



HAL
open science

Improvement of the biodegradability of diatrizoate by electroreduction of its amido groups

W. Zhang, I. Soutrel, Abdeltif Amrane, F. Fourcade, Florence Geneste

► **To cite this version:**

W. Zhang, I. Soutrel, Abdeltif Amrane, F. Fourcade, Florence Geneste. Improvement of the biodegradability of diatrizoate by electroreduction of its amido groups. *Separation and Purification Technology*, 2022, 285, pp.120317. 10.1016/j.seppur.2021.120317. hal-03556873

HAL Id: hal-03556873

<https://hal.science/hal-03556873>

Submitted on 4 Feb 2022

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

Improvement of the biodegradability of diatrizoate by electroreduction of its amido groups

Wei Zhang^{a,b}, Isabelle Soutrel^b, Abdeltif Amrane^b, Florence Fourcade^{b*}, Florence
Geneste^{a*}

^aUniv Rennes, CNRS, ISCR-UMR 6226, F-35000 Rennes, France

^bUniv Rennes, Ecole Nationale Supérieure de Chimie de Rennes, CNRS, ISCR-UMR
6226, F-35000 Rennes, France

Abstract

The occurrence of iodine X-ray contrast media (ICM) in the environment is a global concern since they can be transformed into toxic compounds during water disinfection process. Among the different approaches that have been tested to degrade them, electroreduction offers the advantages to be selective, targeting in a cost-effective way the functional groups responsible for ICM biorecalcitrance and allowing the recovery of iodide ions. We have previously shown that total deiodination of diatrizoate, a highly biorecalcitrant ICM, did not lead to a significant enhancement of its biodegradability. In this work, the electroreduction of amido groups of diatrizoate was attempted to further improve its biodegradability. Electrolyses of diatrizoate and its deiodinated derivative, were investigated in neutral and acidic medium. A total deiodination of diatrizoate was rapidly achieved in both media along with a slower

* Correspondence: Dr Florence Geneste, Florence.geneste@univ-rennes1.fr and Dr Florence Fourcade, Florence.fourcade@univ-rennes1.fr

reduction of the amido groups, as shown by LC-MS/MS analysis of by-products. The biodegradability of the solutions was estimated by the evaluation of the BOD₅/COD ratio. Results show that it was improved after the electroreduction performed in acidic medium. This result was confirmed by a biological treatment on activated sludge over a 21 days period, leading to mineralization yields of 46% and 68% for diatrizoate and 3,5-diacetamidobenzoic acid, respectively, after the electroreduction process.

Keywords. Coupling process, electroreduction, biological treatment, ICM, diatrizoate

1. Introduction

Iodine X-ray contrast media (ICM) are applied for medical intravascular imaging to enhance the visibility of X-ray pictures. Among them, diatrizoate (DTR) is one of the three most extensively used ionic contrast agent. Its structure is composed of three iodine atoms attached on a benzene ring, which is responsible for X-ray absorption, and two acetamide groups and an acetic acid to improve its solubility in water (Scheme 1) [1, 2]. The global consumption of ICM in the world reaches 3500 tons each year due to the high-dose administration (200 g) for a single examination. Owing to its high resistance to metabolism in the human body, diatrizoate, as most injected ICM, is excreted by patient through urines in 24 hours [3, 4]. Without a special treatment to separate them from the hospital wastewater, ICM are usually discharged into wastewater treatment plants (WWTPs). However, they can pass the WWTPs without substantial transformation and be discharged into the water system at a relatively high concentration [5-7]. Although their ecotoxicity has to be confirmed, it has been shown that they are transformed into toxic products during drinking water disinfection process [8]. Considering their environmental risk, it is thus urgent to

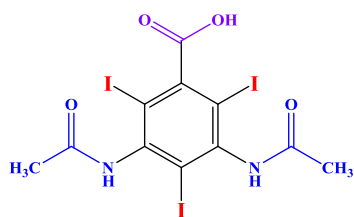
develop methods to remove ICM from the water system.

Various approaches have been attempted to treat ICM such as biological treatments, advanced oxidation processes (AOPs) and reductive deiodination process. The biological treatment of diatrizoate received less attention since it is a strong biorecalcitrant pollutant [9]. AOPs treatment of diatrizoate have been widely investigated and high degradation yields have been achieved. For example, a removal yield of 95% could be obtained by UV/chlorine treatment [10]. Even if the degradation of diatrizoate can be efficient, its mineralization can be difficult to reach due to its recalcitrance to hydroxyl radicals [11, 12]. Moreover, AOP mechanisms generate strong oxidation radicals that non-selectively attack diatrizoate, leading to toxic by-products such as dichloroacetonitrile, chloroform and iodoform [10-13]. Reductive deiodination of diatrizoate has been also investigated since it allows an improvement of its biodegradability and the possible recovery of iodide ions, which is particularly interesting to reduce its production cost [14, 15]. Furthermore, electrochemical reduction processes are particularly promising due to the decreasing cost of renewable electricity, while no extra chemicals is needed [16-18].

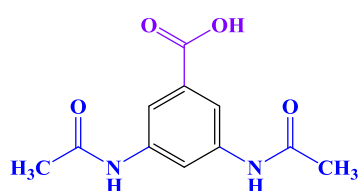
We previously reported that diatrizoate can be totally deiodinated by electroreduction on a graphite felt electrode in a flow cell [15]. The biodegradability of the resulting deiodinated derivative was improved compared to diatrizoate. However, when the electroreduced solution was treated with activated sludge in a laboratory-scale system, the mineralization was still not complete. We previously suspected *N*-aryl amide groups to reduce the biodegradability of deschloroalachlor, the dechlorinated derivative ofalachlor, a pesticide pollutant [19-22]. Regarding this possibility, we decided to investigate the selective electroreduction of the *N*-aryl amide groups of diatrizoate and the biodegradability of the resulting products.

Few articles focused on the electrocatalytic reduction of aromatic amides. Many metals such as lead, mercury, zinc, aluminum, bismuth, copper, nickel, iron, and cobalt have been tested [23, 24]. Among them, lead was highlighted as the best catalyst leading to good performances for amide reduction and having interesting intrinsic properties such as a flexible shape, a chemical inertness against corrosion, a low cost and a relatively high overpotential toward hydrogen evolution. Besides, it has been widely used as cathode for the electrocatalytic reduction of maleic acid into succinic acid [25], and of carbon dioxide into formate [26-28], for dechlorination [29], as well as for reductive deoxygenation of aromatic amides [23, 24, 30]. Thus, aryl amides have been reduced into their aryl amine derivatives in acidic medium with yields ranging from 39 to 63% [23, 24].

To the best of our knowledge, the electroreduction of aromatic amide to treat pollutants has never been attempted. In this regard, the present work aims to investigate the possibility of electroreduction of diatrizoate on lead cathode to enhance its biodegradability.



Diatrizoate (DTR) **1**



3,5-diacetamidobenzoic acid (DABA) **2**

Scheme 1. Structure of diatrizoate and 3,5-diacetamidobenzoic acid

2. Materials and Methods

2.1. Chemicals and Materials

All chemicals used in this work were of analytical grade. Anhydrous sodium sulfate (Na_2SO_4) 99%, sodium diatrizoate acid and 3,5-diacetamidobenzoic acid (DABA) were purchased from Sigma-Aldrich (France). Dimensionally stable anodes (DSA), AC-2004, were supplied by ECS International Electro Chemical Services, France.

2.2. Methods

2.2.1. Electroreduction process

The electroreduction reaction was performed in a cuboid glass cell with a cation exchanged membrane (NafionTM 417, France) dividing into anode and cathode compartments (Fig. 1).



Fig. 1. Electrochemical cell with separated compartments

The total empty volumes for cathodic and anodic compartments are 80 for each of them. Before the electrolysis, lead plate ($2\text{ cm} \times 5\text{ cm}$) was dipped in a solution

containing 30 % hydrogen peroxide and 70% acetic acid for 10 - 15 s to remove the oxide layer, and then rinsed with deionized water. DSA anode with a higher surface than lead was positioned parallel to the cathode. The distance between anode and cathode was fixed to 5 cm for all experiments. Saturated Calomel Electrode (SCE) as a reference electrode was closed to the lead cathode. The working electrode, counter electrode and reference electrode were connected through a potentiostat (VersaSTAT 3 from Elancourt, France). Samples were taken during electrolysis to analyze the released iodine ion concentration.

2.2.2. Analytical procedures

2.2.2.1. HPLC analysis

The residual diatrizoate and the concentration of 3,5-diacetamidobenzoic acid were measured by high performance liquid chromatography (HPLC). Samples were analyzed after filtering through a 0.2 μm membrane filter. The HPLC system consisted of a Waters 996 High Performance Liquid Chromatography equipped with a Waters 996 PDA (Photodiode Array Detector) and a Waters 600 LCD Pump. The separation was conducted on a column Waters C-18 (5 μm ; 4.6 \times 250 mm). Ten microliters of sample were injected with an auto-sampler. The carrier liquids, A composed of 95% of deionized water and 5% acetonitrile as well as 0.1% formic, and B composed of acetonitrile with 0.1% formic were gradient pumped at a flow rate of 0.4 mL min⁻¹. Diatrizoate and 3,5-diacetamidobenzoic acid were detected at a wavelength of 238 nm at a column temperature of 35°C.

2.2.2.2. *IC analysis*

The concentration of released iodide ions was measured by an ion chromatography (IC) system (DIONEX DX120 equipped with a conductivity detector and a DIONEX AS19 (4 × 250 mm) ion-exclusion column). The sample was eluted with potassium hydroxide at a flow rate of 1 mL min⁻¹. The detection was analyzed by conductivity with a Self-Regenerating Suppressor (SRS).

2.2.2.3. *UPLC-MS/MS analysis*

The degradation intermediates were identified by a LC-MS/MS ultra-high pressure liquid chromatography (Acquity UPLC-Waters) combined with a Waters Micromass Quattro Premier (Waters, Manchester UK) triple quadruple mass spectrometer as detector. The mass spectrometer functioned with an electrospray source in positive ionization mode with a cone potential of 30 V. The ionization source conditions were: capillary voltage of 3.0 kV and source temperature of 350°C. The cone and desolvation gas flows were 50 L h⁻¹ and 750 L h⁻¹ respectively. These values were obtained from a nitrogen source, the Peak Scientific NM30LA (Peak Scientific, Inchinnan, United Kingdom). High-purity argon (99.99 %, Air Liquid, Paris, France) was used as collision gas and was regulated at 0.1 mL min⁻¹. Analyses were conducted at full and daughter scan modes with a collision energy of 25 eV. Spectra were obtained ranging from 80 to 500 m/z and the data were analyzed by a Micromass Mass-Lynx 4.1 software. The UPLC system (Waters, Milford, MA, USA) consisted of an Acquity UPLC binary solvent manager, an Acquity UPLC sample manager as well as an Acquity UPLC column heater equipped with a Waters Acquity

UPLC BEH Shield RP18 column (2.1 mm × 100 mm, 1.7 mm particle size) (Milford, MA, USA) maintained at 45 °C. The mobile phase were 0.1% formic acid in acetonitrile as eluent A and 0.1% formic acid in a mixture of MilliQ Water/Acetonitrile (90/10, v/v) as eluent B, pumped at a flow rate of 0.4 mL min⁻¹.

2.2.2.4. Chemical oxygen demand (COD) measurements

The Chemical oxygen demand (COD) was determined by a Nanocolor Test CSB 160 from Macherey-Nagel (Germany). The samples were added in an acidic K₂Cr₂O₇ solution tube then heated at 160°C for 30 min. After cooling, the values were read at 436 nm. Each sample was duplicated and the average value was taken.

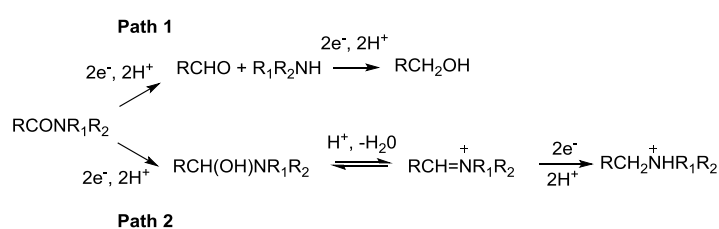
2.2.2.5. Biodegradability evaluation

The biodegradability was evaluated by the ratio of Biological Oxygen Demand in 5 days (BOD₅) to Chemical Oxygen Demand (COD), which was performed in an incubator box set at the temperature of 20°C. Volume sample was considered after the electrochemical processes, it was kept in the fridge (4°C). Activated sludge (AC) were taken from the Beaurade wastewater treatment plant (Rennes, France). Before treatment, activated sludge had to be washed 8-10 times to remove the dissolved organic matter. For BOD₅ test, samples were added in 500 mL brown bottles containing the culture medium with orbital shaking (300 rpm). The composition of the BOD₅ culture medium was: MgSO₄·7H₂O, CaCl₂, FeCl₃·6H₂O, NH₄Cl, KH₂PO₄, K₂HPO₄ at concentrations of 22.5, 27.5, 0.15, 2.0, 6.8, 2.8 g L⁻¹, respectively. The concentration of activated sludge was 0.05 g L⁻¹. All samples were adjusted to pH 7 before BOD₅ measurement and at least duplicated. Then bottles were closed with

sensor caps, which automatically recorded each hour the BOD₅ value. A carbon dioxide scrubber containing solid sodium hydroxide was fixed above the solution.

3. Results and Discussions

Amide compounds have been previously reduced in strong acidic conditions by a 2 electrons, 2 protons transfer into their amino derivative according to Scheme 2 (Path 2) [31].



Scheme 2. Electrochemical reduction of amides

A competitive reaction also occurs, leading to the formation of aldehyde and alcohol (Path 1). The selectivity of the reaction mainly depends on the structure of the amide compound and of the pH of the electrolytic medium [31], strong acidity favoring the cathodic cleavage (Path 1).

3.1. Cyclic voltammetry analysis

The electroactivity of diatrizoate DTR and its deiodinated derivative 5-diacetamidobenzoic acid (DABA) was first studied by cyclic voltammetry analysis in acidic and neutral media. A blank was first performed to study the electrochemical behavior of lead in these media (Fig. 2).

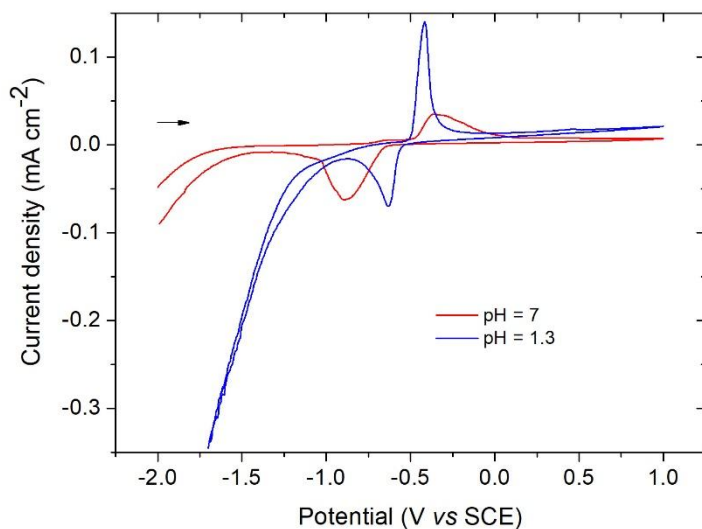


Fig. 2. Cyclic voltammetry on lead (1 mm^2) in $0.1 \text{ mol L}^{-1} \text{ Na}_2\text{SO}_4$ at pH 7 and at pH 1.3 ($1\% \text{ H}_2\text{SO}_4$). Scan rate: 100 mV s^{-1}

In acidic medium, an anodic peak was observed at -0.42 V/SCE , corresponding to the oxidation of metallic lead to PbSO_4 [32]. The resulting reduction peak at -0.63 V/SCE was due to the reduction of PbSO_4 into Pb. At pH=7, a large anodic peak appeared at -0.32 V/SCE , preceded by a small shoulder at around -0.64 V/SCE . This shoulder has been previously attributed to the formation of a thin layer of PbSO_4 on the electrode surface, whereas the large anodic peak has been assigned to the electroformation of $\text{Pb}(\text{OH})_2$ and/or PbO [33, 34]. The reduction peak at -0.88 V/SCE corresponded to the reduction of PbO and PbSO_4 into Pb. A small plateau attributed to the formation of lead hydride PbH_2 was observed just before hydrogen evolution.

0.38 mmol L^{-1} of diatrizoate was then added to the solution to test the catalytic activity of lead. A catalytic effect was observed in acidic medium (Fig. 3) with an onset at -1.44 V/SCE . Since deiodination of DTR occurs at relatively high potential [15], 0.38 mmol L^{-1} DABA was also added to check if other functional groups than carbon-iodine bound could be reduced on lead. An increase of the current was also

observed, showing that a reduction process also occurred on the deiodination compound.

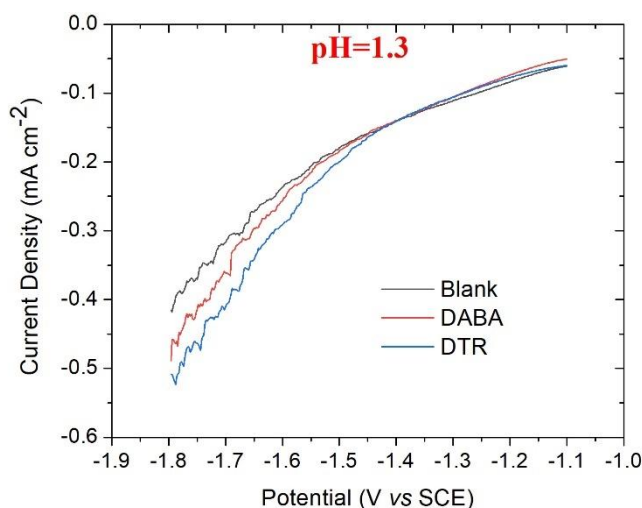


Fig. 3. Linear voltammetric scanning of 0.58 mmol L⁻¹ DTR and DABA in 0.1 mol L⁻¹ Na₂SO₄ with 1% H₂SO₄ (pH 1.3) on lead electrode (1 mm²). Scan rate: 100 mV s⁻¹

The same analysis performed at pH 7 did not show an increase of the current intensity in the presence of diatrizoate, even at a concentration of 1.2 mM. However, to facilitate a further biological treatment, the electroreduction of DTR was first tested at pH 7 at high current densities.

3.2. Electroreduction of diatrizoate and 3,5-diacetamidobenzoic in 0.1 mol L⁻¹ Na₂SO₄

3.2.1. Electroreduction of DTR

Electroreduction of 100 mg L⁻¹ DTR on lead cathode was first performed in 0.1 mol L⁻¹ Na₂SO₄ at a current density of -20 mA cm⁻². The concentrations of DTR and its

dehalogenated derivative, 3,5-diacetamidobenzoic acid DABA were followed by HPLC and the concentration of released iodide ions by linear sweep analysis with a silver rotating disc electrode (Fig. 4a).

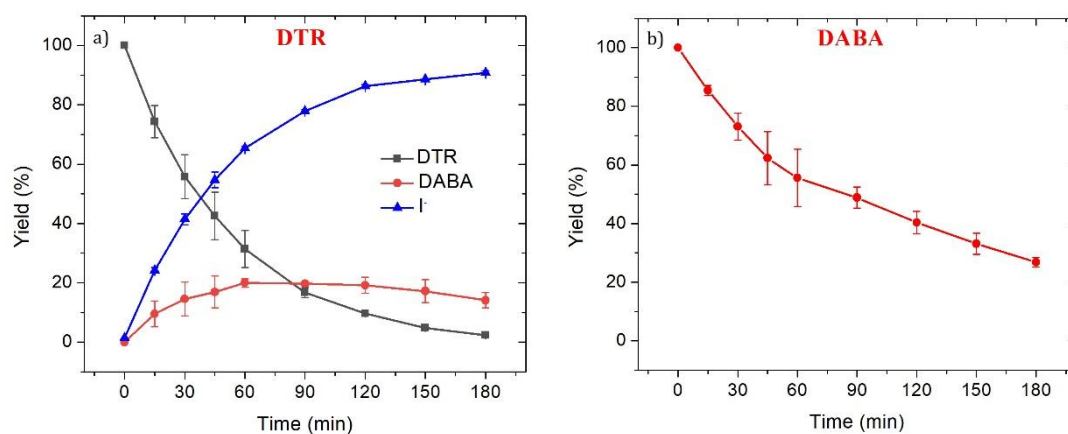


Fig. 4. a) Removal yield of DTR, formation yield of DABA and deiodination yield for electroreduction of 100 mg L^{-1} (0.16 mmol L^{-1}) DTR at -20 mA cm^{-2} in 0.1 mol L^{-1} Na_2SO_4 . b) Removal yield of DABA for electroreduction of 35 mg L^{-1} (0.15 mmol L^{-1}) DABA at -20 mA cm^{-2} in 0.1 mol L^{-1} Na_2SO_4 . Error bars are based on duplicated experiments.

A total removal (99.8%) of DTR was obtained after 3h of electroreduction with a dehalogenation yield of 91%. The yield of 3,5-diacetamidobenzoic acid increased up to 20% and then slowly decreased. These results showed that reductive deiodination of DTR is fast in these conditions, which is consistent with the total deiodination of DTR on graphite felt at -1.3 V/SCE [15]. However, since the concentration of the resulting dehalogenated compound decreased, another reduction process also occurred. To confirm these results, 35 mg L^{-1} 3,5-diacetamidobenzoic acid DABA was

electroreduced in the same conditions (Fig. 4b). A removal yield of 73 % was reached after 3h, confirming the reduction of other functional groups than the carbon-iodine bound in the DTR skeleton. For a better understanding of this reaction, the electroreduction of DABA was investigated in depth.

3.2.2. Electroreduction of DABA

3.2.2.1. Effect of current intensity on DABA electroreduction

The electroreduction of 35 mg L⁻¹ DABA on lead was performed in 0.1 mol L⁻¹ Na₂SO₄ at current densities of -10, -20 and -30 mA cm⁻² and was followed by HPLC monitoring (Fig. 5).

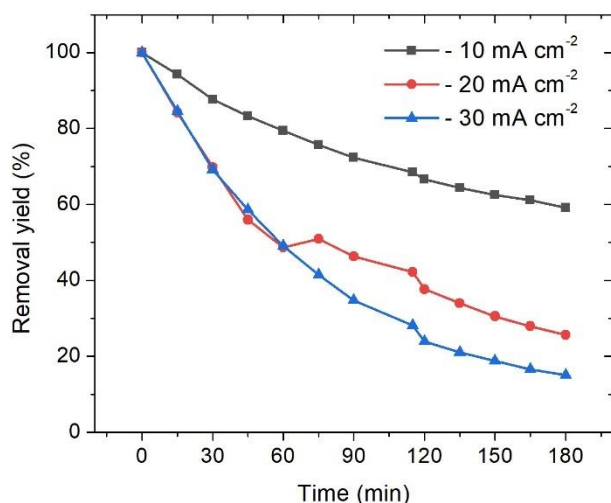


Fig. 5. Effect of current density on the removal yield of DABA for electroreduction of 35 mg L⁻¹ (0.15 mmol L⁻¹) DABA on lead cathode in 0.1 mol L⁻¹ Na₂SO₄ at -10, -20 and -30 mA cm⁻².

The removal yield of DABA increased when the current intensity increased, reaching 85% at -30 mA cm⁻² after 3h of electrolysis. HPLC analysis revealed the presence of

a new peak at 3.61 min (Fig. 6, Fig. SI).

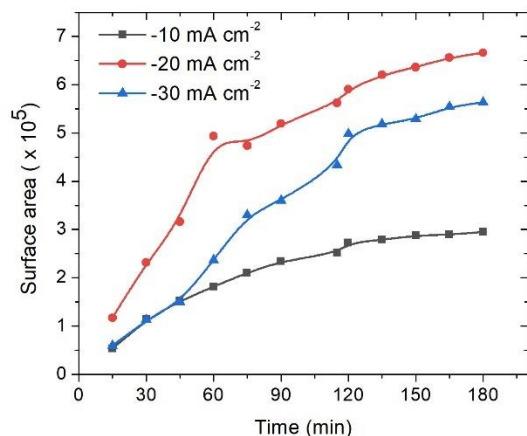


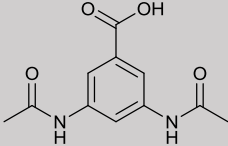
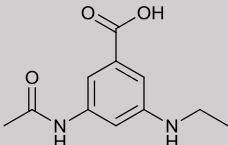
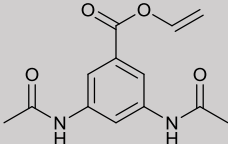
Fig. 6. Surface area of the peak at 3.61 min for electroreduction of 35 mg L⁻¹ (0.15 mmol L⁻¹) DABA on lead cathode in 0.1 mol L⁻¹ Na₂SO₄ at -10, -20 and -30 mA cm⁻².

The surface area of the peak increased with time for all studied current densities and the highest concentration was obtained for a current density of -20 mA cm⁻². Since the removal yields of DABA were nearly similar for -20 and -30 mA cm⁻² current density (74 % and 85%) and the surface area of the new peak was higher for -20 mA cm⁻², these conditions were selected for biodegradability estimation.

3.2.2.2. Identification of by-products

LC-MS/MS was implemented to identify the by-products formed during the electroreduction of DABA. To be sure that all intermediate compounds were present, the electroreduction on lead was performed at -10 mA cm⁻² in 0.1 mol L⁻¹ Na₂SO₄ for 1h. Four main parent compounds were identified (Table 1) with mass (MH⁺) of 237, 223, 180 and 263. To help the identification of the three main by-products, the fragmentations were also considered.

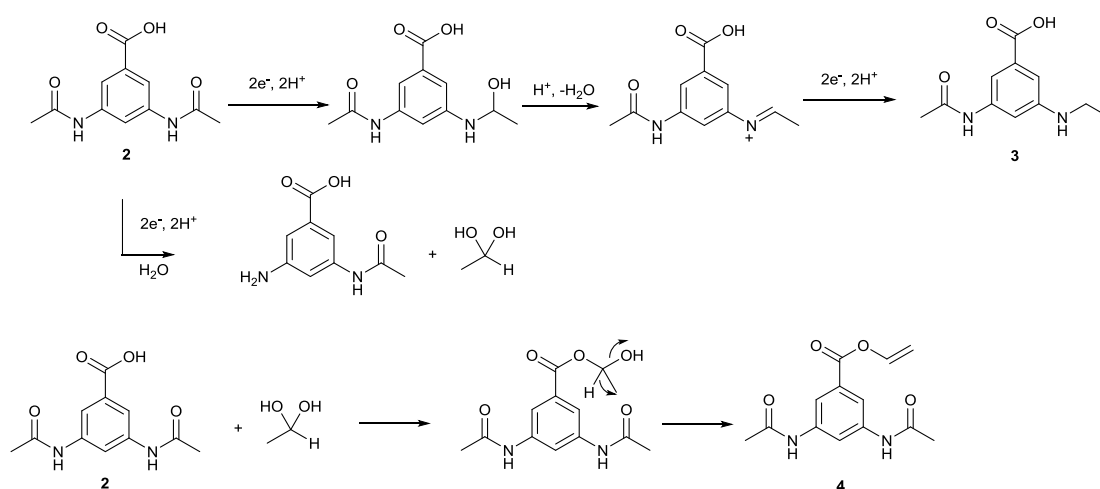
Table 1. LC-MS/MS identification of by-products for electroreduction of 35 mg L⁻¹ (0.15 mmol L⁻¹) at -10 mA cm⁻² in 0.1 mol L⁻¹ Na₂SO₄ for 1h. The structures were deduced from the m/z values and fragments.

Compound	time(min)	Fragments	m/z	Formula	Structure
3,5-DTR	3.27	219.09 191 176.96	237	C ₁₁ H ₁₂ N ₂ O ₄	 <p style="text-align: center;">2</p>
Product-1	2.97	205.06 162.94 120.96	223	C ₁₁ H ₁₄ N ₂ O ₃	 <p style="text-align: center;">3</p>
Product-2	2.84	161.87 137.9 133.81 93.75	180	C ₉ H ₉ NO ₃	-
Product-3	2.69	204.11	263	C ₁₄ H ₂₁ N ₃ O ₂	 <p style="text-align: center;">4</p>

The first parent peak corresponded to 3,5-diacetamidobenzoic acid **2**. The fragment ion at m/z 219 matched with a loss of a OH group and that at m/z 191 with a loss of a COOH group. The cleavage of an amide bond led to the fragment at m/z 177. The compound with a molecular mass (MH⁺) of 223 was identified as compound **3**, corresponding to a reduction of one amido group to an amine. The fragmentation was consistent with the structure of the parent compound. Indeed, the elimination of a OH

group gave a fragment ion m/z of 205, which could lead to the peak at m/z 163 after the cleavage of an amido group. The parent compound at m/z 180 could not be identified. The molecular mass of 263 could correspond to the compound **4**. Indeed, the fragment at m/z 204 could be due to the loss of an amido group.

According to the identification of the by-products, the following synthetic scheme 3 could be proposed:



Scheme 3. Proposed electroreduction pathway based on UPLC-MS/MS analysis.

The formation of **3** consisted in a two-electron reduction involving two protons (Scheme 1 Path 1) [31]. It is consistent with previous results, showing the electroreduction of amido groups on lead [23, 24, 30].

Compound **4** could result from the addition of the hydrated form of acetaldehyde to the carboxylate group of DTR followed by the elimination of water. Indeed, acetaldehyde can be formed by reductive cleavage of an amido group of DTR. Although aniline derivatives resulting from an amide cleavage (Scheme 1 Path 2) were not found, the formation of **4** seems to show that this reaction is in competition

with the electroreduction into amino groups.

3.3. Biodegradability evaluation of DTR and DABA after electroreduction in neutral medium

The biodegradability of DTR and DABA before and after electroreduction on lead cathode at -20 mA cm^{-2} for 3h was evaluated with the ratio of BOD_5 on COD (Fig. 7a). Indeed, a solution is considered as biodegradable if this ratio is higher than 0.4.

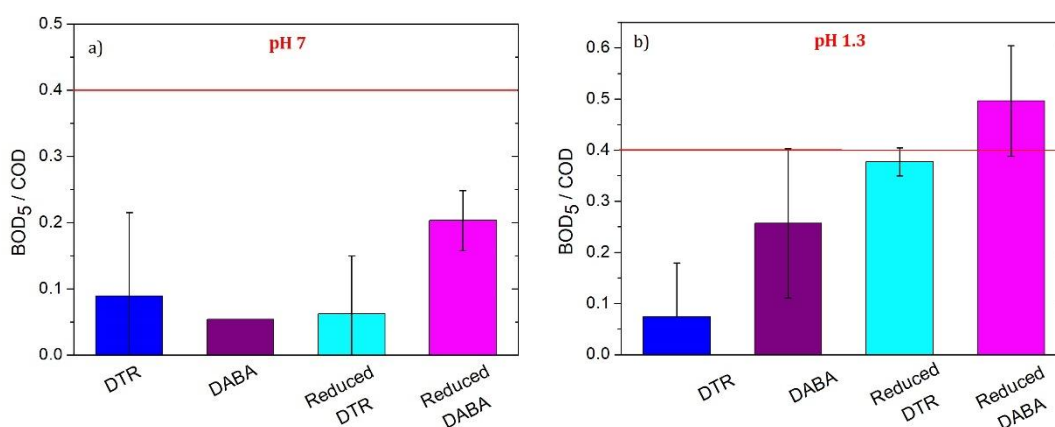


Fig. 7. BOD_5/COD ratio of a) 100 mg L^{-1} DTR and 35 mg L^{-1} DABA before and after electrochemical treatment (electroreduction on lead at -20 mA cm^{-2} in 0.1 mol L^{-1} Na_2SO_4 for 3h) b) 28 mg L^{-1} DTR and 10 mg L^{-1} DABA before and after electrochemical treatment (electroreduction on lead at -30 mA cm^{-2} in 0.1 mol L^{-1} $\text{Na}_2\text{SO}_4 + 1\% \text{ H}_2\text{SO}_4$ for 3h) Error bars are based on duplicate experiments.

As previously observed [15], at neutral pH, DTR exhibited a low BOD_5/COD ratio, showing that it is not biodegradable. Similar results were obtained for its deiodinated

derivative DABA and for the electrolyzed DTR solution. A slight increase was obtained for the electrolyzed DABA solution, although the BOD₅/COD ratio remained very low compared with the threshold value 0.4, showing that the solution is still not biodegradable when electrolyses were conducted at pH 7.

Since electroreduction of DABA was not total in the conditions of electroreduction and the biodegradability was not significantly improved by the electroreduction process, we decided to perform the same experiments in acidic conditions at lower concentrations to enhance the degradation of DABA. Indeed, a proton source should improve the electrocatalytic process and also prevent the oxidation of the lead electrode [23, 24] and a lower concentration can influence the solution toxicity and then its biodegradability [35].

3.4. Bulk electrolyses in 1% H₂SO₄ (pH 1.3)

The electroreduction of 28 mg L⁻¹ (46 μmol L⁻¹) diatrizoate (DTR) was performed in 1 % H₂SO₄ (pH 1.3) on lead cathode at current intensities fixed at -10, -20 and -30 mA cm⁻². The pH of 1.3 was a good compromise between the need to decrease the pH to improve the reaction and the ease of increasing it to pH 7 before further biological experiments. The concentrations of DTR and by-products were followed by HPLC (Fig. 8).

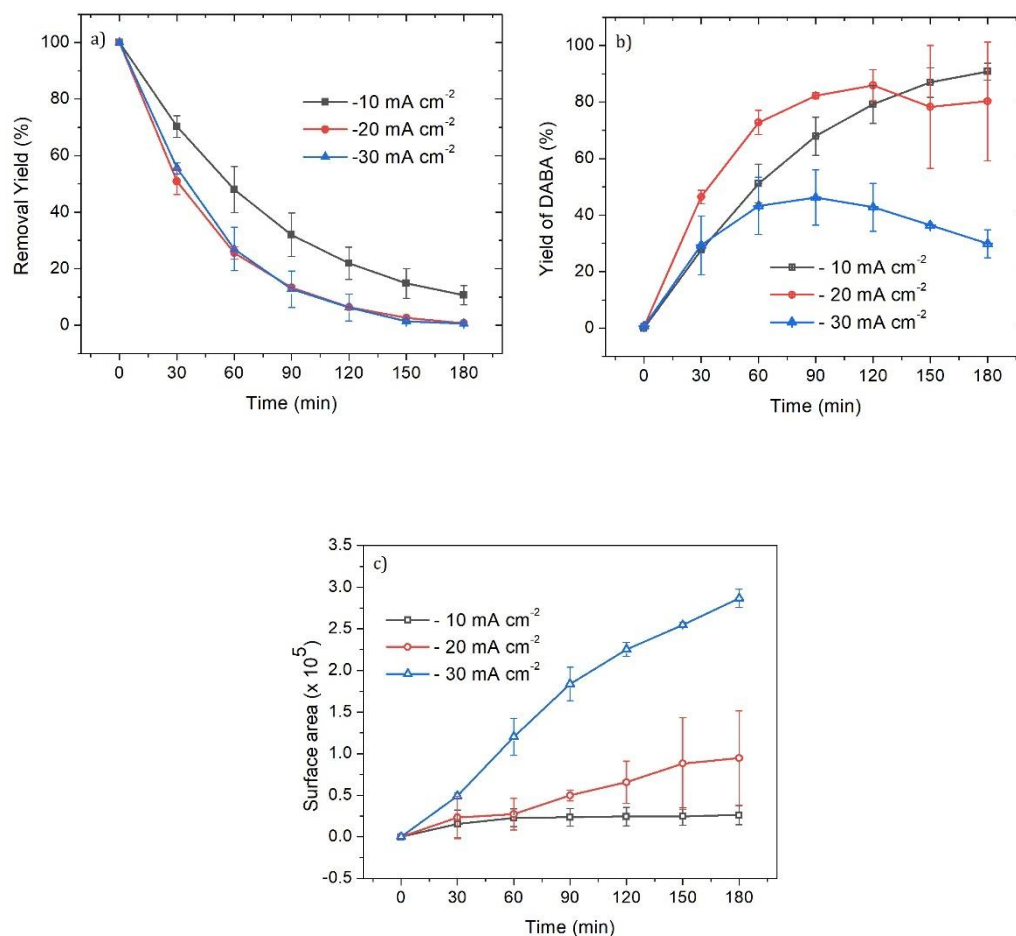


Fig. 8. a) Removal yield of DTR b) yield of DABA and c) surface area of the peak at 3.5 min for electroreduction of 28 mg L⁻¹ (46 μmol L⁻¹) DTR at -10, -20 and -30 mA cm⁻² in 0.1 mol L⁻¹ Na₂SO₄ + 1% H₂SO₄ (pH 1.3). Error bars are based on duplicated experiments.

DTR was totally reduced at -20 and -30 mA cm⁻² after 3h of electrolysis. HPLC analyses of DTR after electroreduction revealed the formation of two new peaks at 3.5 and 4.2 min. The peak at 4.2 min corresponded to 3,5-diacetamidobenzoic acid DABA, showing a reductive deiodination of DTR on lead cathode. It was selectively formed with a yield of 90% after 3h for a current density of -10 mA cm⁻². However,

for a current density of -30 mA cm^{-2} , the concentration of DABA increased until 90 min of electrolysis and decreased to reach a yield of 29% after 3h. Meanwhile, the peak at 3.5 min increased when the electrolysis was performed at -30 mA cm^{-2} , whereas it remained very small at -10 mA cm^{-2} . These results show that the reductive deiodination of DTR happened first, as for electrolyses performed at pH 7 and further reduction occurred when a higher current density was used.

To go further in the second electroreduction process of DTR, its dehalogenated derivative 10 ppm ($42 \mu\text{mol L}^{-1}$) 3,5-diacetamidobenzoic acid was electroreduced in the same conditions (Fig. 9). Its removal yield increased with the current density, reaching 85% for -30 mA cm^{-2} after 3h. The peak at 3.5 min was the only by-product detected by HPLC and corresponded to the same by-product observed by HPLC during DTR electroreduction. It increased with time for the highest current density. This result clearly showed that this main by-product with a retention time of 3.5 min did not contain iodine atom.

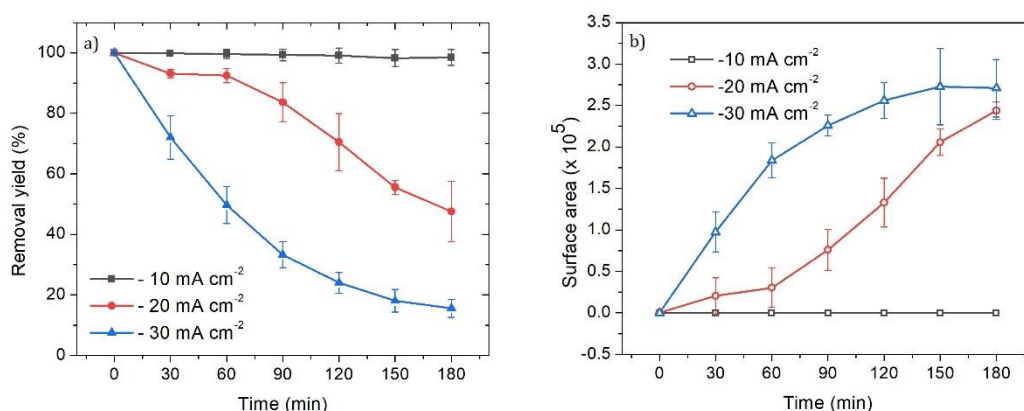


Fig. 9. a) Removal yield of DABA and b) surface area of the peak at 3.5 min for electroreduction of 10 mg L^{-1} ($42 \mu\text{mol L}^{-1}$) DABA at -10 , -20 and -30 mA cm^{-2} in $0.1 \text{ mol L}^{-1} \text{ Na}_2\text{SO}_4 + 1\% \text{ H}_2\text{SO}_4$ (pH 1.3). Error bars are based on duplicated

experiments.

The electroreduction of DABA in acidic medium at -30 mA cm^{-2} gave rise to a removal yield of 85% as in neutral medium, even if the initial concentration was 3.5 lower. The peak area of the main by-product was also lower (2.7×10^5 against 6.7×10^5) due to the lower initial concentration of DABA. Contrary to what was expected, these results seem to show that the efficiency of the electroreduction process was not significantly improved in acidic medium.

3.5. Biodegradability evaluation of DTR and DABA after electroreduction in acidic medium

Although the efficiency of the reduction of DABA was similar in neutral and acidic conditions, comparison of BOD₅/COD ratio in both media highlighted some differences (Fig 7b). The BOD₅/COD ratio remained low for DTR even if the concentration was divided by four but slightly increased for 10 mg L^{-1} DABA, although the solution was still not biodegradable. In contrast, an enhancement of the biodegradability was clearly obtained after electroreduction of DTR and DABA with BOD₅/COD ratios of 0.38 and 0.50, respectively. Since reduction processes in neutral and acidic media were equally effective, we attributed the higher biodegradability of the solution to the lower initial concentrations of DTR and DABA used for the electrolyses performed in acidic medium. Indeed, it has been previously shown that the toxicity and then the biodegradability can be linked to the concentration of the compounds in solution [35, 36]. Due to these encouraging results and since the values were close to 0.4, a biological treatment was performed to confirm the biodegradability of the electrolyzed solutions when lower concentrations of DTR and

DABA were used.

3.6. Biological treatment

A biological treatment of 28 mg L^{-1} ($46 \text{ } \mu\text{mol L}^{-1}$) DTR, 10 mg L^{-1} ($42 \text{ } \mu\text{mol L}^{-1}$) DABA and the corresponding electroreduced solutions was carried out with activated sludge over a period of 21 days. The biodegradation of the solutions were monitored by TOC measurement (Fig. 10).

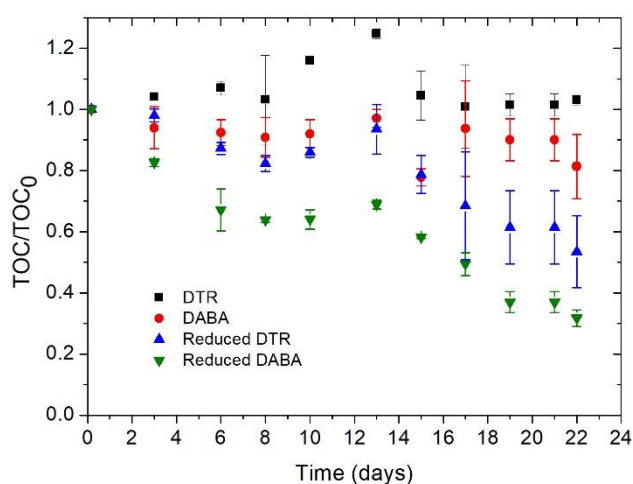


Fig. 10. Time-courses of TOC values during activated sludge culture on 28 mg L^{-1} DTR, 10 mg L^{-1} DABA solutions before and after electrochemical treatments (electroreduction on lead at -30 mA cm^{-2} in $0.1 \text{ mol L}^{-1} \text{ Na}_2\text{SO}_4 + 1\% \text{ H}_2\text{SO}_4$ for 3h). Error bars are based on duplicate experiments.

As previously observed [15, 37], DTR was not mineralized, confirming its biorecalcitrance. The TOC value of the deiodinated compound DABA was also

relatively stable, which underlines that a total deiodination of DTR is not enough to significantly improve the biodegradability of DTR [15]. After electroreduction, the mineralization yields rapidly increased, confirming the BOD₅/COD results and reached after 5 days around 10% and 40% for DTR and DABA, respectively. Then, they were stable until the 13th day and increased again to 46% and 68%, respectively after a 21 days culture period. The 8 days period of stability probably corresponded to an acclimatization step of microorganism to the residual by-products [38]. Interestingly, the biodegradation of by-products after electroreduction of DTR and DABA was significantly improved compared with the initial compounds, especially for electrolyses performed on DABA. These results show that the total removal of DABA, even if it does not correspond to a total reduction of both amido groups, was enough to significantly increase the biodegradability of the solution. Therefore, the simultaneous reduction of the carbon-iodide bounds with amide reduction is promising to achieve a total degradation of DTR.

4. Conclusions

The electroreduction of diatrizoate was performed at high current densities on lead electrode to reduce both the carbon-iodine bounds and the amido groups of the molecule for biodegradability improvement. A total deiodination of DTR was rapidly achieved along with slower electroreduction of amido groups. For a better understanding of the second reduction process, the electroreduction of the deiodinated derivative 3,5-diacetamidobenzoic acid (DABA) was also studied. A removal yield of 85% of DABA was achieved in neutral and acidic medium, although a 3.5 times lower initial concentration of DABA was used in acidic medium, showing that a higher

concentration of protons did not improve significantly the electroreduction process. The analysis of by-products by LC-MS/MS revealed that the amido group was reduced into a methylene amine but could also undergo an amide cleavage. The biodegradability estimated by the BOD₅/COD ratio was low for both DTR and DABA solutions but was improved after the electroreduction process performed in acidic medium. Such enhancement of the biodegradability was not observed after the electroreduction performed in neutral medium. This difference of behavior could be explained by the lower concentration of remaining DTR and DABA in the electrolytic medium. The enhancement of biodegradability by electroreduction was confirmed by the biological treatment on activated sludge that led to mineralization yields of 46% and 68%, for electroreduced DTR and DABA, respectively, after 21 days of incubation. These first results underline the interest of the selective electroreduction of amido groups to improve the biodegradability of ICM. Therefore, the coupling between a selective electrochemical process and a biological treatment is a promising approach to degrade ICM. Since lead is not beneficial to the environment although used as cathode in this electrochemical process, other electrode with high overpotential toward hydrogen evolution would be interesting to test for ICM wastewater treatment.

Acknowledgement

The authors thank the China Scholarship Council (CSC) for a Ph.D. grant.

References

- [1] S.-B. Yu, A.D. Watson, Metal-based X-ray contrast media, *Chem. Rev.*, 99 (1999) 2353-2378.
- [2] K.G. Estep, K.A. Josef, E.R. Bacon, C.R. Illig, J.L. Toner, D. Mishra, W.F. Blazak, D.M. Miller, D.K. Johnson, J.M. Allen, 1, 3, 5-Trialkyl-2, 4, 6-triiodobenzenes: Novel X-ray contrast agents for gastrointestinal imaging, *J. Med. Chem.*, 43 (2000) 1940-1948.
- [3] T. Steger-Hartmann, R. Länge, H. Schweinfurth, Environmental risk assessment for the widely used iodinated X-ray contrast agent iopromide (Ultravist), *Ecotoxicol. Environ. Saf.*, 42 (1999) 274-281.
- [4] S. Pérez, D. Barceló, Fate and occurrence of X-ray contrast media in the environment, *Anal. Bioanal. Chem.*, 387 (2007) 1235-1246.
- [5] W. Seitz, W.H. Weber, J.-Q. Jiang, B.J. Lloyd, M. Maier, D. Maier, W. Schulz, Monitoring of iodinated X-ray contrast media in surface water, *Chemosphere*, 64 (2006) 1318-1324.
- [6] M. Carballa, F. Omil, J.M. Lema, M.a. Llombart, C. García-Jares, I. Rodríguez, M. Gomez, T. Ternes, Behavior of pharmaceuticals, cosmetics and hormones in a sewage treatment plant, *Water Res.*, 38 (2004) 2918-2926.
- [7] J.E. Drewes, P. Fox, M. Jekel, Occurrence of iodinated X-ray contrast media in domestic effluents and their fate during indirect potable reuse, *J. Environ. Sci. Health., Part A*, 36 (2001) 1633-1645.
- [8] C.H. Jeong, E.J. Machek, M. Shakeri, S.E. Duirk, T.A. Ternes, S.D. Richardson, E.D. Wagner, M.J. Plewa, The impact of iodinated X-ray contrast agents on formation and toxicity of disinfection by-products in drinking water, *J. Environ. Sci.*, 58 (2017) 173-182.
- [9] W. Kalsch, Biodegradation of the iodinated X-ray contrast media diatrizoate and

iopromide, *Sci. Total Environ.*, 225 (1999) 143-153.

[10] C.-Y. Hu, Y.-Z. Hou, Y.-L. Lin, A.-P. Li, Y.-G. Deng, Degradation kinetics of diatrizoate during UV photolysis and UV/chlorination, *Chem. Eng. J.*, 360 (2019) 1003-1010.

[11] L. Zhou, C. Ferronato, J.-M. Chovelon, M. Sleiman, C. Richard, Investigations of diatrizoate degradation by photo-activated persulfate, *Chem. Eng. J.*, 311 (2017) 28-36.

[12] M.N. Sugihara, D. Moeller, T. Paul, T.J. Strathmann, TiO₂-photocatalyzed transformation of the recalcitrant X-ray contrast agent diatrizoate, *Appl. Catal., B*, 129 (2013) 114-122.

[13] C.-Y. Hu, Y.-Z. Hou, Y.-L. Lin, A.-P. Li, Y.-G. Deng, Degradation kinetics of diatrizoate during UV photolysis and UV/chlorination, *Chem. Eng. J.*, 360 (2019) 1003-1010.

[14] J. Radjenovic, V. Flexer, B.C. Donose, D.L. Sedlak, J. Keller, Removal of the X-ray contrast media diatrizoate by electrochemical reduction and oxidation, *Environ. Sci. Technol.*, 47 (2013) 13686-13694.

[15] W. Zhang, I. Soutrel, A. Amrane, F. Fourcade, F. Geneste, Electrochemical Processes Coupled to a Biological Treatment for the Removal of Iodinated X-ray Contrast Media Compounds, *Front. Chem.*, (2020) in press.

[16] N. Singh, B.R. Goldsmith, Role of Electrocatalysis in the Remediation of Water Pollutants, *ACS Catal.*, 10 (2020) 3365-3371.

[17] F. Geneste, Catalytic electrochemical pre-treatment for the degradation of persistent organic pollutants, *Curr. Opin. Electrochem.*, 11 (2018) 19-24.

[18] W. Zhang, I. Soutrel, A. Amrane, F. Fourcade, F. Geneste, Electro-reductive deiodination of iohexol catalyzed by vitamin B12 and biodegradability investigation,

- J. Electroanal. Chem., 897 (2021) 115559.
- [19] W. He, J.-M. Fontmorin, P. Hapiot, I. Soutrel, D. Floner, F. Fourcade, A. Amrane, F. Geneste, A new bipyridyl cobalt complex for reductive dechlorination of pesticides, *Electrochim. Acta*, 207 (2016) 313-320.
- [20] W. He, J.-M. Fontmorin, I. Soutrel, D. Floner, F. Fourcade, A. Amrane, F. Geneste, Reductive dechlorination of a chloroacetanilide herbicide in water by a Co complex-supported catalyst, *Mol. Catal.*, 432 (2017) 8-14.
- [21] W. He, Y. Lou, E. Verlato, I. Soutrel, D. Floner, F. Fourcade, A. Amrane, M. Musiani, F. Geneste, Reductive dehalogenation of a chloroacetanilide herbicide in a flow electrochemical cell fitted with Ag- modified Ni foams, *J. Chem. Technol. Biotechnol.*, 93 (2018) 1572-1578.
- [22] Y.-Y. Lou, F. Geneste, I. Soutrel, A. Amrane, F. Fourcade, Alachlor dechlorination prior to an electro-Fenton process: Influence on the biodegradability of the treated solution, *Sep. Purif. Technol.*, 232 (2020) 115936.
- [23] S. Swann Jr, The Electrolytic Reduction of Amides: I. N, N- Dimethylvaleramide and Acetanilide, *Trans. Electrochem. Soc.* , 84 (1943) 165.
- [24] S. Swann Jr, J. Bresee, R. Strehlow, Electrolytic Reduction of Amides: II. p- Aminobenzanilide and Anisamide, *J. Electrochem. Soc.*, 113 (1966) 513.
- [25] S. Vaghela, G. Ramachandraiah, P. Ghosh, D. Vasudevan, Electrolytic synthesis of succinic acid in a flow reactor with solid polymer electrolyte membrane, *J. Appl. Electrochem.*, 32 (2002) 1189-1192.
- [26] M. Alvarez-Guerra, A. Del Castillo, A. Irabien, Continuous electrochemical reduction of carbon dioxide into formate using a tin cathode: Comparison with lead cathode, *Chem. Eng. Res. Des.*, 92 (2014) 692-701.
- [27] B. Innocent, D. Liaigre, D. Pasquier, F. Ropital, J.-M. Léger, K. Kokoh, Electro-

reduction of carbon dioxide to formate on lead electrode in aqueous medium, *J. Appl. Electrochem.*, 39 (2009) 227.

[28] S.Y. Choi, S.K. Jeong, H.J. Kim, I.-H. Baek, K.T. Park, Electrochemical reduction of carbon dioxide to formate on tin–lead alloys, *ACS Sustain. Chem. Eng.*, 4 (2016) 1311-1318.

[29] S. Zhang, J.F. Rusling, Dechlorination of polychlorinated biphenyls by electrochemical catalysis in a bicontinuous microemulsion, *Environ. Sci. Technol.*, 27 (1993) 1375-1380.

[30] C. Edinger, S.R. Waldvogel, Electrochemical deoxygenation of aromatic amides and sulfoxides, *Eur. J. Org. Chem.*, 2014 (2014) 5144-5148.

[31] H. Lund, O. Hammerich, *Organic Electrochemistry*, Marcel Dekker, New York, 2001.

[32] M. Taguchi, H. Sugita, Analysis for electrolytic oxidation and reduction of PbSO₄/Pb electrode by electrochemical QCM technique, *J. Power Sources*, 109 (2002) 294-300.

[33] E.E. Abd El Aal, Cyclic voltammetric behavior of the lead electrode in sodium sulfate solutions, *J. Power Sources*, 102 (2001) 233-241.

[34] E.F. El-Sherbini, S.S. Abd El Rehim, Electrochemical behaviour of lead electrode in Na₂SO₄ solutions, *Mater. Chem. Phys.*, 88 (2004) 17-22.

[35] N. Prado, C. Monteleon, J. Ochoa, A. Amrane, Evaluation of the toxicity of veterinary antibiotics on activated sludge using modified Sturm tests - application to tetracycline and tylosine antibiotics, *J. Chem. Technol. Biotechnol.*, 85 (2010) 471-477.

[36] S. Yahiat, F. Fourcade, S. Brosillon, A. Amrane, Removal of antibiotics by an integrated process coupling photocatalysis and biological treatment - Case of

tetracycline and tylosin, *Int. Biodeterior. Biodegrad.*, 65 (2011) 997-1003.

[37] A. Haiß, K. Kümmerer, Biodegradability of the X-ray contrast compound diatrizoic acid, identification of aerobic degradation products and effects against sewage sludge micro-organisms, *Chemosphere*, 62 (2006) 294-302.

[38] N.A. Salles, F. Fourcade, F. Geneste, D. Floner, A. Amrane, Relevance of an electrochemical process prior to a biological treatment for the removal of an organophosphorous pesticide, phosmet, *J. Hazard. Mater.*, 181 (2010) 617-623.