Photocatalytic oxidation of psychoactive drug duloxetine: degradation kinetics, inorganic ions and phytotoxicity evaluation

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ABSTRACT

Pharmaceutically active compounds, emerging extensively in ecosystems as pollutants, have become an important environmental and public health issue, since they can contaminate drinking water and pose threat to wildlife and human health. Therefore, efforts should be made in order to establish proper methods for their inactivation or elimination in the environment. The photocatalytic oxidation of psychoactive drug Duloxetine (DLX) has been investigated. In the case of heterogeneous photocatalytic oxidation, the effect of TiO₂ P25 concentration (0.1–1 g L⁻¹), initial concentration of H₂O₂ (0.25-0.2 g L⁻¹) and Fe³⁺ (0.00175-0.014 g L⁻¹) and pH of the solution (3-10) on initial reaction rates were evaluated, while for homogeneous photocatalytic oxidation the effect of the amount of H₂O₂ (0.25–0.2 g L⁻¹) and Fe³⁺ (0.00175–0.014 g L⁻¹) were investigated. Additionally, the conversion of the heteroatoms in the molecule of DLX to inorganic ions (NO₃⁻, NH₄⁺, SO₄²⁻) during photocatalytic process has been observed and phytotoxicity testing, using three plant species, was carried out in order to examine the effect of photocatalytic oxidation on the toxicity of DLX. According to the results presented in this study, both heterogeneous and homogeneous photocatalytic oxidation is an efficient methodology for DLX degradation.

Keywords: duloxetine; photocatalytic oxidation; psychoactive drug; urban wastewater

1. Introduction

Pharmaceutical active compounds (PhACs), are turning into an essential ecological and public health issue, because of their expanding use and subsequent release in the aquatic environment and their potential effects on wildlife and humans. Drug residues that have been indentified recently in urban surfaces, groundwater and drinking water (Hofman-Caris et al., 2017; Pereira et al., 2017), they are mostly introduced in the sewage system through excretion of unmetabolized compounds after medical use or inappropriate disposal (Xekoukoulotakis et al. 2010). However, effluents of wastewater treatment plants (WWTPs) are generally recognized as the major emission pathway of pharmaceuticals into the environment (Ramirez et al., 2009), due to the fact that conventional WWTPs are designed to remove solids and nutrients and to reduce the biological oxygen demand of the effluent (Bu et al., 2016). Even though, pharmaceuticals usually do not have toxic impact on aquatic organisms, since they are detected at low or very low concentrations (ng to μg per liter), concerns have been raised for chronic exposure, since their continuous input to the environment exceed their degradation rate, acting, thus, as slightly persistent pollutants (Puckowski et al. 2016).

Psychoactive compounds are a group of drugs used to treat symptoms of neurological disorders such as depression, schizophrenia, bipolar disorder, or anxiety disorders. According to Global Burden of Disease Study, jointly published by World Health Organization (WHO) and other groups, neuropsychiatric disorders have emerged as priority health problems worldwide, projected to be the second most frequent disease by 2020 (Menken, Munsat, and Toole 2000), while Almeida et al. showed that lifetime prevalence of such disorders can reach over 25% (Almeida et al., 1997). Moreover, according to WHO, around 10% of fully grown persons suffer from such disorders at any point in their lifetime (Sara-
ceno 2002). Many previous studies showed that, exposure of aquatic organisms to psychoactive compounds may affect reproduction (Brooks et al., 2003), endocrine function (van der Ven et al., 2006), or photosynthesis (Escher et al. 2006). Furthermore, recent studies showed that psychoactive drugs may alter fish and benthic invertebrates behavioral responses (Rosi-Marshall et al., 2015), and also, induce fish gene expression profiles associated with human idiopathic autism (Thomas et al., 2012).

During the current economic crisis in Greece, the consumption of psychoactive substances has risen significantly, resulting in an increase in their concentration in urban wastewater (Thomaidis et al., 2016). The presence of psychoactive substances and/or their metabolites is confirmed, not only in urban wastewater, but also in aquatic plants and animal tissues (Rodayan et al., 2016; Chu et al., 2007), underlining the possible consequences for human health and ecosystems balance. Pollution of European waters by pharmaceuticals is a ubiquitous phenomenon (Tiedeken et al., 2017; Barbosa et al., 2016) and is receiving great attention, as stated in the EU Directive 2013/39/EU19 on priority substances in the field of water policy (EU 2013). Notwithstanding, wastewater management remains an open issue for the members of the European Union, since legislations is incomplete and in need of update (et al., 2012). Given the increasing interest about aquatic environment contamination by psychoactive compounds, legislation has recently begun to acknowledge this potential problem in order for environmental regulations to require explicit testing for any PhACs in water bodies.

In order to solve various problems encountered by conventional methods for the degradation of organic species resistant to them, many researchers have turned their attention to a particular group of oxidation techniques called Advanced Oxidation Processes (AOPs). Among AOPs, heterogeneous and homogeneous photocatalytic oxidation has been studied extensively and it has been demonstrated that can be an alternative to conventional methods for the removal of organic pollutants from water and air (Thakur et al., 2017; Bogatu et al., 2017). Meanwhile, photocatalytic oxidation has the potential to use solar energy, integrating, in this way, mild energy into environment protection.

The aim of this work was the application of the preceding methods at the degradation of psychoactive drug Duloxetine (DLX), a serotonin norepinephrine reuptake inhibitor (SNRI) widely used for patients with depression and neuropathic pain. The study provides a further insight to oxidation kinetics (i.e. determination of factors affecting oxidation rate) and estimates mineralization degree towards the corresponding oxidation process. In addition, the conversion of the heteroatoms in the molecule of DLX to inorganic ions (NO\textsubscript{3}\textsuperscript{-}, NH\textsubscript{4}\textsuperscript{+}, SO\textsubscript{4}\textsuperscript{2-}) during photocatalytic process has been observed and phytotoxicity testing, using three plant species, was carried out in order to examine the effect of photocatalytic oxidation on the toxicity of DLX.

2. Material and Methods

2.1 Reagents

Duloxetine hydrochloride was product of Sigma-Aldrich and was used as received. NaOH and HCl were used to adjust the pH when necessary. Commercially available TiO\textsubscript{2} catalyst from Evonik, Aerioxide TiO\textsubscript{2} P25 (TiO\textsubscript{2} P25, 70% anatase and 30% rutile with BET surface area of 55 ± 15 m\textsuperscript{2} g\textsuperscript{-1}) was employed in this study. Iron chloride (FeCl\textsubscript{3}·6H\textsubscript{2}O) and hydrogen peroxide (H\textsubscript{2}O\textsubscript{2}, 30% w/v) were purchased from Alfa Aesar and Panreac Quimica, respectively, and they were used without further purification. Double distilled water was used throughout the experimental processes.

2.2 Experimental set-up and procedures

A thermostated pyrex cell of 0.25 L capacity used as the reaction vessel, was fitted centrally with UV-A or visible light of identical dimensions and geometry, and covered with a black cloth to avoid interactions with ambient light. An Osram Dulux® S blue UV-A lamp (9W/78, 350–400 nm) and an Osram Dulux® S blue lamp (9W/71, 400–550 nm) were used as artificial light sources. The photon flux emitted from the lamps was determined actinometry using the potassium ferrioxalate [K\textsubscript{3}Fe(C\textsubscript{2}O\textsubscript{4})\textsubscript{3}·3H\textsubscript{2}O] method (Braun A.M. 1991). The radiation intensities obtained for the UV-A
and the visible lamp were 1.268·10⁻⁴ E min⁻¹ L⁻¹ and 0.724·10⁻⁴ E min⁻¹ L⁻¹, respectively.

In order to check reproducibility of the results, all photocatalytic experiments were performed in duplicate and, in some cases, in triplicate, with the standard deviation never to exceed 5% for degradation experiments and 10% for DOC and inorganic ions analysis.

It must be mentioned that DLX concentration employed in all experiments was substantially higher than the typically one found in environmental samples, in order to assess treatment efficiency within a measurable time scale and accurate determine residual concentrations with the analytical techniques employed in this work.

2.3 Analytical methods
Sample absorbance was scanned in the 200-400 nm wavelength region on a Shimadzu UV-1700 spectrophotometer and changes in the concentration of DLX were monitored via its characteristic absorption band at 289 nm. A Shimadzu V_CSH Total Organic Carbon Analyzer was used for following mineralization by measuring the dissolved organic carbon (DOC), while pH was determined with a Mettler Toledo S20 SevenEasy pH meter. Colorimetric determination with spectrophotometric detection with titanium (IV) oxysulfate-sulfuric acid solution according to DIN 38409 H15, a method based on the formation of a yellow complex from the reaction of titanium (IV) oxysulfate with H₂O₂, was used for residual hydrogen determination.

A Shimadzu system, comprising of an LC-10 AD pump, a CTO-10A column oven and a CDD-6A conductivity detector (0.25 μL flow-cell), was used for inorganic anions determination. Anions separation was performed on an Alltech Allsep column preceded by a guard column of the same material, using a mixture of phthalic acid and lithium hydroxide at 1.5 mL min⁻¹ constant flow. Temperature of column and conductivity cell were constantly held at 35 and 38°C, respectively. Before LC analysis mobile phases were degassed with helium stream. An ion chromatography (IC) unit LC 20 AD (Shimadzu), consisted of an LC-20 pump, an ADsp degasser, a DGU-20A5, a CDD-10 A VP conductivity detector and a CTO-20A oven, was used for inorganic cations identification. Cations separation was held on an IC YK-421 analytical column, with a mobile phase of boric acid, tartaric acid and dipicolinic acid, through adsorption and desorption processes and detected based on their elution time. Before their injection in the column, samples were filtered through a 0.45 mm membrane filter.

2.4 Phytotoxicity evaluation
Phytotoxicity measurements performed in order to determine photocatalytic efficacy to decrease DLX toxicity, using the standard Phytotoxkit microbiobest, a bioassay based on three species of higher plants: the monocotyl Sorgho (Sorghum saccharatum), the dicotyl garden cress (Lepidium sativum) and the dicotyl mustard (Sinapis alba).

This assay measures both the decrease/absence of seed germination and the decrease of roots and/or shoots after 3 days of exposure of the seeds to toxicants or contaminated soils. For this purpose, reference soil (an OECD analogous artificial soil composed of sand, kaolin, peat and pH adjusted with CaCO₃) was added in the lower compartment of the test plates and hydrated with the samples, while control tests were prepared using distilled water. The soil surface was flatten and covered by paper filter. Tests were carried out in three replicates for each sample and for each type of plant. The seeds were left in a dark incubator at 25 °C for 3 days prior to recording and interpretation of results. Digital pictures of the plates were analyzed using ImageJ v1.49 software (Wayne Rasband, national Institutes of Health, USA). The test results were evaluated comparing the mean number of germinated seeds and the mean root/shoot length for the three replicates in the control and in each examined sample. The percentage effect of the tested effluents on seed germination inhibition (GI), root growth inhibition (RGI) and shoot growth inhibition (SGI) was calculated applying the following formula.

% effect = \( \frac{A-B}{A} \times 100 \)  

where A represents the average number of germinated seeds or the average root length in the control water and B represents the same parameters for the tested solution.

3. Results and Discussion
3.1 Heterogeneous photocatalytic oxidation of DLX
3.1.1 Effect of catalyst’s loading

The degradation of an organic compound may influence strongly by TiO₂ dosage, since this parameter is an important factor in slurry photocatalytic processes. In Figure 1 is depicted the effect of varying the quantity of TiO₂ P25 on the observed initial reaction rate \( r_0 \) of DLX degradation and mineralization degree. The proper catalyst amounts were added to 0.01 g L\(^{-1}\) DLX solutions in natural pH (~5.5) and then irradiated with UV-A light. The pH value falls about 1.5 unit after the photocatalytic process. It should be noticed that, preliminary experiments with DLX solution containing TiO₂ in the dark or illuminated DLX solution without the presence of catalyst showed that both illumination and catalyst are essential for the degradation of the drug (data not shown).

The values of the initial degradation rate and mineralization degree in relation to catalyst dosage are depicted in Figure 1. The initial reaction rates were calculated by a linear fit of C-t data obtained during the first minutes of illumination, in order to minimize variations resulting the competitive effects of the intermediate products and pH changes. As it can be seen, the increase of TiO₂ loading from 0.1 to 0.75 g L\(^{-1}\) led to an increase in the initial reaction rate. Moreover, DOC was 95.9% reduced after 120 min of illumination, when heterogeneous photocatalytic oxidation was conducted in the presence of 1 g L\(^{-1}\) TiO₂ P25. The observed trend can be attributed to the evidence that a catalyst dose increase can provide larger surface area for DLX oxidation and, additionally, may increase the active sites amount that can also provide an enhancement to the photocatalytic efficiency (Sriwichai et al., 2014). Nonetheless, above a certain value a slightly decrease in the reaction rate is observed, a behavior similar to our previous studies (Tsoumachidou et al., 2017; Tsoumachidou, Velegraki et al., 2016), which may be due to a possible aggregation of the catalyst (particle-particle interactions) which result in a loss of surface area available for light-harvesting (Sakhivel et al., 2003). Furthermore, there may be an increase in the opacity of the solution leading to a decrease of the light penetration causing the lesser activation of the catalyst resulting in marginal change or even reduced degradation due to the negative contributions of aggregation or reduced energy (Sriwichai et al., 2014; Babuponmusami et al., 2014).

![Figure 1](image.png)

**Figure 1.** Dependence of the initial reaction rate for constant DLX concentration, \( r_0 \) (■) on the concentration of the TiO₂ P25 and DLX mineralization degree (●) at 120 min of photocatalytic process (initial conditions: 0.01 g L\(^{-1}\) DLX, pH ≈ 5.5, T = 25 °C, UV-A illumination).

Considering the above, in order to ensure total absorption of efficient photons and avoid excess catalyst, the optimum catalyst dosage must be determined (Gaya et al., 2008). As shown in Figure 1 the optimum value is approximately 0.75 g L\(^{-1}\). Observing that with lower catalyst concentration was obtained a good efficiency and considering the necessity of a good balance between process efficiency and experimental costs, in the following
experimental runs the catalyst dosage was chosen to be 0.25 g L\(^{-1}\).

### 3.1.2 Effect of electron scavengers

The acceleration of the photocatalytic oxidation by the addition of oxidizing species (e.g. H\(_2\)O\(_2\)) with the potential to capture the photogenerated electrons is an extensively studied procedure (Liu et al., 2016); due to various individual phenomena that are taking place, such as increment of trapped e\(^-\), avoiding, thus, the e\(^-\)/h\(^+\) recombination, enhanced HO\(^-\) and secondary oxidizing species generation, intermediate compounds oxidation rate increment and reduction of potential problems caused by low O\(_2\) concentration decrease (Boroski et al. 2009). It is established that, H\(_2\)O\(_2\) gets a photogenerated electron from the conduction band, promoting, therefore, the charge separation, while, it can also produce HO\(^-\) via superoxide. However, too high peroxide dosage can promote a negative effect on organic pollutants oxidation, since H\(_2\)O\(_2\) is also a semiconductor valence band holes and HO\(^-\) scavenger. Therefore, as have been observed in previous studies, the effect of oxidant additives, such as hydrogen peroxide, can be conflicting depending on the amount and the particular experimental conditions (Rajeshwar et al., 2008).

![Figure 2](image.png)

**Figure 2.** Effect of H\(_2\)O\(_2\) (A) and ferric ions (B) concentration on the initial reaction rate of DLX degradation (●) and DLX mineralization degree (■) at 120 min and 90 min, respectively, of photocatalytic process (initial conditions: 0.01 g L\(^{-1}\) DLX, 0.25 g L\(^{-1}\) TiO\(_2\) P25, pH=5.5, T=25°C, UV-A illumination).

In order to investigate H\(_2\)O\(_2\) role, photocatalytic experimental runs for the oxidation of 0.01 g L\(^{-1}\) DLX in the presence of 0.25 g L\(^{-1}\) TiO\(_2\) P25, were performed with different H\(_2\)O\(_2\) dosage and at the natural pH value of drug solutions. In **Figure 2A** is depicted the effect of H\(_2\)O\(_2\) addition on TiO\(_2\)-mediated photocatalytic mineralization of DLX. In all H\(_2\)O\(_2\)-induced experiments presented in **Figure 2A** over 80% of DLX was eliminated within 20 minutes of reaction. An increase in the concentration of H\(_2\)O\(_2\) from 0.025 to 0.075 g L\(^{-1}\) led to a small increase in the oxidation rate from 0.31 to 0.53 g L\(^{-1}\) min\(^{-1}\), while higher H\(_2\)O\(_2\) dosage resulted to a slightly \(r_o\) reduction down to 0.38 g L\(^{-1}\) min\(^{-1}\), a behavior that has been also obtained in similar studies. Furthermore, mineralization experimental runs showed that DLX dissolved organic carbon was almost 95% reduced after 120 min of illumination for all H\(_2\)O\(_2\) concentrations studied.

Metal ions addition has also been proven to increase photocatalytic effectiveness, since they are able to trap, separate and transfer the photogenerated electrons and holes in semiconductors (Rincón et al., 2007). In an attempt to enhance the efficiency of the TiO\(_2\)-induced photocatalytic process, iron in the form of Fe\(^{3+}\) was introduced in the reaction vessel prior the beginning of the reaction. The facilitation by Fe\(^{3+}\) at DLX degradation and mineralization is shown in **Figure 2B**. The observed enhancing effect could be attributed to interactions between iron species and TiO\(_2\) (Nahar et al., 2009) the efficient electron-hole pair separation, since the addition of suitable scavengers can suppress the photocatalytic process efficiency limitation occurred by the recombination of photogenerated electrons and holes. Ferric ions act as a photo-generated e\(^-\) or h\(^+\) trap, inhibiting, thus, their recombination and enhancing their lifetimes (Sriwichai et al. 2014). Furthermore, Fe\(^{3+}\) reduction to Fe\(^{2+}\) by a photo-generated e\(^-\) in TiO\(_2\) particles can also conduce to the
suppression of electron-hole recombination (Měšťánková et al., 2009).

### 3.1.3 Effect of solution's pH

The pH of the solution seems to have a significant role in the heterogeneous photocatalytic process efficiency (Mir et al., 2014; Lin et al., 2011), since surface charge state and flat band potential, among other catalyst's properties, appear to have a high pH dependence. Additionally, the degradation rate may be enhanced or inhibited by catalyst’s surface and organic molecule electrostatic attraction or repulsion (Helali et al., 2013). Moreover, the size of TiO₂ particles, the catalyst's interaction with solvent molecules and the type of radicals or intermediates formed during photocatalytic process, may be pH-affected. All aforementioned factors can affect organic molecules adsorption onto catalyst and, consequently, influence the observed degradation rates (Tizaoui et al., 2011).

![Figure 3](image-url)

**Figure 3.** Effect of pH on the initial reaction rate of DLX degradation (●) and DLX mineralization degree (●) at 120 min of photocatalytic process (initial conditions: 0.01 g L⁻¹ DLX, 0.25 g L⁻¹ TiO₂ P25, pH ≈ 5.5, T = 25 °C, UV-A illumination).

Solution's initial pH impact on photocatalytic oxidation was studied by performing the experiments with 0.25 g L⁻¹ TiO₂ P25 in 0.01 g L⁻¹ DLX solution, under UV-A illumination and adjusting the solution pH from acidic values to alkaline ones; the results on degradation rate of DLX and its DOC reduction after 120 min of illumination are demonstrated in Figure 3. Usually, at pH values below its pKₐ, an organic compound exists as neutral species or in a cationic form, whereas above this value, gets a negative charge. For that reason and considering that at this area the catalyst's surface is positively charged, it is not likely to achieve great electrostatic attractions between the surface of the catalyst and the organic molecules. With pH increment, substance's negative charging is getting stronger and, therefore, the electrostatic interactions are getting more remarkable, causing, this way, an enhancement in the photocatalytic efficiency (Ahmed et al., 2010). In this case, as it can be seen from Figure 3, when the pH value decreases from 7 to 3 an increase of the DLX degradation rate is observed. Although opposite to the expected, this phenomena can be explained by the fact that DLX has be found in previous studies to be unstable in acidic media (Datar et al., 2014; Sinha et al., 2009).

### 3.2 Homogeneous photocatalytic oxidation of DLX

Fenton reagent is a drawing attention oxidative system, since hydrogen peroxide is wieldy and environmental safe and iron is a very abundant and non-toxic material (Carneiro, Nogueira, and Zanoni 2007). Considering its high ability to generate HO⁻ in a very simple way, as a result of H₂O₂ decomposition by Fe²⁺ in acidic medium, Fenton method
has been widely applied for organic compounds oxidation (Baba et al. 2015). The advantages of photo-Fenton process are the safe and environmentally-benign nature of reagents and relatively simple operating principles as well as short reaction time and the absence of mass transfer limitations (Mirzaei et al. 2017). In addition, with the use of light (UV-A or visible, artificial or natural) the process can be catalytic, since the photo-reduction of Fe$^{3+}$ to Fe$^{2+}$ produces additional hydroxyl radicals and leads to the catalyst regeneration (Eq. (3)) (Baba et al., 2015).

$$\text{Fe}^{2+} + \text{H}_2\text{O}_2 \rightarrow \text{Fe}^{3+} + \text{OH}^- + \text{HO}^+ \quad (2)$$

$$\text{Fe}^{3+} + \text{H}_2\text{O} + \text{h} \rightarrow \text{Fe}^{2+} + \text{H}^+ + \text{HO}^- \quad (3)$$

One of the main parameters that influence the Fenton and photo-Fenton processes is the iron amount. In the majority of cases, the photocatalytic effect is enhanced when the iron concentration is increased, since more hydroxyl radicals’ production is obtained (Abdessalem et al., 2010; Tsoumachidou, Lambropoulou et al., 2017; Malato et al., 2009).

As the iron concentration increases, the regeneration of Fe$^{2+}$ from Fe$^{3+}$ results in the rapid production of additional HO$^-$ (Abdessalem et al., 2010). However, with too high ferric amount in the solution, dark zones can be generated in the photoreactor, since the incident ray is attenuated too strongly along the optical pathlength, reducing, for that reason, the process effectiveness. Moreover, although more radicals can be produced (Eq. (3)), they can be scavenged by reacting with other ferrous ions, which lead to the reduction of their amount (Xu et al., 2009).

As it can be seen in Figure 4A, the increment of ferric ions from 0.00175 to 0.014 g L$^{-1}$ led to an increase on the photo-Fenton induced mineralization percentage of DLX, from 78.7 to 94.8% when the solution was illuminated for 120 min with UV-A light and from 70.8 to 94% under visible illumination. In all homogeneous photocatalytic experimental runs the pH was low enough (pH = 3.2-3.3), since iron precipitates at higher pH.

Iron concentration has proven to be a significant parameter, not only because it can affect the capital costs, but also the operating costs, since shorter reaction times are required (Ioanou et al., 2013). In the presence of high iron amount, the issue of the iron-separation step at the end of the photocatalytic process appears. Consequently, it is considered critical to choose the proper iron concentration in order to achieve as short reaction time as possible, and at the same time not to exceed the limits defined by the direct discharge to biological municipal WWTPs regulations (different amounts are permitted in the EU, USA, Switzerland, etc.) (Gernjak et al., 2003).

It should be mentioned that DLX organic load reduction was found to be always faster in the early stages of the reaction than in the later ones, maybe due to the fact that iron ions catalyse H$_2$O$_2$ to produce HO$^-$ quickly, in the first stages of the photocatalytic oxidation (Yang, Hu, and Ito 1998).

![Figure 4.A](image_url)

**Figure 4.** Influence of ferric ions (A) and H$_2$O$_2$ (B) concentration on the degree of mineralization of DLX under UV-A (●) and visible (●) irradiation after 120 min (initial conditions: (A) 0.01 g L$^{-1}$ DLX, 0.1 g L$^{-1}$ H$_2$O$_2$, pH ≈ 3.3, T = 25 °C (B) 0.01 g L$^{-1}$ DLX, 0.005 g L$^{-1}$ Fe$^{3+}$, pH ≈ 3.3, T=25°C.

The influence of H$_2$O$_2$ concentration was investigated by several previous studies (Navarro et al., 2011;
Abdessalem et al., 2010), with the main findings to be that neither too low H₂O₂ amount nor too high may be applied, since in the first case a Fenton reaction rate reduction is occurred, while, in the second H₂O₂ competes successfully for HO• and becomes decomposed into molecular oxygen and water, while oxidizing the pollutant. However, in general, there is a rather broad concentration interval between both extremes, where none of those two phenomena occurs (Malato et al., 2009).

As shown in Figure 4B, the use of extra H₂O₂ dosage did not result to any remarkable improvement in both UV-A and visible-irradiated process efficiency of DLX photo-Fenton mineralization. Even though, with the oxidant decomposition to be enhanced, a higher amount of HO− is expected, its increment can provide an adverse effect by slowing down the oxidation process, since the competition interactions are enhanced in the case of H₂O₂ excess (Doumic et al. 2015); mainly the recombination of the produced HO−, as well as their recombination with H₂O₂, contributing to the HO− scavenging capacity (Eqs. (4)-(6)) (Navarro et al., 2011; Abdessalem et al., 2010).

\[
\begin{align*}
    \text{HO}^\cdot + \text{HO}^\cdot & \rightarrow \text{H}_2\text{O} \\
    \text{H}_2\text{O}_2 + \text{HO}^\cdot & \rightarrow \text{H}_2\text{O} + \text{HO}_2^\cdot \\
    \text{HO}_2^\cdot + \text{HO}^\cdot & \rightarrow \text{H}_2\text{O} + \text{O}_2
\end{align*}
\]

(4)-(6)

### 3.3 Inorganic ions

Temporal profiles of sulfur (released as sulfate ions) and inorganic nitrogen (i.e. nitrates, nitrites and ammonium ions) released during heterogeneous and homogeneous photocatalytic mineralization of DLX are shown in Figure 5.

Profiles do not show actual concentrations but they correspond to the percentage of the theoretical amount of sulfur or nitrogen initially present in the molecule.

The heteroatoms contained in the organic compounds are readily converted to inorganic ions that remain in the liquid phase. In some cases, photooxidation of nitrogen is relatively slow and the mineralization of organic pollutants containing nitrogen seems to be complex, since nitrate, nitrite, ammonium ions and free nitrogen are formed (Nohara et al., 1997). As it can be seen in Figure 5A, in the first 30 min of heterogeneous photocatalytic oxidation only 25% of organic nitrogen has converted into inorganic, while total conversion occurred after 120 min of process. In the case of homogeneous photocatalytic mineralization, a complete conversion has not achieved, but almost 70% of organic nitrogen was converted after 120 min (Figure 5B).

On the other hand, high percentage of organic sulfur is converted to inorganic, for both processes, while with photo-Fenton-induced oxidation almost total amount of sulfur is converted to inorganic within the first 30 min of reaction. The decrease of S²⁻ amount, in the case of homogeneous photocatalytic oxidation, may be due to the fact that iron forms a mixture of FeSO₄²⁻ and Fe(SO₄)₃⁻ complexes in the presence of sulfur (Pignatello, Oliveros, and MacKay 2006).

![Figure 5](image_url)

**Figure 5.** Release of inorganic anions during heterogeneous (A) and homogeneous (B) photocatalytic mineralization of DLX (initial conditions: (A) 0.01 g L⁻¹ DLX, 0.25 g L⁻¹ TiO₂ P25, 0.1 g L⁻¹ H₂O₂, pH ≈ 5.5, T = 25 °C, UV-A illumination, (B) 0.01 g L⁻¹ DLX, 0.007 g L⁻¹ Fe²⁺, 0.1 g L⁻¹ H₂O₂, pH = 3.3, T = 25 °C, UV-A illumination).
3.4 Phytotoxicity evaluation

For the investigation of the effect of the TiO$_2$ photocatalytic process on DLX phytotoxicity, samples of the initial drug solution (raw DLA) and at the end of the process (treated DLX) were selected. It should be noted that there was not any inhibition effects by the H$_2$O$_2$, since residual hydrogen peroxide was measured during photocatalytic oxidation and found to be totally eliminated after the reaction. The effects of DLX samples on the plants are depicted in Figure 6 and they are expressed as root growth inhibition (RGI), shoot growth inhibition (SGI) and seed germination inhibition (GI).

As it can be seen from Figure 6A, both TiO$_2$ and photo-Fenton-induced photocatalytic processes improved the...
phyto-compatibility of DLX, since exposure to the treated solutions affected the soot length of Sinapis alba. The raw DLX solution exerted higher toxicity to the above species as it brought on 17.3% shoot growth inhibition. Furthermore, the seed germination of Lepidium sativum was affected, reaching 14.8% after incubation with raw DLX and total toxicity elimination with both treated solutions (Figure 6B).

On the contrary, root and shoot growth germination of Lepidium sativum was negatively affected to photo-Fenton-treated solution exposure, since raw DLX exerted no toxicity to the above parameters, while the treated solution brought a toxic effect of 8.6 and 1.5%, respectively. This observation indicates the need for further photocatalytic processing of DLX, since, apparently, the metabolites that are formed during drug’s degradation are more toxic than the parent compound. The same conclusion is derived from Figure 6C, where it can be seen that the toxic effect at seed germination of Sorghum saccharatum increased from 3.3%, that was after the exposure to raw DLW, to 6.7 and 13.3% after incubation with treated DLX (heterogeneous and homogeneous photocatalytic oxidation, respectively).

4. Conclusions

In the present work the photocatalytic oxidative degradation of DLX, a psychoactive drug detected in urban wastewater, has been demonstrated.

With the use of TiO\textsubscript{2} P25 as photocatalyst under UV-A illumination, quantitative mineralization of the organic molecule occur after 2 hours, while as the catalyst dosage increased the initial reaction rate of the drug degradation increased.

In all experimental runs that H\textsubscript{2}O\textsubscript{2} was added in the drug solution over 80% of DLX was eliminated within 20 minutes of reaction. An increase in the concentration of H\textsubscript{2}O\textsubscript{2} led to a small increase in the reaction rate of oxidation, while a higher concentration of H\textsubscript{2}O\textsubscript{2} resulted to a slightly reaction rate reduction. Moreover, mineralization experiments showed that DOC of DLX was almost 95% reduced after 120 min of illumination for all H\textsubscript{2}O\textsubscript{2} concentrations studied.

No remarkable enhancement was obtained by the attempt to enhance the efficiency of the TiO\textsubscript{2}-induced heterogeneous photocatalytic process by introducing Fe\textsuperscript{3+} in the reaction solution. The study of the effect of drug solution's pH showed that the higher degradation rate occurred at pH 3.

In the case of homogeneous photocatalytic oxidation, the increment of ferric ions led to an increase of DLX mineralization percentage in both UV-A and visible-irradiated process efficiency. On the other hand, the use of extra H\textsubscript{2}O\textsubscript{2} dosage did not result to any remarkable improvement in DLX photo-Fenton-induced mineralization.

Inorganic ions formation analysis showed that total organic nitrogen was converted into inorganic after 120 min of heterogeneous photocatalytic oxidation, while in the case of homogeneous process, almost 70% of organic nitrogen was converted after the same illumination time. On the other hand, high percentage of organic sulfur was converted to inorganic, for both processes, while with photo-Fenton-induced oxidation almost total amount of sulfur is converted to inorganic within the first 30 min of reaction.

Phytotoxicity evaluation showed that both TiO\textsubscript{2} and photo-Fenton-induced photocatalytic processes improved the phyto-compatibility of DLX, since exposure to the treated DLX solutions affected the soot length of Sinapis alba. Moreover, the seed germination of Lepidium sativum was affected, achieving total toxicity elimination with both treated solutions. On the other hand, root and shoot growth germination was negatively affected to photo-Fenton-treated solution exposure, since raw DLX exerted no toxicity to the above parameters, while the treated solution brought a toxic effect of 8.6 and 1.5%, respectively. Additionally, the toxic effect at seed germination of Sorghum saccharatum increased after incubation with treated DLX. According to the above, the need for further photocatalytic processing of DLX is required, since, apparently, the metabolites that are formed during drug’s degradation are more toxic than the parent compound.
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References

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