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Degradation of ibuprofen by hydrodynamic cavitation: Reaction pathways and effect of operational parameters

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ABSTRACT

Ibuprofen (IBP) is an anti-inflammatory drug whose residues can be found worldwide in natural water bodies resulting in harmful effects to aquatic species even at low concentrations. This paper deals with the degradation of IBP in water by hydrodynamic cavitation in a convergent–divergent nozzle. Over 60% of ibuprofen was degraded in 60 min with an electrical energy per order (E_{EO}) of 10.77 kWh m^{-3} at an initial concentration of 200 μg L^{-1} and a relative inlet pressure p_{in} = 0.35 MPa. Five intermediates generated from different hydroxylation reactions were identified; the potential mechanisms of degradation were sketched and discussed. The reaction pathways recognized are in line with the relevant literature, both experimental and theoretical. By varying the pressure upstream the constriction, different degradation rates were observed. This effect was discussed according to a numerical simulation of the hydroxyl radical production identifying a clear correspondence between the maximum kinetic constant k_{OH} and the maximum calculated ·OH production. Furthermore, in the investigated experimental conditions, the pH parameter was found not to affect the extent of degradation; this peculiar feature agrees with a recently published kinetic insight and has been explained in the light of the intermediates of the different reaction pathways.

1. Introduction

The pollution of the aquatic environment has received increasing attention over the last decades because of the presence of emerging compounds (EC), only recently quantified thanks to the development of new analytical techniques [1,2]. These compounds are xenobiotic and bioactive chemicals such as surfactants, pharmaceuticals, endocrine disruptors and illegal drugs that, even at small concentrations, can affect water quality and are potentially harmful for the ecosystem and human health [2]. They reach the environment via the discharge of personal hygiene products, industrial (mainly pharmaceutical) and hospital wastes and medications. Currently, it is estimated that more than 3000 different pharmaceutical compounds are used in the European Union and many of them are susceptible to reach the water cycle [3]. Despite the unpredictable negative impact of these compounds on the environment and human health, at present there is no regulation limiting their concentrations in water streams. Based on the precaution principle, every four years the Environmental European Agency updates the list of priority substances where pharmaceuticals are included as potential pollutants. The EU Water Framework Directive EEA 2013 [4] recognizes pharmaceuticals as a potential risk for the aquatic environment in Europe and states that the Commission shall develop a strategic approach to monitor and regulate this kind of pollution as far as possible.

The compound 2-[3-(2-methylpropyl)phenyl] propanoic acid, marketed as ibuprofen (IBP, CAS number: 79261-49-7), is a Non-Steroidal Anti-Inflammatory Drug (NSAID) (available for over-the-counter sale) used in the treatment of rheumatic disorders, fever, migraine, muscle aches, arthritis and tooth aches. The main sources of environmental IBP contamination include industrial discharges, excretory products of medically treated humans and animals and the improper disposal of unused medications via the toilet [5]. As many other contaminants, its fate in the aquatic environment depends on biodegradability and physicochemical

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properties such as solubility in water, octanol–water and organic carbon–water partitioning coefficient [6]: natural attenuation phenomena mainly include sorption on soils or sediments, sunlight photolysis, and other abiotic transformations (i.e. hydrolysis) [7].

The conventional wastewater treatment plants (WWTP) do not seem to effectively remove the IBP from effluents. Some recent studies demonstrate that the conventional treatments, mainly based on the use of microorganisms, are inadequate to effectively destroy these organic compounds with a complex molecular structure and low concentrations [1]. For this reason, IBP can be found in sewage influents, effluent samples and, consequently, in several surface waters located downstream municipal WWTPs [8–12]. IBP, as other micropollutants, can be removed by membrane filtration or adsorption onto activated carbon; however, these two methods can be inhibited by the natural organic matter present in water, that also affects the fouling potential of the membranes or competes for adsorption. Although there have been numerous studies on the adsorption of aromatic compounds in aqueous solutions, the governing mechanisms must still be established to enhance the effectiveness of the process that still suffers from desorption- and regeneration-related issues [13–15]. IBP is also very resistant towards ozonation techniques [16]. For these reasons, new effective EC degradation techniques, known as Advanced Oxidation Processes (AOPs), are currently being studied. They are based on the generation of the hydroxyl radicals. OH-radicals are very reactive and non-selective species, able to react very rapidly with almost every organic substance. The AOPs include, among others, Fenton-like processes, direct ultraviolet photolysis, cavitation, photocatalysis, ozone-based hybrid processes, electro-oxidation and cavitation processes [17].

Cavitation is the formation, growth and subsequent collapse of microbubbles in a solvent with the consequent release of large magnitudes of energy per unit volume over an extremely short interval of time (10^{-3} ms), resulting in local high pressures (10–500 MPa) and temperatures (1000–10,000 K). The collapses also result in the formation of highly reactive free radicals, the continuous surface and interface cleaning as well as the enhancement of mass transfer rates due to generated turbulence [18]. The phenomenon takes place because of a pressure variation due to the presence of a constriction, designed ad hoc, or to ultrasound waves and is nowadays considered an innovative means to enhance different chemical processes [19]. In the field of pharmaceutical wastewater treatment, the application of ultrasounds was the first to be studied and is widely described in literature [6,20–23]. It shows promising results (also for IBP degradation) particularly if combined with other AOPs: sonophotocatalytic degradation in the presence of homogeneous (Fe^{3+}) and heterogeneous photocatalysts (TiO_2) [10,24].

Hydrodynamic cavitation (HC) has been recently considered as an interesting opportunity to rule out the issue of the high-energy consumption of ultrasounds. Moreover, HC can be considered a sustainable, reliable and easy-to-handle technique. Several papers prove the applicability of HC in degrading emerging organic pollutants [25,26]. Its combination with a chemical AOP seems to have major synergistic effects also in the treatment of pharmaceutical micropollutants, besides the excellent results in terms of power consumption and cost-effective system up-scaling. The most recent research works aim at extending degradation by introducing additional oxidants. Particularly in the case of non-VOC and hydrophilic substances, it is common opinion that, in industrial WWT, the cavitation has to be coupled in hybrid AOP solutions (e.g. with H_2O_2, UV, Fenton). Bagal and Gogate [27] studied the degradation of diclofenac by optimizing a HC hybrid technique (95% degradation using UV/TiO_2/H_2O_2 and hydrodynamic cavitation in a Venturi nozzle at 3 bar and pH 4). It is also worth mentioning the work carried out by researchers from the University of Ljubljana who investigated the use of shear-induced cavitation for hybrid HC/AOP [28,29]. They obtained promising results in degrading four different pharmaceuticals in a roto-cavitating apparatus with the addition of H_2O_2. They also achieved very high efficiency with the combination of biological treatment, UV and HC/H_2O_2 for similar pharmaceutical effluents [30] demonstrating how it is possible to transpose research findings into a directly employable large-scale wastewater treatment and encouraging the research in this field.

The patented Dynajet apparatus is the only example of industrial application of stand-alone HC that is effective in the degradation of a mixture of pharmaceuticals and personal care products; it has also shown that the overall degradation extent for different compounds increases linearly with the logKow while the pressure has a non-linear effect on the kinetics [31]. As long as cavitation alone is not ready to solve the issue of EC degradation, in our opinion it is important to keep performing stand-alone and cavitation experiments and to investigate its phenomenology by coupling the experiments with numerical simulations and by analyzing the degradation mechanisms. To this purpose, our paper is part of a wider study carried out by our research group focused on the experimental and theoretical insight of HC as an advanced oxidation process [25,32]. Hereby we present the experimental results on the degradation of IBP through hydrodynamic cavitation in a convergent–divergent nozzle reactor. The effects of inlet pressure and pH are addressed and discussed by referring to a consolidated mathematical model [32] as well as other relevant literature. The identification of different reaction intermediates in this work, allowed the investigation of the reaction kinetics and the identification of a possible mechanism of degradation that, in the light of a thermodynamic insight and other experimental evidences, might explain the peculiar the pH effect observed.

2. Experimental

2.1. Apparatus

Fig. 1 depicts the experimental setup. It consists of a closed-loop reactor comprising a holding tank of 1.5 L volume with a cooling system and two pipelines: the main line consists in the reactor provided with two pressure gauges measuring the inlet pressure (p_1) and the fully recovered downstream pressure (p_2); the second one is used to recirculate the solution bypassing the reactor. The dimensions of the nozzle are shown in Fig. 2. Two control valves regulate the gauge pressure and the flow rate (p_1 = 0.20–0.65 MPa; Q = 0.2–0.4 $m^3\ h^{-1}$) in the main line; the inside diameter of both the main and the by-pass lines is 12 mm while the constriction diameter is 2 mm. Further details can be found in Capocelli et al. [24]. The IBP initial concentration was 200 μg/L, the initial pH was varied in the range 2–9; the temperature was kept during the experiments below the limit of Tw = 25 °C. During testing, 1 ml samples were drawn from the test reservoir and analyzed as described in the next section.

2.2. Materials and methods

An ibuprofen sodium salt of analytical grade with purity higher than 98% purchased from Sigma–Aldrich (UK) was used for the experimental activities. The analytical measurement of the total IBP in solution was performed by Gas-Chromatography coupled with Mass Spectrometry (GC/MS) after a solid phase extraction (SPE) step. The SPE step consists in the isolation of the pharmaceuticals from the water samples through a

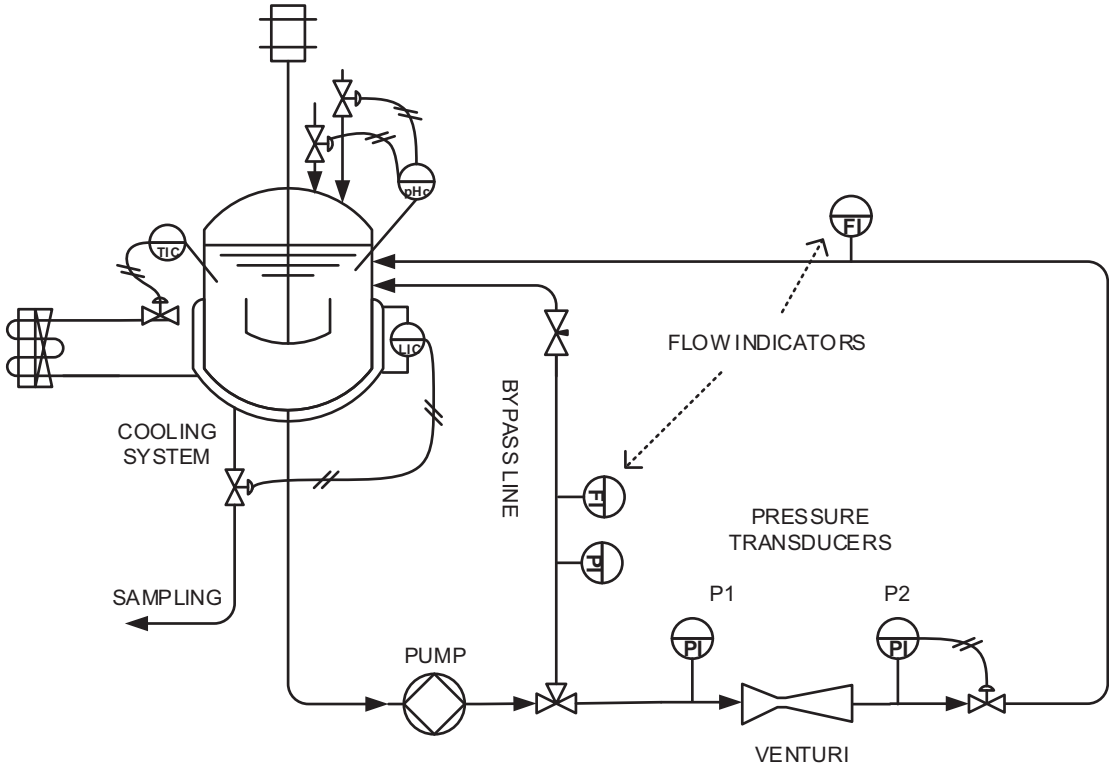

Fig. 1. Layout of the experimental apparatus.

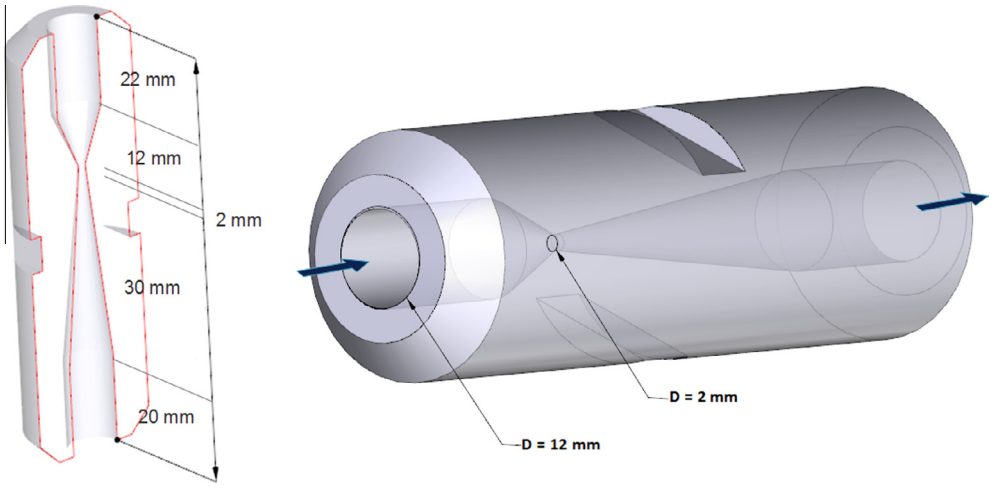

Fig. 2. Details of the Venturi reactor.

reversed-phase cartridge (Oasis HLB 1 cc, Waters) and the extraction of samples under vacuum at flow rate of 5 ml/min. Each cartridge was pre-conditioned with 3 ml of methanol (analytical grade, Fluka) followed by 5 ml of ultra-pure water. After extraction, the cartridges were dried under vacuum for 10 min. Ibuprofen was eluted with 3 ml of methanol. The extracts were dried under a stream of nitrogen. The final volume extracted is 100 µl. The determination of IBP in solution was made by means of a 7890A gas chromatograph with a mass spectrometric detector MSD5975C (Agilent Technologies, USA), equipped with a capillary column HP-5MS (5% phenylmethylsiloxane; length of 30 m; internal diameter of 0.25 mm, film thickness of 0.25 µm). The analytes were detected and quantified in selective ion monitoring (SIM). A calibration (5 points) with standard solutions in CH_2Cl_2 was used for quantification. The oven temperature was held at 70 °C for 1 min, then programmed at 10 °C/min to 300 °C which was held isothermally for 2 min. The temperatures of injection port, transfer line, source and quadrupole temperature were 250, 280, 230 and 150 °C.

The byproducts generated by the HC treatment were identified by a LC–MS/MS analysis [33]. The LC analysis was performed using a Waters 2690 HPLC system (Milford, MA, USA) coupled to a triple quadrupole mass spectrometer (Waters Micromass Quattro), equipped with a Z-spray ESI interface (Manchester, UK). The analysis was carried out in multiple reaction monitoring mode, both in positive and negative electrospray ionization mode.

3. Results and discussion

3.1. Kinetic effect of inlet pressure

The degradation of IBP was evaluated through sampling at different times, at four pressure levels; the system appeared ineffective outside of the range of pressure considered (0.20–0.55 MPa). The dimensionless concentration of IBP versus the experiment duration is shown in Fig. 3.

The degradation extent increases with different rates depending on the inlet gauge pressure p_{in}: the final conversion is higher at p_{in} = 0.35 MPa, followed by p_{in} = 0.45 MPa, p_{in} = 0.55 MPa and p_{in} = 0.20 MPa respectively. The IBP degradation rate was written according to the analysis by Capocelli et al. [25] by assuming that the reactor is a stationary source of hydroxyl radicals and the kinetics can follow an exponential equation with a pseudo-first order constant as shown in Eq. (1) where $[OH]_s$ is the stationary ·OH concentration. Therefore, by assuming a solution residence time τ in the reactor, the IBP conversion follows the Eq. (2) law. Fig. 4 represents the reactor model to write a first order degradation rate for the observable concentration IBP(t) through the mass balance to the tank control volume, Eq. (3). It is obtained by assuming that the reaction occurs only in the reactor of volume V' (representative of the cavitation extension) and the time scale of the variation in concentration is far higher than the residence time into the lines, in similarity to our previous work [25].

$$d[\mathrm{IBP}]/dt = -k_{\mathrm{OH}}[\mathrm{OH}]_s[\mathrm{IBP}] = -k[\mathrm{IBP}] \quad (1)$$

$$\mathrm{IBP}(\tau) = \mathrm{IBP}(0) \cdot \exp(-k \cdot \tau) \quad (2)$$

$$\left.\frac{d\,\mathrm{IBP}}{dt}\right|_{obs} = \frac{W(1-\varphi)}{V} \cdot \left[\exp\left(-k \cdot \frac{V'}{W \cdot (1-\varphi)}\right) - 1\right] \cdot \mathrm{IBP} \quad (3)$$

where W is the total flow rate and φ is the recirculating ratio. Therefore, by interpolating the experimental results of Fig. 3, it was possible to estimate the intrinsic reaction kinetic constant k from Eq. (3). The operating values describing the experiments reported in Table 1: the flow rate of fluid passing through the constriction, the cavitation number and the estimated pseudo-first order kinetic constant. On this basis the electrical energy consumption per order E_{EO} [34] has been calculated for all the inlet pressure and it is found to be minimum for p_{in} = 0.35 MPa. Although the value can be reduced by the implementation of hybrid techniques, it is comparable with the range indicated by the literature for UV treatment [35]. The results in Table 1 highlight the intrinsic role that the inlet

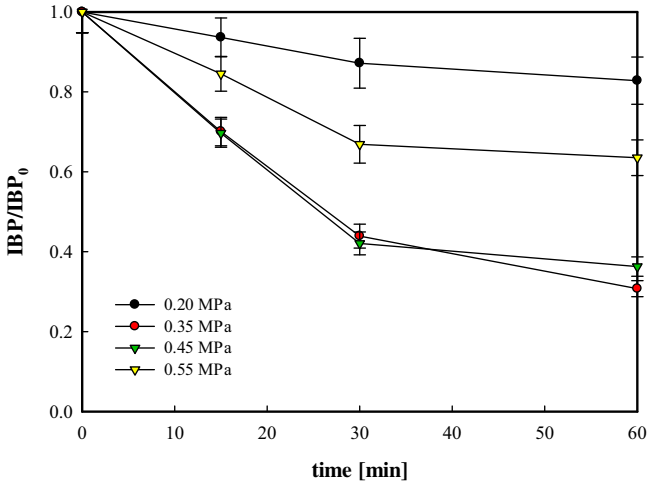

Fig. 3. IBP concentration versus time at different p_{in}.

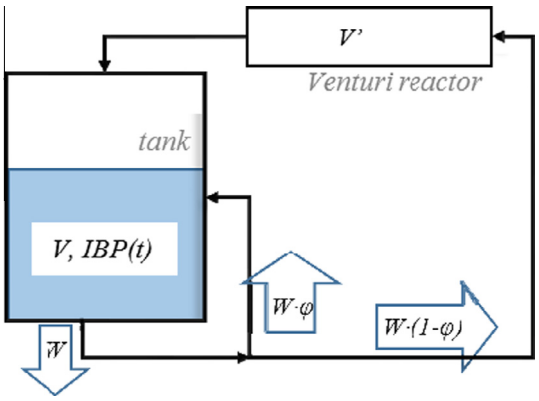

Fig. 4. Reactor model for the kinetics estimation.

Table 1
Experimental parameters and results. pH = 6. T = 25 °C.

Inlet pressure, p_{in} (bar gauge)	Flow rate into the Venturi, W $(1-\varphi)$ (L/min)	Cavitation number, C_v (–)	Pseudo-first order rate constant, k (min^{-1})	Electrical energy per order, E_{EO} (kWh m^{-3})
2	3.504	0.56	0.0152	28.83
3.5	4.17765	0.39	0.0850	10.77
4.5	4.62675	0.32	0.0928	14.07
5.5	5.07585	0.26	0.0383	45.68

pressure has in the optimization of cavitation as an AOP. The non-monotonic effect of p_{in} (and cavitation number) observed in this study, has already been mentioned in the literature as in the case of HC combined with heterogeneous photocatalysts of diclofenac sodium [27]. Vichare et al. [36] also showed similar results for KI decomposition in different orifice plates finding the best operating conditions at a certain value of p_{in}. Yan and Thorpe [37] identified a critical cavitation number for the onset of chocked cavitation, which corresponds to the decreased efficiency for p_{in} > 4.5 bar, seen in this paper. Also Capocelli et al. [32] gave a quantitative evaluation of the reasons beyond the optimal operating conditions. The first one, more intuitive, is that there are more passes through the HC reactor when p_{in} is higher. Secondly, the intrinsic dependence of the degradation on the p_{in}, is ascribable to the production of hydroxyl radicals by the collapse of the cavitating bubbles (seen as a stationary sources of ·OH [22]).

As a matter of fact, IBP conversion occurs mainly by reaction with hydroxyl radicals at the bubble–liquid interface; indeed the concentration inside the bubbles and in the liquid bulk is negligible due to its hydrophobic and low volatility character ($\log K_{ow}$ = 0.56 at neutral pH and H = 1.51 atm m^3/mol at 25 °C) [22]. Therefore, in Fig. 5 we propose a comparison between the observed kinetic constant and the estimated hydroxyl radical production rate. The pseudo-first order constants are taken from Table 1; the global ·OH production (grey line) is obtained by integrating the single bubble dynamics over the initial nuclei population as reported in our previous work [22]. The model consists of: (i) Rayleigh–Plesset equation for the radial motion of the spherical bubble; (ii) equation for the diffusive flux of water vapor (gas diffusion is ignored); (iii) overall energy balance applied at the cavitation bubble open system; (iv) continuity equation; (vi) Bernoulli equation; (vii) cavitation event rate as the number of nuclei of a specific size that actually cavitate as a consequence of the hydraulic regime. Main assumption: (1) fragmentation and coalescence phenomena are neglected; (2) uniform pressure and temperature inside the bubble; (3) temperature–pressure profile stationary in the liquid bulk and at the bubble wall; (4) at the collapse stage, the bubble content

is modeled as a reactive mixture of compounds that cannot get out of the bubble. The equilibrium mole fraction of the entrapped chemical species at the peak conditions reached during the transient collapse is estimated by minimizing the Gibbs free-energy.

The peak of production, recognized around p_{in} = 0.45 MPa, corresponds to the maximum of IBP conversion rate observed in our experiment and gives a further confirmation of the reaction mechanism (above mentioned and summarized by Eq. (1)) triggered by the ·OH released from the collapsing bubbles.

3.2. Reaction pathways

In order to identify the reaction intermediates, the samples collected at various time intervals were analyzed by means of LC–MS/MS. Fig. 6 shows the temporal sequence of intermediate formation for three different sampling times; by extending the treatment time, six prominent peaks were identified. The results show the formation (clearly visible at t = 60 min) of products from the ·OH attack on both the propanoic acid and isobutyl substitutes of the IBP structure as:

A. 2-[4-(2-hydroxyisobutyl)phenyl]propionic acid
B. 2-[4-(1-hydroxyisobutyl)phenyl]propionic acid;
C. 2-hydroxy-2[4-(2methylpropyl)phenyl]propanoic acid.

The reaction pathways with the intermediate chemical structures are shown in Fig. 7. They are generated in the first place by the acidic hydrogen abstraction, which dominates over the other ·OH addition mechanisms and can produce two different benzylic carbon radicals at C11 and C24 and a tertiary radical C14 (as proven by products A, B and C). The identification of these intermediates also corroborates the assumption of the kinetic mechanisms of Eqs. (1) and (2) and are in line with the experimental observations in the literature [22,24], where no oxidation product for the primary carbons is observed. By extending the duration of the experiments, the decomposition of compound A, through the cleavage of the C1–C2 bond of the isobutyl functional group, forms two intermediates, 2-(4-methylphenyl)propanoic acid (D) and ketone acetone. Furthermore, the decarboxylation of the compound C results in the formation of 4-isobuthylacetophenone (E) releasing formic acid. The disappearance of compounds A and B generating D and E is visible in Fig. 6 at t = 150 min.

In order to explain the observed pathways it is interesting to analyze the thermodynamic and kinetic parameters of the reaction between ibuprofen and hydroxyl radical identified by Xiao et al.

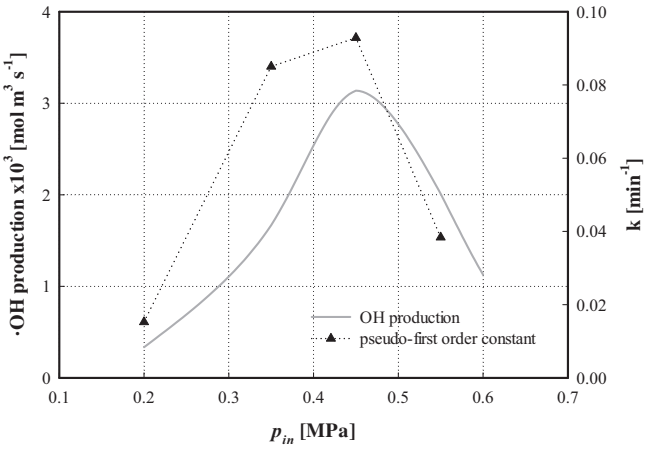

Fig. 5. Effect of the inlet pressure p_{in}: observed pseudo-first order kinetic constant of IBP conversion (▲); theoretical, stationary ·OH rate of production in the experimental apparatus (grey line).

[38]. Through the density functional theory, these authors compared all the possible mechanisms of ·OH on IBP attack: the bond dissociation energy (BDE) in case of hydrogen abstraction for the benzylic carbons (C11 and C24) characterizing the intermediates B and C is lower than that of the primary carbons. A similar BDE can be found for H abstraction at the tertiary carbon (C14) for the generation of compound A. Table 2 reports the resulting kinetic constants. Subsequent secondary reactions of the carbon radical could lead to exothermic decarboxylation and secondary oxidation leading to a ketone group with a low activation barrier: the second step, decarboxylation, can occur very easily and can contribute to the formation of the oxidized products as in the literature [10,21] and expressed in the intermediates D and E. The time sequence of the reaction products observed in this paper endorses the hypothesized mechanisms, also confirming that the technique can be an effective AOP initiator for IBP degradation like in other sonochemical environments [10]. Although it is not possible to observe the mineralization of the pollutant in the experimental conditions investigated, the HC treatment promotes the oxidation of IBP and its biodegradability by triggering the hydroxylation process followed by a second step of de-methylation or de-carboxylation.

3.3. The effect of initial pH

As it is a crucial parameter in wastewater treatment, the pH was varied to investigate its effect on the IBP degradation. The molecule studied is a non-volatile compound and the region of degradation would be outside the cavitation bubbles. The IBP concentration at the hydrophobic bubble interface is clearly influenced by the solution pH: above its pK_a value of 4.9, the anionic IBP [A^-] is predominant, while at lower values IBP is principally found in its molecular form [HA]. In our experiments, the degradation of both ionic and neutral forms of IBP was carried out choosing initial pH values of 2, 9 and 6. The results are illustrated in Fig. 8; the gauge inlet pressure was fixed at the optimized value of p_{in} = 0.45 MPa. As a general remark it can be said that pH does not affect the IBP degradation rate; a slight decrease in the degradation rate of IBP under acidic media was observed following the order pH 9 > pH 6 > pH 2. This could be tell-tale of two phenomena acting in contrast.

On the one hand, an increased hydrophobicity (protonation of the carboxylic group) results in a higher concentration at the interface of cavitation bubbles, thus leading to a faster degradation rate at low pH. On the other hand, the reaction pathway IBP → C → E (see Fig. 7) has been significantly observed only for the anionic form of IBP. This experimental evidence is supported by the result of quantum mechanics recently developed by Xiao et al. [38] While the energy profile for the common ·OH reactions is similar in case of neutral and anionic IBP, the energy barrier for H25 abstraction in position C24 (see C → E in Fig. 7) is significantly lower than for neutral IBP: $\Delta G^\circ = -7.67 \cdot 10^{-2}$ kcal/mol against ΔG° = 6.56 kcal/mol [38]. The results of k_{OH}, showing the preferable pathway that generates observable concentration of compounds C and E from the anionic IBP, are visible in Table 2. Therefore, in our case, the pH influences the reaction mechanisms with a phenomenology that differs from the classical view of the sonochemical degradation of non-VOC substances seen as an issue of hydrophobicity and pollutant adsorption at the bubble interface [10,22].

4. Conclusions

This study presents the evidence that hydrodynamic cavitation alone is effective in the degradation of ibuprofen, both in the

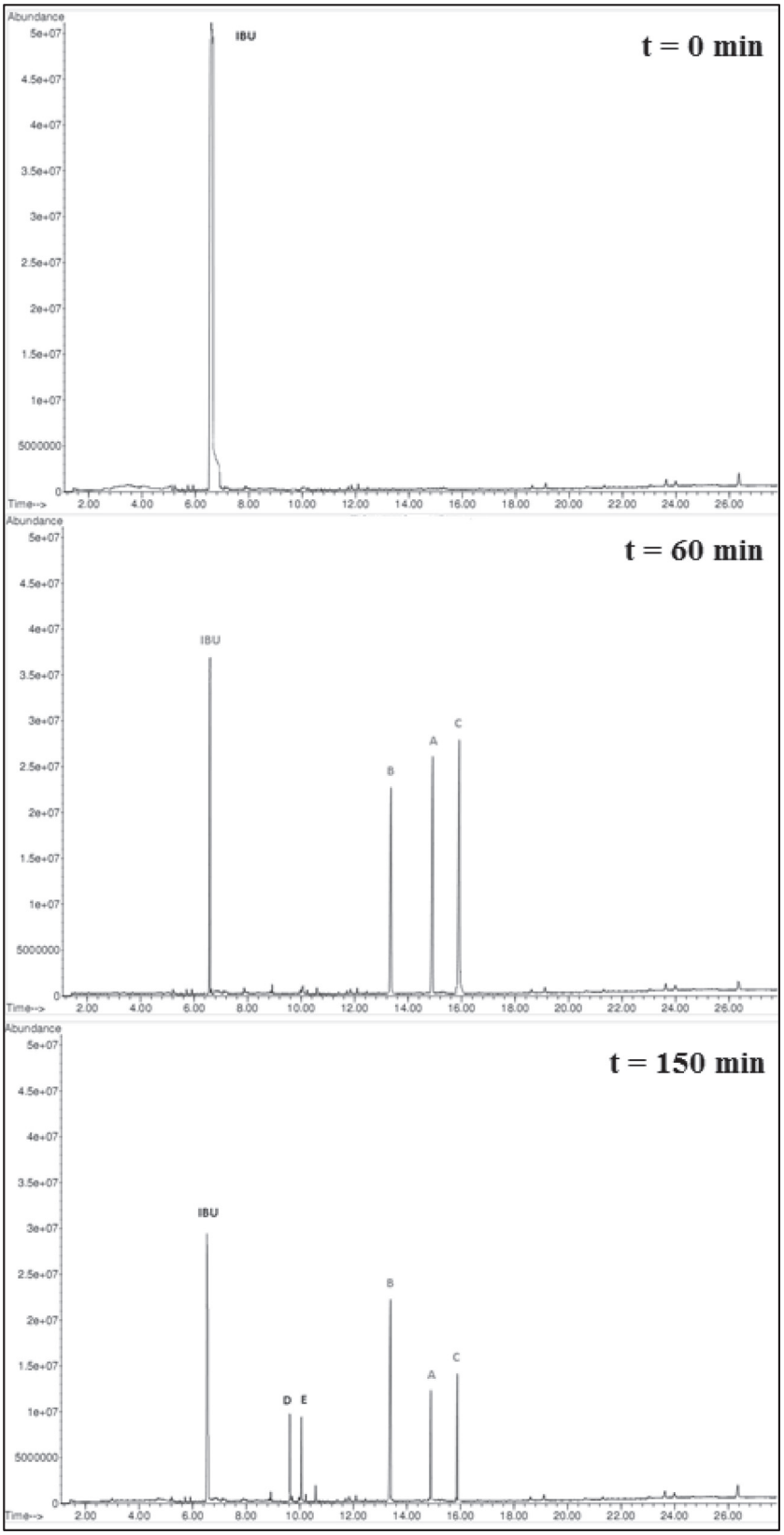

Fig. 6. Temporal sequences of intermediate identification. p_{in} = 0.45 MPa. Time 0–150 min.

neutral and dissociated form. More than 60% of ibuprofen was degraded within 60 min with a E_{Eo} of 10.77 kWh m^{-3} at an initial concentration of 200 μg L^{-1} and a relative inlet pressure p_{in} = 0.35 MPa without the use of additive chemicals. The effect of inlet pressure has been addressed and compared with literature data: the peaks of the kinetic constants related to p_{in} seem to

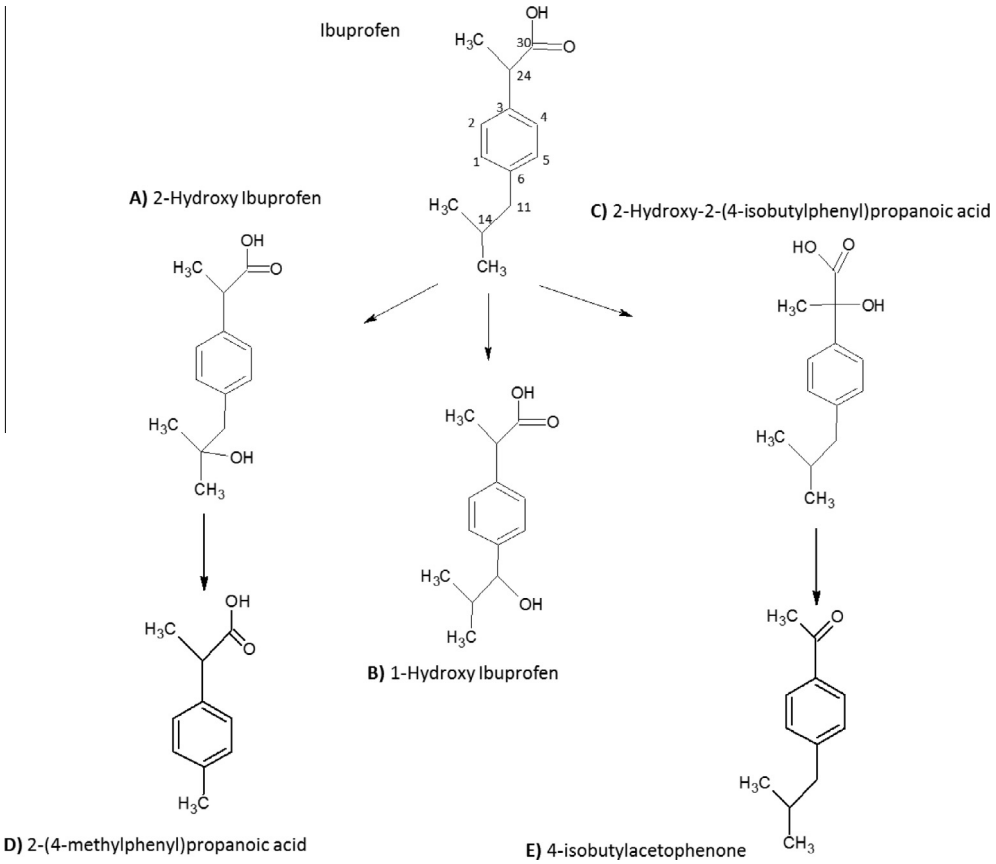

Fig. 7. Intermediates identified and possible degradation pathways by hydrodynamic cavitation of IBP.

Table 2
Rate constants for the transition state species involved in the reactions of neutral and anionic form of ibuprofen with OH extracted from the work of Xiao et al. [38].

Reaction mechanism	Atom number	k ($M^{-1}\,s^{-1}$)
Neutral form		
H abstraction	C11	$2.76 \cdot 10^9/9.59 \cdot 10^7$
	C14	$1.50 \cdot 10^9$
	C24	$1.41 \cdot 10^8$
Nucleophilic attack	C30	$2.74 \cdot 10^{-3}$
Anionic		
H abstraction	C24	$7.40 \cdot 10^9$
Nucleophilic attack	C30	$3.88 \cdot 10^{-2}$

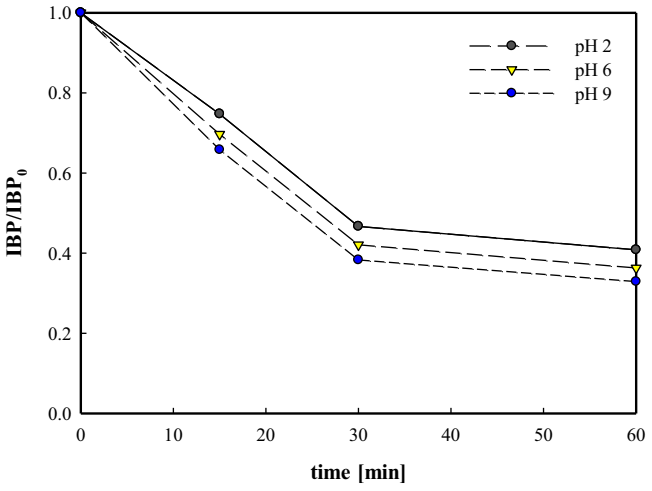

Fig. 8. Conversion of ibuprofen at different pH.

match the numerical simulation of the hydroxyl radicals produced in the Venturi reactor, which is an additional proof that the IBP degradation passes through the collision with ·OH at the interface of the collapsing bubbles. The experimental work therefore represents the basis to design and develop advanced treatment schemes including hydrodynamic cavitation for the treatment of pharmaceutical wastewaters.

Furthermore the paper reports an innovative comprehensive approach by coupling experimental observations with numerical simulations and thermodynamic analysis (from pertinent literature) in order to figure out the process phenomenology from different complementary point of views. A possible reaction scheme has been identified and explained in the light of quantum mechanics as in the literature. The comparison with the rigorous thermodynamic model also validates the reaction mechanisms and indirectly the simulation model. Thanks to the experimental methods and the literature review, a degradation pathway has been identified and, for the first time, related to a plausible explanation of the pH effect. Although this differs from the data in the literature, it can be fully explained on the basis of the kinetic analysis developed. The mechanisms identified at different pH should be studied in further experiments at different IBP initial concentrations in order to figure out the behavior in both kinetic or diffusion controlled regimes.

References

[1] J. Rivera-Utrilla, M. Sànchez-Polo, M.A. Ferro-Garcia, G. Prados-Joya, Pharmaceuticals as emerging contaminants and their removal from water. A review, Chemosphere 93 (2013) 1268–1287.
[2] D. Fatta-Kassinos, M.I. Vasquez, K. Kümmerer, Transformation products of pharmaceuticals in surface waters and wastewater formed during photolysis and advanced oxidation processes – degradation, elucidation of byproducts and assessment of their biological potency, Chemosphere 85 (2011) 693–709.

[3] EEA, Pharmaceuticals in the environment – results of an EEA workshop, Ann. Rev. Environ. Resour. 35 (1) (2010) 57–75.
[4] EEA, Directive 2013/39/EU Of The European Parliament And Of The Council. Official Journal of the European Union, 2013(July), pp. 1–17.
[5] M. Melillo, V.M. Gun'ko, S.R. Tennison, L.I. Mikhalovska, G.J. Phillipas, J.G. Davies, A.W. Lloyd, O.P. Kozynchenko, D.J. Malik, M. Streat, S.V. Mikhalovsky, Structural characteristics of activated carbons and ibuprofen adsorption affected by bovine serum albumin, Langmuir 20 (2004) 2837–2851.
[6] R. Xiao, D. Diaz-Rivera, L.K. Weavers, Factors influencing pharmaceutical and personal care product degradation in aqueous solution using pulsed wave ultrasound sonochemical degradation of pharmaceuticals and personal care products in water, Ind. Eng. Chem. Res 52 (2013) 2824–2831.
[7] D. Vione, P.R. Maddigapu, E. De Laurentiis, M. Minella, M. Pazzi, V. Maurino, C. Minero, S. Kouras, C. Richard, Modelling the photochemical fate of ibuprofen in surface waters, Water Res. 45 (2011) 6725–6736.
[8] A. Ziylan, N.H. Ince, The occurrence and fate of anti-inflammatory and analgesic pharmaceuticals in sewage and fresh water: treatability by conventional and non-conventional processes, J. Hazard. Mater. 187 (2011) 24–36.
[9] R.K. Szabó, Decomposition of some pharmaceuticals by advanced oxidation processes (Ph.D. thesis), University of Szeged, 2010.
[10] F. Méndez-Arriaga, R.A. Torres-Palma, C. Petrier, S. Esplugas, J. Gimenez, C. Pulgarin, Mineralization enhancement of a recalcitrant pharmaceutical pollutant in water by advanced oxidation hybrid processes, Water Res. 43 (2009) 3984–3991.
[11] G.T. Güyer, N.H. Ince, Degradation of diclofenac in water by homogeneous and heterogeneous sonolysis, Ultrason. Sonochem. 18 (2011) 114–119.
[12] C.P. Silva, M. Otero, V. Esteves, Processes for the elimination of estrogenic steroid hormones from water: a review, Environ. Pollut. 165 (2012) 38–58.
[13] S. Canzano, S. Capasso, M. Di Natale, A. Erto, P. Iovino, D. Musmarra, Remediation of groundwater polluted by aromatic compounds by means of adsorption, Sustainability 6 (2014) 4807–4822.
[14] A. Erto, A. Lancia, D. Musmarra, A modelling analysis of PCE/TCE mixture adsorption based on ideal adsorbed solution theory, Sep. Purif. Technol. 80 (2011) 140–147.
[15] P. Iovino, S. Canzano, S. Capasso, A. Erto, D. Musmarra, A modeling analysis for the assessment of ibuprofen adsorption mechanism onto activated carbons, Chem. Eng. J. 277 (2015) 360–367.
[16] M.G. Antoniou, G. Hey, S. Rodríguez Vega, A. Spiliotopoulou, J. Fick, M. Tysklind, J. la Cour Jansen, H.R. Andersen, Required ozone doses for removing pharmaceuticals from wastewater effluents, Sci. Total Environ. 456–457 (2013) 42–49.
[17] S. Chakma, V.S. Moholkar, Intensification of wastewater treatment using sono-hybrid processes: an overview of mechanistic synergism, Indian Inst. Chem. Eng. (2015) 1–23.
[18] K.S. Suslick, M.M. Mdleleni, J.T. Ries, Chemistry induced by hydrodynamic cavitation, J. Am. Chem. Soc. 119 (1997) 9303–9304.
[19] F.J. Keil, Modeling of Process Intensification, Wiley-VCH, Weinheim, 2007.
[20] R. Xiao, Z. He, D. Diaz-Rivera, G.Y. Pee, L.K. Weavers, Sonochemical degradation of ciprofloxacin and ibuprofen in the presence of matrix organic compounds, Ultrason. Sonochem. 21 (2014) 428–435.
[21] R. Xiao, Z. Wei, D. Chen, L.K. Weavers, Kinetics and mechanism of sonochemical degradation of pharmaceuticals in municipal wastewater, Environ. Sci. Technol. 48 (2014) 9675–9683.
[22] F. Méndez-Arriaga, R.A. Torres-Palma, C. Petrier, S. Esplugas, J. Gimenez, C. Pulgarin, Ultrasonic treatment of water contaminated with ibuprofen, Water Res. 42 (2008) 4243–4248.
[23] M. Capocelli, E. Joyce, A. Lancia, T.J. Mason, D. Musmarra, M. Prisciandaro, Sonochemical degradation of estradiols: incidence of ultrasonic frequency, Chem. Eng. J. 210 (2012) 9–17.
[24] J. Madhavan, F. Grieser, M. Ashokkumar, Combined advanced oxidation processes for the synergistic degradation of ibuprofen in aqueous environments, J. Hazard. Mater. 178 (2010) 202–208.
[25] M. Capocelli, M. Prisciandaro, A. Lancia, D. Musmarra, Hydrodynamic cavitation of p-nitrophenol: a theoretical and experimental insight, Chem. Eng. J. 254 (2014) 1–8.
[26] V.K. Saharan, A.B. Pandit, P.S.S. Kumar, S. Anandan, Hydrodynamic cavitation as an advanced oxidation technique for the degradation of acid red 88 dye, Ind. Eng. Chem. Res. 51 (2012) 1981–1989.
[27] M.V. Bagal, P.R. Gogate, Degradation of diclofenac sodium using combined processes based on hydrodynamic cavitation and heterogeneous photocatalysis, Ultrason. Sonochem. 21 (2014) 1035–1043.
[28] M. Petkovšek, M. Zupanc, M. Dular, T. Kosjek, E. Heath, B. Kompare, B. Širok, Rotation generator of hydrodynamic cavitation for water treatment, Sep. Purif. Technol. 118 (2013) 415–423.
[29] M. Zupanc, T. Kosjek, M. Petkovšek, M. Dular, B. Kompare, B. Širok, M. Strazar, E. Heath, Shear-induced hydrodynamic cavitation as a tool for pharmaceutical micropollutants removal from urban wastewater, Ultrason. Sonochem. 21 (2014) 1213–1221.
[30] M. Zupanc, T. Kosjek, M. Petkovšek, M. Dular, B. Kompare, B. Širok, Z. Blazeka, E. Heath, Removal of pharmaceuticals from wastewater by biological processes, hydrodynamic cavitation and UV treatment, Ultrason. Sonochem. 20 (2013) 1104–1112.
[31] G. Loraine, G.L. Chahine, Oxidation of pharmaceuticals and personal-care products in water using hydrodynamic cavitation, in: 245th American Chemical Society Meeting, 2013.
[32] M. Capocelli, D. Musmarra, M. Prisciandaro, A. Lancia, Chemical effect of hydrodynamic cavitation: simulation and experimental comparison, AIChE J. 60 (2014) 2566–2572.
[33] C. Lacey, G. McMahon, J. Bones, L. Barron, A. Morrissey, J.M. Tobin, An LC–MS method for the determination of pharmaceutical compounds in wastewater treatment plant influent and effluent samples, Talanta 75 (2008) 1089–1097.
[34] J.R. Bolton, K.G. Bircher, W. Tumas, C.A. Tolman, Figures-of-merit for the technical development of advanced oxidation technologies for both electric and solar driven system, Pure Appl. Chem. 73 (627–637) (2001) 2001. UPAC.
[35] H. Andersen, K. Hansen, T. Kosjek, Photochemical treatment of pharmaceuticals, in: 6th IWA World Water Congress and Exhibition, Vienna, 2008.
[36] N.P. Vichare, P.R. Gogate, A.B. Pandit, Optimization of hydrodynamic cavitation using a model reaction, Chem. Eng. Technol. 23 (2000) 683–690.
[37] Y. Yan, R.B. Thorpe, Flow regime transitions due to cavitation in the flow through an orifice, Int. J. Multiphase Flow 16 (1990) 1023–1045.
[38] R. Xiao, M. Noerpel, H.L. Luk, Z. Wei, R. Spinney, Thermodynamic and kinetic study of ibuprofen with hydroxyl radical: a density functional theory approach, Int. J. Quantum Chem. 114 (2014) 74–83.