Environmental Science
Water Research & Technology

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Journal:	*Environmental Science: Water Research & Technology*
Manuscript ID	EW-ART-05-2018-000290.R1
Article Type:	Paper
Date Submitted by the Author:	30-Jul-2018
Complete List of Authors:	Huang, Ying; University of Cincinnati, Department of Chemical and Environmental Engineering Liu, Yiqing; University of Cincinnati, Department of Chemical and Environmental Engineering; Southwest Jiaotong University, Faculty of Geosciences and Environmental Engineering Kong, Minghao; University of Cincinnati, Department of Chemical and Environmental Engineering Xu, Elvis Genbo ; University of California, Riverside, Department of Environmental Sciences Coffin, Scott ; University of California, Riverside, Department of Environmental Sciences Schlenk, Daniel; University of California, Riverside, Environmental Sciences Dionysiou, Dionysios; University of Cincinnati, Environmental Engineering and Science

Water impact statement

UV/H_2O_2 can efficiently degrade contaminants of emerging concern, such as diclofenac and triclosan, in both Milli-Q water and field water samples. The cytotoxicity of diclofenac increased and remained high for triclosan during and after the treatment of UV/H_2O_2 at UV fluence of 640 mJ cm^{-2}, indicating the generation of toxic transformation products that may elicit enhanced risk to human health, and/or the environment.

Efficient degradation of cytotoxic contaminants of emerging concern by UV/H_2O_2

Ying Huang[1,#], Yiqing Liu[1,2,#], Minghao Kong[1], Elvis Genbo Xu[3], Scott Coffin[3], Daniel Schlenk[3], and Dionysios D. Dionysiou[1*]

[1]Environmental Engineering and Science Program, Department of Chemical and Environmental Engineering, University of Cincinnati, Cincinnati, Ohio 45221, USA

[2]Faculty of Geosciences and Environmental Engineering, Southwest Jiaotong University, Chengdu 611756, China

[3]Department of Environmental Sciences, University of California, Riverside, California 92521, USA

[#]Contributed equally to this work with: Ying Huang, Yiqing Liu

*Corresponding Author:

Dionysios D. Dionysiou, Ph.D.

Tel.: +1-513-556-0724, Fax: +1-513-556-4162

E-mail: dionysios.d.dionysiou@uc.edu

Abstract

The degradation kinetics and cytotoxicity of two commonly detected contaminants of emerging concern (CECs), diclofenac and triclosan, in UV/H_2O_2 systems were investigated in this study. The second-order rate constants of hydroxyl radical ($^{\bullet}OH$) with diclofenac ($k_{^{\bullet}OH,diclofenac}$) and triclosan ($k_{^{\bullet}OH,triclosan}$) varied at different reaction pH (5.3-8.5) in the range of 7.6-9.1 × 10^9 M^{-1} s^{-1} and 7.0-4.4 × 10^9 M^{-1} s^{-1}, respectively. The pH plays a crucial role in the UV/H_2O_2 treatment for the destruction of diclofenac, triclosan, and four additional CECs (estrone, 17β-estradiol, 17α-ethynylestradiol, and bisphenol A), affecting the ionic state of the CECs (based on pKa) and scavenging the $^{\bullet}OH$ by increasing the concentration of hydroxide. The impacts of H_2O_2 concentration, common inorganic ions (i.e., HCO_3^-, NO_3^-, Cl^-, and SO_4^{2-}), and natural organic matter (NOM) were studied as well. Field water samples from the local water works and Lake Harsha were utilized as reaction matrices to assess the possibility of applying UV/H_2O_2 to decompose diclofenac and triclosan in surface water. Cytotoxicity of diclofenac and triclosan was not reduced during treatment even though concentrations of the compounds were diminished, indicating the formation of toxic transformation products. Overall, UV/H_2O_2 is useful to degrade CECs, such as diclofenac and triclosan, in both Milli-Q water and field water samples, but higher UV fluence might be needed to reduce the cytotoxicity of CECs after UV/H_2O_2 treatment.

Keywords

Hydroxyl radical, UV-AOPs, Hydrogen peroxide, Contaminants of emerging concern, Cytotoxicity

1. Introduction

Increasing attention has been paid to the continual detection of CECs in the aquatic environment, which may cause potential adverse impacts on human health and the environment[1-3]. Conventional treatment technologies such as filtration do not sufficiently degrade CECs[4, 5]. As a result, advanced oxidation processes (AOPs) have been proposed to effectively and completely degrade CECs in water and wastewater[1].

UV/H_2O_2, as a $^{\bullet}OH$-based AOP, is widely used to effectively degrade CECs in water and wastewater treatments[6-9]. Advantages include the fact that $^{\bullet}OH$ can be formed through direct dissociation of H_2O_2 with high $^{\bullet}OH$ quantum yield ($\Phi = 1.0$) as shown in eq. 1[10], and no sludge is formed after treatment. However, UV/H_2O_2 can be greatly affected by the constituents of the treated water matrix, such as carbonate/bicarbonate ions (CO_3^{-}/HCO_3^{-}). CO_3^{-}/HCO_3^{-} can react with $^{\bullet}OH$ to generate carbonate radical ($CO_3^{\bullet-}$), lowering the concentration of $^{\bullet}OH$, and thus inhibiting the decomposition of contaminants by $^{\bullet}OH$-oxidation[6]; while the produced $CO_3^{\bullet-}$ may also react with target compounds at a high rate, resulting an enhanced degradation of contaminants[11]. Nitrate (NO_3^{-}), prevalent in wastewater and surface water, leads to the formation of nitrated contaminants during UV and UV/H_2O_2 processes, but does not significantly produce mutagenic activity[12]. Therefore, water quality parameters should be carefully considered when evaluating the application of UV/H_2O_2 for the degradation of CECs.

$$h\upsilon + H_2O_2 \rightarrow 2\ {}^{\bullet}OH \qquad \Phi = 1.0 \qquad (1)$$

Diclofenac, a common non-steroidal anti-inflammatory drug, and triclosan, an extensively used antimicrobial agent, are frequently detected in surface water and effluents of wastewater treatment plants, and may pose a risk to aquatic organisms[13-18]. Moreover, diclofenac and triclosan stimulated concern by an expert panel in California due to the inefficient reduction

of the compounds in water and wastewater treatment processes[19]. The feasibility of UV/H_2O_2 in degrading diclofenac has been reported by several studies[18, 20-22], while the literature of triclosan degradation by UV/H_2O_2 is more limited[18]. Most previous research has focused on the degradation of CEC mixtures, while the performance of UV/H_2O_2 on the decomposition of diclofenac/triclosan and the impacts of water quality parameters (such as pH, alkalinity, and NOM) have received less attention. In addition, CECs usually cannot be mineralized during most oxidation processes and can be transformed to different byproducts, which might be more toxic than the parent compounds[18, 23]. Therefore, the evaluation of oxidation processes should be based on both monitoring the reduction of parent contaminants and evaluating the biological activity of remaining transformation products[3].

In current study, the degradation of diclofenac and triclosan by UV/H_2O_2 was comprehensively investigated with the following aims: i) to investigate the influence of pH, H_2O_2 dosage, common inorganic ions (HCO_3^-, NO_3^-, Cl^-, and SO_4^{2-}), and NOM on the decomposition of diclofenac and triclosan in UV/H_2O_2 process, respectively; ii) to determine the second-order rate constants of diclofenac and triclosan with $^{\bullet}OH$ at various pH conditions; iii) to assess the application of UV/H_2O_2 to degrade diclofenac and triclosan in different field water samples; and iv) to analyze the cytotoxicity of diclofenac and triclosan before, during, and after the UV/H_2O_2 treatment. Although the concentrations of selected CECs used in this research were higher than detected in the water treatment processes, the data provided insight into the degradation kinetics and cytotoxicity of selected CECs, supporting the applications of UV/H_2O_2 as a promising AOP to decompose CECs in surface water treatment.

2. Experimental section

2.1 Chemicals.

Diclofenac, triclosan, atrazine, estrone, 17β-estradiol, 17α-ethynylestradiol, bisphenol A, sodium chloride, sodium sulfate (anhydrous) were ordered from Sigma-Aldrich (St. Louis, MO). Potassium phosphate monobasic, dibasic potassium phosphate, sodium hydroxide, sodium nitrate, sodium bicarbonate, hydrogen peroxide (30 wt%), and acetonitrile were ACS grade and purchased from Fisher Scientific (Hampton, NH, USA). The humic acid standard I (HA, Suwannee River) and fluvic acid standard II (FA, Suwannee River), which were used as NOM, were purchased from International Humic substances Society (St. Paul, MN). All chemicals used were of reagent grade or higher. Purified water (18 MΩ cm) used in experiments was obtained from a Milli-Q system (Milli-pore Corp., Billerica, MA). Field water samples were collected from the local water works and Lake Harsha, and the general water quality parameters are summarized in Table S1.

2.2 Analysis methods.

Diclofenac, triclosan, and atrazine were measured using an Agilent 1100 HPLC equipped with a C_{18} Discovery HS column (2.1 mm × 150 mm × 5 μm, Supelco). HPLC conditions to determine diclofenac, triclosan, and atrazine can be found in Text S1. Total organic carbon (TOC) was measured by a Shimadzu VCSH-ASI TOC analyzer. Alkalinity was determined according to the American Water Works Association (AWWA) Standard Method[24]. EPA Method 415.3 was used to calculate the specific UV absorbance at 254 nm ($SUVA_{254}$) of field water samples[25].

2.3 Photochemical experiments.

Bench-scale photolysis experiments were performed under two 15 W low-pressure (LP) Hg UV lamps (Cole-Parmer) or one UV light-emitting diode (LED) lamp (Aquisense Technologies) with a primary UV emission of λ_{max} = 254 nm. Average UV fluence rate through

the reaction solution was measured to be 0.1 mW cm^{-2} for LP lamps and 0.13 mW cm^{-2} for LED lamps[26]. LP lamps were used to generate UV irradiation unless specified, which are extensively applied for disinfection in drinking water and wastewater treatments. A typical experiment was as follows: 1 μM of diclofenac or triclosan and 1 mM of H_2O_2 were added into a Petri dish, covered with a quartz cover (Quartz Scientific Inc., OH), and placed under UV irradiation with constant stirring. The pH was stabilized by 10 mM of phosphate buffer through the reactions. Analysis was prepared by sampling 200 μL of the reaction solution after each UV fluence interval and mixing with 200 μL of methanol. All kinetic experiments were performed in triplicate at room temperature (21 ± 1 □). The standard deviation from replicates was shown as error bar in figures.

To determine the second-order rate constants of •OH with diclofenac and triclosan at different pH values, competition kinetic approaches were conducted using 1 μM of atrazine (ATZ) as the reference substance for •OH ($k_{\cdot OH,ATZ} = 2.4 \times 10^9$ M^{-1} s^{-1})[27] and using 10 mM of phosphate buffer to adjust the pH. The $k_{\cdot OH,CEC}$ were calculated using eq. 2, where k'_{CEC} and k'_{ATZ} are the observed degradation rate constants of the selected CECs (diclofenac and triclosan) and ATZ.

$$k_{\cdot OH,CEC} = k_{\cdot OH,ATZ} \times \frac{k'_{CEC,UV/H_2O_2/buffer} - k'_{CEC,UV/buffer}}{k'_{ATZ,UV/H_2O_2/buffer} - k'_{ATZ,UV/buffer}} \quad (2)$$

2.4 Cytotoxicity analysis

Samples collected before, during, and after UV/H_2O_2 treatment were evaluated for cytotoxicity. In this study, cytotoxicity of samples were analyzed using GeneBLAzer CYP1A1-bla LS-180 cells (Life Technologies, CA) with the 3-(4,5-dimethylthiazol-2-yl)-2,5-diphenyltetrazolium bromide (MTT) assay[28]. The MTT assay was chosen because it is a widely-used and robust assay to examine the cell viability[29, 30]. Cultured cells were collected and plated

in a 96-well plate at a density of $2x10^4$ cells 100 μL^{-1}. 24 hours later, the cells were treated with vehicle control, 10% DMSO positive control or water samples and were incubated for an additional 16 hours. The media was removed and replaced with 50 ul of MTT solution (5mg mL^{-1} in PBS) and incubated an additional 4 hours. The MTT solution was then replaced with 200 ul of 1:1 ethanol: dimethyl sulfoxide and shaken for 10 minutes to solubilize the MTT crystals. Absorbance was measured using an automatic plate reader (SpectraMax+ 384, Molecular Devices, San Jose, CA). Green (650 nm) and blue (595 nm) absorbance were measured. The resulting blue: green ratio provides a normalized reporter response, with the higher value indicating lower cytotoxicity.

A typical process to prepare samples was as follows: after the addition of catalase used to quench the residual H_2O_2, the sample was passed through a 500 mg solid phase extraction (SPE) C18 column (Oasis HLB, Waters). The analytes were eluted with 8 mL of pure methanol (MeOH) and reconstituted in 1 mL of MeOH after drying under nitrogen gas. All sample groups were analyzed in triplicate at the concentration of 2.5% MeOH.

3. Results and discussion

3.1 Contribution of $^{\bullet}OH$ to the degradation of diclofenac and triclosan by UV/H_2O_2

The comparison for the destruction of diclofenac and triclosan by three different treatment methods, including UV only, low-pressure (LP)-UV/H_2O_2, and light-emitting diode (LED)-UV/H_2O_2, is illustrated in Figs. 1a and b, respectively. Degradation of diclofenac and triclosan by H_2O_2 oxidation was 1.3% and 8.0% after 120 min, respectively (data not shown). Both diclofenac and triclosan could be degraded by direct UV photolysis at 254 nm, which was largely promoted by the addition of 1 mM of H_2O_2 due to the production of $^{\bullet}OH$. To further confirm the role of $^{\bullet}OH$, 10 mM of MeOH ($k_{^{\bullet}OH,MeOH} = 9.7 \times 10^8\ M^{-1}\ s^{-1}$ [31]) was added into the

reaction solutions as the scavenger for •OH. For both diclofenac and triclosan, the extent of degradation in the UV/H_2O_2/MeOH system was almost the same as in the UV only system, indicating the complete scavenging of •OH by 10 mM of MeOH. The k_{obs} for diclofenac in UV only and UV/H_2O_2 processes were determined to be $(5.62 \pm 0.19) \times 10^{-3}$ and $(2.17 \pm 0.05) \times 10^{-2}$ cm^2 mJ^{-1}; and the k_{obs} for triclosan in UV only and UV/H_2O_2 processes were determined to be $(2.99 \pm 0.06) \times 10^{-3}$ and $(1.74 \pm 0.01) \times 10^{-2}$ cm^2 mJ^{-1}. Therefore, the contribution of UV direct photolysis *vs* •OH oxidation for the degradation of diclofenac and triclosan in the UV/H_2O_2 process was 25.9%: 74.1% and 17.2%: 82.8%, which was calculated following methods reported by Liu *et al.*[7]. The results indicated the important contribution of •OH in the decomposition of diclofenac and triclosan in the UV/H_2O_2 process. In addition, LP-UV/H_2O_2 and LED-UV/H_2O_2 had similar degradation rates of diclofenac/triclosan, suggesting the feasibility of alternating LP-UV with LED-UV in the activation of H_2O_2, which has many advantages over the Hg lamp, including superior stability, flexible forms, long lifetimes, and low costs.

3.2 Effect of pH

The reaction pH becomes a crucial parameter for the decomposition of diclofenac and triclosan in the UV only and UV/H_2O_2 system because of its influence on the dominant species of organic contaminants, the form of H_2O_2, and the concentration of OH^-. Reaction pH values were selected as 5.3, 5.9, 6.6, 7.4, and 8.5, and adjusted by phosphate buffer. Though the reactions between •OH and HPO_4^{2-} ($k_{\cdot OH,HPO_4^{2-}} = 1.5 \times 10^5$ M^{-1} s^{-1}) or $H_2PO_4^-$ ($k_{\cdot OH,H_2PO_4^-} = 2.0 \times 10^4$ M^{-1} s^{-1}) have relatively lower second order rate constants[31], the total concentration of phosphate ions (10 mM) was 10^4 times higher than the initial concentration of target contaminants (1 μM), which would lead to competition for •OH between organic contaminants and phosphate ions. Therefore, the reaction pH was selected within the buffer capacity of

phosphate buffer to obtain the similar influence of phosphate ions. At current pH conditions, its effect on the H_2O_2 species could be excluded because H_2O_2 forms hydroperoxide anion (HO_2^-) only when pH is higher than 11.6[26, 31].

In the UV/H_2O_2 system, the degradation of diclofenac was slightly increased in the pH range of 5.3-6.6 and decreased significantly when the pH was increased from 6.6 to 8.5, as shown in Fig. 2a. Contrasted to diclofenac, the decomposition of triclosan slightly decreased in the pH range of 5.3-7.4 but increased significantly when the pH was raised from 7.4 to 8.5, as depicted in Fig. 2b. At higher pH, the concentration of OH^- increased, which can react with $^{\bullet}OH$ at an extremely high rate as 1.2×10^{10} M^{-1} s^{-1} [31]. Both UV photolysis and $^{\bullet}OH$-oxidation contributed to the degradation of diclofenac and triclosan in UV/H_2O_2; thus, it is important to specify the effects of pH on UV photolysis and $^{\bullet}OH$-oxidation.

In the UV-only process, the destruction of diclofenac was independent of pH (data not shown), while the degradation of triclosan increased significantly when the pH increased from 6.6 to 8.5 (data not shown). The reason might be the pKa of diclofenac and triclosan are 4 and 7.9, respectively, and this may affect the dominant species of diclofenac and triclosan in different pH conditions. For diclofenac, it would be anionic ($C_{13}H_{10}Cl_2N$-COO^-) with the carboxylic group deprotonated at the selected pH range of 5.3-8.5, leading to similar molar absorption coefficient at 254 nm (data not shown). For triclosan, the percentage of the anions with the phenolic group deprotonated ($C_{12}H_7Cl_3O$-O^-) would increase with increasing pH from 5.3 to 8.5, as a result, the molar absorption coefficient at 254 nm would be higher (data not shown).

The second-order rate constants of $^{\bullet}OH$ with diclofenac ($k_{^{\bullet}OH,DCF}$) and triclosan ($k_{^{\bullet}OH,TCS}$) at selected pH values (5.3, 5.9, 6.6, 7.4, and 8.5) were determined to explore the effects of pH on $^{\bullet}OH$-oxidation and are shown in Table 1. Results on competition kinetics

experiments using atrazine as a reference compound are shown in Figs. S1 and S2. Consistent to the phenomenon discussed above, the highest $k_{\cdot OH,DCF}$ was 9.09×10^9 M^{-1} s^{-1} at pH 7.4. This value approximated to the $k_{\cdot OH,DCF}$ in literature as $(9.29 \pm 0.11) \times 10^9$ M^{-1} s^{-1}, which was determined from a plot of k_{obs} vs concentration of diclofenac at pH = 7[23]. Unexpectedly, the highest $k_{\cdot OH,TCS}$ was 8.88×10^9 M^{-1} s^{-1} at pH 5.9, and $k_{\cdot OH,TCS}$ decreased as pH increased from 5.9 to 8.5. Even though the k_{obs} of triclosan in UV/H_2O_2 was highest at pH 8.5, the $k_{\cdot OH,TCS}$ was lowest in the pH range of 5.5 to 8.4. The measured $k_{\cdot OH,TCS}$ were different than the previous reported value as 5.4×10^9 M^{-1} s^{-1}, which was measured by Fenton reaction at pH 3.5[32]. Therefore, the degradation efficiency of diclofenac was lower in the pH range of 6.6-8.5 due to the quenching effect of OH^-. However, the positive enhancement on direct UV photolysis of triclosan at high pH (8.5) exceeded the inhibition effect of increased OH^- on the $^{\bullet}OH$-oxidation of triclosan.

Since pH had completely different effects on diclofenac and triclosan in the UV/H_2O_2 system, more CECs, namely estrone, 17β-estradiol, 17α-ethynylestradiol, and bisphenol A, were tested to further study the influence of pH as shown in Figs. 2c-2f. The pKa of estrone, 17β-estradiol, 17α-ethynylestradiol, and bisphenol A are 10.77, 10.71, 10.33, and 9.2, respectively. Only estrone and 17β-estradiol were degraded by UV photolysis, and at low rates (Figs. S3 and S4), which were not affected by the change of pH from 5.3 to 8.5. Similar to diclofenac, the degradation of estrone, 17β-estradiol, 17α-ethynylestradiol, and bisphenol A were lower in the UV/H_2O_2 process as the pH was increased from 5.3 to 8.5.

Therefore, it is hypothesized that the effect of pH on the degradation of a certain CEC by UV/H_2O_2 is related to the pKa of the CEC. If the pKa is beyond the selected pH range, the degradation rate of the CEC by UV/H_2O_2 may be inhibited as the pH increases due to the

scavenging of $^{\bullet}OH$ by increasing OH^-. If the pKa is within the selected pH range, the molar absorption coefficient at 254 nm of the CEC should be considered and monitored to precisely investigate the impact of pH on UV/H_2O_2.

3.3 Effect of initial concentration of H_2O_2

The dosage of H_2O_2 plays a significant role on evaluating the possible application of UV/H_2O_2 in water and wastewater treatment. The correlation between the initial concentration of H_2O_2 and the k_{obs} of diclofenac and triclosan are shown in Fig. 3. For both diclofenac and triclosan, the extent of degradation and k_{obs} increased as the H_2O_2 dosage increased; however, there was no proportional growth between k_{obs} and H_2O_2 dosage (0.1-2 mM), which is different than results of previous reports[7, 33]. In previous studies, the k_{obs} increased linearly with the H_2O_2 dosage in the range of 0.1-0.9 mM for microcystin-LR and in the range of 0.1-0.5 mM for oxytetracycline. The nonlinear growth between k_{obs} and H_2O_2 might be due to the competition of excess H_2O_2 with diclofenac/triclosan for produced $^{\bullet}OH$ (eqs. (3)-(5))[31].

$$^{\bullet}OH + H_2O_2 \rightarrow HO_2^{\bullet} + H_2O \qquad k = 2.7 \times 10^7\ M^{-1}s^{-1} \tag{3}$$

$$^{\bullet}OH + HO_2^{\bullet} \rightarrow O_2 + H_2O \qquad k = 6.6 \times 10^9\ M^{-1}s^{-1} \tag{4}$$

$$^{\bullet}OH + {}^{\bullet}OH \rightarrow H_2O_2 \qquad k = 5.5 \times 10^9\ M^{-1}s^{-1} \tag{5}$$

3.4 Effect of common inorganic anions

The presence of HCO_3^- was found to have significant inhibition on the decomposition of organic contaminants by $^{\bullet}OH$-based AOPs[6, 33, 34]. However, as shown in Fig. 4a, the addition of HCO_3^- (0.5-6 mM) only slightly inhibited the degradation of diclofenac and triclosan in the UV/H_2O_2 system. The pH was near 8.5 when HCO_3^- was added, thus, the pH of the control group was maintained at 8.5 by 10 mM of phosphate buffer to exclude the impacts of pH on the degradation of diclofenac and triclosan. CO_3^{2-}/HCO_3^- can react with $^{\bullet}OH$ to produce $CO_3^{\bullet-}$ at

high rate, as presented in eqs. (6)-(7)[31]. $CO_3^{\bullet-}$ has an affinity to electron-rich aromatic compounds, especially aniline and phenolic groups[35-37]. Therefore, the aromatic amine group on diclofenac and the phenolic group on triclosan might lead to the contribution of $CO_3^{\bullet-}$ to the degradation of diclofenac and triclosan by UV/H_2O_2 in the presence of HCO_3^-.

$$^{\bullet}OH + CO_3^{2-} \rightarrow CO_3^{\bullet-} + OH^- \qquad k = 3.9 \times 10^8\ M^{-1}s^{-1} \qquad (6)$$

$$^{\bullet}OH + HCO_3^- \rightarrow CO_3^{\bullet-} + H_2O \qquad k = 8.5 \times 10^6\ M^{-1}s^{-1} \qquad (7)$$

Although NO_3^- is a recognized photosensitizer, it had a slight inhibitory effect on the degradation of diclofenac but had no influence on the degradation of triclosan in the range of 1-10 mM (Fig. 4b). A partial reason for this phenomenon might be the low reaction rate of NO_3^- with $^{\bullet}OH$ ($k_{^{\bullet}OH,NO_3^-} < 5 \times 10^5\ M^{-1}\ s^{-1}$)[38]. Although direct photolysis of NO_3^- could produce $^{\bullet}OH$, nitrite ions (NO_2^-) that react with $^{\bullet}OH$ at a high rate constant (eq. (8)) would be formed as well. The supplement and consumption of $^{\bullet}OH$ due to the presence of NO_3^- might be about the same in the degradation of diclofenac and triclosan.

$$^{\bullet}OH + NO_2^- \rightarrow NO_2^{\bullet} + OH^- \qquad k = 9.1 \times 10^9\ M^{-1}s^{-1} \qquad (8)$$

Similar to the influence of NO_3^-, Cl^- (1-10 mM) also slightly inhibited the decomposition of diclofenac but had no effect on the decomposition of triclosan (Fig. 4c). At neutral conditions, Cl^- can react with $^{\bullet}OH$ to form $ClOH^{\bullet-}$, which will disassociate to regenerate $^{\bullet}OH$ (eqs. (9)-(10))[31]. Therefore, the influence of Cl^- on removing diclofenac and triclosan by UV/H_2O_2 was limited.

$$^{\bullet}OH + Cl^- \rightarrow ClOH^{\bullet-} \qquad k = 4.3 \times 10^9\ M^{-1}s^{-1} \qquad (9)$$

$$ClOH^{\bullet-} \rightarrow {}^{\bullet}OH + Cl^- \qquad k = 6.1 \times 10^9\ M^{-1}s^{-1} \qquad (10)$$

For both diclofenac and triclosan, SO_4^{2-} (0.5-3 mM) did not affect their degradation in the UV/H_2O_2 system (Fig. 4d), because they are nonreactive with $^{\bullet}OH$[8].

3.5 Effect of NOM

The influence of isolated NOM on the decomposition of diclofenac and triclosan in UV-only and UV/H_2O_2 systems at pH 7.4 were investigated using 5 mg L^{-1} HA or FA. As described in Fig. 5a, the presence of HA or FA did not affect the degradation of diclofenac by direct UV photolysis but did significantly inhibit its degradation by $^{\bullet}$OH-oxidation. For triclosan, the HA or FA both dramatically inhibited the destruction in UV only and UV/H_2O_2 system as shown in Fig. 5b. Under UV irradiation, NOM could function as a photosensitizer to generate reactive oxygen species such as $^{\bullet}$OH, $^1O_2$, and $O_2^{\bullet-}$ [39]. Nevertheless, in the $^{\bullet}$OH-based oxidation processes, NOM can also compete with organic contaminants for $^{\bullet}$OH due to a high second-order rate constant ($k_{^{\bullet}OH,NOM} = 2.2 \times 10^8$ L (mol C)$^{-1}$ s^{-1})[40]. Therefore, the remarkable inhibition of isolated NOM on the reduction of diclofenac and triclosan in UV/H_2O_2 system indicated that the production of $^{\bullet}$OH through the photolysis of NOM was surpassed by the scavenging effect of $^{\bullet}$OH by NOM.

3.6 Degradation of diclofenac and triclosan in field water samples

When various field water samples were used as reaction matrices, the k_{obs} of diclofenac by UV was similar to that of Milli-Q water; however, the k_{obs} in UV/H_2O_2 process was much lower than in Milli-Q water as shown in Fig. 6a. The results indicated the main constituents, alkalinity and NOM, did not affect the direct photolysis of diclofenac at 254 nm, but they had a more pronounced inhibition of the $^{\bullet}$OH-oxidation of diclofenac. For RAW, FLIN, GACI, and CUVI water samples, the k_{obs} in CUVI was the highest due to it having the lowest TOC, indicating the necessity of GAC as a pre-UV/H_2O_2 treatment. Nevertheless, the k_{obs} in RAW (the highest TOC) was higher than in FLIN and GACI probably because of the $CO_3^{\bullet-}$ generated by the high alkalinity. Additionally, LH had the highest TOC and alkalinity among five field water samples, and the k_{obs} was similar to that in RAW, confirming the adverse effects of alkalinity and

NOM on the degradation of diclofenac by $^{\bullet}OH$-oxidation.

In contrast to diclofenac, the degradation rate of triclosan by UV was inhibited only in field water samples (RAW and LH) as shown in Fig. 6b. Similar to diclofenac, the decomposition of triclosan by UV/H_2O_2 was reduced significantly in field water samples. Due to the addition of NaOH into the stock solution of triclosan, the pH of reacted RAW and LH samples was 8.2 and 8.9, respectively. Through the comparison of the degradation of triclosan and the constituents in different reaction matrices, the presence of large amounts of NOM as well as other constituents such as alkalinity slowed down the decomposition of triclosan by UV-only and UV/H_2O_2 in field water samples.

3.7 Cytotoxicity analysis of treated diclofenac and triclosan

Although diclofenac and triclosan could be effectively degraded by UV/H_2O_2, it is important to analyze the toxicity of the resulting solution. For both diclofenac and triclosan, the reactions of $^{\bullet}OH$ and UV photolysis led to complex mixtures of transformation products at low concentration, which were difficult to isolate. The cytotoxicity of the solutions before, during, and after UV/H_2O_2 treatment was evaluated by the MTT assay. Since the pH values of the collected field water samples were in the range of 7.2-7.8, the reaction pH for cytotoxicity analysis was kept at 7.4 using phosphate buffer.

For the degradation of diclofenac in the UV/H_2O_2 treatment, significantly higher cytotoxicity was observed at the low UV fluence (160 mJ cm^{-2}; p = 0.0145), while it declined slightly as the UV fluence increased to 640 mJ cm^{-2} (Fig. 7a). One-way ANOVA analysis results of the cytotoxicity of diclofenac during the treatment of UV/H_2O_2 are presented in Table S2. The changes of cytotoxicity throughout the UV/H_2O_2 process suggested the generation of transformation products with increased cytotoxicity compared to the parent contaminant. Five

possible transformation products of diclofenac in the UV/H_2O_2 system were reported previously (Fig. S7), namely 2-(8-chloro-9*H*-carbazol-1-yl)acetic acid, 2-(8-hydroxy-9*H*-carbazol-1-yl)acetic acid, 2-(8-hydroxy-3-oxo-3*H*-carbazol-1-yl)acetic acid, 8-methyl-9*H*-carbazol-1-ol, and 8-hydroxy-1-methyl-3*H*-carbazol-3-one[41]. These transformation products had a carbazole group and were generated through ring closure, dechlorination, hydroxylation, and decarboxylation processes. With both LP-UV/H_2O_2 and medium pressure (MP)-UV/H_2O_2 (200-800 nm), these five transformation products were formed with the UV fluence of 300 mJ cm^{-2}. The enhanced cytotoxicity at high UV fluence (640 mJ cm^{-2}) indicated that even more toxic transformation products may have been generated during the oxidation of diclofenac in the UV/H_2O_2 process.

For triclosan, cytotoxicity was not significantly changed either by UV photolysis or by $^{\bullet}$OH-oxidation as shown in Fig. 7b (p = 0.8986). One-way ANOVA analysis results of the cytotoxicity of triclosan during the treatment of UV/H_2O_2 was summarized in Table S3. The possible transformation products (Fig. S8) and degradation mechanisms of triclosan in $^{\bullet}$OH-based oxidation process and UV photolysis (316 nm) were also published previously and included 2,7-dichloro-dibenzodioxin/2,8-dichloro-dibenzodioxin (DCDD) and 4,5'-dichloro-[1,1'-biphenyl]-2,2'-diol ($(OH)_2$PCB-13)[42, 43]. In the presence of $^{\bullet}$OH at a high concentration, dioxins and dioxin-like compounds, such as DCDD, are easily produced from triclosan and are more stable than triclosan[42]. DCDD can also be formed through the photocyclization of triclosan as a halogenated phenoxy-phenol[32, 43]. Moreover, the $(OH)_2$PCB-13 could be formed in abundance through biradical intermediates with photochemical excitation[43]. The DCDD and $(OH)_2$PCB-13 are extremely toxic and carcinogenic[42-44], which may not have acute toxicity.

From the current results, the transformation products of diclofenac and triclosan in the UV/H_2O_2 system with the UV fluence of 640 mJ cm^{-2} cytotoxicity was observed, indicating

potential risk to human health and the ecosystem. Therefore, further remediation for diclofenac and triclosan are needed to reduce the risks associated with UV/H_2O_2 treatments.

4. Conclusion

This study investigated the respective degradation kinetics and cytotoxicity of diclofenac and triclosan in both UV and UV/H_2O_2 systems. Although diclofenac and triclosan could be degraded by direct UV photolysis, $^{\bullet}$OH-oxidation significantly contributed to the destruction of these two contaminants in UV/H_2O_2 system (74.1% and 82.2%, respectively). The pH had different effects for the degradation of different CECs by UV/H_2O_2, which was related to the pKa of CECs. With the increase of pH from 5.3 to 8.5, the degradation of diclofenac, estrone, 17β-estradiol, 17α-ethynylestradiol, and bisphenol A decreased in the UV/H_2O_2 system but increased for triclosan in both UV only and UV/H_2O_2 processes. The second-order rate constants of $^{\bullet}$OH with diclofenac at pH 5.3, 5.9, 6.6, 7.4, and 8.5 were 7.57×10^9, 7.56×10^9, 8.43×10^9, 9.09×10^9, and 8.67×10^9 M^{-1} s^{-1}. The second-order rate constants of $^{\bullet}$OH with triclosan at pH 5.3, 5.9, 6.6, 7.4, and 8.5 were 7.02×10^9, 8.88×10^9, 5.66×10^9, 5.14×10^9, and 4.43×10^9 M^{-1} s^{-1}. The k_{obs} of diclofenac and triclosan was enhanced with increasing initial concentration of H_2O_2 while excess H_2O_2 could compete with contaminants for the $^{\bullet}$OH. The presence of common inorganic anions (i.e., NO_3^-, Cl^-, and SO_4^{2-}) at various concentrations did not affect the degradation of diclofenac and triclosan under current reaction conditions (1 mM of H_2O_2 at pH 7.4); while the presence of HCO_3^- slightly inhibited their decomposition. NOM represented by HA and FA significantly inhibited the destruction of diclofenac and triclosan in both UV-only and UV/H_2O_2 system. Except for the UV photolysis of diclofenac, the degradation rate of diclofenac and triclosan by UV only and UV/H_2O_2 were diminished when various field water samples were used as reaction matrices. Furthermore, the cytotoxicity of diclofenac increased

during the treatment. However, the cytotoxicity did not significantly change for triclosan after the treatment. This study provides further understanding and theoretical support for the application of UV/H_2O_2 to degrade diclofenac and triclosan in surface water.

Conflicts of interest

There are no conflicts to declare.

Acknowledgements

The authors acknowledge financial support from the U.S. Geological Survey (USGS)-Water Resources Research Institute (WRRI) (2015SC101G) for this research. Ying Huang acknowledges support from the China Scholarship Council (CSC) scholarship (201306270057). Minghao Kong acknowledges support from the CSC scholarship (201608110134). We are thankful to the local water works for collecting and sending water samples.

References

1. M. Salimi, A. Esrafili, M. Gholami, A. J. Jafari, R. R. Kalantary, M. Farzadkia, M. Kermani and H. R. Sobhi, *Environ. Monit. Assess.*, 2017, **189**, 22.
2. K. A. Maruya, D. Schlenk, P. D. Anderson, N. D. Denslow, J. E. Drewes, A. W. Olivieri, G. I. Scott and S. A. Snyder, *Integrated Environmental Assessment and Management*, 2014, **10**, 69-77.
3. H. Mestankova, A. M. Parker, N. Bramaz, S. Canonica, K. Schirmer, U. von Gunten and K. G. Linden, *Water Research*, 2016, **93**, 110-120.
4. S. K. Behera, H. W. Kim, J. E. Oh and H. S. Park, *Science of the Total Environment*, 2011, **409**, 4351-4360.

5. V. Matamoros, C. Arias, H. Brix and J. M. Bayona, *Water Research*, 2009, **43**, 55-62.

6. J. Deng, Y. Shao, N. Gao, S. Xia, C. Tan, S. Zhou and X. Hu, *Chemical Engineering Journal*, 2013, **222**, 150-158.

7. Y. Liu, X. He, Y. Fu and D. D. Dionysiou, *Chemical Engineering Journal*, 2016, **284**, 1317-1327.

8. N.-y. Gao, Y. Deng and D. Zhao, *Journal of Hazardous Materials*, 2009, **164**, 640-645.

9. T. J. Wei Li, Kenneth Ishida and Haizhou Liu, *Environmental Science: Water Research & Technology*, 2017, **3**, 128.

10. J. H. Baxendale and J. A. Wilson, *Transactions of the Faraday Society*, 1957, **53**, 344-356.

11. Y. Liu, X. He, X. Duan, Y. Fu and D. D. Dionysiou, *Chemical Engineering Journal*, 2015, **276**, 113-121.

12. S. Semitsoglou-Tsiapou, M. R. Templeton, N. J. D. Graham, S. Mandal, L. Hernández Leal and J. C. Kruithof, *Environmental Science: Water Research & Technology*, 2018, DOI: 10.1039/C7EW00389G.

13. M. Bock, J. Lyndall, T. Barber, P. Fuchsman, E. Perruchon and M. Capdevielle, *Integrated environmental assessment and management*, 2010, **6**, 393-404.

14. H. Singer, S. Müller, C. Tixier and L. Pillonel, *Environmental Science & Technology*, 2002, **36**, 4998-5004.

15. D. Stülten, S. Zühlke, M. Lamshöft and M. Spiteller, *Science of the Total Environment*, 2008, **405**, 310-316.

16. R. Oliveira, I. Domingues, C. Koppe Grisolia and A. M. V. M. Soares, *Environmental Science and Pollution Research*, 2009, **16**, 679-688.

17. M. Schmitt-Jansen, P. Bartels, N. Adler and R. Altenburger, *Analytical and Bioanalytical Chemistry*, 2007, **387**, 1389-1396.

18. O. Rozas, C. Vidal, C. Baeza, W. F. Jardim, A. Rossner and H. D. Mansilla, *Water Research*, 2016, **98**, 109-118.

19. K. A. Maruya, D. Schlenk, P. D. Anderson, N. D. Denslow, J. E. Drewes, A. W. Olivieri, G. I. Scott and S. A. Snyder, *Integrated environmental assessment and management*, 2014, **10**, 69-77.

20. S. Giannakis, F. A. G. Vives, D. Grandjean, A. Magnet, L. F. De Alencastro and C. Pulgarin, *Water Research*, 2015, **84**, 295-306.

21. I.-Y. Kim, M.-K. Kim, Y. Yoon, J.-K. Im and K.-D. Zoh, *Desalination and Water Treatment*, 2014, **52**, 6211-6218.

22. D. Vogna, R. Marotta, A. Napolitano, R. Andreozzi and M. d'Ischia, *Water Research*, 2004, **38**, 414-422.

23. H. Yu, E. Nie, J. Xu, S. Yan, W. J. Cooper and W. Song, *Water Research*, 2013, **47**, 1909-1918.

24. E. W. Rice, L. Bridgewater, A. American Water Works, F. Water Environment and A. American Public Health American Public Health Association, Washington, D.C, 22nd, 2012 edn., 2012.

25. B. Potter and J. Wimsatt, *National exposure research laboratory office of research and development. US Environmental Protection Agency, Cincinnati Ohio*, 2005.

26. X. He, M. Pelaez, J. A. Westrick, K. E. O'Shea, A. Hiskia, T. Triantis, T. Kaloudis, M. I. Stefan, A. A. de la Cruz and D. D. Dionysiou, *Water research*, 2012, **46**, 1501-1510.

27. H. V. Lutze, S. Bircher, I. Rapp, N. Kerlin, R. Bakkour, M. Geisler, C. von Sonntag and

T. C. Schmidt, *Environmental Science & Technology*, 2015, **49**, 1673-1680.

28. A. A. Van de Loosdrecht, R. H. J. Beelen, G. J. Ossenkoppele, M. G. Broekhoven and M. M. A. C. Langenhuijsen, *Journal of Immunological Methods*, 1994, **174**, 311-320.

29. B. I. Escher, M. Allinson, R. Altenburger, P. A. Bain, P. Balaguer, W. Busch, J. Crago, N. D. Denslow, E. Dopp, K. Hilscherova, A. R. Humpage, A. Kumar, M. Grimaldi, B. S. Jayasinghe, B. Jarosova, A. Jia, S. Makarov, K. A. Maruya, A. Medvedev, A. C. Mehinto, J. E. Mendez, A. Poulsen, E. Prochazka, J. Richard, A. Schifferli, D. Schlenk, S. Scholz, F. Shiraishi, S. Snyder, G. Su, J. Y. M. Tang, B. v. d. Burg, S. C. v. d. Linden, I. Werner, S. D. Westerheide, C. K. C. Wong, M. Yang, B. H. Y. Yeung, X. Zhang and F. D. L. Leusch, *Environmental Science & Technology*, 2014, **48**, 1940-1956.

30. W. Li, E. Xu, D. Schlenk and H. Liu, *Environmental Science: Water Research & Technology*, 2018, DOI: 10.1039/C8EW00107C.

31. G. V. Buxton, C. L. Greenstock, W. P. Helman and A. B. Ross, *Journal of Physical and Chemical Reference Data*, 1988, **17**, 513-886.

32. D. E. Latch, J. L. Packer, B. L. Stender, J. VanOverbeke, W. A. Arnold and K. McNeill, *Environmental Toxicology and Chemistry*, 2005, **24**, 517-525.

33. X. He, M. Pelaez, J. A. Westrick, K. E. O'Shea, A. Hiskia, T. Triantis, T. Kaloudis, M. I. Stefan, A. A. de la Cruz and D. D. Dionysiou, *Water Research*, 2012, **46**, 1501-1510.

34. G. Zhang, X. He, M. N. Nadagouda, K. E. O'Shea and D. D. Dionysiou, *Water Research*, 2015, **73**, 353-361.

35. R. A. Larson and R. G. Zepp, *Environmental Toxicology and Chemistry*, 1988, **7**, 265-274.

36. T. Umschlag and H. Herrmann, *Acta Hydrochimica Et Hydrobiologica*, 1999, **27**, 214-

222.

37. Y. Liu, X. He, X. Duan, Y. Fu, D. Fatta-Kassinos and D. D. Dionysiou, *Water research*, 2016, **95**, 195-204.

38. P. Neta and R. E. Huie, *The Journal of Physical Chemistry*, 1986, **90**, 4644-4648.

39. M. Welker and C. Steinberg, *Environmental Science & Technology*, 2000, **34**, 3415-3419.

40. P. Westerhoff, S. P. Mezyk, W. J. Cooper and D. Minakata, *Environmental Science & Technology*, 2007, **41**, 4640-4646.

41. K. Lekkerkerker-Teunissen, M. J. Benotti, S. A. Snyder and H. C. van Dijk, *Separation and Purification Technology*, 2012, **96**, 33-43.

42. Y. Gao, Y. Ji, G. Li and T. An, *Water research*, 2014, **49**, 360-370.

43. S. Kliegman, S. N. Eustis, W. A. Arnold and K. McNeill, *Environmental Science & Technology*, 2013, **47**, 6756-6763.

44. M. Van den Berg, L. Birnbaum, A. T. Bosveld, B. Brunström, P. Cook, M. Feeley, J. P. Giesy, A. Hanberg, R. Hasegawa, S. W. Kennedy, T. Kubiak, J. C. Larsen, F. X. van Leeuwen, A. K. Liem, C. Nolt, R. E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Waern and T. Zacharewski, *Environmental Health Perspectives*, 1998, **106**, 775-792.

Legends

Table 1. Second-order rate constants of $^{\bullet}OH$ with diclofenac and triclosan at different pH values.

pH	Diclofenac (M^{-1} s^{-1})	Triclosan (M^{-1} s^{-1})
5.3	$(7.6 \pm 0.1) \times 10^9$	$(7.0 \pm 0.5) \times 10^9$
5.9	$(7.6 \pm 0.3) \times 10^9$	$(8.90 \pm 0.2) \times 10^9$
6.6	$(8.4 \pm 0.2) \times 10^9$	$(5.7 \pm 0.2) \times 10^9$
7.4	$(9.1 \pm 0.2) \times 10^9$	$(5.1 \pm 0.2) \times 10^9$
8.5	$(8.7 \pm 0.1) \times 10^9$	$(4.4 \pm 0.5) \times 10^9$

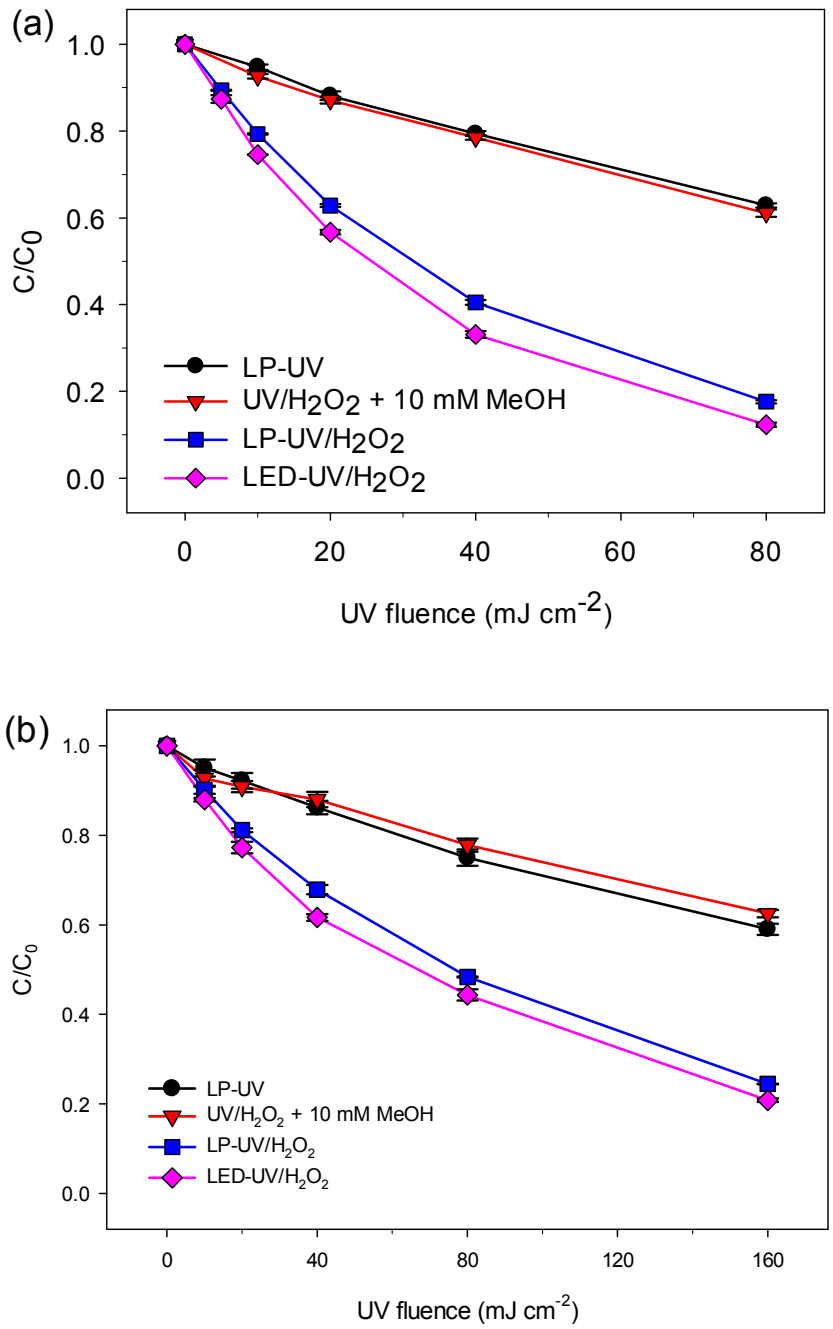

Fig. 1. Degradation of diclofenac (a) and triclosan (b) by UV only, UV/H_2O_2, and UV/H_2O_2/MeOH. Reaction conditions: $[Diclofenac]_0$ = $[Triclosan]_0$ = 1 μM, $[H_2O_2]_0$ = 1 mM, $[MeOH]_0$ = 10 mM. No phosphate buffer was added.

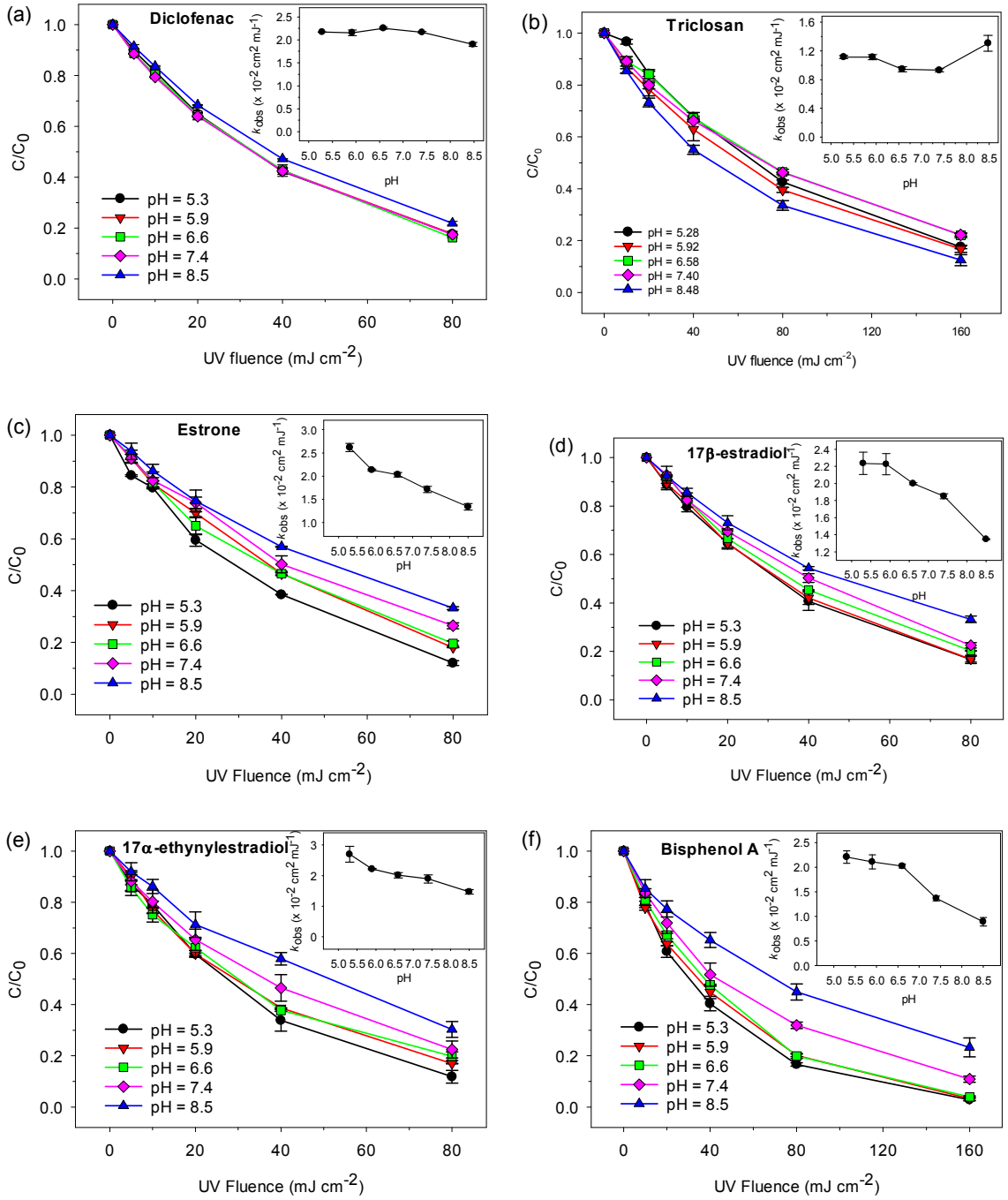

Fig. 2. Effect of pH on the degradation of diclofenac (a), triclosan (b), estrone (c), 17β-estradiol (d), 17α-ethynylestradiol (e), and bisphenol A (f) by UV/H_2O_2. Reaction conditions: $[CEC]_0$ = 1 μM, $[H_2O_2]_0$ = 1 mM.

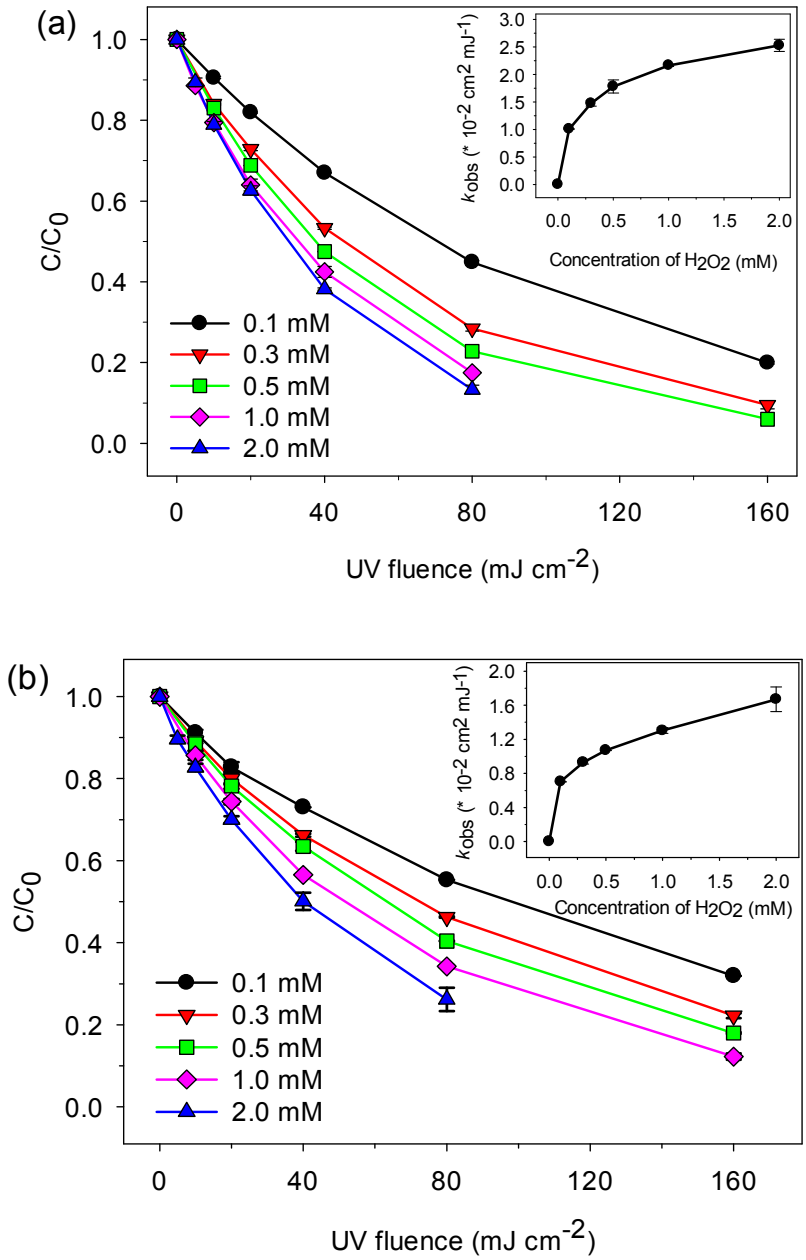

Fig. 3. Effect of initial concentration of H_2O_2 on the k_{obs} of diclofenac (a) and triclosan (b) in UV/H_2O_2 system. Reaction conditions: $[Diclofenac]_0 = [Triclosan]_0 = 1$ μM, $[H_2O_2]_0 = 1$ mM, pH = 7.4 (adjusted by 10 mM of phosphate buffer).

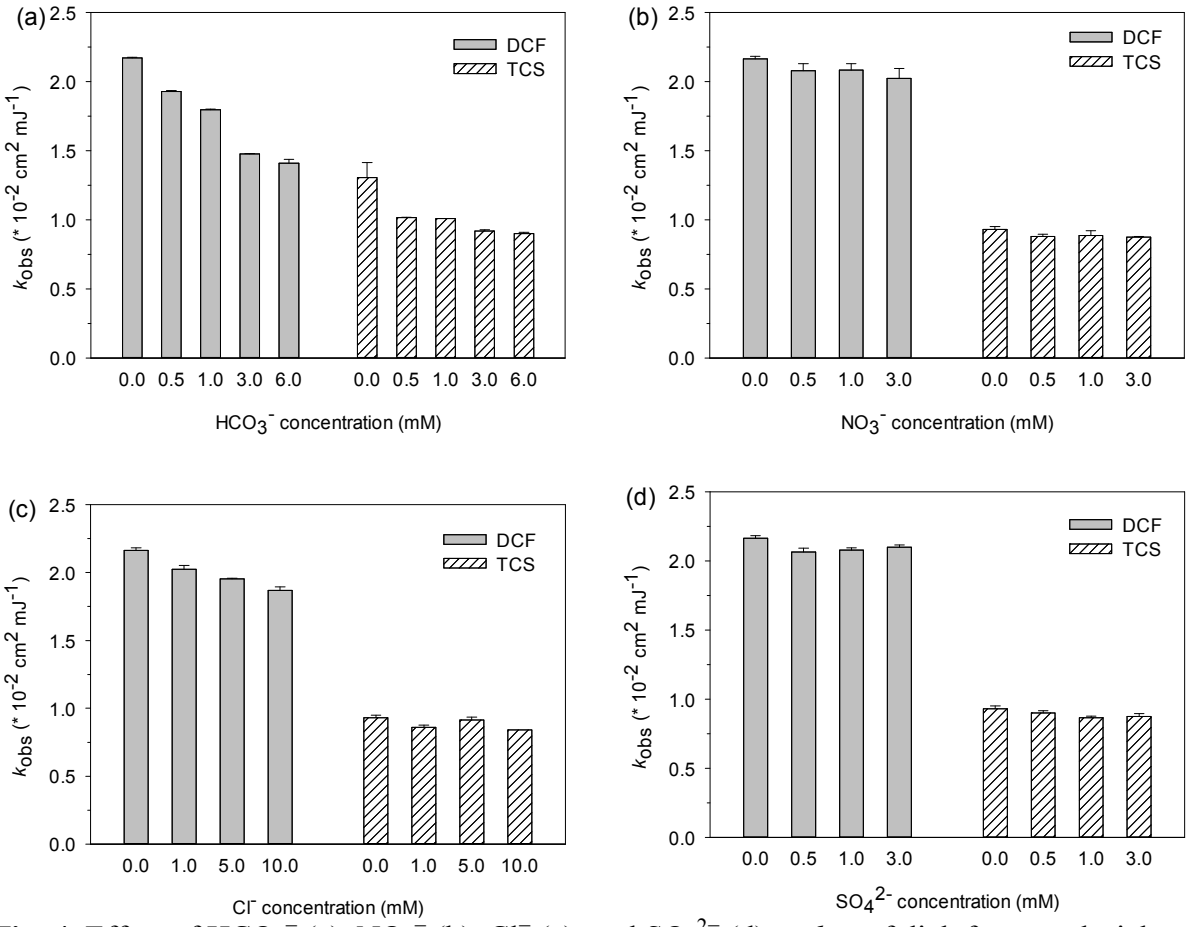

Fig. 4. Effect of HCO_3^- (a), NO_3^- (b), Cl^- (c), and SO_4^{2-} (d) on k_{obs} of diclofenac and triclosan in UV/H_2O_2 system. Reaction conditions: $[Diclofenac]_0 = [Triclosan]_0 = 1$ μM, $[H_2O_2]_0 = 1$ mM, pH = 8.5 for (a), pH = 7.4 for (b), (c), and (d).

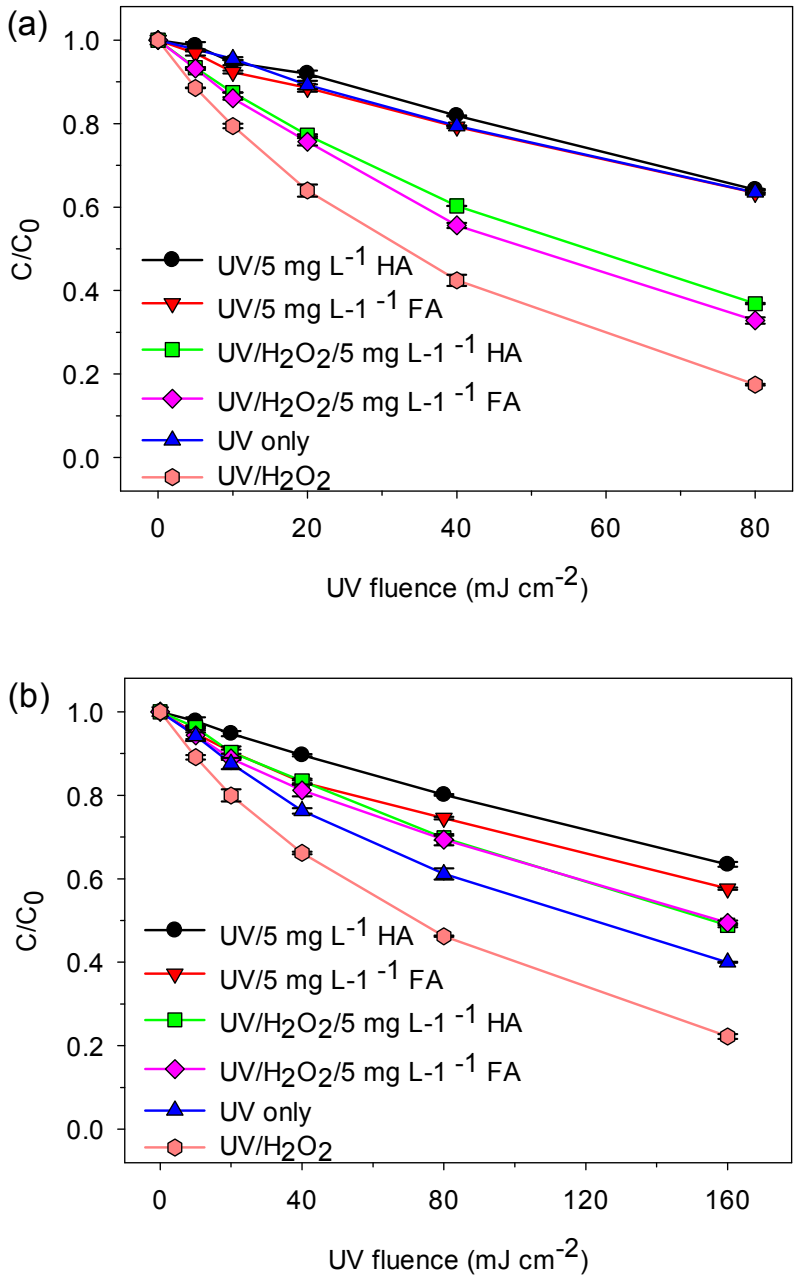

Fig. 5. Effect of NOM on the degradation of diclofenac (a) and triclosan (b) by UV only and UV/H_2O_2. Reaction conditions: $[Diclofenac]_0 = [Triclosan]_0 = 1$ μM, $[H_2O_2]_0 = 1$ mM, [FA] = [HA] = 5 mg L^{-1}, pH = 7.4 (adjusted by 10 mM of phosphate buffer).

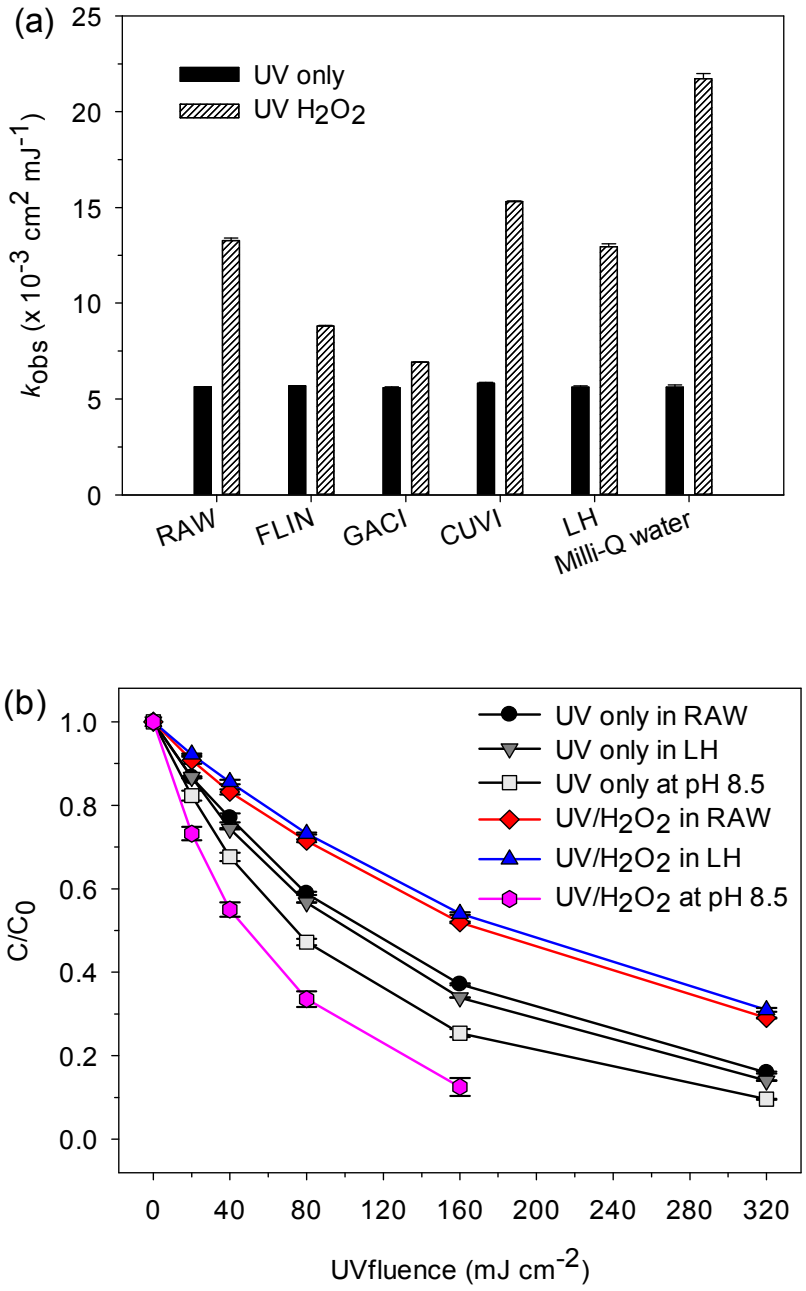

Fig. 6. Degradation of diclofenac (a) and triclosan (b) by UV only and UV/H_2O_2. Reaction conditions: $[Diclofenac]_0 = [Triclosan]_0 = 1$ μM, $[H_2O_2]_0 = 1$ mM. (Abbreviation: RAW, FLIN, GACI, and CUVI were collected from the local water works, Mar. 31, 2015, representing water samples of raw water from Ohio River, sand filtration influent, granular activated carbon (GAC)

influent, and GAC effluent. LH represented the water samples from Lake Harsha, Ohio, Mar.27, 2015.)

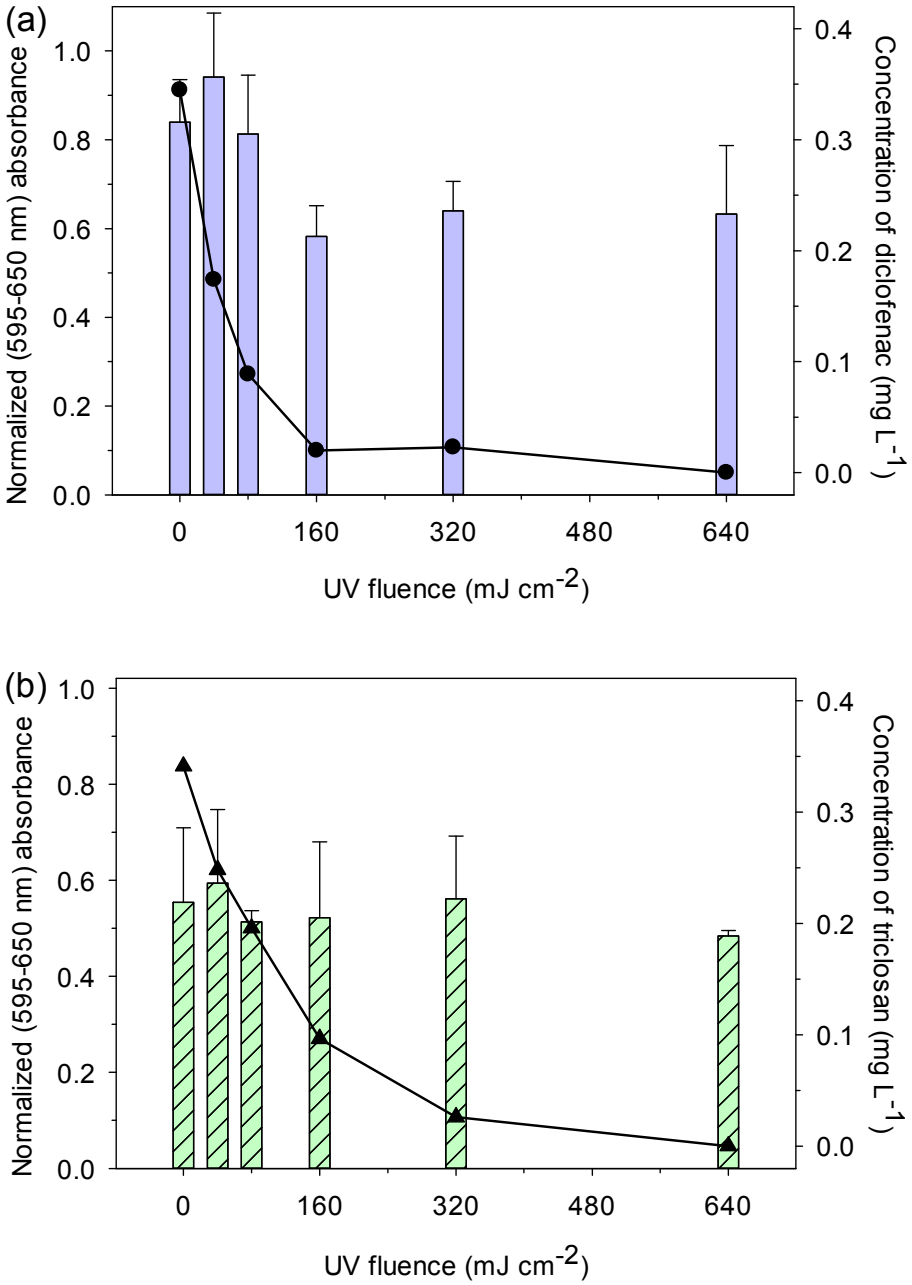

Fig. 7. Cytotoxicity of diclofenac (a) and triclosan (b) treated with UV/H_2O_2 by the MTT assays. *Left y* axis represents cytotoxicity by bars; and *right y* axis represents concentration by lines. The higher the bar, the lower the toxicity. Reaction conditions: $[Diclofenac]_0 = [Triclosan]_0 = 1$ μM, $[H_2O_2]_0 = 1$ mM, pH = 7.4 (adjusted by 10 mM of phosphate buffer).

Graphic Abstract