Chapter 7

Nano based photocatalytic degradation of pharmaceuticals

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Abbreviations

Pharmaceuticals	Abbreviations
Acyclovir	ACY
Amiloride	AMI
Amoxicillin	AMX
Ampicillin	AMP
Atenolol	ATL
Bendroflumethiazide	В
Carbamazepine	CBZ
Chloramphenicol	CAP
Ciprofloxacin	CIP
Cloxacillin	CLX
Codeine	CDN
Diclofenac	DCF
Erythromycin	ERY
Erythromycylamine	ERYA
Ethacrynic acid	ETA
Flumequine	FLU
Furosemide	F
Ibuprofene	IBP
Levofloxacin	LEVO
Lincomycin	LNC
Lorazepam	LZP
Metronidazole	MT
Moxifloxacin	MOX
Naproxen	NPX
Norfloxacin	NOR
Ofloxacin	OFL
Oxacillin	OXA
Paracetamol	PRC
Praziquantel	PZQ
Rifampicin	RIF
Sulfanethazine	SMT
Sulfamethoxazole	SMX
Tetracycline	TC
Tylosine	TYL
Vancomycin	VAN

Introduction

The release of pharmaceuticals in water bodies has dealt a growing attention over the last years (Rizzo et al., 2009; Xekoukoulotakis et al., 2010; Van Doorslaer et al., 2015; Lofrano et al., 2014, 2016; Agarwal et al., 2017). They are mostly introduced in the sewage system through excretion of un-metabolized compounds after medical use or inappropriate disposal and then conveyed into the

wastewater treatment plants (WWTPs) (Teixeira et al., 2016). Pharmaceutical and pharmaceutical residues include antibiotics, anticonvulsants, antipsychotic, analgesics, beta-blocker, lipid regulators, and antihistamines ranging in the range of ng to mg per liter (Zuccato et al., 2005). Since conventional WWTPs are not designed to treat water polluted with pharmaceuticals present at trace levels, the applied treatments are mostly ineffective in their removal. As a consequence, they reach the aquatic system and can be detected in groundwater (Barnes et al., 2008;), drinking water (Benotti et al., 2009), surface water (Hirsch et al., 1999; Batt et al., 2006; Yan et al., 2013; Yang et al., 2011), sediment (Zhou et al., 2011) and agricultural lands (Hu et al., 2010; Karci and Balcioglu, 2009). Although, pharmaceuticals do not generally present acute toxic effects on aquatic organisms due to their low concentrations, concerns have been raised for chronic exposure due to their continuous and uncontrolled release into the environment acting as slightly persistent pollutants (Isidori et al., 2009; Lofrano et al., 2014). Among them, antibiotics represent one of the most urgent environmental problems, primarily due to the potential for the development of antimicrobial resistance among microorganisms (Akiyama et al., 2010; Fuentefria et al., 2011).

The limitations of conventional WWTPs in removing these bio-recalcitrant molecules point toward the urgent need for improved WWTPs such as Advanced Oxidation Processes (AOPs), a special class of oxidation techniques characterized by production of 'OH radicals (Lofrano et al., 2016). Amongst several AOPs, heterogeneous photocatalysis has proven its potential in degrading pharmaceutic compounds from aqueous matrices (Zhang et al., 2010; Lofrano et al., 2014, Van Doorslaer et al., 2015) representing a promising alternative for their removal due to its capacity to utilize the solar radiation as the light source, thus reducing significantly electric power required and therefore saving treatment costs and to operate without pH adjustment.

The elimination of mother compounds does not necessarily result in toxicity removal, since the photocatalytic degradation can produce intermediate by-products, which can still exert adverse biological effects. Therefore, to evaluate the overall behaviour and efficiency of the process, it is worth to assess not only the removal of a specific compound, but also the whole ecotoxicity

potential (Rizzo et al., 2009; Libralato et al., 2010a, 2016; Carotenuto et al., 2014; Lofrano et al., 2014). So far, ecotoxicity data for AOPs treated solutions of pharmaceuticals are scarce or missing, making their environmental risk assessment difficult.

Because the use of AOPs may be a promising approach for pharmaceuticals wastewater treatment, a great attention is focused on the potential of many types of nanoparticles (NPs) for photocatalysis applications. Amongst all NPs, TiO₂ is the most employed photocatalyst because of its favourable (photo-)chemical properties and low toxicity (Libralato et al., 2013; 2014), and ZnO is considered one of the most promising alternative because of its unique characteristics, such as direct and wide band gap in the near-ultra violet (UV) spectral region, strong oxidation ability, good photocatalytic property, and a large free-exciton binding energy (Lee et al., 2016). Recently, CuO and Ga₂O₃ have been also tested as alternative to more conventional photocatalysts (El-Sayed et al., 2014).

This chapter is aimed at describing the state of the art in the heterogeneous photocatalytic degradation of pharmaceuticals using different NPs. Since it is nearly impossible to review comprehensively photocatalysis studies, we have tried to summarize an overview of some of the more fundamental aspects, which are in their own right extremely scientifically interesting and which also need to be better understood in order to make significant progress and development with applications.

Fundamentals of heterogeneous photocatalysis

In the heterogeneous photocatalytic process, UV is utilized as an energy source and certain NPs act as a semiconductor (SC) photo-catalyst. For UV irradiation, its corresponding electromagnetic spectrum can be classified as UV-A, UV-B and UV-C, according to its emitting wavelength. The UV-A range has its light wavelength spans from 315 to 400 nm (3.10e3.94 eV), while UV-B has wavelength range of 280e315 nm (3.94e4.43 eV) and the germicidal UV-C ranges from 100 to 280 nm (4.43e12.4 eV) Illuminated SCs, by photons having an energy level that exceeds their band gap energy ($E > E_g$) excite electrons (e^-) from the valence band to the conduction band and holes (h^+) are produced in the valence band (reaction 1). The photo-generated valence band holes react with either water (H₂O) or hydroxyl ions (OH⁻) adsorbed on the catalyst surface to generate hydroxyl radicals ($^{\bullet}$ OH), which are strong oxidant (reactions 2 and 3). The photo-generated electrons in the conduction band may react with oxygen to form superoxide ions (O_2^-) (reaction 4). The superoxide ions can then react with water to produce hydrogen peroxide and hydroxyl ions (reaction 5). Cleavage of hydrogen peroxide by the conduction band electrons yields further hydroxyl radicals and hydroxyl ions (reaction 6).

- $SC + h\nu \to SC (e^- + h^+)$ (1)
- $SC(h^+) + H_2 0 \rightarrow H^+ + 0H^{\bullet}$ (2)
- $SC(h^+) + OH^- \rightarrow OH^{\bullet}$ (3)
- $SC(e^-) + O_2 \to O_2^-$ (4)
- $0_2^- + H_2 0 + H^+ \rightarrow H_2 0_2 + 0H^-$ (5)
- $\mathrm{H}_{2}\mathrm{O}_{2} + e^{-} \to \mathrm{OH}^{\bullet} + \mathrm{OH}^{-} \tag{6}$

The hydroxyl ions can then react with the valence band holes to form additional [•]OH. Degradation of organic substances can be achieved by their reaction with [•]OH or direct attack from the valence band holes. Recombination of the photo-generated electrons and holes may occur and indeed it has been suggested that pre-adsorption of substrate (organic substance) onto the photocatalyst is a prerequisite for highly efficient degradation (Elmolla and Chadhuri, 2010).

Nano-TiO₂ based photocatalysis

Titania based catalysts have proven to be useful and efficient photocatalysts for the abatement of pharmaceuticals (Lofrano et al., 2014; 2016). During the photocatalytic degradation process, the

efficiency of the system is related to some experimental variables that is mainly light irradiation intensity, catalyst and target compound concentrations, and pH. Typically, the rate of the photocatalytic degradation increases with the increase in light intensity and catalyst concentration up to a plateau value. Lofrano et al. (2016) reported that the degradation rate of 25 mg L^{-1} CAP increased when the concentration of TiO₂ increased up to 1.6 g TiO₂ L^{-1} . Beyond this value, the removal efficiency decreased (Figure 1).



Figure 1 Photocatalytic kinetic curves (A) and removal percentage (B) of CAP (25 mg L^{-1}) after 5, 10, 30, 60 and 120 min at 0.0, 0.1, 0.2, 0.4 and 1.6 g L^{-1} of TiO₂ at pH 5.5. Dark experiments at 1.6 g L^{-1} of TiO₂ (only Figure 1A – No significant differences could be observed in dark experiments at various TiO₂ concentrations investigated); 3.2 g L^{-1} of TiO₂ kinetic curve was not reported due to a

removal efficiency decrease (Fig. 1B) compared to 1.6 g L^{-1} of TiO₂ (Lofrano et al., 2016, with kind permission of Elsevier).

A similar trend was observed by Chatzitakis et al. (2008) during the photocatalytic degradation of 50 mg L^{-1} CAP. They raised TiO₂ concentration from 0.25 to 4 g L^{-1} observing a decrease in the initial reaction rate meaning that the photo-oxidation reached the saturation level. The optimum quantity depends on the nature of the organic compound and the photoreactor's geometry.

Regarding the pollutant concentration, it has been observed that the degradation rate does not increase linearly. Instead, at relatively high pollutant concentration, the slopes of the degradation curve decrease gradually (Giraldo-Aguirre et al., 2015). In Figure 2, it is shown the treatment of 5, 10 and 20 mg L⁻¹ DCF considering 250 mg/L of Degussa P25 TiO₂ as catalyst. Degradation decreased with increasing initial concentration, i.e. at 60 min corresponded to 99.5%, 61% and 46% at 5, 10 and 20 mg L⁻¹ DCF, respectively (Achilleos et al., 2010).



Figure 2 Effect of initial DCF concentration (5, 10 and 20 mg L^{-1}) treated by 250 mg L^{-1} Degussa P25 TiO₂: (a) concentration–time profiles; (b) comparison between TOC and carbon contained in DCF (Achilleos et al., 2010, with kind permission from Elsevier).

Generally the best degradation occurs at pH values at which the higher adsorption of the pollutant onto the catalyst surface occurs (Van Doorslaer et al., 2011). In some cases, the pollutant adsorption can also be detrimental. Consequently, to maximize the efficiency of the process an evaluation and optimization of the experimental parameters should be always carried out (Giraldo-Aguirre et al., 2015).

Several researches are also studying the influence that the presence of coexistent organic compounds and inorganic salts has to TiO₂-based photocatalysts (Guillard et al., 2003; Lair et al., 2008). The general and unanimous criteria are the detrimental effects that adsorption of inorganic ions plays on heterogeneous photocatalysis by trapping photogenerated holes and scavenging OH⁻ during photocatalytic processes (Carbajo et al., 2016). Giraldo-Aguirre et al. (2015) evaluated the photocatalytic degradation of OXA and interfering substances observing a slight inhibition related to excipients in commercial formulation or inorganic ions in minerals.

Depending on process conditions, the photocatalysis can achieve a complete mineralization of pollutants. Paola et al. (2006) reported that the use of UV/TiO₂ was able to achieve complete mineralization of 50 mg/L LNC antibiotic in 8 h. Abellán et al. (2007) reported 82% removal of 100 mg L⁻¹ SMX by degradation and 23% total organic carbon (TOC) reduction by UV/TiO₂ in 6 h. Palominos et al. (2008) reported that the complete removal of FLU by TiO₂ photocatalysis (0.5 g TiO₂ L⁻¹) occurred after 30 min at pH 6.

Nano ZnO based photocatalysis

Daneshvar et al. (2004) reported that ZnO could be a suitable alternative to TiO_2 its photocatalytic capability is anticipated to be similar to that of TiO2. Moreover, ZnO is relatively cheaper compared to TiO_2 whereby the usage of TiO_2 are uneconomic for large scale water treatment

operations. Since ZnO can absorb a larger fraction of the solar spectrum than TiO_2 , it is considered more suitable for photocatalytic degradation assisted by sunlight (Sakthivel et al., 2003). The major drawbacks of ZnO are the wide band gap energy and photocorrosion. The light absorption of ZnO is limited in the visible light region which is due to its wide band energy. This results in fast recombination of photogenerated charges and thus caused low photocatalytic efficiency (Gomez-Solís et al., 2015).

ZnO has proven to be a potential photocatalyst to treat various type of organic pollutants such as phenols (Assi et al., 2015), fungicides (Khodia et al., 2001), herbicides (Mijin et al., 2009), pharmaceuticals (Elmolla and Chaudhuri 2010; Palominos et al., 2009).

As in TiO₂based phototacatalysis, ZnO concentration, pH and irradiation time strongly influence the process efficiency. Elmolla and Chaudhuri (2010) reported that the optimum operating conditions for complete degradation of AMX, AMP and CLX antibiotics in an aqueous solution containing 104, 105 and 103 mg L⁻¹, respectively were: 0.5 g ZnO L⁻¹, irradiation time 180 min and pH 11. The effect of pH on pharmaceuticals degradation can be explained by taking into consideration the properties of both the catalyst and antibiotics at different pHs. For ZnO, the zero point charge is 9.0 \pm 0.3 and hence the ZnO surface is positively charged at pH < 9 and is negatively charged at pH > 9. Degradation of pharmaceuticals compounds typically increases with ZnO concentration presumably due to increase of °OH production. However, increasing ZnO concentration above such limits does not produce any significant improvement. This may be due to decreasing UV light penetration as a result of increasing turbidity and thus decreasing the photoactivated volume of the suspension (Daneshvar et al., 2004).

Few studies supported the assertion that ZnO is a better photocatalyst than TiO₂ (Chatzitakis et al., 2008) (Figure 3). After 90 min of illumination, photocatalysis with 1 g L⁻¹ of ZnO resulted to the approximately complete (90%) degradation of CAP 50 mg L⁻¹ (Chatzitakis et al., 2008). Although ZnO is more efficient than TiO₂ in visible light photocatalytic degradation of some organic compounds in aqueous solution, it is not stable as TiO₂.



Figure 3 Photodegradation of 50 mg L⁻¹ CAP as a function of irradiation time in the presence of 1 g L⁻¹ \blacksquare TiO₂-P25, \bullet TiO₂ (A), \blacktriangle ZnO, \blacktriangledown TiONa, \blacklozenge without catalysts (from Chatzitakis et al., 2008, with kind permission of Elsevier).

Nano CuO based photocatalysis

Copper(II) oxide has low cost and toxicity. As the photocatalytic mechanism suggests both photocatalyst and light source are necessary for the degradation reaction. Compared to TiO₂ and ZnO where an operative wavelength < 400 nm is required, CuO photocatalysis is based on UV-C. The effect of various concentrations of nano-CuO (0.05-0.3 g L⁻¹) can change the photocatalytic degradation rate of metronidazole (1 mg L⁻¹) as investigated by El Sayed et al. (2014). As previously reported for TiO₂ and ZnO, it could be observed that the initial removal rate increased with the increase in catalyst concentration up to a maximum after about 120 min and remaining almost constant thereafter. The optimum catalyst concentration for the degradation of metronidazole was set at 0.2 g L⁻¹.

Doped catalysts

The recombination of photogenerated hole and electron is the major disadvantage in semiconductor photocatalysis. This recombination step lowers the quantum yield and causes energy wasting. Therefore, the recombination process should be inhibited to ensure efficient photocatalysis. Metal doping could counter the recombination problem by enhancing the charge separation between

electrons and holes. In addition, the dopants may trap electrons, reducing the chances of electronehole recombination that deactivates the photocatalytic system (Lee et al., 2016)

By doping different NPs specifically constructed to be used as catalysts, it is possible to maximize the solar light utilization in AOPs. To date, several methods for achieving visible-light-driven photocatalysis or for increasing the lifetime of the photoproduced electron-holes pairs are widely investigated. It is known that pure TiO₂ exhibits relatively high activity only under ultraviolet (UV) light irradiation ($E_g = 3.2 \text{ eV}$), leading to a low utilization of solar light. To improve the absorption ability of TiO₂ in the visible region, several strategies have been involved including the decrease of TiO₂ particle size and the doping of TiO₂ and several other commercially available nanocrystalline semiconductors (e.g. ZnO or CuO) with metal ions or non-metallic species (N, C, S, B, P, F, or I) (Song et al., 2008). The enhanced activity of TiO₂ towards visible light is also achieved by the doping with N, S or C application of the supports (Al₂O₃, SiO₂) and, mainly by the formation of the nanocomposites, with carbon nanotubes (CNT) (Czech et al., 2015). However, metal-doped photocatalysts, which better exploit solar light, suffer from the problem of releasing metal pollutant species, sometimes extremely toxic metals, due to photocorrosion phenomena. On the other hand, the majority of photocatalytic action in non-metal doped photocatalysts illuminated by solar light is still generated by UV-C since the contribution from the visible part of the spectrum is small. Moreover, the stability and long-term efficacy of non-metal doped TiO₂ photocatalysts have not been tested.

Nitrogen doping via sol gel synthesis using diverse anion precursors (e.g. amines, nitrates, ammonium salts, ammonia and urea) (Pelaez et al., 2012) has been verified as a very efficient modification route of TiO_2 photocatalyst, activated in the visible spectrum via the formation of localized energy states within the band gap of TiO_2 or even oxygen deficiencies created during the reaction with the anion precursor (Moustakas et al., 2013). Three possible scenarios, including substitutional or interstitial N doping and defects formed due to disorder in the nanoparticle surface are sketched in Fig. 4 a–c.

An alternative route for the visible light activation of TiO_2 has been recently put forward based on the calcination of TiO_2 in the presence of urea (Mitoraj et al., 2008) The resulting materials were shown to consist of TiO_2 core covered by a poly(tri-s-triazine) shell, *in situ* formed on the TiO_2 surface by the thermal decomposition of urea (Mitoraj et al., 2010) as sketched in Fig. 4 d.



Figure 4 Sketch of possible modification routes for TiO_2 to become visible light active: (a) substitutional N doping, (b) interstitial N doping, (c) defect formation by surface doping, (d) polymeric carbon-nitride/TiO₂ composite structure and (e) sensitization of the surface by carbonacious species in the form of a surface capping layer (from Moustakas et al., 2013 with kind permission by Elsevier).

Suspended catalysts or thin films?

The photocatalysts could be used in slurry or supported forms. Many researchers investigated immobilization of NPs, mainly TiO_2 , on different support materials to improve the photocatalytic activity and make the separation of treated effluent more effective (Lu et al., 2011, Yap et al., 2011; Vilar et al., 2013). The most important properties of a suitable support are as follows: chemical inertness, high specific surface area and transparency to UV radiation. Coating surfaces with nanoparticles has relatively low improvement on photocatalytic reaction because of the low particles dispersion and limited mass transfer between the pollutants molecules and the catalyst

(Zhu et al., 2013; Hinojosa-Reyes et al., 2013). When the photocatalyst is dispersed as slurry inside the reactor, higher degradation rates are achieved due to high solid to liquid contact area but high axial flow rates are necessary to prevent the catalysts from settling. Furthermore the powders are not easy to precipitate and recover from water, preventing their regeneration and reuse (Paschoalino et al., 2012). To overcome this problem, magnetically separable composite photocatalysts such as TiO_2/Fe_3O_4 and $TiO_2/SiO_2/Fe_3O_4$ have been prepared and applied to the degradation of pollutants (Hu et al., 2011).