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Technical Note

Ferrate(VI) oxidation of propranolol: Kinetics and products

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HIGHLIGHTS

- ▶ Reaction kinetics of Fe(VI) and propranolol is similar to secondary amines.
- ▶ Complete transformation of propranolol by Fe(VI) in water.
- ▶ Opening of aromatic ring of propranolol by Fe(VI).
- ▶ Hydroxylation of amine moiety of propranolol by Fe(VI).

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ABSTRACT

The oxidation of propranolol (PPL), a β-blocker by ferrate(VI) (Fe(VI)) was studied by performing kinetics, stoichiometry, and analysis of the reaction products. The rate law for the oxidation of PPL by Fe(VI) was first-order with respect to each reactant. The dependence of second-order rate constants of the reaction of Fe(VI) and PPL on pH was explained using acid–base equilibrium of Fe(VI) and PPL. The required molar stoichiometry for the complete removal of PPL was determined to be 6:1 ([Fe(VI)]:[PPL]). The identified products using liquid chromatography–tandem mass spectrometry were oxidized product (OP)-292, OP-308, and OP-282. The formed OPs could possibly compete with the parent molecule to react with Fe(VI) and thus resulted in a non-linear relationship between degradation of PPL and the added amount of Fe(VI). Rate and removal studies indicate the Fe(VI) is able to oxidize PPL and hence can also oxidize other β-blockers, e.g., atenolol and metoprolol.

1. Introduction

Emerging pollutants such as pharmaceuticals in the aquatic environment are of increasing concern to the ecosystem sustainability. Sources of pharmaceuticals in the surface water include industry, hospital, and domestic wastewaters containing unused or human excreted drugs (Deblonde et al., 2011). β-Blockers (β-andrenergic receptor antagonists) are among the important class of pharmaceuticals, which have been found at low concentrations up to μg L^{-1} in surface waters (Benotti et al., 2009; Martínez Bueno et al., 2012). The levels of β-blockers are expected to increase with the increase of the world population and its age and high concentrations might be ecotoxicologically harmful. Some studies have shown effect of β-blockers on the aquatic organisms (Huggett et al., 2002; Küster et al., 2010). Mixtures of β-blockers have additive effects on the organisms and thus even low concentrations may contribute to the toxicity. A β-blocker, propranolol (1-(1-methylethylamino)-3-(1-naphthyloxy)propan-2-ol, PPL), which has significantly higher toxicity to aquatic organisms compared to other β-blockers, atenolol and metoprolol (Fent et al., 2006), is the focus of the present study.

Beta-blockers are not completely removed by conventional treatment processes in a wastewater treatment plant (WWTP) due to their recalcitrant nature (Gabet-Giraud et al., 2010). For example, PPL was removed about 20% by WWTP (Gros et al., 2010). On the other hand, no evidence for biotransformation or biologically mediated removal of PPL was observed while a very slow abiotic process removing 20% of PPL over 79 d (Barbieria et al., 2012). Alternate treatment methods such as advanced oxidation processes have been tested to remove β-blockers in water (Benitez et al., 2011; Isarain-Chávez et al., 2011; Chen et al., 2012; Köhler et al., 2012; Sirés and Brillas, 2012). Treatment of PPL has been studied by photo-catalytic oxidation, ozonation, UV/O_3, UV/H_2O_2, and O_3/H_2O_2, and electrochemically (Andreozzi et al., 2004; Benner et al., 2008; Piram et al., 2008; Sharma, 2008; Benner and Ternes, 2009a,b; Kim et al., 2009;

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Sirés et al., 2010; Romero et al., 2011). The present paper reports the oxidative degradation of PPL by environmentally friendly oxidant, ferrate(VI) ($Fe^{VI}O_4^{2-}$, Fe(VI)).

In the last decade, several studies have shown potential of Fe(VI) as oxidant, coagulant, and disinfectant in treating water and wastewater (Jiang, 2007; Li et al., 2008; Lee et al., 2009; Lee and von Gunten, 2010; Sharma, 2010, 2011, 2013; Makky et al., 2011; Sharma et al., 2011; Jiang et al., 2012; Filip et al., 2011). However, only handful studies have been performed on the kinetics, stoichiometry, and products of the oxidation of pharmaceuticals by Fe(VI) (Sharma et al., 2006a,b; Hu et al., 2009; Anquandah et al., 2011; Zimmermann et al., 2012). The kinetics and removal study on the β-blockers is limited only to atenolol, which is also at one pH (i.e., pH 8.0) (Lee and von Gunten, 2010). The objectives of the current paper are to: (i) determine the kinetics of the reaction of Fe(VI) and PPL as a function of pH, and (ii) determine stoichiometry and identify oxidized products (OPs) of the reaction between Fe(VI) and PPL.

2. Experimental

2.1. Reagents

Propranolol, sodium acetate, sodium borate, ammonium acetate, and sodium hydrogen phosphate were purchased either from Sigma–Aldrich (St. Louise, MO, USA) or Fisher-Scientific (Austin, TX, USA) with purity higher than 97%. HPLC-grade solvents of acetonitrile and methanol were obtained from Fisher-Scientific. Solid potassium ferrate (K_2FeO_4) of ~98% purity used in the experiments was synthesized by the wet method (Luo et al., 2011). Fe(VI) solutions were prepared by addition of solid K_2FeO_4 to 1 mM $Na_2B_4O_7 \cdot 10H_2O$/5 mM Na_2HPO_4 at pH 9.0. Concentrations of Fe(VI) in the solution were determined spectroscopically at a wavelength of 510 nm using an UV–Vis spectrophotometer (Agilent 8453, Santa Clara, CA, USA). A molar absorption coefficient, $\varepsilon_{510nm} = 1150$ M^{-1} cm^{-1} was used to determine Fe(VI) concentration at pH 9.0 (Luo et al., 2011). All solutions were prepared using doubly distilled water that had been passed through an 18 MΩ cm Milli-Q (Millipore, Waters Alliance, Milford, MA, USA) water purification system. Stock solutions of PPL were prepared by dissolving the solid compound in the 10 mM Na_2HPO_4 buffer solution. The solution pH was adjusted to the desired pH by adding either NaOH or phosphoric acid. Spectral measurements of the solutions to determine the concentration of Fe(VI) were conducted on the UV–Vis spectrophotometer.

2.2. Kinetic studies

A stopped-flow spectrophotometer (SX-18 MV, Applied Photophysics, Surrey, UK) with a photomultiplier detector was used for the kinetic studies in which the PPL was in excess. Kinetic traces were collected at a wavelength of 510 nm to determine the pseudo-first-order rate constants. Data collected from the stopped-flow spectrophotometer were analyzed using the nonlinear least-square algorithm of the SX-18MV Global Software (Applied Photophysics, Surrey, UK). Rate constants obtained represent the average of six replicate runs. The pseudo-first-order rate constants for the reaction were obtained by subtracting the observed rate constants from the rate constants for self-decomposition of Fe(VI) at each studied pH (Sharma, 2010).

2.3. Stoichiometry and products studies

The stoichiometry experiments for the oxidation of PPL by Fe(VI) oxidation were carried out by mixing equal solution volumes of 10 mL and the reaction mixtures were maintained at pH 9.0. The concentration of PPL were kept at 0.1 mM and the concentrations of Fe(VI) varied from 0.1 to 0.6 mM. The Fe(VI) concentration was monitored until no residual Fe(VI) was detectable. The solutions were filtered using 0.45 μm nylon filters into HPLC vials. The concentration of the PPL in the resulting reaction mixtures was quantified by the use of an HPLC (Waters Alliance 2695, Milford, MA, USA) with a RP-C_{18} column (5 μm; 250 × 10 mm, Alltima, Deerfield, IL, USA) at a wavelength of 292 nm. A binary mobile phase consisting of 35% Solvent A (20 mM ammonium acetate) and 65% Solvent B (acetonitrile) at a flow rate of 0.5 mL min^{-1} and injection volume of 70 μL were used in an isocratic elution mode.

The OPs from the reaction of Fe(VI) with PPL were analyzed by using LC in combination with tandem MS (LC–MS/MS). The concentration of PPL was also analyzed using LC–MS/MS. The instrumentation consists of an Accela UPLC pump and a HTC-PAL autosampler LC system attached to TSQ Quantum Access QqQ Mass Spectrometer (Thermo Scientific, San Jose, CA, USA). Chromatographic separation of PPL and its transformation products was achieved on a Hypersil Gold column C18 (1.9 μm; 50 mm × 2.1 mm, Thermo Scientific, San Jose, CA, USA) using a binary gradient mobile phase of acetonitrile and 0.1% formic acid in water. Mobile phase gradient cycle is shown in Supplemental material (SM), Table SM-1. For the ionization of analyte molecules, an Ion Max API Heated Electrospray Ionization source operated in the positive mode with the capillary temperature of 350 °C, vaporizer temperature of 400 °C and a spray voltage of 3500 V was used. Data were acquired in the full scan mode (40–350 m/z) for the identification of PLL oxidation products and MS/MS information on the identified products was obtained in a secondary analysis by isolating the precursor ions detected in the full scan run and monitoring for all product ions. Quantitative measurements were done in selected reaction monitoring mode with three specific transitions. The optimized parameters for mass analysis are given in Table SM-2.

3. Results and discussion

3.1. Kinetics

Initially, experiments were carried out at pH 8.1 and 25 °C to determine the rate law of the reaction between Fe(VI) and PPL. The rate expression for the reaction of Fe(VI) with PPL can be expressed as the following equation:

$$\frac{-\mathrm{d}[\mathrm{Fe(VI)}]}{\mathrm{d}t} = k[\mathrm{Fe(VI)}]^m[\mathrm{PPL}]^n \tag{1}$$

where [Fe(VI)] and [PPL] are the concentrations of Fe(VI) and PPL, m and n are the orders of the reaction with respect to the concentrations of Fe(VI) and PPL, respectively and k is the overall reaction rate constant. The kinetic studies were carried out under pseudo-order conditions with PPL in excess. Eq. (1) can be re-written as:

$$\frac{-\mathrm{d}[\mathrm{Fe(VI)}]}{\mathrm{d}t} = k_1[\mathrm{Fe(VI)}]^m \tag{2}$$

where $k_1 = k\ [\mathrm{PPL}]^n$. The decrease in absorbance of Fe(VI) as a function of time could be fitted nicely to single exponential decay, which indicates that the rate is first-order with respect to the concentration of Fe(VI), i.e., $m = 1$. Different values of k_1 were determined from the slopes of linear plots at various concentrations of PPL at different pHs. The values of k_1 increased linearly with the concentration of PPL (Fig. SM-1). The plot of log k_1 versus log [PPL] was linear ($R^2 = 0.99$) and a slope of 0.84 ± 0.09 (Fig. SM-1) was obtained. This indicates that the reaction was first-order with respect to the [PPL], i.e., $n = 1$ (Fig. SM-1). The rate law for the reaction between Fe(VI) and PPL can then be expressed as:

$$\frac{-\mathrm{d}[\mathrm{Fe(VI)}]}{\mathrm{d}t} = k[\mathrm{Fe(VI)}][\mathrm{PPL}] \tag{3}$$

Similar experiments were also conducted to determine the rate constants at different pH. A plot of the values of k as a function of pH is presented in Fig. 1. A value of k for oxidation of PPL at pH 8 was 20 $M^{-1}\ s^{-1}$, which is slightly higher than that of the oxidation of atenolol by Fe(VI) ($k \sim 7\ M^{-1}\ s^{-1}$) (Lee and von Gunten, 2010). Other pharmaceuticals containing amine moieties, tramadol, enrofloxacin, and ciprofloxacin have values of k as 7.4, 24, and $1.7 \times 10^2\ M^{-1}\ s^{-1}$, respectively, at pH 8.0 (Lee et al., 2009; Anquandah et al., 2011; Zimmermann et al., 2012). Comparatively, sulfonamides had generally higher rates than those of amines containing pharmaceuticals (Sharma et al., 2006a,b).

There was decrease in rate constant with decrease in pH from 9.3 to 8.0, followed by slight decrease as pH was lowered to 7.0. Similar trend of the pH dependence was also observed for the oxidation of secondary amine, dimethylamine, by Fe(VI) (Carr, 2008; Lee and von Gunten, 2010). PPL is a secondary amine molecule and oxidation of it by Fe(VI) may thus involve the amine moiety. The pH dependence behavior in rate may be explained by considering the following equilibria of Fe(VI) and the PPL in the studied pH range.

$$\mathrm{HPPL}^+ \rightleftharpoons \mathrm{H}^+ + \mathrm{PPL}, \quad \mathrm{p}K_\mathrm{a} = 9.5\ (\text{Benner et al., 2008}) \tag{4}$$

$$\mathrm{H_2FeO_4} \rightleftharpoons \mathrm{H}^+ + \mathrm{HFeO_4^-}, \quad \mathrm{p}K_\mathrm{a2} = 3.5 \tag{5}$$

$$\mathrm{HFeO_4^-} \rightleftharpoons \mathrm{H}^+ + \mathrm{FeO_4^{2-}}, \quad \mathrm{p}K_\mathrm{a3} = 7.23 \tag{6}$$

Considering the studied pH range of the reaction between Fe(VI) and PPL, the three species of Fe(VI) (i.e., H_2FeO_4, $HFeO_4^-$, and FeO_4^{2-}) with the two species of PPL (i.e., $HPPL^+$ and PPL) could possibly react with each other. The pH dependence can therefore be modeled by the following equation:

$$k[\mathrm{Fe(VI)}]_\mathrm{tot}[\mathrm{PPL}]_\mathrm{tot} = \Sigma_{i=1,2,3,\ j=1,2} k_{ij}\alpha_i\ \beta_j[\mathrm{Fe(VI)}]_\mathrm{tot}[\mathrm{PPL}]_\mathrm{tot} \tag{7}$$

where $[\mathrm{Fe(VI)}]_\mathrm{tot} = [\mathrm{H_2FeO_4}] + [\mathrm{HFeO_4^-}] + [\mathrm{FeO_4^{2-}}]$; $[\mathrm{PPL}]_\mathrm{tot} = [\mathrm{HPPL}^+] + [\mathrm{PPL}]$; α_i and β_j represent the species of Fe(VI) and PPL, respectively; i and j are each of the species of Fe(VI) and PPL respectively and k_{ij} is the species-specific second-order rate constant for the reaction between the Fe(VI) species i and the PPL species j. While there were six possible reactions from the expression of Eq. (7), only two of the following three reactions (8)–(10) were needed to fit the experimentally observed rate constants.

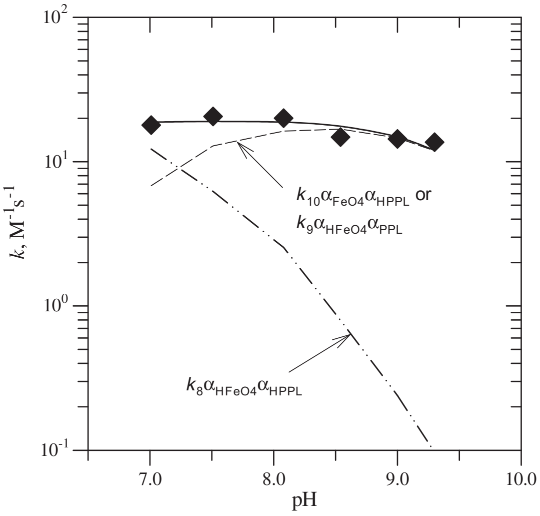


Fig. 1. The observed second-order rate constants (k, $M^{-1}\ s^{-1}$) as a function of pH for the reaction between Fe(VI) and PPL at 25 °C.

$$\mathrm{HFeO_4^-} + \mathrm{HPPL}^+ \rightarrow \mathrm{Fe(OH)_3} + \mathrm{Product(s)} \quad k_8 = (1.8 \pm 0.50) \times 10^1\ \mathrm{M^{-1}\ s^{-1}} \tag{8}$$

$$\mathrm{HFeO_4^-} + \mathrm{PPL} \rightarrow \mathrm{Fe(OH)_3} + \mathrm{Product(s)} \quad k_9 = (3.6 \pm 0.24) \times 10^2\ \mathrm{M^{-1}\ s^{-1}} \tag{9}$$

$$\mathrm{FeO_4^{2-}} + \mathrm{HPPL}^+ \rightarrow \mathrm{Fe(OH)_3} + \mathrm{Product(s)} \quad k_{10} = (2.0 \pm 0.13) \times 10^1\ \mathrm{M^{-1}\ s^{-1}} \tag{10}$$

Reactions (9) and (10) introduce proton ambiguity and thus the experimental values of k could be fitted by using either reactions (8) and (9) or reactions (8) and (10) (solid line in Fig. 1). Contribution of the individual reactions to the rate constant is also shown (dashed lines, Fig. 1). In the alkaline medium, one of the two reactions (9) and (10) mainly contributed to the rate constants while reaction (8) was largely controlling the rate of the oxidation of PPL by Fe(VI) at a neutral pH. Because deprotonated amines usually reacts faster than protonated amines with oxidizing agents, it is likely reactions (8) and (9) might be contributing the rates of the oxidation of PPL by Fe(VI).

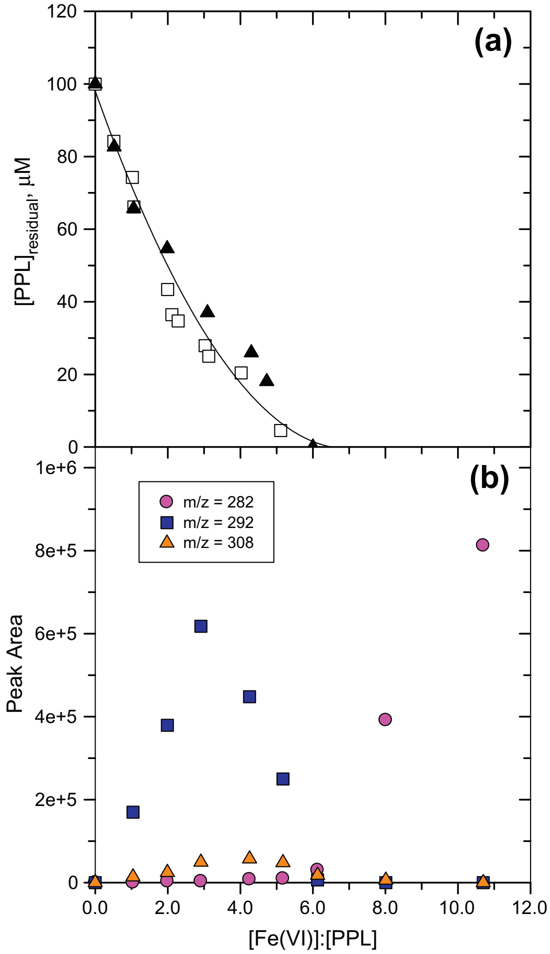
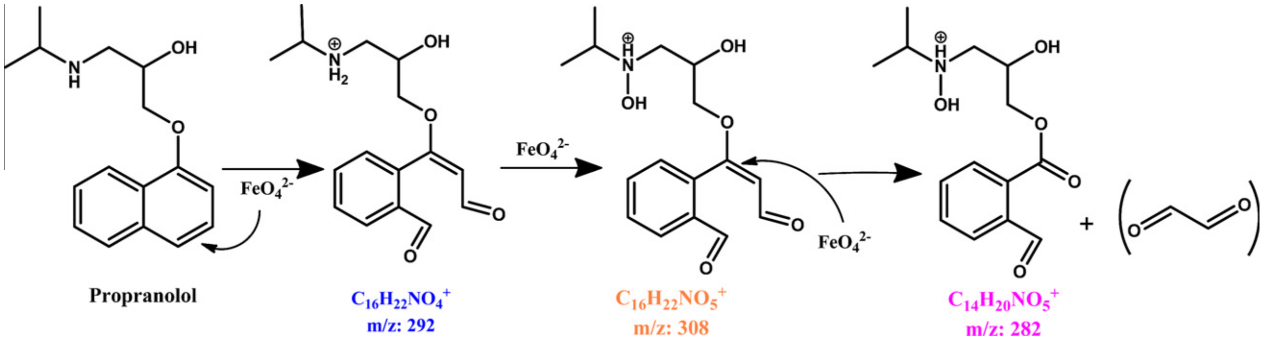


Fig. 2. Removal of PPL (a) and integrated peak areas of products (b) versus molar ratios of Fe(VI) to PPL for the reaction at pH 9.0 (□ – LC-PDA; ▲ – LC–MS).



Scheme 1. Oxidation of propranolol by Fe(VI) ion.

3.2. Stoichiometry

The stoichiometry of the reaction between Fe(VI) and PPL was studied to estimate the dose of Fe(VI) required for complete removal of PPL. In fact, the required does of F(VI) might vary depending on the solution pH. In this study, however, the stoichiometry of the reaction between Fe(VI) and PPL at pH 9.0 was determined by mixing the solutions of Fe(VI) and PPL in which the concentration of PPL was fixed and the concentration of Fe(VI) was varied (Fig. 2a). An increase in the concentration of the Fe(VI) resulted in a decrease in the concentration of PPL. Significantly, the decrease in the concentration of PPL was not linear and required concentration of Fe(VI) for complete removal of PPL was ~6 times over the concentration of PPL. The linear portion of the decreasing curve suggests that almost 50% of the PPL could be removed by using twice the amount of Fe(VI). The remaining 50% removal of PPL needed four times the concentration of Fe(VI). This indicates Fe(VI) in the reaction mixture was not only consumed by the PPL but also by the OPs of the reaction. Furthermore, the rates of the reactions of Fe(VI) with OP(s) are similar to cause the competition with the reaction of Fe(VI) with the parent molecule.

3.3. Identification of OPs

Finally, a study on the products of the oxidation of PPL by Fe(VI) was carried out at pH 9.0. Because of the amine group in the PPL, tests were performed to learn if inorganic products were formed. Ammonia was tested using ion-selective electrode and no detection level of ammonia was made. Tests were also negative for the NO_2^- and NO_3^- ions, hence, the amine group of PPL was not transformed to inorganic oxy-nitrogen ions. However, several OPs were identified at different molar ratio of Fe(VI) to PPL. A plot of peak areas of the identified OPs versus the molar ratio is shown in Fig. 2b. Mass spectra of the identified OPs from the oxidation of PPL by Fe(VI) are given in Figs. SM-2–4. MS/MS spectra of OPs were found to be similar to the previously reported spectra (Benner et al., 2009b).

The product, OP-292 was initially formed and the maximum peak area of this product was observed at a molar ratio of 3:1 ([Fe(VI)]:[PPL]). An increase in the peak area of OP-308 was also observed after a molar ratio of 2:1. The maximum of the OP-308 was at a molar ratio of >4. At a molar ratio of 6, complete removal of OP-292 and OP-308 were observed (Fig. 2). At this molar ratio, PPL was also completely removed (see Fig. 2a). At molar ratios of >6, a new product, OP-282 was formed (Fig. 2b). The observed trend indicates that the oxidation of PPL involves cleavage of an aromatic ring. Scheme 1 shows the proposed plausible pathway for the formation of OPs during the reaction between Fe(VI) and PPL.

Oxidation products shown in Scheme 1 suggest that the Fe(VI) is attacking both moieties, naphthalene and secondary amine group of PPL. The initial attack of Fe(VI) on the double bond of the activated aromatic ring system of PPL on each side leads to opening of a ring via 1,3-dipolar cycloaddition, which resulted in OP-292 having two aldehyde groups (Scheme 1). The opening of the nitrogen-containing organic molecules by Fe(VI) has also been observed previously (Sharma et al., 2006a,b; Yong et al., 2006; Anquandah et al., 2011). This initial step is similar to oxidation of PPL by O_3 (Benner and Ternes, 2009b). The simultaneous formation of OP-308 indicates that the initial step (Fig. 2b) was followed by the attack of Fe(VI) to the amine moiety of the PPL to form the second oxidized product. The hydroxylation of the secondary amine has been suggested previously (Carr, 2008). When both OP-292 and OP-308 degraded completely, the third OP, OP-282 appeared which could possibly be from the oxidation of the double bond in α-position of the aldehyde groups; causing the cleavage of the group to yield the third product (Scheme 1).

4. Conclusions

The reaction between Fe(VI) and PPL followed second-order kinetics and the dependence of the second-order rate constant on pH was successfully explained using the species-specific rate constants of the reactions of Fe(VI) with PPL. The identification of OPs of PPL allowed proposing a simple reaction pathway. The primary product of the oxidation was OP-292, which was then oxidized to OP-308 via oxidation of its amine moiety. Further oxidation of OP-308 yielded OP-282. The OPs have aldehyde groups, which can be further oxidized by adding Fe(VI) beyond the initial required stoichiometry. However, formation of potentially toxic degrades such as aldehydes could be of concern and future studies may include toxicity tests of the products from the reaction between Fe(VI) and the β-blocker.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.chemosphere.2012.12.001.

References

Andreozzi, R., Campanella, L., Fraysse, B., Garric, J., Gonnella, A., Lo Giudice, R., Marotta, R., Pinto, G., Pollio, A., 2004. Effects of advanced oxidation processes (AOPs) on the toxicity of a mixture of pharmaceuticals. Water Sci. Technol. 50, 23–28.

Anquandah, G.A.K., Sharma, V.K., Knight, D.A., Batchu, S.R., Gardinali, P.R., 2011. Oxidation of trimethoprim by ferrate(VI): kinetics, products, and antibacterial activity. Environ. Sci. Technol. 45, 10575–10581.

Barbieria, M., Lichab, T., Nödlerb, K., Carreraa, J., Ayoraa, C., Sanchez-Vilac, X., 2012. Fate of β-blockers in aquifer material under nitrate reducing conditions: batch experiments. Chemosphere 89, 1272–1277.

Benitez, F.J., Acero, J.L., Real, F.J., Roldan, G., Casas, F., 2011. Comparison of different chemical oxidation treatments for the removal of selected pharmaceuticals in water matrices. Chem. Eng. J. 168, 1149–1156.

Benner, J., Ternes, T.A., 2009a. Ozonation of metoprolol: elucidation of oxidation pathways and major oxidation products. Environ. Sci. Technol. 43, 5472–5480.

Benner, J., Ternes, T.A., 2009b. Ozonation of propranolol: formation of oxidation products. Environ. Sci. Technol. 43, 5086–5093.

Benner, J., Salhi, E., Ternes, T., von Gunten, U., 2008. Ozonation of reverse osmosis concentrate: kinetics and efficiency of beta blocker oxidation. Water Res. 42, 3003–3012.

Benotti, M.J., Trenholm, R.A., Vanderford, B.J., Holady, J.C., Stanford, B.D., Snyder, S.A., 2009. Pharmaceuticals and endocrine disrupting compounds in US drinking water. Environ. Sci. Technol. 43, 597–603.

Carr, J.D., 2008. Kinetics and product identification of oxidation by ferrate(VI) of water and aqueous nitrogen containing solutes. ACS Symp. Ser. 985 (Ferrates), pp. 189–196.

Chen, Y., Li, H., Wang, Z., Li, H., Tao, T., Zuo, Y., 2012. Photodegradation of selected β-blockers in aqueous fulvic acid solutions: kinetics, mechanism, and product analysis. Water Res. 46, 2965–2972.

Deblonde, T., Cossu-Leguille, C., Hartemann, P., 2011. Emerging pollutants in wastewater: a review of the literature. Int. J. Hyg. Environ. Health 214, 442–448.

Fent, K., Weston, A.A., Caminada, D., 2006. Ecotoxicology of human pharmaceuticals. Aquat. Toxicol. 76, 122–159.

Filip, J., Yngard, R.A., Siskova, K., Marusak, Z., Ettler, V., Sajdl, P., Sharma, V.K., Zboril, R., 2011. Mechanisms and efficiency of the simultaneous removal of metals and cyanides by using ferrate(VI): crucial roles of nanocrystalline iron(III) oxyhydroxides and metal carbonates. Chem. Eur. J. 17, 10097–10105.

Gabet-Giraud, V., Miège, C., Choubert, J.M., Ruel, S.M., Coquery, M., 2010. Occurrence and removal of estrogens and beta blockers by various processes in wastewater treatment plants. Sci. Total Environ. 408, 4257–4269.

Gros, M., Petrović, M., Ginebreda, A., Barceló, D., 2010. Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes. Environ. Int. 36, 15–26.

Hu, L., Martin, H.M., Arce-bulted, O., Sugihara, M.N., Keating, K.A., Strathmann, T.J., 2009. Oxidation of carbamazepine by Mn(VII) and Fe(VI): reaction kinetics and mechanism. Environ. Sci. Technol. 43, 509–515.

Huggett, D.B., Brooks, B.W., Peterson, B., Foran, C.M., Schlenk, D., 2002. Toxicity of select beta adrenergic receptor-blocking pharmaceuticals (β-blockers) on aquatic organisms. Arch. Environ. Contam. Toxicol. 43, 229–235.

Isarain-Chávez, E., Garrido, J.A., Rodríguez, R.M., Centellas, F., Arias, C., Cabot, P.L., Brillas, E., 2011. Mineralization of metoprolol by electro-fenton and photoelectro-fenton processes. J. Phys. Chem. A 115, 1234–1242.

Jiang, J.Q., 2007. Research progress in the use of ferrate(VI) for the environmental remediation. J. Hazard. Mater. 146, 617–623.

Jiang, J., Zhou, Z., Pahl, O., 2012. Preliminary study of ciprofloxacin (cip) removal by potassium ferrate(VI). Sep. Purif. Technol. 88, 95–98.

Kim, I.H., Yamashita, N., Kato, Y., Tanaka, H., 2009. Discussion on the application of UV/H_2O_2, O_3 and O_3/UV processes as technologies for sewage reuse considering the removal of pharmaceuticals and personal care products. Water Sci. Technol. 59 (5), 945–955.

Köhler, C., Venditti, S., Igos, E., Klepiszewski, K., Benetto, E., Cornelissen, A., 2012. Elimination of pharmaceutical residues in biologically pre-treated hospital wastewater using advanced UV irradiation technology: a comparative assessment. J. Hazard. Mater. 239–240, 70–77.

Küster, A., Alder, A.C., Escher, B.I., Duis, K., Fenner, K., Garric, J., Hutchinson, T.H., Lapen, D.R., Péry, A., Römbke, J., Snape, J., Ternes, T., Topp, E., Wehrhan, A., Knackerk, T., 2010. Environmental risk assessment of human pharmaceuticals in the European union: a case study with the β-blocker atenolol. Integr. Environ. Assess. Manage. 6, 514–523.

Lee, Y., von Gunten, U., 2010. Oxidative transformation of micropollutants during municipal wastewater treatment: comparison of kinetic aspects of selective (chlorine, chlorine dioxide, ferrate^{VI}, and ozone) and non-selective oxidants (hydroxyl radical). Water Res. 44, 555–566.

Lee, Y., Zimmermann, S.G., Kieu, A.T., Gunten, G.V., 2009. Ferrate (Fe(VI)) application for municipal wastewater treatment: a novel process for simultaneous micropollutant oxidation and phosphate removal. Environ. Sci. Technol. 43, 3831–3838.

Li, C., Li, X.Z., Graham, N., Gao, N.Y., 2008. The aqueous degradation of bisphenol A and steroid estrogens by ferrate. Water Res. 42, 109–120.

Luo, Z., Strouse, M., Jiang, J.Q., Sharma, V.K., 2011. Methodologies for the analytical determination of ferrate(VI): a review. J. Environ. Sci. Health A 46, 453–460.

Makky, E.A., Park, G., Choi, I., Cho, S., Kim, H., 2011. Comparison of Fe(VI) (FeO_4^{2-}) and ozone in inactivating *Bacillus subtilis* spores. Chemosphere 83, 1228–1233.

Martínez Bueno, M.J., Ulaszewska, M.M., Gomez, M.J., Hernando, M.D., Fernández-Alba, A.R., 2012. Simultaneous measurement in mass and mass/mass mode for accurate qualitative and quantitative screening analysis of pharmaceuticals in river water. J. Chromatogr., A 1256, 80–88.

Piram, A., Salvador, A., Verne, C., Herbreteau, B., Faure, R., 2008. Photolysis of β-blockers in environmental waters. Chemosphere 73, 1265–1271.

Romero, V., De La Cruz, N., Dantas, R.F., Marco, P., Giménez, J., Esplugas, S., 2011. Photocatalytic treatment of metoprolol and propranolol. Catal. Today 161, 115–120.

Sharma, V.K., 2008. Oxidative transformations of environmental pharmaceuticals by Cl_2, ClO_2, O_3, and Fe(VI): kinetics assessment. Chemosphere 73, 1379–1386.

Sharma, V.K., 2010. Oxidation of nitrogen-containing pollutants by novel ferrate(VI) technology: a review. J. Environ. Sci. Health A 45, 645–667.

Sharma, V.K., 2011. Oxidation of inorganic contaminants by ferrates (Fe(VI), Fe(V), and Fe(IV)) – kinetics and mechanisms – a review. J. Environ. Manage. 92, 1051–1073.

Sharma, V.K., 2013. Ferrate(VI) and ferrate(V) oxidation of organic compounds: kinetics and mechanism. Coord. Chem. Rev. 257, 495–510.

Sharma, V.K., Mishra, S.K., Nesnas, N., 2006a. Oxidation of sulfonamide antimicrobials by ferrate(VI) [$Fe^{VI}O_4^{2-}$]. Environ. Sci. Technol. 40, 7222–7227.

Sharma, V.K., Mishra, S.K., Ray, A.K., 2006b. Kinetic assessment of the potassium ferrate(VI) oxidation of antibacterial drug, sulfamethoxazole. Chemosphere 62, 128–134.

Sharma, V.K., Luther III, G.W., Millero, F.J., 2011. Mechanisms of oxidation of organosulfur compounds by ferrate(VI). Chemosphere 82, 1083–1089.

Sirés, I., Brillas, E., 2012. Remediation of water pollution caused by pharmaceutical residues based on electrochemical separation and degradation technologies: a review. Environ. Int. 40, 212–229.

Sirés, I., Oturan, N., Oturan, M.A., 2010. Electrochemical degradation of β-blockers. Studies on single and multicomponent synthetic aqueous solutions. Water Res. 44, 3109–3120.

Yong, E.E., Sharma, V.K., Ray, A.K., 2006. Ferrate(VI): green chemistry oxidant for degradation of cationic surfactant. Chemosphere 63, 1785–1790.

Zimmermann, S.G., Schmukat, A., Schulz, M., Benner, J., von Gunten, U., Ternes, T.A., 2012. Kinetic and mechanistic investigations of the oxidation of tramadol by ferrate and ozone. Environ. Sci. Technol. 46, 876–884.