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DOI: https://doi.org/10.1016/j.jece.2023.110699

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# Journal of Environmental Chemical Engineering

journal homepage: www.elsevier.com/locate/jece



# Ketoprofen and diclofenac removal and toxicity abatement in a real scale sewage treatment Plant by photo-Fenton Process with design of experiments

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ARTICLE INFO

Editor: Despo Fatta-Kassinos

*Keywords:* Emerging contaminants Factorial experimental design Advanced oxidation processes Toxicity

#### ABSTRACT

The disposal of effluent and sludge containing emerging contaminants (ECs) are the main problems associated with most traditional treatment methods. Non-steroidal anti- inflammatory drugs (NSAIDs) require more complete treatments integrated with conventional processes to enable circular economy and the water-energy-food (WEF) nexus. Different options to traditional technologies have been employed in the treatment of effluents, such as advanced oxidation processes (AOPs). However, few studies use real scales integrated with existing technologies. This work evaluated the use of AOPs for the degradation of ketoprofen (KET), diclofenac (DCF), and the DCF+KET mixture in an anaerobic effluent matrix, post-treated by an Upflow Anaerobic Sludge Blanket Reactor (UASB) in a real-scale Sewage Treatment Plant (STP). A Factorial Experimental Design was proposed and after 2 h of treatment the efficiency was evaluated by the decrease in total organic carbon (TOC), decrease of drugs concentration, physicochemical and ecotoxicity parameters. As a result, the photo-Fenton process showed greater efficiency in the degradation of KET (97.2%  $\pm$  1.6), DCF (72.7%  $\pm$  4.9), and the mixture of the two drugs DCF+KET (66.0%  $\pm$  2.5), improved the quality of the effluent in the removal of COD, BOD, and turbidity, and reduced toxicity in tests performed with Artemia sp. and Lactuca sativa. The integration between AOPs and UASB as a sewage treatment technology widely used as primary and secondary treatment, allowed to optimize the sewage treatment system for an ideal arrangement. As a consequence, it possesses the capacity to eliminate conventional organic matter and efficiently degrade emerging pollutants.

# 1. Introduction

Emerging contaminants (ECs) in environmental matrices represent a great concern due to their widespread use and continuous discharge into the environment [1]. Even at trace level concentrations [2,3] the greatest concern is linked to the adverse effects that these compounds can cause to the environment and the health of living beings, including acute and chronic toxicity [4,5] since the exposure can happen in short, medium and long periods of time [6].

As is known, the conventional treatment used in most sewage treatment plants (STPs) cannot completely eliminate them [7], especially anaerobic systems, which are widely used due to their low operating cost (OPEX), with emphasis on upflow anaerobic sludge blankets (UASBs) due to their compactness and low investment cost (CAPEX). Additionally, the high hydraulic retention time (HRT), when compared to biological processes with advanced oxidative processes, and the disposal of effluent and ECs containing sludge are the main problems associated with most of these traditional methods [8], which may make

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https://doi.org/10.1016/j.jece.2023.110699

Received 29 April 2023; Received in revised form 28 July 2023; Accepted 1 August 2023 Available online 3 August 2023 2213-3437/© 2023 Elsevier Ltd. All rights reserved. the reuse of the effluent and the recovery of by-products (biosolids, biomass, etc.) unfeasible due to possible risks to human health. As a result, ECs, especially non-steroidal anti-inflammatory drugs (NSAIDs), and their metabolites, have been detected in various environmental matrices (water, soil, sludge, sewage, effluents, etc.), in concentrations ranging from nanograms to micrograms per liter [9].

NSAIDs are one of the EC classes of greatest concern for aquatic ecosystems, even at low concentrations (ng  $\tilde{L}^{-1}$  or  $\mu g \: L^{-1}$ ) as evidenced by several studies that aimed to identify and quantify these contaminants [10,11]. The presence of NSAIDs associated with the reuse of water is a potential risk, causing health threats when discarded in bodies of water and reused in agriculture, industry, aquaculture, urban use for non-potable purposes and recreation [12,13], requiring more complete treatments that are integrated with conventional processes to enable the circular economy and the nexus concept (water, energy and food). As there is no full knowledge of the potential risks that reuse can induce, especially with regard to organic compounds in treated effluents [14] and there is no doubt that wastewater recovery is an important component of sustainable water resources management [15], it is essential to add technologies that have the ability to completely eliminate these contaminants in wastewater before their release into the environment [16].

Combining DCF with other drugs in water samples can considerably increase their toxic effects. [17]. Therefore, further studies are needed aiming at the application of oxidation processes that guarantee the satisfactory removal of these drugs, in order to preserve the quality of the water in the environment [18]. The concern with the conservation of aquatic ecosystems has stimulated studies for the development of efficient methods of waste removal [19,20].

Different options to traditional technologies have been employed in the treatment of effluents, such as advanced oxidation processes (AOPs), which have stood out for their ability to successfully remove various organic compounds [21,22]. However, few studies use real scales integrated with existing technologies [23,24] and have been studied as treatment methods for urban and industrial wastewater containing different types of organic pollutants with a high degree of toxicity [25].

The Fenton process is one of the most common methods of AOPs. It involves the combination of hydrogen peroxide (H<sub>2</sub>O<sub>2</sub>) and ferrous ions (Fe<sup>2+</sup>) to destroy organic pollutants [26]. However, this process generates a large volume of Fe(OH)<sub>3</sub> sludge, which requires additional and more costly treatment [27]. To overcome this limitation, more advanced methods have been developed, such as photo-Fenton (PF), which utilizes ultraviolet (UV) light to generate hydroxyl radicals and regenerate the ferrous ions [28]. Thus, the irradiation of Fenton reaction systems not only regenerates the catalyst (Fe<sup>2+</sup>), but also produces additional hydroxyl radicals, responsible for the degradation of organic matter [29].

In some cases, the use of these advanced processes for removing contaminants and treating effluents may lead to the generation of toxic compounds and more toxic than the initial samples [30]. In this sense, in addition to the evaluation of routinely investigated physicochemical parameters, ecotoxicological analyzes must be carried out [31]. Furthermore, by removing toxicity, complete mineralization is not necessary, which consequently reduces the application costs of these processes, which is the main limitation for working on a real scale.

This study aimed to evaluate the efficiency of Fenton and photo-Fenton processes for polishing anaerobic effluent (sewage + landfill leachate), post-treated by a real-scale UASB reactor and enriched with NSAIDs (KET, DCF, and the DCF + KET mixture). Physicochemical parameters were analyzed. Two bioindicators from different trophic levels, Lactuca sativa and Artemia sp., were used in the toxicity tests. This work is the first to provide information on the degradation aspects of KET and DCF in anaerobic effluent from a post-UASB reactor using blacklight through a statistical design of experiments (DOE).

# 2. Materials and methods

### 2.1. Reagents

Ketoprofen (KET; CAS 22071–15–4) and diclofenac (DCF; CAS 15307–79–6) with  $\geq$  99% purity and hydrogen peroxide (30% purity) were purchased from Sigma-Aldrich. Ferrous sulfate heptahydrate (> 99% purity), formic acid (98–100% purity), sulfuric acid (95–98% purity) and sodium hydroxide (99% purity) were purchased from Synth. Acetonitrile with LC–MS grade was purchased from J.T. Baker and methanol with LC–MS grade was purchased from Panreac.

# 2.2. Effluent sampling and physicochemical tests

The anaerobic effluent (sewage + landfill leachate), post-treated by a UASB reactor in real scale (SW) was collected at the Los Angeles Municipal Sewage Treatment Plant (STP) located in the municipality of Campo Grande, capital of the State of Mato Grosso do Sul, with latitude 20°39'27.27"S and longitude 54°50'52.36"W (Fig. 1). Its main treatment is UASB reactors with an operating capacity of 1.100 L s<sup>-1</sup> receiving the leachate produced by the municipal landfill, which has an average flow of 5.05 L s<sup>-1</sup>, corresponding to 1% of the total effluent throughout the day. The treated effluent is released into the Anhanduí River (classification IV; [32]). Raw sewage and SW were evaluated for their main physicochemical characteristics and for the quantification of the drugs diclofenac and ketoprofen presence (Table 1), they were stored under appropriate conditions at 4 °C, before treatments with AOPs. The determination of the drugs DCF and KET concentration was obtained using the solid phase extraction technique (SPE), as described in the supplementary material. The physicochemical characterization was performed according to the methods and techniques described in the Standard Methods for the Examination of Water and Wastewater [33].

The UASB reactors showed efficient removal of the physicalchemical parameters BOD, TN and N-NH<sub>4</sub> in 80.9%, 83.5% and 94.2% respectively, in addition to removing 99.9% of DCF and 98.8% of KET (Table 1). Even with high drug removal values, these residual values, in  $\mu$ g L<sup>-1</sup>, are considered very high and present a potential health threat [2, 3], therefore it is necessary to integrate UASB and AOP to polish this effluent.

# 2.3. Experimental Procedures

The UASB post-reactor effluent (SW), consisting of domestic sewage and 1% landfill leachate, was characterized in terms of physicochemical parameters and enriched with individual solutions of the drugs KET and DCF at a concentration of 25 mg  $L^{-1}$  and/or the mix containing both drugs (DCF+KET), each at a concentration of  $12.5 \text{ mg L}^{-1}$ . It was necessary to add a quantity of drugs in the effluent to monitor the degradation and mineralization process. In addition, this allowed monitoring the reduction of toxicity through the decay of pharmaceutical concentrations. As a source of radiation, blacklight lamps (UVA; 320-400 nm, TechLux, Power 27 W), 1-3 lamps were used in parallel with each other. The photochemical reactor (Fig. 2) consisted of two borosilicate glass tubes with an external diameter of 26 mm, 2 mm thick and 125 mm long, interconnected by flexible PVC transparent hoses. The tubes were positioned transversely to the lamps, 600 mL of drugenriched SW (SW-KET, SW-DCF or SW-KET+DCF) was irradiated. Aliquots of 7 mL were withdrawn for TOC and HPLC analysis, but the withdrawn volume did not exceed 10% of the total volume. Temperature (initial 23.0  $\pm$  1.5 and final 35.0  $\pm$  1.0  $^{\circ}\text{C}$ ) and pH (initial pH 3.0  $\pm$  0.1 and final pH 2.7  $\pm$  0.1) were not kept constant and were monitored in situ. Ferrous sulfate heptahydrate (FeSO<sub>4</sub>·7 H<sub>2</sub>O) was used as a source of Fe<sup>2+</sup> and was added directly into the reactor over the recirculating effluent. Fe<sup>2+</sup> concentrations varied between 3 and 15 mg  $L^{-1}$ (values within the limit established by Brazilian legislation, CONAMA 430/2011) (Table 2). Hydrogen peroxide was added gradually at a flow



Fig. 1. – Los Angeles Sewage Treatment Plant UASB Reactors. For the experiments, the reactors of the first phase were used, which are the oldest reactors. Source: Águas Guariroba, MS, Brazil. Available in: www.aguasguariroba.com.br/UASB\_reactors.

# Table 1 Physicochemical characteristics of the domestic effluent.

Parameters	Raw sewage*	SW*	Efficiency (%)	Release limits standard <sup>**</sup>	Methods
Temperature	$26.6~\pm$	$\textbf{26.2} \pm$	-	< 40 °C	-
(°C)	4.4	4.7			
рН	7.4 $\pm$	7.0 $\pm$	-	6.0 – 9.0	4500 B
	0.1	0.2			
Turbidity	117.5	$\textbf{28.5}~\pm$	75.7	-	2130 B
(NTU)	$\pm$ 17.08	12.0			
TOC (mg $L^{-1}$ )	181.65	127.35	29.9	-	NPOC
	$\pm 0.34$	$\pm 0.43$			
$COD (mg L^{-1})$	754.6	338.5	55.1	-	5220 B
	$\pm$ 59.73	$\pm$ 8.0			
BOD (mg $L^{-1}$ )	333.8	$68.9~\pm$	80.9	< 120	5210 B
	$\pm$ 30.78	40.0			
TN (mg $L^{-1}$ )	127.7	$21.1~\pm$	83.5	-	4500 C
	$\pm$ 15.87	3.2			
$NO_{2}^{-}$ (mg L <sup>-1</sup> )	0.023	$2.4 \pm$	-	-	4500 B
	$\pm 0.005$	1.3			
$NO_{3}^{-}$ (mg L <sup>-1</sup> )	2.0 $\pm$	2.8 $\pm$	-	-	4110 B
	0.25	0.8			
N-NH4 (mg	76.0 $\pm$	4.4 $\pm$	94.2	20	4500 B
L <sup>-1</sup> )	5.14	1.2			and C
TP (mg $L^{-1}$ )	7.9 $\pm$	4.4 $\pm$	44.3	-	4500 B5
	0.79	0.9			and 4500
					Е
TS (mg $L^{-1}$ )	756.8	674.0	10.9	-	2540B
	±	$\pm$ 122.7			
	120.06				
Fe (mg $L^{-1}$ )	$\textbf{2.8}~\pm$	$1.3~\pm$	43.8	15	3030 E
	0.006	0.002			and 3111
					В
DCF ( $\mu g L^{-1}$ )	412.6	0.5 $\pm$	99.9	-	SPE
	$\pm \ 0.000$	0.002			
KET ( $\mu g L^{-1}$ )	90.3 $\pm$	1.1 $\pm$	98.8	-	SPE
	0.003	0.003			

 $^*$  n = 14 samples; (  $\pm$  ) standard deviation; Raw Sewage = UASB reactor inlet effluent; SW = UASB post-reactor effluent; TOC = Total organic carbon; COD = Chemical Oxygen Demand; BOD = Biochemical oxygen demand; TN = total nitrogen; N-NH<sub>4</sub> = ammonium nitrogen; TP = total phosphorus; TS = total solids; DCF = diclofenac; KET = ketoprofen; NPOC = non-purgeable organic carbon; SPE = solid phase extraction.

<sup>\*\*</sup> Release limit values in class IV river [32,34]. There are no limits for Turbidity, TOC, COD, TN, NO<sub>2</sub>, NO<sub>3</sub>, TP, TS, DCF and KET in Brazilian legislation for class IV rivers.



Fig. 2. Scheme of the blacklight photochemical reactor.

# Table 2

Levels of the  $2^4$  factorial design with added center point for the photo-Fenton process.

Independent variables		Levels	
	-1	0	+1
$[H_2O_2] (mg L^{-1})$	10	20	30
$[Fe^{2+}]$ (mg L <sup>-1</sup> )	3	9	15
Irradiation time (min) <sup>a</sup>	-1	0	+ 1
Radiation dose (RD) (mW s $cm^{-2}$ )	0.22	0.46	0.70

<sup>a</sup> Time (min) for degradation (-1 = 5; 0 = 17.5; +1 = 30) and time (min) for mineralization (-1 = 20; 0 = 85; +1 = 150)

of 1.0 mL min<sup>-1</sup> with the aid of a peristaltic pump over a period of 20 min·H<sub>2</sub>O<sub>2</sub> concentrations varied between 10 and 30 mg L<sup>-1</sup>. The concentration of H<sub>2</sub>O<sub>2</sub> was monitored via Merck quant peroxide analytical test strips (Test Peroxides, Merck Merck quant). The residual hydrogen peroxide concentration was less than  $2 \text{ mg L}^{-1}$  in all experiments.

For the Fenton and photo-Fenton processes, the effluent pH was adjusted to 3.0 with 1 mol  $L^{-1}$  H<sub>2</sub>SO<sub>4</sub> solution. To determine the applied radiation dose (RD) and radiation intensity (RI), the photon flux reaching the solution to be degraded by the chemical actinometry procedure was determined [35,36], involving the irradiation of a 0.15 mol  $L^{-1}$  potassium ferrioxalate complex in which the Fe<sup>2+</sup> concentration generated during the irradiation interval is determined

spectrophotometrically (Kasuaki mod. IL227) at 510 nm by measuring the absorbance of the complex formed  $[Fe(phen)_3]^{2+}$ . The photon flux presented the values of  $1.7\times10^{-5}$  Ein  $s^{-1}$  for 1 activated lamp,  $1.9\times10^{-5}$  Ein  $s^{-1}$  for two activated lamps, and  $2.3\times10^{-5}$  Ein  $s^{-1}$  for three activated lamps. The reactor flow rate was monitored during the reactions  $(12.7\pm2.34\,L\,min^{-1})$  and aliquots were collected at predetermined times.

# 2.4. Factorial design with added center point

A 2<sup>4</sup> factorial experimental design was applied to investigate the variables that influenced the degradation of KET and the mineralization of organic matter in SW by the photo-Fenton process, in order to optimize the process by finding the experimental conditions with the highest degradation efficiency and lowest consumption of reagents. Therefore,  $H_2O_2$  and Fe<sup>2+</sup> ion concentration, RD and irradiation time were chosen as independent variables (Table 2). The selection of threshold intervals for the four variables under investigation was determined through a combination of preliminary experiments and literature searches [24]. Nineteen experiments with triplicates at the center point were randomly generated using Statistica Statsoft software version 10. The significance of each model was identified using the p-value (p < 0.05). Statistical validation was obtained by analysis of variance (ANOVA) with a confidence level of 95%.

# 2.5. Ecotoxicity tests

#### 2.5.1. Artemia sp

For the hatching of Artemia sp cysts, a synthetic solution of sea salt was used at a concentration of 36 g L<sup>-1</sup> with pH 8–9 aerated for 48 h. After hatching, 10 individuals were added to each 10 mL beaker containing 3 mL of the sample at 100%, 75%, 50%, 25%, 12.5% and 6.25% (v/v) concentrations [37]. The temperature was maintained at 25  $\pm$  2 °C, with a light photoperiod of 16 h and a dark photoperiod of 8 h, in a static system. Assays were performed in triplicate. The pH was adjusted to 7.0 using 1 mol L<sup>-1</sup> NaOH solution whenever necessary. Saline solution was used as a negative control and for sample dilution. For the positive control, 1% potassium dichromate was used. The results obtained were expressed in average Lethal Concentration (LC<sub>50</sub>) and Toxic Unit (TU).

For the  $LC_{50}$  calculation, PROBIT analysis was conducted using StatPlus AnalystSoft Version v8 [38,39]. To calculate the TU, the Eq. (1) was used [40]:

$$TU = (1/LC_{50}) \times 100 \tag{1}$$

# 2.5.2. Lactuca sativa

Lactuca sativa seeds (Aurelia cultivar; 95% germination; Topseed brand; Lot no. 065044) without chemical treatment were used for phytotoxicity assays. For each sample, 20 seeds were placed in 90 mm diameter Petri dishes with germination paper as support on the bottom. Four milliliters of samples diluted with mineral water (100%, 75%, 50%, 25%, 12.5% and 6.25% (v/v)) were added and the plates were sealed and stored in the dark for 72 h a 24  $\pm$  1 °C [41]. After the exposure time, the root size (hypocotyl and radicle) of Lactuca sativa seeds that germinated were measured. For the LC<sub>50</sub> calculation, PROBIT analysis was conducted using StatPlus AnalystSoft Version v8. To calculate the TU, the Eq. (1) was used.

The calculation of the Germination Index (GI%) was in accordance with Eqs. (2) and (3).

RGR=RLS/RLC (2)

$$GI(\%) = (RGRx(SGS)/((SGC)))x100$$
(3)

Where: RGR is relative root growth; RLS is the total length of the root of the sample; RLC is the total root length in the negative control; SGS is the number of germinated seeds in the sample and SGC is the number of germinated seeds in the negative control [42,43].

# 3. Results and discussion

#### 3.1. Optimization by Factorial experimental design

Factorial designs based on the Response Surface methodology allow studying the effect of a large number of controlled independent variables (factors) on a measured variable for different levels of the factors (response) with a relatively small number of experiments and simple calculations of the results analysis [44].

The 2<sup>4</sup> factorial experimental design with triplicate in the central point was applied to determine the variables and experimental conditions that most influence the degradation of Ketoprofen in SW, denominated by the acronym SW-KET. Table S2 (supplementary material) shows the results (% degradation and mineralization) for the different experiments.

Considering the percentage of drug removal and the % of mineralization, with the lowest consumption of reagents, the experiment that presented results considered optimal was run 15 (Table S2, supplementary material), with 97.2% of KET degradation in 30 min of experiment (10 mg L<sup>-1</sup> of H<sub>2</sub>O<sub>2</sub>; 15 mg L<sup>-1</sup> of Fe<sup>2+</sup>; RD=0.70 mW s cm<sup>-2</sup>;  $Q_{uv}$ = 8.2 kJ L<sup>-1</sup>) at a 95% significance level.

The effect of the variables on the process is reflected in the values and signals of the estimated regression coefficients, i.e., positive coefficients indicate that the KET degradation efficiency increases with the increase of concentration of the respective variable within the investigated range, while negative coefficients indicate that degradation efficiency increases at lower concentrations. The positive coefficients of the interacted variables indicate a synergistic effect, while the negative coefficients reveal an antagonistic effect. Any effect that crosses the vertical line (reference line) is considered potentially important.

The Pareto chart for KET degradation (Fig. 3) shows that all independent variables and interactions  $[H_2O_2 \ x \ Fe^{2+}]$  and  $[H_2O_2 \ x \ RD]$  were important for the treatment of the effluent by the photo-Fenton process, as they reached the reference line p=0.05.

The RI (kWcm<sup>-2</sup>) was calculated to better evaluate the effect of radiation in relation to the exposure time of the samples [45]. Therefore, for (i) RD = 0.70 mW s cm<sup>-2</sup>; RI = 0.39 kWcm<sup>-2</sup>, (ii) RD = 0.46 mW s cm<sup>-2</sup>; RI = 0.26 kWcm<sup>-2</sup>, (iii) RD = 0.22 mWs cm<sup>-2</sup>; RI = 0.12 kWcm<sup>-2</sup>.

The  $H_2O_2$  concentration and RD were the most statistically significant variables for the photo-Fenton process. The concentration of  $H_2O_2$  significantly interacted with other variables such as RD and Fe<sup>2+</sup>. The



Fig. 3. Pareto Chart for Degradation.

presence of  $Fe^{2+}$  ions is essential for the occurrence of oxidation, as these are catalysts for the decomposition of  $H_2O_2$  leading to the generation of active species, mainly HO<sup>•</sup> [46–49].

The results of [29] demonstrated that the reduction of  $Fe^{3+}$  to  $Fe^{2+}$  in the presence of  $H_2O_2$  is an important catalytic pathway for the Fenton reaction. The variables [ $H_2O_2$ ] and [RD] exhibited positive coefficients, showing that the % of degradation increases as we increase the values of these variables. The interactions [ $H_2O_2$ ] x [Fe] and [RD] x [ $H_2O_2$ ] had negative coefficients, indicating to be antagonistic to the degradation efficiency (Fig. 3).

High concentrations of  $H_2O_2$  can react with HO• and form the weaker oxidants,  $HO_2^{\bullet}$  and  $O_2^{\bullet-}$  (Eqs. 4 and 5). The HO<sub>2</sub> that was produced react with HO<sup>•</sup> and form water and oxygen (Eq. 6) and dimerized to  $H_2O_2$  (Eq. 7) [50]:

$$H_2O_2 + HO^{\bullet} \rightarrow HO_2^{\bullet} + H_2O \tag{4}$$

 $H_2O_2 + HO^{\bullet} \rightarrow O_2^{\bullet-} + H_2O \tag{5}$ 

 $\mathrm{HO}_{2}^{\bullet} + \mathrm{HO}^{\bullet} \to \mathrm{H}_{2}\mathrm{O} + \mathrm{O}_{2} \tag{6}$ 

$$2HO_2^{\bullet} \rightarrow H_2O_2 + O_2 \tag{7}$$

Therefore, a high concentration of  $H_2O_2$  can reduce the oxidative capacity of the photo-Fenton reaction by decreasing the amount of  $HO^{\bullet}$  in the system. This behavior may be responsible for the negative effect of the interaction between [Fe] and  $[H_2O_2]$ .

Similar interaction effect between Fe and  $H_2O_2$  was obtained by [51] for the removal of COD in textile industry effluent by means of advanced photo-Fenton oxidative processes aided by different light sources. [50] also observed the importance of the variables Fe and  $H_2O_2$  and the interaction between them in the photo-Fenton reaction for the degradation of the pesticide carbofuran.

Response surface graphs were generated based on the dependent variable for better evaluation of the degradation behavior (Fig. 4). In the antagonistic relationship between  $[H_2O_2] \times [Fe]$  (Fig. 4a), it was possible to observe that at the lowest concentration of  $H_2O_2$  (10 mg L<sup>-1</sup>) and the highest concentration of  $Fe^{2+}$  (15 mg L<sup>-1</sup>), it was already possible to obtain a high % of KET degradation (97.2%), in 30 min (run 15, Table S2). This antagonistic effect can be observed when only one of the variables is decreased, e.g., for (i)  $[H_2O_2] = 30 \text{ mg L}^{-1}$  and  $[Fe^{2+}] = 15 \text{ mg L}^{-1}$ , in 5 min (run 8, Table S2), there was 79.8% degradation; and for (ii)  $[H_2O_2] = 10 \text{ mg L}^{-1}$  and  $[Fe^{2+}] = 15 \text{ mg L}^{-1}$ , in 5 min (run 7, Table S2), there was an increase to 90.7% in degradation, an effect that shows that at lower concentrations of  $H_2O_2$  we can already observe a high % of degradation. The presence of  $H_2O_2$  allows the regeneration of Fe<sup>2+</sup>, which activates HO•, which in turn attack organic compounds, increasing the % of degradation [52].

The relationship between  $[H_2O_2]$  and RD proved to be antagonistic to the efficiency of the photo-Fenton process (negative coefficient). This can occur due to the high formation of hydroxyl radicals by photo-Fenton and photolysis of hydrogen peroxide, the formation of secondary reactive species that compete with hydroxyl radicals in the degradation of the organic compound, or the formation of unwanted byproducts that can inhibit the reaction. High concentrations of  $H_2O_2$ can scavenge hydroxyl radicals [50,53].

The antagonistic effect of the relationship  $[H_2O_2] \times [RD]$  (Fig. 4b) can be observed, e.g., for (i)  $[H_2O_2] = 10 \text{ mg L}^{-1}$  and  $[RD] = 0.22 \text{ mW s cm}^{-2}$  (RI = 0.12 kWcm<sup>-2</sup>) (run 9, Table S2) resulting in 35.3% degradation, but when we increase only one of the variables, such as the applied radiation, e.g.- (ii)  $[H_2O_2] = 10 \text{ mg L}^{-1}$  and  $[RD] = 0.70 \text{ mW s cm}^{-2}$  (RI = 0.39 kWcm<sup>-2</sup>) (run 13, Table S2), degradation efficiency increases to 82.5%. This demonstrates that a low concentration of  $H_2O_2$  is enough to obtain high KET degradation rates, in this sense, being unnecessary to increase both variables simultaneously.

RI, as well as RD, is a key factor in the photo-Fenton reaction because its interaction with  $H_2O_2$  and with  $Fe^{2+}$  increases the performance in the formation of HO• favoring the attack of organic compounds and destroying the bonds, requiring a lower consumption of reagents, such as  $Fe^{2+}$  and  $H_2O_2$ . Similar results were obtained by [54] who also studied three levels of irradiance (18, 32 and 46 W m<sup>-2</sup>) with 3 concentrations of Fe (8, 20 and  $32 \text{ mg L}^{-1}$ ) in the degradation of the pesticide Pyrimethamyl in the photo-Fenton reaction. The main reaction pathways of HO• with organic compounds include hydrogen abstraction from aliphatic carbon, addition of double bonds and aromatic rings, and electron transfer [37]. The estimated regression coefficients were used to generate an empirical model of the degradation percentage efficiency (%DEG), considering the variables and their mutual relationships %  $DEG = 76.2316 + 15.8375[Fe^{2+}] + 33.0625[H_2O_2] + 20.7125[RD]$ + 17.5625[Time] - 17.3125[Fe<sup>2+</sup>][H<sub>2</sub>O<sub>2</sub>] - 18.0375[H<sub>2</sub>O<sub>2</sub>][RD]; R<sup>2</sup> = 0.9738

These interaction effects between the variables would not be manifested if conventional methods had been used and the variables had their values defined in each experiment, in which resides the advantage of using a factorial design in experiments [55]. The  $R^2$  was 0.9738 indicating that the model was satisfactory, with good fit between experimental and predicted values.

# 3.2. Mineralization of organic matter

The Pareto chart (Fig. 5) showed that H<sub>2</sub>O<sub>2</sub>, RD, Fe<sup>2+</sup> and Time were fundamental to mineralize 39.5% of the organic matter present in the effluent within an irradiation time of 150 min ( $Q_{uv} = 40.9$  kJ L<sup>-1</sup>) (run 15; Table S2), as they were statistically significant exceeding the



Fig. 4. Degradation % Response Surface plots for the significant interactions between  $H_2O_2 \times Fe^{2+}$  (a) and  $H_2O_2 \times RD$  (b).



Standardized Effect Estimate (Absolute Value)

Fig. 5. Pareto Chart for Mineralization.

reference line (p < 0.05). Friedrich et al. (2012) [56] had similar mineralization results (40% mineralization after 120 min of experiment) in their study of the degradation of phenol by Fenton's reaction.

The variables [RD] and  $[H_2O_2]$  stood out because they appeared in first and second place in importance, respectively (Fig. 5). [57] showed the importance of radiation intensity for the photo-Fenton process in the degradation of Diphenhydramine (DPH), using LED UVA lamps.

The variables  $[H_2O_2]$  and  $[Fe^{2+}]$  showed negative signals, indicating that they have an antagonistic effect, and RD and Time variables showed positive signals, indicating synergism. With the increase in RD and consequently RI, the generation of HO• increases, therefore a smaller amount of  $H_2O_2$  will be required, reducing the consumption of reagents [58].

When we have iron at its highest concentration  $[15 \text{ mg L}^{-1}]$ , the irradiation time and the RI can be lower without causing damage to the % of mineralization [54].

For the statistically significant interaction (2by3) (Fig. 5), it was possible to draw up the response surface graph of the % of mineralization between the Fe x RD variables (Fig. 6a). To better understand the variables behavior, a response surface graph was generated for the Fe x  $H_2O_2$  variables (Fig. 6b) where we observe the importance of RI in the process for increasing the mineralization of organic matter in the effluent. The synergistic effect between [Fe<sup>2+</sup>] and [RD] can be seen in (Fig. 6a), with the increase of RD and Fe<sup>2+</sup> concentrations there is an increase in mineralization efficiency. With the increase in the RD there is a significant increase in the percentage of mineralization, as the RD accelerates the process of reduction of Fe<sup>3+</sup> to Fe<sup>2+</sup> and the decomposition of  $H_2O_2$ , consequently generating more HO• (Eqs. 8 and 9), mineralizing the organic matter present in the effluent with greater efficiency.

$$Fe^{2+} + H_2O_2 \rightarrow Fe^{3+} + HO\bullet + OH^-$$
(8)

$$Fe (OH)^{2+} + hv \rightarrow Fe^{2+} + HO \bullet$$
(9)

The estimated regression coefficients were used to generate an empirical model of % mineralization (%MIN) efficiency, considering the variables and their mutual relationships.

%MIN =  $16.9289 - 10.2337[H_2O_2] - 7.4188[Fe^{2+}] + 11.9437[RD] + 8.3087$ [Time] + 7.9837[Fe<sup>2+</sup>][RD]; R<sup>2</sup> = 0.9294

# 3.3. Optimized variables for the effluent with the addition of drugs a) KET b) DCF c) DCF+KET

After the  $2^4$  factorial experimental design with SW-KET, the optimized conditions were applied (run 15, Table S2) for the effluent enriched with the drug DCF (SW-DCF) and for the effluent enriched with the mixture DCF+KET (SW-DCF+KET). Fenton and photo-Fenton processes were performed. UVA and UVA/H<sub>2</sub>O<sub>2</sub> were used as "blanks" and reactions were performed in triplicate.

The photo-Fenton-KET process achieved better performance in the mineralization of organic matter showing that the variables  $Fe^{2+}$ ,  $H_2O_2$  and RI contribute to increase the % of mineralization (Fig. 7a, b and c) and the % of degradation (Fig. 8a, b and c). The regeneration of  $Fe^{2+}$  and the absorption of UV radiation by  $H_2O_2$  contribute to the formation of HO•, which attack organic compounds, increasing the % of mineralization and degradation of drugs in the effluent [59].

Mineralization in the photo-Fenton-KET (39.5%  $\pm$  0.61) (Fig. 7a) and photo-Fenton-DCF (38%  $\pm$  1.5) reactions (Fig. 7b) in the effluent were greater than in the mixture DCF+KET (27.1%  $\pm$  3.53) (Fig. 7c), showing that the greater organic load makes it difficult to transform these compounds into CO<sub>2</sub> and H<sub>2</sub>O.

Light was a fundamental factor in the mineralization of KET (Fig. 7a), because when comparing Fenton with photo-Fenton, an increase in the % of mineralization was verified (23.5–39.5%). The mineralization and degradation of DCF (Fig. 7b and Fig. 8b) and of DCF+KET (Fig. 7c and Fig. 8c) were lower than the percentages of mineralization and degradation of KET (Fig. 7a and Fig. 8a) in all processes used. Similar results were obtained by [60], which obtained a % of mineralization and KET degradation greater than DCF when the drugs were subjected to degradation with different photocatalysts under UV light.

The photo-Fenton process was the most efficient process to degrade the DCF+KET mixture. There was a decrease in the % of degradation for



Fig. 6. Mineralization response surface plots for relationships between  $[Fe^{2+}] \times [RD]$  (a) and  $[Fe^{2+}] \times [H_2O_2]$  (b).



\* Time in minutes for processes Fenton (absence of light)

\* Time in minutes for processes Fenton (absence of light)



\* Time in minutes for processes Fenton (absence of light)

Fig. 7. Mineralization of EB with KET (a). DCF (b) and DCF+KET (c) by AOPs.

the photo-Fenton reactions with the presence of the drug DCF and DCF+KET. This is due to the presence of HO• sequestering agents, such as Cl and N in the structure of the DCF molecule, because according to [61] and [49], they increase the scavenging effects of HO• inhibiting the process and decreasing the degradation percentage, nevertheless, a percentage of 97.2% of degradation was obtained for KET, 72.7%, for DCF, and 66.0% for DCF+KET. We can also consider that when the organic load (DCF+KET) is increased, the degradation efficiency decreases. This can also be explained by the deactivation of the active sites on the catalyst surface, due to adsorption and/or coverage of the active sites, resulting in a decrease in the formation of reactive oxidant species and degradation efficiency [62]. In addition, the increase in the amount of intermediate species formed in the presence of higher concentrations of contaminants compete with oxidative radicals [63].

# 3.4. Results of Physicochemical Parameters after treatments

Turbidity is related to the amount of solids and contaminants that can decrease the light absorption capacity [64]. The presence of blacklight UVA radiation significantly influenced the increase in the % turbidity removal compared to the % removal results in Fenton processes (absence of light). There was a removal of 79%, 91% and 97% of the effluent turbidity for the reactions of photo-Fenton-KET, photo-Fenton-DCF and photo-Fenton-DCF+KET, respectively. COD and BOD are related to the organic matter dissolved in the effluent. All the variables, Fe<sup>2+</sup>, H<sub>2</sub>O<sub>2</sub>, RD and the experiment time were important to improve the physicochemical characteristics of the effluent, in which these variables were statistically significant for the mineralization of the organic load present in the effluent. The photo-Fenton process was the most efficient with high percentages of COD and BOD removal. For COD there were 78%, 62% and 78% organic load removal for photo-Fenton-KET, photo-Fenton-DCF and photo-Fenton-DCF+KET, respectively [51] showed high efficiency of the photo-Fenton reaction for the removal of COD in textile effluent. For the BOD parameter, the photo-Fenton processes reached the reuse water limit established by the Brazilian environmental legislation CONAMA 430/2011 (below 20 mg  $L^{-1}$ ) with a removal of 64% (photo-Fenton-KET), 72% (photo--Fenton-DCF) and 65% (photo-Fenton-DCF+KET). Comparing the COD and BOD results from the Fenton reactions (absence of light) with the photo-Fenton reactions in this study, an increase in the percentage of removal of these parameters was observed when the effluent was treated by the photo-Fenton reaction. For the Fenton reaction, the COD ranged from 30.8% to 64.8% removal and the BOD ranged from zero to 60.2% removal. In photo-Fenton, the COD ranged from 61.5 to 77.8 and the BOD ranged from 64.2% to 71.7% removal.

This can be explained because the Fenton reaction (Fenton, 1894)





Fig. 8. Degradation of KET(a), DCF(b) and DCF+KET(c) in urban domestic effluent by AOPs.

can be significantly improved by the incorporation of irradiation (ultraviolet or visible), which characterizes the photo-Fenton process. This method allows the photochemical regeneration of  $Fe^{2+}$  ions, which

results in greater degradability of the compounds, since Fe<sup>2+</sup> ions act as catalysts in the formation of hydroxyl radicals, which can lead to greater degradation of the pollutant [65].

#### Table 3

Results of physicochemical parameters of effluents with drugs and post treatments with AOPs.

Samples	Conductivity (mS $cm^{-1}$ )	Turbidity (NTU)	$COD (mg L^{-1})$	$\begin{array}{c} \text{BOD} \ (\text{mg} \ \ \ \ \ \ \ \ \ \ \ \ \ \ \ \ \ \ \$	Mineralization (%)	Degradation (%)	Turbidity	Efficiency (%) COD	BOD
SW	$1.8\pm0.1$	$\textbf{45.0} \pm \textbf{1.2}$	229.5 + 7 5	$95.0\pm5.5$	-	-	-	-	-
SW-KET	$\textbf{2.9} \pm \textbf{0.0}$	$21.5 \pm 7.8$	293.0 + 8.1	$\textbf{55.9} \pm \textbf{7.2}$			-		-
SW-DCF	$3.0 \pm 0.1$	$21.5 \pm 7.8$	$\begin{array}{c} 208.0 \\ \pm \ 6.8 \end{array}$	$56.6\pm8.4$	-	-	-	-	-
SW-DCF+KET	$3.1\pm0.2$	$21.5 \pm 7.8$	$\begin{array}{c} 338.1 \\ \pm \ 7.6 \end{array}$	$\textbf{57.0} \pm \textbf{6.6}$	-	-	-	-	-
UVA-KET (blank)	$\textbf{2.8} \pm \textbf{0.1}$	$10.2\pm2.3$	$\begin{array}{c} 128.0 \\ \pm \ 4.3 \end{array}$	$\textbf{72.0} \pm \textbf{5.8}$	$\textbf{8.8} \pm \textbf{3.5}$	$\textbf{56.3} \pm \textbf{4.9}$	52.6	56.3	-
UVA-DCF (blank)	$3.3\pm0.1$	$14.1\pm1.9$	$\begin{array}{c} 160.0 \\ \pm \ 3.8 \end{array}$	$30.0\pm3.5$	$11.0\pm0.2$	$\textbf{47.0} \pm \textbf{0.3}$	34.4	23.1	47.0
UVA-DCF+KET (blank)	$2.1\pm0.2$	$20.0 \pm 2.0$	$\begin{array}{c} 256.0 \\ \pm \ 6.3 \end{array}$	$\textbf{84.0} \pm \textbf{5.2}$	$\textbf{4.4} \pm \textbf{1.2}$	$\textbf{42.2} \pm \textbf{1.4}$	7.0	24.3	-
UVA/H <sub>2</sub> O <sub>2</sub> KET (blank)	$1.4\pm0.3$	$\textbf{9.9} \pm \textbf{4.5}$	$\begin{array}{c} 228.7 \\ \pm \ 5.4 \end{array}$	133.4 ± 4.9	$14.3\pm0.5$	$\textbf{79.0} \pm \textbf{2.8}$	54.0	21.9	-
UVA/H <sub>2</sub> O <sub>2</sub> DCF (blank)	$\textbf{3.4} \pm \textbf{0.9}$	$12.5\pm3.6$	$\begin{array}{c} 144.0 \\ \pm \ 4.5 \end{array}$	$\textbf{27.5} \pm \textbf{2.8}$	$12.0\pm1.5$	$62.0 \pm 0.2$	41.9	30.8	51.4
UVA/H <sub>2</sub> O <sub>2</sub> – DCF+KET (blank)	$1.6\pm0.4$	$\textbf{22.7} \pm \textbf{0.9}$	$\begin{array}{c} 240.0 \\ \pm \ 4.2 \end{array}$	$36.0\pm2.2$	$6.1\pm1.3$	$51.6\pm0.7$	-	29.0	36.8
Fenton -KET	$1.7\pm0.1$	$\textbf{4.7} \pm \textbf{1.7}$	$\begin{array}{c} 103.0 \\ \pm \ 2.2 \end{array}$	$58.2 \pm 2.6$	$23.5\pm3.2$	$\textbf{89.4} \pm \textbf{0.4}$	78.1	64.8	-
Fenton -DCF	$1.5\pm0.2$	$\textbf{4.7} \pm \textbf{0.8}$	$\begin{array}{c} 144.0 \\ \pm \ 2.1 \end{array}$	$\textbf{22.5} \pm \textbf{1.3}$	$33.0\pm7.5$	$68.0 \pm 0.9$	78.1	30.8	60.2
Fenton-DCF+KET	$2.2\pm0.2$	$25.0 \pm 4.2$	$\begin{array}{c} 200.0 \\ \pm \ 3.1 \end{array}$	$40.0\pm0.7$	$22.0\pm5.7$	$59.5 \pm 1.5$	-	40.8	29.8
PF-KET	$8.7\pm0.2$	$\textbf{4.6} \pm \textbf{1.7}$	$65.0 \pm 1.2$	$20.0\pm2.4$	$39.5\pm 6.5$	$\textbf{97.2} \pm \textbf{1.1}$	79.0	77.8	64.2
PF-DCF	$\textbf{9.2}\pm\textbf{0.1}$	$\textbf{2.0} \pm \textbf{0.9}$	$80.0 \pm 0.9$	$16.0 \pm 1.8$	$\textbf{38.0} \pm \textbf{1.5}$	$\textbf{72.7} \pm \textbf{4.9}$	91.0	61.5	71.7
PF-DCF+KET	$\textbf{9.4}\pm\textbf{0.1}$	$\textbf{0.6} \pm \textbf{0.1}$	$80.0 \pm 1.3$	$20.0\pm1.5$	$\textbf{27.1} \pm \textbf{3.5}$	$66.0 \pm 2.5$	97.0	76.3	64.9

 $(\pm) =$  standard deviation

Even decreasing the % of degradation and mineralization of the photo-Fenton process in the effluent with DCF+KET, the physicochemical parameters were not affected and the photo-Fenton process showed greater efficiency in improving the quality of the effluent. Table 3.

## 3.5. Acute Toxicity and Phytotoxicity

#### 3.5.1. Artemia sp

The assessment of the toxicity of the by-products generated by the AOPs is extremely important, especially in processes that do not lead to the total mineralization of the target compound, as by-products that are more toxic than the original substance can result at the end of the process and complement physicochemical analyzes of wastewater quality, as well as provide a deeper understanding of treatment performance [66]. Acute toxicity testing is usually the first step in the investigation of toxicity [67,68]. Larvae of the microcrustaceans Artemia sp. have been widely used because they are easy to handle and show an acute toxicity response to persistent pollutants. [69-71]. Artemia sp. was used to evaluate the toxicity of the initial effluent solution post UASB reactor (SW) (SW = effluent without addition of NSAIDs), 25 mg  $L^{-1}$  of KET (SW-KET), 25 mg  $L^{-1}$  of DCF (SW-DCF) and 12.5 mg  $L^{-1}$  of KET + 12.5 mg L<sup>-1</sup> of DCF (SW-DCF+KET), as well as solutions treated by UVA, UVA/H2O2, Fenton and photo-Fenton advanced oxidation processes for KET, DCF and DCF+KET. After the tests, it was possible to calculate the Toxic Unit (TU) (Fig. 9).

Results were obtained for Artemia sp., with toxicity levels expressed in toxic units (TUs), calculated as  $TU = (1 / LC_{50}) \times 100$ , where  $LC_{50}$  is the concentration that kills half of the tested organisms.

The Fenton process reduced the SW-KET toxicity (Fig. 9a), but not enough to make it non-toxic, indicating that KET degradations do not generate toxic substances and the higher the % of degradation, the lower the toxicity of the effluent. Only photo-Fenton was able to annul the initial toxicity of SW-KET, with a removal of 94%. This decrease is consistent with [72], who demonstrated a decrease in effluent toxicity when the photo-Fenton process was applied. In (Fig. 9b), UVA,  $UVA/H_2O_2$  made the SW-DCF effluent more toxic, indicating that UV radiation generated more toxic substances than those initially present [30,73].

Similar to UVA and UVA/H<sub>2</sub>O<sub>2</sub>, Fenton also increased effluent toxicity after treatment (Fig. 9b). The photo-Fenton process removed 79% of the toxicity, decreasing the TU from 1.4 to 0.3, making the effluent without acute toxicity. The same occurred with the DCF+KET mixture (Fig. 9c), which only the photo-Fenton reaction was able to reduce the toxicity by 97.2%, a level classified as non-toxic (TU=0.4). In this case, we can observe that the interaction of DCF with KET produced more toxic degradation by-products [74] in the UVA, UVA/H<sub>2</sub>O<sub>2</sub> and Fenton reactions (Fig. 9c), evidencing the importance of efficient AOPs for the degradation and removal of pharmaceutical compounds, since these compounds are found mixed in the treated effluents of the treatment stations. KET and DCF are among the NSAIDs with the highest risk of drug interactions that can cause adverse effects [75].

# 3.5.2. Lactuca sativa

The germination percentage of the negative control seeds was above 90% (95–97%) with low variability in the elongation of the radicles in the positive control. The coefficient of variation (CV) of rootlets in the positive control was less than 30% (CV=18–22%), indicating that the assays were valid [76].

For the SW-KET (Fig. 10a), the Fenton process made the effluent with mild toxicity (TU=0.5) and the photo-Fenton process reduced 79% of the toxicity (TU=0.3), leaving the effluent with TU< 0.4, as recommended [40].

For the SW-KET (Fig. 10a), the Fenton process made the effluent with mild toxicity (TU=0.5) and the photo-Fenton process reduced 79% of the toxicity (TU=0.3), leaving the effluent with TU< 0.4, as recommended [40].

For the SW-DCF (Fig. 10b), only the photo-Fenton process was able to remove 99% of the toxicity, making the effluent without acute



Fig. 9. TU and classification of effluent toxicity after treatment with AOPs with bioindicator Artemia sp. \*Classification according to Personne et al., 2003. \* \*values with p-value significance (p < 0.05). No acute toxicity; Sight acute toxicity; Acute toxicity; Acute toxicity; Acute toxicity; Key High acute toxicity; Key High acute toxicity;



Fig. 10. TU and effluent toxicity classification after treatment by AOPs in SW-KET (a), SW-DCF (b), SW-DCF+KET (c) using the bioindicator<br/>Lactuca sativa. \*Classification according to [40]. values with p-value significance (p < 0.05).O No acute toxicity; O Slight acute toxicity; R Acute toxicity; R High acute toxicity; R Very high acute toxicity.

toxicity.

Fig. 10c showed that the interaction between DCF+KET drugs made the effluent less toxic than the effluents with the isolated drugs. The Fenton process was already able to remove the effluent toxicity (80% removal; TU=0.3), but the photo-Fenton efficiency (87% removal; TU=0.2) was even more expressive for Lactuca sativa roots.

The Germination Index (GI) (Fig. 11) is the phytotoxicity index that combines seed germination and root growth, thus reflecting a more complete estimate of toxicity and has been widely used to assess toxicity on seed germination and seed elongation roots [77]. Particularly, GI has been used to assess the toxicity of Lactuca sativa to domestic effluents [78] and several other effluents, such as textile [79] anaerobic digestion [80] and oily effluents [81].

Raw effluent (SW), SW-KET and all treatments for SW-KET showed GI values above 80% (Fig. 11a) which, according to [82], indicates that there was no inhibitory effect. We can highlight the photo-Fenton reaction (Fig. 11a) which presented 97  $\pm$  0.8% of GI, getting closer to the negative control (red line in 100%).

In Fig. 11b only the UVA/H<sub>2</sub>O<sub>2</sub> processes (63.5  $\pm$  3.0%) and Fenton (41.5  $\pm$  7.9%) were below 80% of GI after the initial effluent treatment SW-DCF, making the sample more toxic, confirming that these processes can form more toxic degradation products, as shown in Fig. 11b. However, photo-Fenton also proved to be the best process for reducing toxicity, as indicated by the increase in the GI value (89–92%) relative to SW-DCF after 120 min of treatment. The photo-Fenton process (92.5  $\pm$  11.1%) (Fig. 11c) also proved to be the most efficient for reducing toxicity, because there was no significant inhibitory effect on root growth above 80% as recommended.

Although the toxicity of sulfate anions to Lactuca sativa has been reported in the literature [83], the concentration of sulfate anions used in this study, originating from ferrous sulfate  $(5.9 \times 10^{-5} \text{ M} - 9.87 \times 10^{-5} \text{ M})$ , and sulfuric acid for pH correction, is much lower than that reported in electrooxidation studies that use sodium sulfate as the electrolyte (0.1–0.05 M). Furthermore, since there was a reduction in the toxic

character through the Fenton and photo-Fenton reactions, sulfate anions likely did not contribute to the toxicity.

All other advanced oxidation processes carried out in effluents enriched with DCF and KET proved to be toxic, as they indicated GI values below 80%. Confronting the results of toxicity, both in Artemia sp. and in Lactuca sativa, with the results of the physicochemical parameters it was verified that the high efficiency of removal of the toxicity of the effluents with the drugs by the photo-Fenton process is also in agreement with the % of COD removal, proving that a high amount of organic matter influences the toxicity of the effluent and when the photo-Fenton process reduced the organic matter, from 60% of removal it was already possible to obtain an effluent without toxic characteristics.

# 4. Conclusions

The integration of photo-Fenton and UASB reactor optimized the sewage treatment system, removing organic matter, effluent toxicity, and degrading emerging pollutants efficiently. Within 15 min, over 85% of the KET drug was removed, reaching 97.2% by the experiment's end. Recommended conditions include 15 mg  $L^{-1}$  Fe<sup>2+</sup>, 10 mg  $L^{-1}$  H<sub>2</sub>O<sub>2</sub>, RI = 0.39 kWcm<sup>-2</sup>, 15-minute treatment time, and a flow rate of 12.7 L min<sup>-1</sup>. Factorial experimental design enhanced degradation and mineralization efficiency in the photo-Fenton process. Photo-Fenton effectively remediated pharmaceutical compounds in urban domestic effluents, even with landfill leachate. It reduced toxicity (TU < 0.4) in Artemia sp. and Lactuca sativa and removed physicochemical parameters (79% Turbidity, 61.5% COD and 64% BOD removal). Additionally, assessing physicochemical parameters and effluent toxicity is crucial for evaluating emerging contaminants' impact. Advanced technologies improve treatment processes in large and small cities, enabling efficient removal of conventional and emerging contaminants, reducing toxic byproducts, and enabling safe water reuse and nutrient recovery. Despite requiring careful operation, compact treatment options effectively



Fig. 11. GI (%) of effluent toxicity after treatment with AOPs in SW-KET (a), SW-DCF (b), SW-DCF+KET (c) using the bioindicator Lactuca sativa.

enhance effluent quality regarding contaminants and toxicity.

# CRediT authorship contribution statement

Priscila S. Cavalheri: Conceptualization, Methodology, Experimentation, Validation, Investigation, Writing – original draft, Writing – review & editing. Beatriz dos Santos Machado: Methodology and Experimentation. Thalita Ferreira da Silva: Formal analysis and Methodology and Experimentation. Karla Regina Warszawsk: Formal analysis. Carlos Eduardo Nazário: Resources. Fernando J. C. Magalhães Filho: Writing – review & editing. Rodrigo P. Cavalcante: Writing – review & editing. Silvio César de Oliveira: Resources. Amilcar Machulek Junior: Resources, Writing – review & editing, Supervision.

# **Declaration of Competing Interest**

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

# Data Availability

Data will be made available on request.

#### Acknowledgments

The authors wish to thank the Brazilian funding agencies Conselho

Nacional de Desenvolvimento Científico e Tecnológico (CNPq) Coordenação), de Aperfeiçoamento de Pessoal de Nível Superior (CAPES -Código de Financiamento 001), and Fundação de Apoio ao Desenvolvimento do Ensino, Ciência e Tecnologia do Estado de Mato Grosso do Sul (FUNDECT).

# Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.jece.2023.110699.

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