular tetA and ampC, respectively, after 120 min at 21.42 mA cm^{-2} with 1.0 mmol L^{-1} Fe^{2+}. These results are process- and condition-specific, but show why antibiotic removal, ARB inactivation and ARG degradation should be measured separately. Future PhAC EO studies should include residual antibacterial assays, viable ARB counts, qPCR of selected ARGs and transformation-product toxicity instead of relying only on LC–MS disappearance.

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5.5 Comparison with established micropollutant treatment technologies

EO should not be described as universally superior to ozonation, activated carbon adsorption or UV/H_2O_2. These technologies have different operating principles and maturity. Ozonation and powdered/granular activated carbon are established as tertiary micropollutant stages in several full-scale settings. A Swiss technical assessment covering 13 WWTP micropollutant stages (six ozone, five powdered activated carbon and two granular activated carbon plants) reported average annual removal of 75–90% for the monitored micropollutant mixtures. Electricity use was 0.02–0.10 kWh m^{-3}, with ozone systems tending to use more electricity than activated-carbon systems; annual operating costs were 0.02–0.09 CHF m^{-3}, rising to approximately 0.07–0.27 CHF m^{-3} when capital costs were included (VSA, 2025). These are Swiss plant-level figures and should not be transferred directly to Indian cost conditions. UV/H_2O_2 has demonstrated real-matrix treatment but is sensitive to UV transmittance and radical scavenging. For the real wastewater comparison by Cibati et al. (2022), EEO was 0.9–1.5 kWh m^{-3} $order^{-1}$ for urban secondary effluents and greywater, but 7.3–9.1 kWh m^{-3} $order^{-1}$ for hospital and pharmaceutical-industry wastewater. The industrial effluent showed only 17% overall pharmaceutical removal under the reported optimum. These figures illustrate that energy demand and treatment outcome depend strongly on matrix composition. They are not directly comparable with the EO studies in Table 3, for which equivalent EEO values were generally not reported. Table 3 compares EO with treatment technologies that have established plant-scale use or real-wastewater performance data. The values are not a head-tohead trial: the Swiss operating figures and the real-matrix UV/H_2O_2 EEO values come from different systems and should be interpreted as context rather than directly ranked (VSA, 2025; Cibati et al., 2022).

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351 Table 3. Contextual comparison of electrochemical oxidation with ozonation, activated-carbon adsorption and UV/H_2O_2 for micropollutant 352 treatment. Quantitative ranges are source-specific and not directly comparable across technologies (VSA, 2025; Cibati et al., 2022).

Technology	**Quantitative evidence available**	**By-product / residuals profile**	**Technology readiness for PhAC polishing**	**Relative strengths and constraints**
Electrochemical oxidation (EO)	PhAC studies report high compound-specific removal in optimized bench tests; normalized kWh m^{-3} and EEO are often NR	Possible organic transformation products; active chlorine, AOX, chlorate and perchlorate in chloridecontaining matrices; electrode leaching/material stability must be checked	Mostly laboratory, batch and selected realwastewater experimental studies; no long-duration full-scale PhAC EO evidence was established in this review	Modular and electrically controlled; can suit concentrated or sourceseparated streams; energy, electrode life and products/toxicity need validation

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Ozonation	Swiss WWTP assessment: 0.02–0.10 kWh m^{-3} electricity across micropollutant stages; ozone generally toward higher electricity; operating cost 0.02–0.09 CHF m^{-3}	Ozone transformation products; bromate risk where bromide is present; off-gas destruction and residual toxicity monitoring are needed	Commercial/full-scale micropollutant stages documented in Switzerland and other jurisdictions	Mature and established for tertiary polishing; requires ozone generation, gas transfer and control of oxidation byproducts

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PAC / GAC adsorption	Swiss plants: micropollutantstage electricity within overall 0.02–0.10 kWh m^{-3} range; activated-carbon dose 5–20 mg L^{-1}; operating cost 0.02–0.09 CHF m^{-3}	Transfers contaminants to spent or regenerated carbon rather than destroying all organics; spent carbon management is required	Commercial/full-scale micropollutant stages documented	Mature; effective broadspectrum polishing; media replacement/regeneration and residual solids handling determine life-cycle impacts
UV/H_2O_2	Real wastewater EEO: 0.9–1.5 kWh m^{-3} $order^{-1}$ in urban/greywater; 7.3–9.1 in hospital/industrial effluent (Cibati et al., 2022)	Transformation products can remain; toxicity should be assessed; UV transmittance and radical scavenging affect performance	Commercial AOP components, with realmatrix performance studies; reported performance depends on stream and UV conditions	Can oxidize photo-reactive and recalcitrant PhACs; matrix scavenging and energy demand may be high in industrial/hospital streams

The comparison supports a conditional positioning of EO. EO may be most defensible for concentrated, saline or source-separated streams; selected hospital or pharmaceutical-industry effluents; treatment of reverse-osmosis concentrate; or decentralized facilities where modularity and on-site oxidant generation have value. For large municipal tertiary treatment, where ozonation and PAC/GAC have established plant-scale operating records, EO must demonstrate competitive energy use, electrode replacement cost and by-product control under local conditions before claims of superiority or sustainability are justified.

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6. Energy consumption and current efficiency

Electrical energy is a central design and operating constraint for EO. The energy required depends on cell voltage, current, treatment time, treated volume, electrode gap and area, electrolyte conductivity, hydrodynamics and the extent of oxidation achieved. At fixed current, increasing voltage increases electrical power; at fixed voltage, current density depends on current divided by the active electrode area. Raising current density may accelerate parent removal, but beyond the useful operating range additional charge can be diverted to oxygen evolution, heating or other side reactions. Longer treatment can increase mineralization while increasing electricity use and the risk of secondary oxidation products.

Three energy metrics should be distinguished and reported whenever possible. Volumetric specific energy consumption is $SEC_v = U \times I \times t / (1000 \times V)$, where U is cell voltage (V), I is current (A), t is time (h) and V is treated volume (m^3); the result is kWh m^{-3}. COD-normalized specific energy is $SEC_COD = U \times I \times t / (1000 \times \Delta COD \times V)$, where ΔCOD is the COD removed in kg m^{-3}; the result is kWh kg^{-1} COD removed. Electrical energy per order (EEO) is $EEO = U \times I \times t / [1000 \times V \times \log_{10}(C_0/C_t)]$, where C_0 and C_t are initial and final concentrations of the selected target compound; units are kWh m^{-3} $order^{-1}$. The batch and flow-through EEO expressions should be selected consistently with the reactor configuration and the definition of order used (Bolton et al., 2001). Instantaneous current efficiency (ICE) and mineralization current efficiency (MCE) express the fraction of applied charge that is used for the target oxidation. For COD expressed in g O_2 L^{-1}, a commonly used MCE form is $MCE\ (\%) = [F \times V_L \times \Delta COD / (8 \times I \times t_s)] \times 100$, where F is the Faraday constant (96,485 C mol^{-1}), V_L is solution volume (L), ΔCOD is the COD decrease (g O_2 L^{-1}), I is current (A) and t_s is elapsed time (s). The exact stoichiometric formulation should match whether COD is measured as oxygen equivalent and whether the measurement is interval-based or cumulative. Authors should state assumptions and analytical units so that the calculation can be reproduced.

The PhAC studies in Table 1 illustrate a major reporting gap: several report voltage or current density and parent removal but omit cell voltage, current, treated volume or energy consumption. Without these data, kWh m^{-3} or EEO cannot be reconstructed reliably. The ibuprofen study measured general current efficiency and MCE, but the available abstract reports COD and TOC ranges rather than a common normalized energy value (Ciríaco et al., 2009). The Ti_4O_7-nanotube reactive electrochemical membrane study on actual landfill-leachate reverse-osmosis concentrate reported 7.6 kWh kg^{-1} COD removed and 267.3 h accelerated service-life testing; it also reported COD removal of 82% and TOC removal of 68% under its stated optimal conditions. This is useful engineering evidence for concentrated organic streams, but it is not a pharmaceutical-specific trial and should not be presented as direct PhAC performance (Qian et al., 2024).

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A numerical ranking of BDD, MMO, PbO_2 and modified electrodes by energy demand is not scientifically defensible from the present PhAC dataset because studies use different pollutants, current densities, electrolyte levels, reactor volumes and treatment endpoints. BDD can offer stronger mineralization but is capital-intensive; MMO can be robust and may operate with lower cell voltage but can favour oxygen evolution or active-chlorine mediation; PbO_2 may be less costly but lead management and electrode replacement affect life-cycle cost; and modified electrodes may shorten treatment time but require durability and leaching data. The appropriate comparison is therefore a normalized, side-by-side experiment or a complete mass-and-energy balance, not a comparison of isolated removal percentages.

7. Process scale-up and real-world applicability

Scale-up requires more than increasing electrode dimensions. Reactor selection controls current distribution, inter-electrode potential drop, mass transfer, bubble removal, hydraulic residence time, heat transfer and fouling. Reported reactor configurations include undivided batch cells, divided cells, stirred tanks, recirculating flow-through cells, filter-press/flow-by reactors, porous flowthrough reactive electrochemical membranes and three-dimensional particle-electrode systems. Flow-through systems can improve contact with porous electrodes, while filter-press systems offer modular electrode stacking; both require attention to pressure drop, current distribution, channeling and cleanability (Cornejo et al., 2024).

For pharmaceutical wastewater, the real-hospital study by Ouarda et al. (2019) is a relevant matrix demonstration, but it used an experimental reactor and 120-min treatment. It does not establish continuous full-scale performance. A pilot-scale review of electrochemical wastewater technologies describes pre-pilot and pilot demonstrations for anodic oxidation and electro-Fenton in broader organic wastewater applications, but the existence of pilot EO technology for organics should not be equated with a long-term PhAC-specific pilot demonstration (Clematis et al., 2023). In the PhACfocused evidence assembled here, long-duration, continuous-flow operation on real pharmaceutical wastewater remains a key gap.

Electrode lifetime must be evaluated with both operational ageing and accelerated testing. Relevant failure modes include coating delamination, passivation, corrosion of the substrate, loss of catalytic activity, fouling and leaching of constituent metals. Moradi et al. (2020) reviewed service life and stability across EAOP electrodes and emphasized that electrode composition and protective coatings influence durability. Qian et al. (2024) reported 267.3 h from accelerated service-life testing for a Ti_4O_7-nanotube reactive electrochemical membrane treating actual landfill-leachate concentrate, together with 20-cycle performance tests. This is an encouraging non-pharmaceutical example, but a single accelerated test cannot replace multi-month or multi-year operation on representative wastewater.

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A cost per cubic metre cannot be estimated for the PhAC EO studies in Table 3 because complete electricity consumption, electrode depreciation/replacement, electrolyte consumption, sludge or concentrate management and maintenance costs are not consistently reported. The Swiss micropollutant-stage figures for ozone/PAC/GAC provide useful benchmarks (VSA, 2025), but they are not transferable cost estimates for EO or for Indian electricity and materials prices. A credible EO techno-economic assessment should report installed electrode area, cell voltage, current, flow rate, treated volume, kWh m^{-3}, kWh kg^{-1} COD, electrode mass loss and replacement interval, electrolyte/chemical use, pretreatment, post-treatment and effluent-disposal cost.

On the evidence available, EO is most plausibly competitive for concentrated and difficult streams where conventional treatment is inhibited or where a small-volume side stream can be treated selectively: hospital wards or source-separated hospital effluent, pharmaceutical manufacturing streams after equalization/pretreatment, saline brines and reverse-osmosis concentrates, and decentralized facilities requiring modular treatment. For municipal secondary effluent at very large flow, EO has not yet shown a consistent energy and cost advantage over established ozonation or PAC/GAC stages. Each application requires a matrix-specific pilot evaluation and an assessment of the final effluent and residuals.

8. Hybrid and integrated treatment strategies

Hybrid treatment should be designed as an explicit treatment train in which each unit operation addresses a known limitation. An EO–MBR sequence can be configured in two ways. When EO is used as pretreatment, a controlled oxidation dose may transform recalcitrant PhACs into more biodegradable products before a membrane bioreactor (MBR); this approach is appropriate only when EO products are shown to be biodegradable and not inhibitory to biomass. When EO follows an MBR, the biological step removes bulk biodegradable COD and solids, reducing the organic radical demand entering the electrochemical polishing stage. The latter arrangement may be preferable for municipal secondary effluent because it reduces the electrical load imposed by bulk organic matter, although it still requires demonstration of transformation-product and toxicity control.

EO followed by granular activated carbon (GAC) is a plausible polishing train when EO is used to oxidize the parent compound and GAC is used to adsorb residual intermediates or hydrophobic transformation products. Adsorption should not be counted as destruction; spent carbon must be regenerated or managed. Similarly, EO followed by nanofiltration or reverse osmosis may be considered where water reuse is the objective, but the membrane concentrate then requires a defined residuals-management route. EO can be applied to the concentrate stream rather than the entire feed to reduce treated volume, provided the salt load does not create unacceptable halogenated byproducts or membrane/electrode scaling.

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An EO–ozone sequence or parallel ozone/EO train may be considered where the two oxidants provide complementary reactivity, but ozone can also generate transformation products and does not remove all toxicity by default. UV/H_2O_2 can be used as an alternative or polishing AOP where UV transmittance and oxidant dose are suitable; the real-wastewater results of Cibati et al. (2022) show that hospital and industrial matrices may need substantially higher EEO than urban secondary effluent. A process train should therefore be selected on the basis of target PhACs, water quality, EEO, by-products and toxicity, rather than assuming that adding more AOP stages automatically improves sustainability.

For high-strength pharmaceutical manufacturing wastewater, equalization and removal of suspended solids should precede EO to stabilize loading and protect electrodes. A practical sequence may be primary clarification/equalization → biological treatment or physicochemical pretreatment → EO polishing → GAC or biological post-treatment, with toxicity and TOC measurements at each stage. For RO concentrate, the train may be biological/chemical pretreatment → membrane separation → EO of concentrate → residuals polishing; the Ti_4O_7-REM study provides relevant evidence for actual landfill-leachate concentrate but should be treated as transferable engineering evidence, not as pharmaceutical validation (Qian et al., 2024).

9. Future perspectives and prioritized research needs

The next phase of EO research should prioritize reproducible performance data and environmental safety over isolated demonstrations of very high parent removal. The following priorities are justified by the gaps identified across the studies reviewed here.

1. Low-cost, stable and safer electrodes. Develop and compare BDD alternatives, Ti_4O_7, doped SnO_2, MMOs and modified PbO_2 using matched reactor conditions. Report coating composition, substrate, active area, leaching, polarization curves, current efficiency and electrode mass loss. For lead-containing anodes, quantify dissolved lead through the full ageing cycle and include end-of-life management.
2. Standardized energy and performance reporting. Every study should report cell voltage, current, current density, electrode area and spacing, treated volume or flow rate, pH, conductivity, electrolyte concentration, treatment time, COD, TOC and parent compound concentration. SEC (kWh m^{-3}), SEC_{COD} (kWh kg^{-1} COD), EEO and MCE/ICE should be calculated with explicit equations and units. This would enable reproducible comparison across BDD, MMO, PbO_2 and modified electrodes (Begildayeva et al., 2026; Bolton et al., 2001).
3. Continuous-flow and long-duration demonstrations. Move from small batch cells to flow-by, flow-through, filter-press and reactive electrochemical membrane reactors with realistic hydraulic loading, current distribution and maintenance protocols. Pilot campaigns should last

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long enough to capture fouling, scaling, coating degradation and seasonal matrix variability; accelerated-life tests should be accompanied by real-time operation.

4. Real wastewater and India-specific validation. Test hospital and pharmaceutical-industry effluents from multiple facilities, including influent variability, dissolved organic matter, chloride, bicarbonate, sulfate, suspended solids and competing contaminants. The selected Indian occurrence values in Figure 1 justify targeted monitoring, but contemporary, geographically representative datasets are still required before design concentrations can be specified.
5. Transformation-product and ecotoxicological monitoring. Combine targeted LC–MS/MS with non-target screening, COD/TOC, AOX, chlorate/perchlorate and bioassays. Assess toxicity of the treated mixture rather than selected products alone, and conduct post-treatment measurements to detect rebound or transformation after discharge.
6. Antimicrobial-resistance endpoints. For antibiotic-containing effluents, quantify parent antibiotic concentrations, residual antibacterial activity, viable antibiotic-resistant bacteria, intracellular and extracellular ARGs, and the abundance of relevant mobile genetic elements. EO and electro-Fenton should not be assumed to provide equivalent ARG removal.
7. Hybrid trains and renewable electricity. Compare MBR→EO, EO→MBR, EO→GAC and EO treatment of NF/RO concentrates using a common mass balance. Couple EO to low-carbon grid supply or renewable electricity where feasible, while accounting for intermittency, storage, electrode utilization and the carbon intensity of electricity used.
8. Techno-economic and life-cycle assessment. Calculate total cost per m^3 and per kg of PhAC/COD removed, including electricity, electrode amortization, electrolyte, pretreatment, post-treatment, replacement, sludge/concentrate handling and monitoring. Life-cycle assessment should include electrode manufacture, critical materials, carbon footprint, spent adsorbent, brine and hazardous by-products. ―Sustainable‖ should be used only after these burdens are quantified against established alternatives.

10. Conclusions

EO can achieve high parent-compound removal for selected pharmaceuticals under optimized laboratory conditions, including paracetamol, carbamazepine, oxytetracycline, cephalexin and ofloxacin. These percentages are not directly comparable because initial concentration, anode type, current density or voltage, electrolyte, pH, treatment time and reactor differ. In studies that report multiple endpoints, parent removal can exceed COD or TOC removal; therefore, disappearance of a pharmaceutical must not be equated with complete mineralization or detoxification.

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Electrode selection and electrolyte chemistry determine not only oxidation rate but also the fate of transformation products. BDD can support strong oxidation and mineralization, but its cost and possible oxychlorine formation require attention. MMO electrodes offer established durability but may favour active-anode or mediated pathways. PbO_2 can be effective but requires lead-leaching controls. Chloride and bromide electrolytes may accelerate removal while producing halogenated transformation products and, at high charge, chlorate and perchlorate. Antibiotic EO can reduce residual antibacterial activity or inactivate selected resistant bacteria, but EO alone has not consistently removed intracellular or extracellular ARGs.

Real hospital wastewater studies demonstrate feasibility beyond synthetic solutions, while UV/H_2O_2 and Swiss ozone/PAC/GAC benchmarks show that matrix effects, energy demand and technology maturity must be considered. The present evidence does not justify claiming that PhAC-specific EO is already a cost-competitive full-scale replacement for established tertiary treatment. Its strongest near-term opportunities are likely source-separated or concentrated streams, selected hospital and pharmaceutical industrial effluents, saline brines and decentralized applications, subject to pilot validation.

The priority is to establish continuous-flow performance, electrode lifetime, energy per treated volume and per mass removed, transformation-product toxicity, antimicrobial-resistance endpoints and cost per m^3 under real wastewater conditions. Integration with renewable electricity and hybrid treatment should be evaluated through techno-economic analysis and life-cycle assessment. EO should be described as sustainable only where the combined impacts of electricity, electrode manufacture and replacement, chemical use, by-products and residuals management are demonstrably favourable compared with ozonation, activated carbon and UV/H_2O_2.

Abbreviations

AOX, adsorbable organic halogens; AOP, advanced oxidation process; ARB, antibiotic-resistant bacteria; ARG, antibiotic-resistance gene; BDD, boron-doped diamond; CEC, contaminant of emerging concern; COD, chemical oxygen demand; DSA, dimensionally stable anode; EAOP, electrochemical advanced oxidation process; ECO, electrocatalytic oxidation (as termed in Osman and Uğurlu, 2025); EEO, electrical energy per order; EO, electrochemical oxidation; GAC, granular activated carbon; ICE, instantaneous current efficiency; MBR, membrane bioreactor; MCE, mineralization current efficiency; MMO, mixed-metal oxide; NF, nanofiltration; NR, not reported; OEP, oxygen-evolution overpotential; OTC, oxytetracycline; PAC, powdered activated carbon; PhAC, pharmaceutically active compound; RO, reverse osmosis; SEC, specific energy consumption; TOC, total organic carbon; WWTP, wastewater-treatment plant.

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