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Peroxi-electrocoagulation for Treatment of Trace Organic Compounds and Natural Organic Matter at Neutral pH

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Peroxi-electrocoagulation for treatment of trace organic compounds and natural organic matter at neutral pH†

Donald R. Ryan, a Patrick J. McNamara, b Claire K. Baldus, a Yin Wang b and Brooke K. Mayer a

Iron-based oxidation technologies can be advantageous for mitigating trace organic compounds (TOrCs) during water and wastewater treatment due to their production of hydroxyl radicals. However, iron-based oxidation often occurs at acidic pH to promote Fenton’s reaction, which limits the processes’ feasibility for treatment applications. This study focused on utilizing iron-electrocoagulation (EC) paired with ex situ H2O2 addition (peroxi-electrocoagulation [EC:H2O2]) to promote oxidative reactions at neutral pH conditions. The hydroxyl radical probe para-chlorobenzoic acid (pCBA) was used to gauge oxidant activity and serve as a representative TOrC. The impact of water pH, current density, iron dose, H2O2 dose (i.e., [H2O2]initial/[Fe2+]generated ratio), and the presence of natural organic matter (NOM) were evaluated. Multivariable regressions showed that high levels of H2O2 relative to iron (i.e., [H2O2]initial/[Fe2+]generated ratio >0.7) inhibited the rate of pCBA oxidation, likely due to additional radical quenching from extra H2O2. Oxidation of pCBA was confirmed at neutral pH conditions, indicating that EC:H2O2 may potentially serve as a multi-mechanistic treatment technology capable of oxidation. Experiments were also conducted in real-world water samples to gauge EC:H2O2 applications for treating groundwater, river water, and primary treated wastewater. Overall, H2O2 addition enhanced the oxidative degradation of TOrCs while still removing NOM. The one exception was the primary effluent sample, which had the highest degree of oxidant scavenging of all matrices tested. The electrical energy per order (EEO) metric demonstrated that EC:H2O2 is competitive with other TOrC oxidation technologies, with the added benefit of NOM mitigation in the same unit process.

Environmental significance
As trace organic compounds are increasingly being monitored in drinking water, technologies are needed that can mitigate their risks. Iron-electrocoagulation paired with hydrogen peroxide can potentially serve as an oxidative technology for these contaminants, and can simultaneously remove trace organics and bulk organics such as natural organic matter within the same unit process. This combined process can be particularly advantageous for rural and decentralized systems due to multiple treatment processes occurring within the same reactor and favorable energy requirements as compared to oxidation technologies such as UV-H2O2 and ozonation.

1. Introduction
Iron has expansive applications for water and wastewater treatment. Different iron-based treatment pathways proceed depending on the valence state of the iron (e.g., ferrous [Fe2+] or ferric [Fe3+]). Iron speciation varies as a function of pH and the presence of dissolved oxygen in water. Ferric iron predominates in the oxygen-rich neutral and basic pH conditions that are typical for water and wastewater treatment. During coagulation, iron is dosed as ferric chloride or ferric sulfate targeting removal of turbidity and natural organic matter (NOM).1 Alternately, Fe2+ predominantly exists in acidic conditions, and can mediate oxidative treatment via Fenton’s reaction, which produces hydroxyl radicals (HO') that can oxidize trace organic compounds (TOrCs).2-4 Accordingly, iron-based treatments typically feature either non-destructive removal or oxidative destruction due to dominant pathways under different pH conditions.5 Research is needed to simultaneously promote both non-destructive and oxidative destructive pathways through Fenton’s reaction at circumneutral pH for treating multiple classes of contaminants such as bulk organics (i.e.,
Radical quenching, reagent quenching

Radical production

Fenton’s reaction relies on non-complexed Fe$^{2+}$ and H$_2$O$_2$ as reagents to form HO$^\cdot$ (Reaction 1, Table 1). Hydroxyl radicals are highly reactive (2.8 V vs. standard hydrogen electrode) and can react with Fenton’s reagents (Reactions 4 and 5) at faster rates ($10^8$ M$^{-1}$ s$^{-1}$) than the radicals are generated (40–80 M$^{-1}$ s$^{-1}$), which terminates Fenton’s reaction due to oxidant and reagent depletion and hinders treatment effectiveness. However, in acidic conditions (pH 2–4), soluble Fe$^{3+}$ can be recycled into Fe$^{2+}$ (Reaction 2), thereby continuing HO$^\cdot$ generation without reagent depletion.

At neutral pH conditions in water and wastewater treatment, the feasibility of Fenton’s reaction is limited for several reasons: 1. Iron speciation shifts toward Fe$^{3+}$, which is less soluble and more prone to floc formation compared to Fe$^{2+}$, resulting in termination of the Fenton’s reaction cycle by inhibiting regeneration of Fe$^{2+}$ required for oxidant generation.

2. Dissolved oxygen readily oxidizes Fe$^{2+}$ in neutral and basic pH conditions (Reaction 3). Each increase in pH unit increases the oxidation rate of Fe$^{2+}$ 100-fold, leading to less available Fenton’s reagents.\(^a^\,\text{b}\)

3. Anionic ligands in natural waters (e.g., OH$^-$ and CO$_3^{2-}$) form complexes with Fe$^{2+}$, which decreases the amount of non-complexed Fe$^{3+}$ available to react with H$_2$O$_2$ to generate oxidants.\(^c\)

Accordingly, pH limitations restrict Fenton applications to a narrow pH range (pH 2–4), which impedes implementation in water and wastewater treatment due to the intensive pH adjustments to acidify and neutralize waters before and after treatment. Additionally, acidic waters can enhance corrosion of infrastructure and shift the pH of natural waters following discharge.\(^d\)

To facilitate Fenton oxidation at neutral pH, the key premise relies on generating or stabilizing the Fe$^{2+}$ needed to react with H$_2$O$_2$ to form HO$^\cdot$. Accordingly, electrochemical water treatment processes, such as electrocoagulation (EC), may be used for Fenton oxidation at neutral pH by generating non-complexed Fe$^{2+}$ via anodic dissolution of iron electrodes.\(^e\)

Continuous generation of Fe$^{2+}$ can be advantageous for Fenton oxidation at neutral pH by minimizing the need for Fe$^{3+}$ reduction to Fe$^{2+}$ via H$_2$O$_2$ (Reaction 2). Prior research has also shown that EC alone can generate HO$^\cdot$ to treat TOrCs through the in situ generation of Fe$^{3+}$ at the anode and H$_2$O$_2$ production at the cathode.\(^c\,\text{d}\) During electrolysis, the microenvironment near the anode surface is acidic.\(^f\) Consequently, Fenton reactions may occur at the vicinity of the anode surface even if the bulk solution pH is circumneutral, potentially leading to oxidative conditions at neutral pH between H$_2$O$_2$ and the iron anode surface. Supplemental addition of H$_2$O$_2$ as a radical promotor, known as peroxy-electrocoagulation (EC:H$_2$O$_2$), can further enhance EC’s oxidizing capacity and serve as a multi-mechanistic process. During EC:H$_2$O$_2$, Fe$^{2+}$ is continually generated at low concentrations (nM s$^{-1}$ based on Faraday’s law) over the course of electrolysis, such that non-complexed Fe$^{2+}$ is available for oxidation by H$_2$O$_2$. As a result, less Fe$^{2+}$ is “wasted” as a Fenton’s reagent by non-radical generating side reactions such as ligand complexation or oxygenation.\(^g\)

This combination of Fe$^{2+}$ reagent generation and minimal reliance on Fe$^{3+}$ reduction to Fe$^{2+}$ can make EC:H$_2$O$_2$ an advantageous dosing method compared to ex situ reagent dosing in Fenton applications. Of note, while in situ Fe$^{2+}$ dosing can be advantageous, ex situ H$_2$O$_2$ dosing may still be needed, depending on EC:H$_2$O$_2$ reactor design.

Pratap and Lemley (1998, 1994)\(^h\,\text{i}\) demonstrated point-of-concept use of EC:H$_2$O$_2$ for remediation of the herbicides atrazine and metalochlor at neutral pH conditions. Since the inception of EC:H$_2$O$_2$, research has primarily focused on coagulation/flocculation during industrial wastewater treatment for removing bulk organic pollutants (such as chemical oxygen demand) at high concentrations (mg L$^{-1}$ levels).\(^j\,\text{k}\,\text{l}\,\text{m}\,\text{n}\) However, these high-strength wastewater studies do not translate well to municipal wastewater and drinking water treatment applications. For example, environmental waters have lower conductivity, fewer oxidant scavengers, higher dissolved oxygen, and neutral pH conditions, all of which impact the oxidative efficiency of EC:H$_2$O$_2$ and speciation of iron in water. Considering iron’s treatment capabilities, EC:H$_2$O$_2$ may also

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**Table 1** Fenton’s reaction. Iron species are color-coded to reflect the valence state: blue represents ferrous iron (Fe$^{2+}$) and orange represents ferric iron (Fe$^{3+}$)\(^a\).

<table>
<thead>
<tr>
<th>Reaction</th>
<th>Chemical reaction</th>
<th>Role</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Fe$^{2+}$ + H$_2$O$_2$ → Fe$^{3+}$ + HO$^\cdot$ + OH$^-$</td>
<td>Radical production</td>
</tr>
<tr>
<td>2</td>
<td>Fe$^{3+}$(_{aq}) + H$_2$O$_2$ → Fe$^{2+}$ + HO$_2^\cdot$ + H$^+$</td>
<td>Ferrous regeneration via ferric reduction</td>
</tr>
<tr>
<td>3</td>
<td>Fe$^{2+}$ + $\frac{1}{4}$O$_2$ + 2OH$^-$ + $\frac{1}{2}$H$_2$O → Fe(OH)$_3$ (s)</td>
<td>Oxygenation of ferrous iron, reagent quenching</td>
</tr>
<tr>
<td>4</td>
<td>Fe$^{2+}$ + HO$^\cdot$ → Fe$^{3+}$ + OH$^-$</td>
<td>Radical quenching, reagent quenching</td>
</tr>
<tr>
<td>5</td>
<td>H$_2$O$_2$ + HO$_2^\cdot$ → HO$_3$ + H$_2$O</td>
<td>Radical quenching, reagent quenching</td>
</tr>
</tbody>
</table>

\(a\) Reactions adapted from ref. 2, 3 and 6–9.
offer an opportunity for simultaneous treatment of TOCs and bulk organics (e.g., NOM and chemical oxygen demand) in a single unit process as the Fe^{3+} produced following Fenton’s reaction can subsequently contribute to physical removal (i.e., non-destructive removal) of contaminants through coagulation, flocculation, and sedimentation processes.

The goal of this research was to evaluate EC:H2O2 for simultaneous treatment of both TOCs and NOM at neutral pH conditions. To vet oxidation, para-chlorobenzoic acid (pCBA) was selected as the representative TOC, and also served as a HO’ probe for advanced oxidation process (AOP) effectiveness.23 The relative impacts of current density (i.e., iron dosing rate), H2O2 dose, and the corresponding [H2O2]initial/[Fe^{2+}]generated ratio were tested in synthetic matrices. Experiments were conducted to differentiate non-destructive removal via EC-only from oxidative destructive removal and to assess the contribution of potential oxidants generated in EC:H2O2 such as HO’ and H2O2. Experiments were also conducted using surface water, groundwater, and wastewater sources to evaluate the influence of water quality parameters (i.e., dissolved organic carbon [DOC], pH, conductivity, and ions) and the feasibility of EC:H2O2 for different treatment applications. Finally, electrical energy per order of magnitude reduction (EEO) was calculated for all matrices to provide a means of comparing EC:H2O2 energy requirements relative to other advanced oxidation processes.

2. Materials and methods

2.1. Experimental protocols for EC:H2O2 tests of pCBA removal

The EC:H2O2 batch experiments were conducted for 15 minutes of electrolysis with 150 rpm mixing (G = 180 s^-1) in 4 mM HCO3⁻ buffer solutions containing 400 µg L^-1 pCBA. The pCBA concentration of 400 µg L^-1 was selected based on reliable analytical quantification of >90% removal at a target pCBA concentration below that of mg L^-1-level background oxidant scavengers (i.e., NOM). Electrolysis was performed in 200 mL polypropylene beakers using 1020 steel iron electrodes (VMe-scavengers), (99%, Sigma Aldrich, St. Louis, MO) was selected as the HO’ probe due to its resistance to sorption on iron flocs and frequent use as a radical probe to demonstrate the treatability of TOCs by HO’ exposure.23,26-28 Compared to other TOCs, pCBA is classified as having “moderate reactivity” with HO’, as reported by Gerrity et al. (2012),23 which is similar to TOCs of concern such as atrazine and 1,4-dioxane.

Three reactor inputs – current density, H2O2 dose, and the corresponding [H2O2]initial/[Fe^{2+}]generated ratio – were evaluated to gauge their relative influence on treatment. For EC experiments, the current density was synonymous with the iron loading rate. The iron applied in each test was varied by adjusting the current density (and consequently the iron loading rate). For these experiments, the current density ranged from 3 to 15 mA cm^-2 (charge loading rate = 12–60 Coulomb L^{-1} min^{-1}, iron loading rate = 3.5-17.3 mg Fe^{2+} per L-min). The H2O2 stock (ACS reagent grade, Sigma Aldrich, St. Louis, MO) was added at the beginning of EC:H2O2 experiments at levels ranging from 10 to 200 mg H2O2 per L to assess pCBA treatment resulting from a fixed amount of H2O2 available for Fe^{2+} to generate radicals. The corresponding [H2O2]initial/[Fe^{2+}]generated was 0.3–1.6 based on current density and H2O2 inputs. Notably, the [H2O2]initial/[Fe^{2+}]generated ratio reflects the total H2O2 added at the beginning of the reaction, divided by the amount of Fe^{2+} generated by EC (estimated by Faraday’s Law) by the end timepoint when pCBA removal ceased due to H2O2 depletion. The end timepoint of the pCBA degradation reaction was determined as the time point at which less than 10% difference in pCBA removal compared to the preceding time point was observed, likely indicating depletion of H2O2. Samples were collected every 2.5 minutes for 10 minutes, with a final sample at 15 minutes for kinetic analyses. Kinetic curves were fit to at least four data points (R^2 > 0.95 for all) to calculate first order rate constants for pCBA degradation for samples collected prior to H2O2 depletion, assessed as noted above.

2.2. Removal pathway control experiments

Experiments were conducted under the same electrolysis and current density conditions described in Section 2.1 to isolate the impact of different system inputs and delineate the potential treatment pathways in EC:H2O2, including oxidation by HO’ and H2O2 as well as physical removal by sorption to iron flocs. For HO’ oxidation controls, methanol was spiked in stoichiometric excess (12 mM MeOH) of pCBA and H2O2 to quench HO’ that would otherwise react with pCBA. In this case, H2O2 is reactive with electron-dense compounds and unlikely to react with unsaturated alcohols such as methanol. For no electricity controls (e.g., H2O2 controls), 30 mg H2O2 per L was spiked into the reactors containing the iron electrodes and stirred for 15 minutes to assess the potential pCBA removal due to H2O2 under treatment conditions without electricity in addition to potential losses via sorption to the electrode surface.

Kinetic analyses were conducted to estimate the competition between H2O2 and O2 as a function of H2O2 inputs and water chemistry conditions ([H2O2] dose, O2, and pH). These tests assessed the feasibility of HO’ generation under neutral pH conditions and informed mechanistic analyses (oxidation by O2 limits HO’ production by generating Fe^{3+}). The relative rates of oxidation and associated rate constants are provided in the ESI S4.

2.3. Water quality conditions

All EC:H2O2 experiments were conducted in 4 mM bicarbonate solution (with the exception of the environmental waters) to simulate buffered conditions for neutral pH environmental conditions.

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wastewater containing alkalinity, and to supply an electrolyte for electrochemical reactions (Table 2). Environmental waters were sampled to assess the impact of water quality and treatment performance in real waters relative to synthetic waters containing different NOM sources (Table 2). These analyses are important for informing the role of other environmentally relevant water quality parameters such as NOM characteristics and concentration, conductivity, and divalent cations, all of which can impact treatment efficacy. A sample from the Milwaukee River (Milwaukee, WI) was used to test the impact of NOM and mid-range conductivity water. Groundwater from a drinking water well in West Bend, WI, was tested to reflect low dissolved organic carbon (DOC) and high conductivity conditions. Finally, primary effluent from an urban water reclamation facility in Milwaukee, WI, was tested for the impact of high DOC due to anthropogenic NOM and other oxidant scavengers (such as bulk chemical oxygen demand). The wastewater also served as a point of comparison to previous EC:H2O2 wastewater studies. For DOC quantification experiments, a sedimentation phase was required after EC:H2O2 to allow the flocs to settle prior to DOC analysis. Batch tests were performed as described in Section 2.1 followed by an additional tapered flocculation phase (10 minutes at 40 rpm [G = 25 s\(^{-1}\)] and 10 minutes at 20 rpm [G = 9 s\(^{-1}\)]) and a 20 minute sedimentation period to remove flocs (method adapted from Ryan et al. (2020)).

### 2.3.1. Analytical measurements

Liquid chromatography-mass spectrometry was utilized to quantify pCBA (method adapted from Vanderford et al. (2007)). All pCBA samples were filtered through 0.22 μm PTFE syringe filters (Agela Technologies, Wilmington, DE) prior to analyses. Additional information on chromatography and mass spectrometry conditions is provided in ESI S1.† The H\(_2\)O\(_2\) concentrations before and after each experiment were measured using Hach Model Hyp-1 test kits. The DOC was measured \(\text{via} \) a Shimadzu TOC – VCSN based on U.S. EPA Method 415.3. All DOC samples were filtered through 0.45 μm PTFE filters (Agela Technologies) prior to analyses. ICP-MS (7700 series, Agilent Technologies, Santa Clara, CA) was used to measure cations in real-world water samples. Alkalinity was measured \(\text{via} \) titration using Hach Model 2443-89 test kits.

### 2.3.2. Electrical energy per order

Electrical energy per order of magnitude removal (\(E_{EO}\)), as shown in eqn (1), was estimated to provide a figure of merit for comparing energy requirements (kW h m\(^{-3}\)-order) for EC:H\(_2\)O\(_2\) to other oxidative treatment technologies.\(^{29}\) The voltage reading was recorded for each current density during each test to calculate power (power = voltage \(\times\) current). Pseudo-first order rate constants were used to normalize treatment times across experiments as different reactor inputs and water quality conditions required different treatment times for 90% pCBA removal.

\[
E_{EO} = \frac{P}{V \times 0.4343 \times 3600 \times 1000}
\]

where \(P\) is power in \(W\), \(V\) is volume in \(m^3\), and \(k\) is the pseudo-first order rate constant for pCBA removal in \(s^{-1}\). The coefficient of 0.4343 = log(C\(_{\text{final}}\)/C\(_{\text{initial}}\)) for one order of magnitude reduction. The conversion factor 3600 is used to convert seconds to hours, and 1000 is used to convert W to kW.

### 2.3.3. Data analysis and interpretation

GraphPad Prism (version 9.5.1.) software was used to conduct one-way and two-way ANOVA followed with Tukey’s multiple comparison post-hoc test, Pearson correlations, and multivariable linear regressions. Multivariable linear regressions were used as explanatory models to evaluate the contributions of system inputs (H\(_2\)O\(_2\), Fe\(^{2+}\), and [H\(_2\)O\(_2\)]\(_{\text{initial}}/[\text{Fe}^{2+}]_{\text{generated}}\) and the impact of water quality parameters. Independent variables for the EC:H\(_2\)O\(_2\) process were selected based on Pearson correlations and normalized using the min-max method. This min-max normalization method was conducted to minimize the artificial impacts of independent variables on the dependent variable due to different scales and ranges of inputs (e.g., rate constants were on the order of \(10^{-3}\) s\(^{-1}\)), whereas H\(_2\)O\(_2\) ranged from 10 to 100 mg L\(^{-1}\).\(^{29}\) The independent variables for reactor inputs (pH, [H\(_2\)O\(_2\)]\(_{\text{initial}}/[\text{Fe}^{2+}]_{\text{generated}}\), and current density) were selected for the multivariable linear regression model based on their correlation to the dependent variables: pCBA removal, pseudo-first order rate constant, and \(E_{EO}\). For environmental waters, DOC\(_{\text{initial}}\), alkalinity, pH, and conductivity were selected as the independent water quality variables. All independent variables selected for multivariable linear regression were not multicollinear with other variables based on variance inflation factors < 5 for all regressions.\(^{29}\)

---

**Table 2** Water quality parameters

<table>
<thead>
<tr>
<th>Water matrix</th>
<th>Initial pH</th>
<th>H(_2)O(_2) demand,(^\text{a}) mg L(^{-1}) (% H(_2)O(_2) removal)</th>
<th>DOC, mg-C per L</th>
<th>Alkalinity, mg L(^{-1}) as CaCO(_3)</th>
<th>Conductivity, (\mu S) cm(^{-1})</th>
<th>Ca(^{2+}), mg L(^{-1})</th>
<th>Mg(^{2+}), mg L(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bicarbonate buffer</td>
<td>8.3(^{e})</td>
<td>0 (0%)</td>
<td>0.75(^{d})</td>
<td>210</td>
<td>370(^{d})</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Groundwater</td>
<td>7.30</td>
<td>10 (33%)</td>
<td>3.3</td>
<td>400</td>
<td>1430</td>
<td>70</td>
<td>40</td>
</tr>
<tr>
<td>River water</td>
<td>8.4</td>
<td>5 (15%)</td>
<td>7.0</td>
<td>240</td>
<td>755</td>
<td>30</td>
<td>20</td>
</tr>
<tr>
<td>Primary</td>
<td>7.1</td>
<td>23 (75%)</td>
<td>55</td>
<td>280</td>
<td>1400</td>
<td>40</td>
<td>15</td>
</tr>
</tbody>
</table>

\(\text{a}\) H\(_2\)O\(_2\) demand is reported as the decrease in H\(_2\)O\(_2\) concentration after 15 minutes (the length of batch experiments), where the initial concentration was 30 mg L\(^{-1}\) H\(_2\)O\(_2\). All model waters were prepared in Milli-Q water with 4 mM HCO\(_3\)\(^{-}\). \(\text{b}\) pH varied from 3 to 10.3 depending on the experiment. The unadjusted pH was 8.3. \(\text{d}\) For NOM tests, NOM was added as International Humic Substance Society Suwannee River NOM. \(\text{e}\) Conductivity varied for pH tests due to the addition of acid (HCl) or base (NaOH) for pH adjustment. At pH 6.3, conductivity = 450 \(\mu S\) cm\(^{-1}\). At pH 10.3, conductivity = 750 \(\mu S\) cm\(^{-1}\). For unadjusted pH, conductivity = 370 \(\mu S\) cm\(^{-1}\).
3. Results and discussion

3.1. Para-chlorobenzoic acid removal for hydroxyl radical validation

Removal of pCBA during EC:H2O2 primarily proceeded via oxidation at neutral pH conditions due to the system’s combination of iron and H2O2 (Fig. 1). EC-only controls yielded an average pCBA removal of approximately 15%, presumably due to the low levels of HO• that can be generated during EC alone.13,14 For EC:H2O2 + MeOH experiments, the high MeOH concentration (12 mM) scavenged the oxidants and resulted in negligible pCBA degradation. This scavenging indirectly underscores the role of homogeneous oxidants (such as HO•). Negligible pCBA removal in the EC:H2O2 + MeOH test further indicates that pCBA does not sorb to iron flocs. The ‘No Electricity Control’ experiments demonstrated that potential reactions between H2O2 and the iron electrode surface had minimal removal relative to the EC:H2O2 conditions with electricity (p < 0.0001, one-way ANOVA) at circumneutral pH conditions. Overall, these data demonstrate that the addition of H2O2 can enhance oxidant production in EC:H2O2 relative to EC alone and induce oxidative processes at neutral pH conditions.

The occurrence of oxidation at neutral pH conditions during EC:H2O2 is important in the context of Fenton literature since traditional Fenton oxidation proceeds at highly acidic pH 3 conditions. These conventional Fenton conditions limit the feasibility of EC:H2O2 applications as the high acidity can damage infrastructure, enhance corrosion, and incur chemical costs for acidifying and neutralizing water during treatment.

3.2. The impact of reactor inputs on pCBA degradation during EC:H2O2: removal and kinetics

Following oxidant verification, the impact of EC:H2O2 reactor inputs and water quality were assessed. The discussion centers on the role of [H2O2]initial/[Fe2+]generated ratios, current density, and pH. Multivariable linear regressions were used to parameterize the contribution of all inputs.

3.2.1. The impact of H2O2 dose, current density, and iron dose on pCBA removal at neutral pH conditions

The efficacy of H2O2 dose for pCBA removal varied as a function of the [H2O2]initial/[Fe2+]generated ratio (Fig. 2A). The presence of H2O2 only improved treatment when Fe2+ was also present in the system (REM, Fe2+.removal = 0.003, p = 0.99 Pearson correlation, Table S13†). With 10–40 mg L−1 H2O2, there was a positive correlation between pCBA removal and H2O2 dose during EC:H2O2 when iron was also present in the system (REM, Fe2+.removal = 0.84, p < 0.05 Pearson correlation, Table S14†). Once H2O2 exceeded 30 mg L−1 in the presence of Fe2+, pCBA removal began to plateau around 50–60% pCBA removal for [H2O2]initial/[Fe2+]generated ratios ranging from 0.3 to 0.7. In contrast, the higher [H2O2]initial/[Fe2+]generated ratio of 1.6 resulted in less pCBA removal compared to the same H2O2 dose applied at lower ratios. Alternately, for H2O2 concentrations greater than 40 mg L−1, the H2O2 dose did not significantly correlate (REM, Fe2+.40removal = -0.377, p = 0.136, Pearson correlation, Table S15†) and resulted in less pCBA removal. For example, 50 mg L−1 H2O2 had approximately 60% pCBA removal when applied at [H2O2]initial/[Fe2+]generated = 0.35; when the ratio increased to [H2O2]initial/[Fe2+]generated = 1.6, pCBA removal decreased to 40% for all H2O2 doses. The inhibition of pCBA removal at higher H2O2 levels aligns with the scavenging impact of H2O2 and competition between matrix constituents. Although more H2O2 can be beneficial for HO• generation via Fenton’s reaction, higher H2O2 levels lead to a higher degree of oxidant scavenging and decreased radical availability for pCBA removal (ESI S3).†

The key role of current density in this study was to adjust the iron loading rate to add Fe2+ as Fenton’s reagent (Fig. 2B). When no H2O2 was present (i.e., EC-only), pCBA removal was consistently less than 20% regardless of current density (Fig. 2C). Hence, EC offered effective pCBA removal only when H2O2 was present, indicating that the [H2O2]initial/[Fe2+]generated ratio was the key driver of treatment efficiency. During EC:H2O2, pCBA removal improved with increases in current density up to 7.4 mA cm−2 (Rdensity = 3 to 7.4 mA cm−2 = 0.63, p = 0.008, .

Fig. 1 Mechanisms for pCBA removal during EC:H2O2 at 7.40 mA cm−2. A series of controlled batch experiments were run in 4 mM bicarbonate buffer at pH 8.3 for 15 minutes. In ‘EC only,’ electrolysis was conducted using iron electrodes with no peroxide addition. For ‘EC:H2O2 + MeOH’, methanol was spiked in stoichiometric excess of pCBA (12 mM MeOH:2.5 μM pCBA) to quench oxidants that would otherwise degrade pCBA. In “No Electricity (H2O2 + electrodes)”, 30 mg L−1 H2O2 was spiked into the solution with the iron electrodes and mixed for 15 minutes. All experiments were conducted in duplicate and error bars indicate ±1 standard deviation. EC-only results are the average of all duplicate experiments for each EC-only control, including current densities of 3.5 mA cm−2, 5.5 mA cm−2, 11.1 mA cm−2, and 15 mA cm−2, where n = 8.
higher removal (\(\text{HO}_2\)) additional iron no longer improves treatment. Here, the lowest level of iron is needed for this system, and beyond that level, removal for higher current density may suggest that a minimum between \(\text{Fe}^{2+}\) and \(\text{H}_2\text{O}_2\). This excess is a result of \(\text{Fe}^{2+}\) being generated (estimated by Faraday’s law) over the course of the EC: \(\text{H}_2\text{O}_2\) experiment, and (c) current density, which is proportional to the iron experiments were conducted in duplicate and error bars indicate ±1 standard deviation.

In summary, \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) ratios were the key driver for \(\text{pCBA}\) removal where lower ratios (0.33–0.7) had higher removal (\(\beta_{\text{H}_2\text{O}_2/\text{Fe}^{0.0-0.7}=0.77, p<0.0001\), multivariable linear regression: “% \(\text{R}\), low ratio, neutral pH”) from minimal \(\text{HO}^-\) scavenging, and higher ratios (1.6) decreased removal (\(\beta_{\text{H}_2\text{O}_2/\text{Fe}^{0.3-1.6}=-0.42, p=0.0008\), multivariable linear regression: “% \(\text{R EC: H}_2\text{O}_2\) neutral pH”). This finding is important when considering material requirements including the \(\text{ex situ}\) \(\text{H}_2\text{O}_2\) additions and the power demands associated with iron generation. For this system, \(\text{H}_2\text{O}_2\) levels determined the treatment capacity because \(\text{pCBA}\) removal ceased after depletion of the one-time dose of \(\text{H}_2\text{O}_2\) at the start of the test, whereas the \(\text{Fe}^{3+}\) was continually generated via electrolysis.

It is important to note that \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) ratios do not translate to the actual ratio of \(\text{H}_2\text{O}_2\) relative to \(\text{Fe}^{2+}\) at any timepoint during the test. During EC: \(\text{H}_2\text{O}_2\), \(\text{H}_2\text{O}_2\) is initially in large excess to \(\text{Fe}^{2+}\) as \(\text{Fe}^{2+}\) is formed during EC, which may drive the rate of oxidant formation resulting from interactions between \(\text{Fe}^{2+}\) and \(\text{H}_2\text{O}_2\). This excess is a result of \(\text{Fe}^{2+}\) being generated at mN levels (e.g., 2500 nM s\(^{-1}\) for 7.4 mA cm\(^{-2}\) based on Faradays law) during electrolysis, which highlights the benefits of using iron electrolysis for \(\text{Fe}^{2+}\) dosing to avoid side reactions and encourage efficient \(\text{Fe}^{2+}\) utilization by \(\text{H}_2\text{O}_2\).

### 3.2.2. The impact of \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) and current density on \(\text{pCBA}\) oxidation rate during EC: \(\text{H}_2\text{O}_2\)

Pseudo-first order kinetic modeling offered good data fits, enabled comparison to other AOP processes in the literature, and was used in \(\text{E}_{\text{EC}}\) calculations. For a fixed current density of 5.5 mA cm\(^{-2}\), \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}=0.35, 0.5, \text{and } 0.7 \text{ had similar pseudo-first order rate constants (}1.1 \times 10^{-3} \text{ to } 1.3 \times 10^{-3} \text{ s}^{-1}\) before \(\text{H}_2\text{O}_2\) depletion (Fig. 3A and Table S5†). As the ratio increased, the rate of \(\text{pCBA}\) removal declined, which corroborates the removal findings in Section 3.2.1. Notably, for \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}=0.35 \text{ and } 0.5, \text{ pCBA removal stagnated after 7.5 minutes and 10 minutes, respectively. Accordingly, }\text{H}_2\text{O}_2\text{ should be continually dosed at lower concentrations in EC: }\text{H}_2\text{O}_2\text{ operations in order to continue oxidative reactions without adding excess }\text{H}_2\text{O}_2\text{ that can lead to quenching.}

As shown in Fig. 3, for a fixed \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}=0.5, \text{ the rate constants were comparable for current densities of }5.5, 7.4, and 11.1 \text{ mA cm}^{-2} (1.3 \text{ to } 1.6 \times 10^{-3} \text{ s}^{-1}, \text{ Table S5†}). \text{ However, 3 mA cm}^{-2} \text{ had the lowest rate of removal (}8.4 \times 10^{-4} \text{ s}^{-1}, \text{ Table S5†)}). This trend aligns with the removal data, in which the removal plateaued as current (i.e., iron loading rate) increased, indicating that additional iron after a threshold level no longer improved treatment.

Overall, the ratio had the highest influence on rate of removal based on multivariable linear regressions (\(\beta_{\text{H}_2\text{O}_2/\text{Fe}^{0.0-0.7}=0.77, p<0.0011, \text{ multivariable linear regression: “}k\text{, EC: }\text{H}_2\text{O}_2\text{, low ratio, neutral pH”)} (Fig. 3C)). Considering both major inputs in terms of \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) ratios, pseudo-first order rate constants were grouped into three clusters for a range of current densities at neutral pH conditions (Fig. 3C) to understand the general impact of different ratio levels. The clusters were EC-only conditions (i.e., no \(\text{H}_2\text{O}_2\), \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}=0.3-0.7\), and \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) ratios greater than 0.7. The lower \(\text{[H}_2\text{O}_2\text{]}_{\text{initial}}/\text{[Fe}^{2+}\text{]}_{\text{generated}}\) ratios of 0.3–0.7 had the highest rate constants, ranging from \(1.1 \times 10^{-3} \text{ to } 1.6 \times 10^{-3} \text{ s}^{-1}\). The higher ratios of >0.7 to 1.6 resulted in lower rate constants, ranging from \(5.8 \times 10^{-4} \text{ to } 7.7 \times 10^{-4} \text{ s}^{-1} (\beta_{\text{H}_2\text{O}_2/\text{Fe}^{0.3-1.6}=-0.59 \pm 0.11, p<0.0001, \text{ multivariable linear regression: “}k\text{, EC: }\text{H}_2\text{O}_2\text{, only, neutral pH”}). For all cases, experiments containing \(\text{H}_2\text{O}_2\) had faster rates of \(\text{pCBA}\) removal compared EC-only controls (\(1.4 \times 10^{-4} \to 2.7 \times 10^{-4} \text{ s}^{-1}\)). However, higher levels of \(\text{H}_2\text{O}_2\) (greater than 40 mg L\(^{-1}\))
did not increase the rate of pCBA removal, likely due to radical quenching by H$_2$O$_2$. This is notable given that the effective H$_2$O$_2$ doses found in this study are less than reagent demands in other EC:H$_2$O$_2$ studies for industrial treatment applications.$^{21-23}$ Overall, these findings indicate that less H$_2$O$_2$ may be required than previously thought for effective oxidation during EC:H$_2$O$_2$.

### 3.2.3. The impact of pH on oxidation rate.

The impact of pH on pCBA oxidation was assessed to evaluate the interplay of Fe$^{2+}$, H$_2$O$_2$, and pCBA over a range of acid/base conditions. As pH decreased, pCBA removal increased and the rate of oxidation accelerated (Fig. 4). At basic pH 10.3 conditions, minimal removal was observed. At pH 6.3, the maximum rate of pCBA degradation was observed (amongst the circumneutral pH levels tested), $k = 4.6 \times 10^{-3}$ s$^{-1}$ (1.6 times faster than the rate at pH 8.3). Removal of pCBA ceased after 5 minutes (as indicated by the stagnation of the kinetic curve), likely due to depletion of H$_2$O$_2$ at these conditions. Measurements of the H$_2$O$_2$ remaining after 5 minutes demonstrated roughly 70% loss of the initial 30 mg L$^{-1}$ H$_2$O$_2$ at pH 6.3 and 95% loss at pH 3 (ESI S7†). Depletion of the H$_2$O$_2$ helps explain the stagnated pCBA removal, likely due to decreased formation of oxidants. Accordingly, H$_2$O$_2$ should be continually dosed at lower concentrations in EC:H$_2$O$_2$ to encourage continuous oxidative reactions and improve TOC treatment.

The acidic conditions (pH 3, encouraging Fenton’s reactions) resulted in the greatest and fastest pCBA removal (>99% removal, to below the detectable limit). However, at pH 3, enhanced corrosivity led to >99% pCBA removal even without electricity, wherein increased iron dissolution was visually observed. For no electricity controls, pCBA removal was likely due to non-faradaic iron dissolving from the electrodes (30 mg Fe per L) and reacting with $ex$ $situ$ H$_2$O$_2$ (98% H$_2$O$_2$ removal; ESI S7†) to generate HO·. The no electricity control resulted in a $[H_2O_2]_{initial}/[Fe^{2+}]_{generated}$ ratio of 1.5. Although this ratio was toward the upper end of ratios tested, the pH 3 conditions were expected to enhance the rate of reaction. For circumneutral conditions, the no-electricity controls had minimal pCBA removal, indicating no oxidant generation in the absence of electricity at the conditions tested.

Overall, the pCBA removal trends agree with the kinetic modeling performed to estimate the competition between H$_2$O$_2$ and O$_2$ for oxidizing Fe$^{3+}$ (ESI S4†). The modeling scenarios included 0 to 200 mg L$^{-1}$ H$_2$O$_2$ concentrations. At pH 6.3, the rate of Fe$^{3+}$ oxidation by H$_2$O$_2$ was up to 10 orders of magnitude higher than Fe$^{2+}$ oxidation by O$_2$, suggesting that there was minimal competition for ferrous oxidation between H$_2$O$_2$ and O$_2$. Accordingly, these kinetic analyses support that HO· generation was driven by Fe$^{3+}$ oxidation via H$_2$O$_2$ (not O$_2$). Removal was minimal at pH 10.3, likely due to enhanced O$_2$ activity (Fig. 4). As pH increases, the inhibition of HO· generation due to O$_2$ becomes more apparent given that the oxidation of Fe$^{3+}$ by O$_2$ is second order with respect to [OH·] (based on Stumm and Lee, 1961) and increases 100-fold for each pH unit increase (ESI S4†).

The pseudo-first-order rate constants were used to estimate the HO· concentration (ESI S2†). For pH 8.3, [HO·] ranged from 2–4.1 × 10$^{-13}$ M for the $[H_2O_2]_{initial}/[Fe^{3+}]_{generated}$ ratios of 0.33 to 0.7. At pH 6.3, when pCBA treatment was more effective, [HO·] was approximately 9 × 10$^{-13}$ M. These estimates of radical concentrations can be applied to future studies to compare the [HO·] yield for a range of TOC oxidation technologies such as UV/H$_2$O$_2$.

### 3.2.4. Multivariable linear regression analysis of EC:H$_2$O$_2$ process inputs.

To evaluate the roles of independent variables, multivariable linear regressions were conducted to consider the influence of all reactor input experiments at neutral pH and for variable pH experiments. The key parameters incorporated into
the regression were \( \frac{[H_2O_2]_{initial}}{[Fe^{2+}]_{generated}} \) ratios, pH, and current density. These independent variables were selected based on preliminary Pearson correlations and were not multicollinear (ESI 56).

Overall, \( \frac{[H_2O_2]_{initial}}{[Fe^{2+}]_{generated}} \) ratios, pH, and current density were significantly correlated to pCBA removal (\( p = 0.028, 0.008, \) and <0.0001, respectively, multivariable regression “% R all”). The most influential parameter for pCBA removal was pH (\( \beta \text{pH} = -0.91 \pm 0.15 \), multivariable linear regression: “% R, all”), where lower pH led to higher pCBA removal. The ratio of \( \frac{[H_2O_2]_{initial}}{[Fe^{2+}]_{generated}} \) and current density had smaller impacts relative to pH, but similar magnitude of contributions to one another (\( \beta \text{pH} [H_2O_2]/[Fe^{2+}] = 0.22 \pm 0.09, \beta \text{current density} = 0.36 \pm 0.09, \) multivariable linear regression: “% R, all”).

A separate regression was performed for experiments with \( [H_2O_2]_{initial} \left/ [Fe^{2+}]_{generated} \right. \) ratios = 0–0.77 to rank the inputs that yielded higher rate constants and higher pCBA removal. For these tests, pH still had the greatest influence (\( \beta \text{pH} = -0.79 \pm 0.08, p < 0.0001 \), multivariable linear regression: “\( k \), all”) followed by \( [H_2O_2]_{initial}/[Fe^{2+}]_{generated} \) ratios (\( \beta [H_2O_2]/[Fe^{2+}] = 0.38 \pm 0.05, p < 0.0001 \), multivariable linear regression: “\( k \), all”). However, variations in current density alone had an insignificant influence on the rate of pCBA removal (\( \beta \text{current density} = 0.09 \pm 0.05, p = 0.073 \), multivariable linear regression: “\( k \), all”), implying that pH and \( [H_2O_2]_{initial}/[Fe^{2+}]_{generated} \) ratios are the key parameters influencing oxidant production.

3.3. Co-treatment of pCBA and NOM using EC:H₂O₂ to treat environmental waters and synthetic matrices

3.3.1. pCBA removal in NOM-containing waters. In environmental source waters (i.e., river water and groundwater), EC:H₂O₂ oxidized pCBA, indicating that EC:H₂O₂ can treat real waters containing relatively low DOC levels typical of natural source waters in addition to synthetic matrices (Fig. 5A). The matrices with the lowest levels of DOC (groundwater and bicarbonate [no DOC]) had similarly high pCBA removal (\( p = 0.8, \) ANOVA: post hoc Tukey multiple comparison), whereas the matrices containing moderate DOC levels (river water and SR-NOM) had less pCBA removal and performed similarly to one another (\( p > 0.99, \) ANOVA: post hoc Tukey multiple comparison). Notably, the synthetic matrices had similar removal efficacies to the real waters in spite of increased complexity in real water sources.

The initial concentration of DOC had a small impact on pCBA removal (\( \beta \text{DOC} = -0.07, p = 0.34 \), not including primary effluent) for matrices containing low-to mid-range DOC levels (<10 mg-C per L) that reflect drinking water source matrices. This trend implies that the presence of NOM may not heavily impede pCBA oxidation when treating typical environmental source waters.

Compared to real-world waters and synthetic matrices, the primary effluent had the least pCBA removal. Decreased removal was likely due to high H₂O₂ demand (Table 2) and high DOC levels. In this case, the high H₂O₂ demand rapidly depleted the H₂O₂ that was initially dosed into the reactor, which hindered HO⁻ production. Thus, for EC:H₂O₂ applications, the water’s H₂O₂ demand should be accounted for to gauge potential negative impacts on process performance. For example, Serra-Clusellas et al. (2021) demonstrated TOrC mitigation via EC:H₂O₂ at pH 3 in municipal tertiary treated wastewater containing ng L⁻¹ TOrCs by using elevated 220–440 mg L⁻¹ H₂O₂. mg L⁻¹ doses (resulting in \( [H_2O_2]_{initial}/[Fe^{2+}]_{generated} \) ratios of 1.7 to 2 during treatment), which offset oxidant scavenging by wastewater constituents.
A multivariable regression of all test matrices showed that DOC$_{\text{initial}}$ and pH were the key water quality parameters that impacted pCBA removal ($\beta_{\text{DOC}} = -0.72 \pm 0.18, p = 0.008$ and $\beta_{\text{pH}} = -0.81 \pm 0.08, p < 0.0001$, respectively). Pearson correlations showed that DOC$_{\text{initial}}$ and H$_2$O$_2$ demand were multicolinear ($R^{2}_{\text{DOC vs. H}_2\text{O}_2 \text{demand}} = 0.893, p < 0.05$, Pearson correlations). As anticipated, higher DOC levels typical of wastewater impeded treatment efficacy, whereas lower DOC conditions improved radical yield and offered less competition for oxidants. However, it is important to note that other water matrix constituents beyond DOC (including chemical oxygen demand, reduced metals, and sulfides, which were not assessed in this study) also likely contributed to H$_2$O$_2$ depletion and impeded DOC removal.

### 3.3.2. DOC removal in environmental waters.

In terms of bulk organics, EC:H$_2$O$_2$ appeared to offer similar levels of DOC removal compared to EC-only, with the added benefit of TORC mitigation based on pCBA removal (Fig. 5C). The favorable reproducibility of DOC removal via EC:H$_2$O$_2$ replicates relative to single EC-only as a point of reference suggests that DOC
removal primarily proceeds through non-destructive pathways as EC-only was previously shown to have minimal pCBA removal via oxidants at the conditions tested.

The river water and SR-NOM matrices are of particular interest for DOC removal given that they are representative of surface waters that could be treated for drinking water. Using EC:H2O2, DOC removal for the river water complied with recommendations in the US Environmental Protection Agency’s Enhanced Coagulation Guidance manual (>30% DOC removal for matrices containing >120 mg L⁻¹ as CaCO3 alkalinity). The synthetic SR-NOM matrix only had effective DOC removal when pH was 6.3. At pH 8.3, EC:H2O2 formed no flocs or precipitates in SR-NOM, indicating unsuccessful coagulation, precipitation, and subsequent sedimentation of flocs (ESI S9†). This difference between real and synthetic waters suggests that other constituents in environmental waters (such as divalent cations, i.e., calcium and magnesium) may improve coagulation processes in real waters by promoting ionic interactions between NOM and ions that promote co-sorption to flocs, as shown for a calcium-fulvic acid-goethite iron mineral system. Overall, the addition of H2O2 during EC:H2O2 can enhance treatment applications by simultaneously treating TOrCs such as pCBA as well as bulk organics such as DOC in a single unit process in lieu of a multi-stage treatment train such as coagulation/flocculation/sedimentation followed by filtration and oxidation to achieve both non-destructive removal and oxidative destruction of contaminants.

3.4. Engineering implications: rate constants and electrical energy per order

3.4.1. Pseudo-first order rate constants for treating environmental waters. Pseudo-first order reaction rates are key figures of merit for evaluating operational parameters by accounting for matrix-specific scavengers. The pseudo-first order rates for pCBA removal were 1.3 × 10⁻³ s⁻¹ and 1.6 × 10⁻³ s⁻¹ for river water and groundwater, respectively (Fig. 5B and Table 4). These values satisfy the proposed break-even point k = 2.1 × 10⁻⁵ s⁻¹ for TOrC treatment technologies to be competitive based on technoeconomic analyses.

3.4.2. Electrical energy per order: impact of reactor inputs assessed in bicarbonate buffer. In terms of energy requirements, higher current densities resulted in higher Eₖₒ values (p current density = -0.36 ± 0.09, p < 0.0001, multivariable regression: “Eₖₒ, all”), whereas [H₂O₂]initial/[Fe²⁺]generated ratios had less impact (β[H₂O₂]/[Fe²⁺] = -0.36 ± 0.09, p = 0.16). For the lower current densities, 3 and 5 mA cm⁻², the Eₖₒ was 0.62 ± 0.02 and 1.22 ± 0.05 kW h m⁻³, respectively, when operated at pH 8.3 conditions. The higher current densities of 7.4 to 15 mA cm⁻² had Eₖₒ values ranging from 3.13 ± 0.13 to 12.54 ± 2.12 kW h m⁻³ due to the additional electrical loading (Table 3). When pH decreased to 6.3, the Eₖₒ decreased from 2.86 ± 0.2 kW h m⁻³ to 0.9 ± 0.044 kW h m⁻³ for the same current density of 7.4 mA cm⁻². This improvement in energy efficiency was likely due to a combination of the faster rate of removal at pH 6.3 and the solution’s increased conductivity due to chloride addition (HCl was used for pH adjustment).

At circumneutral pH, current densities above 3 mA cm⁻² thus exceeded the recommended 1 kW h m⁻³ Eₖₒ threshold to be competitive with conventional HO₂-mediated advanced oxidation processes (AOPs) such as UV/H₂O₂ and ozone-based AOPs.⁷⁶ Accordingly, EC:H₂O₂ may be operated at lower current densities for more favorable energy demands. However, the benchmark Eₖₒ values for conventional AOPs rely on preliminary treatment technologies such as coagulation and membrane filtration to remove oxidant scavengers, primarily DOC. Additional DOC removal technologies add materials and energy demands to overall treatment of TOrCs that are not accounted for in standalone Eₖₒ values for conventional AOPs. Alternately, EC:H₂O₂ offers the benefit of simultaneous TOrC and DOC treatment, which can minimize preliminary treatment needs and decrease overall energy inputs compared to conventional AOP treatment trains.

Although EC:H₂O₂ was higher than the Eₖₒ benchmark for conventional AOPs under high current conditions, EC:H₂O₂ generally resulted in a lower range of Eₖₒ values (0.6 to 12.5 kW h kW h m⁻³ [Table 3]) compared to pCBA mitigation using other electrochemical technologies such as boron-doped diamond electrooxidation. For example, Lanzarini-Lopes et al. (2017)²⁷ reported Eₖₒ values for pCBA mitigation ranging from 39.3 kW h m⁻³ to 332 kW h m⁻³ for electro-oxidation current densities from 16.6 to 100 mA cm⁻². Consistent with this study, increasing current density in the electrochemical treatment process yielded higher, less favorable Eₖₒ values.

3.4.3. Electrical energy per order: impact of water quality. The Eₖₒ values for different water matrices ranged from 0.7 to 7.5 kW h m⁻³ (Table 4) as a function of water quality, pCBA removal, and the voltage input to achieve the fixed current of 7.4 mA cm⁻². Of the environmental waters, groundwater had the lowest Eₖₒ at 1.0 ± 0.13 kW h m⁻³, while the river water Eₖₒ was 1.91 ± 0.21 kW h m⁻³. The matrix with the highest Eₖₒ was SR-NOM (7.57 ± 0.20 kW h m⁻³) due to low pCBA

Table 3: Electrical energy per order of magnitude pCBA removal (kW h m⁻³) for EC:H₂O₂ operated in bicarbonate buffer. Values are the averages of duplicate experiments ± one standard deviation

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removal and low matrix conductivity. Primary effluent had the second highest $E_{EO}$ of 6.49 ± 1.34 kW h m$^{-3}$. Notably, primary effluent had the least pCBA removal of all waters tested (<20%); however, the water’s high conductivity led to low voltage input, leading to a relatively low $E_{EO}$ in spite of the poor pCBA treatment performance.

Multivariable regressions were used to assess how water quality in environmental and synthetic water matrices influenced $E_{EO}$. The $E_{EO}$ trends followed the removal trends, where DOC concentration and alkalinity increased $E_{EO}$ by decreasing pCBA removal and increasing treatment inputs ($\beta_{DOC} = 1.2 \pm 0.15$, $p < 0.0001$; $\beta_{alkalinity} = 0.45 \pm 0.11$, $p = 0.0013$, multivariable regression: “$E_{EO}$, water quality”). In terms of water quality parameters, DOC$_{initial}$ had the largest negative influence on $E_{EO}$. For parameters that improved $E_{EO}$, higher water matrix conductivity improved $E_{EO}$ by reducing the electrochemical cell’s power demands ($\beta_{conductivity} = -0.29 \pm 0.1$, $p < 0.0001$, multivariable regression: “$E_{EO}$, water quality” ESI S10†). Accordingly, groundwater required the lowest $E_{EO}$ of the environmental waters likely due to the low DOC concentration and high conductivity. The energy demands of the EC:H$_2$O$_2$ system operated at 7.4 mA cm$^{-2}$ were in the range of competitive performance (e.g., 1 kW h m$^{-3}$ according to Miklos et al. (2018)†) for several water matrices (Table 4), making EC:H$_2$O$_2$ a promising option for scaled applications for treating ToRCs in environmental waters such as groundwater and river water, with the added benefit of DOC removal in the same reactor.

### 3.5. Conclusions

The goal of this research was to evaluate EC:H$_2$O$_2$ as a combined destructive and non-destructive treatment technology at neutral pH. This performance was assessed as a function of reactor inputs and solution pH. The treatment efficacy of environmental source waters containing varying levels of NOM, scavengers, and ionic constituents was also evaluated. Neither current density nor H$_2$O$_2$ alone promoted pCBA oxidation, although the combination of these parameters heavily influenced performance. At neutral pH conditions, $[\text{H}_2\text{O}_2]_{\text{initial}}/\left[\text{Fe}^{2+}\right]_{\text{generated}}$ ratio was the key driver of oxidative performance, where ratios <0.7 had higher pCBA removal and higher ratios (0.7–1.6) decreased pCBA removal, likely due to H$_2$O$_2$ scavenging.

For water quality, pH was the key driver of improved removal, where lower pH conditions minimized the competition between H$_2$O$_2$ and O$_2$ for oxidation of Fe$^{2+}$ to better encourage radical generation. When treating groundwater and river water, EC:H$_2$O$_2$ had both oxidative treatment of ToRCs and non-destructive treatment of DOC. The pseudo-first order rate constants and $E_{EO}$ values demonstrated that EC:H$_2$O$_2$ can be competitive with other AOPs for ToRC treatment based on energy requirements and treatment performance (depending on current density and water quality, e.g., low DOC, high conductivity waters are easier to treat), with an added benefit of DOC removal due to coagulation and flocculation in the same reactor.

In real world treatment trains, EC:H$_2$O$_2$ could be operated to promote both oxidative and non-destructive treatments in a single process, which could replace multiple conventional unit processes. However, post-EC:H$_2$O$_2$ particle separation via sedimentation or rapid sand filtration would be needed to separate the iron flocs from solution. Additionally, a final disinfection step would likely be needed to ensure sufficient pathogen inactivation and maintain disinfectant residual. Future assessment of the performance of the full treatment train and the related energy efficiency can help to inform treatment train comparisons.

Future work is needed to evaluate EC:H$_2$O$_2$ treatment trains from a systems-engineering perspective wherein the additional benefits such as DOC removal and in situ chemical generation are parameterized to compare against the treatment costs associated with conventional treatment trains. These findings are needed to quantify the benefits of utilizing EC:H$_2$O$_2$ for combined treatment and provide a more comprehensive comparison of EC:H$_2$O$_2$ to current AOP technologies as well as conventional treatment trains. Additionally, the byproducts generated during EC:H$_2$O$_2$ should be evaluated. For example, the co-dissolution of regulated metals from iron electrodes (e.g., manganese) could add secondary contamination. Ex situ H$_2$O$_2$ addition is another challenge for decentralized EC:H$_2$O$_2$ treatment. Accordingly, research is needed to inform reactor setups for EC:H$_2$O$_2$ and to explore H$_2$O$_2$ dosing technologies such as air-diffusion cathodes that can promote in situ H$_2$O$_2$ generation, thereby decreasing ex situ chemical additions and enhancing the process’ potential as a small footprint decentralized treatment technology.

### Author contributions

<table>
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<tr>
<th>DRR: conceptualization, data curation, methodology, laboratory analyses, investigation, visualization, writing – original draft; writing – review &amp; editing. PJM: methodology, funding acquisition, project administration, supervision, resources, validation, visualization, writing – review &amp; editing. CKB: investigation, laboratory analysis, writing – original outline, writing – review &amp; editing. YW: funding acquisition, resources, writing – review &amp; editing. BKM: methodology, funding acquisition, resources, writing – review &amp; editing.</th>
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| Table 4 | Figures of merit for pCBA treatment in varying water matrices, including pseudo-first order rate constants ($k$) and electrical power per order ($E_{EO}$) of magnitude removal values for EC:H$_2$O$_2$. For all experiments, current density = 7.4 mA cm$^{-2}$ and H$_2$O$_2$ = 50 mg L$^{-1}$. pH 3 is not included due to insufficient data points to model a pseudo-first order rate constant prior to H$_2$O$_2$ depletion. pH 10.3 is not shown due to poor removal that did not provide viable data for pseudo-first order rate constants to estimate $E_{EO}$ values. | 1584 | Environ. Sci.: Adv., 2023, 2, 1574–1586 | © 2023 The Author(s). Published by the Royal Society of Chemistry |
| Water matrix | $k$, s$^{-1}$ | $E_{EO}$, kW h m$^{-3}$ |
| Bicarbonate buffer (pH 8.3) | 1.2 $\times$ 10$^{-3}$ | 2.86 ± 0.20 |
| Bicarbonate buffer (pH 6.3) | 4.7 $\times$ 10$^{-3}$ | 0.68 ± 0.005 |
| Bicarbonate buffer + NOM | 6.3 $\times$ 10$^{-3}$ | 7.57 ± 0.20 |
| Bicarbonate buffer + NOM (pH 6.3) | 2.9 $\times$ 10$^{-2}$ | 1.13 ± 0.08 |
| Groundwater | 1.6 $\times$ 10$^{-3}$ | 1.00 ± 0.13 |
| River water | 1.3 $\times$ 10$^{-3}$ | 1.91 ± 0.21 |
| Primary effluent | 2.5 $\times$ 10$^{-4}$ | 6.49 ± 1.34 |
acquisition, project administration, supervision, resources, validation, visualization, writing – review & editing.

Conflicts of interest

There are no conflicts of interest to declare.

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