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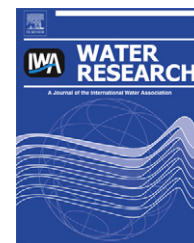
Photodegradation of naproxen and its photoproducts in aqueous solution at 254 nm: A kinetic investigation

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ABSTRACT

The kinetics of photodegradation of the non steroidal anti-inflammatory drug naproxen (+)-S-2-(6-methoxynaphthalen-2-yl)propanoic acid, an emerging organic pollutant, was studied in aqueous solutions under deaerated and aerated conditions. The photolysis experiments were carried out under monochromatic irradiation (λ = 254 nm) at pH = 7.0 and T = 25 °C.

Simplified reaction schemes of photodegradation of naproxen are proposed in absence and in presence of oxygen respectively. The schemes take into account the photolysis of naproxen and its photoproducts and the reactions of the measured species with oxygen dissolved in the liquid bulk. According to these schemes, two kinetic models were developed which correlate the experimental data, for runs performed in absence and in presence of oxygen, with a fair accuracy and allowed to estimate the best values for the unknown kinetic parameters.

The calculated quantum yield of direct photolysis of naproxen under deaerated media is in good agreement with the one previously reported. Under aerated conditions, the generation of singlet oxygen has also been taken into account.

The obtained results, under the adopted conditions, indicated a marked influence of dissolved oxygen on the photodegradation rates of naproxen and the relative distribution of the major reaction intermediates.

1. Introduction

The reuse of urban wastewater in agricultural field, after a proper treatment, may represent a suitable solution for water shortage in many parts of the world including the Mediterranean area (Post et al., 2008; Angelakis and Bontoux, 2001; Angelakis et al., 1999).

The more stringent standards indicate that, after a first biological process, the effluents have to be submitted to a disinfection procedure mainly to reduce the presence of pathogenic microrganisms under specified limits (Koivunen, 2005; Kretschmer et al., 2000).

Different processes are currently adopted for water sterilisation before the reuse, mainly UV irradiation, ozonation, chlorination and oxidant dosage as peracetic acid (Gehr et al., 2003; Caretti and Lubello, 2003; Radziminski et al., 2002; Paraskeva and Graham, 2002; Liberti et al., 2000).

Generally the water streams are submitted to one of the above-mentioned processes until the presence of total and faecal coliforms is reduced below the limits imposed in the specific country. It is noteworthy to observe that the conditions adopted for sterilization are quite mild and this would suggest a limited conversion of chemical species present.

Only a little attention, different from country to country, is generally reserved to the presence, at very low concentrations (micrograms/litre or nanograms/liter), of organic micropollutants which can be adsorbed by the plants receiving the treated water streams. The xenobiotics may have bioaccumulating properties when applied to crops through irrigation (Korte et al., 2000; Marty, 1999) or exert toxic effects on the same plants, bacteria and algae (Levine and Asano, 2004; Singh et al., 2004; Dizer et al., 2002; Ono et al., 1996) and may end up in humans through food ingestion (Beek et al., 2000).

Among the classes of micropollutants present in STP effluents, pharmaceuticals have gained during the last decade the attention of many researchers mainly due to the concern of possible effects on living aquatic organisms and humans. Although no conclusive assessments are so far available on this topic, the presence of these unregulated xenobiotics in STP effluents to be reused for irrigation in agricultural field is undesirable based on the precautionary principle. However, before planning the adoption of proper treatments to remove these species from water streams, an evaluation of their fate during the sterilization procedure may be useful. A literature search indicates the existence of some studies in which the reactivity of pharmaceuticals with hypochlorite is investigated at the conditions adopted during the sterilization process for water reuse. For example, it has been reported that acetaminophen is transformed into toxic compounds during chlorination in wastewater treatment plants (Bedner and MacCrehan, 2006a) whereas N-chloramines are produced from the reaction between hypochlorite and fluoxetine or metoprolol (Bedner and MacCrehan, 2006b). Moreover, Della Greca et al. (2009) found that atenolol reacts with hypochlorite with the formation of chlorinated products. Regarding the sterilization based on UV irradiation, Kim et al. (2009a) recently reported the results of an investigation on 30 pharmaceuticals and personal care products found in an effluent from secondary sedimentation tanks in a municipal wastewater treatment plant which were submitted to UV (at 254 nm) treatment. The data given by these authors indicate that at very severe conditions of irradiation with a fluence quite higher than those adopted during the sterilization procedure for reuse, a great part of the studied species were not effectively removed. Moreover, no indications on the photoproducts have been reported in the study.

The presence of naproxen, (+)-S-2-(6-methoxynaphthalen-2-yl)propanoic acid, a non steroidal anti-inflammatory drug (NSAID), has been often documented in different STP effluents at concentration levels between 25 ng l^{-1} and 33.9 µg l^{-1} (Santos et al., 2010; Bueno et al., 2009; Vieno et al., 2005; Andreozzi et al., 2003; Metcalfe et al., 2003; Miao et al., 2002; Ternes et al., 1998).

The ecotoxicity of naproxen and its photoproducts was assayed on some aquatic organisms, such as bacteria, microcrustaceans and algae (Isidori et al., 2005; Della Greca et al., 2004). The reported results indicated that some photoderivatives are more ecotoxic than naproxen, both in acute and chronic conditions, whereas genotoxic and mutagenic effects were not found.

The EC$_{50}$ values of naproxen were respectively 21.2 µg l^{-1} from the test ToxAlert and 35.6 µg l^{-1} with the Microtox test (La Farre et al., 2001).

Its behaviour in aqueous solutions under UV irradiation at a wavelength of 254 nm has been recently investigated (Kim et al., 2009a,b). It has been observed that the removal of naproxen, with UV intensity lamps of 0.384 mW cm^{-2}, was around 30% and 90% in 10 and 73 min respectively. Unfortunately due to the severe conditions adopted, the results achieved during this study could not be used to predict the fate of this species during the sterilization procedure for reuse. Kinetic indications on naproxen removal during irradiation at 254 nm were also reported by others (Pereira et al., 2007a,b; Benitez et al., 2009; Meite et al., 2010). The quantum yield and the photoproduct distribution depend in a sensible way on the adopted conditions (aerobic versus anaerobic medium).

In particular, the quantum yields of direct photolysis (at 254 nm) in water, under aerated media, reported in three previous papers, are not in good agreement: $9.3 \cdot 10^{-3} \pm 2.7 \cdot 10^{-4}$ mol E^{-1} (Pereira et al., 2007a,b), $49.8 \cdot 10^{-3}$ mol E^{-1} (Benitez et al., 2009) and $24.0 \cdot 10^{-3} \pm 5.0 \cdot 10^{-3}$ mol E^{-1} (Meite et al., 2010). These discrepancies in the quantum yields can be probably attributed to the influence of the concentration of dissolved oxygen as observed by Meite and coauthors who, under controlled operative conditions (pH 5.5–6.0, T = 25 °C, photon flux 10^{-7} E l^{-1} s^{-1}), estimated the following values of quantum yields: $1.5 \cdot 10^{-2}$ mol E^{-1}, $2.40 \cdot 10^{-2}$ mol E^{-1} and $5.60 \cdot 10^{-2}$ mol E^{-1} for dissolved oxygen concentrations of 0.01 mM, 0.26 mM and 1.3 mM respectively.

Moreover, no indications were found in the literature on the products formed during the photolysis/photooxidation of naproxen at 254 nm, whereas detailed reaction schemes may be found for its photolysis/photooxidation in aqueous solution under solar UV irradiation conditions (Musa and Eriksson, 2008; Boscà et al., 2001, 1990; Moore and Chappuis, 1988).

These findings demonstrated the harmful effects of the investigated compounds and suggested the opportunity to consider also the photoderivatives in ecotoxicology assessment of naproxen.
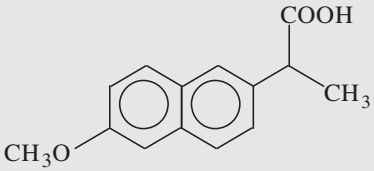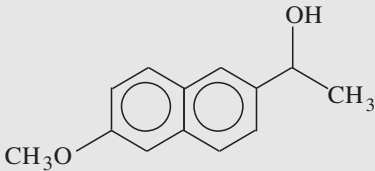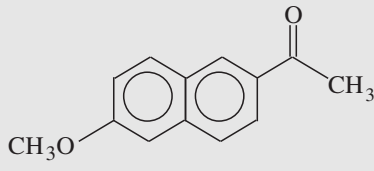
Therefore, the present work aims at a new, and more detailed, characterization of the kinetics of naproxen photodegradation and its photoproducts under UV irradiation at 254 nm in deaerated and aerated aqueous solution at pH = 7.0.

2. Experimental

Naproxen solutions in water were irradiated with a nominal 17 W low-pressure mercury monochromatic lamp emitting at 254 nm (Helios Italquartz) in a 0.420 l (V) annular photoreactor, with a path length (L) equal to 2.20 cm. The reactor was thermostated at 298 K and wrapped with an aluminium foil. At the top, the reactor had more inlets for feeding reactants and gas stream and outlets for withdrawing samples. A scheme of the experimental device has been reported elsewhere (Andreozzi et al., 2000).

The power output of the lamp (I_o) was $2.13 \cdot 10^{-6} \pm 1.12 \cdot 10^{-7}$ E s^{-1} as measured by hydrogen peroxide actinometry (Nicole et al., 1990). The pH of the aqueous solutions was regulated at 7.0 with perchloric acid and/or sodium hydroxide and monitored by means of an Orion 420A^{+} pH-meter (Thermo).

Table 1 – Chemical structure of naproxen and its major photoproducts along with their molar absorptivities at 254 nm.

naproxen
NP
$\varepsilon_{NP}^{254} = 4.18 \cdot 10^3 \ M^{-1} \ cm^{-1}$

1-(6-methoxy-2-naphthyl)ethanol
NP
$\varepsilon_{NP_1}^{254} = 4.35 \cdot 10^3 \ M^{-1} \ cm^{-1}$

2-acetyl-6-methoxynaphthalene
NP
$\varepsilon_{NP_2}^{254} = 3.06 \cdot 10^4 \ M^{-1} \ cm^{-1}$

In all the experiments the solution was preventively sparged with air. During the runs a gaseous stream of air was continuously fed, at a flow rate of $0.3 \ l \ min^{-1}$, to the irradiated magnetically stirred solution.

The concentration of the dissolved oxygen, reported above as $[O_2]$, was calculated according to the Henry's law ($P_{O_2} = H \cdot x$) with $H = 4.38 \cdot 10^4$ atm (Perry and Green, 1997). If an air stream at $P = 1$ atm was used then $x = 4.79 \cdot 10^{-6}$. A value of $2.66 \cdot 10^{-4}$ M for the concentration of oxygen dissolved in the liquid bulk can be thus obtained.

The concentrations of naproxen and its photoproducts, 1-(6-methoxy-2-naphthyl)ethanol and 2-acetyl-6-methoxynaphthalene, named as NP, NP_1 and NP_2 respectively (Table 1), were evaluated by HPLC analysis. For this purpose, the HPLC apparatus (Agilent 1100) was equipped with a diode array UV/Vis detector ($\lambda = 220$ nm) and a Synergi Max-RP column (Phenomenex), using a mobile phase of 60% aqueous buffer and 20% acetonitrile, flowing at $1.0 \ ml \ min^{-1}$. One liter of buffer was made by 10 ml of phosphoric acid solution (5.05 M), 50 ml of methyl alcohol and water for HPLC.

For each experimental run, a starting concentration around 10^{-5} M of the selected substrate was used to prepare the aqueous solution.

The molar absorptivities for NP, NP_1 and NP_2 at $\lambda = 254$ nm (ε_i^{254}) were calculated by measuring the absorbances of solutions at known concentration (pH $= 7.0$) with UV–Vis Diode Array spectrophotometer (HP 8452 A) equipped with a quartz cell of a 1 cm path length (Table 1).

Naproxen was purchased from Sigma Aldrich whereas 1-(6-methoxy-2-naphthyl)ethanol and 2-acetyl-6-methoxynaphthalene from Alfa Aesar Co. with a purity 98% w/w. All reagents were used as received.

The experiments were conducted in duplicate with a resulting standard deviation lower than 3.5% and for each reaction time the average value of concentration was reported in the diagrams.

3. Results and discussion

Preliminary runs were carried out at dark under aerated conditions on solutions containing NP, NP_1 and NP_2 species. No reactivity at all was observed (data not shown).

The results of UV_{254}-photolysis of NP in aerated aqueous solution are shown in Fig. 1a. A complete removal of this species was achieved, at the adopted experimental conditions, after 30 min of irradiation with the formation of NP_1 and NP_2 as major intermediates.

In a second experiment in which the same solution was submitted to UV irradiation under a continuous feeding of pure oxygen, no appreciable changes were observed in the concentration profiles of NP, NP_1 and NP_2 (data not shown).

That is the UV irradiation at 254 nm of naproxen, in aerated or oxygenated aqueous solutions, converted it mainly into NP_1 which is one of its possible intermediate and this was successively transformed into the corresponding ketone (NP_2).
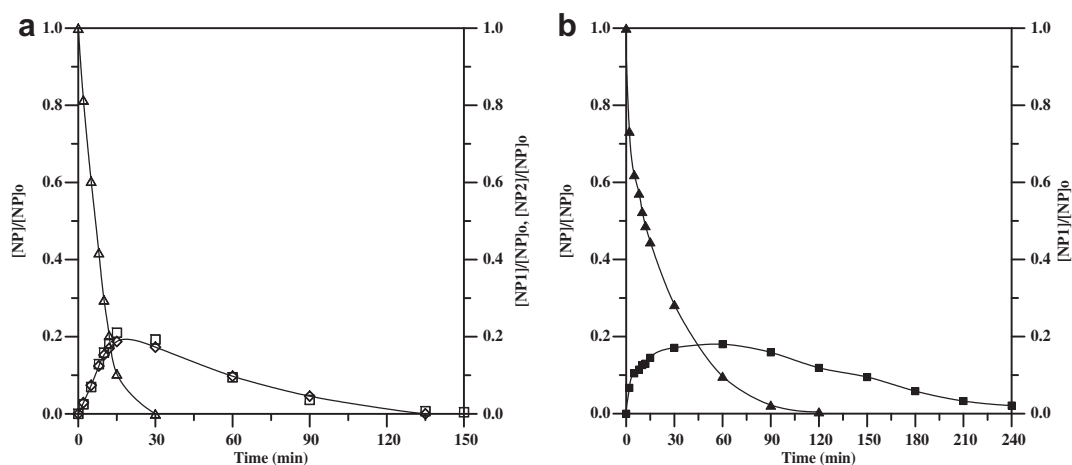
Fig. 1 – UV-photolysis of naproxen under aerated (fig. 1a) and deaerated (fig. 1b) conditions. pH = 7.0. T = 25 °C $[NP]_o = 2.3 \cdot 10^{-5}$ M ($\triangle$, $\blacktriangle$) NP, ($\square$, $\blacksquare$) NP_1, ($\diamond$) NP_2.

These results are in agreement with those reported by others on the photodegradation of naproxen but for wavelength irradiations higher than 300 nm (Boscà et al., 1990; Costanzo et al., 1989; Moore and Chappuis, 1988).

To investigate the role exerted by dissolved oxygen, some runs were repeated after having purged oxygen from the solution through a helium bubbling. In Fig. 1b the results collected during one of these runs are reported.

A decrease of the reactivity of both NP and NP_1 was observed with a formation of NP_2 at trace level only. For both aerated and no aerated conditions it can be inferred that the first step of UV_{254}-photolysis of NP is always a decarboxylation reaction to give NP_1 which is preferentially oxidized to NP_2, if oxygen is present in the aqueous solution, as occurs for higher wavelengths (Boscà et al., 2001).

It was suggested in the literature that "long lifetime triplet transient states", formed during the irradiation of NP, transfer readily energy to oxygen molecules, leading to the formation of an excited oxygen state as singlet oxygen ($^1O_2$) (Quintero and Miranda, 2000; Martinez and Scaiano, 1989). Since no indications were reported on the generation of singlet oxygen for irradiation of NP_1 and NP_2 photoproducts, some attempts were done to evaluate the real occurrence of this species in the reacting system and its effects.

Further runs were thus carried out to analyse the behaviour of both the species, NP_1 and NP_2, either in the absence and presence of oxygen in the solution or when a well known quencher for singlet oxygen ($^1O_2$) and for triplet transient species, 1,4-diazabicyclo[2,2,2]octane, named DABCO (Mac and Wirz, 2002; Bhattacharyya and Das, 1986; Monroe, 1985; Foote et al., 1970) was added to the oxygenated aqueous solution (Fig. 2a,b and 3 respectively).

The results shown in these figures undoubtedly indicate that both the reactivities of NP_1 and NP_2 species decrease after purging oxygen from the solution (Fig. 2a and 3, full triangles).

The addition of the DABCO seems to slightly reduce the reactivity NP_1 during the UV_{254}-photolysis (Fig. 2a, black circles), while it does not influence the consumption profile for UV_{254}-photolysis of NP_2 (Fig. 3, black circles). Moreover, no production of NP_2 was evidenced when the UV-photolysis of NP_1 was carried out in aerated condition but in presence of DABCO (Fig. 2b, black circles), probably due to the ability of DABCO to quench NP_1 excited states whose decomposition leads to the formation of NP_2. The irradiation of NP_2 seems not to generate singlet oxygen (Fig. 3, black squares and circles).

4. Kinetic modelling

4.1. *UV-photolysis in the deaerated system*

On the basis of above-reported observations an attempt to model the system behaviour was firstly done by considering the simple UV_{254}-photolysis of the investigated species (NP, NP_1 and NP_2) that occurs when oxygen is purged from the solution.

The following scheme was thus proposed for the UV_{254}-photolysis of naproxen in aqueous solution:

According to the results reported in the previous paragraph the proposed scheme accounts for the formation of NP_1 from
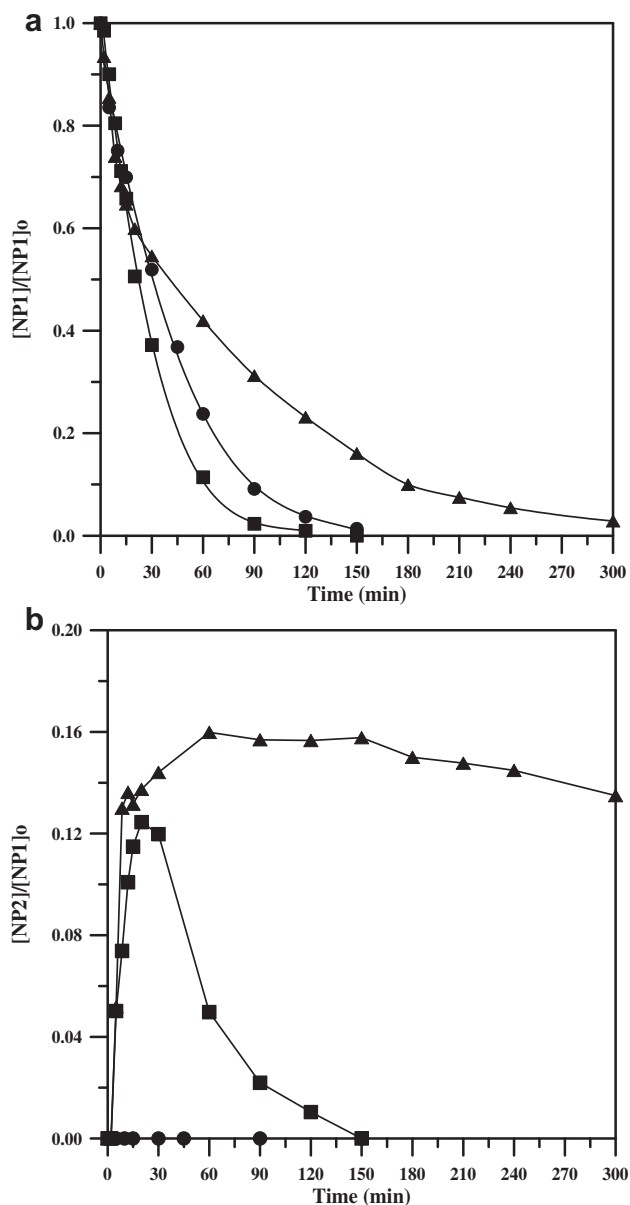
Fig. 2 − UV_{254}-photolysis of NP_1 under deaerated (▲), aerated without DABCO (■) and aerated with DABCO (●) conditions. $[NP_1]_o = 2.5 \cdot 10^{-5}$ M. pH = 7.0, T = 25 °C (2a) NP_1 consumption, (2b) NP_2 formation.

UV_{254}-photolysis of NP (at a selectivity lower than 1) and for its photolytic decomposition which only in part gives rise to the generation of NP_2. The disappearance of the latter by photolysis was included in the scheme. The formation of unidentified species was foreseen from NP_1 and NP_2 precursors and the photolysis of NP_2. This pathway is consistent with the mechanisms proposed for the NP photodegradation in deaerated media: NP_1 is generated through an electron ejection from the NP singlet excited state followed by decarboxylation. On the other hand, NP excited state could give rise to the formation of radical intermediates which can abstract hydrogen to form other unidentified species (Boscà et al., 2001, 1990).
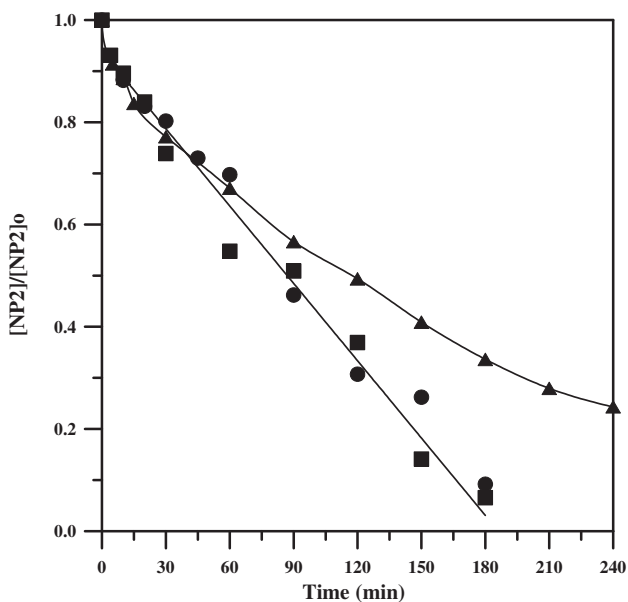
Fig. 3 — UV$_{254}$-photolysis of NP$_2$ under deaerated (▲), aerated without DABCO (■), aerated with DABCO (●) conditions. [NP$_2$]$_o$ = 1.7·10^{-5} M. pH = 7.0, T = 25 °C.

According to Scheme 1 and taking into account the equation rate for a photolytic process, in which the rate of photolysis of a substrate may be related to the photon flux absorbed by the solution and the quantum yield (Tang, 2003; Leifer, 1988), the following material balance equations could be written for all the involved species and for the pseudo-component X:

$$\frac{d[NP]}{dt} = -\frac{I_o \cdot \Phi_{NP}}{V} \cdot E \cdot f_{NP} \tag{1}$$

$$\frac{d[NP^*]}{dt} = \frac{I_o \cdot \Phi_{NP}}{V} \cdot E \cdot f_{NP} - (k_{NP*} \cdot [NP] + k_1) \cdot [NP^*] \tag{2}$$

$$\frac{d[NP_1]}{dt} = k_{NP^*} \cdot [NP^*] \cdot [NP] - \frac{I_o \cdot \Phi_{NP_1}}{V} \cdot E \cdot f_{NP_1} \tag{3}$$

$$\frac{d[NP_1^*]}{dt} = \frac{I_o \cdot \Phi_{NP_1}}{V} \cdot E \cdot f_{NP_1} - \left(k_{NP_1^*} + k_2\right) \cdot [NP_1^*] \tag{4}$$

$$\frac{d[NP_2]}{dt} = k_{NP_1^*} \cdot [NP_1^*] - \frac{I_o \cdot \Phi_{NP_2}}{V} \cdot E \cdot f_{NP_2} \tag{5}$$

$$[X] = [NP]_o - [NP] - [NP_1] - [NP_2] \tag{6}$$

where

$$E = 1 - e^{-2.3 \cdot L \cdot \left(\varepsilon_{NP}^{254} \cdot [NP] + \varepsilon_{NP_1}^{254} \cdot [NP_1] + \varepsilon_{NP_2}^{254} \cdot [NP_2] + \varepsilon_X^{254} \cdot [X]\right)} \tag{7}$$

For ε_X^{254} a value of 6.58·10^3 M^{-1} cm^{-1} was obtained by measuring the absorbances (A_{tot}) of the samples withdrawn from the reactor at different reaction times and using the following equation:

$$A_{tot} = \varepsilon_{NP}^{254} \cdot [NP] + \varepsilon_{NP_1}^{254} \cdot [NP_1] + \varepsilon_{NP_2}^{254} \cdot [NP_2] + \varepsilon_X^{254} \cdot [X] \tag{8}$$

The terms f_{NP}, f_{NP_1} and f_{NP_2} can be written as:

$$f_{NP} = \frac{\varepsilon_{NP}^{254} \cdot [NP]}{A_{tot}} \quad f_{NP_1} = \frac{\varepsilon_{NP_1}^{254} \cdot [NP_1]}{A_{tot}} \quad f_{NP_2} = \frac{\varepsilon_{NP_2}^{254} \cdot [NP_2]}{A_{tot}} \tag{9}$$

Applying the steady-state hypothesis for NP* and NP$_1^*$ transient species:

$$[NP^*]_{ss} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP} \cdot E \cdot f_{NP}\right)}{k_{NP^*} \cdot [NP] + k_1} \tag{10}$$

$$[NP_1^*]_{ss} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP_1} \cdot E \cdot f_{NP_1}\right)}{k_{NP_1^*} + k_2} \tag{11}$$

the Equations (3) and (5) become:

$$\frac{d[NP_1]}{dt} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP} \cdot E \cdot f_{NP}\right)}{[NP] + k_a} \cdot [NP] - \frac{I_o \cdot \Phi_{NP_1}}{V} \cdot E \cdot f_{NP_1} \tag{12}$$

$$\frac{d[NP_2]}{dt} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP_1} \cdot E \cdot f_{NP_1}\right)}{1 + k_b} - \frac{I_o \cdot \Phi_{NP_2}}{V} \cdot E \cdot f_{NP_2} \tag{13}$$

where $k_a = k_1/k_{NP^*}$ and $k_b = k_2/k_{NP_1^*}$

As it is clear from the equations reported above (1, 12 and 13), there are too many unknown parameters (k_a, k_b, Φ_{NP}, Φ_{NP_1} and Φ_{NP_2}) to be estimated by means of the data collected in a single photolytic run starting from an aqueous solution of naproxen. To overcome the difficulties associated with this assessment, the results collected by following the evolution of the more simple subsystems were firstly taken into account. That is, for example, a first attempt was done to estimate the best value of quantum yield Φ_{NP_2} by focussing the attention on the results collected during some photolytic runs in which deaerated aqueous solutions containing NP$_2$ species were irradiated. Successively, the data collected when NP$_1$ species
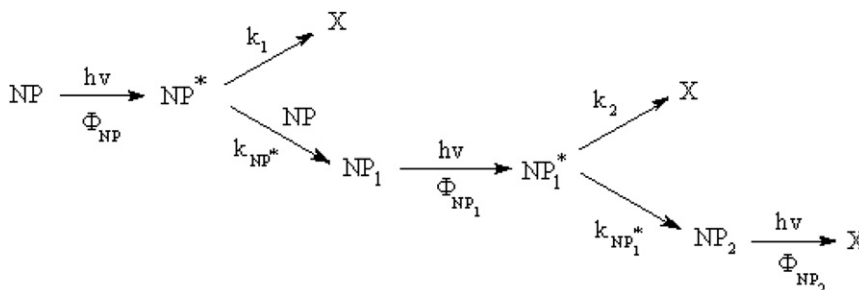
Scheme 1 — UV$_{254}$-photolysis of NP species under deaerated conditions. X represents unidentified species.

were submitted to photolysis in the absence of oxygen were considered and so on. In each case, the best estimation of the values of the parameters was done, for each subsystem, through an iterative optimization procedure (Marquardt approach) which minimized the squares of the differences between calculated and experimental concentrations of each species (objective function) (Reklaitis et al., 1983):

$$\Omega = \sum_{l=1}^{m} \sum_{j=1}^{n} \sum_{i=1}^{h} \left(y_{i,j,l} - c_{i,j,l} \right)^2$$

in which the terms y and c are the calculated and experimental concentrations whereas m, n and h are the number of experimental data recorded in each experiment, the number of the involved species and the number of the experiments used in a simple optimization procedure respectively.

The MATLAB routine "ode45", which is based on the Runge–Kutta method with adaptive step-size, was used to solve the set of ordinary differential equations (ODEs).

The percentage standard deviation for the j-th component was calculated as follows:

$$\sigma_j(\%) = \frac{1}{\bar{c}} \times \sqrt{\frac{\sum_{i=1}^{n} \left(y_i - c_i \right)^2}{n - p}} \times 100$$

where $\bar{c}$ is the average measured concentration, c_i the experimental concentration of j-th component for the i-th reaction time, n the number of the experimental points and p the unknown parameters to be estimated.

The parameter confidence limits were calculated by means of Student's t variable, for a fixed degree of freedom (ν) and a level of confidence as those corresponding to the intersection between the objective function and the straight line:

$$\Omega = \Omega_k \cdot \left(1 + \frac{t^2}{\nu} \right)$$

being Ω_k the value which the objective function assumes for the best estimated value of the k unknown parameter.

In Table 2 the values estimated for all the unknown parameters are shown along with their uncertainties and percentage standard deviations.

The values of quantum yields of direct UV$_{254}$-photolysis for NP$_1$ and NP$_2$ species were estimated for the first time in the present paper, whereas the value calculated for Φ_{NP} ($1.297 \cdot 10^{-2} \pm 1.23 \cdot 10^{-3}$ mol E^{-1}) is in good agreement with that previously reported ($1.5 \cdot 10^{-2} \pm 5.0 \cdot 10^{-3}$ mol E^{-1}) by Meite et al. (2010). From the comparison of the three parameters Φ_{NP}, Φ_{NP_1} and Φ_{NP_2} it can be assessed that, at the same experimental starting conditions, the photoreactivity at 254 nm follows the order: NP > NP$_1$ > NP$_2$ even if the order of molar absorptivities is $\varepsilon_{NP_2}^{254} > \varepsilon_{NP_1}^{254} > \varepsilon_{NP}^{254}$.

Some examples of comparison of experimental and calculated data for the subsystems considered in the estimation procedures are reported in Fig. 4a–b. The analysis of the data reported in Table 2 (low uncertainties on the estimated parameters) and in the figures indicates that the model derived from Scheme 1 adequately describes the behaviour of the investigated system under deaerated conditions.

4.2. UV$_{254}$-photolysis in the aerated system

On the basis of collected experimental results, a new kinetic scheme can be obtained by adding to the reactions already considered for the direct photolysis of NP$_i$ species in deaerated conditions (Scheme 1, black arrows) those occurring when oxygen is present in the reacting system (Scheme 2, blue arrows):

This scheme is in agreement with the findings previously reported on the photodegradation of NP in presence of oxygen. NP undergoes a photodecarboxylation step, leading to a decarboxy-naproxen radical, which combines readily with dissolved molecular oxygen, in aerated solution, yielding a peroxy radical which further reacts to yield 1-(6-methoxy-2-napthyl)ethanol and 2-acetyl-6-methoxynaphthalene as major photooxidation products (Boscà et al., 2001) and other photoproducts such as 2-ethyl-6-methoxy-naphthalene and 2-hydroperoxyl-6-methoxy-naphthalene (Hsu et al., 2006; Della Greca et al., 2004). According to these findings, in Scheme 2, under aerated irradiated conditions, NP gives prevalently NP$_1$ and NP$_2$ species and minor photoproducts, indicated as X. The NP$_1$ and NP$_2$ intermediates undergo a similar photo-oxidation process. Moreover, since there are some evidences that major products generated by the NP photolysis are a source of singlet oxygen (Partyka et al., 2001; Costanzo et al., 1989), it is considered the possibility that NP$_1$ reacts with oxygen dissolved to produce singlet oxygen via photo-unstable intermediate (Y). The singlet oxygen can be quenched by the aqueous media to its ground state (triplet oxygen) or attack NP$_2$ species.

Although it was reported in the literature that NP is a moderate singlet oxygen photosensitizer (Martinez et al., 1998; De la Pena et al., 1997) any quantitative attempt done in the present work to include, in Scheme 2, a reaction of singlet oxygen by irradiation of NP failed.

Under aerated conditions, taking into account Scheme 2, the above-described ODEs (1, 12–13), developed for the photodegradation of NP in absence of oxygen, were properly modified and reported below along with the mass balance on Y, Y*, ^{1}O$_2$ and X species:

Table 2 – Best estimated values of unknown kinetic constants along with the single ($\bar{\sigma}_i$) and overall ($\bar{\sigma}_{tot}$) average standard deviations for the UV$_{254}$-photolysis of NP, NP$_1$ and NP$_2$, under un-aerated conditions, in aqueous solution at pH = 7.0.

Φ_{NP} (mol E^{-1})	Φ_{NP_1} (mol E^{-1})	Φ_{NP_2} (mol E^{-1})	k_a (mol l^{-1})	k_b (dimensionless)
$1.29 \cdot 10^{-2} \pm 1.23 \cdot 10^{-3}$	$3.80 \cdot 10^{-3} \pm 3.20 \cdot 10^{-4}$	$3.73 \cdot 10^{-4} \pm 1.70 \cdot 10^{-5}$	$2.70 \cdot 10^{-2} \pm 5.01 \cdot 10^{-3}$	0.607 ± 0.13
$\bar{\sigma}_{NP}$ (%)	$\bar{\sigma}_{NP_1}$ (%)		$\bar{\sigma}_{NP_2}$ (%)	$\bar{\sigma}_{tot}$ (%)
1.26	2.2		0.23	3.06
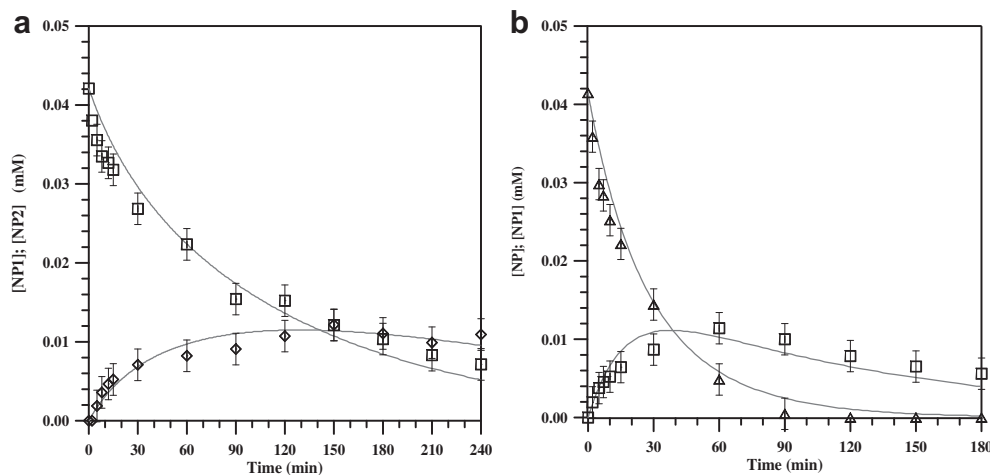
Fig. 4 — Predicted (solid lines) and experimental (symbols: △ NP, NP₁, ◇ NP₂) concentration-time profile for NPᵢ species UV₂₅₄-photolysis under deaerated conditions at pH = 7.0 and T = 25 °C, 4a: [NP₁]ₒ = 4.21·10⁻⁵ M; 4b: [NP]ₒ = 4.14·10⁻⁵ M.

$$\frac{d[NP]}{dt} = -\frac{I_o \cdot \Phi_{NP}}{V} \cdot E \cdot f_{NP} - \left(k_{NP/O_2} + k'_{NP/O_2} + k''_{NP/O_2}\right) \cdot [NP] \cdot [O_2] \quad (14)$$

$$\frac{d[NP_1]}{dt} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP} \cdot E \cdot f_{NP}\right)}{[NP] + k_a} \cdot [NP] - \frac{I_o \cdot \Phi_{NP_1}}{V} \cdot E \cdot f_{NP_1} + k'_{NP/O_2} \cdot [NP] \cdot [O_2]$$
$$+ -k_{NP_1/O_2} \cdot [NP_1] \cdot [O_2] \quad (15)$$

$$\frac{d[NP_2]}{dt} = \frac{\left(\frac{I_o}{V} \cdot \Phi_{NP_1} \cdot E \cdot f_{NP_1}\right)}{1 + k_b} - \frac{I_o \cdot \Phi_{NP_2}}{V} \cdot E \cdot f_{NP_2} + k''_{NP/O_2} \cdot [NP] \cdot [O_2]$$
$$+ -k''_{NP_2/O_2} \cdot [NP_2] \cdot [O_2] - k_3 \cdot [NP_2] \cdot [^1O_2]_{ss} \quad (16)$$

$$\frac{d[Y]}{dt} = k_{NP_1/O_2} \cdot [NP_1] \cdot [O_2] - \frac{I_o \cdot \Phi_Y}{V} \cdot E \cdot f_Y \quad (17)$$

$$[X] = [NP]_o - [NP] - [NP_1] - [NP_2] - [Y] \quad (18)$$

where $[O_2]$ indicates the concentration of $^3O_2$ dissolved in the liquid bulk (see Scheme 2).

The concentrations for $^1O_2$ and Y^* were calculated assuming a steady-state hypothesis for these species:

$$\frac{d[^1O_2]_{ss}}{dt} = -[^1O_2]_{ss} \cdot (k_d + k_3 \cdot [NP_2]) + k_4 \cdot [Y^*]_{ss} \cdot [O_2] = 0 \quad (19)$$
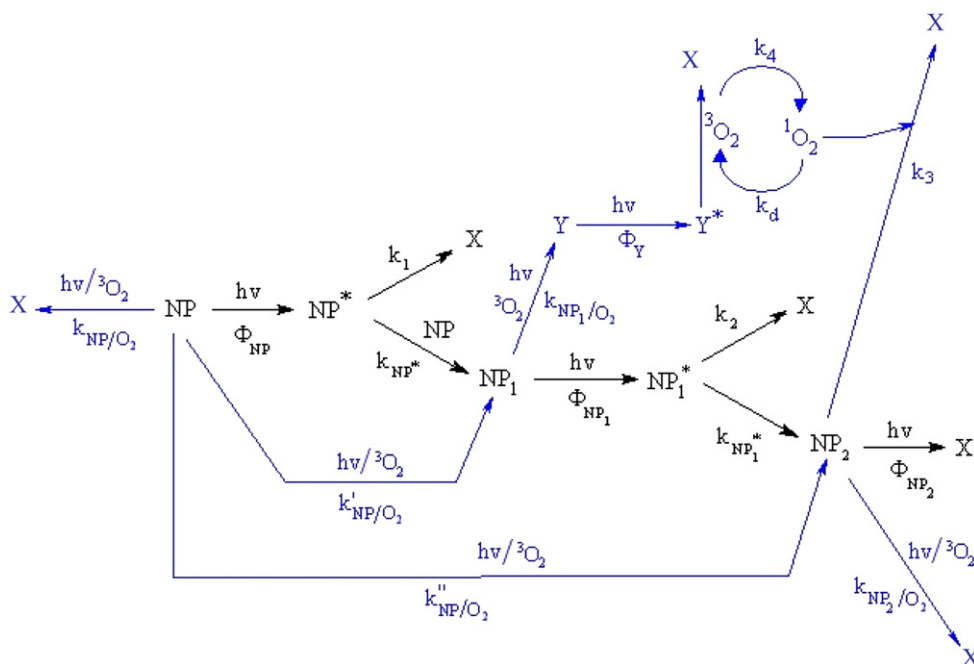
Scheme 2 — UV₂₅₄-photolysis of NP species under aerated (blue arrows) and deaerated (dark arrows) conditions. X and Y represent unidentified species. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

k_{NP/O_2} (M^{-1} s^{-1})	k'_{NP/O_2} (M^{-1} s^{-1})	k''_{NP/O_2} (M^{-1} s^{-1})	k_{NP_1/O_2} (M^{-1} s^{-1})	k_{NP_2/O_2} (M^{-1} s^{-1})	k_3 (M^{-1} s^{-1})	Φ_Y (mol E^{-1})
$0.817 \pm 6.67 \cdot 10^{-2}$	$0.172 \pm 4.50 \cdot 10^{-2}$	$0.283 \pm 3.67 \cdot 10^{-2}$	$0.175 \pm 2.83 \cdot 10^{-2}$	$4.33 \cdot 10^{-2} \pm 8.33 \cdot 10^{-3}$	$7.07 \cdot 10^9 \pm 2.30 \cdot 10^8$	$3.31 \cdot 10^{-3} \pm 5.96 \cdot 10^{-4}$

$\overline{\sigma}_{NP}$ (%)	$\overline{\sigma}_{NP_1}$ (%)	$\overline{\sigma}_{NP_2}$ (%)	$\overline{\sigma}_{tot}$ (%)
1.46	3.04	1.57	6.10

$$\frac{d[Y^*]_{ss}}{dt} = -k_4 \cdot [O_2] \cdot [Y^*]_{ss} + \frac{I_o \cdot \Phi_Y}{V} \cdot E \cdot f_Y = 0 \tag{20}$$

with

$$f_Y = \frac{\varepsilon_Y^{254} \cdot [Y]}{A_{tot}} \tag{21}$$

$$A_{tot} = \varepsilon_{NP}^{254} \cdot [NP] + \varepsilon_{NP_1}^{254} \cdot [NP_1] + \varepsilon_{NP_2}^{254} \cdot [NP_2] + \varepsilon_X^{254} \cdot [X] + \varepsilon_Y^{254} \cdot [Y] \tag{22}$$

$$E = 1 - e^{-2.3 \cdot L \cdot \left(\varepsilon_{NP}^{254} \cdot [NP] + \varepsilon_{NP_1}^{254} \cdot [NP_1] + \varepsilon_{NP_2}^{254} \cdot [NP_2] + \varepsilon_X^{254} \cdot [X] + \varepsilon_Y^{254} \cdot [Y] \right)} \tag{23}$$

It was reasonably assumed that $\varepsilon_{NP_1}^{254} = \varepsilon_Y^{254} = 4.35 \cdot 10^3$ M^{-1} cm^{-1}. For the kinetic constant of physical quenching of singlet oxygen (k_d) a value of 2.5×10^5 s^{-1} was considered (Wilkinson et al., 1993).

Also in this case, several kinetic constants were present for which no values were known "a priori" (($k_{NP/O_2}, k'_{NP/O_2}, k''_{NP/O_2},$

$k_{NP_1/O_2}, k_{NP_2/O_2}, k_3, \Phi_Y$)) and the same procedure "step-by-step", previously adopted for deaerated process, was adopted for their estimation. Firstly, the k_{NP_2/O_2} value was separately estimated using experimental data collected during photolytic run of aerated solutions containing NP_2 species (t = 0, $[NP_2]_o = 1.15 \cdot 10^{-5}$ M). Successively, the data collected from two runs, at different starting NP_1 concentrations ($[NP_1]_o = 2.00 \cdot 10^{-5}$ and $4.40 \cdot 10^{-5}$ M), when aerated solutions were submitted to photolysis, were used to estimate the best values for $k_{NP_1/O_2}, k_3, \Phi_Y$ parameters. Finally, the model was applied to the data collected from three photolytic runs with different NP initial concentrations, ($[NP]_o = 1.40 \cdot 10^{-5}$, $2.40 \cdot 10^{-5}$ and $4.50 \cdot 10^{-5}$ M), carried out on aerated solution of NP, with the purpose to identify the best values for $k_{NP/O_2}, k'_{NP/O_2}, k''_{NP/O_2}$ constants.

The kinetic values found for all the unknown parameters along with their uncertainties and percentage standard deviations for the photodegradation of NP ad its photoderivatives
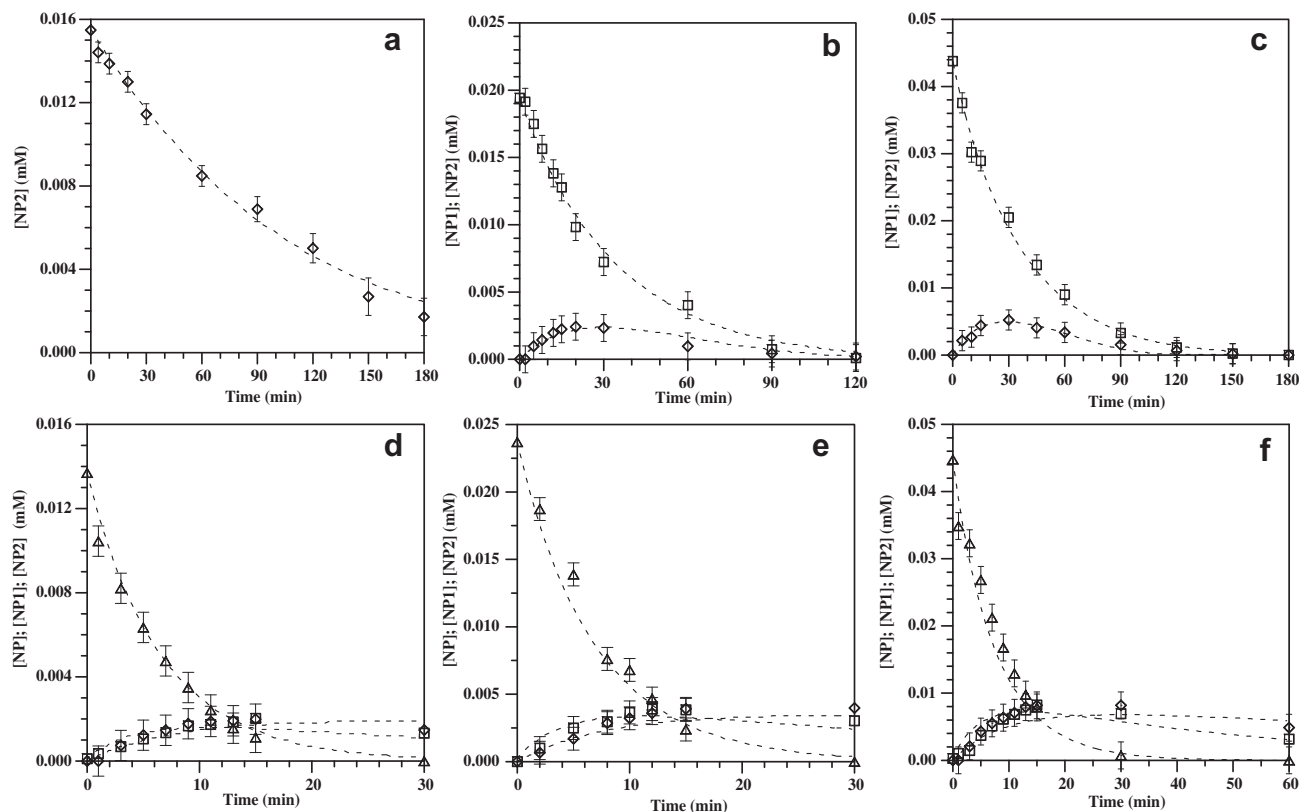
Fig. 5 — Predicted (dashed lines) and experimental (symbols: △ NP, ▢ NP_1, ◇ NP_2) concentration-time profile for NP_i species UV_{254}-photolysis under aerated conditions at pH = 7.0 and T = 25 °C, 5a: $[NP_2]_o = 1.15 \cdot 10^{-5}$ M; 5b: $[NP_1]_o = 1.94 \cdot 10^{-5}$ M; 5c: $[NP_1]_o = 4.38 \cdot 10^{-5}$ M; 5d: $[NP]_o = 1.37 \cdot 10^{-5}$ M; 5e: $[NP]_o = 2.37 \cdot 10^{-5}$ M; 5f: $[NP]_o = 4.48 \cdot 10^{-5}$ M.

under aerated conditions are shown in Table 3. The overall percentage standard deviations and those on the NP_i species, are lower than those associated to the experimental determination of the concentrations of the measured species.

The kinetic constants values for the photoreactions of oxygen with NP to give NP_1 (k'_{NP/O_2}) and NP_2 (k''_{NP/O_2}) are very similar and close to the value estimated for the attack of oxygen to NP_1 under irradiation (k'_{NP_1/O_2}). Their reaction rates are obviously influenced by the concentration of oxygen dissolved in the aqueous bulk.

This fact is in agreement with Meite's results (Meite et al., 2010) who reported that the value of the quantum yield of naproxen photolysis at 254 nm, estimated by using a simple model of the type $Ln[C]/[C]_o = -2.3 \cdot \varepsilon_\lambda \cdot \ell \cdot \Phi_\lambda \cdot I_o(\lambda) = -k_{app} \cdot t$, increases at increasing oxygen concentration in the liquid bulk. This observation may explain the discrepancies found in the literature for naproxen quantum yield values reported by different Authors (Meite et al., 2010; Benitez et al., 2009; Pereira et al., 2007a,b).

The k_3 parameter is the kinetic constant for the attack of singlet oxygen to NP_2 species leading to unknown species. The estimated value for it ($7.07 \cdot 10^9 \pm 2.30 \cdot 10^8$ M^{-1} s^{-1}) is not very different from that ($4.0 \cdot 10^9$ M^{-1} s^{-1}) reported in the literature for the reaction between singlet oxygen and nabumetone (Martinez and Scaiano, 1989), which is structurally very similar to NP_2:
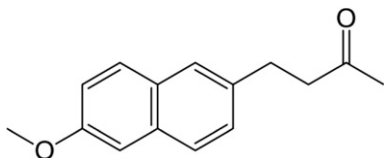
The k_3 value is four order larger than the rate constant for interaction of NP with singlet oxygen ($1.1 \cdot 10^5$ M^{-1} s^{-1}) reported by others (Packer et al., 2003) thus indicating that NP does not compete with NP_2 for this reacting species.

A visual comparison of the calculated and experimental concentration data for the NP, NP_1 and NP_2 species is shown in the figures reported below (Fig. 5a–f). A good capability of the model to simulate the system behaviour appears from the analysis of these diagrams.

5. Conclusions

The kinetics of photodegradation of naproxen was investigated under both deaerated and aerated media at pH = 7.0 and T = 25 °C.

Simplified reaction schemes of photodegradation of naproxen were proposed, in absence and in presence of oxygen, which take into account the photolysis reactions of naproxen and its derived photoproducts and contemporary the reactions of the measured species with oxygen dissolved in the liquid bulk. According to these schemes, two proper kinetic models were developed to simulate the system behaviour.

In particular, when oxygen was admitted in the reacting solution, the generation of singlet oxygen and its reactions with some of the intermediates identified or simply put forward in the solution were considered.

The proposed kinetic models predict the experimental data with a good accuracy and allowed to estimate, for the first time, the best values for the unknown kinetic parameters.

The calculated quantum yield of direct photolysis of naproxen under deaerated media ($1.29 \cdot 10^{-2} \pm 1.23 \cdot 10^{-3}$ mol E^{-1}) is in good agreement with the one previously reported by others ($1.50 \cdot 10^{-2} \pm 5.0 \cdot 10^{-3}$ mol E^{-1}).

The obtained results, under the adopted conditions, indicated a marked influence of dissolved oxygen on the photodegradation rates of naproxen and the relative distribution of the major reaction intermediates. The estimated kinetic constants values for the attack of oxygen to NP and NP_1, under irradiation, are very similar.

The calculated value for the kinetic constant of singlet oxygen attack to NP_2 ($7.07 \cdot 10^9 \pm 2.30 \cdot 10^8$ M^{-1} s^{-1}) was found to be close to the value of $4.0 \cdot 10^9$ M^{-1} s^{-1} reported for the reaction between singlet oxygen and nabumetone which is structurally very similar to NP_2.

REFERENCES

Andreozzi, R., Marotta, R., Paxeus, N., 2003. Pharmaceuticals in STP effluents and their solar photodegradation in aquatic environment. Chemosphere 50, 1319–1330.

Andreozzi, R., Caprio, V., Insola, A., Marotta, R., 2000. The oxidation of metol (N-methyl-p-aminophenol) in aqueous solution by UV/H_2O_2 photolysis. Water Research 34 (2), 463–472.

Angelakis, A.N., Marecos, Do Monte M.H.F., Bontoux, L., Asano, T., 1999. The status of wastewater-reuse practice in the Mediterranean basin: need for guidelines. Water Research 33 (10), 2201–2217.

Angelakis, A.N., Bontoux, L., 2001. Wastewater reclamation and reuse in European countries. Water Policy 3, 47–59.

Bhattacharyya, K., Das, P.K., 1986. Nanosecond transient processes in the triethylamine quenching of benzophenone triplets in aqueous alkaline media. Substituent effect, ketyl radical deprotonation, and secondary photoreduction kinetics. The Journal of Physical Chemistry 90 (17), 3987–3993.

Bedner, M., MacCrehan, W.A., 2006a. Transformation of acetaminophen by chlorination produces the toxicants 1,4-benzoquinone and n-acetyl-p-benzoquinone imine. Environmental Science and Technology 40, 516–522.

Bedner, M., MacCrehan, W.A., 2006b. Reactions of the amine-containing drugs fluoxetine and metoprolol during chlorination and dechlorination processes used in wastewater treatment. Chemosphere 65, 2130–2137.

Beek, B., Bohling, S., Bruckmann, U., Franke, C., Johncke, U., Studinger, G., 2000. The assessment of bioaccumulation. In: Beek, B. (Ed.), The Handbook of Environmental Chemistry, vol. II (Part J). Springer-Verlag, Berlin, Heidelberg, pp. 239–273.

Benitez, F.J., Real, F.J., Acero, J.L., Roldan, G., 2009. Removal of selected pharmaceuticals in waters by photochemical processes. Journal of Chemical Technology & Biotechnology 84, 1186–1195.

Boscà, F., Marìn, M.L., Miranda, M.A., 2001. Photoreactivity of the nonsteroidal anti-inflammatory 2-arylpropionic acids with

photosensitizing side effects. Photochemical and Photobiological Sciences 74 (5), 637–655.

Boscà, F., Miranda, M.A., Vañò, L., Vargas, F., 1990. New photodegradation pathways for naproxen, a phototoxic non-steroidal anti-inflammatory drug. Journal of Photochemistry and Photobiology A: Chemistry 54, 131–134.

Bueno, M.J.M., Agüera, A., Hernando, M.D., Gómez, M.J., Fernández-Alba, A.R., 2009. Evaluation of various liquid chromatography-quadrupole-linear ion trap-mass spectrometry operation modes applied to the analysis of organic pollutants in wastewaters. Journal of Chromatography A 1216, 5995–6002.

Caretti, C., Lubello, C., 2003. Wastewater disinfection with PAA and UV combined treatment: a pilot plant study. Water Research 37, 2365–2371.

Costanzo, L.L., De Guidi, G., Condorelli, G., Cambria, A., Famà, M., 1989. Molecular mechanism of naproxen photosensitivation in red blood cells. Journal of Photochemistry and Photobiology B: Biology 3, 223–235.

De la Pena, D., Marti, C., Nonell, S., Martinet, L.A., Miranda, M.A., 1997. Time-resolved near infrared studies on singlet oxygen production by the photosensitizing 2-arylpropionic acids. Photochemical and Photobiological Sciences 65 (5), 828–832.

Della Greca, M., Iesce, M.R., Pistillo, P., Previtera, L., Temussi, F., 2009. Unusual products of the aqueous chlorination of atenolol. Chemosphere 74, 730–734.

Della Greca, M., Brigante, M., Isidori, M., Nardelli, A., Previtera, L., Rubino, M., Temussi, F., 2004. Phototransformation and ecotoxicity of the drug Naproxen-Na. Environmental Chemistry Letters 1, 237–241.

Dizer, H., Wittekindt, E., Fischer, B., Hansen, P.D., 2002. The cytotoxic and genotoxic potential of surface water and wastewater effluents as determined by bioluminescence, umu-assays and selected biomarkers. Chemosphere 46 (2), 225–233.

Foote, C.S., Denny, R.W., Weaver, L., Chang, Y., Peters, J., 1970. Quenching of singlet oxygen. Annales of the New York Academy of Sciences 171, 138–148. International conferences on singlet molecular and its role in environmental sciences. Part III biological applications.

Gehr, R., Wagner, M., Veerasubramanian, P., Payment, P., 2003. Disinfection efficiency of peracetic acid, UV and ozone after enhanced primary treatment of municipal wastewater. Water Research 37, 4573–4586.

Hsu, Y.H., Liou, Y.B., Lee, J.A., Chen, C.Y., Wu, A.B., 2006. Assay of naproxen by high-performance liquid chromatography and identification of its photoproducts by LC-ESI MS. Biomedical Chromatography 20, 787–793.

Isidori, M., Lavorgna, M., Nardelli, A., Parrella, A., Previtera, L., Rubino, M., 2005. Ecotoxicity of naproxen and its phototransformation products. Science of The Total Environment 348 (1–3), 93–101.

Kim, I., Yamashita, N., Tanaka, H., 2009a. Photodegradation of pharmaceuticals and personal care products during UV and UV/H_2O_2 treatments. Chemosphere 77, 518–525.

Kim, I., Yamashita, N., Tanaka, H., 2009b. Performance of UV and UV/H_2O_2 processes for the removal of pharmaceuticals detected in secondary effluent of a sewage treatment plant. Journal of Hazardous Materials 166, 1134–1140.

Korte, F., Kvesitadze, G., Ugrekhelidze, D., Gordeziani, M., Khatisashvili, G., Buadze, O., Zaalishvili, G., Coulston, F., 2000. Organic toxicants and plants. Ecotoxicology and Environmental Safety 47, 1–26.

Koivunen, J., 2005. Inactivation of enteric microorganisms with chemical disinfectants, UV irradiation and combined chemical/UV treatments. Water Research 39, 1519–1526.

Kretschmer, N., Ribbe, L., Gaese, H., 2000. Wastewater reuse for agriculture. In: Institute of Technology in the Tropics (Ed.), Technology Resource Management & Development. Special

Issue: Water Management. ISSN: 1618-3312, vol. 2, pp. 37–64. Cologne.

La Farre, M., Ferrer, B., Ginebreda, A., Figueras, M., Olivella, L., Tirapu, L., Vilanova, M., Barcel, D., 2001. Determination of drugs in surface water and wastewater samples by liquid chromatography–mass spectrometry: methods and preliminary results including toxicity studies with Vibrio fischeri. Journal of Chromatography A 938, 187–197.

Leifer, A., 1988. The Kinetics of Environmental Aquatic Photochemistry: Theory and Practice. ACS Professional Reference Book, Washington.

Levine, A.D., Asano, T., 2004. Recovering sustainable water from wastewater. Environmental Science and Technology, 201–208.

Liberti, L., Lopez, A., Notarnicola, M., Barnea, N., Pedahzur, R., Fattal, B., 2000. Comparison of advanced disinfection methods for municipal wastewater reuse in agriculture. Water Science and Technology 42 (1–2), 215–220.

Mac, M., Wirz, J., 2002. Salt effects on the reactions of radical ion pairs formed by electron transfer quenching of triplet 2-methyl-1,4-naphthoquinone by amines. Optical flash photolysis and step-scan FTIR investigations. Photochemical and Photobiological Sciences 1, 24–29.

Martinez, L.J., Scaiano, J.C., 1989. Characterization of the transient intermediates generated from the photoexcitation of nabumetone: a comparison with naproxen. Photochemical and Photobiological Sciences 68 (5), 646–651.

Martinez, L.A., Braun, A.M., Oliveros, E., 1998. Effect of the microenvironment on the efficiency of singlet oxygen ($O_2(^1\Delta_g)$) production by photosensitizing anti-inflammatory drugs. Journal of Photochemistry and Photobiology B: Biology 45 (2–3), 103–112. http://www.sciencedirect.com/science/article/pii/S1011134498001687-cor1#cor1.

Marty, F., 1999. Plant vacuoles. The Plant Cell 11, 587–600.

Meite, L., Szabo, R., Mazellier, P., De Laat, J., 2010. Cinétique de phototransformation de polluants organiques émergents en solution aqueuse diluée. Revue des sciences de l'eau 23 (1), 31–39.

Metcalfe, C.D., Koenig, B.G., Bennie, D.T., Servos, M., Ternes, T.A., Hirsch, R., 2003. Occurrence of neutral and acidic drugs in the effluents of Canadian sewage treatment plants. Environmental Toxicology and Chemistry 22 (12), 2872–2880.

Miao, X.S., Koenig, B.G., Metcalfe, C.D., 2002. Analysis of acidic drugs in the effluents of sewage treatment plants using liquid chromatography–electrospray ionization tandem mass spectrometry. Journal of Chromatography A 952, 139–147.

Monroe, B.M., 1985. In: Frimer, A.A. (Ed.), Singlet Oxygen, vol. 1. CRC Press, Boca Raton, FL, pp. 177–224.

Moore, D.E., Chappuis, P.P., 1988. A comparative study of the photochemistry of the non-steroidal anti-inflammatory drugs, naproxen, benoxaprofen and indomethacin. Photochemical and Photobiological Sciences 47 (2), 173–180.

Musa, K.A.K., Eriksson, L.A., 2008. Theoretical study of the phototoxicity of naproxen and the active form of nabumetone. Journal of Physical Chemistry A 112, 10921–10930.

Nicole, I., De Laat, J., Doré, M., Duguet, J.P., Bonnel, C., 1990. Use of UV radiation in water treatment: measurement of photonic flux by hydrogen peroxide actinometry. Water Research 24, 157–168.

Ono, Y., Somiya, I., Kawaguchi, T., Mohri, S., 1996. Evaluation of toxic substances in effluents from a wastewater treatment plant. Desalination 106, 255–261.

Packer, J.L., Werner, J.J., Latch, D.E., McNeill, K., Arnold, W.A., 2003. Photochemical fate of pharmaceuticals in the environment: naproxen, diclofenac, clofibric acid, and ibuprofen. Aquatic Sciences 65, 342–351.

Paraskeva, P., Graham, N.J.D., 2002. State of the art-review: ozonation of municipal wastewater effluents. Water Environment Research 74, 569–581.

Partyka, M., Au, B.H., Evans, C.H., 2001. Cyclodextrins as phototoxicity inhibitors in drug formulations: studies on model

systems involving naproxen and b-cyclodextrin. Journal of Photochemistry and Photobiology A: Chemistry 140, 67–74.

Pereira, V.J., Linden, K.G., Weinberg, H.S., 2007a. Evaluation of UV irradiation for photolytic and oxidative degradation of pharmaceutical compounds in water. Water Research 41, 4413–4423.

Pereira, V.J., Weinberg, H.S., Linden, K.G., Singer, P.C., 2007b. UV degradation kinetics and modelling of pharmaceutical compounds in laboratory grade and surface water via direct and indirect photolysis at 254 nm. Environmental Science and Technology 41, 1682–1688.

Perry, R.H., Green, D.W., 1997. Perry's Chemical Engineers' Handbook, seventh ed. McGraw-Hill, New York.

Post, J., Petta, L., Kramer, A., Al Baz, I., 2008. EMWater Guide and Recommendations on Wastewater Treatment and Water Reuse. Efficient Management of Wastewater, pp. 19–28, (Chapter 2).

Quintero, B., Miranda, M.A., 2000. Mechanisms of photosensitization induced by drugs: a general survey. Ars Pharmaceutica 41 (1), 27–46.

Radziminski, C., Ballantyne, L., Hodson, J., Creason, R., Andrews, R.C., Chauret, C., 2002. Disinfection of Bacillus subtilis spores with chlorine dioxide: a bench-scale and pilot-scale study. Water Research 36 (6), 1629–1639.

Reklaitis, G.V., Ravindran, A., Regsdell, K.M., 1983. Engineering Optimization. Wiley, New York.

Santos, L.H.M., Araujo, A.N., Fachini, A., Pena, A., Delerue-Matos, C., Montenegro, M.C., 2010. Ecotoxicological aspects related to the presence of pharmaceuticals in the aquatic environment. Journal of Hazardous Materials 175, 45–95.

Singh, K.P., Mohan, D., Sinha, S., Dalwani, R., 2004. Impact assessment of treated/untreated wastewater toxicants discharged by sewage treatment plants on health, agricultural, and environmental quality in the wastewater disposal area. Chemosphere 55, 227–255.

Tang, W.Z., 2003. In: Physicochemical Treatment of Hazardous Wastes. CRC Press. Lewis Publishers (Chapter 7).

Ternes, T.A., Stumpf, M., Schuppert, B., Haberer, K., 1998. Simultaneous determination of antiseptics and acid drugs in sewage and river waters. Vom Wasser 90, 295–309.

Vieno, N., Tuhkanen, T., Kronberg, L., 2005. Seasonal variation in the occurrence of pharmaceuticals in effluents from a sewage treatment plant and in the recipient water. Environmental Science and Technology 39, 8220–8226.

Wilkinson, F., Helman, W.P., Ross, A.B., 1993. Quantum yields for the photosensitized formation of the lowest electrochemically excited singlet state of molecular oxygen in solution. Journal of Physical and Chemical Reference Data 22, 113–262.